European Commission

Environmental impact assessments of innovative bio-based product

Task 1 of "Study on Support to R&I Policy in the Area of Bio-based Products and Services "

Written by COWI A/S and Utrecht University

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Environmental impact assessment of innovative bio-based products

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ABSTRACT

The aim of this project is to provide science-based facts and evidences on the environmental impacts of innovative bio-based products and mostly plastic products compared to petrochemical counterparts, in order to support the future bioeconomy policy and decision-making at EU level. Seven cradle-to-grave LCA case studies were carried out covering three major commercialised bio-based polymers, namely i) bio-based PET ("Beverage Bottles"), ii) PLA ("Single-use cups", "Single-use Cutlery", "Packaging films") and iii) starch plastics ("Clips", "Mulch films" and "Carrier bags"). Primary data are gathered from industry based on the real supply chain. This also includes the biomass used by the industry presently. Based on normalised and weighted results of 16 impact categories (excluding direct & indirect land use changes), for the bio-based products the most important impact categories were found to be climate change, abiotic depletion (fossil fuels) and human toxicity (cancer effects). Together, these three impact categories accounted for approximately 30-60 % of the total cradle-to-grave impacts and were highly associated with the direct and indirect (predominantly fossil) energy use in the cradle-to-user phase, and the direct emissions from the end-of-life phase. Based on a literature review of fossil fuel polymers, it is concluded that only five environmental impact categories are considered to have low discrepancy in their LCA results and could be used for comparison. These five impact categories are climate change, abiotic depletion (fossil fuels), particulate matters, photochemical ozone formation and terrestrial eutrophication. Compared to their fossil counterparts, the seven bio-based products investigated offer benefits with regard to climate change and abiotic depletion but have a higher impact in particulate matters in some cases (e.g. PET[1]); while the comparison for photochemical ozone formation and terrestrial eutrophication varies case by case. The chosen EoL scenario was found to have a strong influence on the overall impact savings of the bio-based products, especially significant for climate change impact.

[1] Due to the combustion of cane trash in the harvesting phase of sugarcane. It should be noted that burning cane trash will be soon phased out in Brazil.

RÉSUMÉ SCIENTIFIQUE

Le présent projet vise à fournir des éléments factuels et des preuves scientifiques concernant les impacts environnementaux des bioproduits innovants, et surtout des produits en plastique, par rapport à leurs homologues pétrochimiques, afin de contribuer à la politique bioéconomique à venir et à la prise de décision en la matière sur le plan de l'UE. Sept études de cas d'analyse du cycle de vie « du berceau à la tombe » ont été réalisées, couvrant trois des principaux biopolymères commercialisés, à savoir : i) le BioPET (« Bouteilles de boisson »), ii) l'acide polylactique (« Gobelets à usage unique », « Couverts à usage unique », « Films d'emballage ») et iii) les plastiques d'amidon (« Clips », « Films de paillage » et « Sacs en plastique »). Les données primaires ont été collectées auprès du secteur, sur la base de la chaîne d'approvisionnement réelle. Ceci inclut aussi la biomasse utilisée par le secteur à l'heure actuelle. Sur la base des résultats normalisés et pondérés de 16 catégories d'impacts (à l'exclusion des changements de l'affectation des sols directs et indirects), pour les bioproduits, les catégories d'impact qui se sont avérées les plus importantes ont été le changement climatique, la diminution des ressources abiotiques (combustibles fossiles) et la toxicité pour l'être humain (effets cancérigènes). Ensemble, ces trois catégories d'impacts représentaient environ entre 30 et 60 % des impacts au cours du cycle de vie et se sont avérées fortement associées à l'utilisation énergétique, directe et indirectement (surtout fossile) lors de la phase « du berceau à l'utilisateur », ainsi que des émissions directes de la phase de fin de vie. L'analyse de la littérature concernant les polymères des combustibles fossiles a permis de conclure que seulement cinq catégories d'impacts environnementaux peuvent être considérées comme présentant des faibles divergences pour ce qui est de leurs résultats dans l'analyse du cycle de vie et pourraient être utilisées à des fins de comparaison. Ces cinq catégories d'impacts sont le changement climatique, la diminution des ressources abiotiques (combustibles fossiles), les particules, la formation photochimique de l'ozone et l'eutrophisation terrestre. Par rapport à leurs homologues fossiles, les sept bioproduits objets de recherche présentent des bienfaits pour ce qui est du changement climatique et la diminution des ressources abiotiques, mais ont un impact plus élevé sur les particules dans certains cas (par exemple, le PET[[2]]). En revanche, la comparaison concernant la formation photochimique de l'ozone et l'eutrophisation terrestre varie en fonction des cas. Le scénario fin de vie choisi s'est avéré avoir une forte influence sur les économies d'impact d'ensemble des bioproduits, notamment pour ce qui est du changement climatique.

[2] Dû à la combustion de résidus de canne lors de la phase de récolte de la canne à sucre. Il faut noter qu'au Brésil la combustion de cette matière sera éliminée progressivement.

EXECUTIVE SUMMARY

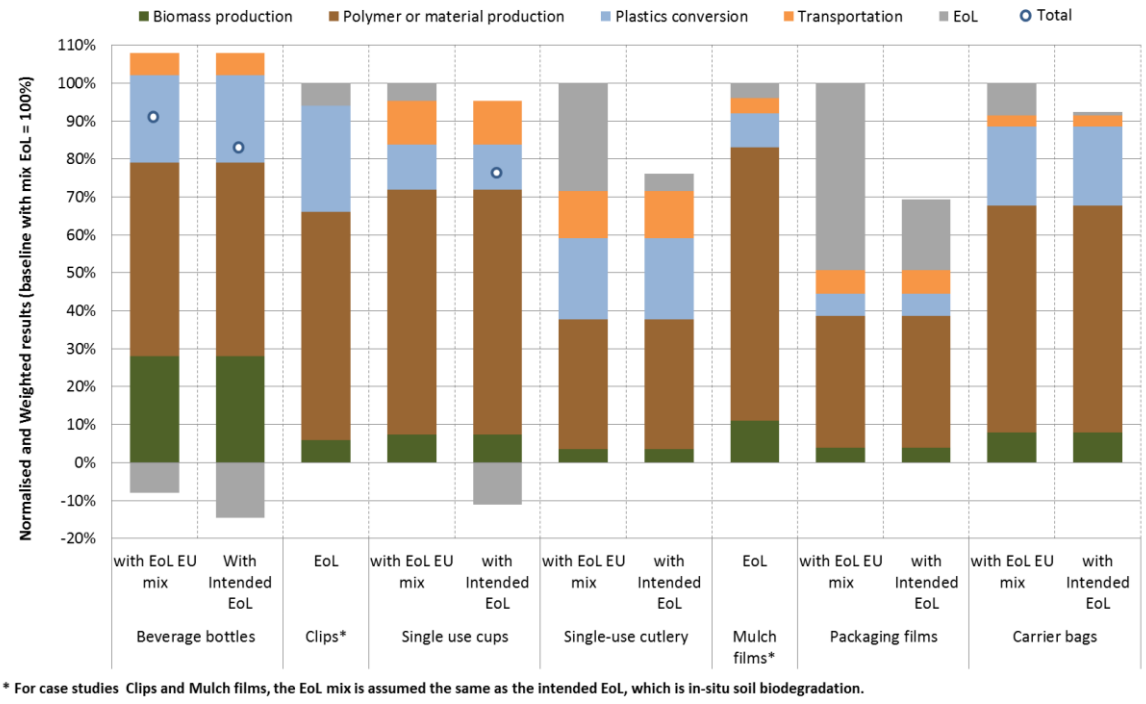
The aim of this project is to provide science-based facts and evidences on the environmental impacts of innovative bio-based products and mostly plastic products for comparison with petrochemical plastic counterparts, in order to support the future bio-economy policy and decision-making at the EU level, to support the implementation of the plastics strategy.

In this study, seven LCA (life cycle assessment) case studies were carried out covering three major commercialised bio-based polymers, namely i) bio-based PET (polyethylene terephthalate, case study "Beverage Bottles"), ii) PLA (polylactic acid, case studies "Single-use cups", "Single-use Cutlery" and "Packaging films") and iii) starch plastics (case studies "Clips", "Mulch films" and "Carrier bags").

Based on the normalised and weighted results of the seven cradle-to-grave LCA case studies for 16 impact categories, without including any possible effects of direct and indirect land use changes, we found that for the bio-based products the most important environmental impact categories i are climate change, abiotic depletion (fossil fuels)[3] and human toxicity (cancer effects). Together, these three impact categories account for approximately 30-60 % of the total cradle-to-grave environmental impacts of the studied bio-based products. These three impact categories are highly associated with (predominantly fossil) energy use in the cradle-to-user phase, and the direct emissions from the EoL (End-of-Life) phase.

Figure 1 shows the overview of the breakdown of the weighted LCA results for the cradle-to-grave environmental impacts. The weighting is carried out based on the weighting factors recommended by the Product Environmental Footprint Category Rules (PEFCR) guidance version 6.3 including all 16 impact categories.

[3] In the latest PEFCR guidance (v6.3) this impact category is re-named to "Resources (fossil fuels)".



* For case studies Clips and Mulch films, the EoL mix is assumed the same as the intended EoL, which is in-situ soil biodegradation.

Figure 1. Cradle-to-grave environmental impacts of seven bio-based products, weighted results based on 16 PEFCR impact categories (default weighting factors from PEFCR guidance version 6.3, including toxicity categories), broken down into the major life cycle stages. The results based on EoL EU mix scenarios are then compared with the results based on intended EoL scenarios. LCA results here exclude the effect of direct or indirect land use changes.

The *biomass production* phase in most cases accounts for less than 10 % of the overall cradle-to-grave environmental impacts (except for bio-based PET). *The manufacturing phase* has the highest contribution of all life cycle stage: it accounts for approximately 50 % of the cradle-to-grave impacts of nearly all seven bio-based products studied. Process energy (electricity and heat) and chemicals used during the manufacturing phase are responsible for the major parts of the total impacts. Bio-based industries are a nascent sector. Thus, immature and with production processes not fully efficient yet. In the future, through process optimisation the contribution of the manufacturing phase to the total environmental impact of bio-based products is expected to decrease. The impacts from *transportation* are in general insignificant. The exceptions include when the global supply chain is very long (e.g. in the case of single-use cutlery). The impacts of the EoL of the bio-based products studied are highly dependent on the applications and the assumptions. Differences are observed between the mixed EoL and intended EoL, for some applications, such as food packaging films and single-use cutlery, the differences are substantial (see Figure 1). Overall, recycling and industrial composting are more favourable than anaerobic digestion, municipal waste incineration and landfilling for the seven bio-based products studied. Note that this is average European technologies, hence an optimised anaerobic digestion plant could prove more favourable than both recycling and industrial composting.

The environmental impacts shown in Figure 1 excluded the effects of land use changes. In this study, an attempt was made to account for land use changes. Direct land use changes were accounted for based on the PEFCR Guidance v6.3 recommendations. For the impacts of indirect land use changes a deterministic approach based on historical deforestation data (2000-2010) was developed within this study, building on existing

frameworks. It resulted in an average indirect land use change (ILUC) factor of 4.0 t CO_2e ha-1 y-1, with a range of [1.22 – 5.20] t CO_2e ha-1 y-1. The main impacts affected by the inclusion of ILUC were climate change and photochemical ozone formation. When the effects of land use changes are accounted for, the impacts of biomass production do increase. However, the land use changes share to the overall cradle-to-grave impacts is relatively marginal: on average 14% for GWP_{100}, 10% for photochemical ozone formation, and between 0.01-2.4% for all other impact categories.

To conduct a comparative LCA between bio-based and petrol-based products, it is important that insights of all options compared are obtained. In this study, a deeper insight was gained for the bio-based polymers, whereas for the petrochemical polymers the Eco-profiles appear as a black box. In Chapter 6 of this report, we reviewed the Eco-profiles' data and compared the LCA results of all 16 PEFCR categories with other similar databases for the petrochemical polymers used in the case studies. This comparison shows that out of 16 impact categories, five categories are considered to have low discrepancy in LCA results. These five categories are used for the overall comparisons in this study, namely, climate change, abiotic depletion (fossil fuels), particulate matter (PM), Photochemical ozone formation and terrestrial eutrophication.

Figure 2 shows the results of the comparison between the bio-based products with their petrochemical counterparts. We conclude that for all seven cases, from cradle-to-grave the bio-based products offer environmental benefits in two impact categories: *climate change* and *abiotic depletion (fossil fuels)*. All seven bio-based products have a higher impact in *particulate matter* (PM) compared to the petrochemical counterparts. PM impacts are all higher for the bio-based. But the results deviate from the median value significantly. Thus, the median value of 87% higher PM impact for the bio-based in Figure 2 should be taken as an indicative value and should be treated with caution. The impacts of particulate matter are especially (six times) higher for bio-based PET bottles compared to petrochemical PET due to the burning of cane trash in the harvesting phase of sugarcane. In the case of PLA packaging films and cups, the particulate matter impacts are over two times as much as the impact of their PP counterparts. Compared to PP, PLA has a slightly higher energy requirement in the manufacturing phase and has much less avoided impact from the EoL phase. For all other cases, particulate matter is only marginally higher (3-8%) for the bio-based products than for their petrochemical counterparts. For the remaining two impact categories i.e., photochemical ozone formation and terrestrial eutrophication some bio-based products (i.e. bio-based PET bottles, bio-based carrier bags and PLA cups, films and cutleries) do not offer impact reductions, whereas for some other cases, especially where in-situ soil biodegradation is assumed as EoL waste management (i.e. bio-based mulch films and clips), substantial impact reductions are observed. Therefore, no generic trend was observed for the comparisons for these two categories. The results are case-dependent. It is also observed that the choice of EoL has a strong influence on the overall savings of the bio-based products. The influence is especially significant for climate change impact.

		Savings*, with EoL EU mix	Savings*, with intended EoL
Lower	Climate Change (GWP 100a)	-14% (-81% to 38%)	-66% (-87% to -14%)
	Abiotic depletion (fossil fuels)	-45% (-72% to -7%)	-46% (-72% to -3%)
Higher	Particulate matter	87% (2% to 603%)	94% (-2% to 597%)
Inconclusive	Photochemical ozone formation	23% (±79%)	12% (-79% to 58%)
	Terrestrial eutrophication	66% (-77% to 143%)	61% (-77% to 129%)

*Median savings based on the eight comparisons of the seven case studies (two comparisons were made for single-use cups).

Figure 2. Comparing cradle-to-grave environmental impacts of the seven bio-based products (EoL EU mix) with their petrochemical counterparts, results indicated as environmental impact median savings of seven bio-based products (negative values stand for savings). The comparisons between the bio-based products with *intended EoL* are listed in the second column. Savings are without considering the effect of direct and indirect land use changes.

In the *biodiversity assessment*, the novel Chaudhary 2015 methodology has been used for assessment of land use impacts on terrestrial biodiversity associated with the cultivation of selected feedstocks. Biodiversity is a relatively new topic in LCA and the methodology applied here is not yet mature enough to support political decision-making. The main shortcomings are related to ecoregion level assessments, where specific data are key but to a very large extend still unavailable. Specific locations of cultivation and local practices must be considered for all feedstocks because the impact may vary greatly with location. With the current methodologies available and the mentioned lack of data, it is not possible to compare between different feedstocks from different locations. More work and the establishment of methodological consensus is needed.

The *socio-economic assessment* was carried out based on a literature review, and not linked to quantitative modelling. Overall, the tentative conclusion is that there seems to be a tendency for bio-based products having better socio-economic performance compared to their petrochemical counterparts.

To improve future assessments of bio-based and fossil plastics, *better and transparent data* on the production and associated impacts of fossil plastics are needed; also, better monitoring of the composition of biogenic waste streams and behaviour for various EoL options would allow a more accurate assessment of possible future EoL scenarios. We recommend incentivizing the industries involved to collect and share these data.

Numerous *methodological issues* need to be tackled linked to the LCA of plastics, including comprehensive assessment of land use impacts (also of fossil fuels) and the timing of emissions. Also, the impact of littering, which is not included in the End of Life modelling, due to a missing standardised methodology, both missing possible impact categories as well as missing data on e.g. how the plastics (bio-)degrade in different environmental compartments. Furthermore, more general improvements regarding the assessment of biodiversity and socio-economic impacts would allow a more complete assessment of the overall effects. We recommend that these issues are further investigated. Moreover, unanticipated results from one case indicating that certain imported feedstocks performed better that EU produced counterparts, warrant further investigation even if no conclusions can be made from the result of that case.

To support the *formulation of policies on bio-based products*, we recommend that a larger number of case studies is carried out, including different feedstocks and different end-uses (including durable products) to obtain a more comprehensive overview of the

structural strong and weak spots in comparison to the fossil-based alternatives. The bio-based industry is still in its infancy stage. There are on-going innovation efforts to optimise manufacturing process. The efforts would likely lead to a lower environmental impact for the future bio-based products than today. The potential (decrease in) environmental impacts of bio-based plastics due to technological progress and the exogenously changing energy system should be assessed.

Ultimately, this study provides an excellent starting point for comparing bio-based and fossil plastics, as well as a good indication of the likely impacts and benefits of bio-based products compared to the fossil-based conventional products, but this study should be supplemented to provide a better basis before any formulation of general policies for bio-based plastics.

RÉSUMÉ

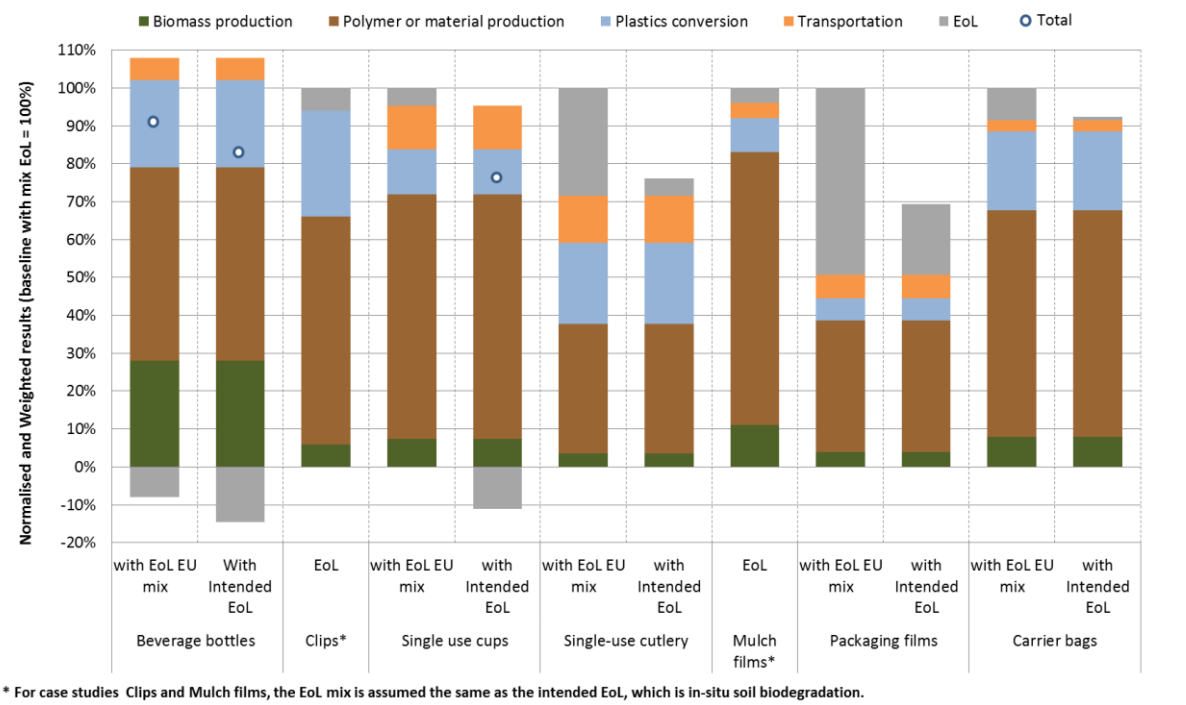
Le présent projet vise à fournir des éléments factuels et des preuves scientifiques concernant les impacts environnementaux des bioproduits innovants, et surtout des produits en plastique, par rapport à leurs homologues pétrochimiques, afin de contribuer à la politique bioéconomique à venir et à la prise de décision en la matière sur le plan de l'UE, dans l'objectif de soutenir la mise en œuvre de la stratégie sur les plastiques.

Dans le cadre de ce travail, sept études de cas d'analyse du cycle de vie ont été réalisées, couvrant trois des principaux biopolymères commercialisés, à savoir : i) le BioPET (polytéréphtalate d'éthylène, étude de cas « Bouteilles de boisson »), ii) l'acide polylactique (études de cas « Gobelets à usage unique », « Couverts à usage unique » et « Films d'emballage ») et iii) les plastiques d'amidon (études de cas « Clips », « Films de paillage » et « Sacs en plastique »).

Sur la base des résultats normalisés et pondérés de 7 études de cas d'analyse de cycle de vie « du berceau à la tombe » concernant 16 catégories d'impacts (à l'exclusion des effets éventuels des changements de l'affectation des sols directs et indirects), nous avons conclu que, pour les bioproduits, les catégories d'impacts environnementaux les plus importantes sont le changement climatique, la diminution des ressources abiotiques (combustibles fossiles)[4] et la toxicité pour l'être humain (effets cancérigènes). Ensemble, ces trois catégories d'impacts représentent environ entre 30 et 60 % des impacts environnementaux du cycle de vie entier des bioproduits étudiés. Ces trois catégories d'impacts sont fortement associées à l'utilisation énergétique (surtout fossile) lors de la phase « du berceau à l'utilisateur », ainsi que des émissions directes de la phase de fin de vie.

Figure 3 donne un aperçu de la ventilation des résultats pondérés de l'analyse des impacts environnementaux au cours du cycle de vie entier. La pondération se fait sur la base des facteurs de pondération recommandés par la version 6.3 des orientations « Product Environmental Footprint Category Rules (PEFCR) », incluant l'ensemble des 16 catégories d'impacts.

[4] Dans les dernières orientations PEFCR (v6.3), cette catégorie d'impacts est rebaptisée « Ressources (combustibles fossiles) ».



* For case studies Clips and Mulch films, the EoL mix is assumed the same as the intended EoL, which is in-situ soil biodegradation.

Figure 3. Impacts environnementaux « du berceau à la tombe » de sept bioproduits, résultats pondérés sur la base des 16 catégories d'impacts PEFCR (facteurs de pondération par défaut de la version 6.3 des orientations PEFCR, y compris les catégories de toxicité) ventilés en fonction des étapes majeures du cycle de vie. Les résultats sur la base des scénarii mix UE de fin de vie sont ensuite comparés aux résultats sur la base des scénarii de fin de vie envisagée. Les résultats de l'analyse de cycle de vie ici présentées excluent les effets des changements, directs ou indirects, de l'affectation des sols.

La phase de *production de biomasse* représente, dans la plupart des cas, moins de 10 % des impacts d'ensemble du cycle de vie entier (sauf dans le cas du BioPET). La *phase de fabrication* est celle qui contribue le plus sur l'ensemble du cycle de vie : elle représente environ 50 % des impacts « du berceau à la tombe » pour quasiment la totalité des bioproduits étudiés. L'énergie (électricité et chaleur) et les produits chimiques de traitement utilisés lors de la phase de fabrication sont à l'origine des parties les plus importantes des impacts d'ensemble. Les bio-industries constituent un secteur naissant. Aussi, elles n'ont pas atteint la maturité et leurs processus de production ne sont pas encore pleinement efficients. À l'avenir, grâce à l'optimisation des processus, la contribution de la phase de fabrication à l'impact environnemental total des bioproduits devrait diminuer. Les impacts du *transport* sont, en général, insignifiants. Parmi les exceptions à cette règle figurent les cas où la chaîne d'approvisionnement est très longue (par exemple, pour les couverts à usage unique). Les impacts de la fin de vie des bioproduits étudiés dépendent fortement des applications et des hypothèses. On observe des différences entre la fin de vie mixte et la fin de vie envisagée, et pour certaines applications, comme les films d'emballage alimentaire et les couverts à usage unique, les différences sont de taille (voir Figure 3). Dans l'ensemble, le recyclage et le compostage industriel s'avèrent plus avantageux que la macération anaérobique, l'incinération des déchets municipaux et les décharges, et cela pour les sept bioproduits étudiés. À noter qu'il s'agit là de technologies européennes ordinaires, de sorte qu'une installation de macération anaérobique pourrait s'avérer plus avantageux que le recyclage et le compostage industriel.

Les impacts environnementaux illustrés dans la Figure 3Figure 1 excluent les effets des changements de l'affectation des sols. Dans le cadre de cette étude, une tentative a été

entreprise pour tenir compte des changements des affectations des sols. Les changements directs de l'affectation des sols ont été pris en considération sur la base de la version 6.3 des recommandations des orientations PEFCR. Pour ce qui est des impacts des changements indirects de l'affectation des sols, une approche déterministe basée sur des données historiques concernant la déforestation (2000-2010) a été développée dans le cadre de cette étude, se fondant sur des cadres existants. Le résultat est un facteur moyen de changement indirect de l'affectation des sols (CIAS) de 4,0 t CO_2e hectare-1 a-1, allant de [1,22 à 5,20] t CO_2e hectare-1 a-1. Les principaux impacts affectés par l'inclusion du CIAT ont été le changement climatique et la formation photochimique de l'ozone. Lorsque les effets du changement de l'affectation des sols sont pris en compte, les impacts de la production de biomasse augmentent. Néanmoins, la part des changements de l'affectation des sols sur l'ensemble des impacts du cycle de vie entier sont relativement marginaux : une moyenne de 14 % pour le PRG_{100}, de 10 % pour la formation photochimique de l'ozone et d'entre 0,01 et 2,4 % pour toutes les autres catégories d'impacts restantes.

Pour réaliser une analyse du cycle de vie comparative des bioproduits et des produits à base de pétrole, il importe de disposer de connaissances concernant l'ensemble des options comparées. Dans le cadre de cette étude, nous sommes parvenus à mieux comprendre les biopolymères, alors que pour ce qui est des polymères pétrochimiques, les éco-profils ressemblent à des boîtes noires. Dans le chapitre 6 du présent rapport, nous avons étudié les données des éco-profils et comparé les résultats de l'analyse du cycle de vie pour l'ensemble des 16 catégories PEFCR à d'autres bases de données similaires pour les polymères pétrochimiques utilisés dans les études de cas. Cette comparaison met en évidence que sur les 16 catégories d'impact, cinq catégories sont considérées comme présentant un faible écart pour ce qui est des résultats de l'analyse du cycle de vie. Ces cinq catégories sont utilisées pour les comparaisons d'ensemble dans cette étude, à savoir notamment : le changement climatique, la diminution des ressources abiotiques (combustibles fossiles), les particules, la formation photochimique de l'ozone et l'eutrophication terrestre.

Figure 2Figure 4 montre les résultats de la comparaison opérée entre les bioproduits et leurs homologues pétrochimiques. Nous concluons que pour les sept études de cas, sur leur cycle de vie entier, les bioproduits fournissent des avantages environnementaux pour deux catégories d'impact : le *changement climatique* et la *diminution des ressources abiotiques (combustibles fossiles).* L'ensemble des sept bioproduits ont un impact plus élevé pour ce qui est des *particules*, par rapport à leurs homologues pétrochimiques. Les impacts en termes de particules sont tous plus élevés pour les bioproduits. Néanmoins, les résultats s'écartent de la valeur moyenne de façon significative. Aussi, la valeur moyenne de 87 % pour l'impact en matière de particules plus élevé des bioproduits indiquée dans la Figure 4 devrait être considérée comme une valeur purement indicative et avec prudence. Les impacts en termes de particules sont particulièrement (six fois) plus élevés pour les bouteilles en BioPET, par rapport à celles en PET pétrochimique, et cela en raison de l'incinération de déchets de canne à sucre lors de la phase de récolte de cette dernière. Dans le cas des films d'emballage et des gobelets en acide polylactique, les impacts en matière de particules sont plus de deux fois plus élevés que ceux de leurs homologues en polypropylène. Par rapport au polypropylène, l'acide polylactique s'avère légèrement plus énergivore lors de la phase de fabrication et présente un impact évité bien plus faible de la phase de fin de vie. Dans tous les autres cas de figure, les particules émises sont seulement marginalement plus élevées (3-8 %) pour les bioproduits que pour leurs homologues pétrochimiques. S'agissant des deux catégories d'impact restantes, à savoir la formation photochimique de l'ozone et l'eutrophication terrestre, certains bioproduits (les bouteilles en BioPET, les bio-sacs en plastique et les gobelets, les films et les couverts en acide polylactique) ne présentent pas de réduction d'impact, alors que dans quelques autres hypothèses, notamment lorsque la biodégradation dans le sol *in situ* est présumée comme étant la méthode de gestion des déchets en fin de vie (à savoir, les bio-films de paillage et les bio-clips), des réductions substantielles de l'impact ont été observées. Aussi, nulle

tendance générale n'a été constatée dans le cadre des comparaisons pour ces deux catégories. Les résultants varient en fonction des cas. Il a également été observé que le choix de la fin de vie a une forte influence sur les économies d'ensemble des bioproduits. Cette influence s'avère particulièrement significative pour ce qui est de l'impact sur le changement climatique.

		Savings*, with EoL EU mix	Savings*, with intended EoL
Lower	Climate Change (GWP 100a)	-14% (-81% to 38%)	-66% (-87% to -14%)
	Abiotic depletion (fossil fuels)	-45% (-72% to -7%)	-46% (-72% to -3%)
Higher	Particulate matter	87% (2% to 603%)	94% (-2% to 597%)
Inconclusive	Photochemical ozone formation	23% (±79%)	12% (-79% to 58%)
	Terrestrial eutrophication	66% (-77% to 143%)	61% (-77% to 129%)

*Median savings based on the eight comparisons of the seven case studies (two comparisons were made for single-use cups).

Figure 4. Comparaison des impacts environnementaux « du berceau à la tombe » des sept bioproduits (fin de vie UE mix) avec leurs homologues pétrochimiques, résultats indiqués en tant qu'économies moyennes en termes d'impacts environnementaux pour les sept bioproduits (les valeurs négatives représentent des économies). Les comparaisons entre les bioproduits par rapport à la *fin de vie envisagée* sont listées dans la seconde colonne. Les économies ne tiennent pas compte de l'effet des changements, directs ou indirects, de l'affectation des sols.

Dans le cadre de *l'évaluation de la biodiversité*, la nouvelle méthodologie de 2015 de Chaudhary a été utilisée pour l'analyse des impacts de l'affectation des sols sur la biodiversité terrestre associée à la culture de matières premières choisies. La biodiversité constitue un sujet relativement nouveau dans le cadre de l'analyse du cycle de vie, et la méthodologie appliquée ici n'est pas encore suffisamment mûre pour contribuer à la prise de décision politique. Les principales insuffisances sont associées aux évaluations au niveau de l'écorégion, où les données spécifiques s'avèrent essentielles mais demeurent, dans une grande mesure, non disponibles. Les lieux spécifiques de culture et les pratiques locales doivent être pris en considération pour l'ensemble des matières premières car l'impact peut varier énormément en fonction du lieu. En vue des méthodologies actuellement disponibles et de l'absence de données susvisée, il ne s'avère pas possible d'effectuer une comparaison entre différentes matières premières provenant de lieux divers. Davantage de travail et un consensus quant à la méthodologie sont nécessaires.

L'*évaluation socio-économique* a été réalisée sur la base d'une étude de la littérature, sans quelque association que ce soit à une modélisation quantitative. Dans l'ensemble, la conclusion provisoire est que les bioproduits semblent avoir tendance à dégager de meilleurs résultats sur le plan socio-économique que leurs homologues pétrochimiques.

Afin d'améliorer les évaluations à venir des bioplastiques et des plastiques fossiles, *des données de meilleure qualité et plus transparentes* concernant la production est les impacts associés des plastiques fossiles s'avèrent nécessaires. De même, un meilleur suivi de la composition des flux de déchets biogéniques et du comportement des différentes options de fin de vie permettrait de réaliser une évaluation plus précise des scénarii de fin de vie envisageables pour l'avenir. Nous recommandons d'inciter les industries impliquées à collecter et partager ces données.

De nombreuses *difficultés méthodologiques* doivent être résolues concernant le cycle de vie des plastiques, dont l'évaluation exhaustive de l'impact de l'affectation des sols (y compris pour les combustibles fossiles) et le moment des émissions. Il en va de même

pour l'impact des ordures, non inclus dans la modélisation de la fin de vie en raison de l'absence d'une méthodologie normalisée, ce qui a pour conséquence de ne pas tenir compte de catégories d'impact éventuelles et un manque de données concernant la façon dont les bioplastiques se dégradent dans différents compartiments environnementaux. En outre, des améliorations plus générales concernant l'évaluation de la biodiversité et des impacts socio-économiques permettraient de réaliser une évaluation plus complète des effets d'ensemble. Nous recommandons que ces questions fassent l'objet de davantage de recherches. Par ailleurs, les résultats inattendus d'un cas indiquant que certaines matières premières importées ont dégagé de meilleurs résultats que leurs homologues produits dans l'UE nécessitent la réalisation de davantage de recherches, même si nulle conclusion ne peut être tirée des résultats dudit cas.

Afin de contribuer à la *formulation de politiques concernant les bioproduits*, nous recommandons qu'il soit procédé à un nombre élevé d'études de cas concernant différentes matières premières et des utilisations finales diverses (y compris pour les produits durables), afin d'obtenir un aperçu plus exhaustif des points forts et faibles structurels par rapport aux alternatives fossiles. La bio-industrie se trouve toujours à ses balbutiements. Des efforts d'innovation sont en cours afin d'optimiser les processus de fabrication. Ces efforts déboucheront, probablement, sur des impacts environnementaux plus faibles à l'avenir pour les bioproduits. La diminution potentielle des impacts des bioplastiques induite par les progrès technologiques et l'évolution exogène du système énergétique devrait être évaluée.

Enfin, la présente étude constitue un excellent point de départ pour comparer les bioplastiques et les plastiques fossiles, ainsi qu'une bonne indication des impacts et des avantages probables des bioproduits par rapport aux produits d'origine fossile conventionnels. Néanmoins, elle devrait être complétée, pour fournir une base plus solide, avant toute formulation de politiques générales concernant les bioplastiques. Background, policy context and aim of the study

BACKGROUND

The bio-based industries in Europe and globally are growing, driven by business and consumer demand, as well as policy initiatives promoting more sustainable consumption and reduced environmental footprint. The development and deployment of innovative technologies that can release the potential of the bioeconomy at competitive cost levels is crucial for the scale up to commercial levels. The scale up will enable the transition from fossil to renewable sources.

Bioeconomy is defined by bio-based value chains, meaning value chains that start from agriculture, forests or organic waste. New technologies and traditional methods coupled with biotechnologies applied to a biomass feedstock or waste streams such as the urban or food wastes or agricultural residues, can be converted into high added-value sustainable bio-based products. For the consumer, bio-based products can be both visually and functionally indistinguishable from fossil-based alternatives or entirely new with markedly different look, feel and function. An important characteristic is the fact that bio-based products can be both B2B and B2C and that many of them are ingredients and components that only constitute a small part of a final product yet offers functional and or environmental improvements. In short, this means that the bioeconomy is everywhere around us, and that the bioeconomy is a complex multi-level and multi-sector transition more than a distinct substitution of like-for-like.

Sustainable bio-based products have the potential to deliver many environmental, economic and social benefits. They can help to meet several policy objectives ranging from economic growth, job creation, growing the circular economy and enhanced resource efficiency to climate change mitigation, food security, agriculture modernisation and regional development (see Figure 5). To ensure a secure and sustainable supply of biomass feedstock for biorefineries it is important to note that bio-based value chains and products are not per se sustainable. Issues such as land use, deforestation, water use, emissions, health and energy demand are important for an overall assessment of sustainability. And thereby for the determination of the environmental benefits of the products of the bioeconomy. Considering increasing pressure on our planet and nature, the bioeconomy and its political masters will have to prioritise use of finite bio-based resources to uses that delivers the best environmental and economic benefits.

Innovative, financially solid and growing Bio-based industries in Europe are essential for EUs ability to reap the benefits of the bioeconomy, economic as well as environmental. Already now, bio-based chemicals, plastics, pharmaceuticals, forest-based products such as paper, textiles, biofuels and bioenergy account for an annual turnover of EUR 600 billion (BIC, 2016). Moreover, the European bioeconomy provides jobs for 18.3 million Europeans, over half of them in primary biomass production (agriculture, forestry, and fisheries). Bio-based industries employ 3.2 million people (EuropaBio, 2016).

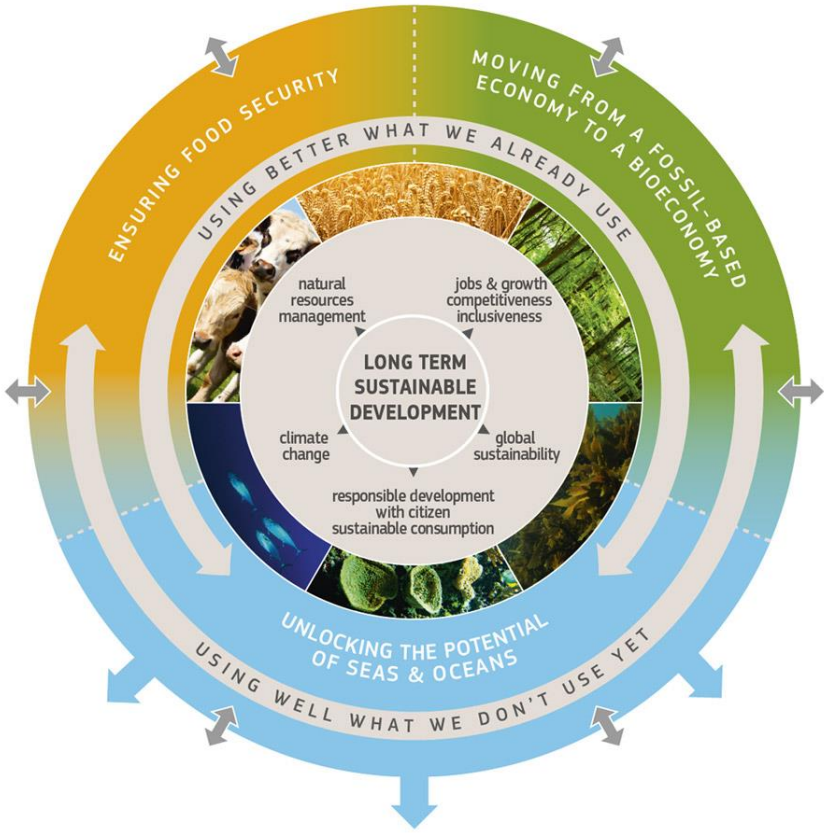


Figure 5. European Bioeconomy Strategy. Source: EC, 2017.

1 Policy context

Being linked to many policy areas, innovative bioeconomy plays an integral part of the Commissions policy agenda for Environment, Climate, Jobs, Growth, Fairness and Democratic Change and will continue to be on the Research and Innovation Agenda. The Europe 2010 Flagship Initiative - Innovation Union (COM, 2010) placed innovation in the heart of the Europe 2020 strategy to maintain Europe's competiveness in a demographically changing and increasing competitive world. One specific policy initiative by the commission to support the 2020 strategy, was the 2012 EU Bioeconomy Strategy (EC, 2012b). The EU Bioeconomy Strategy was revised in second half of 2018, just before the closure of the BIOSPRI project.

Beyond technological barriers, both market acceptance and commercialization need to be supported by the policy and economic context to drive the transition to a bioeconomy. Mirroring this, the 2012 Bioeconomy strategy is structured around three pillars:

- Investments in research, innovation and skills;
- Reinforced policy interaction and stakeholder engagement;
- Enhancement of markets and competitiveness.

Bioeconomy interfaces with a range of policy area such as food, agriculture, energy, climate, land use planning etc. Linked to this fact, there might be a demand from more sides for the same bio-based resources. It is a balance to find the optimum sustainable use of land, sea and associated resources, and the bioeconomy review points to the need for addressing policy coherence to avoid sub-optimization. In this vein, the review recognises that the policy context has changed since 2012, and specifically the Circular

Economy, the Energy Union and the Sustainability Development Goals are highlighted. It can also be noted that the Bioeconomy Panel was created in 2013 to support interactions between different policy areas, sectors and stakeholders in the bioeconomy (e.g. business and primary producers, policy-makers, researchers and civil society organisations). In the Review, it is further concluded that the strategy has mobilised increased funding in research and innovation (R&I) in bioeconomy and contributed to knowledge and education. The 2018 update of the Strategy continues this focus, while recognizing that many bio-based sectors are closer to market now than in 2012 and hence that investments are now needed to complement research grants.

The need for policy coherence is in line with the 2014 policy document "For a European Industrial Renaissance" which states: 'granting access to sustainable raw materials at world market prices to production of bio-based products [...] will require the application of the cascade principle in the use of biomass and eliminating any possible distortions in the allocation of biomass for alternative uses that might result from aid and other mechanisms that favour the use of biomass for other purposes (e.g. energy)'.

Overall, development of innovative bio-based ingredients and products is increasingly foreseen to be part of the toolbox for meeting the objectives in several key EU policy areas. Most notably, the action plan for the Circular Economy sees innovation as instrumental in securing the systemic changes needed for a shift to a circular economy. Biomass and bio-based products is one of five priority areas, which shall boost this transition. The action plan also points out that the bio-based sector has shown its potential for innovation in new materials, chemicals and processes, supporting the transition and that the Commission will continue to promote investment in bioeconomy innovation. In the review of the bioeconomy strategy, the Commission foresees that a new bioeconomy strategy shall feed into this European flagship initiative.

Another priority area in the Circular Economy action plan is 'plastics'. The recently published Plastics strategy aims at stimulating innovation in relation to plastics waste reduction and collection, but also further research and development of alternative sustainable types of feedstocks, including those of bio-based origin. The strategy notes that costs can be an obstacle to development and application of bio-based plastics and that is important to see sustainability in life cycle context as compared to non-bio-based plastics. Innovation in relation to bio-based plastics might strengthen the competiveness of the European industry and provide the same or new plastics as compared to fossil-based plastics, including biodegradable plastics. It should be noted that Horizon 2020 has already funded and will continues to fund development of alternative feedstock, including feedstock for plastic production.

In short, current EU policy action, notably the 2012 Bioeconomy Strategy is a response to and tool for the promotion of a circular Bioeconomy under the 2020 Strategy and looking beyond. The 2018 Update of the Bioeconomy Strategy, together with the renewed Industrial Policy and the Clean Energy for all Europeans Package, take stock on the development of the circular Bioeconomy and propose actions and in some cases legislation for the period up to 2030. Among three main action areas, the Bioeconomy Strategy outlines that further work is needed to *understand the ecological boundaries of the bioeconomy*. The overall aim of this study is drawn from this context. In view of prioritising efforts and support, both at product and policy - several LCA methodological issues need to be tacked, including comprehensive assessment of land use impacts (also of fossil fuels), soil degradation, ecosystem health, biodiversity, the timing of emissions as well as littering, which is not included in the End-of-life modelling. With such information available it should be possible for policy makers, researchers and investors to determine what sectors, products and innovations that holds most promise in terms of high value added and low environmental impact. Numerous *methodological issues* need to be tackled linked to the LCA of plastics, including comprehensive assessment of land use impacts (also of fossil fuels) and the timing of emissions. Also, the impact of littering, which is not included in the End-of Life-modelling, due to a missing standardised

methodology, both missing possible impact categories as well as missing data on e.g. how the plastics (bio-)degrade in different environmental compartments. Furthermore, more general improvements regarding the assessment of biodiversity and socio-economic impacts would allow a more complete assessment of the overall effects.

2 Aim of the study

The aim of this project is to provide science-based facts and evidences on the environmental impacts of innovative bio-based products and mostly plastic products in comparison with petrochemical plastic counterparts. This will support the future bioeconomy policy and decision-making at the EU level, in particular support the implementation of the plastics strategy. A recognised approach for assessing environmental impacts of products is the LCA framework. Hence, this study will answer an identified need for a more standardised, comparable approach to LCA for bio-based value chains and thereby for bio-based products. Such an approach would support identifying the most environmentally friendly and best performing bio-based products, both for new products or as a substitute for a fossil-based alternative, and thereby enable detailed prioritisation of action and support at feedstock, technology and product level. The environmental impact assessments are performed by conducting LCA of seven-selected bio-based product (see the selection procedure and results in deliverable Sub-task 1.1).

Many existing studies have shown that bio-based products offer important environmental benefits especially the impact reductions of non-renewable energy use (NREU) and greenhouse gas (GHG) emissions (Hermann et al., 2007; Weiss et al., 2012). However, many previously published LCA studies on the innovative bio-based products are either perspective studies which cover only carbon and energy indicators (Cok et al., 2014; Patel et al., 2005; Pawelzik et al., 2013; Weiss et al., 2012), or only covers the impacts from cradle to factory gate (Broeren et al., 2017a,b, 2016; Patel et al., 2005; Shen et al., 2012; Tsiropoulos et al., 2015). Also, the potential impact of extracting biomass resources on ecosystems, such as land use, land use changes, biodiversity and water withdrawal, still needs to be well understood (Broeren et al., 2017b). As such, by conducting new complete LCAs for seven bio-based products, this study further contributes to enhancing the understanding of the environmental implications of the transition towards a bio-based economy particularly for plastics applications and to offer learnings on so far underdeveloped LCA impact categories.

The LCAs are breaking new ground on another strategically important topic, namely EoL. In the past decades, many innovative bio-based products have successfully made its way to the market, entered our daily life and eventually ended up in the waste management systems. It is therefore, important to reflect the potential environmental impacts of these newcomers and consider the impact of End of Life (EoL) waste management.

To accommodate the aims of the study in a comparable and transparent manner, the LCAs are carried out to comply with the latest European Commission's Product Environmental Footprint Category Rules (PEFCR), as closely as possible. Since the innovative bio-based products are a relatively new group of products, the study faced the situation that some part of the methodology is not fit for this purpose, concerning impacts of the feedstock production, i.e. Indirect Land Use Change (ILUC), Biodiversity and Ecosystem Impacts, and for littering and End of Life (EoL). Wherever a decision is made other than what is recommended by the PEFCR, it has been documented in the individual case study chapter. In the final conclusions chapter, experience will be summarised, which aims at supporting future research and policy development. Notably, the experiences show that there are still several crucial limitations to LCA (PEF) that means comparison of bio-based and fossil products must be made with care and diligence. The experiences can be used to improve the future PEFCR development for the category of innovative bio-based products.

APPROACH AND METHODOLOGY

This first part of the report covers methodological issues, and clarifies and explains choices made, tools and methods used, and critical assumptions adopted. The second part of the report consists of individual chapters for each of the seven product cases.

The methodological issues covered in this first part of the report are, further to scope and other overarching issues:

- LCI Modelling
- EoL Inventory modelling
- Land Use modelling
- Interpretation, uncertainty and sensitivity analysis
- Generic LCAs data for fossil-based plastics
- Biodiversity Assessment

The end of the report, after presentation of the seven cases, includes a section on combined observations and closing remarks, a list of abbreviations and references covering all parts, and 6 Annexes.

1 Introduction and scope definition

This chapter introduces several critical overall methodological issues of relevance across all the impact category or modelling specific issues which are covered in the succeeding chapters. The overall methodological issues are *Scope*, *Data*, *Case study Selection*, *Functional Units and Product Systems*, and *Impact Assessment methods*, covered in that order.

The LCAs were conducted in accordance with the ISO guidelines (ISO 14040 and ISO 14044) and EN 16760 (CEN, 2015). Having the aim of the study in mind, we strived to stay as close as possible to PEFCR guidance (version 6.3), for which many detailed modelling and methodological decisions were made. However, the LCAs should not be a fully compliant with the latest PEFCR guidance, because of different decision contexts of the current study and the PEFCR guidance is not developed to accommodate niche applications which is the main group of products analysed in this study. Lessons learned from this study is summarised in the final chapter of this report Observations*/Conclusions*.

1.1 Scope

Based on the goals of the LCA studies, the studied products are assumed to be used and disposed of in Europe whereas the supply chain of biomass production and the manufacturing processes can be extended globally.

Technological scope is defined as the status-quo production technologies as of the status of the year 2017. The background technologies are average technologies reflecting the 2000's and the 2010's.

The temporal scope is defined as current (2017-2018) and in 5-10 years near future.

The LCAs cover the full life cycle, i.e. from cradle-to-grave.

1.2 General Considerations on Data

The ambition of the project is to gather to the extent possible primary data from relevant industries to conduct full LCAs for the selected seven bio-based products, using consistent methodology and reflecting the status-quo environmental impacts to support policy decisions. The environmental impacts are then compared with those of reference products made from fossil fuel resources.

Literature data, rather than primary data, are used for the impacts of the reference products made from fossil fuels. The project team acknowledges that this could potentially create an imbalanced view. Efforts were made to obtain transparency from the reported literature data. A critical review was conducted for such a purpose. The method of critical review is documented in section 6 of this chapter. The result of the critical review is documented in Chapter *Critical review of petrochemical references*.

1.3 Selection of case studies

Currently, there is large focus on development and production of bio-based products from numerous stakeholders. Thus, it is important to carry out a thorough, transparent and trustworthy environmental assessment of bio-based product groups, their different possible feedstocks, and the EoL scenarios as well as their market performance and potential for deployment and other relevant aspects. Furthermore, it is important to gain knowledge about the potential environmental impacts of selected bio-based product groups to support - if justifiable - the applications having better environmental performance than the current fossil solutions. This requires environmental life cycle impact assessment of relevant products and their functionalities. The performance of each product within each of these criteria will help to guide the selection of promising case studies, and it is therefore vital that criteria most suitable to guide this, are identified and used for this assessment.

To ensure the highest relevance and highest potential value of this project scoring criteria has been defined and selected for the prioritisation of the most relevant case studies (see Table 1). All but one of these main assessment aspects derives from relevant EU projects on bio-based products, regulation, EU prioritisation within bio-based products and our own expert experience from previous projects, as also shown in Table 1. The main references supporting these criteria are the state of play and future potential of the Bio-based economy in Europe (European Commission, 2011), the European Commission Communication on bioeconomy (European Commission, 2012a) and the strategy *Innovating for Sustainable Growth: A Bioeconomy for Europe* (European Commission, 2012b). The only additional criterion that is proposed directly by the project Consortium is *available data*, as this will determine the practical feasibility of determining the environmental and socio-economic impact of bio-based products.

The scoring of the case studies is presented in Table 2. Each sub-criteria has been assigned a weighting factor determined by DG-RTD. It was decided that availability of data should not influence the selection of cases, so these sub-criteria were assigned a weighting factor of 0.

Table 1. Scoring model for case studies. The scoring is based on the induced consequences entailed by the introduction of the bio-based product.

Overall criteria	Sub-criteria	0	1	2	3
Market potential	Types of jobs created	Jobs created in rural areas (R) and jobs created in industrial areas (I) expressed as relative potential			
	No. of jobs created	No jobs created	0-5,000 jobs created	5,000-50,000 jobs created	Over 50,000 jobs created
	Market size/volume in specific sectors in terms of total amount of products and incomes, both total and relative (to present market)	Lower than 1,000t bio-products	Between 1,000 and 10,000t bio-product	Between 10,000 and 50,000t bio-products	Over 50,000t bio-products
	Relative projected market growth[5]	Negative	Relative growth between 0 and 5 %/yr	Relative growth between 5 and 10 %/yr	Relative growth over 10 %/yr
Promise for deployment	Projected availability of feedstock (in terms of volume required, length of supply response & eventual technical/environmental/economic supply restrictions)	Considerable delay in feedstock response to demand OR amount required is greater than available stock. Considerable supply restrictions are foreseen.	Considerable delay in feedstock response to demand OR amount required is comparable to available stock. Some supply restrictions are foreseen.	Slight delay in feedstock response to demand OR available stock exceeds reasonably amount required. Some supply restrictions are foreseen.	Required feedstock volume can respond immediately to demand OR available stock exceeds largely amount required. No or very small supply restrictions foreseen.
	Possibility of exchanging the feedstock with a residual feedstock	Technically challenging OR involve prohibitive additional costs OR residual feedstock	Technically possible but technical challenges can be foreseen, with	Technically possible with available technology, with considerable	Technically possible with available technology. Minor additional costs only

[5] Relative growth considering the overall growth of the specific sector.

Overall criteria	Sub-criteria	0	1	2	3
		"available" at level 0 or 1 only (see above).	considerable additional costs (considerable payback time). Residual feedstock "available" at level 2 (see above).	additional costs (considerable payback time). Residual feedstock "available" at level 2 (see above)	(reasonable payback time). Residual feedstock "available" at level 2 or 3 (see above).
	Development status of technology	The technology has not been fully developed and has not been tested	The technology has been developed but not yet tested in large scale	The technology has been developed and tested in large scale	The technology has been tested and is in operation
	Ensuring food security and safety	The product will threaten the food security and food safety	The product can possibly threaten the food security and food safety	The product has no impact on food security and food safety	The product might have potential benefits for improving the food security and food safety
	LCIA data on potential environmental impacts	No data available for this specific product. Data for analogous products	Limited data from various sources. Data not validated	Generic LCA data and to a minor degree producer specific data. Data validated	LCA data from the main contributors to the potential environmental impacts. Data validated
	Life cycle inventory data about feedstock	No data available for this specific feedstock. Data for analogous products	Limited data from other projects. Data not validated	Data from several data sources regarding feedstock production are available including quantification of data uncertainty. Data validated.	Complete data from more sources related to feedstock. Data validated
Available data	Life cycle inventory data about conversion technology	No data available for this specific conversion technology. Data for	Limited data from only one conversion technology. Data not	Data from several, applicable conversion technologies	Complete data from several conversion technology providers.

Overall criteria	Sub-criteria	0	1	2	3
		analogous technology	validated		Data validated
	Life cycle inventory data about EoL scenarios	No data available for the specific EoL scenario	Limited data from only one EoL scenario. Data not validated	Data from several scenarios of EoL handling. Data validated	Complete data on several EoL scenarios from various sited in EU. Data validated
	Product with innovative functionalities	The bio-based product offers reduced functionality compared to the conventional product	The functionality of the bio-based product is equal to the conventional product	The bio-based products offer an improvement in functionality over the conventional product, but does not open up new applications or markets	The bio-based product offers innovative functionality compared to the conventional product, and can potentially be used in new applications or markets
	New collaborations to enhancing synergies and coherence as compared to the production of the fossil-based reference product	No new matches/collaborative partners	Only few new matches/collaborative partners	More than 5 new matches/collaborative partners	More than 10 new matches/collaborative partners
Innovation	Use of scarce resources	Larger than the conventional product	At the same level as the conventional product	Lower than the conventional product	Significantly lower than the conventional product
Potential sustainability benefits	Direct and indirect land use change	Larger than the conventional product	At the same level as the conventional product	Lower than the conventional product	Significantly lower than the conventional product
	Relevance for key policies	No relevance for the subject matter or subjects of any of the four key policies	Only indirect relevance for any of the policy areas	Direct relevance for 1-2 policy areas	Direct relevance for 3-4 policy areas
Policy opportunity	Relevance for key policies	No relevance for the subject matter or subjects of any of the	Only indirect relevance for any of	Direct relevance for 1-2 policy areas	Direct relevance for 3-4 policy areas

Overall criteria	Sub-criteria	0	1	2	3
		four key policies	the policy areas		

Table 2. Scoring of case studies.

		Criteria															Scoring	
		Market potential				Promise for development				Available data				Inno-vation	Potential sustainability benefits			
		Sub-Criteria															Total score	Total weighted score
Case number and type of bio-based product	Bio-based feedstock[a]	Types of jobs created[b]	Size of job creation	Market size	Relative project growth (interest of European bio-based industries – JRC study)	Projected feedstock availability	Possibility of exchange with residual feedstock[c]	Development status of technology	Ensuring food security and safety	LCIA data on potential environmental impact	Life cycle inventory data about feedstock	Conversion technology inventory data	EoL inventory data	New collaborations[d]	Use of scarce resources	(In)direct land use change	Relevance for key EU policies		
Weighting factor per sub-criteria			1	1	3	1	1	1	1	0	0	0	0	1	1	1	3		
1. Bio-based PET bottles	Sugarcane, maize*	R>I, I≥R	2	3	2	3	2	3	1	2	3	2	1	2	2	0	3	31	33
2. Clips, binders and seeding pots	Potato*	R>I	1	1	3	3	3	3	1	2	3	2	1	2	2	0	3	30	34
3. Single-use cups for cold drinks (with low barrier properties)	Maize, sugarcane	I≥R, R>I	1	2	2	3	2	3	1	2	3	2	1	2	2	0	3	29	31

4. Carpet	Maize*	I≥R	1	2	1	3	1	2	1	1	3	1	1	2	2	0	3	24	26
5. Food rigid packaging- food containers	Straw, sugarcane bagasse*	I>R	1	2	2	3	2	2	2	1	3	1	1	2	2	3	3	30	34
6. Beverage bottles (future)	Maize, sugarcane	I≥R, R>I	2	3	1	3	2	1	1	2	3	1	1	3	3	0	3	29	30
7. Single use cutlery	Sugarcane, maize	R>I, I≥R	1	2	2	3	2	2	1	1	3	2	0	2	2	0	3	26	30
8. PVC pipes	Sugarcane*	R>I	2	3	0	3	3	2	1	2	3	2	1	1	2	0	2	27	23
9. Milk packaging	Maize, sugarcane	I≥R, R>I	2	3	2	3	2	3	1	2	3	2	2	1	2	0	2	30	29
10. Agriculture mulching film	Potato, maize*	R>I, I≥R	1	1	3	3	3	3	1	2	3	2	1	2	2	0	2	29	31
11. Food packaging films	Sugarcane	R>I	2	3	2	3	2	3	1	2	3	2	2	1	2	0	3	31	32
12. Single use plastic carrier bags	Potato*	R>I	1	1	3	3	3	3	1	2	3	2	1	2	2	0	3	30	34
13. Car interior panel	Hemp/flax*	R>I	1	2	1	3	0	2	1	1	2	2	0	1	2	0	2	20	21
14. Cellulose insulation	Cellulose (Residual grass from unused EU grassland), (news)paper*	I>R	1	2	1	2	3	2	2	2	2	1	1	1	2	2	1	25	23
15. Liquid epoxy resin adhesives	Glycerine (glycerol as by-product from	I>R	1	1	2	3	3	2	2	1	3	2	0	1	2	0	1	24	24

	biodiesel production*																		
16. Lubricants	Vegetable oils: Rapeseed oil, palm oil	R>I	1	1	3	3	1	3	1	0	3	0	0	1	2	0	3	22	31
17. Single use gloves	Rubber (from rubber tree)	R>I	1	1	1	2	0	3	1	1	1	2	1	1	3	0	2	20	21

a Many of the proposed cases are not 100 % bio-based; these are indicated by a '*'. For these cases, only the bio-based feedstock share of the product has been scored, unless otherwise noted under the sub-criteria.

b This category is scored qualitatively and indicates whether the types of jobs created are mainly rural (R) or industrial (I).

c In the cases where the feedstock is already residual, the scoring for this category is set to 3 by default.

d New collaborations to enhancing synergies and coherence as compared to the fossil-based reference product.

e Scarce resources for this scoring essentially encompasses the use of fossil resources. As earlier noted, some of the bio-based product cases do involve the use of fossil resources.

1.4 Functional units and product systems for the seven cases

Most technologies considered have reached TRL (technology readiness level) nine in the initial selection processes. Still, a few ex-ante assessments were carried out for technologies that are very close to TRL 9. Those assessments are meant to provide perspectives for the promising near future alternatives. They should not be used to meet the primary goal of the study.

A summary of functional units defined for the seven cases are shown in Table 3. The functional units were first proposed by the project team, following the four criteria of *what (function), how much (quantity), how long (duration) and how well (quality)* defined by the PEF guidelines (as well as in the ISO standards).

The definition of functional units has also been discussed and reviewed among academics, policy makers and industrial experts. The detailed definition processes can be found in the individual case study chapters.

For each case study, a so-called "baseline" bio-based case is defined to reflect the status-quo commercial production of the product. Primary data are gathered from industry based on the real supply chains; this also includes the biomass used by the industry presently. Next to the baseline for most cases, alternative scenarios of bio-based products are also explored to see the changes in impacts by sourcing biomass from different feedstocks and/or from different regions.

Table 3. Overview of the functional units and product systems for the seven case studies.

Cases	Functional unit	Bio-based baseline (material and biomass sources)	Bio-based Alternatives	Fossil fuel-based references
1.Beverage Bottles	Packaging of water in 100 0.5-litre bottles providing a shelf life of at least 9 months.	30 % Bio-based PET. Biomass sourced from Brazilian sugarcane.	1. 30 % Bio-based PET. Biomass sources from mixed European ethanol crops. 2. 30 % Bio-based PET. Biomass sourced from European lignocellulosics.	Petrochemical PET
2. Horticultural clips	45 000 single-use clips used for horticultural purposes for one hectare of land	Starch plastics made from European biomass	Starch plastics made from potato waste from food industry	Petrochemical PP
3. Single-use cups	1 000 single-use drinking vessels each for 200 ml of cold beverage.	PLA made from US maize and Thai sugarcane	1. PLA made from European maize. 2. Bio-based PP made from UCO.	1. Petrochemical PP 2. Petrochemical PET
4. Single-use cutleries	1 000 sets of disposable cutleries each consisting of a knife, a fork and a soup spoon.	PLA made from US maize and Thai sugarcane	n/a	Petrochemical PS
5. Agricultural mulch films	Providing field mulching for one hectare of land for six months	Starch plastics made from European biomass	n/a	Petrochemical LDPE
6. Food packaging films	100 m^2 of transparent film packaging for fresh vegetables for one week.	PLA made from US maize and Thai sugarcane	1. PLA made from European maize. 2. Bio-based PP made from UCO	Petrochemical PP
7. Single-use carrier bags	1 single-use all-purpose lightweight plastic carrier bag with the volume of 20 liters and 10 kg weight holding capacity	Starch plastics made from European biomass	1. Bio-based LDPE made from sugarcane from Brazil 2. Bio-based LDPE made from mix EU ethanol crops	Petrochemical LDPE

Abbreviations:

n/a: not applicable or not available.

UCO: Used cooking oil.

PET: polyethylene terephthalate. PLA: polylactic acid. PP: polypropylene. PS: polystyrene. UCO: used cooking oil. LDPE: low-density polyethylene.

1.5 Impact assessment methods

Since the goal of this study is to support policy-making at the EU level, it is important to cover a wide range of environmental impacts. In this study, all 16 mid-point environmental impact categories recommended by the PEFCR guidance draft version 6.3 (European Commission, 2017) are included, with a few exceptions:

- Particulate matter. By the time this study was conducted, it was not clear which impact assessment model would be recommended for this category. We therefore adopt the impact category but used the model recommended by the PEF guidance (2013) (which is also included in ILCD 2015, Benini et al., 2014).

- Land use impact category. The LANCA model and the characterisation factors, recommended by PEFCR guidance 6.2 for the assessment of land use impacts, are currently not available in SimaPro (version 8.4) and it is unlikely to be included in the next update. The project team decided to use the land transformation indicator (indicated as "Soil Organic Matter") recommended by the PEF guidance (2013) instead. Table 4 provides an overview of the impact categories selected in this study.

- Abiotic depletion (AD) and Abiotic depletions (AD fossil fuels) are renamed into "Resources (minerals and metals" and "Resources (fossils) in PEFCR guidance version 6.3. The impact assessment models remain the same.

In addition to the 16 PEFCR recommended impact categories, the project team added two indicators as additional information. Non-renewable energy use (NREU) and an end-point indicator Global Temperature Change Potential (GTP).

Non-renewable energy use (NREU) is defined as total cumulative primary fossil fuels and nuclear energy demand. The indicator is selected because cumulative fossil energy demand is a good proxy of many energy-related impacts (Huijbregts et al., 2006). Note that the current energy indicator in the LCA "Abiotic depletion (fossil fuels)" is defined as cumulative fossil fuels demand including all direct and indirect energy use, in primary energy term. NREU is not a PEFCR guidance recommended indicator. It overlaps AD (fossil fuels). However, NREU is one of the most reported environmental indicators in literature. It is a useful indicator to crosscheck all the energy-related impacts categories with reported values in literature.

GTP is an end-point environmental indicator. Unlike climate change impact, which indicates the radiative forcing of a substance over a given time period and relative to that of CO_2, GTP is "the ratio of change in global mean temperature at a chosen point in time from the substance of interest relative to that from CO_2" (Myhre et al., 2013). Note that GTP has a higher uncertainty than climate change impact (Myhre et al., 2013). In this project, we choose GTP as an additional climate change impact indicator. The full list of GTP characterisation factors can be found in the appendix of Chapter 8 of the IPCC 5th assessment report (WGI).

Table 4. List of impact categories and assessment models used in this study.

Impact Category	Unit	Impact assessment Models
PEFCR guidance recommended mid-point impact category		
Climate change	kg CO_2 eq.	GWP 100a IPCC 2013, with carbon climate feedback[a]
Ozone depletion	kg CFC-11 eq	ILCD 2011 (WMO, 1999)
Human toxicity, non-cancer effects	CTUh	USEtox (recommended + interim) (Rosenbaum et al., 2008)
Human toxicity, cancer effects	CTUh	USEtox (recommended + interim) (Rosenbaum et al., 2008)
Particulate matter[b]	kg PM2.5 eq	ILCD 2011 (Rabl et al., 2014)
Ionizing radiation HH (Human health)	kBq U235 eq	ILCD 2011 (Frischknecht et al., 2000)
Photochemical ozone formation	kg NMVOC eq	ILCD 2011 (van Zelm et al., 2008)
Acidification	molc H+ eq	ILCD 2011 (Posch et al., 2008)
Terrestrial eutrophication	molc N eq	ILCD 2011 (Posch et al., 2008)
Freshwater eutrophication	kg P eq	ILCD 2011 (Struijs et al., 2009)
Marine eutrophication	kg N eq	ILCD 2011 (Struijs et al., 2009)
Freshwater ecotoxicity	CTUe	USEtox (recommended + interim) (Rosenbaum et al., 2008)
Land transformation[c]	kg C deficit	Soil Organic Matter model (Milà i Canals et al., 2007)
Water use	m3	AWARE factors (Boulay et al., 2015)
Abiotic depletion[d]	kg Sb eq	ILCD 2011/CML-IA baseline (van Oers et al., 2002)
Abiotic depletion (fossil fuels)[d]	MJ	ILCD2011/CML-IA baseline (van Oers et al., 2002)
Indicators included in the study but not listed by the PEFCR		
Non-renewable energy use. (NREU)	MJ	Cumulative Energy Demand
GTP 100 (end-point)	kg CO_2 eq.	IPCC 2013, WG I, Chapter 8 (Myhre et al., 2013)

a See Table 13 in PEFCR guidance v6.3 (European Commission, 2017). The CFs of methan (fossil)

is 36.75 kg CO_2 eq., methane (biogenic) is 34 kg CO_2 eq.

b In the meantime the impact assessment model recommended has been updated by using Fantke et al., 2016. This information was not available at the time when this study was conducted.

c Indicator taken from the current PEF guideline. PEFCR requires using the LANCA model, which is currently not available in SimaPro.

d "Abiotic depletion" and "Abiotic depletion (fossil fuels)" are renamed into "Resources (mineral and metal" and "Resources (Fossils)" in PEFCR guidance v6.3.

2 Life cycle inventory modelling

2.1 Overall approach

The life cycle inventory models in this study are built based on the so-called modular approach. Figure 6 illustrates a generic flowsheet of a bio-based product system. The inventory models are divided into five modules in a generic concept:

- Biomass feedstock production (including models of direct and indirect land use changes);
- Production of intermediates or platform chemicals where harvested biomass is converted into intermediate chemicals or materials;
- Conversion of material into final product and distribute to the end users;
- Use phase; and
- EoL waste management.

Key unit processes are identified within each module. Primary data on the activity level are collected for the unit processes. The LCA results are presented at the module-level breakdown in the interpretation step.

The modular approach allows to investigate any "drop-in" replacements to a certain pathway, e.g. a different type of biomass feedstock. The modular approach and the drop-in replacement can only be conducted if the subsequent chemical conversion technologies remain unchanged. Several alternative scenarios of bio-based products are constructed using this modular approach.

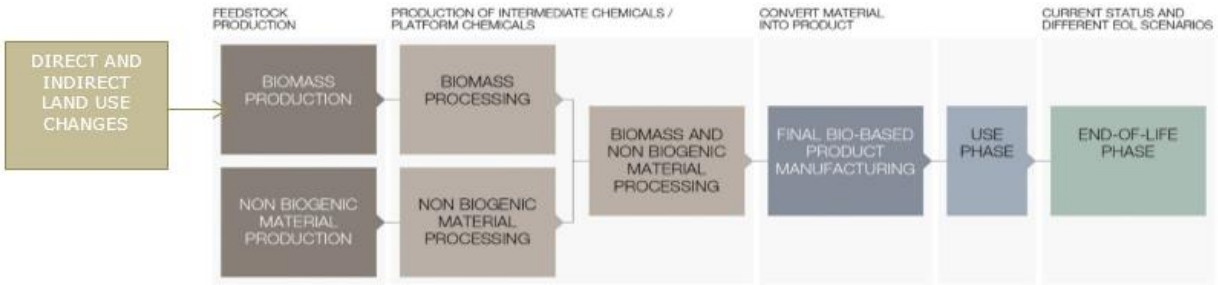


Figure 6. Generic process diagram of the life cycle of bio-based products.

The biogenic carbon removal is accounted for in the cradle-to-factory gate phase. This means that the embedded biogenic carbon provides a credit for the bio-based products. The embedded carbon will then be released again in the atmosphere at the EoL stage. These biogenic carbon emissions can be in the form of CO_2 if fully oxidised at EoL, or partially oxidised and form CO and/or CH_4 or stored in a compost. In short, we present the full carbon cycle for different life cycle stages.

In the assessment of land use changes, the biogenic carbon flows are modelled with "land use change" elementary flows as suggested by PEFCR guidance.

2.2 Biomass feedstock production and conversion to final products

The selected cases involve the following intermediates: bio-based polylactic acid (PLA), polyethylene terephtalate (PET), low density polyethylene (LDPE), and starch plastics (first priority cases). Second priority cases include polypropylene (PP) from EU maize and lignocellulose-based PET; these second priority cases are however not discussed herein.

To select the feedstock to consider for these intermediates, as well as the overall production process, an effort was made to reflect the current European market. This means that feedstock (type and location) were not selected based upon crop production statistics, but upon the leading companies producing the above-mentioned intermediates for the European market.

Market reports, interviews with industrial experts and published literature were used to identify the major current industrial producers of these bio-based intermediates for the European market. Non-disclosure confidentiality agreements (NDA) were signed with some of these key companies, so the data used in this study is an attempt to reflect the exact feedstock (type and region of origin) currently being used to produce the bio-based intermediates used for the product cases targeted herein. Proxy were built up to represent companies that are key players and for which no NDA could be obtained.

Some intermediates involved more than one key producer. When this was the case, a weighted average was made (based on the companies installed capacities). For example, if there are two key leading companies producing the intermediate PLA (e.g: company 1 with an installed capacity of 33 % and company 2 with an installed capacity of 67 %), with company 1 using feedstock A (from country Z) and company 2 using feedstock B (from country Y), then the following was considered: 33 % of the PLA is from feedstock A (country Z), and 67 % from feedstock B (country Y).

Table 5 details the feedstock selected in this study (type, location), per intermediate and case study. In a nutshell, 8 "feedstock x location" combinations are involved: (i) sugarcane x Brazil; (ii) sugarcane x Thailand; (iii) maize x USA (Nebraska and Iowa); (iv) maize x Italy; (v) maize x Europe's average[6] (vi) potato x Germany; (vii) wheat x Europe's average; (viii) sugar beet x Europe's average.

[6] Maize, wheat and sugar beet from "Europe's average" belong to a fictive case (feedstock for European ethanol). For the case of maize, this also applies for the fictive "European PLA from maize" variant. For these three feedstocks, an analysis of production data within the Eurostat database was made, and a 5-y average was used over the last 5 years of available data. On this basis, the following mixes were selected: (i) for wheat: France (48 %), Germany (33 %), United Kingdom (19 %); (ii) for sugar beet: France (48 %), Germany (36 %) and Poland (16 %); for maize: France (36 %), Romania (28 %), Hungary (18 %) and Spain (18 %). In all these cases, the selected countries represent at least 50 % of the EU production (average over 2013-2017). It should be noted that the countries selected herein for sugar beet may not be representative in the medium-term, due to abandon of the sugar beet quota.

Table 5. Feedstock included in the study, per case and key intermediate (priority cases only).

Intermediate	Final product case (case ID)	Associated feedstock[a]	Data source[c]
Ethanol -> Ethylene	Beverage bottles (case study 1)	Sugarcane (BR)	Data from 2 companies
		EU ethanol feedstock, 1st generation (36 % maize, 27 % sugar beet, 37 % wheat) [b]	Not based upon industrial production, fictional
	Bio-based LDPE single use carrier bags (case study 7)	Sugarcane (BR)	Literature data from industry sources
		EU ethanol feedstock, as described for case 1[b]	Not based upon industrial production
Starch/TPS[d]	Horticultural application (clips) (case study 2)	Potato (DE) and maize (IT).	Data from 1 company + proxy
		Potato waste (peels and other leftovers from the food industry)	Proxy
	Agricultural mulching film (case study 5)	Maize (IT)	Data from 1 company
	Single-use carrier bags (case study 7)	Potato (DE) and maize (IT)	Data from 1 company + proxy
Lactic Acid	Single-use cups (cold) (case study 3)	Maize (US) and sugarcane (TH)	Data from 2 companies
	Food packaging films (case study 6)	Maize (US, avg. EU) and sugarcane (TH)	Data from 2 companies
	Single-use cutlery (case study 4)	Maize (US) and sugarcane (TH)	Data from at least 4-5 companies

a BR: Brazil; EU: European Union; IT: Italy; US: United States; TH: Thailand;

b Extrapolated from Epure (2017). This average from 2016 production may underestimate the proportion of sugar beet ethanol, given the recent release of the sugar beet quota. Yet, the time scope of the LCA is the immediate short-term, and the current estimates were considered as the best representation for this time horizon;

c Company names and the considered shares for each when making weighted averages are not disclosed herein in virtue of the signed NDAs;

d Thermoplastic starch.

2.3 Data sources and verification procedure

It is the ambition of the project to gather primary data from industry to conduct full LCAs for the selected bio-based products, using consistent methodology and reflecting the status-quo environmental impacts to support policy decisions. The environmental impacts are then compared with those of reference products made from fossil fuel resources.

Literature data, rather than primary data, are used for the impacts of the reference products made from fossil fuels. The project team acknowledges that this could potentially create an imbalanced view. Efforts were made to obtain transparency from the reported literature data. A critical review was conducted for such a purpose. The method of critical review is documented in section 6 of this chapter. The result of the critical review is documented in Chapter "Critical review of petrochemical references".

The foreground data are gathered with available literature and/or directly primary data from relevant companies and stakeholders involved. The foreground models are constructed in MS Excel (from cradle-to-user) and EASTECH for the grave stage (i.e. EoL). This allows ensuring a proper mass balance of the overall substance flows as the initial feedstock undergoes a series of conversion processes. If literature data, rather than company data, needs to be used as foreground data, thorough reviews are conducted to obtain plausible ranges reported by literature. The selected data are verified by independent sources wherever possible. If the literature provides none plausible ranges or data could not be easily verified by any independent sources, sensitivity analyses are conducted in the interpretation step.

The background data are retrieved from state-of-the-art available LCA databases, e.g. Ecoinvent v3, Gabi 2017, PlasticEurope's eco-profiles 2012-2017, Agri-Footprint and EASTECH among the most relevant.

The foreground data are imported into SimaPro to carry out the impact assessment calculation (and to use the database as the background data).

For each case study report, a summary table is provided for both the sources used for fore- and background data.

2.4 Data quality requirements and registration

For all activity level data, the data quality is scored/registered based on three parameters: the technological representativeness (TeR), the geographical representativeness (GR) and time-representativeness (TiR). Table 6 shows the scoring criteria recommended by the PEFCR for these three parameters.

Table 6. Assign scores to parameters in the DQR formula when secondary datasests are used (source: PEF)

	TiR	TeR	G_R
1	The EF report publication date happens within the time validity of the dataset.	The technology used in the EF study is exactly the same as the one in scope of the dataset.	The process modelled in the EF study takes place in the country the dataset is valid for.
2	The EF report publication date happens not later than 2 years beyond the time validity of the dataset.	The technologies used in the EF study are included in the mix of technologies in scope of the dataset.	The process modelled in the EF study takes place in the geographical region (e.g. Europe) the dataset is valid for.
3	The EF report publication data happens not later than 4 years beyond the time validity of the dataset.	The technologies used in the EF study are only partly included in the mix of technologies in scope of the dataset.	The process modelled in the EF study takes place in one of the geographical regions the dataset is valid for.
4	The EF report publication data happens not later than 6 years beyond the time validity of the dataset.	The technologies used in the EF study are similar to those included in the scope of the dataset.	The process modelled in the EF study takes place in a country that is not included in the geographical region(s) the dataset is valid for, but sufficient similarities are estimated based on expert judgement.
5	The EF report publication data happens late than 6 years after the time validity of the dataset.	The technologies used in the EF study are different from those included in the scope of the dataset.	The process modelled in the EF study takes place in a different country than the one dataset is valid for.

The data quality for EoL modelling in the EASTECH model has been analysed regarding the PEF requirements. EASETECH provides a data quality score inspired by a quality assessment of processes system introduced by Weidema and Wesnæs (1996). This score was translated to match the PEF quality scoring. Data from other sources than EASETECH (Ecoinvent, scientific, articles, internet etc.) have been scored based on the PEF requirements. Data quality scoring is only registered for the activity level processes for each EoL technology (see Annex 1).

2.5 Procedure of multi-functionality processes

Multi-functionality processes are handled according to the following decision hierarchy (according to the PEFCR):

1) Subdivision or system expansion should be applied to avoid allocation;

2) Allocation based on a relevant underlying physical relationshio; and

3) Allocation based on some other relationships (e.g. economic values).

The detailed decisions of each occurrence of multi-output processes are described in each case study report chapter.

3 End of Life inventory modelling

The end of life (EoL) treatment of the products, i.e. from end user to grave (final disposal), is modelled in the LCA modelling software EASETECH.

Six different EoL treatment technologies representing average European waste treatment have been modelled:

- Mechanical plastic recycling (referred to as recycling)
- Municipal solid waste incineration with and without energy recovery (MSWI) (referred to as incineration)
- Municipal solid waste landfill (referred to as landfilling)
- In-situ field-based biodegradation
- Industrial composting
- Anaerobic digestion

Littering has not been included as an EoL disposal option in this LCA study, even though littering is a serious challenge in the EU. Littering, and the reasons for excluding this from the study, are further discussed in section 3.1.4.

The following sections describe the EASETECH software as well as input data, material- and energy substitution and other general assumptions for the EoL modelling. Input data includes detailed material composition for each product system, which EASETECH utilises to calculate the environmental impacts.

The robustness of the results has been tested by applying a range of sensitivity analyses. The scenarios applied in the sensitivity analyses are described in chapter 5.

Details regarding the seven case studies can be found in the case study specific chapters later in this report. Additional technical details and values for the modelling are presented in Annex 1 and additional results are presented in Annex 4.

3.1 Scope and scenarios

The seven case studies each have several product systems:

- A bio-based baseline;
- Up to two types of bio-based alternatives; and
- The petrochemical reference(s),

Each product system has several relevant EoL scenarios, an intended EoL option for the bio-based product systems and an EoL mix.

3.1.1 Relevant EoL scenarios modelled individually (100 % scenarios)

For each case study, the relevant EoL technologies will be modelled for the relevant product systems. For each specific product all relevant EoL technologies are evaluated. For each product this may be up to five different EoL technologies. Treatment of the products in all relevant EoL technologies has been modelled assuming 100 % treatment in that EoL technology. An overview of the EoL options for each case study's product systems can be seen in Table 7. The different types of plastic have different EoL options, as specified below:

Bio-based non-biodegradable plastic product systems (PET, PP and LDPE in case studies 1, 3, 6 and 7) have the following EoL options:

- Recycling
- Incineration
- Landfilling

Bio-based biodegradable plastic product systems (Starch plastic, PLA mix and PLA in case studies 2, 3, 4, 5, 6 and 7) have the following EoL options:

- Recycling (only PLA)
- Incineration
- Landfilling
- Industrial composting
- Anaerobic digestion

Bio-based biodegradable plastic product systems in agricultural applications (Starch plastic in case studies 2 and 5) have the following EoL options:

- Incineration
- Landfilling
- In-situ field-based biodegradation

Petrochemical references (PET, PP, PS and LDPE in all case studies) have the following EoL options:

- Recycling (except case study 2)
- Incineration
- Landfilling

3.1.2 Intended End of Life

For each baseline product system and bio-based alternative an "intended EoL" option was selected. The intended EoL options were chosen as follows:

- Biodegradable starch field application products (case study 2, clips, and case study 5, mulch film): In-situ field-based biodegradation was chosen as the intended EoL technology, as this is the EoL technology recommended by the producer.
- Biodegradable PLA (case study 3, cups): Recycling *and* industrial composting were chosen as the intended EoL technologies.
- Biodegradable PLA with food leftovers (case study 2, clips and case study 4, single-use cutlery): Industrial composting was chosen as intended EoL technology.
- Biodegradable starch plastic (case study 7, carrier bags): Industrial composting was chosen as the intended EoL technology.
- Bio-based non-biodegradable plastics (case study 1, beverage bottles, and as alternative product systems in several other case studies): Recycling was chosen as the intended EoL technology, since this is assumed to be the EoL technology recommended by the producer.

The choices of intended technology have in many cases been confirmed by the industry, i.e. either the producers or the waste management sector. Additionally, the intended EoL choice is in line with the European Circular Economy package (2018), which focuses on collection of plastics for recycling and sets targets for separate organic waste collection in 2023.

The intended EoL technologies for the bio-based products are shown in Table 7, and are highlighted in the EoL modelling results with a black box in the case study chapters.

3.1.3 European End of Life mix

In addition to modelling the individual EoL options (assuming 100 % EoL treatment by the specific EoL technologies), mixed EoL scenarios, representing the mix of current EoL treatment technologies in the EU, have also been modelled as shown in Table 7.

The EoL mix scenarios are based on:

- Bio-based non-biodegradable plastic product systems (PET, PP and LDPE in case studies 1, 3, 6 and 7): The estimated current EoL technology mix, as for the petrochemical reference, since the waste disposal is assumed to be similar for these waste streams.

  The collection efficiency of the bio-based non-degradable plastic (the citizen's ability to separate plastic from MSW) is assumed to be similar to the current collection efficiency of petrochemical plastic for recycling.

- Bio-based biodegradable plastic product systems (Starch plastic, PLA mix and PLA in case studies 2, 3, 4, 5, 6 and 7): The estimated current EoL technology mix, as for the petrochemical reference, since the current limited market penetration of the bio-based products makes it difficult to estimate the current EoL technology mix. According to Geyer et al., (2018), the bio-based plastics represent less than 1 % of the market. A literature search has confirmed that there is little information on the current waste management of bio-based plastics. Therefore, the bio-based products are often assumed to follow the petrochemical products (Gironi, 2010).

  The collection efficiency of the bio-based biodegradable plastic is assumed to be similar to the current collection efficiency of petrochemical plastic for recycling.

  Recycling is not always suitable for bio-based biodegradable plastic and therefore the following treatment technologies are applied for the part of the waste stream where the petrochemical plastic is separated for recycling:

- Industrial composting for starch plastic and PLA mix;
- Recycling and industrial composting for PLA; and
- In-situ for agricultural products.
- Petrochemical references (PET, PP, PS and LDPE in all case studies): The estimated current EU EoL technology mix.

The EoL technology mix is mostly based on data from European Commission (2018), where it is stated that general treatment for plastic waste within the EU is 39 % incineration, 31 % landfill and the remaining 30% is collected for recycling. This figure of 30 % for plastic waste collected for recycling is further supported by 2014 data from Plastics Recyclers Europe (2015). However, there is a difference between polymer types in terms of how much is recycled, depending on the economic value of the recycled polymer and the recycling technology available for the polymer. A part of the plastic waste that is collected for recycling is sorted out and not recycled. In this study it is assumed that all plastic rejects are incinerated. The sorting and technology efficiencies

for each of the polymer types used in this study are presented in detail in chapter 3.4.4 Plastic recycling.

In cases where the collected waste to recycling is different from 30 %, the share to incineration and landfilling is calculated with the division between these technologies remaining the same, unless otherwise noted.

The assumptions are further explained in the case study chapters. The petrochemical product systems should provide the references closest to the current waste management systems, while empirical data for waste management of the bio-based (and biodegradable) products is very limited.

Some of the case studies are chosen partly due to current low recycling or high degree of littering of the specific product. Therefore, the application of a general recycling percentage for bio-based as well as petrochemical plastic (based on how much plastic is currently collected for recycling in the EU) in the EoL technology mix may be questioned for some case studies. However, keeping the share of recycling constant between different product systems in each case study ensures that the comparison of bio-based and petrochemical products is not biased by assumed differences in the waste system.

Table 7 presents the EoL scenarios modelled for each product system under each case study. This includes the definition of relevant EoL technologies, the 'intended' EoL technology option and the assumed EoL technology mix for each product system. In the full LCA modelling of the case studies (cradle-to-grave modelling) the EoL technology mix scenarios are applied. For plastic recycling and industrial composting, the rejects (the difference between collection rate and treatment rate) are incinerated, see Figure 8 and Figure 12.

Table 7. Overview of the EoL options for each product system, their intended EoL option and the estimated EoL mix of EU.

Case study #	Product system	Relevant EoL scenarios modelled individually	Intended EoL option	Estimated EoL mix in EU	Comment and references to the estimated EoL mix in EU
		100 % scenarios		Collection rate (Treatment rate). Used in the cradle-to-grave modelling.	
1. Beverage bottles	Bio-based PET	Recycling Incineration Landfilling	Recycling	Recycling: 60 % (45 %) Incineration: 20 % Landfilling: 20 %	Same EoL mix as PET bottles (EPBP, n.d.); PlasticsEurope, 2016b; PlasticsEurope, 2013; European Commission, 2018). Explained further in the case study chapter.
	Petrochemical PET	Recycling Incineration Landfilling		Recycling: 60 % (45 %) Incineration: 20 % Landfilling: 20 %	
2. Single-use clips	Bio-based starch plastics	Incineration Landfilling In-situ	In-situ	In-situ: 100 %	Estimated EoL mix, see case study chapter for further explanation.
	Petrochemical PP	Incineration Landfilling		Incineration: 56 % Landfilling: 44 %	Same EoL mix as plastic waste excluding recycling. Based on factors reported by European Commission (2018).
3. Single-use cups for cold drinks	Bio-based PLA	Recycling Incineration Landfilling Industrial composting Anaerobic digestion	Industrial composting and recycling	Industrial composting: 15 % (11 %) Recycling 15 % (11 %) Incineration: 39 % Landfilling: 31 %	EoL mix for plastic waste based on data from European Commission (2018). See further explanations for the recycling in chapter 3.4.4.
	Bio-based PP	Recycling Incineration Landfilling	Recycling	Recycling: 30 % (21 %) Incineration: 39 % Landfilling: 31 %	EoL mix for plastic waste based on data from European Commission (2018). See further explanations for the recycling in chapter 3.4.4.
	Petrochemical PP	Recycling Incineration Landfilling		Recycling: 30 % (21 %) Incineration: 39 % Landfilling: 31 %	
	Petrochemical PET	Recycling Incineration Landfilling		Recycling: 30 % (21 %) Incineration: 39 % Landfilling: 31 %	

4. Single-use cutlery	Bio-based PLA mix	Incineration Landfilling Industrial composting Anaerobic digestion	Industrial composting	Industrial composting: 30 % (21 %) Incineration: 39 % Landfilling: 31 %	EoL mix for plastic waste based on data from European Commission (2018). see case study chapter for further explanation.
	Petrochemical PS	Recycling Incineration Landfilling		Recycling: 30 % (7.5 %) Incineration: 39 % Landfilling: 31 %	EoL mix for plastic waste based on data from European Commission (2018). See further explanations for the recycling in chapter 3.4.4.
5. Agricultural mulching film	Bio-based starch plastic	Incineration Landfilling In-situ	In-situ	In-situ: 100 %	Estimated EoL mix, see further explanation in the case study chapter.
	Petrochemical LDPE	Recycling Incineration Landfilling		Recycling: 5 % (2 %) Incineration: 53 % Landfilling: 42 %	EoL mix for agricultural plastic, see case study chapter for further explanation.
6. Food packaging films	Bio-based PLA	Recycling Incineration Landfilling Industrial composting Anaerobic digestion	Industrial composting	Industrial composting: 15 % (11 %) Recycling 15 % (11 %) Incineration: 39 % Landfilling: 31 %	EoL mix for plastic waste based on data from European Commission (2018). See further explanations for the recycling in chapter 3.4.4.
	Bio-based PP	Recycling Incineration Landfilling	Recycling	Recycling: 30 % (21 %) Incineration: 39 % Landfilling: 31 %	EoL mix for plastic waste based on data from European Commission (2018). See further explanations for the recycling in chapter 3.4.4.
	Petrochemical PP	Recycling Incineration Landfilling		Recycling: 30 % (21 %) Incineration: 39 % Landfilling: 31 %	
7. Single-use carrier bags	Bio-based LDPE	Recycling Incineration Landfilling Reuse as carrier bag Reuse as waste bag	Recycling	Recycling: 30 % (21 %) Incineration: 39 % Landfilling: 31 %	EoL mix for plastic waste based on data from European Commission (2018). See further explanations for the recycling in chapter 3.4.4.
	Bio-based starch plastic	Incineration Landfilling Industrial composting Anaerobic digestion	Industrial composting	Industrial composting: 30 % (21 %) Incineration: 39 % Landfilling: 31 %	EoL mix for plastic waste based on data from European Commission (2018), see case study chapter for further explanation.

		Reuse as carrier bag Reuse as waste bag Reuse as organic waste bag			
	Petrochemical LDPE	Recycling Incineration Landfilling Reuse as carrier bag Reuse as waste bag		Recycling: 30 % (21 %) Incineration: 39 % Landfilling: 31 %	EoL mix for plastic waste based on data from European Commission (2018). See further explanations for the recycling in chapter 3.4.4.

3.1.4 Excluding plastic littering from the End of Life modelling

Many plastic products are designed to resist degradation, even though the products are often only used once before being disposed of. Although a lot of the waste ends in the intended EoL technology, e.g. recycling or incineration, a lot of waste is littered either intentionally or through illegal dumping. Littering is a serious problem throughout the world, and accumulating amounts of plastic are seriously affecting ecosystems.

Littered plastic is often addressed as either macro plastic (larger than 5 mm), micro plastic (smaller than 5 mm) or nano plastic (smaller than 100 nm). Globally, more macro plastics than micro plastic is littered; however, large regional differences apply (Pant et al., 2018). After littering, the plastic will with varying speed and to varying extents disintegrate into smaller pieces (macro plastic – micro plastic – nano plastic) and eventually it might degrade. This is caused by mechanical forces and by UV light (Moore, 2008). The rate of disintegration and degradation depends on the type of plastic and the environmental conditions.

According to Plastic Waste Systems (PWS, n.d), the main reasons for the magnitude of the littering problem are:

- Plastic has little perceived residual value after use,
- Plastic products often have no other use than the one single use they were designed for (e.g. packaging),
- The plastic products are cheap to manufacture,
- It is difficult to avoid buying plastic items or items wrapped in plastic.

Some data for plastic items found on beaches and in the ocean are available. Based on these data, it has been estimated, that 5-10 million tonnes of plastic is littered to land and oceans every year, corresponding roughly to 1-3 % of the produced plastic items worldwide (Pant et al., 2018). With an increasing amount of plastic produced every year, this is a growing challenge to be dealt with.

The new proposed single-use plastics directive by the European Commission addresses the issue looking at the 10 single-use plastic products most often found on European beaches and in oceans. The European Commission estimates that these products constitute 43 % of all marine litter (European Commission, 2018a). Among these ten single-use products, four are addressed in this study (cutlery, cups for cold beverages, beverage bottles and carrier bags).

This study addresses biodegradable plastic products. Biodegradable plastic is not necessarily easily (bio-)degradable in the environment, since the plastic is often designed for degradation in industrial treatment facilities, i.e. industrial composting, requiring certain conditions regarding temperature, oxygen level etc. The products in question are often tested according in standards like EN 13432 and EN 14045. Since these industrial composting conditions are not necessarily available in the environment, where the biodegradable plastic is littered, the degradation of the bio-based plastic may not be very different from degradation of fossil-based plastic. However, the Open-Bio project researched how bio-based products have properties which regarding littering in e.g. marine environments, mitigate some of the negative impacts (CalRecycle, 2012; Open-Bio, 2016). Furthermore, other products like agricultural mulch film comply with the standard EN 17033 making them biodegradable on farm land.

Even though littering is a highly relevant aspect, it has not been modelled in this LCA study since no standard impact category/categories have been developed for modelling and assessing littering in LCA. The effects from littering might also be addressed via some of the existing impact categories such as those related to toxicity and perhaps land use (visual effects). Modelling of littering would require a wide range of data, of which

many are currently not available. The data gaps for modelling of littering include e.g. (Pant et al., 2018):

- Fate, exposure and effect modelling for littered macro and micro plastics (specifically depending on geography and environmental compartment);
- Product specific degradation rates and leakage of additives (also for conventional waste treatment, especially landfilling);
- Quantification of physical impacts (e.g. effects from entanglement and ingestion of larger plastic particles);
- Quantification of chemical impacts (e.g. effects due to micro plastic formation and chemicals/additives in the plastic);
- Quantification of biological impacts (e.g. micro plastics as carriers of germs/alien species).

Thus, it would require further data and development of existing and new impact categories to account for the full environmental impacts from littering of plastic to different ecosystems. Development of a new LCA methodology is not part of this study and the effect of littering is therefore not included in the LCA assessment.

However, the issue of littering has high public and political attention and the issue will therefore be discussed at a qualitative level when relevant in the presentation of the EoL modelling results. Proper waste management is a key solution to plastic littering (STAP, 2011). Jambeck et al. (2015) emphasise the connection between improper waste management and the amount of plastics entering the sea.

3.2 Modelling tool

EASETECH has been chosen as the modelling tool for the EoL LCA, because EASETECH provides a state-of-art software tool within LCA modelling for waste management (Clauvreul et al., 2014). Compared to other LCA tools, EASETECH focuses on material flow modelling and specified EoL technologies allowing the user to apply either standard data or case specific data.

EASETECH was developed at the Technical University of Denmark. The tool is used for environmental impact assessment for different waste systems, including comparison of different scenarios for EoL treatment.

EASETECH provides a possibility for defining the exact detailed chemical composition of the waste fractions investigated. This provides an advantage compared to most other LCA tools, where the waste fractions are often included as more standard fractions with specific composition, e.g. general Municipal Solid Waste (MSW) or a general plastic. Differentiating between different types of plastic would thus not be possible in simpler LCA tools. Based on the material inputs, the mass flow, resource use and emissions from processes, impacts can be estimated. Where waste material utilisation, energy recovery or material substitution is possible, the model will account for potential savings by avoiding the corresponding production of virgin materials.

EASETECH is equipped with a database that includes central technologies and sub processes within waste management, such as incineration, recycling, landfilling, anaerobic digestion, electricity, transportation etc. For each treatment technology, several parameters can be modified. The database contains a range of specific treatment scenarios based on actual data from existing treatment plants. These may be used directly or modified due to the user's specific data.

EASETECH is designed for the easy set-up of waste management scenarios. The processes modelled can handle heterogeneous material flows in different pathways and calculate associated emissions. An example is that landfilling consists of a large number

of sub processes, where the user can choose between different types of landfills, precipitation level, gas collection and utilisation systems, landfill cover systems etc.

EASETECH also contains a detailed module for modelling the use of organic waste for agricultural purposes. This module can process data on carbon sequestration in soil, nitrogen emissions from various fertiliser products and utilisation of compost and digestate for fertiliser substitution.

3.3 EoL-reference flow, definition and composition

The main input to EASETECH is the detailed chemical composition of each material. The input materials assessed in this project are the plastic products with associated contamination, including possible food leftovers or soil. For each case study, an overview of the material compositions will be given in the case study chapters and the full material compositions are provided in Annex 1. The combination of the input materials and their weight describes the EoL-reference flows for the EoL-modelling:

- 1 kg plastics + 100 g contamination + X kg possible food leftover or soil

An overview of the plastics, the contamination and the possible food leftover or soil is presented in Table 8. The amount of food leftovers and additional soil can be observed in Table 9 and Table 10.

Table 8. Overview of each product system's plastics, contamination and possible food leftover or soil.

Case study #	Plastic	Contamination	Food leftover / soil
1. Beverage bottles	Bio-based PET	Organic	
	Petrochemical PET	Organic	
2. Single-use clips	Bio-based starch plastics	Soil	Additional soil
	Petrochemical PP	Soil	Additional soil
3. Single-use cups for cold drinks	Bio-based PLA	Organic	
	Bio-based PP	Organic	
	Petrochemical PP	Organic	
	Petrochemical PET	Organic	
4. Single-use cutlery	Bio-based PLA mix	Organic	Food leftover
	Petrochemical PS	Organic	Food leftover
5. Agricultural mulching film	Bio-based starch plastic	Soil	Additional soil
	Petrochemical LDPE	Soil	Additional soil
6. Food packaging films	Bio-based PLA	Organic	Vegetable food leftover
	Bio-based PP	Organic	Vegetable food leftover
	Petrochemical PP	Organic	Vegetable food leftover
7. Single-use carrier bags	Bio-based LDPE	Organic	
	Bio-based starch plastic	Organic	
	Petrochemical LDPE	Organic	

3.3.1 Plastic composition

Each case study includes two, three or four different material compositions depending on the number of product systems in the specific case study (see Table 7). The chemical composition of the petrochemical materials (PET, PP, PS, LDPE) is obtained from Götze et

al. (2016) – based on waste characterisation of different fractions in residual and source segregated household waste.

The chemical composition of the bio-based non-biodegradable plastics (bio-based PET, PP, LDPE) is based on the same data by exchanging the fossil carbon with biogenic carbon content in the material composition in EASETECH. The composition of bio-based biodegradable plastics (starch plastic and PLA) is obtained from producers or literature (further details in the specific case study chapters).

3.3.2 Contamination composition

The composition of petrochemical and the bio-based non-biodegradable plastics from Götze et al. (2016) already account for a certain level of contamination with residual waste, since these analyses are performed on plastics from waste characterization surveys (e.g. MSW). Therefore, no further contamination is added when these data are applied. Based on stoichiometric calculation and assessment of the inherent substances, the general contamination has been estimated to be approximately 10 % of the material weight. In Götze et al. (2016) plastic sorted from residual waste as well as from source segregated plastic waste was analysed and contamination ranged from 2 % to 18 %.

For the bio-based biodegradable material compositions the same level of contamination (10 %) was added. This contamination consists of organic waste (composition is standard in EASETECH based on Petersen et al., 2011 and Petersen et al., 2012).

Hence, the reference flow of 1 kg of plastic will include 100 g of contamination, which is factored into the model.

For the two case studies concerning waste from products used in agriculture (Case study 2, Clips, and Case study 5, Mulching film), the contamination consists of soil and other garden waste (EASETECH, based on Boldrin & Christensen 2008 and Lehtomäki & Björnsson 2006). This will be further addressed below in section 3.3.4 Additional soil.

3.3.3 Food leftover composition

In the two case studies concerning cutlery (Case study 4) and food packaging film (Case study 6) the product system will contain food leftovers in addition to the residual waste contamination. The newly adapted EU Circular Economy package with legislative targets for separate collection of organic waste by 2023 also relates to the two case studies, as organic waste shall be collected separately in all Member States.

Several papers (Silvennoinen et al., 2015, Razza et al., 2008, Calderón et al, 2010 and WRAP, 2013) have highlighted that food leftovers often follow the product and hence are treated in the same EoL technology as the plastic product. This is due to the fact that food leftovers are discarded with the product, e.g. cutlery or the food packaging film.

An example could be a sports event, where food is served on bio-based plates with bio-based cutlery, and where used plates and cutlery should be sorted into an organic waste fraction. In this case the bio-based plates and cutlery will end up in the organic fraction together with the food leftovers.

For the petrochemical plastic products, the food waste is assumed to follow the product in case of landfilling or incineration of the products. However, in case of source segregation of the petrochemical plastic for recycling it is assumed that the organic waste is treated with the residual waste fraction (split between incineration and/or landfilling). In case study 2 a sensitivity analysis is performed letting the food leftovers in the recycling scenario go to industrial composting.

To model this, the food leftovers are added to the EoL-reference flow (bio-based as well as petrochemical product systems) and generally treated in the applied EoL technology of the product.

The estimated amounts of food leftovers are presented below in Table 9. The estimates are based on a certain amount of food leftovers per functional unit. Since the weight of

the different products varies, the amount of food leftovers per EoL reference flow differs between the product systems. This is further explained in the specific case study chapters.

Table 9. The weight of food leftover for case studies 4 and 6 shown per EoL-reference flow (1 kg of plastic) and per functional unit. The range for the food leftover is shown in parentheses and is used in a sensitivity analysis. The numbers refer to weight in kg.

Case study	Flow	Baseline product system		Alternative and reference product systems	
		Plastic	Food leftover	Plastic	Food leftover
4. Single-use cutlery	Per EoL-reference flow	1 (PLA)	5.2 (2.5-8)	1 (PS)	6.1 (2.9-9.4)
	Per functional unit (1 000 sets)	13.6 (PLA)	70.7 (34-108.8)	10.1 (PS)	61.6 (29.3-94.9)
6. Food packaging film	Per EoL-reference flow	1 (PLA)	15.0 (4.3-25.6)	1 (PP)	15.0 (4.3-25.6)
	Per functional unit (100 m² film)	3.12 (PLA)	46.8 (13.4-79.9)	3.12 (PP)	46.8 (13.4-79.9)

To illustrate the impacts from the food leftovers, a section discussing the difference with a reference flow with and without food leftovers is provided. The results for the "pure plastic product" (including contamination, but without food leftovers) may be relevant in cases where food leftovers are separated from the plastic waste.

3.3.4 Additional soil

Two case studies concerning waste from products used in agriculture (Case study 2, Clips, and Case study 5, Mulching film) the EoL-reference flow contains contamination as well as additional soil. The soil will be attached to the product if collected for treatment and the soil will therefore follow the product to recycling, incineration or landfilling.

The additional soil is assumed to be 3 kg of soil mix (Plasticulture, 2018) per one kg plastic, see Table 10.

Table 10. The weight of additional soil for case studies 3 and 5 shown per EoL-reference flow (1 kg of plastic) and per functional unit. The numbers refer to weight in kg.

Case study	Flow	Baseline product system		Alternative and reference product systems	
		Plastic	Soil	Plastic	Soil
2. Single-use clips	Per EoL-reference flow	1 (Starch)	3	1 (PP)	3
	Per functional unit (45 000 clips)	72	216	54	162
5. Agricultural mulch film	Per EoL-reference flow	1 (Starch)	3	1 (LDPE)	3
	Per functional unit (6 000 m² film)	91.4	274.2	189	567

3.4 Average End of Life technologies

In this project the EoL technology options represent six average European waste treatment options, which have been modelled based on pre-defined EASETECH templates. The following sections describe the overall modelling approach, the system boundaries, multifunctionality and cut-offs.

The EoL technologies are modelled as "average European technologies" covering the most common variations within a technology, excluding less common technologies (e.g. chemical recycling). For example, for waste incineration technology the average EU energy recovery has been defined by assessing the energy recovery of incineration plants in the different EU countries (including plants both with and without energy recovery).

Each EoL-technology will be described by a flow chart and a "mass and energy flow table", see Figure 7 and Table 11. The dark blue boxes represent processes that are different for each EoL technology. The red boxes refer to flows which will be reported in the results sections under relevant case studies. Dotted lines indicate avoided processes and green boxes represent substitution of avoided products. Full lines represent induced processes and flow. Grey boxes are emissions to the environment or output.

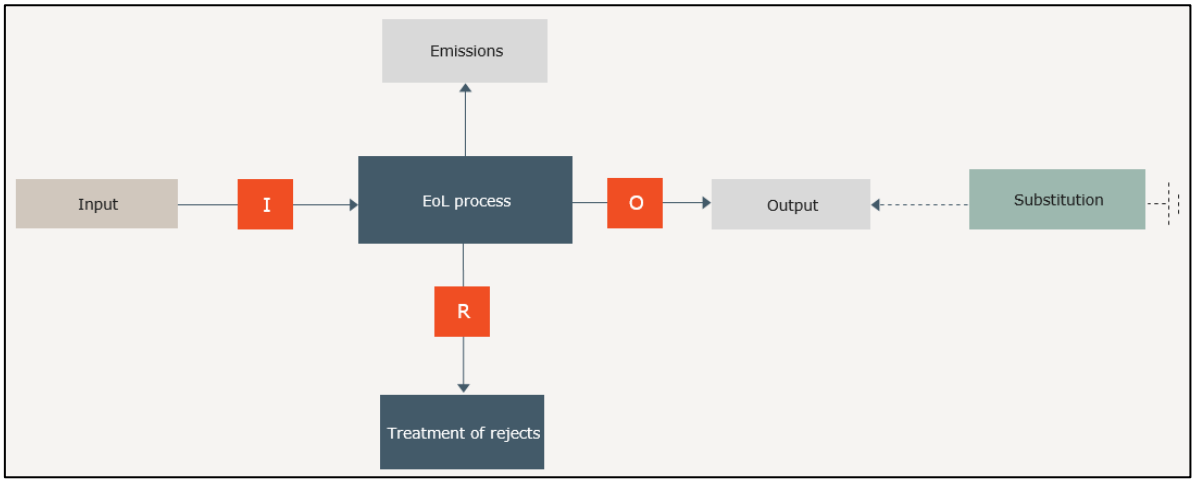


Figure 7. Flow chart example for EoL technologies.

The rows in Table 11 are marked with letters (I, O, R) corresponding to the specific flows (orange boxes) in the flow chart. The mass and energy balance will differ from case to case and the table is therefore not filled out in the following general description of the EoL technologies. However, in the case studies, the tables will be filled out presenting the specific flows. For each case study there are several EoL technologies that will be presented in the material and energy flow tables.

Table 11. Example of material and energy flow for EoL.

Techno-logy	Box	Process	Mass kg	Bio carbon kg	Fossil carbon kg	Gas m^3CH_4	Energy substitution MJ	Fertiliser substitution kg NPK
Material	I							
Techno-logy 1	R							
	O							

The mass and energy flow table include recycled material, recovered energy and mass disposed, and is hence compliant with the PEFCR guidance draft version 6.3 (European Commission, 2017) with reference to the circular footprint formula (CFF). In addition, the mass and energy flow tables show the produced amount of gas and the mass flow of biogenic and fossil carbon. The results are not just shown for the input/output of the system, but for critical steps (orange boxes in the flow charts) in the products' EoL treatment to enable the reader to follow the EoL processes.

3.4.1 System factors, general and product system specific

The EoL-treatment modelling includes a range of factors, of which some factors are general, and some depend on the specific product system (product system specific factors). For each technology it is described (see section 3.4.2 - 3.4.10) whether EASETECH models the process emissions based on the amount of treated waste, or the specific waste composition.

EASETECH includes a range of EoL-technologies based on data collected from specific cases (plants). These technology modules are adapted to reflect the defined average EoL-technology and the external processes are updated, when possible, primarily based on the newest data from the ELCD database 3.2 and Ecoinvent database 3.4.

The general factors include external processes, such as consumption of materials and energy, e.g. at the EoL treatment plant or for transportation. Other general factors are process emissions determined per kg of treated waste or depending on specific elements in the material.

An iterative process has been applied to adjust general default EASETECH factors that vary largely within one treatment technology in the EU and have a significant impact. Examples of such factors are electricity consumption, diesel oil combusted in trucks or consumption of water in specific treatment technologies, e.g. composting. The basis for defining these general factors is existing default data in EASETECH combined with literature study to define the average European technology.

The most important parameters for each EoL-treatment technology are presented in the sections below (3.4.2 – 3.4.10). Details on general factors, i.e. consumption of materials and energy, are described more in detail in Annex 1.

The specific product system factors are parameters that are dependent on the specific waste composition and therefore will differ in each case study and/or product system. The factors are described under each EoL technology below and further described in the case study chapters. This is for instance the sorting and technology efficiency at the plastic recycling facility, which heavily depends on the plastic type and product.

Modelling of the carbon balance is important for the EoL assessment, since the carbon balance is essential for the climate change impact category. For several of the bio-based (non-degradable) plastics, the only chemical difference to their petrochemical reference is the origin of the carbon.

Impacts from biological as well as fossil carbon is accounted for in the model with very similar impact factors. For example, the climate change impacts from incineration of plastic depends only on the chemical composition and not the origin of the carbon. This methodology is a consequence of the fact that the carbon absorption in plants grown for production of bio-based plastic is included in the in the cradle-to-gate results. In the two agricultural cases (Case study 2 clips and case study 5 mulching film) the removal of soil containing biogenic carbon is also included in the carbon balance.

The EoL treatment in the different scenarios results in secondary materials and/or energy, e.g. recycled plastic, compost, digestate or recovered electricity/heat, substituting conventional materials and energy. This is described for each EoL technology below.

3.4.2 Collection and transportation

Collection of waste is in the EoL-LCA split into the collection of waste from the waste producer (households, service sector or agricultural farms) and the transportation to the treatment. The collection is defined as the route for the collection truck. The diesel consumption for collection is defined per tonnes of waste collected and would differ depending on waste type and collection area (e.g. city or rural area). Transport of the waste is defined as the transport distance from the collection route (defined "centre point"[7]) to the treatment facility and the diesel consumption mainly depend on distance and the type of truck used for the transport.

The actual collection of waste is generally not included in this EoL-modelling, because the waste collection is assumed to be similar in all EoL- scenarios (not depending on the chosen EoL-technology).

However, for the agricultural products (case study 2, clips, and case study 5, mulching film) the bio-based biodegradable products left in the field are not collected, in contrast to the petrochemical clips, that are assumed to be collected and transported to either incineration or landfill. In the in-situ biodegradation scenarios, an avoided collection is therefore included in the EoL- scenario to account for the saved environmental impact from the avoided collection of the bio-based biodegradable products.

Transport of all materials was modelled with a '28-32 tonne EURO5 standard truck, highway use'. The distances for the transportation in the case studies have been estimated based on Eisted et al. (2009). The estimated distances to the treatment facilities are presented in Table 12.

The emissions from transportation include CO_2, CO, NOx, SO_2, particulate matter, and hydrocarbons due to diesel consumption for the transport. The emissions include production of fuel, combustion of fuel and metal emissions from wear of tyres and brakes.

Table 12. Transportation distances for EoL treatments.

	Collection point to treatment facility (km)	Treatment facility to treatment of residues (km)	Treatment facility to fly ash treatment	Treatment facility to bottom ash treatment	Treatment facility to use on land (km)

[7] A theoretical midpoint of the collection area, in order to calculate the distances to treatment facilities.

			(km)	(km)	
Plastic recycling	2 000	150	-	-	-
MSW incineration	150	-	500	150	-
Landfill	150	-	-	-	-
In-situ	-	-	-	-	-
Industrial composting	150	150	-	-	150
Anaerobic digestion	150	150	-	-	150

3.4.3 Energy, all EoL-technologies

Some of the included EoL-treatment technologies produce energy (electricity and/or heat). The energy produced (electricity and heat) is assumed to substitute for "marginal technologies in Europe". This substituted energy is defined by combining the marginal energy technology for each EU country to an "EU marginal mix".

The marginal EU electricity mix is acquired from the Ecoinvent database 3.4 and thus represents a mix of electricity sources used in Europe. The Ecoinvent marginal electricity process is based on a methodology reported in Iltten et al. (2012) with data from 2014/2015 extrapolated to 2017.

The marginal heat in Europe is assumed to be natural gas, as natural gas remains the largest contributor to heat production in Europe (Eurostat., 2012 & Honoré., 2018). The dataset is acquired from the Ecoinvent database 3.4 (data excludes Switzerland). The dataset includes shares of heat supplying activities from combined heat and power (CHP) plants and pure heat plants have been estimated based on IEA (2013) for the year 2009. Overall, the marginal heat mix thus consists of approximately 70 % heat from combined heat and power plants and 30 % heat from pure heat plants. The heat dataset includes primarily data from 2011 extrapolated to year 2017.

The choice of the marginal energy source is an important parameter for the overall results, since energy substitution from energy recovery, especially from waste incineration, largely affects the environmental performance of the EoL technologies.

The generated electricity in the different EoL technologies, is modelled to be fully exported to the electricity grid. The same is applied for heat production. Examples are electricity production at the incineration plant and the heat produced from the biogas at anaerobic digestion plants. In some facilities the heat and the electricity are used directly at the plant, but this is not accounted for. This would solely mean that the grid loss would be saved, thus a few percentages of the production.

3.4.4 Plastic recycling

The plastic is transported to the recycling facility. The whole recycling facility is assumed to be situated in a single geographical location, and not split into separate facilities, e.g. sorting/ washing facility and shredding/ extrusion facility at different locations. Hence our modelling does not account for transportation between different plastic recycling locations, which could be the case for some recycling facilities in Europe. Residues from the plastic recycling facility will be transported to incineration.

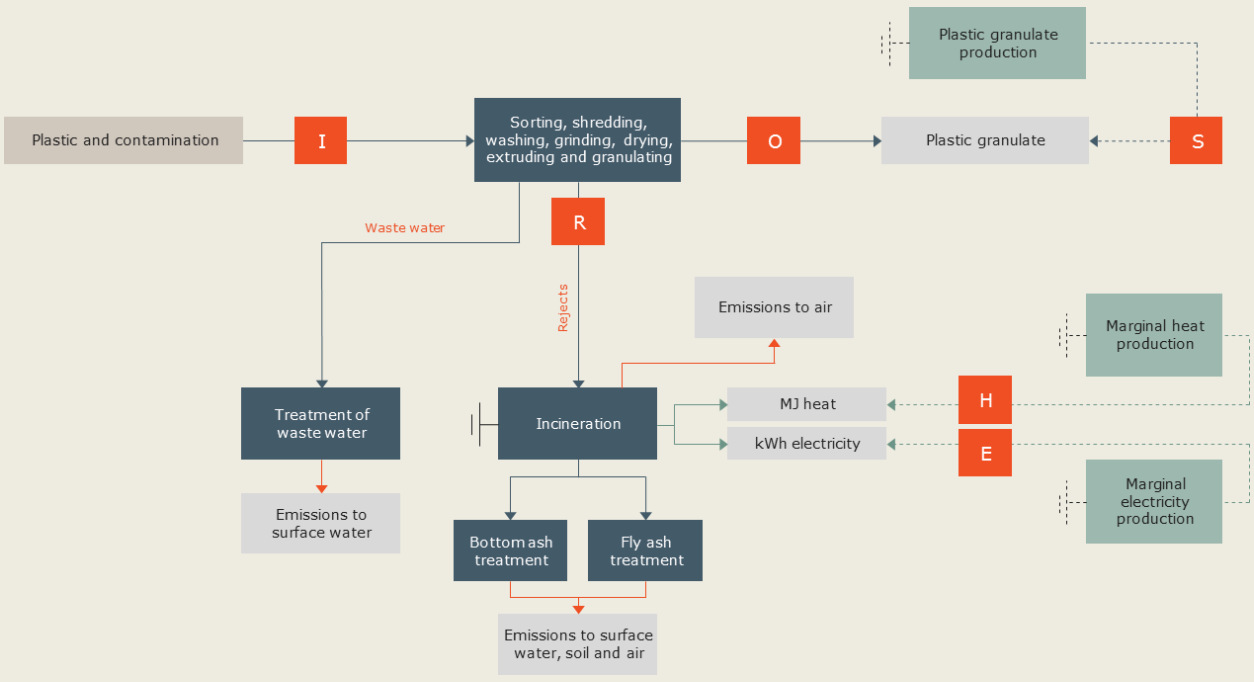
At the plastic recycling facility, the plastic is sorted, shredded, washed, grinded, dried, extruded and granulated into pellets. For this process water, steam, electricity and soap is consumed (based on Rigamonti et al., 2013). No additional consumption of materials is accounted for.

Only mechanical plastic recycling is modelled, thus excluding feedstock recycling (also known as chemical recycling) from the study since only 0.3 % of post-consumer plastic waste is currently chemically recycled in the EU (PlasticsEurope, 2013).

Treatment of waste water from the recycling facility is modelled as treated waste water with direct emissions of COD, BOD5, cadmium, nitrogen, phosphorus and TOC to surface water (EASETECH).

The recycled plastic is assumed to substitute for virgin plastic on at the EU market. When modelling this substitution, the amount of virgin plastic substituted per amount of recycled plastic is included in the modelling as the "substitution rate for substitution". Quantification of this factor is described below.

The flowchart for plastic recycling is presented in Figure 8. The corresponding outline of the material and energy flow table (not containing data, as these will be specific for each case study) is presented in Table 13.



Dark blue boxes represent processes. Dotted lines indicate avoided processes, while full lines represent induced processes and flow. Transport is not illustrated. The red boxes refer to flows which will be reported in the results sections under relevant case studies. Grey boxes are emissions to the environment or output products and green boxes represent substitution of avoided products.

Figure 8. Flowchart for the EoL technology mechanical plastic recycling, which gives an overview of the processes modelled in relevant case studies.

Table 13. Material and energy flow table for plastic recycling (not filled out, since the data for this will be specific for each case study.

Technology	Box	Process	Mass kg	Bio carbon kg	Fossil carbon kg	Energy substitution MJ
Material	I	Material input				

Recycling	O	Recycled plastic				
	S	Substituted plastic				
	R, E, H	Rejects				

3.4.4.1 Losses and substitution of plastic in recycling

The collected plastic for recycling has several losses during the recycling process:

- Sorting and technology efficiency – the material losses, which consist of
    - Sorting efficiency (both coarse sorting (e.g. a co-mingled fraction were foreign objects are sorted out and where e.g. plastic is sorted from metals) and fine sorting (sorting into polymers))
    - Technical efficiency (a.k.a. A factor) (losses during shredding, washing, grinding, drying, extruding and granulating)
- Substitution rate – the quality loss (a.k.a. B factor)

The recycling rate is sometimes defined to as the collected plastic multiplied with solely by the sorting efficiency and other times by both the sorting and the technology efficiency. In this study the recycling rate is defined as:

- Recycling rate = Collection rate x Sorting efficiency x Technical efficiency

Table 14 presents the collection rate, the sorting and technology efficiency and the actual recycling rate of the plastic for each case study. Furthermore, the references that have been used to determine the recycling rates used in this report are presented in Table 14.

The collection rate of plastic is likely to be equal for each type of plastic product no matter which polymer the product is produced of. Within each case study the collection rate for the EoL pathways is assumed to be the same, as consumers often will not see any difference between polymer types. PLA is an exception, where a part of the PLA is assumed to enter the organic waste stream.

The plastic that is collected for recycling normally undergoes normally both coarse and fine sorting, where a part of the collected plastic is sorted out as rejects and some or all foreign objects. The amount of rejects (see Figure 8 above) depends on the sorting and technology efficiency, which is in this study is specifically defined for each case study.

After the sorting steps, the pre-sorted plastic waste undergoes recycling where there also is also some material loss.

Losses in the sorting facility depend on the input and the product output, which product the recycled plastic is intended for. Note that these losses are for the combined polymers, not for specific products. The same polymers do not have the same sorting and technology efficiency, as this is also dependent on product type.

The rejects are transported to incineration (assumption based on Tonini, 2018).

Table 14. Collection rates, sorting efficiencies, technical efficiencies and recycling rates from literature.

Polymer	**Collection rate**	**Sorting efficiency**	**Recycling rate** (incl. sorting, but excl. technical eff.)	**Technical efficiency**	**Combined sorting and technology efficiency**	**Recycling rate** (incl. sorting and technology)

Post-consumer plastic	31.1 %[1] 37 %[2]					
Plastic packaging	40.1 %[1]		15 %[a, 2]			
Polyolefins (PO) and PET	EU: 44 %[2] FR: 44 %[2] DE: 76 %[2] UK: 38 %[2] ES: 41 %[2] IT: 55 %[2]	FR: 48 %; DE: 47 %; UK: 58 %; ES: 76 %; IT: 76 % (Estimated[2])	FR: 21 %[2] DE: 36 %[2] UK: 22 %[2] ES: 31 %[2] IT: 42 %[2]	97 %[4] 90.3 %[3], 93 %[5], 90 %[6] (HDPE) 80 %[5], 75.5 %[6] (PET)	50 %[7] (PO) 70 %[7] (PET)	
PS		Very few plants recycle[2]			50 %[7]	

a excluding plastics exported, which is assumed not recycled
1 Plastics Europe, 2017
2 Plastic Recyclers Europe, 2017
3 Replast, n.d.
4 SWEREC, 2007
5 Franklin Associates, 2011
6 Giugliano et al., 2011
7 Dall et al., 2003

Based on Table 14, the estimated collection rates, sorting and technology efficiencies and recycling rates used in this study are presented in Table 15. For case study 1, 4 and 5 there is specific case study dependent literature, not presented above, which also has an influence on the selected rates and efficiencies. This is presented in further detail in the case study chapters.

Table 15. The estimated collection rates, sorting and technology efficiencies and the recycling rate for plastic recycling in all case studies.

Case #	Plastic type	Collection rate	Sorting and technology efficiency	Recycling rate (collection rate x sorting and technology efficiency)
1	PET	60 % [1]	76 % [2]	45 %
	Biobased PET	60 % [1]	76 % [2]	45 %
3	PLA	15 % [3]	70 % [4]	11 %
	PET	30 % [5]	70 % [4]	21 %
	PP	30 % [5]	70 % [4]	21 %
	Biobased PP	30 % [5]	70 % [4]	21 %
4	PS	30 % [5]	25 % [6]	7.5 %
5	LDPE	5 % [7]	37 % [8]	2 %
6	PLA	15 % [3]	70 % [4]	11 %

	PP	30 % [5]	70 % [4]	21 %
	Biobased PP	30 % [5]	70 % [4]	21 %
7	LDPE	30 % [5]	70 % [4]	21 %
	Biobased LDPE	30 % [5]	70 % [4]	21 %

1 Estimated based on EPBP, n.d.; PlasticsEurope, 2016b; PlasticsEurope, 2013; European Commission, 2018
2 Estimated based on EPBP, N.D.; PlasticsEurope, 2016; PlasticsEurope, 2015; PlasticsEurope, 2013a; PlasticsEurope, 2013b
3 Estimated to be equal to (5) but divided equally to the intended EoL technologies based on Seyring et al., 2015 and Eurostat 2017c.
4 Estimated based on Plastic Recyclers Europe, 2017, Replast, n.d., SWEREC, 2007, Franklin Associates, 2011, Giugliano et al., 2011 and Dall et al., 2003
5 Estimated based on European Commission, 2018; Plastics Europe, 2017 and Plastic Recyclers Europe, 2017
6 Estimated based on Dall et al., 2013 and Plastic Recyclers Europe, 2017
7 Estimated based on PRE, 2017; IFFPG, 2018; RIGK, 2018; EuPC, 2018
8 Based on Briassoulis et., al, 2013.

The total substituted material is represented by "S" and the total recycled material is represented by "O" in Figure 8 and Table 13. The following equations are used to calculate the volume of recycled material and substituted plastic:

- **S** total substituted material (kg) = **I** EoL-reference flow (kg) x Sorting and technology efficiency (%) x Substitution rate (%)
- **O** total recycled material (kg) = **I** EoL-reference flow (kg) x Sorting and technology efficiency (%)

The substitution of virgin plastic depends on the quality of the recycled material (Vadenbo et al., 2017), which is a consequence of the quality of the collected plastic and the applied sorting technology. The economic value of recycled plastic is a good indicator for the quality of the recycled plastic.

The substitution rate for substitution of recycled plastic (based on PE and PET) is set to 81 % according to Rigamonti et al. (2013) for all case studies. It may in reality vary between products; however, since the values of sorting and technology efficiency for each case study were carefully chosen, this market response figure was assessed to be a relevant estimate for all cases.

Throughout this study it is assumed that recycling of PLA substitutes the petrochemical reference (PET or PP). It is possible, that recycling of PLA will substitute virgin PLA in the future (European Bioplastics., 2016), which would change the environmental benefits of the substitution.

The substitution processes modelled were based on PlasticsEurope's ecoprofiles representing conventional petrochemical plastic (APME, 1998). The specific substitution assumptions are presented below for each case study in Table 16, and further details are provided in Annex 1 (e.g. PlasticsEurope's Ecoprofile process names).

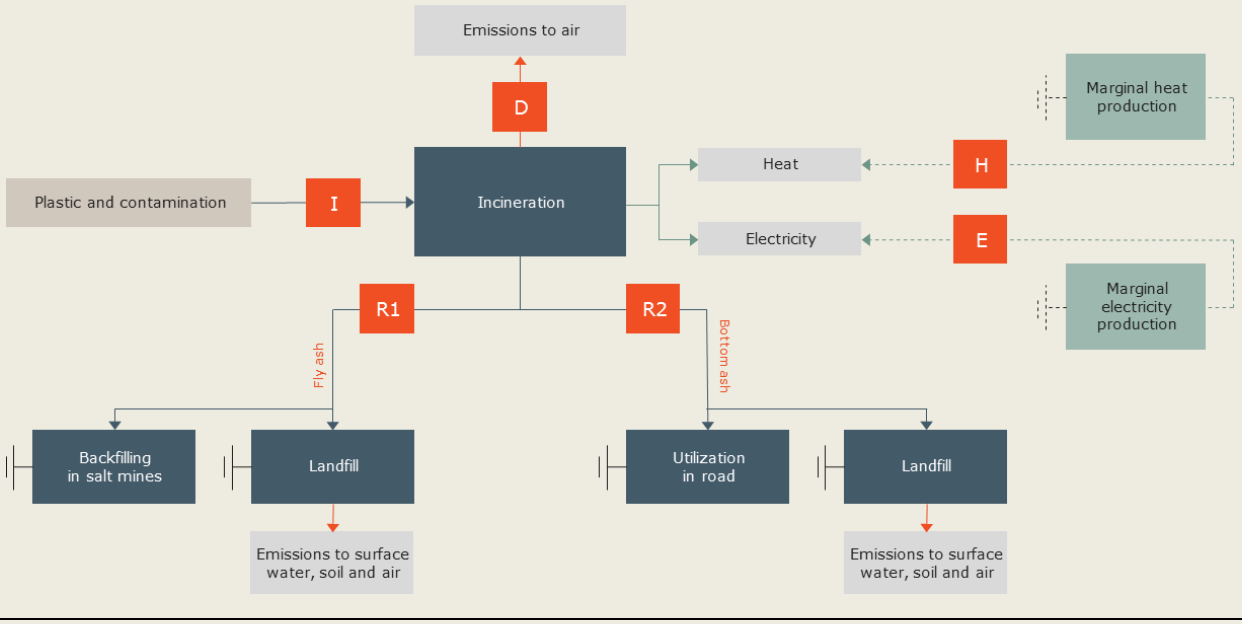
Table 16. Substitution of recycled plastic (based on PlasticsEurope's ecoprofiles).

Case study #	Recycled plastic	Substituted plastic (PlasticsEurope's ecoprofiles)
1, 3	Petrochemical PET	Petrochemical PET
1	Bio-based PET	Petrochemical PET

3	PLA	Petrochemical PET
3	Bio-based PP	Petrochemical PP
3, 6	Petrochemical PP	Petrochemical PP
4	Petrochemical PS	Petrochemical PS
5, 7	Petrochemical LDPE	Petrochemical LLDPE
6	PLA	Petrochemical PP
7	Bio-based LDPE	Petrochemical LLDPE

3.4.5 Waste incineration with and without energy recovery

Waste incineration is modelled with and without energy recovery as one technology, and the flowchart for the process is presented in Figure 9. The corresponding outline of the material and energy flow table (not containing data, as these will be specific for each case study) is presented in Table 17.



Dark blue boxes represent processes. Dotted lines indicate avoided processes, while full lines represent induced processes and flow. Transport is not illustrated. The red boxes refer to flows which will be reported in the results sections under relevant case studies. Grey boxes are emissions to the environment or output products and green boxes represent substitution of avoided products.

Figure 9. Flowchart for the EoL technology incineration, which gives an overview of the processes modelled in relevant case studies.

Table 17. Material and energy flow table for incineration (not filled out, since the data for this will be specific for each case study).

Technology	Box	Process	Mass kg	Bio carbon kg	Fossil carbon kg	Gas m^3CH_4	Energy substitution MJ
Material	I	Material input					
Incineration w/wo energy recovery	E, H	Energy production					
	R1	Fly Ash					

	R2	Bottom ash					
	D	Direct emissions					

The municipal solid waste incineration (MSWI) model is based on the chemical composition of the waste fraction, including e.g. the energy, water and ash content for the individual materials.

The EASETECH model for incineration includes a separate mass flow for each element (carbon, nitrogen etc.) consisting of distribution coefficients for each element distributing the specific element between the different outputs of the incineration plant (air, water, fly ash, bottom ash). For example, the carbon in the waste is divided into emissions to air, fly ash, bottom ash etc. The energy produced is based on the specific material's energy content. The most important input values (such as energy content) are presented for each product system in the case study chapters.

For process emissions, default EASETECH figures are used, i.e. the consumption of chemicals and energy related to the amount of waste incinerated. The parameters in Table 18 are applied for modelling the average European incineration plant and are explained further in Annex 1.

If one would like to interpret the results solely for incineration with energy recovery or solely for incineration without energy substitution, the only difference between the two technologies is the energy recovery. To interpret incineration with energy recovery the energy substitution (heat and electricity) should be multiplied by 1.65 (factor between the amount of waste treated in plants with and without energy recovery). To interpret incineration without energy recovery the energy substitution should be omitted.

Table 18. Parameters for modelling of average European waste incineration.

Parameter	Value	Reference
Heat efficiency	22 %	Calculated based on CEWEP (2013), providing data on 314 incineration plants with energy recovery. This includes 17 European countries and the amount of waste treated in these plants corresponds to 85% of the total waste in the EU25 (59 mio. tons waste (average 2007-2010) with 34.59 % heat efficiency and 14.89 % electricity efficiency). Eurostat (2017a) provides data related to incineration plants without energy recovery (35.7 mio. tons of waste (average of 2006, 2008 and 2010)).
Electricity efficiency	9 %	
Fly ash utilised in salt mines	43 %	Manders, 2008
Fly ash landfilled	57 %	Manders, 2008
Bottom ash recycled in road construction	60 %	Manders, 2008
Bottom ash landfilled	40 %	Manders, 2008

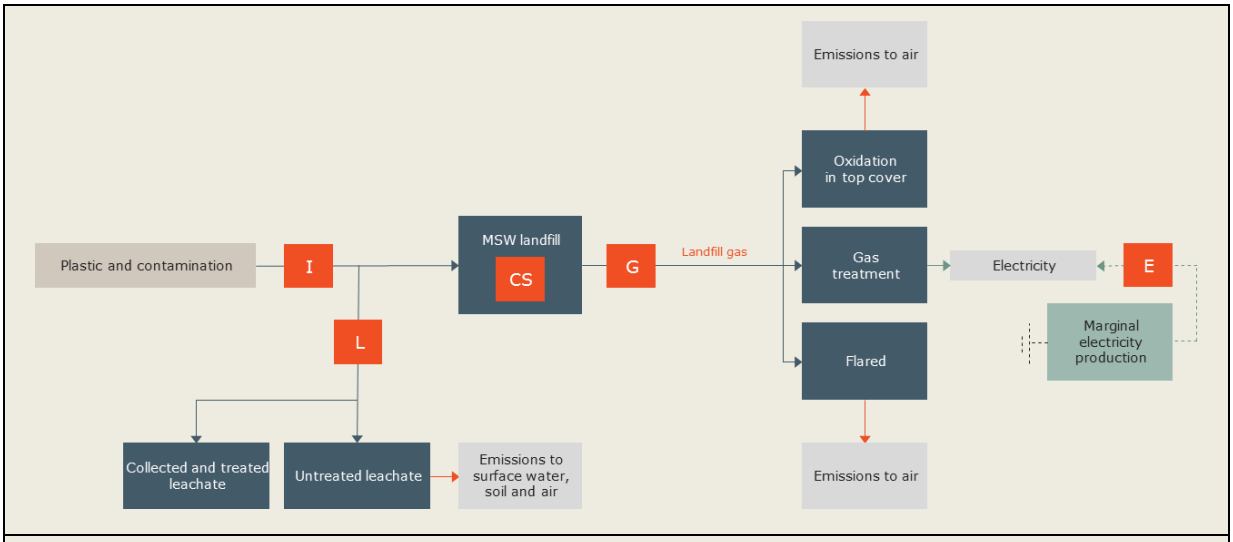
The recycling of ash in road construction and utilization in salt mines substitute gravel at a 1:1 ratio (Astrup, 2008 and Astrup & Hyks, 2008).

3.4.6 Landfilling

The landfill model is based on an EASETECH template for municipal solid waste (MSW) landfilling. The landfill modelling includes construction and operation, collection of landfill gas, collection of leachate, direct emissions of landfill gas, direct emissions of leachate and energy production from a share of the produced landfill gas. The modelling includes 100 years of landfilling.

Some modifications have been applied as described below to consider the difference between landfilling MSW compared to landfilling plastics.

The flowchart for landfilling is presented in Figure 10. The corresponding outline of the material and energy flow table is presented in Table 19 (not containing data, as these will be specific for each case study).



Dark blue boxes represent processes. Dotted lines indicate avoided processes, while full lines represent induced processes and flow. Transport is not illustrated. The red boxes refer to flows which will be reported in the results sections under relevant case studies. Grey boxes are emissions to the environment or output products and green boxes represent substitution of avoided products.

Figure 10. Flowchart for the EoL technology landfilling, which gives an overview of the processes modelled in relevant case studies.

Table 19. Material and energy flow table for landfilling (not filled out, since the data for this will be specific for each case study.

Technology	Box	Process	Mass kg	Bio carbon kg	Fossil carbon kg	Gas m^3CH_4	Energy substitution MJ
Material	I	Material input					
Landfill	L1	Leachate					
	L2, E	Landfill gas					
	CS	Storage Landfill					

The process emissions were initially based on default EASETECH parameters for a MSW landfill. Several processes, which have substantial impact on the results when comparing landfilling of plastic to landfilling of MSW, were further assessed and the default values for landfilling of MSW were adapted to landfilling of plastic. Furthermore, values were evaluated and modified to represent average European conditions rather than the standard landfill in EASETECH. The two most important adaptions are described below (details are given in Annex 1):

- Leachate profile: The leachate profile from the landfill was changed to represent leachate from plastic waste. The leachate composition when landfilling plastic was based on Manfredi et al. (2010). The COD and BOD concentrations were based on the TOC concentration (Manfredi et al., 2010). The COD and BOD concentrations were estimated to be different between non-degradable (both petrochemical and

bio-based) and biodegradable plastic. Due to biodegradation biodegradable plastic is assumed to generate higher leachate concentrations.

- Precipitation: The amount of leachate depends on different parameters including precipitation, which was set to 200 mm per year to represent average European conditions (Manfredi et al., 2010).

A share of the leachate is assumed to be collected and treated in a wastewater treatment plant, whereas the remaining part (not collected) ends up as direct emissions to surface water. The leachate collection rate for the first 80 years of the landfill's lifetime is 99.9 %, and for the remaining 20 years 87 % is collected. The leachate production is modelled in EASETECH as distributed into four-time periods, considering that the leachate will be less contaminated over time.

The amount of landfill gas produced is calculated by the k rate (1st order decay rate for methane generation), which is a product system specific factor.

The composition of the landfill gas is assumed to be similar to landfill gas from an average MSW landfills (EASETECH).

The gas collection varies over the life time of the landfill. The landfill gas is collected at varying rates between 35-75 % during the first 55 years (three periods). For the last 45 years no gas collection is assumed.

For the collected landfill gas, it is assumed that 22 % of the gas is flared, 29 % is used for energy recovery and 49 % is leaked (OpenLCA Nexus, 2015). This data was validated between 2015 and 2018 by OpenLCA Nexus (2015). The electricity efficiency of the landfill gas utilisation is assumed to be 37 % (Christensen, 2011).

The landfill gas, which is not collected, is assumed to be partly oxidised in the landfill top covers. The conditions of the top cover depend on the age of the landfill (daily cover for the first five years, intermediate cover for 5-15 years and final cover for 15-100 years).

Burial tests of biodegradable plastics under anaerobic conditions have shown partial degradation already after three months and Cho et al. (2011) predicted complete degradation occurs after 300 days for a starch-PCL plastic.

3.4.7 Biological treatments

Three biological waste treatment options are included in this study: in-situ field-based biodegradation, industrial composting and anaerobic digestion.

Biodegradability of the products is an important factor, when assessing biological EoL-treatment options. Standards exist for biodegradability, e.g. for bio-degradable plastic, cover industrial composting and home composting under aerobic conditions. However, biodegradable plastic products are also treated by anaerobic waste treatment technologies (landfilling and anaerobic digestion), where the biodegradability must be assumed to be much lower. This will e.g. be the case in countries where biological treatment of food waste consists solely of anaerobic digestion, as in Denmark, or in countries, where organic waste still ends up in landfills.

3.4.7.1 Biodegradation in biological EoL technologies

Various test standards have been developed to describe and measure the biodegradability of plastic products. Many published studies refer to such standards. A short overview of the most commonly used biodegradability standards is presented in Annex 2. The European standard for compostability is EN 13432, where CEN 16760 describes LCA of bio-based products.

Degradation of plastic can occur in several ways e.g. photochemical degradation. However, degradation of plastic in this study generally refers to biodegradation. Definition of biodegradation is according to the OECD:" *the process by which organic*

substances are decomposed by micro-organisms into simpler substances such as carbon dioxide, water and ammonia".

Biodegradation can occur under aerobic as well as anaerobic conditions (composting and anaerobic digestion, respectively). One of the common ways to define the degree of biodegradation is by comparing the carbon in the tested material with the carbon in the produced gas (both CO_2 and if relevant CH_4), according to formula (1),

$$Biodegradation\ rate\ (\%) = \left(\frac{C_g - C_b}{C_i}\right) \cdot 100\ \%\ (1)$$

where,

C_g: Carbon in the produced gas.

C_b: Carbon in the air phase of a blanc vessels to account for background concentrations.

C_i: Carbon contained in the tested material.

According to the standards, there is a distinction between disintegration and biodegradation (EN 13432:2000 certification of compostability). After three months the plastic must be disintegrated, while biodegradation must occur within six months.

During biodegradation under *aerobic conditions (composting),* carbon is transformed into carbon dioxide as shown in formula (2).

$$C + O_2 \rightarrow CO_2\ (2)$$

For *anaerobic conditions (anaerobic digestion)*, carbon is transformed to methane and CO_2 according to formulas (3).

$$2C + 2H_2O \rightarrow CH_4 + CO_2\ (3)$$

In industrial composting plants most of the degraded carbon will be emitted as CO_2. However, it is difficult to obtain 100 % aerobic conditions in the treatment plant and thus a certain methane formation must be expected. This is included in the modelling.

Since the biodegradable plastics are designed to degrade under aerobic conditions, little information is available on degradation of biodegradable plastic under anaerobic conditions. A limited literature review has been performed in the project on this topic (also covering industrial composting for comparison) and the results of that is presented in Annex 2. The results show that the environmental degradability of plastics is complex and is influenced by many factors such as the nature of the plastics (e.g. chemical composition), the physical handling of the plastic (e.g. shredding) and the environmental conditions to which the plastic is exposed (e.g. temperature and retention time). The main findings of the literature review can be summarised as follows:

- The biodegradation rate depends on the type of plastic (chemical composition).
- In-situ: Burial tests show that the plastics in contact with soil (aerobic conditions) will not degrade to a desirable extent within a year.
- Industrial composting: Biodegradable plastics show good biodegradation under aerobic conditions and typically biodegradation of between 63 and 97 % (up to 10 weeks).
- Anaerobic digestion: Biodegradation is possible to some extent, however not to satisfactory levels, at least not within the typical retention time in the anaerobic digestion plants (<30 days). A large share of the plastic will therefore be present in the digestate. For anaerobic digestion plants, complete degradation can only occur if the digestate is further composted after the anaerobic process. The degradation rates under anaerobic digestion were found to be between 10 % and 88 % (up to 20 weeks).

The biodegradation is modelled depending on the product system. The implication for each treatment system is described in further detail below in the specific case studies.

3.4.7.2 Agricultural application of plastic, compost and digestate on field

Anaerobic digestion and composting create digestate and compost, respectively, which is typically applied to agricultural soil.

The environmental impacts from agricultural application of the digestate or compost is included in the LCA system, since this is seen as part of the waste management system. The substitution of fertiliser is included as well as the difference in emissions from conventional fertiliser and organic fertiliser (compost and digestate), respectively.

The environmental impacts from this application are modelled based on an EASETECH template (based on Bruun et al, 2006; Bernstad and Jansen, 2011; Ambus et al., 2001; Hansen et al., 2012; Børgesen et al., 2001; Yoshida et al., n.d.; Klinglmair et al., n.d.). This is further described below. In-situ biodegradation has been modelled in a similar way.

The previous sections described the often-incomplete degradation of biodegradable plastic during typical retention time in the waste treatment facilities (industrial composting or anaerobic digestion). Therefore, compost or digestate applied to agricultural soil may contain partly degraded plastic.

In this study it is assumed that non-degraded biodegradable plastic in residues from biological treatment degrade after field application, and the carbon is distributed between air emissions (CO_2 and CH_4) and carbon stored in soil. A similar approach applies for non-degraded contamination on the products.

The fate of the carbon after agricultural application of the residues depends on soil conditions, which vary significantly across Europe. All biogenic carbon applied to the field is assumed to be distributed as follows: 88.68 % CO_2, 0.01 % CH_4 and 11.31 % stored in the soil. These figures are based on sandy loam soil conditions, based on one dimensional simulation by an agro-ecosystem model (results implemented in EASETECH based on Hansen et al., 2012). The values are simulated based on Danish weather data, which is not representative for average European weather (EASETECH) and is therefore subject to uncertainty when modelling European waste scenarios.

Distribution of nitrogen (N) is not relevant for bioplastics, since the nitrogen content in bio-based biodegradable plastic products is negligible (Dewilde, 2018).

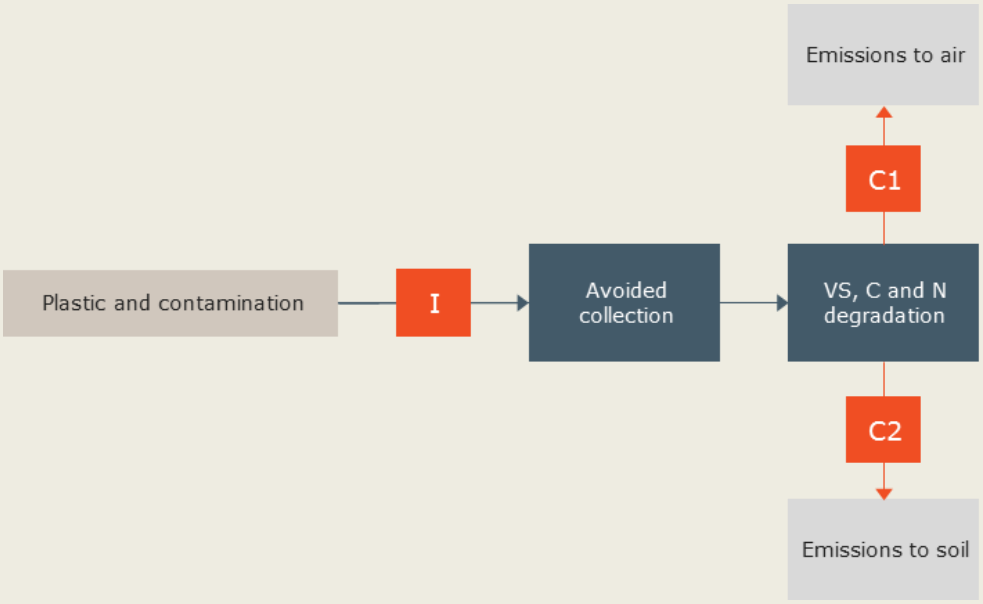
The content of metals (Cr, Zn, Ni, Hg, Ca, Co and Pb) in the residue as well as the substituted conventional fertilisers are modelled as emissions to soil. Thus, the metal emissions are net emissions, i.e. the difference from applying compost or digestate compared to conventional fertilisers.

The additional benefits of the use of compost or digestate to agricultural soil, including reduced leaching, improvement in soil structure and erosion protection are currently not adequately captured by environmental impact models and indicators. The nutrient content in the compost or digestate can to a certain extent substitute conventional fertiliser. Substitution of other products, such as e.g. peat, has not been modelled. Data on production of conventional fertilisers for substitution was obtained from the Ecoinvent database version 3.4. Further details on the agricultural application can be seen in Annex 1.

3.4.8 In-situ field-based biodegradation

The in-situ modelling in EASETECH is based on the process used for application of compost on agricultural land. The agricultural products applied on the field (Case studies 2 and 5 on clips and mulching film) are assumed to be fully degraded within the 100 years of impact modelling.

Figure 11 illustrates the flowchart for 'in-situ field-based biodegradation'. The corresponding outline of the material and energy flow table (not containing data, as these will be specific for each case study) is presented in Table 20.



Dark blue boxes represent processes. Full lines represent induced processes and flows. Transport is not illustrated. The red boxes refer to flows which will be reported in the results sections under relevant case studies. Grey boxes are emissions to the environment.

Figure 11. Flowchart for the EoL technology in-situ field-based biodegradation, which gives an overview of the processes modelled in relevant case studies.

Table 20. Material and energy flow table for in-situ field application (not filled out, since the data for this will be specific for each case study).

Technology	Box	Process	Mass kg	Bio carbon kg
Material	I	Material input		
In-situ	C1	Air emissions		
	C2	Storage in soil		

The in-situ module contains a distribution of carbon to CO_2, methane and carbon stored in the soil. Furthermore, any metals in the material composition are assumed to be emitted to soil. No substitution of fertilisers is relevant for field-based application (Tonini, 2018).

In-situ treatment results in avoided waste collection in contrast to the other EoL treatment technologies for the clips. Therefore, avoided waste collection, i.e. avoided diesel consumption, is included in modelling this EoL technology (and excluded in the other EoL technologies).

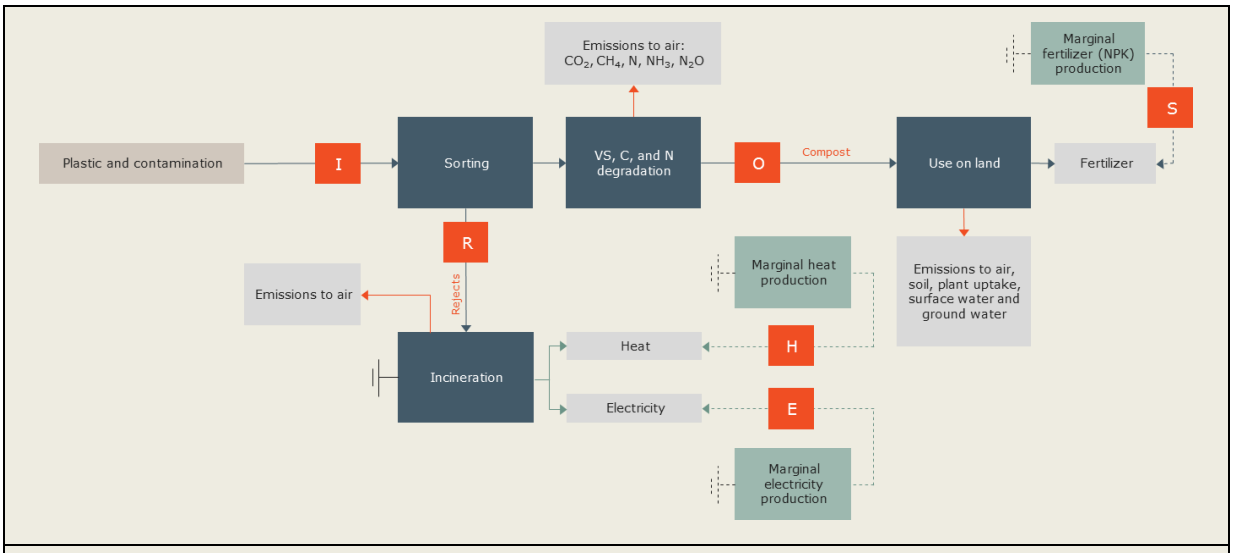
Impacts of microplastics left in the soil are not included in this project, as available data does not allow such impacts to be included in the current LCA impact categories.

3.4.9 Industrial composting

Industrial composting in Europe includes several technology types. The composting technologies can be divided into indoor and outdoor facilities composting (each covering

several types of composting technologies)[8]. The organic waste going for composting is typically divided into two major material fractions: garden waste and food waste. The biodegradable plastics would typically follow the food waste stream. As food waste will typically be treated in an indoor facility, only indoor composting facilities are included in this project (De Wilde and Siebert, 2018).

The specific process used in EASETECH is based on tunnel composting, which is one of the main indoor composting technologies in Europe today (De Wilde et al., 2014). The flow chart for the treatment facility can be seen in Figure 12. The corresponding outline of the material and energy flow table (not containing data, as these will be specific for each case study) is presented in Table 21.



Dark blue boxes represent processes. Dotted lines indicate avoided processes, while full lines represent induced processes and flow. Transport is not illustrated. The red boxes refer to flows which will be reported in the results sections under relevant case studies. Grey boxes are emissions to the environment or output products and green boxes represent substitution of avoided products.

Figure 12. Flowchart for the EoL technology industrial composting, which gives an overview of the processes modelled in relevant case studies.

Table 21. Material and energy flow table for industrial composting (not filled out, since the data for this will be specific for each case study).

Technology	Box	Process	Mass kg	Bio carbon kg	Fossil carbon kg	Energy substitution MJ	Fertiliser substitution kg NPK
Material	I	Material input					
Compost	O, S	Compost					
	R, E, H	Rejects					

Industrial composting facilities often include pre- and/or post-treatment steps. Often the purpose of pre-treatment is to remove larger objects before the materials enter the treatment process. In the pre-treatment the waste fraction can be optimised in regard to C/N ratio, moisture and other characteristics (De Wilde, 2014). Post treatment mainly

[8] e.g. windrow composting, aerated static piles, tunnel composting, in-vessel composting etc.

consists of sieving of the material to ensure a high-quality product with low content of impurities (De Wilde, 2018). If the composting plant has a post treatment step the oversized items are often removed and re-composted. Other plants will have shredded the materials, hence no post-treatment with sieving is performed. The output product is a mature/stable compost that has little readily biodegradable material content (De Wilde, 2014). In some EU countries a national standard for compost exists, where e.g. the amounts of non-composted materials are defined.

The rate of material rejected from the plant (in pre-sorting) is assumed to be 30 % of the biodegradable plastic input (EASETECH and confirmed by OWS and ENS) and 5 % of the organic contamination and food leftover input (EASETECH). All rejected materials are sent for incineration (Tonini, 2018).

Default process emissions data from EASETECH were used for modelling the industrial composting facility. This includes diesel consumption for wheeled loaders, electricity and wood pellets used as structural material, which all depend on the amount of treated waste (not the chemical composition).

Biogenic carbon is transformed into CO_2 and CH_4 in the composting process. The content of biogenic carbon in the plastic product and the degradation rate defines the distribution of the input material between air emissions (degraded material) and compost product (non-degraded material). These factors vary between different plastic types (see specific case studies). The degradation rate for the contamination and food leftovers are standard values taken from EASETECH (for organic waste 73.5 %).

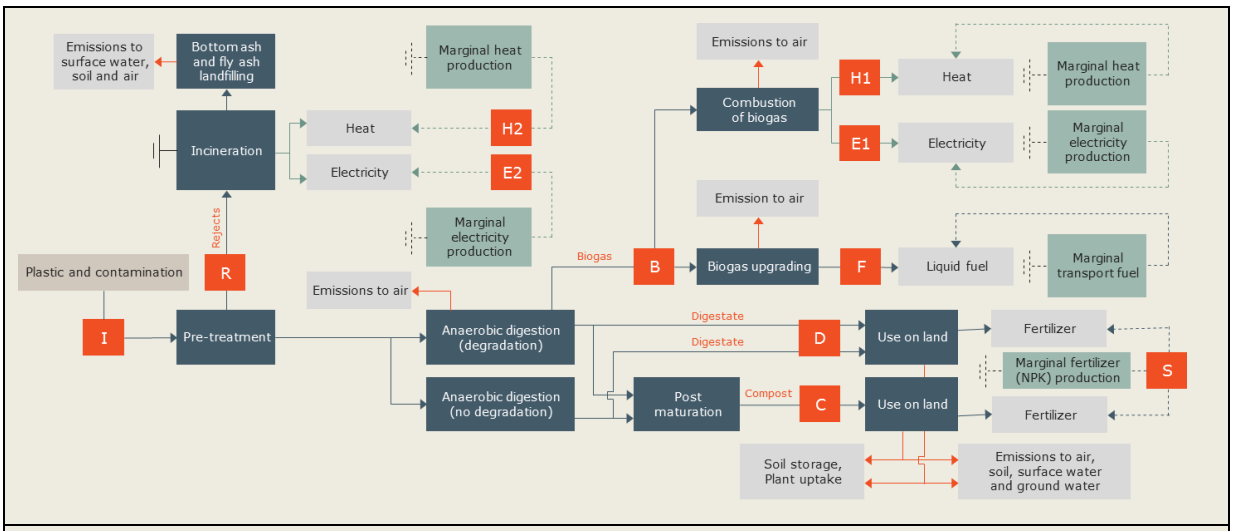
3.4.10 Anaerobic digestion

A range of different technologies for anaerobic digestion exist in Europe today (wet or dry combined with mesophilic or thermophilic). In the EU 23 % of the anaerobic digestion plants are thermophilic wet and 77 % are mesophilic (wet or dry) or thermophilic dry technologies (Bruno and Baera, 2015), which is modelled in a combined scenario.

Some biodegradable plastics are not fully biodegraded under dry and/or mesophilic conditions, while it is assumed to be biodegraded under wet thermophilic conditions. A research on PLA under AD condition showed that degradation will only take place under thermophilic conditions (De Wilde et al, 2016). The biodegradation of bio-based biodegradable plastic is modelled differently depending on the type of AD technology. The contamination consisting of organic waste will degrade and generate biogas in all types of biogas plants.

Default process emissions factors from EASETECH for a standard biogas plant have been used (see Annex 1 for further details).

The flow in the anaerobic digestion scenario is shown in Figure 13. The corresponding outline of the material and energy flow table (not containing data, as these will be specific for each case study) is presented in Table 22.



Dark blue boxes represent processes. Dotted lines indicate avoided processes, while full lines represent induced processes and flow. Transport is not illustrated. The red boxes refer to flows which will be reported in the results sections under relevant case studies. Grey boxes are emissions to the environment or output products and green boxes represent substitution of avoided products.

Figure 13. Flowchart for the EoL technology anaerobic digestion, which gives an overview of the processes modelled in relevant case studies.

Table 22. Material and energy flow table for anaerobic digestion (not filled out, since the data for this will be specific for each case study.

Techno-logy	Box	Process	Mass kg	Bio carbon kg	Fossil carbon kg	Energy substitution MJ	Fertiliser substitution kg NPK
Material	I	Material input					
Anaerobic digestion	B, E1, H1	Biogas					
	C	Compost					
	D	Digestate					
	R1, E2, H2	Rejects					

Anaerobic degradation plants can have a pre-treatment and/or a post-treatment. The purpose of pre-treatment is to remove larger objects and condition the input for the treatment process, e.g. by particle size reduction. Post-treatment mainly consists of sieving of the material to ensure a high-quality product with low content of impurities. The impurities can be larger items of bio-based plastic products or other materials (De Wilde, 2018).

The rate of material rejected from the plant (in pre-sorting) is assumed to be 30 % of the plastic input and 5 % of the organic contamination and food leftover input (EASETECH; De Wilde & Siebert, 2018). All rejected material is sent for incineration (Tonini, 2018).

Anaerobic digestion combined with post-composting/post-maturation is the most suitable option for biodegradable plastics to ensure the largest possible degradation of the bio-based plastic products. This combination of AD and post-composting is a rising trend in Europe (De Wilde, 2018). There are multiple different combinations of AD plants within Europe, and some are more compatible for biodegradable plastic input (De Wilde, 2016). The output fraction from the AD, which includes the nutrient rich substances, could either be used directly on agricultural land (digestate) or further treated (post-maturation)

producing compost. For the "other anaerobic technologies" (other than wet thermophilic plants) 20 % of the digestate is sent for post-maturation and 80 % of the digestate is used directly on the field.

The following Table 23 summarises parameters and their values for the anaerobic digestion plant included in the EASETECH modelling. The biogas utilization (distributed between electricity, heat and/or fuel) is largely dependent on individual Member States, e.g. current energy infrastructure and policies. As an example, Sweden and Switzerland use most of the produced biogas as fuel in the transport sector while other countries produce mainly electricity and/or heat. Heat and electricity are substituted with marginal energy (see section 3.4.3 Energy in all technologies), while upgrading of biogas for transportation is assumed to substitute for conventional petrol.

Table 23. Factors for average European anaerobic digestion plant.

Parameter	Value	Reference
Biogas production		
Gas yield	70 %	EASETECH, data from a number of European plants
Loss of methane in plant	2 %	EASETECH
Methane in the biogas	63 %	EASETECH
Biogas utilisation		
Biogas for combined heat and power plant	90 % of biogas produced	European Biogas Association, 2011
Heat efficiency	50 %	European Commission, 2016
Electricity efficiency	40 %	European Commission, 2016
Biogas upgraded to fuel	10 % of biogas produced	European Biogas Association, 2011

3.5 Data quality, critical assumptions and limitations in End of Life modelling

The most important uncertainties on data quality, critical assumptions and limitations for the EoL modelling are presented below.

Estimated EoL mix, average EU – For each case study, a mix of EoL technologies was chosen to represent the most likely EoL treatment of the product on average across EU countries. The assumed EoL mixes are based on available data for current waste treatment in different EoL technologies, and the distribution of different waste fractions between them. These waste statistics are of varying quality and thus varying uncertainty.

For the bio-based biodegradable plastic products, the most likely EoL mix is even more uncertain. In most EU countries the waste systems have not yet taken these products into account, i.e. it is not clear what pathway the new products should follow. Therefore, the representativeness of the defined EoL technology mix is uncertain with respect to the waste system today and the uncertainty increases for future EU waste systems.

However, to perform an LCA of the case study with only one result for EoL, an EoL mix must be defined and the described EoL mix is the best estimate possible based on the data available.

Energy, marginal technologies – A marginal EU energy mix has been used for modelling the marginal energy (energy substitution). These marginal processes are valid for the time period of 2015-2017. However, the energy sector is currently changing and the pace of implementation of more green/renewable energy contributes to an important uncertainty of the applied data for marginal electricity and heat.

Energy, development of the EoL technologies – Development of the EoL treatment technologies may increase the efficiency of the specific EoL technology, thereby lowering the energy consumption and increasing the energy production (primarily from waste incineration and anaerobic digestion). These changes will affect the environmental

performance of the specific EoL treatment technologies and thereby may alter the environmental "ranking" of the technologies.

Substitution, PLA – In the calculation, it is assumed that recycling of PLA substitutes the petrochemical reference (PET or PP). It is possible, that recycling of PLA will substitute virgin PLA in the future, which would change the effects of substitution (making the recycling of PLA less environmentally beneficial). Similarly, other assumptions concerning substitution may be uncertain due to e.g. market mechanisms.

In-situ field-based biodegradation – The modelling of in-situ degradation of agricultural mulching film and the clips is based on a Danish soil type (sandy loam soil). If another soil type is chosen, the results of the modelling will be different, e.g. with respect to distribution of carbon and nutrient, degradation of organic matter, carbon storage, nutrients losses etc. The in-situ biodegradation modelling focuses on carbon and nutrients with respect to degradation patterns, distribution and losses, but does not take other impacts of e.g. microplastics in the soil into account.

Biodegradation in anaerobic digestion. Biodegradable plastics are not prone to degrade in all types of AD plants. There are many different types of installations in AD plants (e.g. mesophilic/thermophilic, dry/wet, with/without post composting, retention time) that will affect the biodegradability. The degradation will moreover depend on the product type, thickness, additives and biogenic carbon content. To model an average European AD plant is very complex, and the estimations made for modelling this technology are subject to uncertainty. This complexity of the AD biodegradation is supported by De Wilde et al., 2016, as Open-BIO does not propose a label for products suitable for anaerobic digestion due to the complexity of varying components in this technology.

Biodegradation in landfills is uncertain and will depend on the landfill conditions such as temperature, waste composition in the landfill and the plastic type.

The amount of rejects in biological treatment is subject to uncertainties because the pre- and post-sorting of the industrial composting plants and the anaerobic digestion have different technical set-ups. There is not much literature on this matter. The amount of rejected biodegradable plastic in the industrial composting and AD plants is also prone to change if more biodegradable plastic enters into the waste stream. The amount of rejects will be highly dependent on the type of composting or AD facility.

Consumption of materials and energy at the EoL treatment plants – Data has to a large extent been collected from existing databases including LCA databases. Some data may therefore represent a specific plant and could not reflect the most recent technological developments. However, since the LCA should reflect average EU treatment technologies, this is not seen as a major issue.

Littering is not modelled in this LCA, since no standard method exist to perform such a modelling. The justification for this choice is explained further in a previous section (3.1.4 excluding plastic littering from the EoL modelling).

The chemical composition of the biodegradable plastics does not always contain a full chemical composition. It consists of the pure plastic product, i.e. the pellets which are used to produce the products (includes polymer and additives). In any further manufacturing of the products, additional items like ink or labels could be added to the product. This is not included in the modelling in this LCA study, which could give the biodegradable products an advantage in certain impact categories and a disadvantage in other impact categories. The petrochemical and bio-based material compositions are on the other hand based on a chemical analysis of waste plastics, which includes chemical elements from the possible items added to the products (e.g. ink and labels). See the full chemical compositions of the product systems in Annex 1.

4 Land use change methodology

4.1 Definitions

It should be noted that several definitions are applied when it comes to land use changes, in particular for direct land use changes (DLUC). For instance, Marelli et al. (2015) define a land use change to be direct (DLUC) when the demanded crops[9] are grown on uncultivated land (and indirect when grown on already cultivated or used land). Similarly, the PAS2050 (BSI, 2011) defines it as a "change in the use of land at the location of production of the product being assessed", although the methodology it proposes involve a slightly different definition, as later discussed. In the scientific literature, DLUC has also been defined as "all changes in above- and below-ground flows of carbon, nitrogen and phosphorus flows on a particular site, as one land use takes place instead of another" (e.g. Hamelin et al. 2012; Tonini et al. 2012). Finally, the amended Renewable Energy Directive (RED; European Union, 2015) defines DLUC as arising when the production of feedstock has led to "a change from one of the following IPCC land cover: forest land, grassland, wetlands, settlements, or other land, to cropland or perennial cropland", a definition that appears close to the one proposed by PAS2050.

In this study, the definition (and in particular the methodology) of BSI (2011) is used for DLUC, this being a requirement of the latest PEFCR Guidance (v6.3) (European Commission, 2018), as it will later be discussed into more details.

Indirect land use changes (ILUC), on the other hand, occurs when the crops under consideration are grown on already cultivated land. In other words, ILUC arises as changes in overall land demand occur. The rationale (and pre-condition) behind ILUC is that the global agricultural area (as a whole) is still expanding (drivers include increased population, GDP increase of some countries, etc.), and is limited. For example, if the feedstock needed for a given bio-based case study is cultivated at the expense of another crop, that other crop, or rather the service it provided (say protein feed), will still be demanded on the World's market. The main underlying postulate of ILUC is that this relative drop in supply (e.g. of protein feed) is likely to cause a relative increase in agricultural prices, which in turn provides incentives to increase the production elsewhere. Such increased crop production[10] will ultimately stem from agricultural land expansion and/or production intensification (often both in combination). In a nutshell, the point of departure for ILUC to occur is when arable land (whether cultivated or hosting grazing activities) is used for supplying the feedstock of interest. To compensate for the feedstock that was previously supplied from that arable land, a (more or less) long chain of land use changes takes place (e.g. the initial say displaced wheat from country A being replaced by barley from country B, itself displacing maize from country C, and so on…) until there is no more displacement possible, whether because production intensification took place or because non-arable land has been converted to agriculture.

[9] In the case of Marelli et al. (2015), the crops are demanded for biofuels production.

[10] It should be noted that not 100 % of the displaced feedstock (e.g. protein feed) may be compensated by increased production (in the short-to-medium-term). In fact, as displacing the commodity leads to a higher price for it, it is often forecasted that the demand for that commodity will simply decrease, so part of the displaced feedstock is never replaced (Edwards et al., 2010; Hertel et al., 2010a; Laborde, 2011). In other words, this is the effect of higher prices reducing the land demand (and use) but increasing hunger. However, Schmidt et al., (2015) argue that this effect should not be included in LCAs, since it is the long-term effect of the demand that should be guiding for decisions (Weidema et al., 2013). According to these authors, the supply of goods and services should, as long as attempting to model the long-term consequences of decisions, be assumed to be fully elastic, i.e. an increase in demand is to be met by a corresponding (1:1) increase in supply. Since this study uses a deterministic model, short-term effects are not captured, therefore a 1:1 replacement is considered.

Definitions for ILUC are generally consistent with one another. In the updated RED (where the focus is on fuels), ILUC is defined as follows: "Where pasture or agricultural land previously destined for food and feed markets is diverted to biofuel production, the non-fuel demand will still need to be satisfied either through intensification of current production or by bringing non-agricultural land into production elsewhere. The latter case constitutes indirect land-use change […]". In PAS2050, ILUC is defined as "the change in the use of land elsewhere [than the country where the feedstock of interest stems from]", while Marelli et al. (2015) define ILUC as the land-use changes occurring around the world as the result of growing the feedstock of interest on existing arable land (thereby displacing the production that was occurring on that land).

The relevance of addressing DLUC and ILUC separately rather than just addressing the overall land use changes (LUC) has been questioned several times. In many studies addressing the full land use changes impacts, the "ILUC" impact factors developed often englobe the "DLUC" ones (e.g. Schmidt et al., 2015; Tonini et al., 2016).

To refer to the case of changes in flows related to the cultivation of the feedstock involved in this study, the term "field emissions" will be used herein[11]. Field emissions here refers to all emissions taking place "in the field", as a result for example of manure application, mineral fertiliser application, CO_2 uptake by the plants, (eventual) degradation of crop residues on the field, etc., but does not include emissions from other processes such as "use of tractor", "production of mineral fertilisers", etc. "Field emissions" and the emissions from these "others non-field processes due to crop production" are then combined to form the full "crop production" dataset. Additional details on this are supplied in section 4.4 of this report.

4.2 Direct Land Use Changes assessment (DLUC)

This study strives to follow the recommendations of the PEFCR. The latest PEFCR guidance document (v.6.3) indicates that "*for land use changes: all carbon emissions and removals shall be modelled following the modelling guidelines of PAS 2050:2011 (BSI, 2011) and the supplementary document PAS2050-1: 2012* (BSI, 2012) *for horticultural products.*" The PEFCR then literally reproduces (and connects) several sub-sections of the PAS2050. The key ones are reported below:

- "Examples of direct land use changes are the conversion of land used for growing crops to industrial use or conversion from forestland to cropland".

- "While GHG emissions also arise from indirect land use change, the methods and data requirements for calculating these emissions are not fully developed. Therefore, the assessment of emissions arising from indirect land use change is not included".

- "The GHG emissions and removals arising from direct land use change shall be assessed for any input to the life cycle of a product originating from that land and shall be included in the assessment of GHG emissions. The emissions arising from the product shall be assessed based on the default land use change values provided in PAS 2050:2011 Annex C, unless better data is available".

- "The assessment of the impact of land use change shall include all direct land use change occurring not more than 20 years, or a single harvest period, prior to undertaking the assessment (whichever is the longer)".

Finally, the PEFCR guidance (v.6.3) mentions the following regarding soil carbon (that slightly goes beyond what can be found in the PAS2050:2011 and PAS2050-1:2012):

[11] In the PEFCR guidance, these emissions are addressed under "Agricultural modelling" (section 7.10 of the PEFCR guidance v.6.3)

- "Soil carbon uptake (accumulation) shall be excluded from the footprint results as it is highly questionable how the long-term uptakes (beyond 100 years) can be guaranteed in practice".

In PAS2050:2011 (more precisely in Appendix C), two main types of previous land use are considered for DLUC: transformation from grassland (to annual or perennial crop) and transformation from forest land (to annual or perennial crop).

This above methodological guidelines extracted from the PEFCR guidance document (v.6.3) implies a crop-specific reasoning for a given country, rather than an analysis of the "demand for cropland" in that country. To illustrate this, the example of sugarcane in "country X" is taken. Say that the product under assessment is a given bio-based material (as in this study), and that a certain portion of the sugarcane farms in "country X" are now supplying sugarcane to that (new) market, rather than the sugar market as they were previously doing. If these farms, or rather lands, have been under sugarcane for at least 20 years from the date the LCA is made, the PAS2050:2011 method implies that DLUC is zero, even if it can be shown that additional demands of cropland in "country X" tend to take place at the frontier between nature and agriculture. In this example, the methodology of the PEFCR guidance document (v.6.3) disconnects DLUC and ILUC (in this example by ignoring the sugar market; i.e. which new crop(s)/farm(s) will now supply the feedstock no longer supplied by the farms that are now selling their feedstock to the "bio-based materials" market). Relating to the sugarcane example, the methodology proposed by the PEFCR guidance document (v.6.3) (and PAS2050) also involves that while the DLUC of sugarcane in "country X" is zero, the one of another crop with a shorter history in "country X", say maize, may be very high.

It should also be highlighted that the PEFCR guidance (v.6.3) is only concerned with assessing the GHG emissions (and removals) arising from direct land use changes, and these "*shall be assessed based on the default land use change values provided in PAS 2050:2011 Annex C, unless better data is available*". Yet, the direct land use changes methodology in PAS 2050:2011 (supplemented by the PAS2050-1:2012) is only accounting for carbon flows, and these are exclusively translated into CO_2 emissions. This involves that, according to the PEFCR guidance (v.6.3), only the "climate change" impact can be addressed (CO_2 affects this impact category only, according to the impact assessment models considered in Table 4).

In this study, to comply with the PEFCR v6.3 guidance, a 2-steps methodology was followed. The methodological guidelines to calculate DLUC are detailed in section 5.7 of PAS2050:2011 and supplemented in PAS2050-1:2012 (section 5.2.3). These will not be reproduced herein, but refered to.

4.2.1 Step 1: Defining the crop x country combinations and areas affected by DLUC

PAS2050-1:2012 provides, in its section 5.2.3, equations to calculate the area affected by DLUC (and eventually the overall resulting GHG emissions), for a given crop in a given country. In other words, it provides equations to estimate how many of the current hectares of crop A in country X were, in the last 20 years, under grassland and forestland (or under annual/perennial cropland: but this gives no DLUC). To ease these calculations, Blonk Consultants has been contracted to produce a "DLUC tool" providing these values based on the FAO statistics, which have been continuously updated since (the latest version is 2017.2) (BlonkConsultants, 2018). This company is also producing the Agri-footprint life cycle inventory datasets (Durlinger et al., 2017) for a variety of crop and agricultural processes (used in this study), where these DLUC (in terms of area converted) are available. The latest version of these datasets (Agri-footprint v.3.0) can be found in the LCA software SimaPro 8.4.0.0 (there, it is mentioned that the Agri-footptint dataset were last updated 16th June 2017; it uses the "DLUC tool" version 2016.1).

This study used these Agri-footprint v3.0 dataset to retrieve the area converted from grassland and forestland, for each of the "crop x country" combinations considered. The results are shown in Table 24, where it can be observed that only a few "crop x country"

combiations give rise to DLUC (according to this methodology), namely sugarcane from Brazil, maize from Germany, and wheat from France and Germany.

Table 24. Areas of forestland (F) and grassland (G) converted for the "crop x country" combinations considered in this study, as retrieved from the Agri-footprint dataset of 16-06-2017. Values in m^2 forestland or grassland per ha of crop[a]

	Sugarcane, BR	Sugarcane, TH	Maize, US	Maize, IT	Maize, FR	Maize, DE	Maize, PL	Maize, ES	Maize, HU	Maize, RO	Maize, BG	Wheat, FR	Wheat, DE	Wheat, PL	Wheat, CZ[b]	Wheat, ES	Wheat, HU	Wheat, RO	Wheat, BG	Sugar beet, FR	Sugar beet, FR	Sugar beet, FR	Sugar beet, FR	Sugar beet, FR	Potato, DE
F	260	0	0	0	0	0	0	0	0	0	0	0	0	0	-	0	0	0	0	0	0	0	0	0	0
G	0	0	0	0	0	15	0	0	0	0	0	4	8	0	-	0	0	0	0	0	0	0	0	0	0

a Country codes based on ISO 3166-1 (ISO, 2013);

b No figures (empty field) provided for this crop. Therefore, the DLUC was assumed to be as for wheat in Poland.

4.2.2 Calculating the GHG related to DLUC

Building from the figures of Table 24, the data of Annex C of PAS2050:2011 were used to calculate the GHG emissions related to DLUC (as defined by PEFCR). The results are shown in Table 25. Again, it should be noted that this methodology for direct land use changes only allows accounting for CO_2 emissions, and no other substances.

Table 25. DLUC emissions considered in this study.

Crop involvind DLUC in this study	Area of DLUC (m^2 ha^{-1} of crop) (Table 24)	GHG emission factor from Annex C of PAS2050 (t CO_2 eq. ha^{-1} y^{-1})	DLUC emission[a] (kg CO_2 eq. ha^{-1} crop y^{-1})
Sugarcane, BR	260 (forestland)	26	677
Maize, DE	14.8 (grassland)	7	10.4
Wheat, FR	3.9 (grassland)	4.5	1.7
Wheat, DE	7.8 (grassland)	7	5.5

a Calculation example for sugarcane, BR: (26 t CO_2 eq. / ha DLUC*y) *x* (260 m^2 DLUC / ha sugarcane) *x* (ha / 10000 m^2) *x* (1000 kg CO_2 eq. / t CO_2 eq.) = 677 kg CO_2 eq. ha^{-1} sugarcane y^{-1}

4.3 ILUC assessment

The impacts of ILUC, although not to be included in the main LCA results according to the PEFCR guidance document v.6.3, are nevertheless estimated and shown separately as a "non-PEF" impact. Here, a deterministic approach based on historical deforestation data (2000-2010) was used[12]. Therefore, the method does not estimate the impacts of the future cropland demand for EU bio-based products but provides an estimate of the overall deforestation and intensification emissions associated with demanding 1 ha of cropland in the past. It is here the hypothesis that this estimate provides useful insights to anticipate the magnitude of these emissions in the future.

Further, as a consequence of the approach used, the generic "ILUC factor" derived herein is to be seen as a "LUC factor", as it provides the average emissions related to the average annual deforestation/intensification that took place between 2000 and 2010. Therefore, it does not distinguish whether the deforestation is a "direct" or "indirect" result of additional land demand. The consequence of this is that the ILUC factor derived with this methodology should not be used in combination with any DLUC emissions, but on its own reflecting the overall land use change impacts. Doing otherwise would lead to overestimate the land use emissions.

Further, it should be added that the emissions factors for climate change are found in Table 13 of PEFCR v.6.3 were used (steps 4-5 below). This involves that specific (land use changes) flows were created, and that the following climate change impact potentials were used:

- CO_2 (land use changes), air emission: 1 kg kg^{-1} CO_2 eq. (note: it is 0 for "CO_2, biogenic, air emission").
- CO (land use changes), air emission: 1.57 kg kg^{-1} CO_2 eq. (note: it is 0 for "CO, biogenic, air emission").

[12] This methodology is an adaptation (and update) of the methodology proposed by Tonini et al. (2016).

- CH_4 (land use changes), air emission: 36.75 kg kg^{-1} CO_2 eq. (note: it is 34 for "CO, biogenic, air emission").

- N_2O (land use changes), air emission:298 kg kg^{-1} CO_2 eq. (not specified in Table 13 of PEFCR as it is not a C flow; the same factor as used through this study was thus used).

The impacts of ILUC in terms of changed carbon (CO_2, CH_4) and nitrogen flows (NO_x, NH_3, N_2O) (among others) are here assessed based on the following 6-steps methodology (see Figure 14):

- Step 1: Determine the share of the ILUC response due to expansion and to intensification

- Step 2: Determine the types and amount of land (biomes) that underwent expansion in the last 10 years of available deforestation data, and their location in the World (region). As a result, a "biome *x* region" land expansion matrix is produced.

- Step 3: Determine how much of the observed deforestation from step 2 is actually due to the demand for arable land

- Step 4: Estimate the carbon and nitrogen emissions from land expansion, considering the biomes x region matrix of step 2, and the share of deforestation due to arable land demand of step 3

- Step 5: Estimate the emissions due to intensification

- Step 6: Derive a generic ILUC factor

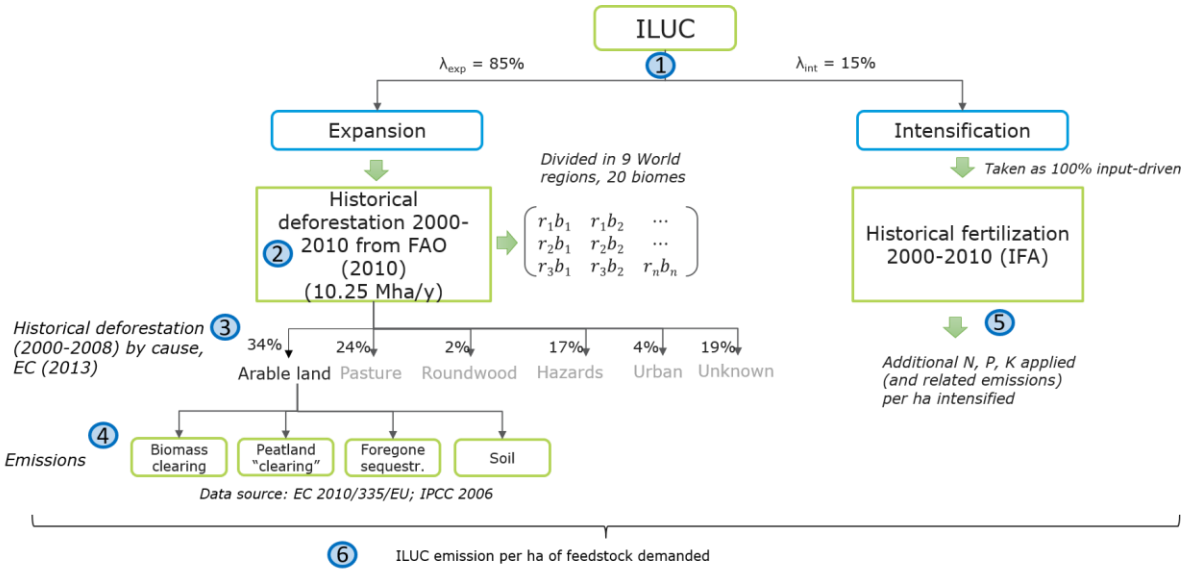


Figure 14. Illustration of the 6-steps deterministic methodology used to assess the impacts of ILUC

4.3.1 Step 1: Share of the response from expansion and intensification

This step endeavour to determine the share of the ILUC response that is due to expansion and intensification. These shares will be re-used in Step 6 of the methodology.

While expansion is straightforward (transformation of nature to agricultural land), intensification may be achieved through three main pathways:

- Input-driven pathway: This refers to any yield increases obtained through changes in farm inputs (e.g. fertilisers, pesticides, irrigation, etc.). The increases in yield obtained this way may however be reversible. They are also characterised by the so-called "diminishing returns", meaning that for each additional unit of input (e.g. fertiliser) applied, the magnitude of the additional yield becomes lower and lower, until it becomes practically negligible.

- Innovation-driven pathway: This refers to any yield increases obtained through technological development (e.g. harvesting technologies allowing to recover more biomass, plant breeding, etc.), and is seen as a more permanent effect (Marelli et al., 2011). However, a lag of ca. 20 years is likely before research and development activities actually translate into yield increases (Edwards et al., 2010). In Edwards et al. (2014), it is referred to as yield increases due to time.

- Multi-cropping pathway: This consists to grow more than one crop on the same hectare of land for a given year, which in some countries allows a harvest all year-round. This represented (in 2010) ca. 18 % of the world's cropland, and higher crop prices can be envisioned to increase the profitability of this practice (Marelli et al., 2011).

While input-driven and multi-cropping intensification are directly driven by crop prices (and thus demand growth), innovation-driven intensification is dependent upon public and private research investments, and as such is more difficult to predict (Kløverpris, 2008). It is also typically excluded in ILUC studies, among other because of the difficulty to predict it. However, it is well acknowledged as the most sustainable way to increase the yields, especially in developing countries (Marelli et al., 2011; Tilman et al., 2011).

To evaluate the share of the ILUC response stemming from intensification and expansion, past time-series data (2002-2012) retrieved from the FAOSTAT database in terms of crop production, crop yield, and area of arable land were used. The global crop production was calculated as the sum of specific crop groups (those found in FAOSTAT), following the approach suggested in Schmidt et al. (2015). Accordingly, expansion was found to represent 37 % of the ILUC response, and intensification 63 %, when considering production data per wet weight (considering production data per dry weight, expansion was found to represent only 25 % of the ILUC response).

However, these results were adapted, at the light of Marelli et al. (2015) and Edwards et al. (2014). If fact, as highlighted in Marelli et al. (2015), the increase in yield with time is of secondary interest; what really matters is how yields respond to crop price changes. Yet, Marelli et al. (2015) argue that available historical data cannot demonstrate that the observed historical intensification has responded to price changes. In other words, historical data on yields could not be correlated to historical data on crop prices.

Similarly, Edwards et al. (2014) argue that by considering "real data" on deforestation, the (mitigating) intensification effect is implicitly accounted for. This is correct, although the emission implications of additional fertiliser use (one major parameter of input-driven intensification) are then not reflected.

Here, based on Marelli et al. (2011), the (minimal) share of intensification (λ_{int}) was taken to 15 % of the ILUC response. The share of expansion (λ_{int}) was thus taken to 85 %. This is seen to be a conservative choice and reflect minimal intensification. Further, this study models intensification as 100 % input-driven (step 4), which may overestimate the impact of increased fertilization and underestimate the (likely beneficial) impact of multi-cropping.

4.3.2 Step 2: Geo-quantification of arable land expansion

This step is probably the one where methodologies differ the most. Two main approaches are typically used: economic equilibrium models and causal deterministic models (Warner et al., 2013). It is beyond the scope of the present study to elaborate on the details, strengths and drawbacks of these respective approaches. For this, the reader is referred

to Warner et al. (2013) as well as to Marelli et al. (2011). Key highlights of these approaches are nevertheless presented below.

The approach relying on economic equilibrium modelling is often used in studies modelling the environmental consequences (or most often the GHG emissions) of ILUC, in particular in the context of biofuels (e.g. Edwards et al., 2010; Laborde, 2011; Marelli et al., 2011; Searchinger et al., 2008; Valin et al., 2015). Such approach is used due to the very nature of the ILUC process: changes in land use result in changes in crop supply that are transmitted across global markets linked by commodity substitutability and competition through numerous interactions. To cope with these, it is argued that sophisticated models allowing to represent the World crop markets are essential. Such econometric models are based whether on computable partial (representing one sector of the economy) or general (representing the whole economy) equilibrium models. They are seen to provide results relevant for the short- and medium-term period (Marelli et al., 2015).

A less complex, but more transparent & reliable over time[13] alternative is the use of a causal-descriptive deterministic model, also often referred to as a biophysical approach. This is the approach that has been used in this study. The vision is to establish a cause-effect relationship between demand for arable land and expansion/intensification effects using historical statistical data about deforestation, natural biome losses (e.g. shrubland, grassland), crop yields, and fertilisers consumption. In other words, the vision is to derive a "generic" emission factor per initial hectares of arable demanded to grow the feedstock.

To determine the type and location of arable land, the approach developed in Tonini et al. (2016) has been used, and further developed. This approach is also along the lines of the methodology presented in Edwards et al. (2014). In the present study, a methodology using features of both approaches have been used (i.e. Tonini et al., 2016 and Edwards et al., 2014).

The starting point is the deforestation that has occurred between 2000-2010 (latest available data), retrieved per World regions from FAO (2010; Table 3)[14]. Accordingly, an average of 10.25 Mha were annually converted, of which 40 % in South America, 34 % in Africa, 12 % in South-East Asia, 6.9 % in Oceania, 4.6 % in Central America, 2.2 % in the "rest of Asia", 0.2 % in East Europe and 0.3 % in "rest of Europe", while the deforestation for United States and "Rest of North America" is negligible.

Based on IPCC (2006a), forest losses were divided into 5 biome categories (tropical, subtropical, temperate, boreal and polar). Except the latter, these were all further sub-divided (e.g. tropical rain forest, topical dry forest, etc.). As a result, all biomes of IPCC (2006a) were included. To link these biomes to the regions of deforestation from FAO (2010) (e.g. "Africa"), it is necessary to distribute the forest loss in each region within each biome of the region. This was done based on the distributions in (FAO, 2000) (reference year: 2000). In total, 82 combinations of regions x biomes were considered (see Table 26)

[13] i.e. the approach can be cross-checked and the results replicated by a third party in e.g. 5 years' time, while this is less likely (though not impossible) with an equilibrium model, be it because the exact version of the model used to generate the results is no longer available a few years after the study has been released, because the third-party modeller would take different choices, etc.

[14] For South-East Asia, out of the total, an area of 0.26 Mha y^{-1} has been considered to reflect peatland losses. This is the average annual figure for peatland losses in Indonesia & Malaysia in 2000-2010, based on FAO (2012).

Table 26. Average annual natural vegetation loss (Mha y^{-1}) per World region 2000-2010 and coverage of biomes (%) in these regions.

Region[a]	Area loss	Tropical					Sub-tropical				Temperate			Boreal			Polar (%)	Total (%)
	Mha y^{-1}	Rain (%)	Moist (%)	Dry (%)	Shrub (%)	Mountain (%)	Humid (%)	Dry (%)	Steppe (%)	Mountain (%)	Oceanic (%)	Continental (%)	Mountain (%)	Coniferous (%)	Tundra (%)	Mountain (%)		
AF	-3.47	33	27	30	6	0	3	0	1	0	0	0	0	0	0	0	0	100
RoA	-0.23	3	3	11	2	0	4	24	1	1	0	17	0	13	5	1	11	100
SEAS	-1.25	62	13	13	1	0	10	0	0	0	0	0	0	0	0	0	0	100
OC	-0.71	16	1	30	11	0	2	6	5	18	3	0	5	0	0	0	4	100
EEU	-0.02	0	0	0	0	0	0	0	0	0	0	0	0	13	1	0	5	100
RoEU	-0.03	0	0	0	0	0	0	0	22	0	0	4	26	8	0	0	13	100
NA	0.00	0	0	0	0	0	0	11	0	1	0	5	0	15	0	1	19	100
CA	-0.47	23	32	9	0	0	15	0	0	3	4	14	0	0	0	0	0	100
SA	-4.07	66	13	12	0	0	5	1	1	0	0	0	1	0	0	0	0	100
Wd	-10.25																	

a "Regions" in this study are defines as follows: AF: Africa; RoA: Rest of Asia; SEAS: South-East Asia (Brunei, Cambodia, Indonesia, Lao, Malaysia, Myanmar, Philippines, Singapore, Thailand, East Timor, Vietnam); OC: Oceania; EEU: East Europe (Countries of the former Soviet Union including Russia); RoEU: Rest of Europe; NA: North America (Bermuda, Canada, Greenland, Saint-Pierre-et-Miquelon, United States); CA: Central America; SA: South America (Argentina, Bolivia, Brazil, Chile, Columbia, Ecuafor, Falkland Island, French Guyana, Guyana, Paraguay, Peru, Suriname, Uruguay, Venezuela); Wd: World

4.3.3 Step 3: Attributing deforestation to the demand for arable land

The next step is to understand how much of this deforestation (2000-2010) was due to the demand for cultivated land (i.e. the driver studied herein), ξ. In an earlier (but recent) effort, this Task has been performed for DG-Environment (European Commission, 2013). According to this, 34 % of the deforestation that has occurred between 2000-2008 was due to the demand for cultivated land. It is here assumed that this value holds true for 2000-2010. The regional (overall) deforestation values of FAO (2010) were thus adjusted accordingly (ξ = 34 %). This aspect was for example not considered in Tonini et al. (2016).

4.3.4 Step 4: Carbon and Nitrogen emissions from expansion

To estimate the amount of carbon lost during the conversion of natural lands (those from Table 26) the above-ground biomass data from European Commission (2010) and IPCC (2006a; Tables 4.4, 4.7, 4.9 and 6.4) were used. Two types of emissions were considered: the initial emissions from land clearing (above-ground biomass), and the emissions from foregone sequestration. The latter reflects the CO_2 uptake that would have otherwise taken place.

The land clearing emissions from eventual carbon and nitrogen losses from below-ground biomass and soil carbon were not considered. This may underestimate the overall ILUC effect. In fact, it has been considered in different studies (e.g. Gibbs et al., 2008; Müller-Wenk and Brandão, 2010; Searchinger et al., 2008) that 25 % of the carbon in the soil (presumably the top layer) is released in the atmosphere as CO_2 during the initial land clearing. This is especially important in the case of boreal biomes, where ca. 65 % of the total C stock is found in the soil (this does not including any carbon from living biomass), while this figure is relatively lower for the temperate (48 %) and tropical biomes (30 %) (Pan et al., 2011). Here, as reflected in Table 26, over 75 % of the historical deforestation considered (2000-2010) occurred in tropical biomes. According to Edwards et al. (2014), this exclusion causes an 8-30 % underestimate in emissions. Edwards et al. (2014) also did not include the soil carbon emissions caused by deforestation on mineral soils.

The emissions (CO_2-C) from foregone sequestration were considered as a loss of ecosystem service over a period of 20 years, considering mature trees only (above-ground biomass > 20 years in IPCC). These emissions are calculated according to Equation 1 (adapted from Equation S17 of Tonini et al., 2016):

$$CEF_{\mathrm{FS,b,r}} = \sum_{b,r=1}^{n} \frac{(1+\mathrm{R})\times \mathrm{AG}_{\mathrm{t>20y\,b,r}} \times \mathrm{C} \times 44 \times SL_r \times LL_r \times \xi}{12} \qquad \text{Equation 1}$$

$CEF_{FS,b,r}$: carbon dioxide emission for foregone sequestration (t CO_2-C ha^{-1} y^{-1})

$AGt_{>20}$: above-ground biomass growth for t>20y (IPCC, 2006a; table 4.9) (t DM $ha^{-1}{}_b$ y^{-1})

C: carbon content in biomass (47 %; IPCC, 2006a) (t C t DM^{-1})

R: below-ground to above-ground biomass ratio (IPCC, 2006a; table 4.4) (%)

SL_r: share of land covered by biome b, in region r (from Table 26) (%)

LL_r: land cover loss of biome b, in region r (%)[15]

[15] Example for tropical rain forest, in region "Africa" (all values can be found in Table 26): 33 % x -3.47 Mha y^{-1} x (1/10.25 Mha y^{-1}) = 11 % .

ξ:portion of deforestation due to the demand for arable land (34 %, see text)

b,r: indicate biome type and geographic region, respectively.

The emissions of CO_2, CO, CH_4, N_2O, NO_x from biomass clearing are calculated as shown in Equation 2, considering the biomass data stipulated in (European Commission, 2010), but the data of IPCC (IPCC, 2006a; Table 4.7 and 6.4) were used when no other data were available.

$$CEF_{LC,b,r} = \sum_{b,r=1}^{n} \frac{AG_{b,r} \times EF_{LC,b} \times SL_r \times LL_r \times \xi}{C \times 1000}$$ Equation 2

$CEF_{LC,b,r}$: Emission (CO_2, CO, CH_4, N_2O, NO_x) for land clearing (t emission ha^{-1} y^{-1})

$AG_{b,r}$: above-ground biomass stock (European Commission, 2010, else IPCC, 2006a; tables 4.7 and 6.4) (t C $ha^{-1}{}_{b,r}$ y^{-1})

$EF_{LC,\ b}$: Emission factor for biomass burning, for CO_2, CO, CH_4, N_2O, NO_x (IPCC 2006b; Table 2.5) (g kg^{-1} DM)

Land clearing also involves clearing of peatland. This was considered only for the annual average peatland loss from Malaysia & Indonesia (0.26 Mha y^{-1}; derived from tables 4 and 7 of Joosten et al., 2012). The emission factors provided in IPCC (IPCC, 2013; tables 2.1, 2.3, 2.5) were considered, as shown in Equation 3:

$$CEF_{PL} = \frac{APL \times EF_{PL} \times \boldsymbol{SPL} \times \xi}{AFL}$$ Equation 3

CEF_{PL}: Emission (CO_2-C, CH_4, N_2O-N) from peatland losses (kg or t emission ha^{-1} y^{-1})

APL: Average annual peat loss (0.26 Mha y^{-1})

AFL: Average annual forest loss (10.25 Mha y^{-1})

EF_{PL}: Emission factor for peatland loss, for CO_2-C, CH_4, N_2O-N (IPCC, 2013; tables 2.1, 2.3, 2.5) (kg or t ha^{-1} y^{-1})

SPL: share of peatland cover in SouthEast Asia (19 %, this can be derived from Table 26 and footnote 14).

It may be argued that peatland losses should not be included (e.g. European Commission, 2010), drawing a parallel from the sustainability criteria laid down by Directive 2009/28/EC (Renewable Energy Directive). The modular database was thus built to easily allow peatland losses inclusion or exclusion (switch) when addressing ILUC (expansion). By default, peatland losses are not included in the ILUC factor calculated herein.

4.3.5 Step 5: Emissions related to intensification

As earlier stated, it is assumed that 100 % of the intensification response is met by increases in fertilisers (N, P and K). Yet, how much additional fertiliser is applied per hectare intensified? Here, emissions from intensification were calculated according to the emission factors presented in Tonini et al. (2016). These are based upon statistical data retrieved from the International Fertiliser Association (IFA) database on the annual change (2000-2010) in global N, P and K fertiliser use and elaborated statistics on the annual change in agricultural (fertilised) land. Accordingly, the following factors were used: 166 kg N ha_{int} y^{-1} (taken as urea), 68 kg P_2O_5 ha_{int} y^{-1} (taken as diammonium phosphate), 47 kg K_2O ha_{int} y^{-1} (taken as potassium chloride). Emissions (N_2O, NH_3, NOx) from the application of urea were taken from Hamelin et al. (2012), while NO_3 were taken as 20 % of the applied N based on Galloway et al. (2004). Losses of P to water compartments were taken as 1 % of the P applied. Emission dataset related to the manufacture of these fertilisers were retrieved from the Ecoinvent database (v3.4).

4.3.6 Step 6: Deriving a generic ILUC factor

In this Step, the emissions found per area expanded (Step 4) and per area intensified (Step 5), are aggregated based on their respective shares for the ILUC response (Step 1). This is summarised in Table 27. To relate these emissions to the case study (demand of ha y^{-1}), emissions are annualised (i.e. distributed equally) over 20 years, in line with most LUC calculations used by the Commission (Edwards et al., 2014).

Table 27. Summary of emission due to indirect land use changes.

	Emission[a] or material input	Value per area expanded or intensified	
		Value	Unit
Expansion (85 %)	$CO_{2,FS}$	0.74	t ha^{-1}_{exp}
	$CO_{2,LC}$	71.6	t ha^{-1}_{exp}
	$CO_{2,PL}$	0.64	t ha^{-1}_{exp}
	$N_2O_{,LC}$	9.08×10^{-3}	kg ha^{-1}_{exp}
	$N_2O_{,PL}$	1.24×10^{-4}	kg ha^{-1}_{exp}
	$NO_{x,\ LC}$	0.073	kg ha^{-1}_{exp}
	$CH_{4,LC}$	0.307	kg ha^{-1}_{exp}
	$CH_{4,PL}$	1.60×10^{-4}	kg ha^{-1}_{exp}
	CO_{LC}	4.71	kg ha^{-1}_{exp}
Intensification (15 %)	N_2O	4.4	kg ha^{-1}_{int}
	NH_3	4	kg ha^{-1}_{int}
	NO_x	6	kg ha^{-1}_{int}
	NO_3-N	33	kg ha^{-1}_{int}
	N-fertiliser	166	kg N ha^{-1}_{int}
	P-fertiliser	68	kg P_2O_5 ha^{-1}_{int}
	K-fertiliser	47	kg K_2O ha^{-1}_{int}

a To air except for NO_3-N, which is an emission to water. FS: Foregone sequestration; LC: Land clearing; PL: Peatland losses.

4.3.7 Deriving an ILUC factor for urban land demand (proxy for petrochemical land use)

In step 3, it was determined that ca. 34 % of the global deforestation (2000-2010) was due to the demand for cultivated land (ξ = 34 %). Yet, one could ask how much deforestation was due to the demand for land by the petrochemical sector. Using the same approach as in step 3, it is not possible to answer this question for the specific petrochemical sector (and doing so is well beyond the scope of this study), but a proxy can be taken using "urban land" (Figure 14). Accordingly, this gives (ξ = 4 %). Using the same methodology as presented herein (steps 1-6), an ILUC factor of 0.12 t CO_{2e} ha^{-1} y^{-1} can be derived.

It should be noted that this should only serve as a very rough proxy to factor in the impact of deforestation driven by "urban land" demand, here assimilated to the land demanded for petrochemicals. It should also be remembered that petrochemicals are also demanded in the bio-based scenarios (among others in the background processes, for example the production of fertilisers); hence it is the difference (bio-based vs fossil-based alternative for a given case study) in land demand for petrochemical activities that should be assessed. Finally, one could argue that products inducing the demand for petrochemical products de facto induce additional risks for environmental damages caused by this industry, e.g. oil spills. Though this reasoning may seem reasonable, quantifying the likelihood of such risks (and any other risks, also those from the bio-based products production) is beyond the scope of state-of-the-art LCAs and of this study. It also mixes the concepts: land use is a real natural constraint (planetary boundary) while oil spills are a risk. Both can be mitigated, but the former cannot (under current food production technologies) be avoided if the overall demand for agricultural land is increasing, while the latter could theoretically be close to null if the right investments are made.

4.3.8 Comparison with other studies

The overall ILUC factor derived in this study, in terms of CO_2 eq., is compared to factors obtained in previous studies (Table 28 and Table 29). It should, however, be noted that it is well beyond the scope of the present study to make an extensive comparison of different ILUC models and their results. Such an analysis can be found in e.g. Woltjer et al. (2017). The idea here is simply to compare the generic factor derived in this study with other literature.

Since most ILUC results relate to the production of biofuels, the ILUC generic factor found in this study (per ha y^{-1}; Table 28) has been roughly translated to an ILUC estimate per MJ. This has been done using US maize bioethanol as a proxy (given all ILUC studies on this particular case), considering the following data (425 l ethanol Mg^{-1} maize DM, based on Wang et al., 2012), with an energy content of 21 MJ l^{-1}, and a yield of 11.9 Mg DM ha^{-1} y^{-1} – as used later in this study for US maize). Further, the ILUC is assumed to be reduced by roughly one-third, to account for the co-produced protein feed (based on Hertel et al., 2010b). This would yield an ILUC factor of 12.6 g CO_2 eq. MJ^{-1}. This crude estimate of our ILUC value in MJ falls within the lower end of the examples from the literature presented in Table 29.

Table 28. Comparison with other ILUC studies with factors per ha y^{-1}.

Study[a]	Model[b]	Scope	Co-product credit	Time (y)	ILUC factor, t CO_2 eq. $ha^{-1}y^{-1}$	Comment
Audsley et al. (2009)	BIO	-	-	-	1.4	General use
Schmidt and Munos (2014)	BIO	-	-	-	1.7	Value for "World average arable land"
Tonini et al. (2016)	BIO	-	-	100	4.1	General use
This study	**BIO**	**-**	**-**	**20**	**4.0**	**General use**

a Reference;

b Bio: Biophysical; PE: Partial Equilibrium; GE: General Equilibrium

Table 29. Comparison with other ILUC studies with factors per MJ.

Study[a]	Model[b]	Scope	Co-product credit	Time (y)	ILUC factor, g CO_2 eq. MJ^{-1}	Comment[c]
Searchinger et al. (2008)	GE	USA	Partly factored in	30	104	
Hertel et al. (2010b)	GE	USA	Factored in	30	27	
Valin et al. (2015)	GE	EU	Factored in	20	14	
Wang et al. (2012)	GE	USA	Unclear	30	9	
EC (2015)		EU	Factored in	20	12 -55	Amended RED; Mean values for sugar-based ethanol to oilseed biodiesel
Laborde (2011)	GE	EU	Factored in	20	38.4	NREAP 2020 scenario
Marelli et al. (2015)	PE+GE	EU	Factored in	20	36	Peat included, NREAP 2020 scenario
Edwards et al. (2014)	BIO	EU	Factored in	20	15 - 123	Cereals-based ethanol to oilseed biodiesel including peat
This study, rough proxy	**BIO**	**-**	**Roughly factored in**	**20**	**12**	**See text**

a Reference;
b Bio: Biophysical; PE: Partial Equilibrium; GE: General Equilibrium;

c See source studies for additional details. RED: renewable Energy directive; NREAP: National Renewable Energy Action Plans.

4.3.9 Impact assessment and interpretation of ILUC

The ILUC "process", as modelled herein, is composed of two sub-processes, namely "land expansion" and "land intensification". The relative share of these two sub-processes for the various impact categories considered in this study is presented in Figure 15. As highlighted in that figure, land expansion clearly dominates the 6 following impacts: global warming, particulate matter, photochemical ozone formation, acidification, terrestrial and marine eutrophication impacts, while land intensification dominate all the other impacts. The reason for the latter is essentially the use of the fertilisers processes (mineral fertiliser production).

The land expansion process is itself composed of two sub-processes, namely foregone sequestration and land clearing (step 4; peatland clearing was excluded by default). For the 6 impacts where land expansion dominates, land clearing represents more than 99 % of the expansion response (this graph is not presented herein). Also, it should be noted that foregone sequestration only includes one substance, namely CO_2, as shown in Table 27. Yet, this substance only affects one impact category: climate change (based on the impact assessment methods used in this study; Table 4).

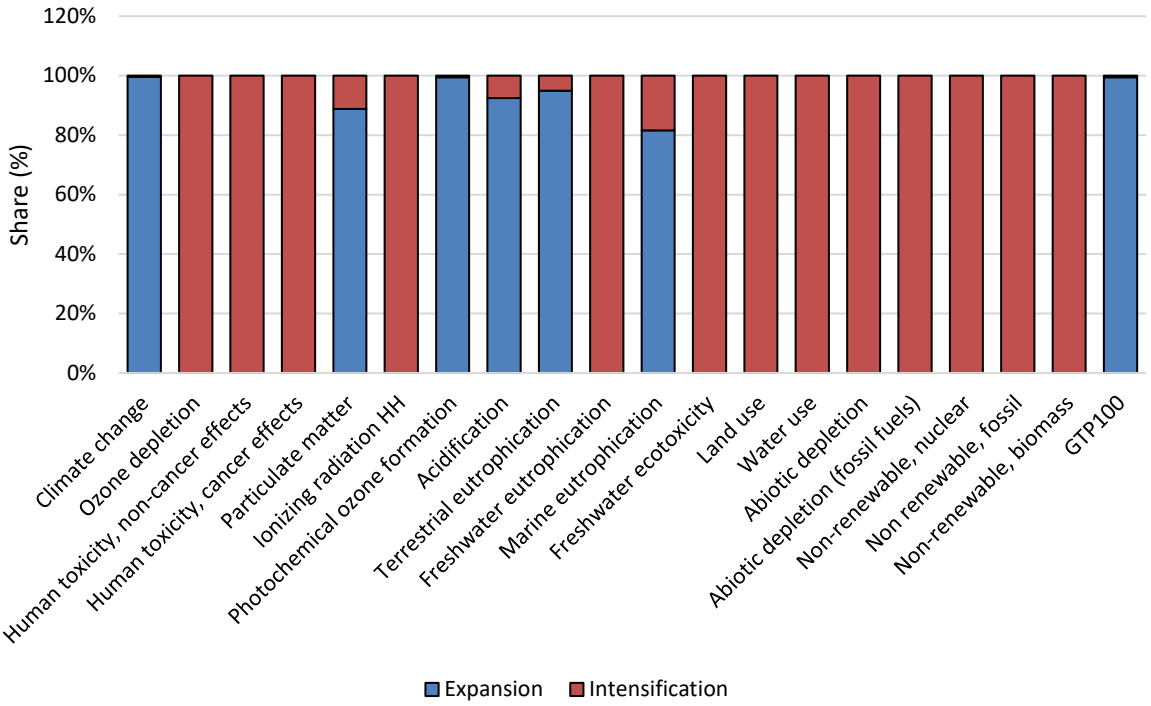


Figure 15. ILUC impact assessment: Share of the expansion and intensification reactions per impact.

Similarly, to Figure 15, Figure 16 presents the breakdown of the intensification process for all impact categories considered in this study. It reflects the importance of the production of additional marginal mineral phosphorus and nitrogen fertilisers to many of the impact categories. Field emissions represent the emissions occurring because of the (additional) fertiliser application (N_2O, NH_3, NOx, NO_3, P losses) to an agricultural field.

Figure 17 presents, for the ILUC process, the key contributing emissions, per impact categories. This allows, at the light of the inventory presented in Table 27, and the results of Figure 15 and Figure 16, to derive an understanding of the complete ILUC results modelled herein.

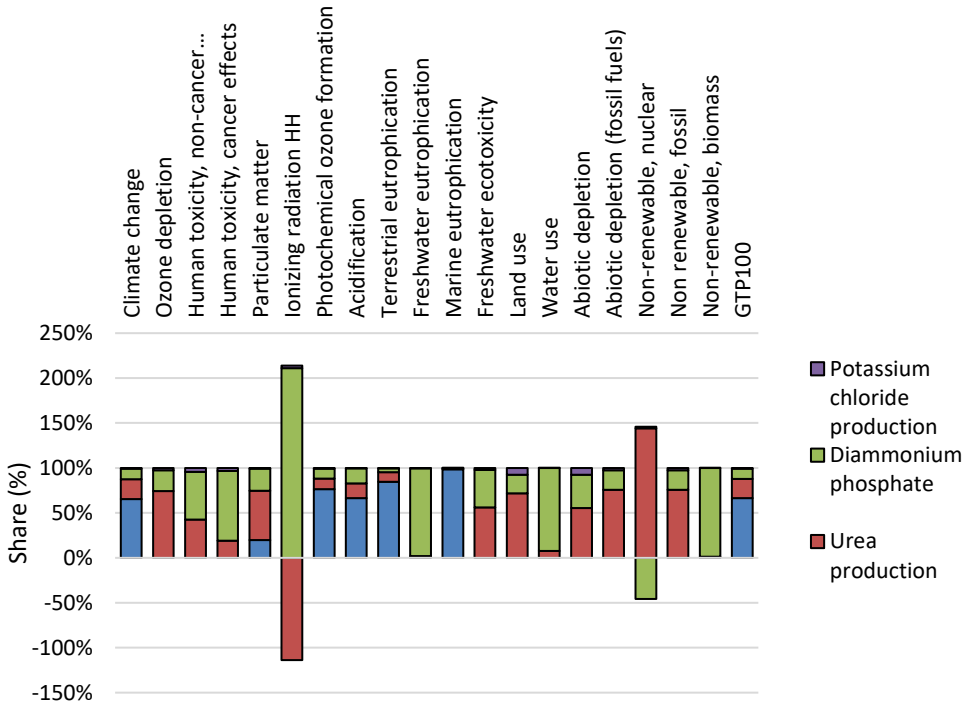


Figure 16. ILUC: Relative breakdown of the intensification effect per impact category.

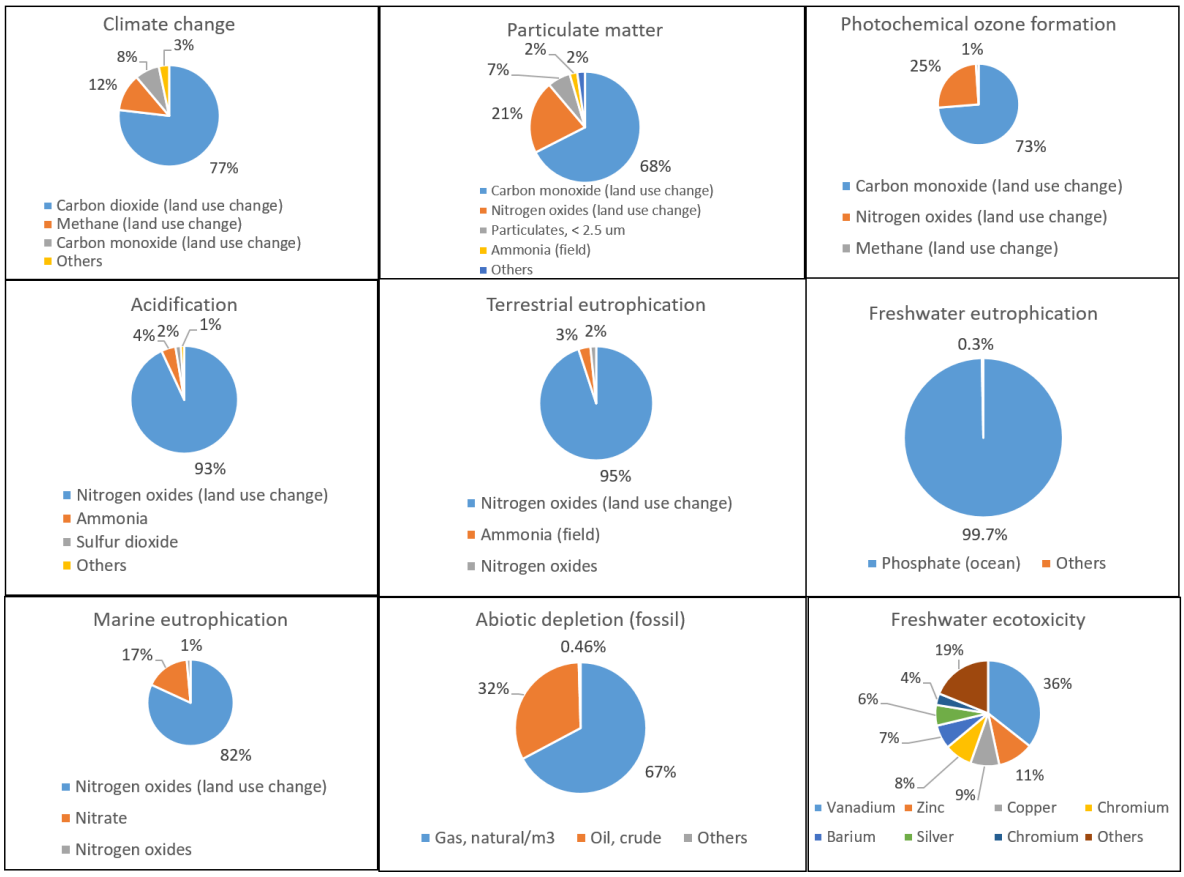


Figure 17. ILUC: Breakdown of the key emission flows contributing to selected impacts assessed in this study.

4.3.10 Limitations of the method

There are several limitations to the estimation method for the impacts of ILUC presented herein. First, it heavily relies on the IPCC guidelines for national GHG inventories (whether directly or through using the updated figures from EC 2010) for the estimation of carbon stocks and emission factors from deforestation in different biomes. The last version of this document (i.e. the one used in this study) was published in 2006. However, this guideline is currently (at the time of writing) being updated; a new methodology report titled "2019 refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories" is planned. The ILUC factor derived in this study could be updated accordingly (steps 4-5).

Similarly, deforestation data from 2000-2010 were used (Forest Resource Assessment report of FAO, 2010) at step 2, since it matches the time scope covered by EC (2013) (step 3). Yet, this Forest Resource Assessment is made every 5 years. Accordingly, updated data (2015) are already available, and even more updated data should be published by FAO in 2020. It is here suggested to update the "shares of deforestation than can be attributed to different demands" (parameter ξ of step 3) for the time scope 2010-2020, along with using the Forest Resource Assessment data 2010-2020, when available. This, of course, also applies for the intensification time scope.

One additional potential limitation relates with the use of an expansion share of 85 % of the ILUC reaction in step 1. This may be overly conservative. In the next section, an attempt was made to calculate a minimal ILUC factor considering that expansion is instead 25 % of the reaction, among others.

Finally, the way intensification was modelled implies increased fertilisers application as the sole factor behind the intensification observed in 2000-2010 (100 % input-driven; this input being entirely fertilisers). This likely overestimates the overall ILUC factor.

On the other hand, it should also be highlighted that soil emissions were not included; according to Edwards et al. (2010), this exclusion causes an 8-30 % underestimation of the overall ILUC emissions.

The way time was dealt with could also be pointed out as a limitation, and it is. Yet, here, the PEFCR Guidance v.6.3 was followed, and this recommends annualizing emissions (i.e. distribute them equally over time) over 20 years, or a single harvest period if it is longer. For example, the absolute global warming potential of gases at a given period (say 2030) could have been used as an alternative to represent the timing of emissions (beyond only land use changes) in this study.

To conclude, it should be remembered that ILUC factors typically only consider carbon emissions (and often only CO_2). This study goes slightly further by considering additional emissions (see steps 4 and 5), but still ignores many other effects of deforestation inducing a net warming (foregone cooling effect of volatile organic compounds released by forests, foregone [local] cooling effect of forest transpiration, drought increases from deforestation-induced changed precipitation patterns, etc.). Of course, if accounted for, these effects should also include the cooling effects induced by deforestation (e.g. through albedo change).

4.3.11 Low – High range for ILUC

At the light of the above section, an attempt was made to derive a minimal and maximal ILUC factor. The rationale is:

- Minimal: Expansion of 25 % and intensification of 75 %, of which 50 % is free of impacts (technology-driven intensification)

- Maximal: Current estimate + 8 %[16] to compensate for the underestimation caused by not including soil emissions

This results in a range of [1.22 – 5.20] t CO_2 eq. ha^{-1} y^{-1}.

4.4 Field emissions and crop production dataset

In this study, state-of-the-art life cycle inventory data were used to model the agricultural production of the feedstock of interest, such as Agri-Footprint v3.0 and Ecoinvent v3.4 inventory data (as further detailed in the case study section). These were cross-referenced with company data, when such data were provided. In Agri-Footprint, climate change due to land use is modelled separately via an emission flow titled "carbon dioxide, land transformation". That flow has been set to zero in the modelling herein (i.e. in all case studies using Agri-Footprint data), to ensure no double-counting would occur between the DLUC/ILUC modelled herein and the one considered in Agri-Footprint. Similarly, the flow "carbon dioxide, to soil or biomass stock" from Ecoinvent has been set to zero.

[16] According to Edwards et al. (2010), excluding soil emissions causes an 8-30 % underestimation of the overall ILUC emissions; here the lower end of this range is considered for the "maximal" ILUC emission.

5 Interpretation, dealing with uncertainty and sensitivity analysis

5.1 Structure of presenting the LCA results

In this study, the cradle-to-grave characterised results (16 impact categories and NREU) are presented in two life-cycle stages: *cradle-to-user* and *EoL*. For single-use applications, the environmental impacts of the use phase are negligible. The LCA results are presented for the two life-cycle stages separately first for the baseline bio-based product system, followed by the bio-based alternative system(s) (if applicable) and the petrochemical reference system(s). Then the comparisons of product systems in each case study are made for the scope cradle-to-grave. To summarise, the following order can be found in Section 3 "*Life cycle impact assessment results and interpretation*" of each case study report:

- Cradle-to-user results (baseline, alternatives and petrochemical references);
- EoL results (baseline, alternatives and petrochemical references);
- Cradle-to-grave results (baseline, alternatives and petrochemical references); and
- Comparison of cradle-to-grave characterised results for all product systems for a selection of impact categories.

In the last step, the selection of impact categories used to compare the characterised results are determined based on the recommendations made based on the review of the LCA/LCI results of petrochemical polymers (see Chapter 6 *Investigation of generic LCA data for fossil-based plastics*).

Following the 16 PEFCR impact categories and NREU, the results of global temperature change potential (GTP) and the effect of land use changes can be found in section 5 of each case study report "*Assessing non-PEF impact categories: considering ILUC and endpoint of GTP*".

5.2 Interpretation

The interpretation is carried out in five steps based on the goal and the research questions of the study: 1) interpretation of characterised results for 16 impact categories, 2) interpretation of normalised and weighted results to identify environmentally hotspots for the baseline, 3) identification of the most environmental friendly EOL option for the bio-based baseline systems, 4) sensitivity analyses to identify the most sensitive input data and assumptions and to check the robustness of the results, and 5) the interpretation of the effects of direct or indirect land use changes.

5.2.1 Breakdown of characterised results into key unit processes

When the LCA results are presented for each life cycle stage, a detailed breakdown is available to indicate the contribution of the most important unit processes of the product system analysed for the selected impact categories. These key unit processes can be found in the flow diagrams in the *life cycle inventory* sections of the case study reports. The selected impact categories are determined by the environmental hotspots identified using normalisation and weighting (see section below). Interpretation is then carried out into activity levels as well as substance levels to be able to explain at least 80 % of the impacts for that impact category.

5.2.2 Use of normalization and weighting to identify environmental hotspots

For the bio-based baseline and petrochemical reference systems, the characterised results are normalised and weighted to identify the most important environmental hotspots. The normalisation and weighting factors used by this study are summarised in Table 30. The per capita EU 27 normalisation factors (2010) are taken from ILCD 2015 (Benini et al., 2014) for

13 impact categories. The three exceptions are "Water use", "Abiotic depletion (fossil fuels)" and "Abiotic depletion". For these three categories, the ILCD 2015 normalisation factors (Benini et al., 2014) do not offer AWARE-model based water use normalisation factors (NFs) and do not distinguish between fossil fuels, minerals and metals. Instead, we adopted the normalisation factors of global per capita for 2010 for these three impact categories as provided by the PEFCR guidance v.6.3. We therefore prioritise to ensure the same impact assessment models are used for characterization and normalisation.

Two sets of weighting factors, with all 16 impact categories and with 13 impact categories (excluding the toxicity categories), are both applied as recommended by the PEFCR guidance v6.3. The weighting factors are listed in Table 30.

Table 30. Normalisation factors (NF) and weighting factors used by this study.

Impact category	Impact assessment Model	Unit	NFs per person EU 27 (2010) ILCD 2015 [a]	FINAL WFs with toxicity (PEFCR guidance v6.3)	FINAL WFs without toxicity (PEFCR guidance v6.3)
Climate change	IPCC, 2013	kg CO_2 eq.	9.22E+03	21.06	22.19
Ozone depletion	World Meteorological Organisation (WMO), 1999	kg CFC-11 eq.	2.16E-02	6.31	6.75
Human toxicity, cancer	USEtox (Rosenbaum et al., 2008)	CTUh	3.69E-05	2.13	-
Human toxicity, non-cancer	USEtox (Rosenbaum et al., 2008)	CTUh	5.33E-04	1.84	-
Particulate matter and respiratory inorganics	Rabl and Spadaro, 2004	kg PM2.5 eq.	3.80E+00	8.96	9.54
Ionising radiation	Frischknecht et al., 2000	kBq U-235 eq.	1.13E+03	5.01	5.37
Photochemical ozone formation	Van Zelm et al., 2008, as applied in ReCiPe, 2008	kg NMVOC eq.	3.17E+01	4.78	5.1
Acidification	Posch et al., 2008	mol H+ eq.	4.73E+01	6.2	6.64
Terrestrial eutrophication	Posch et al., 2008	mol N eq.	1.76E+02	3.71	3.91
Freshwater eutrophication	Struijs et al., 2009	kg P eq.	1.48E+00	2.8	2.95
Marine eutrophication	Struijs et al., 2009	kg N eq.	1.69E+01	2.96	3.12
Land use	Milà i Canals et al., 2007	kg C deficit	7.48E+04	7.94	8.42

Ecotoxicity freshwater	USEtox (Rosenbaum et al., 2008)	CTUe	8.74E+03	1.92	-
Water use	AWARE factors (Boulay et al., 2015)	m^3 water eq water (world)	1.15E+04 (global per capita) [a]	8.51	9.03
Abiotic depletion (fossil fuels)	ADP fossils (van Oers et al., 2002)	MJ	6.53E+04 (global per capita) [a]	8.32	8.92
Abiotic depletion (mineral and metals)	ADP ultimate reserve (van Oers et al., 2002)	kg Sb eq	5.79E-02 (global per capita)[a]	7.55	8.08
NFs are taken from ILCD 2015 (Benini et al. 2014) for per capital EU27; the exceptions are for Water use, Abiotic depletion (fossil fuels) and Abiotic depletion (mineral and metals). The NFs of these three categories are for global per capita for 2010, as recommended by PEFCR guidance 6.3. The ILCD 2015 NFs do not provide a NF of water use based on the AWARE factors and do not distinguish two abiotic depletion categories. Therefore, the NFs for these categories are not available for per capita EU27.					

For bio-based baseline product systems, the normalised and weighted results are calculated to identify the most important environmental impact categories. The categories which contribute to at least 80 % of the total environmental impacts are further interpreted into different life-cycle stages, activity levels and substances/elementary flows.

5.2.3 Use of normalised and weighted results to determine the preferences of EoL options

The normalisation factors and weighting factors provided in Table 30 are also used to determine the most preferable EoL waste management options for the baseline bio-based products from the environmental point of view. This step is carried out when the impacts of the EoL life-cycle stage are presented for each case study (see the relevant sub-section under section 3 of each case study report). Here, single scores are obtained for each EoL option for the baseline bio-based products.

5.2.4 Sensitivity analysis (can be found in section 4 of each case study report)

For the cradle-to-user phase, since primary data based on real supply chains are used in the inventory models, the assumptions are usually focused on energy consumption of end-product manufacturing or transportation distances where company-specific data are not available and generic data are used. In some cases, when there is a big range of weights on the market without any apparent additional functions,[17] the sensitivity analysis is also carried out to test the robustness of the choice of the functional unit.

For the EoL phase, due to a very large number of scenarios it is not possible to perform a full sensitivity analysis, e.g. varying the chosen parameters across all scenarios. Instead, the impacts from the "intended waste management technology" for each case study are further investigated in a sensitivity analysis: for each intended waste management technology one important parameter has been selected for the sensitivity analysis (see Table 31). This

[17] This could be caused by, for example, over engineering of a product due to the restriction of the processing machines (e.g. size and thickness of the moulds).

selection is based on experience from similar LCAs with respect to which parameters significantly affect the overall LCA results of the technology and what parameters are likely to change due to technical improvements in the plastic products as well as the waste management technologies.

For each bio-based product system the chosen parameter is varied to model a "worst case" and a "best case" scenario for the intended EoL technology.

Table 31. Parameters chosen for sensitivity analysis of impacts from the chosen technologies.

Product types	Intended waste management technology	Parameter, sensitivity analysis
Field application products	In-situ degradation	Degradation of carbon in the soil (depending on soil type)
Biodegradable PLA	Recycling[18] Industrial composting	Sorting/technology efficiency Degradation of biogenic carbon during treatment
Biodegradable starch plastic	Industrial composting	Degradation of biogenic carbon during treatment
Bio-based plastic (not biodegradable)	Recycling	Sorting/technology efficiency

For the two cases with additional food leftovers (Case study 4 on cutlery and Case study 6 on packaging films) the amount of food leftover is varied based on several studies and this is presented in the sensitivity analysis in the case study chapters.

In the selection of parameters for sensitivity analysis the focus has been on selecting parameters that can improve when either the product or the EoL treatment technology is improved. A further description of the selected parameters is presented in the case study chapters.

Furthermore, in a few additional case studies a specific sensitivity analysis has been performed, which is described in the case study chapters.

5.2.5 Results and interpretation of GTP and effects of land use changes

In section 5 of each case study report, GTP results and the effects of direct or indirect land use changes are presented.

The GTP results of the bio-based baseline products are then shown with the breakdown of key life cycle stages and unit processes. They are also compared with climate change results to gain insight into the differences between mid-point and end-point climate change impacts.

The results of land use changes, consisting of both direct and indirect LUCs, are presented for all bio-based product systems (to the extent they apply). This allows for comparison with the LCA results without LUCs. A detailed breakdown for all impact categories will be shown to

[18] Except for the PLA mix, where the number of additives make recycling of the material difficult.

illustrate the contribution of LUC to the entire life cycle. To provide a further breakdown, the impacts from ILUC resulting from expansion and agricultural intensification are interpreted for all case studies.

6 Investigation of generic LCA data for fossil-based plastics

As described in the above chapters, the purpose of this study is to perform LCAs on bio-based and fossil-based plastics – enabling comparisons of the two overall types of feedstocks for the same products. A further comparison of the impact categories recommended by ILCD has not been possible, because the publicly available data on fossil-based plastics do not comply with the ILCD requirements for all categories.

For these reasons, a specific and detailed assessment of the fossil-based data has been included in this study and is set out in this chapter.

This chapter contains three main sections: the first one discussing the ranges for environmental impacts from production of fossil-based plastics and the second depicting the identified ranges for each of the fossil-based plastic types included in this study.

In the third main section, the importance of selected upstream activities in production of fossil-based plastics is investigated and described.

6.1 Ranges for environmental impacts from production of fossil-based plastics

This sub-chapter aims at establishing ranges for environmental impacts caused by different producers of plastics in Europe. This is done by comparing the results from different data sources, using the same impact assessment methodology. The main limitations of this approach are described in the following paragraphs.

6.1.1 Databases used for the comparisons

The primary data source selected for the project is the LCAs published by PlasticsEurope, and the results are compared to LCAs published by thinkstep™ for the same plastic types in 2017. Except for the polystyrene eco-profile, the 2018 version of the GaBi databases from thinkstep™ do not include the eco-profiles from PlasticsEurope, the reason being that they do not fully comply with the ILCD requirements and thus also do not fulfil the overall requirements for GaBi data[19].

The eco-profile documentation from PlasticsEurope[20] consists for each type of plastic of a short LCA-report summarizing the main findings and an MS Excel-file with the full inventory of in- and outputs registered by the consultants.

It must be acknowledged that the datasets from thinkstep™ have a different scope (German production or consumption mix with import of 25-50 % of the total is compared to the European average) and makes use of the proprietary data in the GaBi database which is updated at regular intervals.

Except for polystyrene, the most recent datasets from PlasticsEurope have been established by combining information from representative European plastic manufacturers with the (proprietary) data found in the Institut für Energie- und Umweltforschung Heidelberg GmbH (IFEU) database. PlasticsEurope data for polystyrene were compiled by thinkstep™, formerly PE International.

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[19] Regarding distinction between salt water and fresh water emissions, land use, method for calculation of water consumption etc.

[20] Can be downloaded from www.PlasticsEurope.org.

The eco-profiles published by PlasticsEurope have generally been accepted as giving precise information on the performance of European refineries and plastic producers supplying a wide range of monomers and polymers. The eco-profiles are published with relatively long intervals (>5 years). A main update was made in 2011, using data collected in 2009, and this data age is not satisfactory for today's LCA, as outlined in the ILCD/PEF methodology papers from the EU Commission (European Commission, 2017). It is, however, acknowledged that data collection requires large efforts, and some of the eco-profiles published by PlasticsEurope therefore only concern an update of selected background processes or specific issues like water consumption.

As mentioned, the datasets from PlasticsEurope are established by different consultants from GaBi. It is worth noting that relevant consultancy companies like IFEU and thinkstep™ rely on (different) proprietary data in their respective databases to account for background processes like energy production, refinery processes, etc. It is not possible to investigate the basic information in these databases in detail, and important information on e.g. allocation principles is kept in very general terms. It is also worth noting that PlasticsEurope does not have specific requirements for the allocation performed when establishing an eco-profile, only that the approach should be stated clearly. In practice, the consultants try to avoid allocation as far as possible, but both physical (mass, exergy, etc.) and economic allocation has been applied in different datasets.

The ecoinvent databases are used by many LCA practitioners to calculate the impacts from a wide range of industrial processes and materials. The ecoinvent data for plastic production are based on production conditions in 1999-2001, probably the same sample as included in the PlasticsEurope data. The datasets are regularly updated with respect to background processes like fuel production, transport, etc., but the foreground processes have not been subjected to an update. As a result, the impacts calculated using the ILCD methodology in general are significantly different than for the updated datasets from PlasticsEurope calculated in SimaPro. The datasets from ecoinvent are therefore only considered very briefly in the present study through its use in parallel calculations for PET (see Case study 1 PET bottles section 2.1.4).

6.1.2 Impact assessment methodology

The impacts from both average European production and German production are primarily calculated using the ILCD methodology, recommended by the European Commission's Joint Research Centre, JRC, with the changes outlined below. The methodologies are described in some detail in ILCD (2011). The ILCD methodology for climate change has, however, been replaced by the "IPCC AR5, excl. biogenic carbon" method, and the water use is assessed using the AWARE model with a high characterization factor for unspecified water. Assessment of human toxicity and ecotoxicity is done by applying USEtox methods, and depletion of resources makes a distinction between mineral and fossil resources in the assessment using the 2016 impact assessment model from CML. Finally, an assessment of cumulative energy demand is made for non-renewable energy use.

It is noted that the datasets from PlasticsEurope in some cases do not comply with the requirements defined in the ILCD methodologies. One of the drawbacks of most of the PlasticsEurope datasets (polystyrene is an exception) is that they do not make a distinction between emissions to fresh water and to seawater. This makes it difficult or even impossible to calculate the impacts on human health and ecosystems as well as marine and fresh water eutrophication, because the receiving compartment is an important element in determining the fate and effect of an emission.

The next sections describe the high-level differences that can be observed between plastic production in European countries and the differences that occur as a function of refinery complexity in a single country (the United Kingdom, (UK)), where a broad range of refineries were investigated with respect to their carbon efficiency. Then follows an examination of the

differences that are observed when comparing the results calculated from different datasets. For each of the five commodity plastics a plausible range for the environmental impacts is suggested, where possible.

6.1.3 Differences between European countries

Another important question when using average data from European manufacturers is how large the difference is between countries and between single producers in each country. In the first eco-profiles published by PWMI in the early nineties, Ian Boustead describes the ranges applicable to the energy consumption in production of some commodity plastics (PWMI, 1993a,b). Since the emissions from production and consumption of different fuels, together with fugitive emissions, are the major sources of most conventional environmental impacts, the difference in energy consumption is believed to be a good indication also of the ranges of such impacts.

In the 1993 reports for olefin feedstock sources, polyethylene and polypropylene, the following averages and ranges were reported by Ian Boustead:

- Ethylene (all producers): Average 69.47 MJ/kg; range 51-90 MJ/kg
- Ethylene (crackers feeding European pipeline): Average 73.25; range 55-86 MJ/kg
- Polyethylene (all grades): Average 88.83 MJ/kg; range 69-107 MJ/kg
- LDPE: Average 88.55 MJ/kg; range 73-107 MJ/kg
- LLDPE: Average 82.94 MJ/kg; range 74-104 MJ/kg
- PP: Average 80.03 MJ/kg; range 61-104 MJ/kg

The ranges of energy consumption for the monomers and polymers was thus about -/+ 25 %. Ian Boustead points to various operations being carried out in different countries with different amounts of transport, different technologies and different fuel and feedstock mixes as the main reasons for the variations. Additionally, the first methodology report from the Plastic Waste Management Institute (PWMI) points to differences in electricity production efficiency between European countries as a possible source of variation (PWMI, 1992).

Impacts caused by upstream activities, primarily extraction of oil and gas, are addressed in detail in section 6.3. Here it is only mentioned that there are large differences between supplying countries, and that European countries at the same time have different import profiles for oil and gas supplies.

6.1.4 Differences between refineries in the UK

It is assessed that all the causes for the differences between refineries are still relevant in 2018. This judgement is supported by the findings in a master thesis from the Norwegian University of Science and Technology study (NTNU)-study, examining the impacts from refinery processes in nine UK refineries (OBorn, 2012). The study aims to show differences between refineries in great detail, taking into consideration both fuel delivery technologies, plant efficiencies and differences in output of marketable products. The basic figure is the simple calculation of emissions of CO_2 eq. per kg refined fuel. Here, the study shows a variation between best and worst performer of 190 g and 319 g CO_2 eq. per kg fuel. When the

complexity of the refineries was included (by dividing with the so-called Nelson complexity index[21]), the difference was even larger, 18 g vs. 43 g CO_2 eq. per Nelson number.

Combining the findings in the PWMI and NTNU studies it can be concluded that there are large variations both between countries (~ 25 %) and between single refineries in each country (~ 50 %).

6.2 Variations and ranges of environmental impacts from production of selected plastics

In this paragraph, the variations and ranges of environmental impacts from production of selected plastics are described. It includes polystyrene, polypropylene, low density polyethylene and polyethylene terephthalate.

6.2.1. Variations and ranges of environmental impacts from production of polystyrene

The datasets for polystyrene production are well suited for establishing valid ranges because they are available with the same high level of detail in the GaBi database. The datasets are the most recent (2011) data from PlasticsEurope and the newer (2017) GaBi dataset for German production mix. In other words, both datasets give the possibility of examining the contribution from single substances, although with differences pertaining to the requirements outlined in the PlasticsEurope eco-profile methodology document and to the general modelling principles described for the GaBi databases. It is noted in this context that it is thinkstep™ that has established both datasets and the impact assessment, and the comparison between results is therefore only affected by differences in the age and scope[22] of the two datasets. A major difference in this respect is that the European average data addresses two routes for production of monomer styrene (EBSM (ethylbenzene styrene monomer) and POSM (propylene oxide styrene monomer), while the information in the dataset for German production only shows EBSM as a production route. The importance of this cannot be assessed based on the available information.

The impacts from average European production and average German production are shown in Table 32.

[21] The Nelson complexity index (NCI) is a measure to compare the secondary conversion capacity of a petroleum refinery with the primary distillation capacity.[1] The index provides an easy metric for quantifying and ranking the complexity of various refineries and units.

[22] Europe average vs. German production mix.

Table 32. Life cycle impact assessment of 1 kg polystyrene granulate production, kg for average European production (2011) and average German production (2017).

Impact category	Unit	PS PlasticsEurope 2011	GaBi 2017 German production mix
Climate change	kg CO_2 eq.	2.28E+00	1.99E+00
Ozone depletion	kg CFC-11 eq	1.63E-08	3.51E-13
Human toxicity, non-cancer effects	CTUh	8.65E-08	2.20E-08
Human toxicity, cancer effects	CTUh	5.29E-09	2.05E-08
Particulate matter	kg PM2.5 eq	2.70E-04	1.45E-04
Ionizing radiation, human health	kBq U235 eq	4.25E-02	3.55E-02
Photochemical ozone formation	kg NMVOC eq	4.78E-03	3.43E-03
Acidification	molc H+ eq	6.48E-03	3.68E-03
Terrestrial eutrophication	molc N eq	1.40E-02	1.02E-02
Freshwater eutrophication	kg P eq	3.91E-06	3.83E-06
Marine eutrophication	kg N eq	1.30E-03	9.69E-04
Freshwater ecotoxicity	CTUe	1.52E-01	4.73E-01
Land use	kg C deficit	4.01E-02	1.40E-01
Water use	m3	6.27E-01	5.89E-01
Abiotic depletion	kg Sb eq	9.26E-07	4.38E-07
Abiotic depletion (fossil fuels)	MJ	7.47E+01	7.19E+01
Non-renewable energy use. (NREU)	MJ	7.60E+01	7.26E+01

The following paragraphs examine the differences between the two datasets and concludes the range to be included in comparisons of single impact categories.

6.2.1.1 Climate change

The climate change impact potential is very similar for the two datasets. As shown earlier, however, the variation between single producers may very well be higher. A range of +/- 10 % relative to the European average is suggested.

6.2.1.2 Ozone depletion potential (ODP)

The ODP differs by more than four orders of magnitude between the two datasets. In both datasets, R114 is the main contributor, so the difference is only related to the emitted amounts. For ozone-depleting substances, the emissions are probably calculated from site-specific consumption patterns, where backfilling of a cooling tower with e.g. R114 is assumed to be related to the production in the same year. In subsequent years, there may not be a need to backfill, and the contribution to ozone depletion in such years is therefore expected to be zero. It is noted in this context that according to the Montreal Protocol, R 114 must not be produced or consumed since 1996, and it is therefore surprising that the substance is found in both datasets. Taking this into consideration, it is assumed that the most recent inventory from GaBi is also the most accurate for today's technologies, as it includes a broader range of cooling agents. However, it is concluded on the general level that the datasets are not of sufficient quality to establish a valid range for the ODP.

6.2.1.3 Human toxicity; non-cancer effects

The impacts calculated from the two datasets are very similar, with a variation of about 15 %. A closer look at the main contributions, however reveal that they are very different. For the European average, 60 % of the impacts are related to airborne emissions of mercury, while the emission only accounts for 10 % in the German production mix. In the German dataset, emissions of arsenic (V) to freshwater accounts for about 90 % of the impacts, while it in the European dataset only contributes about 10 %. It is not possible to pinpoint the processes from which the two emissions emerge, but it seems unlikely that the differences should occur as a result of allocation procedures or technological differences between average European and average German production. Given the inherent uncertainties in calculation of toxicity impacts, a valid range should be as wide as -/+ 1,000 % (three orders of magnitude) between individual producers. The large variation makes it difficult to use the impact category as a meaningful indicator in a comparison with other products.

6.2.1.4 Human toxicity; cancer effects

The contribution to human cancer effects are four times higher for the German production average than for the European average. In contrast to the non-cancer effects, the main contributors are the same, although the relative contributions differ. For the German production, the main contribution (92 %) comes from emission of chromium to fresh water, while the emission of chromium contributes 67 % to the European average. The second most important contribution to the European average comes from emission of chromium to seawater, which is of little importance (< 2 %) for the German production. It is judged that the factor 4 difference between the two datasets may well reflect that a large part of the European producers emits their wastewater to seawater, compared to the German producers that are situated in areas where wastewater is emitted to fresh water. It is noted in this context that chromium has a factor 15 higher characterization factor when emitted to freshwater than when emitted to seawater. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.1.5 Particulate matter

The contribution from the European average is about double that from the German production average. It is the same substances, sulphur dioxide and $PM_{2.5}$, that are the main contributors

with 65 % and 22 % respectively. The factor 2 difference may therefore very well reflect the degree of flue gas cleaning, combined with differences in the fuel types used for energy production. A range of -/+ 50 % is suggested for this impact category.

6.2.1.6 Ionizing radiation (human health)

The ionizing radiation from average European production is more than 10 times higher than from average German production. In all datasets the contribution is related to emissions of carbon (C14). It is not possible to track the emissions further, and it is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.1.7 Photochemical ozone creation potential (POCP)

The POCP of European and German average production differs by about 25 %. The main sources are the same for both datasets (nitrogen oxides and NMVOC), and a range of -/+ 25 % is suggested for this impact category.

6.2.1.8 Acidification (AP)

The AP of German production is about 60 % of the European average. Sulphur oxide and nitrogen oxides are the main contributors in both averages and given the small difference in energy consumption (calculated, but now shown here), the smaller value for German production is most probably caused by more efficient flue-gas cleaning. A range of -/+ 40 % is suggested.

6.2.1.9 Eutrophication potentials

The eutrophication potentials (terrestrial, marine and freshwater) all vary by less than 25 % from the European average. The main sources are identical, and a range of -/+ 25 % is suggested for all three impact categories.

6.2.1.10 Freshwater ecotoxicity

The impact from German production is three times higher than the European average. The main contribution (40 %) from the German production is emission of (unspecified) chromium to fresh water, while the main contribution to the European average is distributed on a wide range of metal emissions to fresh water. It is noted that emissions of vanadium to air plays a visible role (> 10 %) for the European average. The differences in inventory details indicates that the variation between single producers may be large, and that the large variation makes it difficult to use the impact category as a meaningful indicator in a comparison with other materials. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.1.11 Land use

The results for land use are not directly comparable with the PlasticsEurope dataset as these data were compiled at a time when the current method for assessment of land use was not available. In addition, the two inventories therefore have different levels of detail: the European average only concerns one factor, i.e. arable land, while the German average includes a wide range of national and regional elements that are relevant for calculation of land use impacts according to ILCD requirements. The methodological differences make it impossible to establish a valid range. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.1.12 Water resource depletion

The general uncertainty in the AWARE model has not been investigated, but a range of +/- 50 % is assumed to be valid.

6.2.1.13 Abiotic depletion (minerals)

The abiotic resource depletion for the German production is about 50 % of that for the European average. Lead and sodium chloride (rock salt) are the main contributors in both scenarios, and a range of +/- 50 % is suggested.

6.2.1.14 Abiotic depletion (fossil fuels)

The two values are very similar, but a conservative range of +/- 25 % is suggested.

6.2.1.15 Non-renewable energy use

The two values are very similar, but a conservative range of +/- 25 % is suggested.

6.2.1.16 Ranges for the polystyrene LCA data to be applied in the comparative study

Table 33. Applied ranges for the polystyrene LCA data.

Impact category	Unit	Value	Suggestion for ranges relative to European average
Climate change	kg CO_2 eq.	2.28E+00	+/- 10 %
Ozone depletion	kg CFC-11 eq.	Not applicable	Not applicable
Human toxicity, non-cancer effects	CTUh	Not applicable	Not applicable
Human toxicity, cancer effects	CTUh	Not applicable	Not applicable
Particulate matter	kg PM2.5 eq.	2.70E-04	+/- 50 %
Ionizing radiation, human health	kBq U235 eq.	Not applicable	Not applicable
Photochemical ozone formation	kg NMVOC eq.	4.78E-03	+/- 25 %
Acidification	molc H+ eq.	Not applicable	Not applicable
Terrestrial eutrophication	molc N eq.	1.40E-02	+/- 25 %
Freshwater eutrophication	kg P eq.	3.91E-06	+/- 25 %
Marine eutrophication	kg N eq.	1.30E-03	+/- 25 %
Freshwater ecotoxicity	CTUe	Not applicable	Not applicable
Land use	kg C deficit	Not applicable	Not applicable
Water use	m^3	6.27E-01	+/-50 %
Abiotic depletion	kg Sb eq.	Not applicable	Not applicable
Abiotic depletion (fossil fuels)	MJ	7.47E+01	+/- 25 %
Non-renewable energy use. (NREU)	MJ	7.60E+01	+/- 25 %

These values and ranges provided in Table 33 for 1 kg PS granulate production are used for making comparison with bio-based plastics in Case study 4 cutlery.

6.2.2 Variation and ranges in polypropylene production

The variation and ranges in polypropylene production are developed using the most recent (reference year 2011) European average established by IFEU and published by PlasticsEurope to the 2017 data for German consumption mix published by thinkstep™ in the GaBi database. It is noted in this context that both IFEU (the consultant for PlasticsEurope) and thinkstep™ develop and maintain their own proprietary databases, and that these very well may use different approaches and background data on specific elements. One example, clearly stated in the PlasticsEurope methodology report, is that the IFEU data for production of monomer propylene includes a proprietary model for fluid catalytic cracking (used for about 23 % of the total monomer production), while the GaBi data only concerns steam cracking.

The development of ranges has been done by identifying the most important contributions in the German consumption mix and then crosschecked with information in the MS Excel-spreadsheet provided by PlasticsEurope together with the eco-profile report[23]. The two impact assessment profiles are shown in Table 34.

[23] Can be downloaded from PlasticsEurope.org.

Table 34. Life cycle impact assessment of 1 kg polypropylene granulate production, showing average European production (2011) and average German consumption mix (2017).

Impact category	Unit	PP Industry data 2.0 (Plastics Europe)	GaBi 2017 German production mix
Climate change	kg CO_2 eq.	1.65E+00	1.53E+00
Ozone depletion	kg CFC-11 eq.	3.54E-08	7.46E-13
Human toxicity, non-cancer effects	CTUh	5.52E-08	1.99E-08
Human toxicity, cancer effects	CTUh	6.44E-09	6.35E-08
Particulate matter	kg PM2.5 eq.	3.34E-04	1.22E-04
Ionizing radiation, human health	kBq U235 eq.	5.54E-02	7.96E-02
Photochemical ozone formation	kg NMVOC eq.	5.46E-03	3.02E-03
Acidification	molc H+ eq.	5.12E-03	3.02E-03
Terrestrial eutrophication	molc N eq.	1.24E-02	7.97E-03
Freshwater eutrophication	kg P eq.	5.19E-05	2.60E-06
Marine eutrophication	kg N eq.	1.15E-03	7.76E-04
Freshwater ecotoxicity	CTUe	8.15E-01	4.34E-01
Land use	kg C deficit	8.73E-02	2.09E-01
Water use	m^3	1.46E+00	3.57E-01
Abiotic depletion	kg Sb eq.	1.01E-07	4.45E-07
Abiotic depletion (fossil fuels)	MJ	7.36E+01	6.60E+01
Non-renewable energy use. (NREU)	MJ	7.54E+01	6.75E+01

The following paragraphs examine the differences between the two datasets and concludes the range to be included in comparisons of single impact categories.

6.2.2.1 Climate change

The values for the two scenarios are very similar, although it could be expected that including fluid catalytic cracking in the European average scenario would cause a decrease in climate change compared to other technologies. However, the European average data are older, and this may counterbalance the differences in technology. A range of +/- 25 % is suggested as a conservative range.

6.2.2.2 Ozone depletion potential

The impacts from German PP consumption mix are only a small fraction of the average European impact, where "halogenated hydrocarbons" accounts for 100 % of the impacts, without further detail. It is not possible to assess whether this type of emission is relevant in today's production, but the production of such substances has been discontinued for more than a decade. Taking this into consideration, it is assumed that the most recent inventory from GaBi is the most accurate for todays' technologies, as it includes a broader range of cooling agents. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.2.3 Human toxicity; non-cancer effects

Similar considerations as for polystyrene. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.2.4 Human toxicity; cancer effects

Similar considerations as for polystyrene. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.2.5 Particulate matter/Respiratory inorganics

Similar considerations as for polystyrene, although the difference for polypropylene is a little higher. A range of +/- 60 % is suggested for comparisons with bio-based plastics.

6.2.2.6 Ionizing radiation (human health)

The ionizing radiation is in both datasets related to emissions of carbon (C14), but it is not possible to identify the contributing processes in more detail. This lack of information and knowledge makes it difficult to use the impact category as a meaningful indicator. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.2.7 Photochemical ozone creation potential (POCP)

The POCP of European and German average production differs by approximately 50 %. The main sources are the same for both datasets (nitrogen oxides and NMVOC), but nitrogen oxides appear to be more important for the European average; the emissions are simply larger. A range of -/+ 50 % is suggested for comparisons with bio-based plastics.

6.2.2.8 Acidification (AP)

The AP of German production is about 60 % of the European average. Sulphur oxide and nitrogen oxides are the main contributors in both averages, and the smaller value for German production is most probably caused by less sulphur in the fuels used and/or a more efficient flue-gas cleaning. A range of -/+ 40 % is suggested for comparisons with bio-based plastics.

6.2.2.9 Freshwater eutrophication

The impacts from the German production mix are only 5 % of those from average European production. The difference is related to emissions of phosphate, but the originating process cannot be identified. It is noted here that the databases used for background processes are different in the two scenarios, and it cannot be ruled out that this is of importance. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.2.10 Terrestrial eutrophication

The impacts from the German production mix are about 65 % of the average European production. The difference is primarily due to a lower emission of nitrogen oxides/nitrogen dioxide. A range of -/+ 50 % is suggested for comparisons with bio-based plastics.

6.2.2.11 Marine eutrophication

See terrestrial eutrophication, above. It is noted that emissions of nitrate play a minor role for the German production mix. It has not been possible to investigate the importance of nitrate in the calculation of the European average. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.2.12 Freshwater ecotoxicity

The impacts from the German average are about 50 % of that of the European average. The contribution to the German average – and the absolute value – is the same as for polystyrene production, i.e. that emission of chromium is the largest contributor. It is not possible to examine the inventory for the European average in the same detail, and therefore it is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.2.13 Land use

Similar considerations as for polystyrene. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.2.14 Water resource depletion

The water consumption in the European average scenario is about four times higher than for the German production mix. It has not been possible to examine the impact category in any detail due to lack of data, it is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.2.15 Abiotic depletion (minerals)

The depletion of abiotic minerals is four times higher for the German scenario than for the European average. The inventories appear to be significantly different, but it is not possible to examine the contributions to the European average impact in any detail due to lack of data. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.2.16 Abiotic depletion (fossil fuels)

The difference of 10-15 % is assumed to give a representative picture of the actual range, but a conservative estimate of +/- 25 % is suggested based on previous assessments of ranges.

6.2.2.17 Non-renewable energy use

The difference of 10-15 % is assumed to give a representative picture of the actual range, but a conservative estimate of +/- 25 % is suggested based on previous assessments of ranges.

6.2.2.18 Ranges for the polypropylene LCA data to be applied in the comparative study

Table 35. Applied ranges for the polypropylene LCA data.

Impact category	Unit	Value	Suggestion for ranges relative to European average
Climate change	kg CO_2 eq.	1.65	+/- 25 %
Ozone depletion	kg CFC-11 eq.	Not applicable	Not applicable
Human toxicity, non-cancer effects	CTUh	Not applicable	Not applicable
Human toxicity, cancer effects	CTUh	Not applicable	Not applicable
Particulate matter	kg PM2.5 eq.	3.34E-04	+/- 60 %
Ionizing radiation, human health	kBq U235 eq.	Not applicable	Not applicable
Photochemical ozone formation	kg NMVOC eq.	5.46E-03	+/- 50 %
Acidification	molc H+ eq.	5.12E-03	+/- 40 %
Terrestrial eutrophication	molc N eq.	1.24E-02	+/- 50 %
Freshwater eutrophication	kg P eq.	Not applicable	Not applicable
Marine eutrophication	kg N eq.	Not applicable	Not applicable
Freshwater ecotoxicity	CTUe	Not applicable	Not applicable
Land use	kg C deficit	Not applicable	Not applicable
Water use	m^3	Not applicable	Not applicable
Abiotic depletion	kg Sb eq.	Not applicable	Not applicable
Abiotic depletion (fossil fuels)	MJ	7.36E+01	+/- 25 %
Non-renewable energy use. (NREU)	MJ	7.54E+01	+/- 25 %

These values and ranges provided in Table 35 for 1 kg PP granulate production are used for making comparison with bio-based plastics in Case study 2 clips and Case study 6 food packaging films.

6.2.3 Variation and ranges in LDPE production

The variation and ranges in data for production of low density polyethylene (LDPE) are compared using the most recent (reference year 2011) European average established by IFEU and published by PlasticsEurope (PlasticsEurope, 2016b) to the 2017 data for German production mix published by thinkstep™ in the GaBi database. It is noted in this context that both IFEU (the consultant for PlasticsEurope) and thinkstep™ develop and maintain their own proprietary databases, and that these very well may use different approaches and background data on specific elements. However, both consultancies describe the processes as generic, but with specific process parameters being regarded as confidential.

The development of ranges was done by identifying the most important contributions in the German consumption mix and then crosschecked with information in the MS Excel-spreadsheet provided by PlasticsEurope together with the eco-profile report[24]. The result of this comparison can be seen in Table 36.

[24] Downloadable from https://www.plasticseurope.org/en/resources/eco-profiles.

Table 36. Life cycle impact assessment of LDPE production, per kg LDPE for average European production (2011) and average German production (2017).

Impact category	Unit	LDPE Industry data 2.0 (PlasticsEurope)	GaBi 2017 German production mix
Climate change	kg CO_2 eq.	1.90E+00	1.68E+00
Ozone depletion	kg CFC-11 eq.	2.50E-07	8.53E-13
Human toxicity, non-cancer effects	CTUh	6.57E-08	1.90E-08
Human toxicity, cancer effects	CTUh	6.80E-09	5.11E-08
Particulate matter	kg PM2.5 eq.	3.24E-04	1.29E-04
Ionizing radiation, human health	kBq U235 eq.	1.08E-01	7.61E-02
Photochemical ozone formation	kg NMVOC eq.	8.15E-03	2.50E-03
Acidification	molc H+ eq.	5.13E-03	3.10E-03
Terrestrial eutrophication	molc N eq.	1.34E-02	8.15E-03
Freshwater eutrophication	kg P eq.	6.70E-05	3.92E-06
Marine eutrophication	kg N eq.	1.24E-03	8.15E-04
Freshwater ecotoxicity	CTUe	1.04E+00	4.13E-01
Land use	kg C deficit	1.51E-01	4.25E-01
Water use	m^3	2.72E+00	3.45E-01
Abiotic depletion	kg Sb eq.	2.87E-08	6.52E-07
Abiotic depletion (fossil fuels)	MJ	7.62E+01	6.79E+01
Non-renewable energy use (NREU)	MJ	7.93E+01	6.98E+01

The following paragraphs examine the differences between the two datasets and concludes the range to be included in comparisons of single impact categories.

6.2.3.1 Climate change

The climate change is very similar for the two datasets. As shown earlier the variation between single producers may very well be higher, and a conservative range of +/- 25 % is suggested.

6.2.3.2 Ozone depletion potential

Same considerations as for polystyrene. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.3.3 Human toxicity; non-cancer effects

Same considerations as for polystyrene. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.3.4 Human toxicity; cancer effects

The contribution to human cancer effects are eight times higher for the German production mix than for the European average. The main contribution to the German average comes from emission of (unspecified) chromium to fresh water, but it is not possible to directly compare this to the emissions of chromium ion reported for the European average, which is a factor 100 lower. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.3.5 Particulate matter/Respiratory inorganics

Similar considerations as for polystyrene. A range of -/+ 60 % is suggested for comparisons with bio-based plastics.

6.2.3.6 Ionizing radiation (human health)

Same considerations as for polypropylene. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.3.7 Photochemical ozone creation potential (POCP)

The POCP of European and German average production differs with a factor 3. The main sources are the same for both datasets (nitrogen oxides and NMVOC), but nitrogen oxides appear to be more important for the European average; the emissions are simply larger. At the same time, the relative importance of the emissions of NMVOC is higher in German production, although the absolute value is lower.

6.2.3.8 Acidification (AP)

The AP of German production is about 60 % of the European average. Sulphur oxide and nitrogen oxides are the main contributors in both averages, and the smaller value for German production is most probably caused by less sulphur in the fuels used and/or a more efficient flue-gas cleaning. A range of -/+ 60 % is suggested for comparisons with bio-based plastics.

6.2.3.9 Freshwater eutrophication

Similar considerations as for polypropylene. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.3.10 Terrestrial eutrophication

The impacts from the German production mix are about 60 % of the average European production. The difference is primarily due to a lower emission of nitrogen oxides/nitrogen dioxide. A range of -/+ 50 % is suggested for comparisons with bio-based plastics.

6.2.3.11 Marine eutrophication

The impacts from the German production mix are about 60 % of the average European production. The difference is primarily due to a lower emission of nitrogen oxides/nitrogen dioxide. A range of -/+ 50 % is suggested for comparisons with bio-based plastics.

6.2.3.12 Freshwater ecotoxicity

Similar considerations as for polypropylene. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.3.13 Land use

Similar considerations as for polystyrene. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.3.14 Water resource depletion

The water consumption is ten times higher for the European average. It has not been possible to investigate the category in any detail, and it is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.3.15 Abiotic depletion (minerals)

The depletion of abiotic minerals is more than 20 times higher for the German production mix. There is no readily available explanation for this difference and it is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.3.16 Abiotic depletion (fossil fuels)

The two values are very similar, but a conservative range of +/- 25 % is suggested.

6.2.3.17 Non-renewable energy use

The two values are very similar, but a conservative range of +/- 25 % is suggested.

6.2.3.18 Ranges for the LDPE LCA data to be applied in the comparative study.

Table 37. Applied ranges for the LDPE LCA data.

Impact category	Unit	Value	Suggestion for ranges relative to European average
Climate change	kg CO_2 eq.	1.90E+00	+/- 25 %
Ozone depletion	kg CFC-11 eq.	Not applicable	Not applicable
Human toxicity, non-cancer effects	CTUh	Not applicable	Not applicable
Human toxicity, cancer effects	CTUh	Not applicable	Not applicable
Particulate matter	kg PM2.5 eq.	3.24E-04	+/- 60 %
Ionizing radiation, human health	kBq U235 eq.	Not applicable	Not applicable
Photochemical ozone formation	kg NMVOC eq.	8.15E-03	+/- 75 %
Acidification	molc H+ eq.	5.13E-03	+/- 60 %
Terrestrial eutrophication	molc N eq.	1.34E-02	+/- 50 %
Freshwater eutrophication	kg P eq.	Not applicable	Not applicable
Marine eutrophication	kg N eq.	1.24E-03	+/- 50 %
Freshwater ecotoxicity	CTUe	Not applicable	Not applicable
Land use	kg C deficit	Not applicable	Not applicable
Water use	m^3	Not applicable	Not applicable
Abiotic depletion	kg Sb eq.	Not applicable	Not applicable
Abiotic depletion (fossil fuels)	MJ	7.62E+01	+/- 25 %
Non-renewable energy use. (NREU)	MJ	7.93E+01	+/- 25 %

These values and ranges provided in Table 37 for 1 kg LDPE granulate production are used for making comparison with bio-based plastics in Case study 5 mulching films and Case study 7 carrier bags.

6.2.4 Variation and ranges in PET production

The variation and ranges in production of PET (polyethylene terephthalate) are compared using the most recent (reference year 2015) European average eco-profile established by IFEU and published by PlasticsEurope (PlasticsEurope, 2017) to the 2017 data for German consumption mix published by thinkstep™ in the GaBi database and the compiled PET bottle grade data by the project team (see section for Case study 1: Beverage bottles). It is noted in this context that both IFEU (the consultant for PlasticsEurope) and thinkstep™ develop and maintain their own proprietary databases, and that these very well may use different approaches and background data on specific elements. It is further noted in this context that earlier versions of the eco-profile for PET were developed using a third database, established by Ian Boustead.

Although the eco-profile documentation was published in 2017, it does not conform with ILCD requirements in its presentation of inventory details or impact assessment results.

The MS Excel-file from PlasticsEurope[25] with the inventory does not distinguish between emissions to fresh water and to seawater, and it is therefore not possible to perform an impact assessment that is in conformity with ILCD requirements. This fact alone makes it difficult to suggest valid ranges for environmental impacts, but also other issues are problematic as is evident from the below discussion of the individual impact categories.

The three impact assessment profiles are shown in Table 38.

[25] Can be downloaded from PlasticsEurope.org.

Table 38. Life cycle impact assessment of 1 kg bottle grade PET granulate production, showing average European production (2017) and average German consumption mix (2017).

Impact category	Unit	PET Industry data 2.0 (PlasticsEurope)	Compiled PET bottle grade (see Case study 1, section 2)	GaBi 2017 German production mix
Climate change	kg CO_2 eq.	2.20E+00	2.19E+00	2.67E+00
Ozone depletion	kg CFC-11 eq.	1.07E-05	1.00E-05	6.49E-13
Human toxicity, non-cancer effects	CTUh	1.43E-07	2.95E-07	1.97E-08
Human toxicity, cancer effects	CTUh	5.28E-09	6.64E-08	5.85E-08
Particulate matter	kg PM2.5 eq.	4.19E-04	1.08E-03	1.49E-04
Ionizing radiation HH	kBq U235 eq.	9.58E-01	1.82E-01	5.90E-02
Photochemical ozone formation	kg NMVOC eq.	7.86E-03	7.36E-03	3.59E-03
Acidification	molc H+ eq.	9.55E-03	1.13E-02	3.84E-03
Terrestrial eutrophication	molc N eq.	1.83E-02	1.96E-02	1.14E-02
Freshwater eutrophication	kg P eq.	5.98E-05	4.91E-04	5.28E-06
Marine eutrophication	kg N eq.	1.72E-03	1.91E-03	1.08E-03
Freshwater ecotoxicity	CTUe	2.75E+00	7.97E+00	4.16E-01
Land use	kg C deficit	1.50E-01	1.13E+00	3.22E-01
Water use	m^3	2.40E+00	3.06E+00	1.10E+00
Abiotic depletion	kg Sb eq.	4.83E-08	2.96E-06	7.53E-07
Abiotic depletion (fossil fuels)	MJ	6.35E+01	6.80E+01	7.42E+01
Non-renewable energy use. (NREU)	MJ	6.73E+01	6.96E+01	7.57E+01

The following paragraphs examine the differences between the two datasets and concludes the range to be included in comparisons of single impact categories.

6.2.4.1 Global warming potential

The results show that the contribution from German production of bottle grade PET is 20 % larger than the contribution from average European production as well as the compiled (and applied) dataset. This difference is not readily explained, not at least because the three datasets are both very recent with reference year 2015 and 2016, respectively.

In this context, it is worth noting that the contribution from PET production to climate change over time has been calculated with very different results as shown in Table 39.

Table 39. Values for the Global Warming Potential for bottle grade PET as reported by PlasticsEurope and thinkstep™ since 1995.

Data source	Age	Value kg CO_2 eq./kg	Comment
PlasticsEurope, report 8	1995	2.30	Only CO_2
PlasticsEurope, report 16	1998	4.58	Including CO_2 and CH_4
PlasticsEurope	2001	3.40	Calculated by Ian Boustead
PlasticsEurope, dataset found in GaBi	1999 (2001)	3.27	PET amorphous. Calculated using ILCD characterization as found in GaBi
PlasticsEurope, dataset found in GaBi	2009 (2011)	2.14	Calculated using ILCD characterization as found in GaBi
PlasticsEurope, 2017	2015	2.19	Calculated from SimaPro using ILCD characterization
GaBi dataset for German production	2016	3.01	ILCD compatible data
GaBi dataset for German production mix	2017	2.67	Characterization factors from IPCC AR5, excl. biogenic carbon and including Land Use Change

It is judged that the very large variations are not caused by different scopes or technologies behind the data. Rather, the differences are caused by poor data collection and/or data quality. As shown earlier the variation between single producers may very well be higher, and a conservative range of +/- 25 % is suggested.

6.2.4.2 Ozone depletion potential

There are eight orders of magnitude between the impacts from German production mix and the average European impact. Bromomethane accounts for more than 90 % of the average European impact, but less than 0.01 % in the German production mix.

It is not possible to verify the different consumption pattern, and it is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.4.3 Human toxicity; non-cancer effects

The impacts from average European production are seven times higher as for the German production mix, where emissions of mercury to air is the main source, while the reported

mercury emissions in the European average are 50 times lower. It was not possible to identify the main contribution to the European average, but alone the differences with respect to mercury emissions makes it difficult to use the impact category as a meaningful indicator in a comparison with other materials.

It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.4.4 Human toxicity; cancer effects

Similar considerations as for polystyrene. The large variation between European average production and German production mix makes it difficult to use the impact category as a meaningful indicator in a comparison with other materials. It is noted that the values from the compiled dataset and German production mix only differ by 10 % but may very well be coincidental (see section 5.2.1.3). It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.4.5 Particulate matter/Respiratory inorganics

Similar considerations as for polystyrene. A range of +/- 50 % is suggested. It is noted that the value in the compiled dataset is more than twice as high as the European average, but it has not been possible to investigate this difference in any detail.

6.2.4.6 Ionizing radiation (human health)

The ionizing radiation from the two scenarios is in both datasets related to emissions of carbon (C14), but the contribution to the European average is 15 times higher than the German production mix, and five times higher than in the compiled dataset. It is not possible to identify the contributing processes in more detail. Thus, it is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.4.7 Photochemical ozone creation potential (POCP)

The POCP of European and German average production differs by approximately 50 %. The main sources are the same for both datasets (nitrogen oxides and NMVOC), but nitrogen oxides appear to be more important for the European average; the emissions are simply larger. A range of -/+ 50 % is suggested for this impact category. The value in the compiled dataset is very similar to the European average.

6.2.4.8 Acidification (AP)

The AP of German production is about 40 % of the European average. Sulphur oxide and nitrogen oxides are the main contributors in both averages, and the smaller value for German production can perhaps be attributed to less sulphur in the fuels used and/or a more efficient flue-gas cleaning. A range of -/+ 50 % is suggested. The value in the compiled dataset is 20 % higher than the European average, and it is thus covered by the suggested range.

6.2.4.9 Freshwater eutrophication

The impacts from the German production mix are less than 10 % of that from average European production. The difference is related to emissions of phosphate, but the originating process cannot be identified. It is noted here that the PlasticsEurope dataset does not distinguish between emissions to fresh water and seawater, and this is most probable of importance for the impact category results. The value in the compiled dataset is eight times higher than the European average. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.4.10 Terrestrial eutrophication

The impacts from the German production mix are about 65 % of the average European production. The difference is primarily due to a lower emission of nitrogen oxides/nitrogen dioxide. A range of -/+ 50 % is suggested, and this will also cover the value in the compiled dataset which is slightly higher than the European average.

6.2.4.11 Marine eutrophication

See section about freshwater eutrophication above. It is noted that emissions of nitrogen oxides to air is the largest contributor to the German production mix, while nitrate only plays a minor role. The emissions of nitrogen are four times higher in the European average, and it has not been possible to investigate the importance of nitrate in the calculation of the European average. The value in the compiled dataset is slightly higher. A range of +/- 50 % is suggested to be used with caution in comparisons.

6.2.4.12 Freshwater ecotoxicity

The impacts from average European production are six times higher than from the German average, and the impacts in the compiled dataset are 20 times higher. Emissions of chromium to fresh water is the largest contributor (40 %) to the German average. It is not possible to examine the inventory for the European average in the same detail due to lack of data.

It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.4.13 Land use

Similar considerations as for polystyrene. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.4.14 Water resource depletion

The water use varies with a factor 2 between European average and German production mix, and with a factor 3 between the compiled dataset and German production average.

A factor 2 range cover these differences and can be used with caution in comparisons.

6.2.4.15 Abiotic depletion (minerals)

The depletion of mineral resources is about twenty times higher in the German production mix and more than fifty times higher in the compiled dataset than in the European average. The large differences – and in-transparent dataset make it impossible to suggest a valid range. It is concluded not to consider this impact category in a comparison between fossil- and bio-based plastics.

6.2.4.16 Abiotic depletion (fossil fuels)

The depletion of fossil fuels is 15 % higher for the German production mix, while the value in the compiled dataset is very similar to the European average. A conservative range of +/- 25 % is suggested for comparisons.

6.2.4.17 Non-renewable energy use

Same considerations as for abiotic depletion of fossil fuels. A conservative range of +/- 25 % is suggested for comparisons.

6.2.4.18 Ranges for the PET LCA data to be applied in the comparative study.

Table 40. Applied ranges for the PET LCA data.

Impact category	Unit	Value	Suggestion for ranges relative to European average
Climate change	kg CO_2 eq.	2.19	+/- 25 %
Ozone depletion	kg CFC-11 eq.	Not applicable	Not applicable
Human toxicity, non-cancer effects	CTUh	Not applicable	Not applicable
Human toxicity, cancer effects	CTUh	Not applicable	Not applicable
Particulate matter	kg PM2.5 eq.	1.08E-03	+/- 50 %
Ionizing radiation HH	kBq U235 eq.	Not applicable	Not applicable
Photochemical ozone formation	kg NMVOC eq.	7.36E-03	+/- 50 %
Acidification	molc H+ eq.	1.13E-02	+/- 50 %
Terrestrial eutrophication	molc N eq.	1.96E-02	+/- 50 %
Freshwater eutrophication	kg P eq.	Not applicable	Not applicable
Marine eutrophication	kg N eq.	1.91E-03	+/- 50 %
Freshwater ecotoxicity	CTUe	Not applicable	Not applicable
Land use	kg C deficit	Not applicable	Not applicable
Water use	m^3	3.06	factor 2
Abiotic depletion	kg Sb eq.	Not applicable	Not applicable
Abiotic depletion (fossil fuels)	MJ	67.99	+/- 25 %
Non-renewable energy use. (NREU)	MJ	69.60	+/- 25 %

These values and ranges provided in Table 40 for 1 kg PET bottle grade granulate production are used for making comparison with bio-based plastics in Case study 1 PET bottles and Case study 3 cups.

6.2.5 Non-compliance of data for fossil-based plastics

In this section the non-compliance aspects of the data for the fossil-based plastics are summed up.

First it is noted that the datasets from PlasticsEurope in some cases do not comply with the requirements defined in the ILCD methodologies.

One of the drawbacks of most of the PlasticsEurope datasets (polystyrene is an exception) is that they do not make a distinction between emissions to fresh water and to seawater. This makes it difficult or even impossible to calculate the impacts on human health and ecosystems as well as marine and fresh water eutrophication, because the receiving compartment is an important element in determining the fate and effect of an emission.

Secondly, PlasticsEurope does not have specific requirements for the allocation performed when establishing an eco-profile. PlasticsEurope only requires that the approach should be stated clearly.

The data embedded in ecoinvent for fossil plastic production are based on production conditions in 1999-2001, probably the same sample as included in the PlasticsEurope data. As the foreground processes have not been subjected to an update, the impacts calculated using the ILCD methodology in general are significantly different than for the updated datasets from PlasticsEurope calculated in SimaPro.

6.2.6 Concluding comments on ranges for environmental impacts

Generally – for the four types of plastic investigated – there are seven impact categories that are not applicable for use. These are ozone depletion, human toxicity (cancer and non-cancer effects), ionizing radiation, freshwater ecotoxocity, land use and abiotic depletion.

The reason is mainly that the comparison of datasets with different origin has for each of the four fossil-based plastics revealed large differences. There are several causes for the differences, the main reason probably being that the core datasets from PlasticsEurope in most cases are non-compliant with the requirements defined in the ILCD handbooks. Another reason is the high data age which does not comply with the requirements in ILCD/PEF.

For LDPE, PET and PP the PlasticsEurope datasets make no distinction between emissions to fresh water and to seawater. This makes it difficult or even impossible to calculate the impacts on human health and ecosystems as well as marine and fresh water eutrophication, because the receiving compartment is an important element in determining the fate and effect of an emission.

When data for fossil-based plastics from PlasticsEurope are compared to data from other data sources for plastics that allegedly are ILCD-compliant, and this will inevitably lead to differences that are large for some impact categories, and small for others. Many of the datasets from PlasticsEurope are currently being updated, but no information is available as to when they will be published.

Another issue is the applicability of the ILCD-methodology to compare fossil-based to bio-based plastics. Environmentally important elements like toxic impacts on humans and ecosystems can only be assessed with a high degree of uncertainty. This is an inherent problem caused by inadequate inventories on the one hand and limited knowledge about the toxicological properties of chemicals being emitted to the environment from industrial and agricultural activities on the other hand.

Furthermore, the datasets from PlasticsEurope do not have the same level of detail as many other datasets with respect to the number of emissions being reported and to which compartments they are emitted.

The ecoinvent data for the plastic production processes are relatively old (1999-2001). Only the backgound processes have been subject to updates. The data age makes it less relevant to use these data.

The ILCD-methodology does not address biodiversity directly in any of its elements, and the end-point impacts from land use, consumption of mineral plus fossil resources and fresh water etc. It has therefore been chosen to change the methods for assessing some impacts to methods that are not recommended by ILCD but may be more suitable for the given purpose.

It must, therefore be acknowledged by decision-makers that quantitative LCA cannot give an answer to all questions. Some issues must also be considered in qualitative terms, and it is in all circumstances important to be very cautious when interpreting results obtained by different consultants for different commissioners using slightly different approaches.

6.3 Investigation of the importance of selected upstream activities in production of fossil-based plastics

This sub-chapter examines the importance of different activities in the upstream production of raw materials for fossil-based plastics. Data for plastic granulate as found in the publicly available database from PlasticsEurope and commercial databases from thinkstep™ and Pré is almost exclusively presented as aggregated unit processes, without the possibility of identifying the most important activities for the potential environmental impacts from production of commodity plastics.

The sub-chapter addresses the relative importance with respect to climate change of the activities until the crude oil and gas arrives at the refinery. An example shows that the impacts from such upstream processes accounts for about 25 % of the total impacts from plastic production, but the importance varies from country to country and from plastic type to plastic type. It has not been possible to establish more detail than presenting information on the importance of venting, flaring and fugitive emissions as well as transportation, but it has been possible to establish a range for the upstream impacts of fossil fuels being used in different European countries. This information is used to calculate the importance of raw materials supply relative to the plastic granulate being marketed by European companies. Similarly, quantitative information is not available for other impact categories addressed in the ILCD methodology.

The sub-chapter also addresses issues related to missing information about upstream processes in databases and/or poor prediction power in the impact assessment methodology applied. The focus here is on oil spills, land use and biodiversity, but it must be acknowledged that it is outside the scope of the chapter to establish helpful solutions in any detail.

6.3.1 Extraction from oil and gas

The environmental impacts from extraction of crude oil and natural gas depend on several factors, where flaring and venting are important elements as outlined by Cai *et al.* (2014) and described in some detail below. There is no information available to the reviewer that makes it possible calculate the climate change from oil and gas extraction from a specific supplying country. It is, however, possible when using the GaBi databases from thinkstep™ to calculate climate change impacts from the oil and gas consumption mix for a given country. Such climate change impacts reflect the weighted average of own production and imports from other countries. Basically, this means that any further processing or direct use of oil and gas in a country starts with a specific rucksack, and that the differences between 'country rucksacks' therefore also shall be reflected in different climate change impacts of identical products like commodity plastics.

Table 43 shows the EU ranges and the average climate change impact per kg plastic for 36, resp. 40 countries. Also shown is the climate change impact per MJ consumed for the

extraction of crude oil (Table 41) and gas (Table 42) for the plastic. Table 41, Table 42 and Table 43 were prepared by the project team using data available in the GaBi databases from thinkstep.

Please observe that this figure only considers the average net calorific value of oil and gas - a value that differs between the individual sources.

Table 41. Carbon footprints (CF) associated with extraction of crude oil consumed in European countries.

Crude oil extraction	Unit	Value	Consuming/ receiving country	Largest supplying countries
Highest CF	kg CO_2 eq./kg	0.454	Portugal	Angola and Nigeria
Lowest CF (Europe, incl. N and CH)	kg CO_2 eq./kg	0.063	Norway	Norway
Low CF EU27 Member States	kg CO_2 eq./kg	0.182	Romania	Romania, Kazakstan, Russia
Average	kg CO_2 eq. /kg	0.267		
Net calorific value	MJ	42.33		
CF/MJ	kg CO_2 eq. /kg	0.00631		

Table 42. Carbon footprints (CF) associated with extraction of natural gas consumed in (mostly) European countries.

Natural gas extraction	Unit	Value	Consuming/ receiving country	Largest supplying countries
Highest CF	kg CO_2 eq./kg	0.840	Slovakia	Russia
Lowest CF (Europe)	kg CO_2 eq./kg	0.085	Norway	Norway
Low CF EU27 Member States	kg CO_2 eq./kg	0.138	The Netherlands	The Netherlands
Average	kg CO_2 eq./kg	0.502		
Net calorific value	MJ	44.08		
CF/MJ	kg CO_2 eq./kg	0.0114		

The carbon footprint of extraction of crude oil and natural gas can be used to calculate the relative importance of the activities until the oil and gas is ready to be consumed in a given country.

The following example shows the relative importance of the production of polystyrene in Germany and for average European production. The polystyrene climate change impact and the consumption of oil and gas (in MJ) was calculated using information available in GaBi, and subsequently combined with the carbon footprints for the specific supply to Germany and EU27. It is noted that the consumption details are only available for Germany, and the calculations shall only be used to give a qualified estimate of the relative importance (Table 43).

Table 43. Calculation of the relative importance of fuel extraction in production of polystyrene.

Polystyrene production	Unit	Germany	EU27
Climate change impact of PS	kg CO_2 eq.	2.21	2.24
Crude oil consumption	MJ	42.7	48.8
Natural gas consumption	MJ	27.9	24.9
Carbon footprint of fuels consumed	kg CO_2 eq.	0.555	0.591
CF of fuel extraction/climte change		25 %	26 %

The example shows that fuel extraction accounts for approximately 25 % of the total carbon footprint of polystyrene. It is noted that the relative contribution will be lower when polystyrene is produced in countries where the supply of oil and gas is less demanding in

terms of carbon footprint from fuel extraction, and vice versa. The percentage will also be different for other types of plastic, and the values should only be regarded as indicative.

6.3.2 Flaring and venting

Gas flaring is the general term for the process of burning-off associated gas from wells, hydrocarbon processing plants or refineries, as a means of disposal or as a safety measure to relieve pressure. In all cases, the energy is not recovered. Flaring is recognised as a major environmental problem, contributing to approx. 400 Mt CO_2 per year with 150 billion m^3 of natural gas being flared around the World (Emam, 2015).

Flaring is only used where the gas cannot be utilised or pumped back to the well to increase pressure. More precisely, flaring takes place at sites where there is no demand for the surplus gas, and this is a major reason for the differences observed between oil and gas extraction sites. Cai *et al.* (2014) reports that the flaring intensity, measured in m^3 gas/bbl crude, is five times higher in Nigeria than in the USA and nine times higher than for the countries defined in Cai *et al.* (2014) as "Rest of World". In Norway, there is an enforced policy of zero flaring, while in Uzbekistan, the flaring intensity is 13 times higher than in the US. Flaring thus adds to the environmental impacts from production of fossil fuels. EASAC (2016) quotes a study where there is a difference of 15 g CO_2 eq./MJ between the Well-to-Wheel (WtW) climate change of diesel based on North Sea crude (84 g CO_2 eq./MJ) and heavy Venezuelan crude (99 g CO_2 eq./MJ), but the difference may be even higher, depending on the producing country and/or the specific technology and site.

Depending on the production technology used, the WtW climate change of oil-sands derived petrol and diesel in the USA averaged 100-115 and 99-117 g CO_2 eq./MJ, respectively. Comparative figures for USA conventional crudes were 92 and 91 g CO_2 eq./MJ, respectively (EASAC, 2016).

Gas venting is the discharge of unburned gases into the atmosphere, often carried out in order to maintain safe conditions during the different phases of the treatment process. During venting operations methane, carbon dioxide, volatile organic compounds, sulphur compounds and gas impurities are released. As an example, when methane is emitted, its full global warming potential is released, causing an impact that is about 25 times larger than if the methane was burned.

In recent years, there has been an international trend to reduce gas flaring and venting through the World Bank Global Gas Flaring Reduction (GGFR) partnership and the Global Methane Initiative (GMI). Several countries are now signatories on the GGFR partnership's voluntary standard for flare and vent reduction, and both the GGFR partnership and GMI actively promote demonstration projects to reduce flaring and venting (Emam, 2015).

Cai *et al.* (2014) estimated the combined emissions from venting, flaring and fugitive emissions (the VFF factor) in US production of crude oil to 108 g CH_4 and 120 g CO_2 per million Btu (mmBtu). Assuming that one mmBtu equals 1.05 gJ and that the climate change impact potential of methane is equal to 25 g CO_2 eq., the climate change impact potential of the VFF factor can be calculated to 2.7 g CO_2 eq./MJ. The by far greatest contribution (91 %) to the VFF factor comes from venting of methane, and the contribution from flaring is only 0.1 g CO_2/MJ. Off-shore platforms in shallow water, oil tanks and pneumatic devices were identified as the most important elements from a climate change point of view in the supply of crude oil to the US. Increasing the amount of methane not being destructed in flaring with e.g. two percent does not add significantly to the combined VFF factor.

It has not been possible to examine the extent to which EASAC includes the contributions from venting in their calculations of climate change. Much attention is given to flaring, but the calculations based on the combined VFF factor and its elements indicates that venting is much more important.

6.3.3 Transportation

The International Council on Clean Transportation (ICCT) calculated the emissions from tanker transport to the EU to be 1.1 g CO_2 eq./MJ, without stating the distance. Calculating the impacts from transportation of crude oil from Lagos, Nigeria to Rotterdam in a 10,000-300,000 dwt tanker shows a comparable figure, i.e. 0.96 g CO_2 eq/MJ. Obviously, the impacts depend on the distance, but it is evident that transport plays a visible role when crude oil is transported from African countries and other distant suppliers.

6.3.4 Upgrading of oil sands

There are two major methods of bitumen extraction. Mining of bitumen involves the open-face mining of oil sands ore. The oil ore is transported to a central processing plant, where the bitumen is separated from the sand and subsequently upgraded to synthetic crude oil, which has technical properties similar to those of light crude oil. Mining can only be effective for bitumen deposits located sufficiently close to the surface (up to roughly 75 meters in depth below the surface).

In-situ bitumen extraction refers to the extraction of bitumen directly from the ground. As the bitumen does not flow naturally at the reservoir temperature, the viscosity of the bitumen must first be increased to allow the bitumen to be pumped out of the ground. This is typically achieved by injecting steam into a well, which heats the bitumen, reducing its viscosity and allowing it to be pumped from the well.

CERI (2014) points to the following environmental concerns in upgrading of oil sands extracted by mining:

- Land use change and reclamation: the mining region contains a high number of natural wetlands, which are important habitats and very difficult to restore to their natural condition. In addition, some mine features, such as tailings ponds, are difficult to reclaim.

- Energy use and greenhouse gas emissions: the mining process consumes considerable energy. Fuel is required for the mining vehicles, for heating the water for hydro transport and bitumen extraction, and for the upgrading of bitumen to synthetic crude oil. This energy is typically derived from the burning of fossil fuels, which releases greenhouse gases that contribute to climate change.

- Air pollutant emissions: exhaust from mining vehicles and stack emissions from the processing plants and upgraders can contribute to local and regional air pollution. Exposed mine surfaces can release volatile organic compounds and dust, while contaminated tailings ponds can release pollutants as well through direct evaporation or activity of microorganisms.

- Water use: a substantial amount of water is required for the separation of bitumen from the oil sands ore, which must be obtained from local fresh water sources such as rivers. As some extraction water is tied up in tailings, new water must be used on a regular basis, which removes it from the local fresh water systems.

- Water pollution: tailings pond water contains several contaminants from the oil sands extraction process, some of which are toxic to aquatic life. There is concern about tailings water entering the water system and the effect this might have.

In-situ extraction of bitumen has a somewhat different environmental profile. The following differences appear to be most important:

- Surface blowout and groundwater contamination: in-situ extraction requires that the oil sands formation be deep enough and the cap rock above the formation be strong enough that the steam stays within the reservoir, not reaching the groundwater.

- Land disturbance: since the extraction process takes place mostly underground, the surface footprint of in-situ projects is considerably smaller than for surface mines.

- Air emissions: although in-situ primarily uses cleaner-burning natural gas a fuel source, there are still air pollutant emissions that arise from operation beyond greenhouse gases.

6.3.5 Climate change of upgrading activities

Most of the environmental concerns addressed above cannot be quantified with existing LCA methodologies. It is, however, possible to quantify the climate change impact potential of oil sands mining and upgrading to synthetic crude oil (SCO) using information from Environment Canada Facility GHG Reporting as found in CERI (2014). The greenhouse gas intensity has remained constant from 2004-2012, being about 97 kg CO_2 eq/barrel SCO. With an energy content of 1.05 gJ/barrel SCO, this value corresponds to 0.092 kg CO_2 eq/kg SCO or about 2.3 g CO_2 eq/MJ. This is a little less than the climate change impact potential of conventional oil extraction, but it must be remembered that the value for this is associated with wide ranges.

Subsequent transportation of the SCO through pipelines cannot be quantified with respect to environmental impacts, but there is an obvious risk of large environmental impacts if the pipeline breaks for some reason.

6.3.6 Water consumption in upgrading of bitumen

Hanania et al., (n.d.) estimates 2.8 kg fresh water per kg bitumen, while CERI (2014) reports that 3.2 kg of fresh water is used for every kg of SCO. Being related to Canadian production, the quality and nature of the water is not described according to ILCD requirements, and the values should only be regarded as indicative.

6.3.7 Oil spills

The environmental impacts from oil spills are not considered in conventional LCA, although they for many people are considered very serious. Oil spill and accidents/events where large amounts of oil are emitted to marine waters or different types of soil do have profound and immediate effects on ecosystems and often also on human health.

The following events (see Table 44) are among those with the largest volume being spilled. The spills are caused by sabotage (Kuwait), physical conditions in the well (Lakeview Gusher (USA)), faulty equipment (Deepwater Horizon) (Mexican Gulf), and bad seamanship (Exxon Valdez (Alaska)). Large spills on land or into water can occur if a pipeline breaks for some reason, and especially local spills are caused by ignorance or against better knowledge as is the case when waste oil is emitted from smaller and larger vessels.

Table 44. Selected examples of large oil spills in the last century (based on Wikipedia, 2018).

Event	Year	Tonnes of crude oil (thousands)	Comments
Kuwaiti oil fires	1991	136 000	Oil burned resulting in airborne emissions
Kuwaiti oil lakes	1991	3 400-6 900	Emitted to 300 surface lakes
Lakeview gusher, California USA	1910-1911	1 200	Emitted to surface
Deepwater Horizon, Gulf of Mexico, USA	2010	560-585	Emitted to marine water
Exxon Valdez, Alaska USA	1989	280	Emitted to especially vulnerable waters

Table 44 gives an indication of how large oil spills potentially can be, but volume is a limited indicator of damage or impact. Smaller spills have already proven to have a great impact on ecosystems, such as the Exxon Valdez oil spill because of the remoteness of the site or the difficulty of an emergency environmental response. An example of a catastrophic accident on land is the Lac Mégantic rail disaster, where a 74-car train carrying Bakken Formation crude oil derailed, resulting in a fire and explosion which caused the death of at least 42 people.

It should also be kept in mind that everyday there are several small-scale oil spills all over the world, each of which has an impact on the local environment. It is noted in this context that according to the International Tanker Owners Pollution Federation the number of spills over 7 tons have decreased from about 79 per year in 1970-1980 to 18 per year between 2000-2010 (ITOPF, 2018).

Oil spills at sea are generally much more damaging than those on land, since they can spread for hundreds of nautical miles in a thin oil slick which can cover beaches with a thin coating of oil. These can kill seabirds, mammals, shellfish and other organisms they coat. There is no clear relationship between the amount of oil in the aquatic environment and the likely impact on biodiversity. A smaller spill at the wrong time/wrong season and in a sensitive environment may prove much more harmful than a larger spill at another time of the year in another or even the same environment.

Oil spills on land are more readily containable, e.g. if a makeshift earth dam can be rapidly bulldozed around the spill site before most of the oil escapes, and land animals can avoid the oil more easily.

6.3.8 Land use

Land use is for many decision-makers a relevant parameter when two or more options are analysed in a life cycle perspective. The midpoint characterization factor for land use, in the earliest stage of the cause-effect chain, is mostly expressed as the amount and quality deficit of land occupied or transformed. Some midpoint methods use indicators like soil structure, soil pH or soil organic carbon and the latter approach is recommended by the EU Commission (ILCD, 2011). It is however, acknowledged that it shall be applied with caution because its scope is limited, and that it does not cover effects on biodiversity. There is little doubt that many of the actors in the field of LCA and its applications will appreciate a methodology that

integrates quality aspects beyond soil organic matter, but none of the available methods for this has fulfilled the stringent requirements used when the ILCD methodology was developed.

The LCA impact assessment methods are continuously being improved and associated databases are being extended, and the research needs described in the ILCD methodology report indicate clearly that the priorities for future developments are related to the methods selected for ILCD in the first round.

SOM (Soil Organic Matter) is a key soil quality indicator, especially for assessing the impacts on fertile land use as it influences properties like buffer capacity, soil structure and fertility. In the recommended method, the LCA practitioner is expected to know the location, the timeframe, and the SOM values before and after the land occupation, the SOM value of the reference land system, the relaxation rate, and associated SOM values. Based on this, the LCA practitioner is expected to calculate the characterization factors for the foreground system (ILCD, 2011). Characterization factors for certain land use flows in the background system are provided in Milà i Canals et al. (2007c).

The approach described in the Impact 2002+ methodology was not pre-selected when the ILCD recommendations were established by the group of expert authors behind the methodology. Where the impact in the recommended method is measured by the C-deficit created by an activity, the unit for Impact 2002+ assessments is m2*year. The method thus measures land occupation (how much – and for how long) where the SOM-approach measures the changes in soil quality. The methods be complimentary to each other, and the following values calculated for German consumption mix of fossil-based plastics can be seen as indicative for land occupation (without any quality considerations) as measured by the Impact 2002+ methodology:

- PET: 0.030 m^2*year
- PS: 0.028 m^2*year
- LDPE: 0.060 m^2*year
- HDPE: 0.020 m^2*year
- PP: 0.030 m^2*year

It is noted that the results are calculated from information collected and structured by thinkstep in the GaBi database. Using average data from PlasticsEurope may very well give different results, but it has not been possible to make such calculations. It is further noted that there is no available information relevant for assessment of land use with respect to extraction of shale oil.

6.3.9 Conclusions about the importance of selected upstream activities and production of fossil-based plastics

The main findings in this chapter are summarised below:

- Upstream production of plastic raw materials can account for more than 25 % of the total contribution to climate change impact potentials from cradle-to-gate;
- Depending on supplying countries, the variation between the impacts of upstream raw material production can vary with more than factor 2 between producing Member States;
- Flaring, and especially venting, is an important source of the contribution to climate change. Fugitive emissions play a minor role in this picture;

- Off-shore platforms in shallow water, oil tanks and pneumatic devices were identified as the most important elements in impacts from venting;

- Transport of crude oil has a visible impact, depending on the distance. A credible range is 5-35 % of the total climate change impact potential from extraction and transportation of oil;

- Upgrading of shale oil to synthetic crude oil is associated with a global warming impact that is comparable to that for extraction of oil from conventional sources;

- Neither accidental oil spills nor catastrophes are considered in LCA, although they are the most visible environmental impacts.

Mining and in-situ extraction of shale oil will inevitably affect local ecosystems and most probably also biodiversity. As for oil spills and catastrophes, current LCA methodologies cannot be used to give a quantitative indication of the importance.

7 Biodiversity assessment

7.1 Scope of assessment

The biodiversity assessment methodology described in this chapter addresses the land use impacts on terrestrial biodiversity associated with cultivation of the feedstocks used as a baseline for the case studies carried out in the project, see Table 45. [26] Two assessments will be carried out at different levels of specificity in this study. These are further described in section 7.3.5.

Impacts on terrestrial biodiversity related to urban land uses associated with processing of feedstocks, and production of intermediates as well as bio-based products, and waste management are not considered. However, the agricultural activities included here are likely to be a major contributor to the overall biodiversity loss of the life cycle of bio-based products.

Table 45. Feedstocks included in biodiversity assessments and their origin.

Feedstock	Origin	Case study (baseline) relevance
Sugar cane	Thailand; northeast, central	3, 4, 6
Maize	USA; Nebraska	3, 4, 6
Potato	Europe; Germany	2, 7
Maize	Europe; Italy	2, 5, 7
Sugar cane	Brazil	1

7.2 Choice of characterisation model

The assessment of land use impacts on biodiversity related to the bio-based product case studies is based on the characterisation model presented in Chaudhary et al. (2015), hereafter referred to as the Chaudhary 2015 methodology.

The choice of this characterisation model is based on the recommendation of the UNEP- SETAC Task Force on Biodiversity Land Use. In Global Guidance for Life Cycle Impact Assessment Indicators (2016) the Task Force recommends the methodology after a comprehensive analysis of existing models, test of applicability and consultation with experts (UNEP-SETAC, 2016).

The uniqueness of the methodology can be attributed to its regional and global approach to assessment of biodiversity loss related to land use. Where previous methods have put emphasis on what and how an activity is performed this methodology adds the dimension of where an activity is performed i.e. the consideration that biodiversity varies significantly with location.

The methodology is very new, and applications are few. Practical guidance on application is non-existent at present, which makes application challenging and some "learning by doing" has been necessary in this study. Methodological choices are documented for retrospective reflection.

[26] The scope of the biodiversity assessment here is limited to considering 'the land use impacts on terrestrial biodiversity associated with cultivation of feedstocks' thus impacts related to climate change, acidification, eutrophication etc. also contributing to damage to terrestrial biodiversity are not considered.

In recognition of the widespread use of the ReCiPe 2016 methodology for assessment of land use impacts on biodiversity, a biodiversity assessment has also been performed with this methodology for all case study baseline feedstocks. This assessment is included in Annex 4. A comparison of the results obtained from the two methodologies is also presented here.

7.2.1 Background for choice of characterisation model

The Chaudhary 2015 methodology is a continuation of the work by De Baan et al. (2013) upon which several known LCA land use impact assessment methods are based e.g. ReCiPe 2016 methodology (Huijbregts et al., 2016, 2017; Goedkoop et al., 2009) and Impact World+ (described in Vidal-Legaz et al. (2016)).

For predicting local, regional and global biodiversity loss due to land use species-area relationship models (SARs), which describe the relationship between area and species richness, are central. Until recently, the Classic SAR model has been the most commonly used model, but as a response to some of its known weaknesses the Matrix SAR and the Countryside SAR models have been developed. The ReCiPe 2016 and the Impact World+ methodologies both apply Classical SAR while the Chaudhary 2015 methodology applies Countryside SAR.

The main difference between the Classical SAR model and the Countryside SAR is the latter's consideration of the spatial heterogeneity of biodiversity and differentiated species response to land transformation and occupation.

Land use impacts can be very region specific, for example *'the occupation of cropland in a naturally biodiversity poor region in Europe has a different ecological impact compared to the occupation of cropland in a hot spot of biodiversity in Latin America.'* (Koellner et al., 2013). The Chaudhary 2015 methodology, applying Countryside SAR, enables assessment of regional and global biodiversity loss resulting from land use in 804 terrestrial ecoregions[27] (Figure 18) whereas, for example, the ReCiPe 2016 methodology does not consider spatial differentiation for land use impacts.

Classical SAR operates with the assumption of a binary landscape of either habitat (such as an old-growth forest) or non-habitat (e.g. agricultural land) which means that any anthropogenic activity will result in a 100 % loss of biodiversity. The Countryside SAR on the other hand considers that species have differentiated sensitivity to habitat loss caused by land use changes. Some species show partial or total tolerance to habitats modified by anthropogenic activity whereas others are highly sensitive to habitat loss. The Classical SAR model does not capture these individual differences in the responses of species to land use changes.

The Chaudhary 2015 methodology furthermore presents a unique approach for quantification of potential global extinction of species due to land use changes by combining the Countryside SAR model with vulnerability scores for each ecoregion per taxon based on species-specific threat level and geographic range data. This means that the methodology takes into consideration the richness of threatened endemic species. This consideration is expressed through the global characterisation factors in the methodology.

[27] Ecoregions are defined as relatively large units of land containing a distinct assemblage of natural communities and species, with boundaries that approximate the original extent of natural communities prior to major land-use change. (Olson et al., 2001).

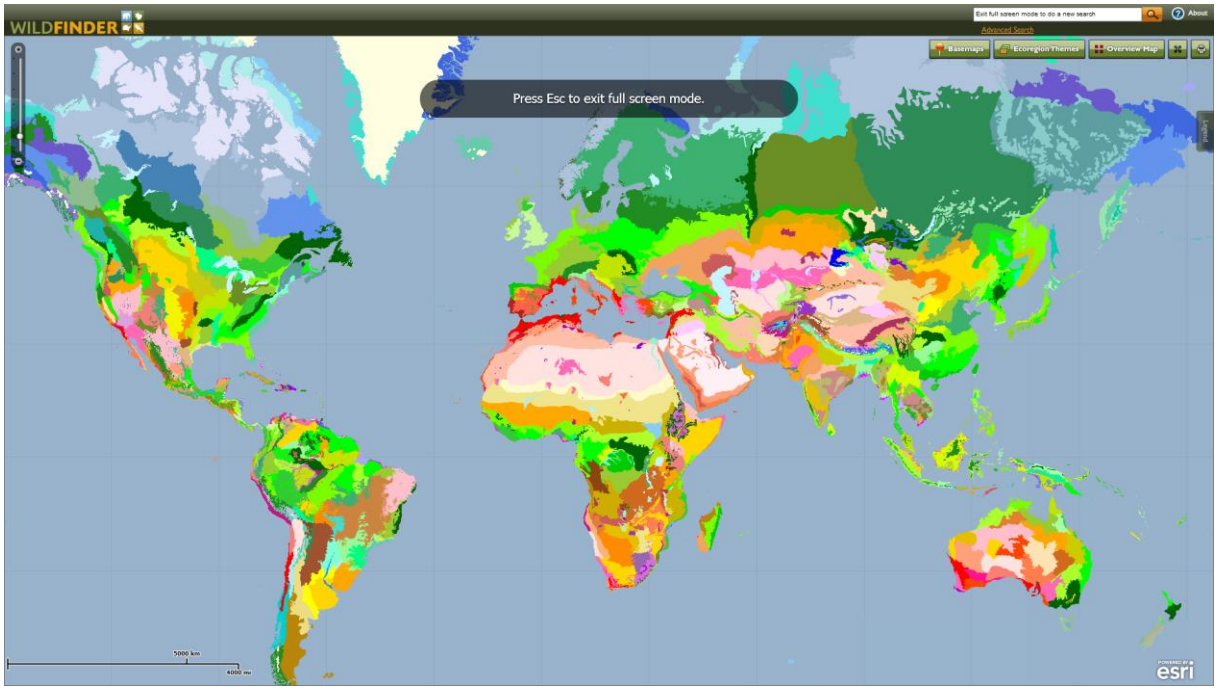


Figure 18. Division of the world into terrestrial ecoregions by Olson et al. (2001) as reproduced by WildFinder (2017). The Chaudhary 2015 methodology enables assessment of regional and global biodiversity loss in 804 terrestrial ecoregions.

7.3 Methodological choices

7.3.1 Land use types

The characterisation model operates with six land use types at present: extensive forestry, intensive forestry, pasture, annual crops, permanent crops, and urban (see Table 46 for further description of these).

This study focuses on biodiversity loss resulting from cultivation of feedstocks and hence impacts caused by urban land use are not considered. Data on the extent and type of land use requirements serve as inventory input for the assessment of biodiversity loss (see chapter on land use change assessment). The feedstocks included in this study are either annual crops or permanent crops.

Table 46. Land use definitions applied by Chaudhary et al. (2015) based on Koellner et al. (2013) and as referred to in present study.

Land use ID	Land use type	Definition	As referred to in modelling
1.2.1	Forest, used, extensive	Forests with extractive use and associated disturbance like hunting, and selective logging, where timber extraction is followed by re-growth including at least three naturally occurring tree species.	Extensive forestry
1.2.2	Forest, used, intensive	Forests with extractive use, with either even-aged stands and clear-cut patches, or less than three naturally occurring species at planting/seeding.	Intensive forestry
4.2	Pasture/meadow	Areas that have been converted to grasslands for livestock grazing or fodder production	Pasture
5.1	Agriculture, arable	Cultivated areas regularly ploughed and generally under a rotation system. Cereals, legumes, fodder crops, and root crops. Includes flower and tree (nurseries) cultivation and vegetables as well as aromatic, medicinal and culinary plants. Excludes permanent pastures.	Annual crops
5.2	Agriculture, permanent crops	Perennial crops not under a rotation system which provide repeated harvests and occupy the land for a long period before it is ploughed and replanted: mainly plantations of woody crops	Permanent crops
7.1	Artificial areas, urban	Areas with infrastructure for living and businesses	Urban

7.3.2 Category indicator

The category indicator of the characterisation model of the Chaudhary 2015 methodology is potential species loss (PSL). The indicator covers the impact on five taxonomic groups; birds, mammals, reptiles, amphibians, and vascular plants[28].

7.3.3 Regional and global impact categories

The characterisation model assesses two categories of impacts PSL_{reg} and PLS_{glo}:

- Regional Potential Species Loss PSL_{reg}
- The unit of the characterisation factor (CF) is:[29]
- Regional species lost/m² land, or
- Regional land PDF/m² (when aggregated across taxa)
- Global Potential Species Loss PSL_{glo}[30]

[28] Vascular plants are only covered by some sets of characterisation factors.

[29] The unit of the characterisation factor refers to an average assessment (on average assessment see section 7.3.4).

- The unit of the characterisation factor (CF) is:
- Global species eq. lost/m² land, or
- Global PDF /m² land (when aggregated across taxa)

When the characterisation factors are aggregated across taxa the term potential disappeared fraction (PDF) of species is used instead of 'species lost' which refers to a specific taxonomic group e.g. birds (see section 7.3.6).

A schematic of the applied characterisation model is presented in Figure 19. Life cycle inventory data is collected for each specific land use type and for a specific geographical unit: ecoregion, country, or continent (see section 7.3.5). In the Chaudhary 2015 methodology 'Local species loss' is used as a stepping stone to assess regional and global impacts and is hence not included as a separate impact category.

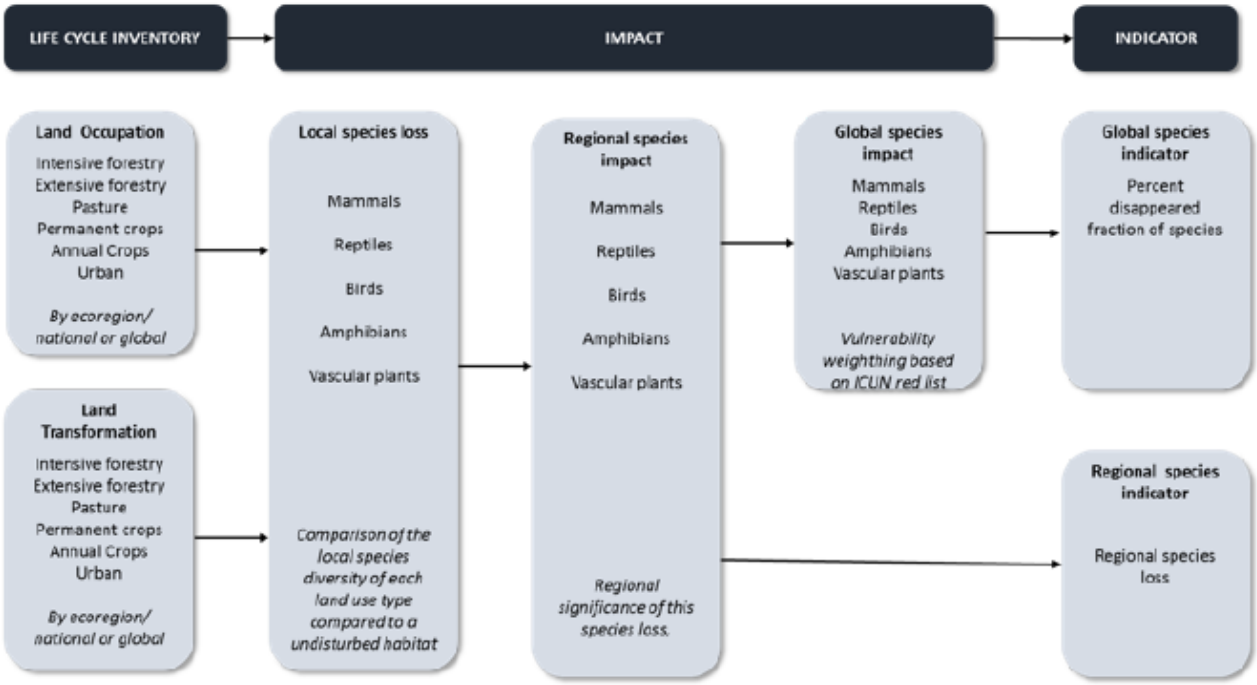


Figure 19. Schematic of the model used for calculation of the recommended category indicators. (UNEP-SETAC, 2016).

Where global characterisation factors are based on threatened endemic species richness, the regional characterisation factors are based on species richness. Regional impact assessment considers biodiversity as a local matter whereas Global biodiversity impact assessment

[30] If the species are endemic to the ecoregion, their loss will translate into global species loss (extinction). To determine an estimate of the permanent global (irreversible) species loss, the regional CFs for each taxon and ecoregion are multiplied by a vulnerability score (VS) of that taxon in that ecoregion. The VS is based on the proportion of endemic species in an ecoregion and the threat level assigned by the IUCN red list for the different taxa and regions. (UNEP-SETAC, 2016).

considers biodiversity a global matter by directing attention towards locations inhabited by vulnerable species in terms of threat level or range restriction.

Global characterisation factors express vulnerability weighted biodiversity loss per unit of land occupied or transformed. So, compared to the potential biodiversity loss predicted with regional characterisation factors, the global characterisation factors add emphasis to endemism and endangerment of species, thus portraying the biodiversity loss through a global lens. High characterisation factors can be regarded as estimates of permanent species extinction caused by land use. When comparing global characterisation factors for different ecoregions a relatively higher characterisation factor for one ecoregion compared to another expresses a higher vulnerability of species inhabiting that ecoregion.

When both the regional and global impact categories are included in the biodiversity loss assessment, it enables the LCA practitioner to consider trade-offs between species richness loss (regional impact category) and threatened endemic species (richness) loss (global impact category) because different hotspots of biodiversity impacts may emerge depending on the chosen impact category. Therefore, both impact categories are included in this study.

7.3.4 Average or marginal assessment

The Chaudhary 2015 methodology proposes the option of performing either average or marginal assessment of land use impacts on biodiversity and hence provide sets of characterisation factors for both types of assessment[31] (see Figure 20).

Consistency between the logic applied in connection with the assessment of land occupied (m^2*yr/kg crop) determined elsewhere in the project (section 4 and yields selected) and the logic applied to the choice of characterisation factor (-sets) for biodiversity assessment is important. In the case study LCAs, the calculation includs the yields from land 'already' occupied, instead of the yields from the 'additional land to be occupied'. The latter would be linked to a demand increase which is not found to be the case.

UNEP-SETAC (2016) provides a general recommendation to perform average assessment: *'Our recommendation is to use average factors (and not marginal) for consistency with other indicator models used in the LCA. However, if it is known that the system in focus causes significant land expansion, we recommend considering marginal values.'*

In this study, average assessment characterisation factors are chosen for biodiversity assessment. The main arguments are that: (a) it is in line with the UNEP-SETAC recommendation; (b) consistency with the way land occupation is modelled in the case study LCAs.

7.3.5 Geographical unit of biodiversity assessment

The Chaudhary 2015 methodology provides a spatial resolution of 804 ecoregions i.e. a specific characterisation factor is provided for each of the five taxa, for each type of land use, and for each ecoregion location. The result is a multitude of characterisation factors, see the schematic presentation in Figure 20.

Applying ecoregion specific characterisation factors however requires very specific information on where the cultivation of feedstocks takes place, which is often not available to the LCA

[31] A comparison of marginal and average characterisation factors for most ecoregions shows that the marginal characterisation factors are approximately 2.5 times higher than for the average characterisation factors for most ecoregions, highlighting the importance of the assessment approach (Chaudhary et al., 2016).

practitioner. Therefore, geographically aggregated characterisation factors have also been developed in the Chaudhary 2015 methodology. These factors have been obtained by aggregating the regional and global characterisation factors for the 804 ecoregions into 245 country average characterisation factors[32]. More specifically, country average characterisation factors are calculated based on shares of ecoregions located within the country. These characterisation factors build on the underlying premise that the cultivation areas of a crop are evenly distributed throughout the country area. In other words, the share of ecoregion affected by cultivation is set as equal to that of the area share of that ecoregion within the country. Furthermore, species richness, endemic species richness, species threat level and remaining natural habitat area may vary significantly in ecoregions within the same country. These differences are averaged out in the country characterisation factor.

The application of these aggregated country averages thus introduces considerable uncertainty to an impact assessment, especially for large countries. The use of aggregated characterisation factors in general also complicates in-depth interpretation. The presented aggregated country factor appears as a black box, making it difficult to investigate what drives the potential impact and gain a true understanding of how activities affect different taxonomic groups. Country averages however enable assessment in the absence of more specific data and should therefore be applied in these situations as well as for background processes.

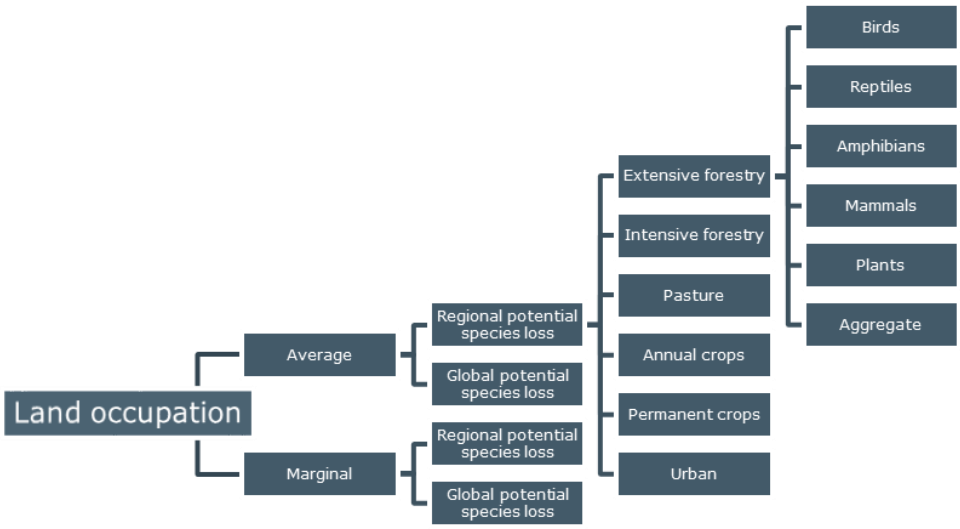


Figure 20. Schematic overview of ecoregion specific characterisation factors developed for impact assessment of land occupation on biodiversity in the Chaudhary 2015 methodology. For each ecoregion, a set of characterisation factors exist for land occupation, which is further distinguished for average or marginal assessment. For each impact category PSL_{reg} and PSL_{glo} there is a specific characterisation factor for each of the 5 taxa for each of the 6 land use types.

In this study biodiversity assessment is carried out at ecoregion level for cultivation of Thai sugar cane and American maize used as base materials for production of PLA (case study 3, 4, 6), and at country level for cultivation of all case study baseline feedstocks (see Table 45). The assessments are carried out in sections 7.4 and 7.5.

[32] Continental global characterisation factors calculated using the Chaudhary 2015 methodology are also made available through UNEP-SETAC (2016).

Sugar cane (Thailand) and maize (USA) are chosen here for more detailed assessment for several reasons. The two feedstocks are cultivated in very different parts of the world and are both used to produce PLA which enters several of the case studies considered in this project. By choosing these two feedstocks, the biodiversity assessment will demonstrate (potential) land use impacts on biodiversity connected to cultivation of feedstock used by the two largest suppliers of PLA resin for bio-based products in the world.

7.3.6 Taxa-aggregated characterisation factors

The five taxonomic groups considered in the characterisation model can be analysed separately or can be aggregated to represent the potentially disappeared fraction (PDF) of species (see Figure 20). The regional and global characterisation factors (CF_{reg}/CF_{glo}) are typically aggregated to provide a single value for potential species lost from land use PSL_{reg} and PLS_{glo} respectively. This provides a good overview of potential impacts.

The relative order of magnitude of characterisation factors differs remarkably depending on the taxa considered within a region for a specific land use type, which indicates that there is not necessarily congruence of biodiversity hotspots defined by different taxa.[33]

In a biodiversity impact assessment, it therefore makes good sense to present results taxa dis-aggregated as well as taxa aggregated, because it allows identification of hot spots in terms of which taxonomic group is mostly affected by the land use in the ecoregions involved. In principle this supports further investigation into specific agricultural practices at the location and possible scope and type of measures that may reduce potential biodiversity impacts. Naturally, the uncertainty of the underlying model must also be considered in this regard as well.

When the geographical unit of the inventory is a country rather than ecoregion, and country characterisation factors are used, the taxa disaggregated result, being an average, does not provide further valuable input to interpretation and decision-making.

In view of this, the biodiversity assessment will provide taxa aggregated and taxa dis-aggregated characterisation results according to the chosen geographical unit of the inventory.

7.3.7 Applied sets of characterisation factors

Given the novelty of the Chaudhary 2015 methodology, there is a lack of completeness of the existing sets of characterisation factors provided by Chaudhary et al. (2015) and UNEP-SETAC (2016) which limits modelling options to some degree. For example, vascular plants are not included in all characterisation factor sets. For this assessment, characterisation factor sets have been chosen from the same source.

The chosen sets of characterisation factors are presented below.

Ecoregion specific characterisation factors (Chaudhary et al.,2015)[34]:

[33] For example: "in one ecoregion we may find a very high global CF for land use type annual crops and reptiles (high endemic richness and high threat level i.e. high VS) and no presence of amphibians at all and low ranks of mammals and birds in corresponding CF list". (Chaudhary et al., 2015)

[34] Supplementary information 2 of Chaudhary et al. (2015)

- Regional occupation characterisation factors calculated using the Countryside SAR model and average approach (units: regional species lost/m^2), five taxa, taxa-aggregated; median values.

- Global occupation characterisation factors calculated using the Countryside SAR model and average approach (units: global species lost/m^2), four taxa, taxa-aggregated; median values.

Country aggregated characterisation factors (Chaudhary et al.,2015)[35]:

- Regional occupation characterisation factors calculated using the Countryside SAR model and average approach (units: regional species lost/m^2), five taxa, taxa-aggregated; median values.

- Global occupation characterisation factors calculated using the Countryside SAR model and average approach (units: global species eq. lost/m^2), four taxa, taxa-aggregated; median values.

7.4 Biodiversity assessment at country level

In this section biodiversity assessment is carried out at country level for cultivation of all case study baseline feedstocks.

7.4.1 Inventory for baseline feedstocks

The area of land occupied in connection with the cultivation of each feedstock is summarised in Table 47. For simplicity, it excludes the cases with average EU crops. For more in-depth understanding of land use data refer to individual case study descriptions and chapter 4 on 'Land use change methodology'.

Table 47. Spatial land use inventory for cultivation of baseline feedstocks with 'country' as the geographical unit. Inventory flows for land occupation is expressed as m^2*years per tonne (t) feedstock fresh weight (fw) (area and time land are occupied for cultivation of feedstock).

Feedstock	Origin	Land use type	Land occupation (m^2*year/t feedstock fw)
Sugarcane	Thailand; Northeast, Central	Permanent	139
Maize	USA; Nebraska, Iowa	Annual	721
Potato	Europe; Germany	Annual	233
Maize	Europe; Italy	Annual	1 060
Sugar cane	Brazil	Permanent	129

7.4.2 Impact assessment at country level

Potential biodiversity loss is assessed based on the relevant global and regional country aggregated characterisation factors (Section 7.3.7) and land requirements of the considered feedstocks (Table 47).

UNEP-SETAC recommends first and foremost global impact assessment and therefore this is presented first in the following (UNEP-SETAC, 2015).

[35] Supplementary information 4 of Chaudhary et al. (2015).

7.4.2.1 Global impact assessment

Figure 21 shows the global potential species loss (PSL_{glo}) assessed for the baseline feedstocks. The global impact assessment predicted the highest potential disappeared fraction of species (PDF) for maize cultivation in Italy, followed by much lower and an almost equivalent PDF for maize and sugarcane cultivation in the USA and Thailand respectively. Cultivation of potato in Germany resulted in the lowest predicted potential PDF of all the feedstocks.

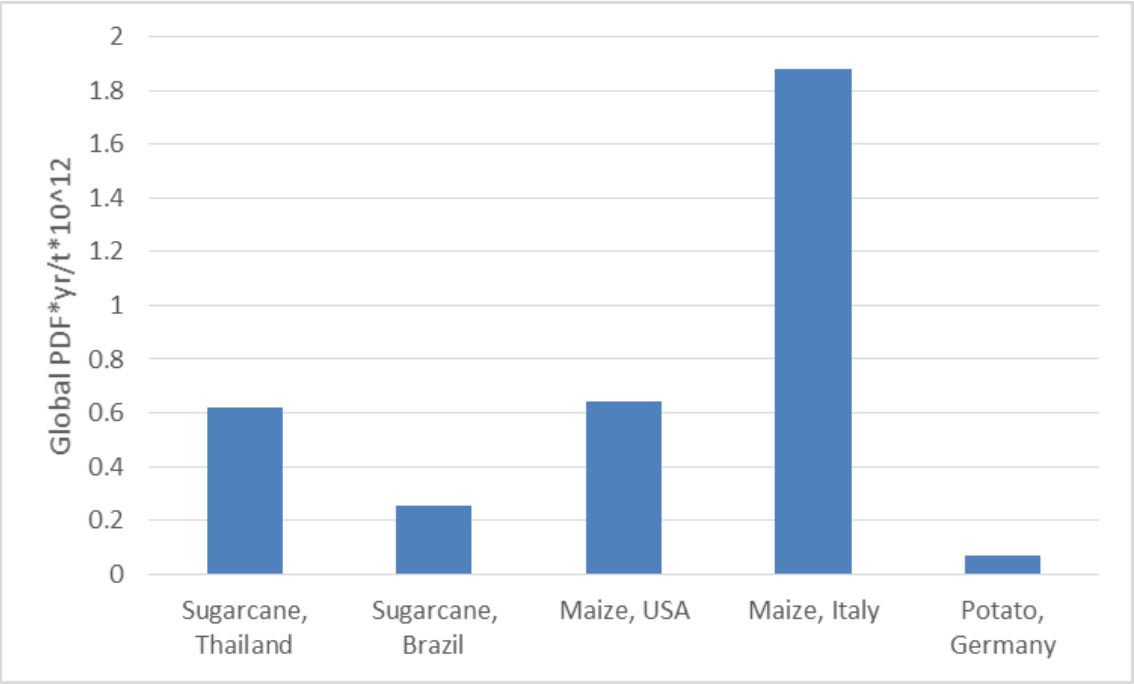


Figure 21. Global potential species loss (PLS_{glo}) per tonne fresh weight feedstock expressed as Global PDF*yr/t. Calculated using occupation average global CFs for aggregated taxa for the case study baseline feedstocks with countries as geographical unit.

The global characterisation factor for Thailand ($4.45*10^{-15}$ Global PDF/m²) is the highest of the five considered, followed by the somewhat lower characterisation factor for Brazil ($1.97*10^{-15}$ Global PDF/m²) and thereafter closely followed by Italy, USA and Germany in that order (See Figure 22). One might have expected the highest potential species loss associated with cultivation of sugar cane in Thailand due to higher species richness. However, the significantly higher land occupation requirements for Italian maize (1060 m²*yr) compared to Thai sugar cane (138.9 m²*yr) per tonne of feedstock combined with a relative high global characterisation factor (Thailand is 2.5 times the magnitude of Italy) drives the impact higher for Italian maize compared to Thai sugar cane. The result is a three times higher potential species loss associated Italian maize than Thai sugar cane.

In this study, significantly higher land requirements for cultivation of maize in Italy and USA compared to sugar cane in Thailand and Brazil influences the difference in predicted biodiversity loss a great deal. The result reflects the fact that the higher vulnerability of species in Thailand (on average) compared to the other countries (as expressed in characterisation factors) affects the potential species loss to a lesser degree than the different land requirements per tonne of feedstock.

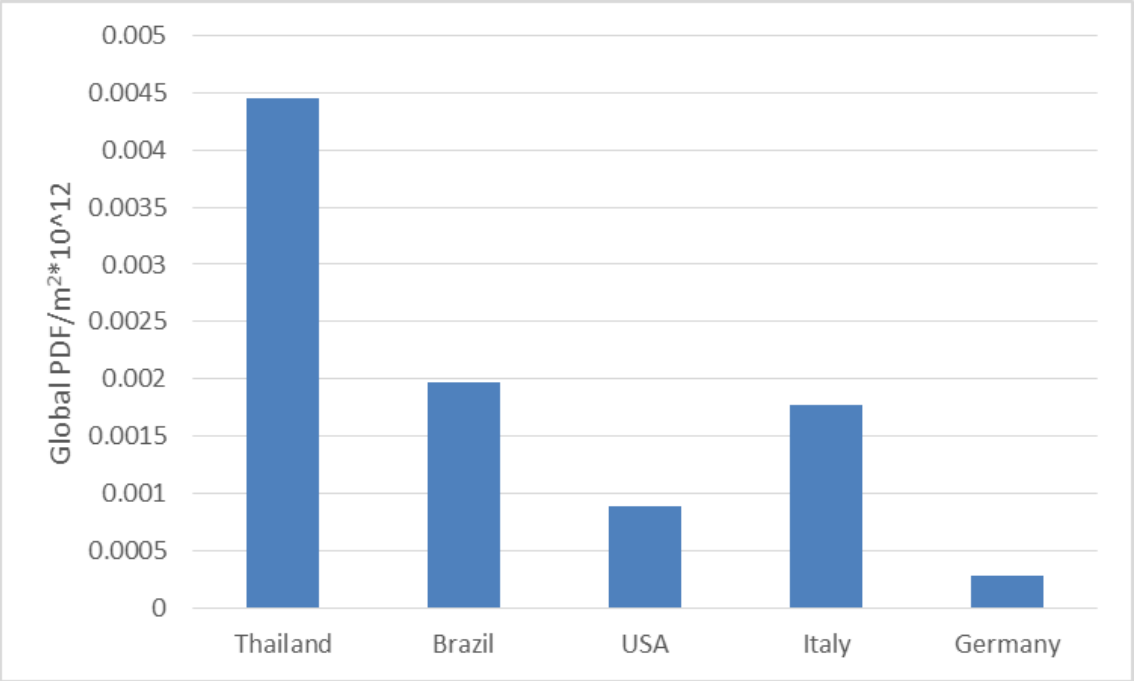


Figure 22. Global country average occupation characterisation factors for four aggregated taxa (mammals, birds, amphibians, reptiles). Unit: Global PDF/m². Chaudhary et al. (2015)

Chaudhary et al. (2015) found that ecoregions in tropical biomes in general have higher global characterisation factors than those in temperate and boreal biomes, primarily because of existing species richness per unit area are higher in tropical biomes. Based on this finding, it may be expected that characterisation factors (averages) for countries with significantly high share of tropical biomes have higher characterisation factors than for those more dominated by temperate and boreal biomes. However, the selection of country characterisation factors included in this study seems not to be entirely in accordance with this observation. Thailand and Brazil are covered largely by tropical biomes (more for Thailand than Brazil) and have higher global characterisation factors compared to the other countries, however Italy also had a surprisingly high global characterisation factor (comparatively). See the biomes represented in the countries considered in this study in Table 48.

Table 48. Biomes represented in the countries where feedstock cultivation takes place. Based on WWF Wildfinder maps (2018).

Feedstock	Location	Represented biomes
Sugar cane	Thailand	Tropical and Subtropical Dry Broadleaf Forests Tropical and Subtropical Moist Broadleaf Forests (Mangroves)
Maize	USA	Temperate broadleaf and mixed forests Temperate grasslands, savannas and shrubs lands Temperate Coniferous Forests Deserts and Xeric Shrublands Mediterranean Forests, Woodlands, and Scrub Flooded Grasslands and Savanna Tropical and Subtropical Coniferous Forests
Potato	Germany	Temperate Broadleaf and Mixed Forests
Maize	Italy	Mediterranean Forests, Woodlands, and Scrub Temperate Broadleaf and Mixed Forests (Temperate Coniferous Forests)
Sugar cane	Brazil	Tropical and Subtropical Moist Broadleaf Forests Tropical and subtropical grasslands, savannas, and shrublands Tropical and Subtropical Dry Broadleaf Forests Deserts and Xeric Shrublands (Mangroves)

Due to the aggregation of ecoregion characterisation factors into an average country characterisation factor it is not possible to pinpoint the cause of the relatively high characterisation factors for Italy. Wildfinder maps does not reveal particularly high species richness or endemism of species (as expected), so the reason is more likely to be found among the other model parameters e.g. a very small share of remaining habitat.

7.4.2.2 Regional impact assessment

Figure 23 shows the regional potential species loss assessed for the baseline feedstocks.

The regional impact assessment predicted the highest potential disappeared fraction of species (PDF) for maize cultivation in Italy, followed by much lower PDF for maize cultivation in USA and sugarcane in Thailand. The predicted PDF for maize cultivation in Italy was five times higher than for sugar cane in Thailand. Cultivation of potato in Germany resulted in the lowest predicted potential PDF ($2.31*10^{-12}$ Regional PDF/m²) of all the feedstocks, the same as for the global PDF, however cultivation of sugar cane in Brazil resulted in an almost equivalent low regional PDF ($2.50*10^{-12}$ Regional PDF/m²).

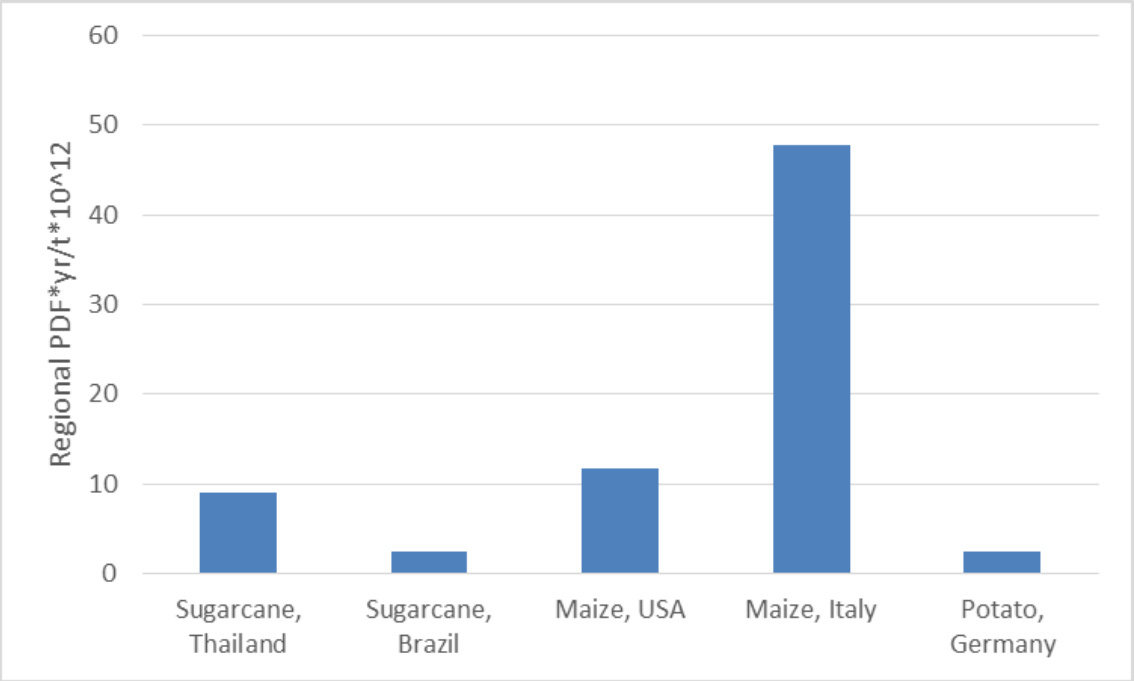


Figure 23. Regional potential species loss (PLS_{reg}) per tonne fresh weight feedstock expressed as Regional PDF*yr/t. Calculated using occupation average regional CFs for aggregated taxa for the case study baseline feedstocks with countries as geographical unit.

The regional impact assessment predicted the same order of country-feedstock impacts (highest to lowest predicted potential disappeared fraction) as the global impact assessment. The two different impact assessments thus pointed to cultivation of maize in Italy as a hot spot. The result however, shows that the relative difference in potential species loss related to cultivation in the different countries changes when adding the vulnerability weighting (from regional assessment shown in Figure 23 to global assessment shown in Figure 21). This reflects the difference in magnitude of global and regional characterization factors. The magnitude of the characterisation factors for Brazil and Thailand increases relative to the characterisation factors for the other countries when shifting from a regional to global view (from regional CFs Figure 24 to Figure 22), indicating the larger vulnerability of ecoregions[36] present in Brazil and Thailand compared to the other countries (on average).

[36] Due the aggregation it is not possible to know whether it is the vulnerability of few or many ecoregions that influences the result.

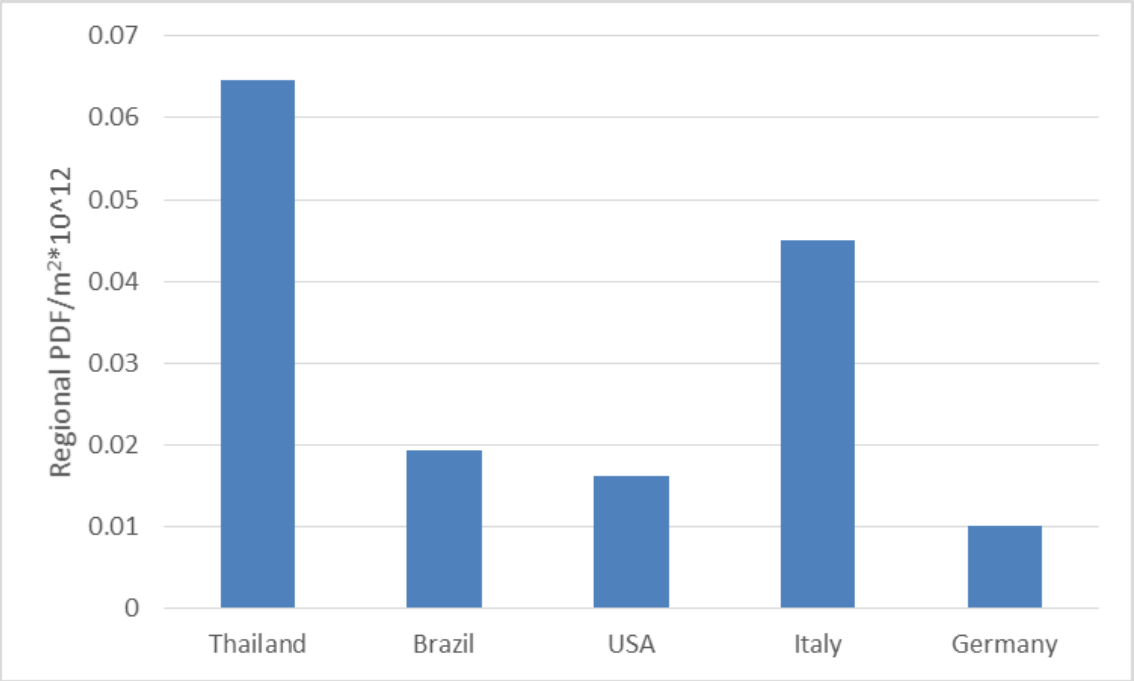


Figure 24. Regional country average occupation characterisation factors for 5 aggregated taxa (mammals, birds, plants, amphibians, reptiles). Unit: Regional PDF/m². Chaudhary et al. (2015)

In the country aggregated characterisation factor sets provided by Chaudhary et al. (2015) (See section 7.3.7) the taxonomic group 'plants' is included in the regional set but not the global set. This difference in inclusion of taxonomic groups may account for some of the differences seen when comparing global and regional characterisation factors (Figure 22 and Figure 24) and predicted biodiversity loss assessed with either one of them (Figure 21 and Figure 23). This is particularly important to keep in mind since Chaudhary et al. (2015) found that *'the regional CFs for each land use type were in general highest for most species rich taxa plants, followed by birds, mammals, and amphibians, and lowest for reptiles, which had low species richness per ecoregion.'* From this line of reasoning, plants are more likely to be the most impacted taxonomic group in the regional impact assessment.

7.4.2.3 Maize cultivated in Italy and USA

Both the regional and global impact assessments show a great difference between impacts related to cultivation of maize in Italy and USA.

Land requirements for cultivation of maize in USA and Italy differ considerably. Maize cultivated in the US requires ¾ of the land required in Italy. Furthermore, the global characterisation factor for Italy is twice as high as for the USA, expressing twice as high potential global PDF per m² land occupied. The result is three times higher predicted global biodiversity loss.

For the regional assessment a similar situation applies. The regional characterisation factor for Italy is three times as high as for the USA, expressing three times more potential regional PDF per m² of land occupied. The result is four times higher predicted regional biodiversity loss.

7.4.4.4 Sugar cane cultivated in Thailand and Brazil

Both the regional and global biodiversity assessments show a significant difference in predicted biodiversity loss between sugar cane grown in Thailand and Brazil (Figure 21 and Figure 23).

The predicted regional and global biodiversity loss was approximately 2.5 and 3.5 times higher for sugarcane grown in Thailand respectively compared to Brazil.

Land requirements are roughly the same for cultivation of sugar cane in Thailand (138.9 m^2*yr) and Brazil (129.0 m^2*yr). The difference in impact can thus be attributed to differences in magnitude of their characterisation factors. The global characterisation factor for Thailand is approximately twice as high as for Brazil and the regional factor is correspondingly approximately three times higher for Thailand than Brazil (Figure 22). It is only possible to speculate as to the reasons for this difference in characterisation factors for the two countries. The share of tropical biomes and species richness of ecoregions (included in the country average) may be higher for Thailand.

7.4.3 Decision support

Potential species loss predicted using country aggregated characterisation factors must be considered with great care and with respect for inherent uncertainty. In terms of biodiversity locality is of the essence and it is therefore a matter of discussion whether biodiversity loss can be meaningfully assessed based on country aggregated characterisation factors (see section 7.3.5). Since some of the same type of feedstocks are produced in different countries (maize and sugar cane) it is obvious to compare them, as has been done above, but it is important to emphasize that the assessment cannot support decisions about which country is best to produce the feedstocks in question at present. Specific locations of cultivation areas (ecoregion location) and knowledge about local agricultural practices must be considered.

The assessment with country aggregated characterisation factors puts the issue of biodiversity loss into the picture, where it otherwise would not have been considered due to lack of specific information. The results showing differences between countries can be used to get an idea of possible hot spots and may guide further investigation.

A biodiversity assessment carried out with the ReCiPe 2016 methodology is presented in Annex 4 and compared to the above impact assessment.

7.5 Biodiversity assessment at ecoregion level

In this section biodiversity assessment is carried out at ecoregion level for cultivation of Thai sugar cane and American maize used as baseline feedstocks to produce PLA (case studies 3, 4, 6).

7.5.1 Detailed spatial inventories for maize (USA) and sugar cane (Thailand)

In this project, information on specific locations of main supplying maize and sugar mills was obtained from the two leading producers of PLA. On this basis two sugar cane mills located in Thailand, Central and Northeast, and one maize mill located in the USA, state of Nebraska (close to the border of Iowa) are considered here.

Sugar mills are typically supplied by many smaller farms rather than a few large farms. The producer informed that one Thai sugar mill was supplied by as many as 4,000 farms[37]. In the USA farms are much larger than in Thailand, but the maize mills are typically also equivalently

[37] Personal communication May 2018.

larger operations. This results in a very large number of farm locations to include in both case studies. This investigation was deemed outside the scope of this study.[38]

Instead it is assumed that the supplying farmers are located in relative close proximity to the mills. For Thai sugar mills it is assumed that cultivation areas are located within a radius of 50 km of the mills.[39] Comparison between sugar cane cultivation areas and location of mills in Thailand supports the assumption that mills typically are located close to cultivation areas (Weerathaworn, 2017). Suppliers are small farms, and considering the bulk volume of sugar cane, and the infrastructure in rural areas of Thailand, it is also likely that it is not transported a long distance, which supports the assumption of a relatively small radius.

Supported by good infrastructure and given the large mill size, the trade area of the American mill is expected to be much larger for maize than for sugar cane. For the maize it was therefore assumed that cultivation areas would be located within a radius of 250 km of the mill location.

The affected ecoregions and their area shares within the chosen radius of the mill locations were determined by comparing and measuring on the biogeographic (interactive) WWF WildFinder maps based on Olson et al. (2001) available at Wildfinder.org (WWF Wildfinder, 2018).[40] This assessment formed the basis for determining what weights should be attributed to each ecoregion characterisation factor in a weighted characterisation factor average for the entire area affected by cultivation.

7.5.1.1 Identification of affected ecoregions - Sugar mill located in Northeast Thailand

If it is assumed that the Thai sugar farmers are located in the vicinity of the sugar mill, then cultivation of sugar cane takes place in either of two ecoregions. Ecoregions IM0121 (Luang Prabang montane rain forests) and IM0202 (Central Indochina dry forests) together cover the area in a radius of 50 km of the sugar mill. In almost all directions (except Northeast) these two ecoregions expand even further, so if sugar farms actually are located even further than 50-100 km from the sugar mill it is still most likely that the same ecoregions are affected. (WWF Wildfinder, 2018)

The ecoregions are located in different biomes[41]. IM0121 is located within the *'Tropical and Subtropical Moist Broadleaf Forests'* biome and IM0202 in the *'Tropical and Subtropical Dry Broadleaf Forests'*, both in the Indo-Malay Realm. (WWF Wildfinder, 2018)

The shares of ecoregions identified within a 50 km radius of the mill are presented in Table 49.

[38] Within the timeframe of this study it was not possible to initiate the required collaboration with the mills in question to obtain this information about exact farm locations. To handle such a magnitude of data skilfully would also require more advanced mapping tools than available for this study.

[39] An assumption of 50 km radius has been applied by the PLA producer for their own studies of other environmental impacts. Personal communication May 2018.

[40] The most accurate way to identify ecoregions by using a radius from a specific location and determining shares of affected ecoregions within this radius is by using a GIS based tool. Such a tool was however not available for current study, so this more manual approach was undertaken.

[41] Different subdivisions of the terrestrial world exit. The text here refers to the subdivisions of Olson et al. (2001). Hence ecoregions are categorised within 14 biomes and eight biogeographic realms according.

Table 49. The shares of identified ecoregions within 50 km radius of the sugar mill located in Northeast Thailand.

Weight of CF (%)	Ecoregion	Name of region	Biome
70	IM0202	Central Indochina dry forests	Tropical and Subtropical Dry Broadleaf Forests
30	IM0121	Luang Prabang montane rain forests	Tropical and Subtropical Moist Broadleaf Forests

7.5.1.2 Identification of affected ecoregions - Sugar mill located in Central Thailand

When the same approach is applied as for the other Thailand mill location then cultivation is likely to take place in either IM0202 (predominately) and IM0210, both located within the *'Tropical and Subtropical Dry Broadleaf Forests'* biome in the Indo-Malay Realm. (WWF Wildfinder maps, 2018)

In almost all directions (except South West) the IM0202 ecoregion expands even further, so in case sugar farms are located even further than 50-100 km from the sugar mill, it is still most likely that the IM0202 ecoregion is affected. (WWF Wildfinder maps, 2018)

The shares of ecoregions identified within a 50 km radius of the mill are presented in Table 50.

Table 50. The shares of identified ecoregions within 50 km radius of the sugar mill located in Central Thailand.

Weight of CF (%)	Ecoregion	Name of region	Biome
90	IM0202	Central Indochina dry forests	Tropical and Subtropical Dry Broadleaf Forests
10	IM0210	South eastern Indochina dry evergreen forests	Tropical and Subtropical Dry Broadleaf Forests

7.5.1.3 Identification of affected ecoregions - Maize mill located in the state of Nebraska in the USA

The states of Nebraska and Iowa are entirely located within the *'Temperate Grasslands, Savannas, and Shrublands'* biome in the Nearctic Realm. If cultivation of maize takes place within a 250 km radius of the maize mill, it will primarily take place in ecoregion NA0805 and secondarily in NA0803 and to a lesser degree in ecoregions NA0810 and NA0809 (see Table 51).

Table 51. The shares of identified ecoregions within 250 km radius of the maize mill located in Nebraska in USA.

Weight of CF (%)	Ecoregion	Name of region	Biome
75	NA0805	Central tall grasslands	Temperate Grasslands, Savannas, and Shrublands
15	NA0803	Central and Southern mixed grasslands	Temperate Grasslands, Savannas, and Shrublands
7	NA0810	Northern mixed grasslands	Temperate Grasslands, Savannas, and Shrublands
3	NA0809	Nebraska Sand Hills mixed grasslands	Temperate Grasslands, Savannas, and Shrublands

In the Eastern direction, ecoregion NA0805 expands further into Iowa, while regions NA0809 and NA0803 expand to the West. If cultivation areas actually are located further into Iowa, ecoregion NA0805 will be more affected, and if they are actually located further into Nebraska, either ecoregions NA0803 and NA0809 may become more affected. Expansion to the South and North may also affect ecoregions NA0807 and NA0805 respectively.

7.5.1.4 Inventory summary

Table 52 summarises the shares of identified ecoregions within a 250 km radius of the maize mill located in Nebraska; a 50 km radius of the sugar mill located in Central Thailand; and a 50 km radius of the sugar mill located in Northeast Thailand. Based on Table 52 relevant ecoregion characterisation factors were weighted into an average characterisation factor representing each of the specific cultivation areas in Northeast Thailand, Central Thailand and Nebraska. These are hereafter referred to as Northeast Thailand location CF, Central Thailand location CF, and Nebraska location CF.

Table 52. Spatial land use inventory for cultivation of maize (USA) and sugar cane (Thailand) with 'ecoregion' as the geographical unit. Inventory flows for land occupation is expressed as m^2*years per tonne (t) feedstock fresh weight (fw).

Feedstock	Origin	Affected ecoregion	Ecoregion share of cultivation area (%)	Land use type	Land occupation (m^2*year/t feedstock fw)
Sugar cane	Thailand; Northeast	IM0202	70	Permanent	139
		IM0121	30		
Sugar cane	Thailand; Central	IM0202	90	Permanent	139
		IM0210	10		
Maize	USA; Nebraska, Iowa	NA0805	75	Annual	721
		NA0803	15		
		NA0810	7		
		NA0809	3		

7.5.2 Impact assessment at ecoregion level

The characterisation factors specially constructed for the three cultivation areas, Nebraska, Northeast and Central Thailand were multiplied by the respective land use requirements resulting in a prediction of potential species loss associated with each specific cultivation area.

7.5.2.1 Global and regional potential species loss (taxa aggregated)

Figure 25 shows the global potential species loss assessed for the maize cultivation area in the USA and the sugar cane cultivation areas in Northeast and Central Thailand. The global impact assessment predicted the highest potential disappeared fraction of species (PDF) for sugar cane cultivation in Thailand with almost identical impact for the two different locations. The predicted biodiversity loss associated with maize cultivation is however within close range of the Thai sugar cane.

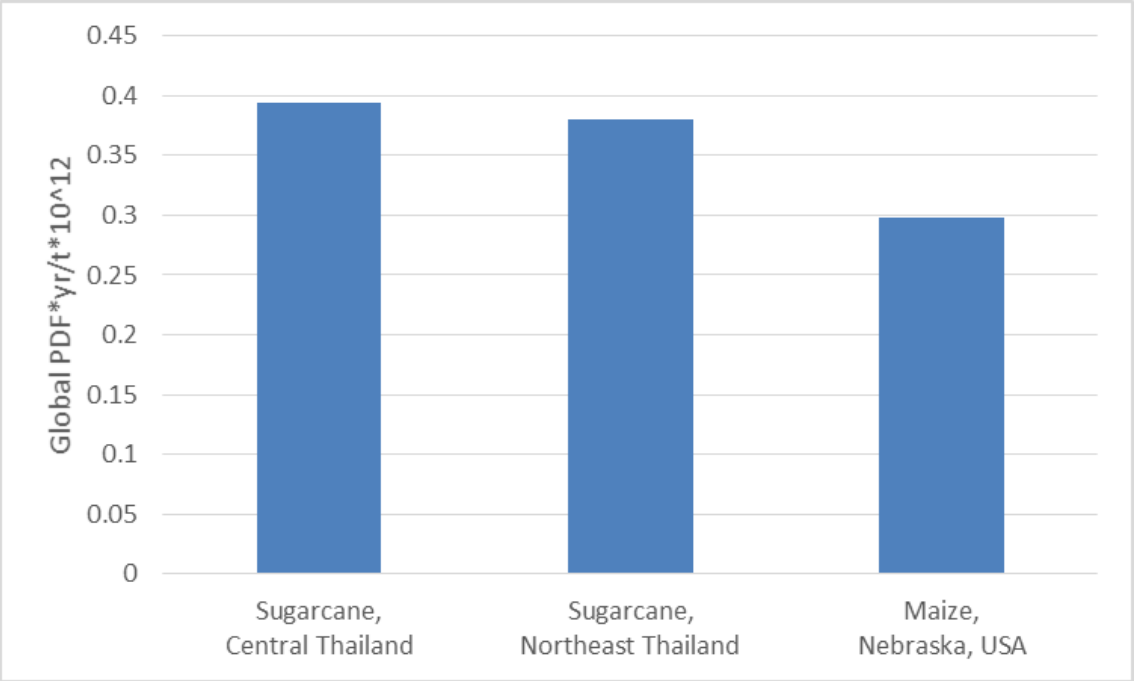


Figure 25. Global potential species loss (PLSglo) per tonne fresh weight feedstock expressed as Global PDF*yr/t. Calculated using occupation average global CFs for aggregated taxa for specific cultivation areas located in Thailand and USA with ecoregions as geographical unit

Figure 26 shows that the biodiversity loss predicted with regional impact assessment results in another ranking of the cultivation areas in terms of magnitude of impact. The highest PDF is here attributed to maize cultivation in the USA.

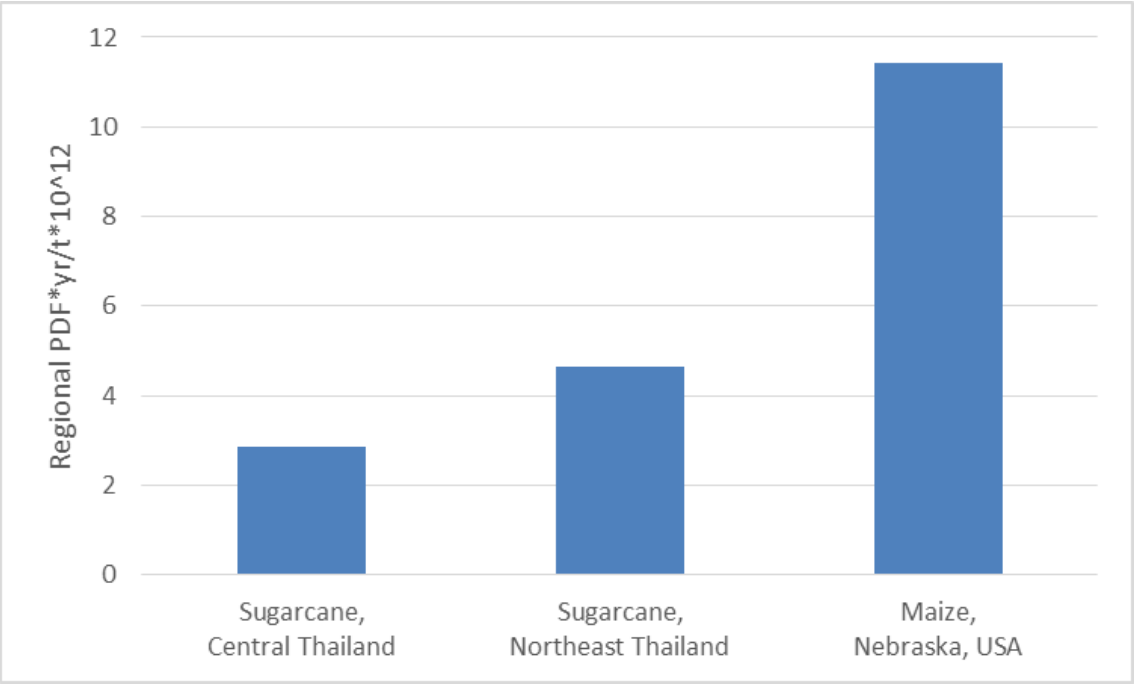


Figure 26. Regional potential species loss (PLSreg) per tonne fresh weight feedstock expressed as Regional PDF*yr/t. Calculated using CFs for aggregated taxa, occupation average regional CFs for specific cultivation areas located in Thailand and USA with ecoregions as geographical unit.

The Northeast Thailand location CF and Central Thailand location CF are approximately 6.5 and 7 times higher than the Nebraska location CF respectively, expressing a larger vulnerability of species in these cultivation areas. This vulnerability is reflected clearly in the predicted global potential PDF, placing less emphasis on the much higher land requirements for cultivation of maize.

In conclusion, from a global perspective sugar cane cultivation is associated with the highest potential species loss per tonne of feedstock while from a regional perspective maize cultivation is associated with the highest potential species loss per tonne of feedstock.

Again, it is important to keep in mind that the regional impact assessment includes plants as an additional taxa compared to the global impact assessment, which may account for some of the differences in results.

7.5.2.1.1 Comparison of results obtained with location specific and country average characterisation factors for Thailand and USA

Comparison of the global taxa aggregated Central Thailand location CF ($2.84*10^{-15}$) and Northeast location CF ($2.74*10^{-15}$) with the Thailand country CF ($4.45*10^{-15}$) shows that the country CF was 1.6 times higher than the location characterisation factors.

The USA CF ($8.92*10^{-16}$) was approximately twice as high as the global taxa aggregated Nebraska location CF ($4.14*10^{-16}$). In this particular case the country average characterisation factors were higher than the determined specific location characterisation factors. The data is however too limited to draw any general conclusions about the difference between location specific and country average characterisation factors.

Had location specific characterisation factors been applied for Thailand and USA instead of the country averages in the first assessment it would not have changed the overall conclusion reached in the country level assessment in section 7.4.2.1 (see Figure 27). Cultivation of maize in Italy remains the main hot spot in terms of potential species loss, but the potential species loss associated with cultivation of sugar cane in Thailand ends up being slightly higher than for cultivation of maize in the USA i.e. the opposite situation of what we see in Figure 21. This relatively small difference in impact associated with cultivation of the two feedstocks for PLA may though be within the uncertainty range of the characterisation factors.

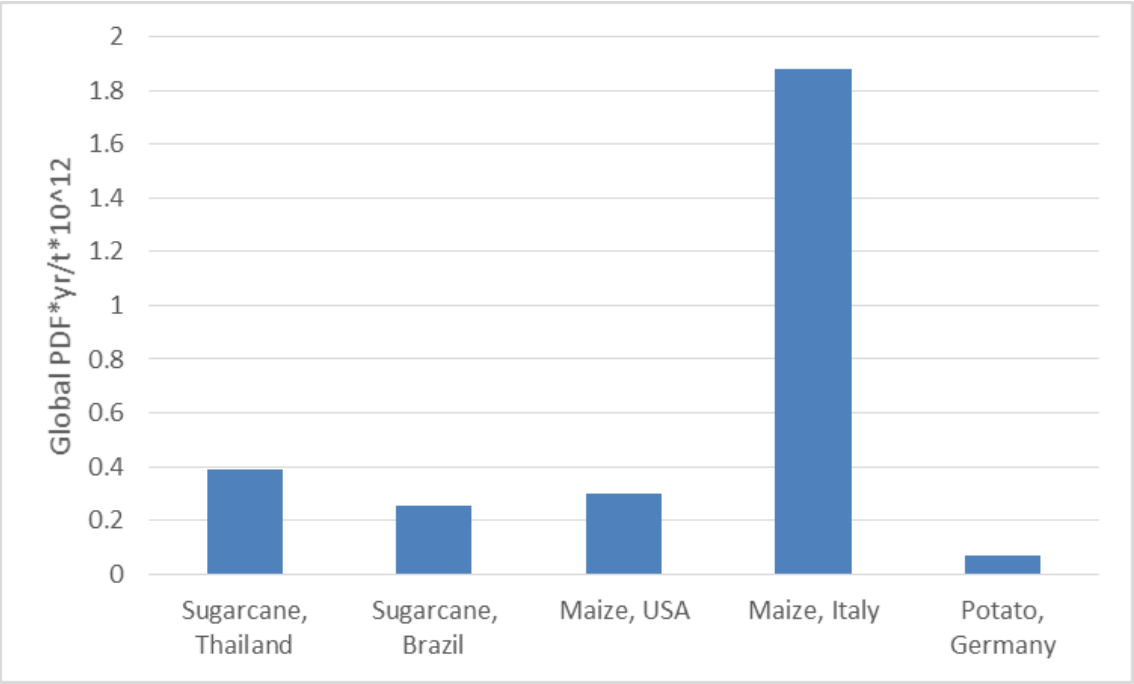


Figure 27. Combined country level assessment and ecoregion level assessment. Global potential species loss (PLS_{glo}) per tonne fresh weight feedstock expressed as Global PDF*yr/t. Calculated using occupation average global CFs for aggregated taxa for the case study baseline feedstocks sugar cane (Brazil), maize (Italy) and potato (Germany) and Central Thailand location CF and Nebraska location CF for sugar cane (Thailand) and maize (USA) respectively.

7.5.2.2 Global and regional potential species loss (taxa dis-aggregated)

Presenting the potential species loss taxa disaggregated allows for a better understanding of which taxa are most affected by cultivation at the different locations. Figure 28 shows that the global impact assessment predicted the highest potential species equivalent loss for the taxonomic group birds in all cultivation areas. For mammals, maize cultivation in the USA caused the greatest potential species equivalent loss, while for amphibians and reptiles sugar cane cultivation in Thailand caused the greatest potential species equivalent loss. Birds are thus the primary impacted taxonomic group for all three cultivation areas while the secondary varies.

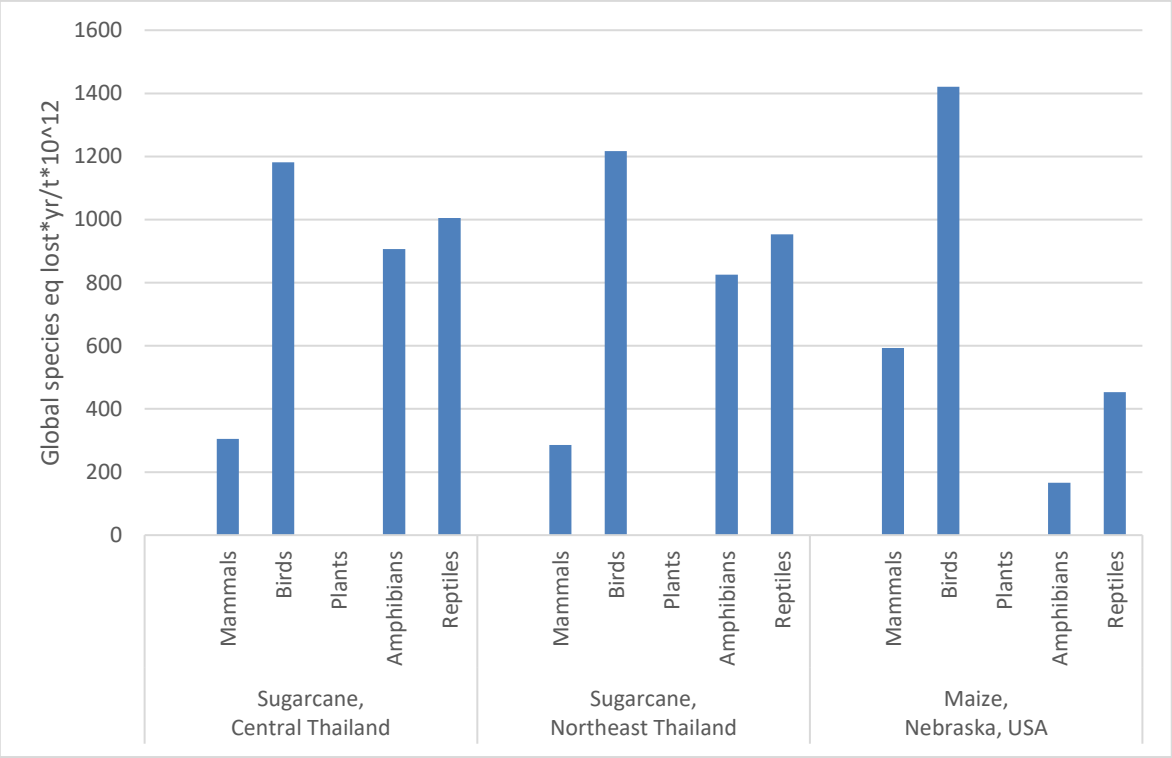


Figure 28. Global potential species loss (PLS_{glo}) per tonne fresh weight feedstock presented taxa dis-aggregated. Calculated using specially constructed characterisation factors for specific cultivation areas located in Thailand and USA, based on occupation average global CFs with ecoregions as geographical unit.

Not entirely unexpectedly, the taxa disaggregated view of the regional impact assessment shows that the highest predicted species loss was for plants in all cultivation areas, see Figure 29. The secondary impacted taxonomic group was birds for all cultivation areas. For mammals, amphibians and reptiles, maize cultivation in the USA caused the highest predicted species loss.

With the inclusion of plants in the regional impact assessment, there is not congruence between the main hot spot pointed out by the regional and global impact assessment in terms of the most impacted taxonomic group. When plants are taken out, the two different impact assessments both point to birds as the most impacted taxonomic group for all cultivation areas.

The regional impact assessment did not predict entirely the same order of impacts according to taxonomic group for the three cultivation areas as the global impact assessment when including plants. For the Thailand cultivation areas, the vulnerability weighting resulted in greater predicted impacts on amphibians and mammals, suggesting higher vulnerability of these two taxonomic groups in the ecoregions of the Thailand cultivation areas compared to the ecoregions of the Nebraska cultivation area.

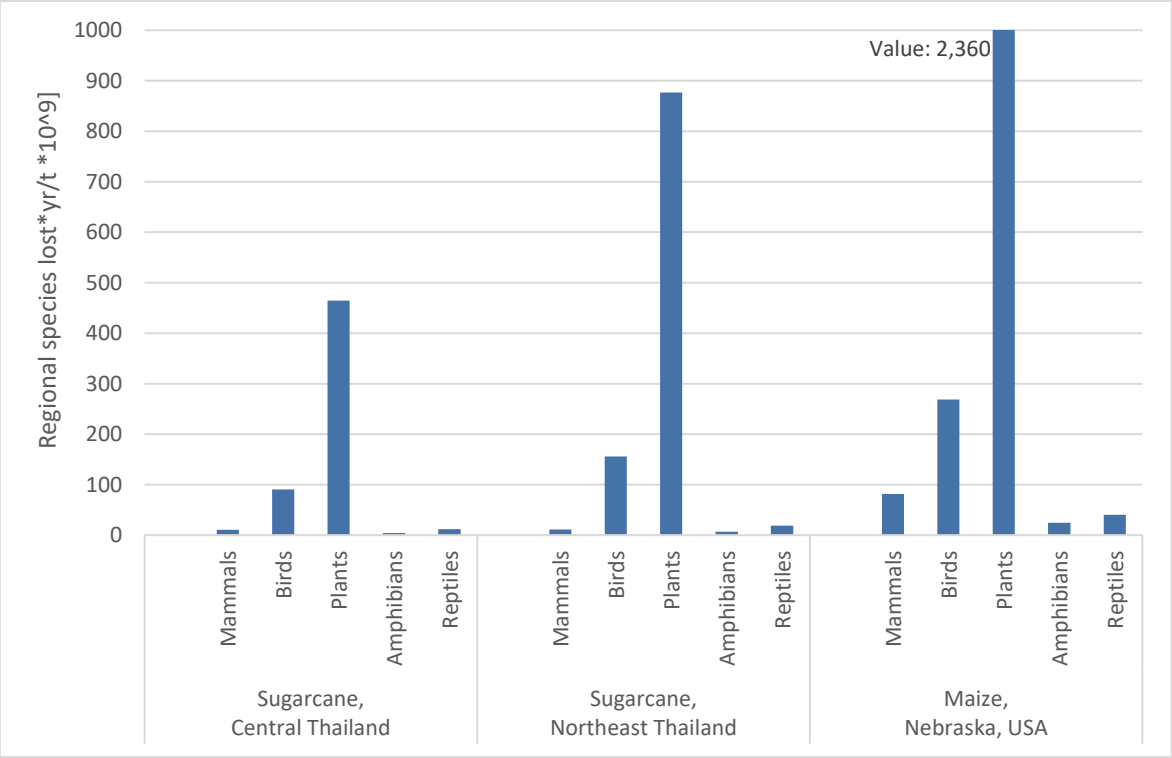


Figure 29. Regional potential species loss (PLS_{reg}) per tonne fresh weight feedstock presented taxa dis-aggregated for 5 taxa. Calculated using specially constructed characterisation factors for specific cultivation areas located in Thailand and USA on the basis of occupation average regional CFs with ecoregions as geographical unit.

7.5.2.3 Sugar cane cultivated in Thailand

From Figure 25 we learn that the global potential species loss is almost the same for the two Thai cultivation areas and from Figure 23 we see that the order of impacts according to taxonomic group predicted by the global impact assessment is the same for the two areas. The impact on birds is higher for the Northeast location, whereas the impacts on amphibians and reptiles are a little higher for the Central location. These differences may however be within the uncertainty range of the characterisation factors.

The results show that Ecoregion IM0202 'Central Indochina dry forests' is identified as the dominat ecoregion in both cultivation areas. The characterisation factor for IM0202 enters the Northeast Thailand location CF and Central Thailand location CF with a share of 70 % and 90 % respectively. Land use requirements are the same for the two Thailand locations (based on the land use assessment carried out in this project).

Wildfinder maps show that ecoregion IM0202 is a dominant ecoregion in Thailand. It is therefore also very likely to contribute a large area share of the country average i.e. the characterisation factor for this ecoregion is likely to be a large contributor to the country average characterisation factor for Thailand used earlier in this study (See comparison of location CF and country CF in section 7.7.2.1.1).

Figure 26 shows that the regional potential species loss is 1.5 times higher for the Northeast cultivation area compared to the cultivation area in central Thailand. The order of impacts according to taxonomic group predicted by the regional impact assessment is the same for the two areas (same situation as for the global impact assessment).

A regional biodiversity perspective suggests that it is slightly more optimal to cultivate sugar cane at the Central Thailand location. However, when adding the vulnerability weighting there is only a slight difference in predicted global potential species loss per tonne for the two cultivation areas and little difference in how the taxonomic groups are affected, making it less signifivant whether sugar cane is cultivated at Northeast or Central Thailand location.

7.5.2.4 Maize cultivated in Nebraska USA

Figure 30 shows how the different ecoregions represented in the Nebraska cultivation area contribute to the global potential species loss predicted for the area in terms of impacts on taxonomic groups. Ecoregions NA0805 'Central grasslands' and NA0803 'Central and Southern mixed grasslands' are the main ecoregions covering the cultivation area. The ecoregion characterisation factors for these two regions entered in to Nebraska location CF with a weight of 75 % and 15 % respectively, which explains the resemblance between the taxa impact profile for the Nebraska cultivation area (Figure 28) and that of ecoregion NA0805 presented in Figure 30[42].

The order of impacts according to taxonomic group predicted by the global impact assessment is the same for the two ecoregions, but the relative difference between impacts of the taxonomic groups in the ecoregions are not entirely the same. For NA0805 the difference between how much birds and mammals are impacted is more pronounced than for NA0803.

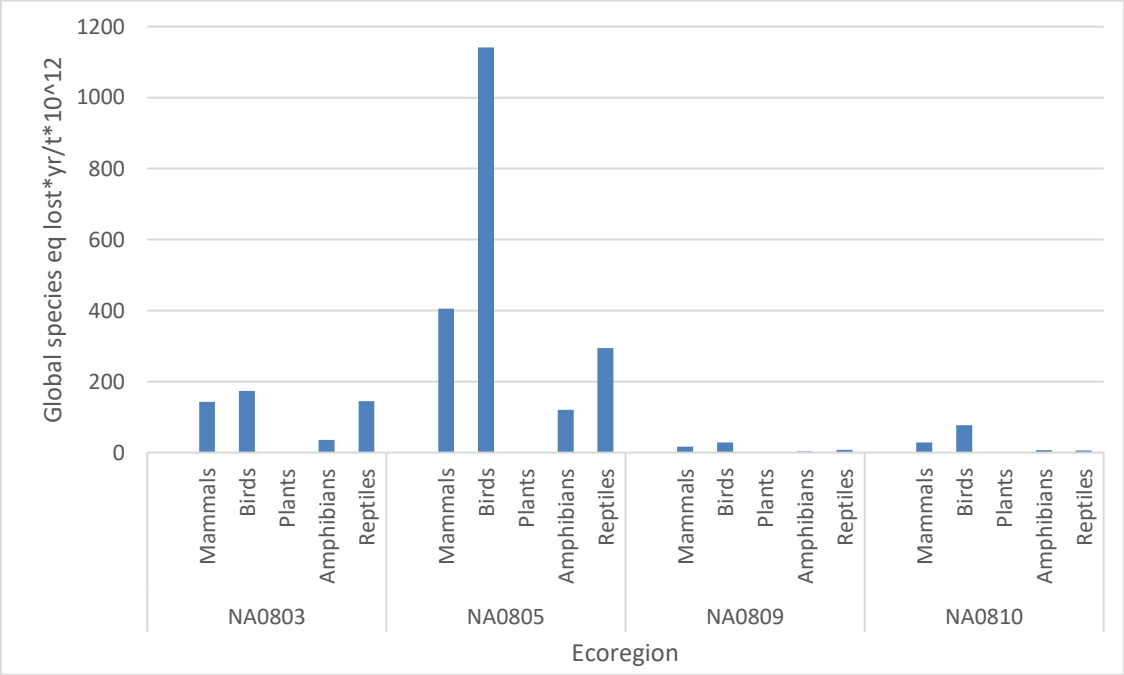


Figure 30. Global potential species loss (PLSglo) per tonne fresh weight feedstock presented taxa dis-aggregated for ecoregions of the Nebraska cultivation area. Calculated using Nebraska location CF on the basis of occupation average global CFs with ecoregions as geographical unit.

[42] A similar resemblance between the taxa impact profile for the Nebraska cultivation area and that of ecoregion NA0805 presents itself for the regional impact assessment. Therefore, for regional assessment refer to the taxa impact profile for the Nebraska cultivation area presented in Figure 29.

7.5.3 Decision support

In this biodiversity assessment locality is narrowed down to a specific radius of mills processing feedstocks, and relevant ecoregions have been identified and ecoregion characterisation factors applied. This provides a more accurate picture of the ecosystem pressure related to cultivation compared to the previous assessment made using country average characterisation factors. However, before entering decision-making about whether to use one or the other feedstock for production of PLA more in-depth investigations must be performed. Local conditions influencing biodiversity not considered by the biodiversity assessment performed here and local agricultural practices must be considered. Precautions to protect biodiversity may already have been put in place and agricultural management may be different than assumed.

Furthermore, uncertainty of applied characterisation factors, land use requirements and identification of ecoregions should be further scrutinised, and the known limitations of the Chaudhary 2015 methodology in terms of predicting biodiversity loss should be considered as well.

7.6 Methodology reflections

Biodiversity is a relatively new topic in LCA and the Chaudhary 2015 methodology applied here is still very young and has not yet been tested in a wide range of product systems, regions and application areas. As such, UNEP-SETAC recommends the method *ad interim, and recommends* that it is not used to support comparative assertions yet (UNEP-SETAC, 2016).

The Chaudhary 2015 methodology provides a quantitative impact assessment, which offers a truly unique and valuable global perspective on biodiversity. In this respect, the Chaudhary 2015 methodology is very promising. A major current drawback however, is that the methodology cannot take differentiated agricultural management practices' effect on biodiversity into account. It is therefore mainly applicable for hot spot analysis at its current stage of development.[43]

In conclusion, the methodology is not yet mature enough as a standalone to support political decision-making, but must be supplemented with other, perhaps qualitative, methods for the time being.

If the methodology in its existing form is accepted for comparative assertions in the future, then more detailed information about the location of cultivation areas will improve the assessment significantly as described in the assessment carried out with eco-region characterisation factors in this project. Locality data will also facilitate qualitative assessment of agricultural practices.

A recent report compiled by experts on biodiversity assessments from academic institutions, civil society organizations, and the private sector on existing methodologies and tools for biodiversity metrics concludes that nearly all of the assessed methodologies and tools are still under development (Neveux et al., 2018). Given this situation there is thus not a clear (quantitative) alternative to the demonstrated Chaudhary 2015 methodology.

[43] Other known methodological challenges of quantifying land use impact on biodiversity in LCA in general and shortcomings of the Chaudhary 2015 methodology are further discussed in (Chaudhary et. al, 2015) (UNEP-SETAC, 2016).

8 Socio-economic assessment

8.1 Scope of assessment

Bio-based products are assessed by many to have beneficial socio-economic properties. The European Commission (2009)[44] emphasises socio-economic benefits such as less dependency on limited and increasingly expensive fossil resources, and the support to rural development. Furthermore, it points to increased industrial competitiveness through innovative eco-efficient bio-based products. This latter benefit has also been emphasised by the JRC (2015)[45] with bio-based sectors showing growth and high labour productivity gains. Similarly, European Bioplastics (2017)[46] highlights the innovative properties of bioplastics and their prospects for creating jobs and growth in Europe.

The seven-case studies on bio-based products – for which LCAs are presented in detail in the following chapters – have been assessed according to socio-economic properties similar to those just mentioned.

The assessment method applied is a simple check of whether a bio-based product has positive or negative socio-economic characteristics that should be noted and possibly be responded to when promoting it.

Assessments are made about how the bio-based products perform – in comparison with the alternative fossil fuel-based products – with respect to:

- Job standards and quality – i.e. working arrangements in the form of safety and health aspects, working time, job security, quality of work contracts, and/or training/educational/innovation content and career perspectives;
- Job creation – i.e. labour intensity of production process;
- Social cohesion and rural development – i.e. opportunities for jobs and income for the poorest people in rural areas.

These three indicators have been selected through a delimitation process, where a gross list of possible socio-economic impacts was reduced to cover issues also covered by the LCAs. There has therefore been no attempt to assess health impacts, the reason being that the LCA do not, with sufficient certainty, find differences in toxicity between the bio-based products and the fossil fuel-based alternatives.

Finally, the assessments of the three socio-economic indicators are done for the value chain of the products, i.e. with a distinction between the input step (i.e. the production of feedstock), the production step (i.e. the production of the bio-based product), and the EoL step (i.e. the disposal of the bio-based product).

8.2 Results of assessment

[44] European Commission (2009), Taking bio-based from promise to market, https://publications.europa.eu/en/publication-detail/-/publication/8f3bff72-a95c-4fc8-a581-89bd591eaf61/language-en

[45] Ronzon et. al., 2017 http://publications.jrc.ec.europa.eu/repository/bitstream/JRC108733/jrc_researchbrief_bioeconomics_2015.pdf

[46] European Bioplastics, 2017 http://docs.european-bioplastics.org/publications/EUBP_Facts_and_figures.pdf

Table 53 summarises the assessments by socio-economic indicator for each of the seven bio-based products.

A first observation across the seven products is that the socio-economic properties differ most between a bio-based product and a fossil fuel-based alternative with respect to the input step. However, this difference depends on whether the feedstock is primary product-based – e.g. maize or sugar cane, or whether it is vegetable waste oil-based. If the former, there is a risk that the feedstock production takes place under inferior working conditions (job standards and quality) in the poorer part of the world

Furthermore, job creation properties – here in the context of social cohesion and rural development – are particularly good if the feedstock is primary product-based.

For the production step of the bio-based products, another general observation is that the training/educational content and thus job quality is often higher for the bio-based products than for the fossil fuel-based alternatives. In particular, vegetable waste oil-based production processes are considered to be innovative. Furthermore, there is a tendency that (parts of) the bio-based production processes are labour intensive. Finally, job creation within some of the bio-based sectors are encouraged by policy/legislative measures that favour the products vis-à-vis fossil fuel-based products.

There are for some bio-based products no EoL issues compared with the fossil fuel-based alternatives. For example, beverage bottles (CS1) and single-use cups for cold drinks (CS3) are chemically identical to fossil fuel-based products. For others there can be some difference. For example, if there is a requirement to collect fossil fuel-based single-use clips (CS3), it will require less labour input if the single-use clips are biodegradable. Similarly, biodegradable agricultural mulching films (CS5) save work processes if they replace films that need to be collected after use.

In conclusion, the socio-economic assessment does overall not lead to socio-economic concerns from promoting the case study bio-based products. There seems to be a tendency for these products to have better socio-economic properties than their fossil fuel-based alternatives.

Table 53. Socio-economic assessments for case study bio-based products.

Case study / socio-economic indicator	Job standards and quality	Job creation	Social cohesion and rural development
1. Beverage bottles	Primary product-based, e.g. sugarcane in Brazil, implying feedstock production under worse working conditions than for fossil fuels (in the EU) If feedstock is vegetable waste oil-based no significant difference. Training/educational content is relatively high in an innovative bio-based production process. Since a bio-based bottle is chemically identical to a petrochemical bottle, there are no EoL issues.	Primary product-based feedstock production is relatively labour intensive. If feedstock is vegetable waste oil-based there is little or no job creation, while there may be lost fossil fuel jobs. The resin production part of the bio-based production process is relatively labour intensive, while the remaining part of the production process do not differ much from the fossil fuel-based alternatives with respect to labour intensity. There are no EoL issues.	Primary product-based feedstock production creates jobs and income in rural areas often characterised as highunemployment areas Hence, if based on vegetable waste oil there is no impact. And, there is no impact (difference from a fossil fuel-based product) for the remaining part of the value chain.
2. Single-use clips	The feedstock is typically potato (or waste from potato)/maize from the US, where production is highly mechanised. Hence, no major difference in job standards and quality compared with fossil fuel production. However, the bio-based production process is more innovative – leading to jobs with higher educational content. Re. EoL: no collection of clips as they are biodegradable.	Bio-based feedstock production is relatively more labour intensive, while the use of waste from potato may be less labour intensive. Jobs are lost in the collection of clips (if the fossil fuel-based alternatives are collected).	Benefits for rural areas (in the US).
3. Single-use cups for cold drinks	Feedstock is either maize or sugar cane, which may imply worse working conditions than for fossil fuels (in the EU). Relatively innovative production process. Since a bio-based cup is chemically identical to a petrochemical bottle, there are no EoL issues.	Bio-based feedstock production is relatively more labour intensive. There are no EoL issues.	Benefits for rural areas (in Brazil and the US).

4. Single-use cutlery	The feedstock is typically potato (or waste from potato)/maize from the US, where production is highly mechanised. Hence, no major difference in job standards and quality compared with fossil fuel production. However, the bio-based production process is more innovative – leading to jobs with higher educational content. Re. EoL: there may be some waste management issues in the future.	Bio-based feedstock production is relatively more labour intensive, while the use of waste from potato may be less labour intensive. The EoL issue seems not to have be fully solve. For example, it may lead to sorting jobs at biogas plants.	Benefits for rural areas (in the US).
5. Agricultural mulching films	The feedstock is typically potato (or waste from potato)/maize from the US, where production is highly mechanised. Hence, no major difference in job standards and quality. However, the bio-based production process, hereunder the polymer blending process, is more innovative – leading to jobs with higher educational content. Re. EoL: no collection of films as they are biodegradable.	Bio-based feedstock production is relatively more labour intensive, while the use of waste from potato may be less labour intensive. Additional innovative jobs are created within polymer blending process that also involves new infrastructure investments. Re. End of life: no collection of films implies fewer jobs compared with the fossil fuel alternative.	Benefits for rural areas (in the US).
6. Food packaging films	Feedstock is either maize or sugar cane, which may imply worse working conditions than for fossil fuels (in the EU). Relatively innovative production process.	Bio-based feedstock production is relatively more labour intensive.	Benefits for rural areas (in Brazil and the US).
7. Single use carrier bags	Bio-based feedstock is either maize or sugar cane, which may imply worse working conditions than for fossil fuels (in the EU). Relatively innovative blending production process (i.e. not purely bio-based).	Bio-based feedstock production is relatively more labour intensive. Job creation is encouraged by policy/legislative measures that favour the products vis-à-vis fossil fuel-based alternative The EoL treatment may lead to sorting jobs at biogas plants.	Benefits for rural areas (in Brazil and the US).

CASE STUDY 1: BEVERAGE BOTTLES

1 Goal and scope definition

1.1 Goal and background

The goal of this case study is to assess the environmental profiles of selected pathways for bio-based PET (polyethylene terephthalate) beverage bottles production and compare them with the fossil-based counterpart.

PET is produced from monoethylene glycol (MEG), which accounts for approximately 30 % by weight, and petrochemical purified terephthalic acid (PTA), which accounts for approximately 70 % by weight. Figure 31 illustrates the transesterification reaction of PET synthesis.

The bio-based PET bottle is chemically identical with the petrochemical PET bottle. The essential difference lies in the MEG. Bio-MEG is used in bio-based PET whereas fossil-based MEG is used in the petrochemical PET. For both types of MEG, the PTA is fossil-based. Therefore, bio-based PET has approximately 30% biogenic content.[47]

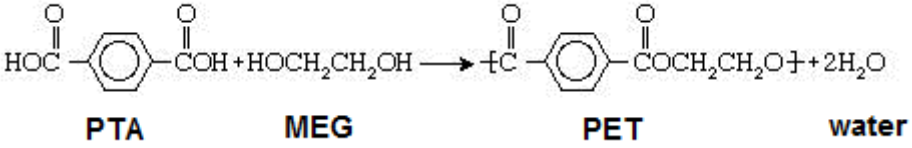


Figure 31. PET production from MEG and PTA.

Bio-MEG is produced from ethanol. Ethanol is produced from fermentation of sugars obtained from biomass, followed by distillation. Ethanol is then catalytically dehydrated to ethylene. Ethylene oxide is obtained by oxidation of ethylene and it is then hydrolyzed to MEG. For fossil-based MEG, ethylene is directly obtained from steam cracking of petroleum fractions such as naphtha. The ethylene obtained is converted first to ethylene oxide and then to MEG.

The PTA part is fossil-based and is produced from paraxylene. Xylenes are obtained from catalytic reforming of petroleum fractions such as naphtha followed by aromatics separation (PlasticsEurope, 2017). Paraxylene isomer is separated and catalytically oxidised to produce terephthalic acid, in the presence of an acetic acid solvent. The crude terephthalic acid attained is purified to produce PTA. So far, no commercial bio-based production of paraxylene exists (Tsiropoulos et al., 2015).

PET is produced by esterification of fossil-based PTA with either bio-MEG or fossil-based MEG. After this, melt polymerization takes place to produce amorphous PET. A subsequent solid-state polymerization is required to achieve bottle grade PET (PlasticsEurope, 2017). The bottle grade PET then can be used to produce bottles by stretch blow moulding.

Bio-based PET shows fast market growth and represents approximately 23 % of the global production capacity of bioplastics (Aeschelmann et al., 2017). Bio-based PET is predominantly used for bottle packaging. The use of bio-based PET for bottles accounts for more than 85 % of the bio-based PET used globally with the rest finding use in technical (e.g. automotive,

[47] It is technically possible to produce 100 % bio-based PET using bio-MEG and bio-PTA. This is not investigated in this project since bio-PTA is currently not commercially produced.

electronics) applications (Grand View Research, 2017). The global demand for PET bottles remains strong with over 4 % annual growth (Euromonitor International, 2015).

1.2 Scope

The geographical scope of the case study is Europe for the purchase, use and disposal of the PET bottle. However, the study considers all processes occurring outside Europe prior to purchase (e.g. feedstock cultivation and harvesting, conversion processes). For the baseline, the cultivation of sugarcane and its conversion to ethanol takes place in Brazil. For the alternative feedstock systems, wheat, maize, sugar beet and lignocellulosic feedstocks are cultivated and converted to ethanol in Europe. For the ethanol to bio-MEG conversion, only one company produces bio-MEG on a large industrial scale, and its production takes place in India. The PTA and bottle grade PET production and stretch blow moulding processes occur in Europe. Further explanation of the choice of geography is given in sections 1.4 (product systems) and 2.1 (inventory analysis).

The temporal scope of the case study is current production (2017-2018) with relevant developments foreseen for the short-term future (5-10 years). The technical scope is the range of technologies at a technology readiness level higher than 8. Currently only sugarcane-based ethanol is used for MEG conversion. Two alternative feedstocks for ethanol are also considered but the conversion to MEG is based on hypothetical scenarios. The production of ethanol from lignocellulosic feedstock has recently become commercialised and more development is expected in the near future.

1.3 Function and functional unit

The function of a beverage bottle is to hold a certain amount of beverage. Packaging of water represents the most important application for PET bottles, representing over 40 % of all soft drink volume, and its share is forecasted to increase further (Euromonitor International, 2016). In 2015, ~ 41 % of all PET bottles used for packaging water had a volume of 0.5 L or less, ~ 33 % had a volume between 0.5 L and 3 L volume. The remaining quarter had a volume greater than 3 L (Nestlé Waters, 2015).

Accordingly, the functional unit of this case is defined as:

- Packaging of water in 100 0.5 L bottles providing a shelf life of at least 9 months, purchased and consumed in Europe.

The average weight of a 0.5 L water bottle is about 10 g (PETRA, 2017). As a result, this functional unit roughly equals 1 kg of PET bottles. It is important to note that this is nearly half of what a typical 0.5 L bottle weighed in 2000 (18.9 g) and there are ongoing efforts to further reduce bottle weight (Nestlé Waters, 2015). The mass associated with the functional unit is identical for both the bio-based and the petrochemical PET bottles since they are chemically identical.

1.4 Product systems

Three bio-based PET product systems are identified in this study. The baseline is production of bio-based PET using sugarcane from Brazil. Although ethanol can be produced from various biomass feedstocks, sugarcane is currently the only feedstock used in the production of ethanol to produce bio-MEG. A company located in India sources ethanol produced from Brazilian sugarcane since Brazil is the world's largest sugarcane and sugarcane-based ethanol producer. The baseline therefore reflects the status-quo.

Two alternative hypothetical product systems are considered. The first feedstock alternative considers the use of European ethanol crops. Based on a 2016 statistical report, in Europe approximately 90 % of fuel ethanol is made from maize, wheat and sugar beet (Epure, 2017).

The combined market mix is applied where 36 % of ethanol is produced from maize, 37 % from wheat and 27 % from sugar beet.

The second alternative feedstock considered is lignocellulosics. There is ongoing and fast development of this second-general ethanol for commercial scale production. The three product systems, the feedstock used, and their source are summarised in Table 54. The bio-based product systems are compared with a reference system of petrochemical PET bottles.

The three product systems, the feedstock used, and their source are summarised in Table 54. The bio-based product systems are compared with the reference system of petrochemical PET bottles.

Table 54. Product systems considered for the case study *Beverage bottles.*

Product system	Feedstock for ethanol	Source of feedstock
Baseline	Sugarcane	Brazil
Alternative 1	36 % maize, 37 % wheat and 27 % sugar beet	Europe, weighted average of dominant crop-producing Member States
Alternative 2	Lignocellulosic (wheat straw)	Europe
Reference system	n/a	Fossil fuel-based, average EU

1.5 System boundaries

A cradle-to-grave approach is taken including the life cycle stages of feedstock production, manufacturing and EoL. The consumer use phase is excluded from the analysis which is the same for both product systems and has negligible impact. Thus, the life cycle can be divided into cradle-to-gate and EoL.

2 Life cycle inventory analysis

2.1 Process description, data, assumptions and multifunctionality

2.1.1 Baseline: bio-based PET from Brazilian sugarcane

As of 2018, only one company produces bio-MEG on a large industrial scale, and its production takes place in India. The production route of this company is used in the baseline. The company uses ethanol produced from sugarcane in Brazil. Ethanol produced from sugarcane molasses in India can also be used in the production of bio-MEG (Tsiropoulos et al., 2014). For this LCA, it is assumed that 100 % ethanol is produced from Brazilian sugarcane which is converted to bio-MEG in India.

The process flow diagram of the baseline system is shown in Figure 32. The processes involved can be divided into the following key unit processes:

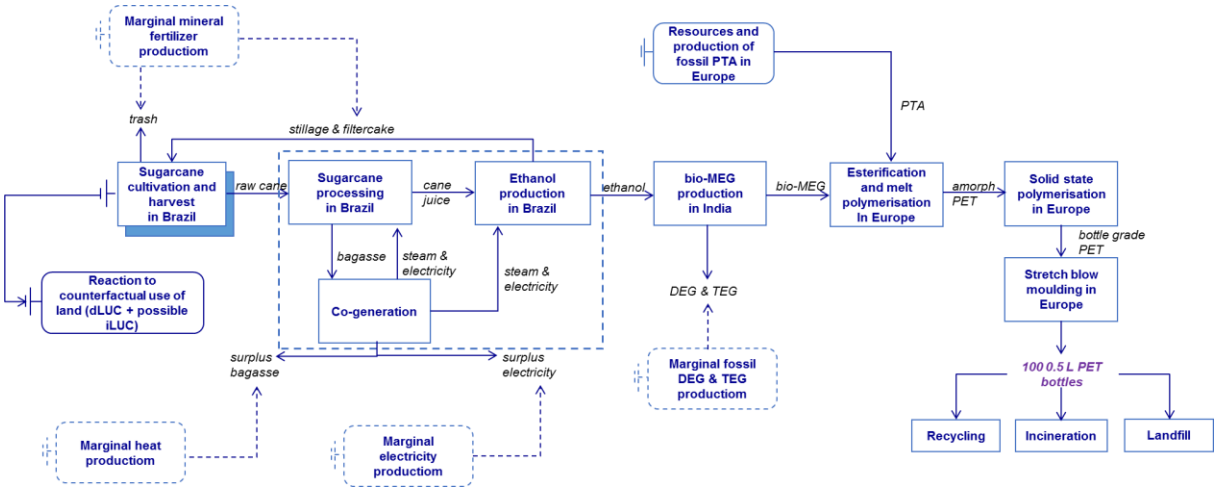


Figure 32. Process flow diagram of baseline product system: bio-based PET production from sugarcane.

Approximately 90 % of the sugarcane production in Brazil originates from the sugarcane cultivated in the south-central region (Seabra et al., 2011). The life cycle inventory represents 168 sugarcane mills in the region (Macedo et al., 2008; Seabra et al., 2011; Tsiropoulos et al., 2014). This is the latest, most comprehensive and reliable data obtained from the sugarcane technology center (CTC, 2009). However, this latest available data reflects the 2008 situation when 65 % pre-burned harvesting was practiced (even with machine harvesting). In other words, in 65 % of the harvested area cane waste (the tops and leaves of the sugarcane plant) was burned in the field during harvesting. The pre-burning practices will be phased out by 2031 based on state law (São Paulo State Law n. 11241/02) (Carvalho, 2016) and was phased out by 2017 by industrial protocol (Tsiropoulos et al., 2014). [48]

2.1.1.1 Sugarcane processing and ethanol production (Brazil)

After harvest, sugarcane is transported to ethanol mills by truck. Cane juice is extracted and fermented to produce 95 vol. % ethanol. The cane extraction residue, bagasse, is burned in a

[48] The straw left on the field provides agronomic gains and yield improvements (Carvalho et al., 2017). In the future, a fraction of the cane waste could be potentially recovered to produce e.g. second-generation ethanol. However, recovery technologies are still not fully established.

co-generation unit to produce steam and electricity to fuel the sugarcane processing and ethanol production processes. The surplus electricity is sold to the grid. The residues of the filter cake and stillage are applied as fertilisers on sugarcanes field to reduce the need for mineral fertilisers (Tsiropoulos et al., 2014).

This unit process combines sugar processing, ethanol production and a co-generation system as shown with a dotted box in Figure 32. Data is obtained from the same literature sources as sugarcane cultivation and harvest (Macedo et al., 2008; Seabra et al., 2011; Tsiropoulos et al., 2014), which represents industrial production in south-central Brazil in 2008.

2.1.1.2 Ethanol transportation from Brazil to India

The bioethanol produced in Brazil is transported to India to be converted into mono-ethylene glycol (MEG). Ethanol is transported by lorry from the ethanol production plant to the Rio Grande port. Then it is transported by bulk chemical carrier to Mumbai port. From there, it is transported by lorry to the bio-MEG production plant. Transportation distances are based on Tsiropoulos et al., 2015.

2.1.1.3 Bio-MEG production (India)

For conversion of ethanol to bio-MEG, confidential, primary data from the producer is obtained (Tsiropoulos et al., 2015). The process involves ethanol dehydration to ethylene, followed by ethylene oxidation and hydration of the resulting ethylene oxide to produce ethylene glycols. Small amounts of heavier glycols (diethylene glycol (DEG) and triethylene glycol (TEG)) are produced as by products. MEG is separated from DEG and TEG by distillation.

2.1.1.4 Transportation of bio-MEG from India to Europe

Bio-MEG produced in India is then sent to Europe. It is first transported by lorry to Mumbai port and then by transoceanic ship to the Rotterdam port.

2.1.1.5 PTA production (Europe)

Fossil-based PTA production takes place in Europe. Industry data 2.0 Terephthalic acid, purified, at plant/RER is used which is based on PlasticsEurope's PTA eco-profile (PlasticsEurope, 2014). The climate change impact potential and NREU impact categories are updated with the breakdown given in the latest PlasticsEurope PET eco-profile (PlasticsEurope, 2017). This process is the same for bio-based and petrochemical PET so the same data is used for the petrochemical reference as described in section 2.1.5.

2.1.1.6 Polymerization (Europe)

MEG is reacted with fossil-based PTA in Europe to produce PET. Amorphous PET is obtained via melt polymerisation. Amorphous PET is further upgraded into bottle grade PET via solid state polymerization. The impact of polymerization is calculated using Ecoinvent 3.3 *Polyethylene terephthalate, granulate, bottle grade {RER}| production | Alloc Def, U* by setting MEG and PTA inputs to 0 to reflect only the impacts coming from the esterification and polymerization processes. The climate change and NREU impact categories are updated with the breakdown given in the latest PlasticsEurope PET eco-profile (PlasticsEurope, 2017). These processes are the same for bio-based and petrochemical PET, so the same data is used for the petrochemical reference.

2.1.1.7 Stretch blow moulding (Europe)

Bottle grade PET is stretch blow moulded to produce PET bottles. The key activity level data is electricity consumption. We compared literature data from different sources and a range from 1.8-9.1 MJe/kg was found where the higher end is based on Ecoinvent 3.3 and the lower end

based on a survey conducted in literature showing range of 1.8 to 6.1 MJe/kg. Ecoinvent 3.3 Stretch blow moulding {RER}| production | Alloc Def, U is used with electricity consumption modified based on conservative values from the literature survey of 6.1 MJe/kg (CalRecycle, 2011).

The foreground data sources used for the baseline bio-based PET bottles are summarised in Table 55. For background data, e.g. grid electricity and heat, other utilities and production of chemicals and materials, Ecoinvent 3.3 is used.

Table 55. Data used in the bio-based PET bottle: the baseline.

Processes	Data sources, foreground	Comments
Sugarcane cultivation	Tsiropoulos et al., 2014 & Seabra et al., 2011	Represents south-central Brazil 2008, based on sugarcane technology centre CTC database (CTC, 2009), with own modification
Bioethanol production	Tsiropoulos et al., 2014 & Seabra et al., 2011	Represents south-central Brazil 2008, based on sugarcane technology centre CTC database (CTC, 2009)
Bio-MEG production	Tsiropoulos et al., 2015	Represents production in India in 2011 based on data from industrial producer
PTA production (fossil)	PlasticsEurope, 2017 & Industry data 2.0	Represents European average, reference year of 2015[a]
Polymerisation to bottle grade	PlasticsEurope, 2017 & Ecoinvent 3.3	Represents European average, reference year of 2015[a]
Stretch blow moulding	CalRecycle 2011 & Ecoinvent 3.3	Modified Ecoinvent 3.3 process. Electricity consumption adapted based on conservative value from literature survey (CalRecycle, 2011)
a Only Climate change and NREU are obtained from PlasticsEurope (PlasticsEurope, 2017), for other impact categories Ecoinvent 3.3 and industry data 2.0 databases are used. Further description is provided in the text.		

2.1.1.8 Biogenic carbon stored in the bottle

From cradle to factory gate, biogenic carbon removal is considered in the carbon accounting (e.g. as defined by PAS 2050). The climate change from cradle-to-gate is calculated as the sum of fossil GHG emissions and biogenic methane emissions minus the biogenic carbon embedded in the product. In the EoL stage the carbon is emitted again and a net balance is calculated for the cradle-to-grave system. Based on the molecular structure of PET and the biogenic carbon content coming from MEG (Figure 31) the biogenic carbon embedded in the bio-based PET bottle is calculated as 0.45 kg CO_2/kg for bio-based PET.

2.1.1.9 Multifunctionality

As described above there are several unit processes where multiple outputs are produced. To handle these multifunctional processes, as explained in the Methodology document, based on ISO 14044 and PEFCR guidance documents, system expansion is applied to avoid allocation in the first place.

Sugarcane processing and ethanol production systems involve co-production of filtercake, stillage, surplus electricity and surplus bagasse (Figure 32). Filtercake and stillage are returned to the field, thus consumed within the system boundaries.

For the surplus bagasse (8.7 kg/t sugarcane), the system is expanded to include heat production from bagasse (with 79 % efficiency) (Seabra et al., 2011). This enables substitution of marginal production of industrial heat in Brazil. It was reported that in Brazil many industries use surplus bagasse to avoid use of fuel oil (Seabra et al., 2011). Thus, for Brazil, displacement of fuel-oil based heat supplied with 92 % efficiency is assumed in this project (Tsiropoulos et al., 2014). Per 1 kg of ethanol 0.9 MJ of fuel-oil based heat is avoided by the surplus bagasse.

Surplus electricity produced from bagasse avoids marginal electricity production in Brazil. The Brazilian power system is supplied 80 % by hydropower and 20 % by thermal power (Seabra et al., 2011). Natural gas is predominantly used in power plants in Brazil for marginal electricity production (Seabra et al., 2011). Thus, the natural gas-based electricity is assumed to be avoided by the surplus electricity (0.16 kWh/kg ethanol). Instead of average Brazilian electricity production, marginal electricity production is considered because in Brazil small changes in demand for electricity will not influence the capacity of electricity production by hydropower.

Furthermore, in bio-MEG production, DEG and TEG are also produced in very small amount as co-products. These are assumed to be substituting fossil-based DEG and TEG.

2.1.2 Bio-based alternative 1: Ethanol from EU crop mix

An alternative hypothetical product system is considered where instead of Brazilian sugarcane (baseline, status-quo), ethanol is produced from European crops. Combined market mix consisting of maize, wheat and sugar beet is considered as described in section 1.4. The process flow diagram of the bio-PET production using European crops of wheat, maize and sugar beet are given in Figure 33; Figure 34; Figure 35.

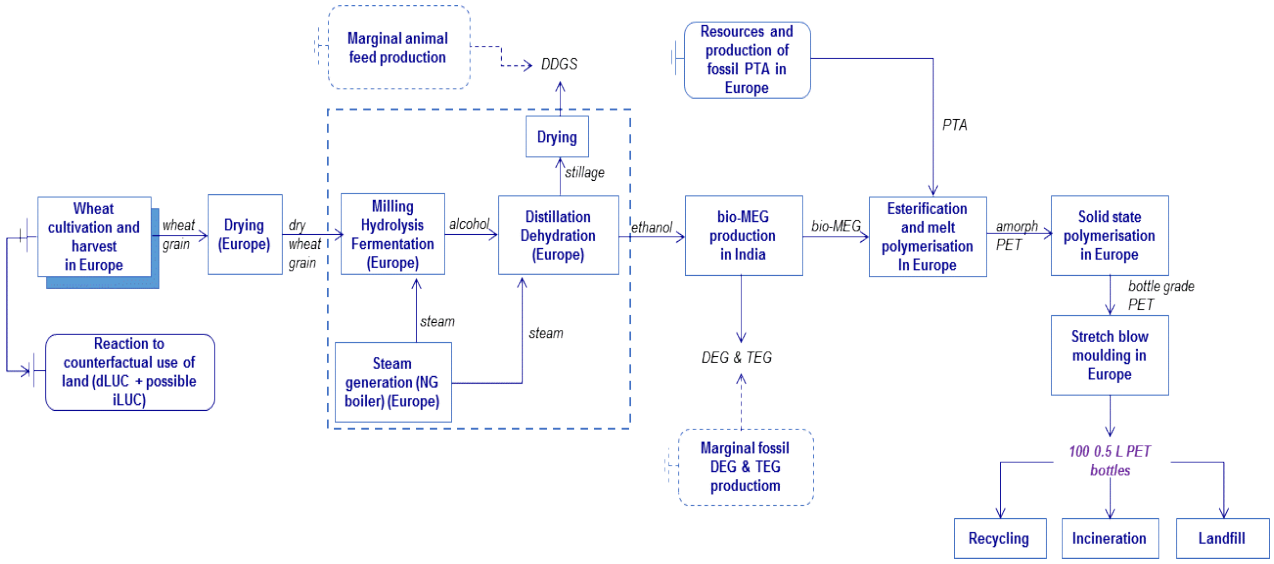


Figure 33. Process flow diagram of bio-based alternative 1: bio-based PET production from European wheat.

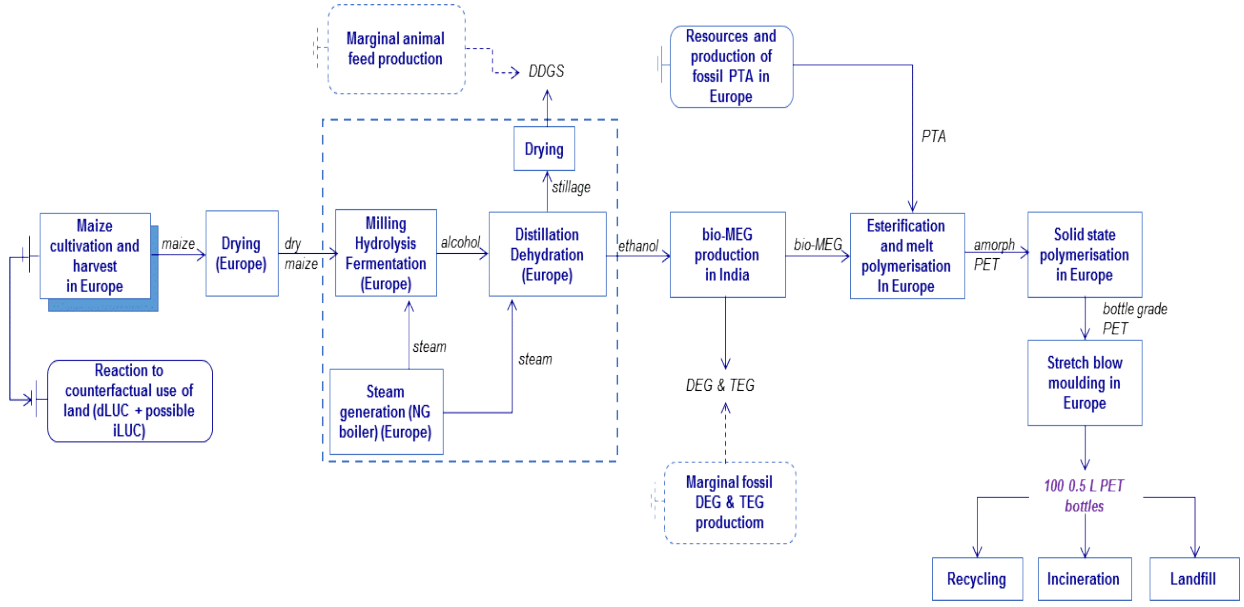


Figure 34. Process flow diagram of bio-based alternative 1: bio-based PET production from European maize.

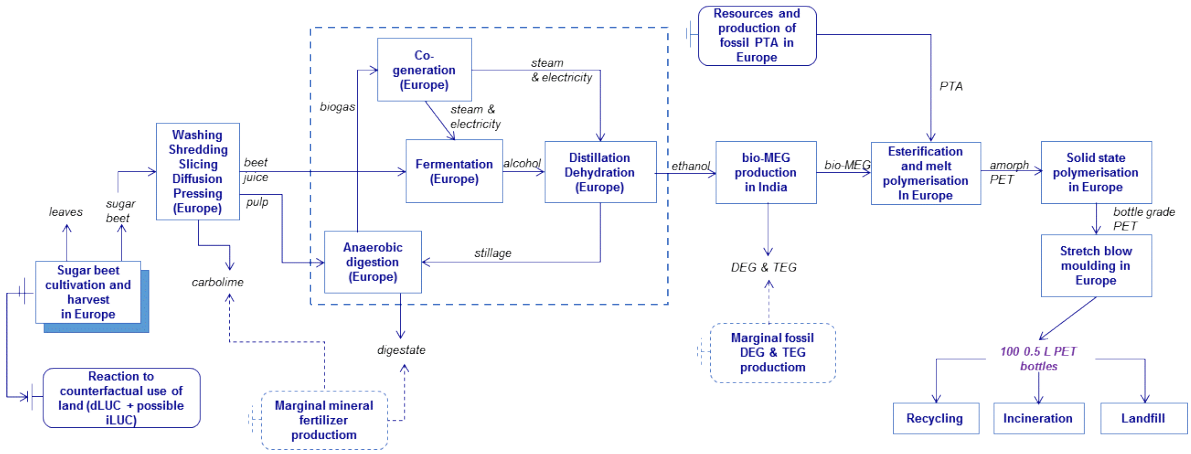


Figure 35. Process flow diagram of bio-based alternative 1: bio-based PET production from European sugar beet.

From ethanol production onward (bio-MEG production, polymerization and stretch blow moulding) the processes are the same as the baseline product system. Therefore, only the inventory corresponding to ethanol production from the three European crops is described below. The foreground data sources used for the bio-based PET bottles in alternative 1 are summarised in Table 56.

Table 56. Data used in the bio-based PET bottle: alternative 1.

Processes	Data sources, foreground	Comments
Wheat, maize and sugar beet cultivation and harvest	AgriFootprint, 2017	Represents country specific crop cultivation. Crop yields derived mainly from 2010-2014 FAO statistics (FAOSTAT, 2014)
Bioethanol production from wheat	Biograce, 2015	Inventory data based on JEC Well-to-Wheels analysis (JEC, 2014)
Bioethanol production from maize	Edwards et al., 2017	Represents averages of data from various sources from years 1999-2015
Bioethanol production from sugar beet	Buchspies and Kaltschmitt, 2017	Represents industrial production in Europe in 2014

2.1.2.1 Wheat, maize and sugar beet cultivation and harvest

For each of the three European crops, Eurostat statistics are collected, and a five-year average is taken. Countries that contribute to at least 80 % of the production are determined and a distribution is made among them. France and Germany are found as the major producers. An agrifootprint database is used for the inventory data for the relevant countries and crops. Then the distribution is used to calculate the weighted average European inventory for each crop. The distribution is as shown in Table 57.

Table 57. Contribution of the EU countries in the modelled average European maize, wheat and sugar beet (country mix made to represent 85 % of the production, 5-yr averages retrieved from EUROSTAT, 2017).

EU country of production	Share assumed in the model for Maize (%)	Share assumed in the model for Wheat (%)	Share assumed in the model for Sugar beet (%)
Germany	7.9	25	29.6
France	23.7	35.3	43
Italy	14.7	-	-
Romania	17.1	8	-
Hungary	12.5	5.1	-
Spain	13.0	5.6	-
Poland	6.4	10.7	14.5
Bulgaria	4.7	5.3	-
Czech Republic	-	5	-
The Netherlands	-	-	7.2
Belgium	-	-	5.7

2.1.2.2 Ethanol production from wheat and maize

Wheat and maize after drying are milled in a dry milling process. By hydrolysis and fermentation alcohol is produced. Subsequently, ethanol is separated from water and impurities by distillation and dehydrated. The stillage from distillation is dried to form dried distiller grains with soluble (DDGS) which is used as animal feed. Inventory data for ethanol production from maize is based on Edwards et al. (2017) and from wheat is based on Biograce (2015).

2.1.2.3 Ethanol production from sugar beet

Sugar beets are washed, sliced and pressed to produce beet juice. In beet processing, beet pulp and carbonation lime are produced as by-products. Carbonation lime is used as a fertiliser product. Beet juice is fermented, followed by distillation and dehydration to produce ethanol. Beet pulp from beet processing and stillage from distillation are anaerobically digested to obtain biogas. Biogas is used internally to supply heat and electricity. Surplus electricity is sold to the grid. Digestate is used as a fertiliser. Inventory data is based on industrial production in Europe (Buchspies and Kaltschmitt, 2017).

2.1.2.4 Multifunctionality

For ethanol production from wheat and maize, the by-product DDGS is produced. To avoid allocation, the system is expanded to include their animal feed replacement. The co-products are displaced with a mix of marginal feed ingredients with the same standardised feed unit.

The marginal protein feed is taken as soybean meal, marginal carbohydrate is taken as maize and marginal oil is taken as palm oil based on Tonini et al. (2016). These choices are based on detailed elaboration of recent demand trends and future projections (FAPRI, 2012; FAOSTAT, 2014).

For ethanol production from sugar beet, the by-products leaving the system boundary are carbolime, digestate and surplus electricity. Digestate and carbolime replaces mineral fertilisers according to their nutrient content (Nmin, Norg, P and K) attained from Buchspies and Kaltschmitt, 2017. The mineral fertiliser equivalents of Nmin, Norg, P and K are 1, 0.15, 0.84 and 1, respectively. The marginal N, P, and K fertilisers considered are urea, diammonium phosphate (DAP), and potassium chloride, respectively. This choice is supported by 2000-2010 trends in demand/consumption based on Tonini et al. (2016). The surplus electricity replaces electricity from the grid (medium voltage electricity across Europe).

2.1.3 Bio-based alternative 2: Ethanol from wheat straw

The second alternative hypothetical product system is ethanol produced from European wheat straw instead of Brazilian sugarcane (baseline, status-quo). The process flow diagram of bio-PET production using European wheat straw is given in Figure 36.

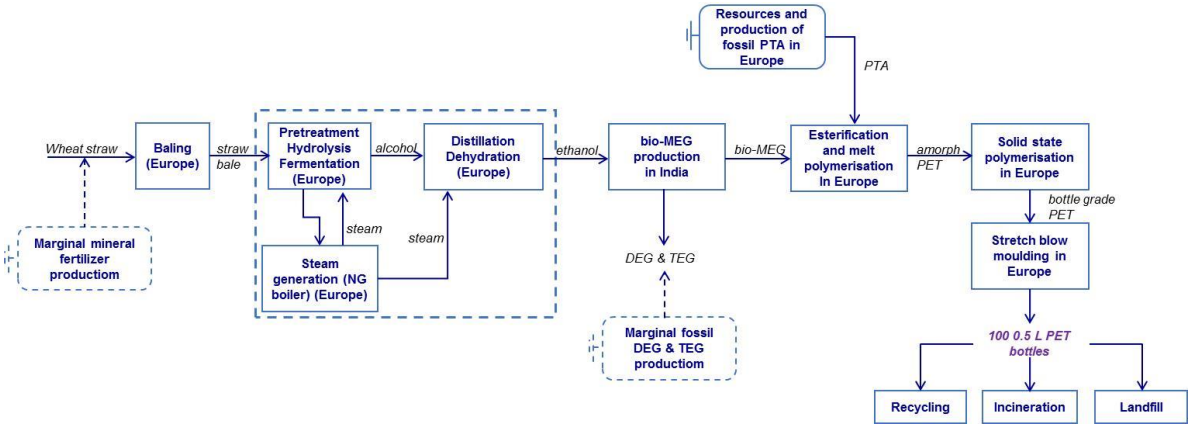


Figure 36. Process flow diagram of bio-based alternative 2: bio-based PET production from European wheat straw.

From ethanol production onward (bio-MEG production, polymerization and stretch blow moulding) the processes are the same as the baseline product system. Therefore, only the inventory corresponding to ethanol production from European wheat straw is described below. The foreground data sources used for the bio-based PET bottles alternative 2 are summarised in Table 58.

2.1.3.1 Wheat straw

Removal of wheat straw from land depletes the soil. It requires N, P and K mineral fertilisers to be applied in proportion to the content of the wheat straw. Tonini et al. (2016) describes a compensatory 40 % replacement for N and 100 % for P and K. The nutrient (N, P and K) content of wheat straw was obtained from this literature source. The marginal N, P, and K fertilisers considered were urea, diammonium phosphate (DAP), and potassium chloride, respectively. This choice is supported by 2000-2010 trends in demand/consumption based on Tonini et al. (2016).

2.1.3.2 Baling

The straw collected goes through baling and is transported by lorry to a nearby ethanol plant. The straw baling and transport is modelled based on Giuntoli et al. (2017).

2.1.3.3 Ethanol production

Wheat straw goes through pre-treatment, hydrolysis and fermentation to produce alcohol. The solid by-products are separated and used for process heat. The alcohol produced goes through distillation and dehydration to produce ethanol. The inventory data is based on Edwards et al. (2014).

Table 58. Data used in the bio-based PET bottle: alternative 2.

Processes	Data sources, foreground	Comments
Wheat straw	Tonini et al. 2016	The amount of mineral fertiliser to be applied based on nutrient content and method described in Tonini et al. 2016
Straw baling	Giuntoli et al. 2017	Inventory data based on GEMIS, 2014
Bioethanol production from wheat straw	Edwards et al. 2017	Inventory data based on Johnson, 2016

2.1.4 The fossil reference systems: petrochemical PET

In 2017 PlasticsEurope released, an eco-profile for petrochemical bottle-grade PET. This data is considered as the main reference since PlasticsEurope provides the most up-to-date eco-profile for European fossil-based plastics. Regarding impact categories, the eco-profile of PlasticsEurope provides information for climate change impact potentials and NREU. However, data for the full list of 16 PEFCR impact categories selected in this project are not available. Furthermore, breakdown of the inventory for the different unit processes is not available. To be consistent with the modelling for the bio-based PET it is important to have the breakdown for all impact categories, therefore we opted to model our own petrochemical PET.

The petrochemical PET bottle production involves fossil-based MEG and fossil-based PTA production, polymerization and stretch blow moulding. The processes of PTA production, polymerisation and stretch blow moulding are the same for the bio-based and the petrochemical route. To be consistent with the modelling of the bio-based PET, we modelled petrochemical PET by replacing bio-MEG with fossil-based MEG. The same environmental impacts estimated for PTA production, polymerization and stretch blow moulding for bio-based PET described in Table 55 are used for petrochemical PET. For fossil-based MEG, Ecoinvent 3.3 *Ethylene glycol {RER}| production | Alloc Def, U* is used. The climate change and NREU impact categories are updated with the breakdown given in the latest PlasticsEurope PET eco-profile (PlasticsEurope, 2017).

To cover all impact categories while providing the most up to date profile, it was necessary to combine different data sources. For climate change and NREU the latest PlasticsEurope eco-profile is used and for the rest of the impact categories either Ecoinvent 3.3 or industry data 2.0 inventories are used (Table 55). We realise the inconsistency of using different data sources. We did several crosschecks with PlasticsEurope to portray a correct breakdown of the unit processes, but this was only possible for climate change and NREU and not for all impact categories.

Fossil-based PTA production is based on PlasticsEurope's PTA eco-profile (PlasticsEurope, 2014). PlasticsEurope data for petrochemical plastics are not transparent and do not comply

with ILCD requirements. For fossil-based MEG, data from Ecoinvent 3.3 is used. The Ecoinvent database is updated with respect to background processes like fuel production, transport, etc., but the foreground processes have not been subjected to an update in the last 10 years. These data were compiled to form the reference PET data in this study using reaction stoichiometry which states 0.32 kg MEG/kg PET and 0.86 kg PTA/kg PET. The compiled data for PET is compared with the available data from different data sources for PET in Chapter 6 (*Investigation of generic LCA data for fossil-based plastics*). Differences are found to be large and some impact categories were found not applicable (see Chapter 6). The data are provided in Table 59 for the suitable impact categories for bottle-grade PET. This data is used in the analysis in Chapter 6 (see Table 38).

Table 59. Compiled petrochemical bottle-grade PET data.

Impact category	Unit	MEG production	PTA production	Polymerization	Total
Climate change	kg CO_2 eq.	0.55	1.32	0.32	2.19
Ozone depletion	kg CFC-11 eq.	Not applicable	Not applicable	Not applicable	Not applicable
Human toxicity, non-cancer effects	CTUh	Not applicable	Not applicable	Not applicable	Not applicable
Human toxicity, cancer effects	CTUh	Not applicable	Not applicable	Not applicable	Not applicable
Particulate matter	kg PM2.5 eq.	3.45E-04	3.04E-04	4.34E-04	1.08E-03
Ionizing radiation HH	kBq U235 eq.	Not applicable	Not applicable	Not applicabl	Not applicable
Photochemical ozone formation	kg NMVOC eq.	1.79E-03	3.91E-03	1.66E-03	7.36E-03
Acidification	molc H+ eq.	2.71E-03	5.07E-03	3.55E-03	1.13E-02
Terrestrial eutrophication	molc N eq.	4.54E-03	9.66E-03	5.44E-03	1.96E-02
Freshwater eutrophication	kg P eq.	Not applicable	Not applicable	Not applicabl	Not applicable
Marine eutrophication	kg N eq.	4.49E-04	9.24E-04	5.37E-04	1.91E-03
Freshwater ecotoxicity	CTUe	Not applicable	Not applicable	Not applicable	Not applicable
Land use	kg C deficit	Not applicable	Not applicable	Not applicable	Not applicable
Water use	m^3	3.16E-01	2.50	2.42E-01	3.06
Abiotic depletion	kg Sb eq.	2 Not applicable	Not applicable	Not applicable	2 Not applicable
Abiotic depletion (fossil fuels)	MJ	15.44	45.15	7.40	67.99
NREU	MJ	16.72	47.29	5.60	69.60

2.2 End of life description, data, assumptions and multifunctionality

Three EoL options for the PET beverage bottles were modelled (see Table 60):

- **Mechanical plastic recycling**; the process includes the energy and material requirements for the transportation to the facility, sorting, cleaning and recycling processes. The recycled PET is assumed to substitute virgin PET production. The rejects from the recycling process are sent to incineration (both with and without energy recovery) and transportation is included. The flowchart for plastic recycling is presented in Figure 8.

- **MSW Incineration with and without energy recovery**; incineration is modelled with a generic MSW incineration plant, which represents average EU waste incineration (with and without energy recovery) with an average heat efficiency of 22 % and electrical efficiency of 9 % (CEWEP, 2013; Eurostat, 2017a). The process includes the direct emissions from waste incineration as well as indirect emissions from production of the input materials, combustion of fossil fuels as well as treatment of bottom ash and fly ash. The flowchart for MSW incineration is presented in Figure 9.

- **MSW landfilling**; the process includes the construction and operation of an average landfill site, direct emissions of landfill gas, landfill gas collection, flaring and recovery system, treatment of collected leachate and direct emissions from uncollected leachate. 29 % of the collected landfill gas is assumed to be utilised to generate electricity (OpenLCA Nexus, 2015). The electricity conversion efficiency is 37 % (Christensen, 2011). The flowchart for MSW landfilling is presented in Figure 10.

The intended EoL for bio-based PET is recycling.

In addition to modelling full (100 %) recycling, incineration and landfilling individually, a mix of these EoL scenarios was also modelled to represent the current situation in Europe. The assumed distribution between EoL technologies for average EoL mix for PET bottles is presented in Table 60 based on statistics and literature (EPBP, n.d.; PlasticsEurope, 2016b; PlasticsEurope, 2013; European Commission, 2018; BPF, 2018)[49]. The collection rate for recycling for the bottles of 60 %, correlates to the collection rates of 42 % to 94 % in five different large EU countries in Figure 6 in Plastics Recyclers Europe (2017).

Table 60. EoL options, the intended EoL option and the estimated EoL mix.

Product system	EoL options modelled individually (100 % scenarios)	Intended EoL option	Estimated EoL collection mix of EU (based on statistics of waste treatments of waste plastics)
Bio-based PET	Recycling Incineration Landfilling	Recycling	Recycling: 60 % Incineration: 20 % Landfilling: 20 %
Petrochemical PET	Recycling Incineration Landfilling		Recycling: 60 % Incineration: 20 % Landfilling: 20 %

The average European EoL technologies are described in the approach and methodology chapter in the section regarding EoL. In that section, factors not related to the specific product system are described, e.g. energy efficiency of incineration plant and consumption of energy and materials for each technology. Furthermore, principles for substitution are described. This includes e.g. that the energy produced in the EoL technologies is substituted with marginal electricity and heat, as well as the recycling substitution methodology.

[49] The amount of recycled PET has been reported 1.68 million tonnes in 2012 (PlasticEurope, 2013), 1.8 million tonnes in 2015 (The European PET Bottle Platform, nd.) and forecasted for 2017 to be 2.1 million tonnes (PlasticsEurope, 2013). PlasticsEurope (2016) reports a PET polymer demand in the EU of 3.1 million tonnes in 2015. No data on the amount of PET bottles incinerated or landfilled were found. In Germany the collection rate in 2015 was 94 %, in France 56 %, in Italy 42 %, in Spain 60 % and in UK 53 % reported by Plastic Recyclers Europe (2017). In the UK RECOUP calculated that the over all collection of plastic bottles in 2017 was 59% (BPF, 2018). Based on the figures above it has been estimated 60 % of the PET bottles are recycled yearly, when considering that most of the PET consumed is for bottles. The share between incineration and landfilling is reported (European Commission, 2018) to be almost the same; hence the remaining 40 % is divided equally between these technologies.

Several product system dependent factors are presented in Table 61 for each relevant EoL technology for beverage bottles. For plastic recycling, the PET in both product systems is substituted with petrochemical amorphous bottle grade PET (the two product systems have the same factors). The recycling rate is calculated by multiplying the collection rate and the sorting and technology efficiency.

Table 61. Values for product system dependent factors for beverage bottles.

EoL Technology	Product system dependent factor	Unit	Bio-based PET (30 %)	Petrochemical PET	Reference
Plastics recycling	Sorting and technology efficiency (amount collected for recycling which is recycled)	% of input	76	76	EPBP, N.D.; PlasticsEurope, 2016; PlasticsEurope, 2015; PlasticsEurope, 2013a; PlasticsEurope, 2013b
MSWI with and without energy recovery	Energy content a	MJ/kg TS	22.91	22.91	Götze et al., 2016
Landfilling	1st order decay rate for methane generation	1/s	0	0	Estimated
Energy content is in lower heating value (LHV).					

Table 62 presents the material composition of the beverage bottles. This composition is utilised as the input to the EoL LCA model. The full chemical composition is presented in Annex 1.

The EoL-reference flow for the beverage bottles is:

- 1 kg of plastic and 100 g contamination

Table 62. Overview of the material composition of beverage bottles (1 kg plastic + 100 g contamination).

Chemical component	Unit	Bio-based PET (30 %)	Petrochemical PET
Water	% of total	3.3	3.3
TS (VS+ash) [a]	% of total	96.8	96.8
VS	% of TS	99.0	99.0
C fossil	% of TS	44.8	64.0
C biogenic	% of TS	19.2	0.0
Reference		Götze et al., 2016 (Modified C)	Götze et al., 2016
a TS: Total solids, VS: Volatile solids.			

The material composition for the bio-based PET and the petrochemical PET are the same, hence the only difference is that 30 % of the carbon in the bio-based PET is biogenic carbon and not fossil carbon. Therefore, the results for the bio-based PET and the petrochemical PET will be very similar (there is a slight difference only in the climate change impact, due to a small difference in characterisation factors between biogenic and fossil C (PEFCR guidance draft version 6.3 (European Commission, 2017)).

3 Life cycle impact assessment results and interpretation

3.1 Cradle-to-gate environmental impacts and interpretation, Bio-Based system(s)

3.1.1 Cradle-to-gate environmental impacts of Bio-Based baseline system: Bio-Based PET bottle from Brazilian sugarcane

The cradle-to-factory gate LCA results from the assessment of 1 functional unit of bio-based PET bottle is presented in Table 63.

Table 63. Cradle-to-factory gate LCA results of bio-based PET bottle (baseline) for 1 functional unit (1 kg of PET bottle (excluding DLUC and ILUC effects)).

Impact category	Unit	Bio-based PET (baseline)
Climate change	kg CO_2 eq.	3.23
Ozone depletion	kg CFC-11 eq.	1.04E-05
Human toxicity, non-cancer effects	CTUh	1.03E-06
Human toxicity, cancer effects	CTUh	1.38E-07
Particulate matter	kg PM2.5 eq.	9.89E-03
Ionizing radiation HH	kBq U235 eq.	5.83E-01
Photochemical ozone formation	kg NMVOC eq.	1.51E-02
Acidification	molc H+ eq.	2.61E-02
Terrestrial eutrophication	molc N eq.	5.59E-02
Freshwater eutrophication	kg P eq.	1.37E-03
Marine eutrophication	kg N eq.	4.81E-03
Freshwater ecotoxicity	CTUe	1.75E+01
Land use	kg C deficit	10.83
Water use	m^3	5.30
Abiotic depletion	kg Sb eq.	1.81E-06
Abiotic depletion (fossil fuels)	MJ	74.09
NREU	MJ	82.26

The breakdown of process contribution can be found in Figure 37. The impacts of bio-based PET (baseline) are broken down into the main unit processes described in section 2.1 inventory analysis.

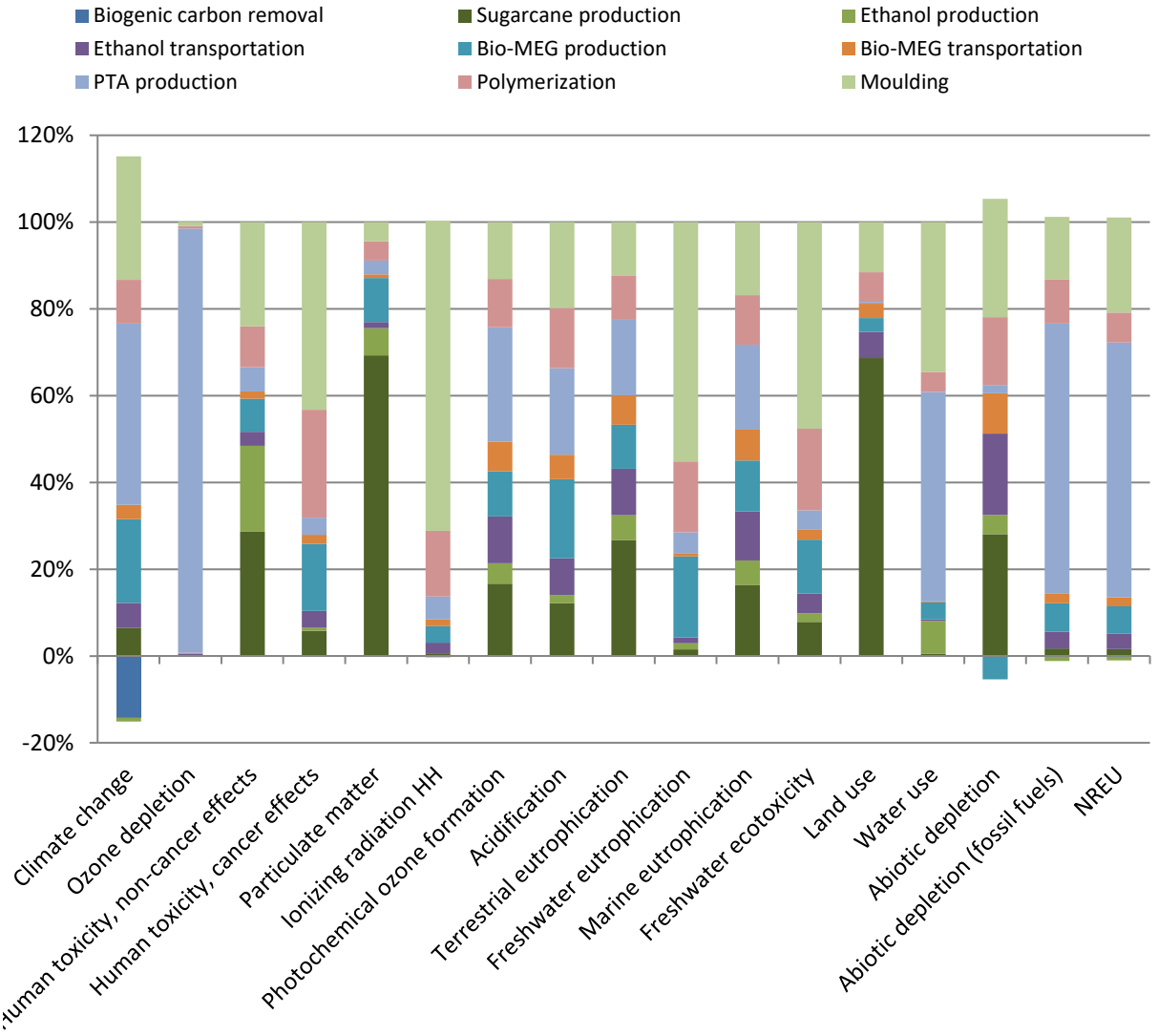


Figure 37. Breakdown of the cradle-to-factory gate baseline results for bio-based PET bottles, 1 functional unit, excluding DLUC and ILUC effects.

Sugarcane production is neither energy intensive (low contribution to NREU), nor carbon intensive (low contribution to greenhouse gas (GHG) emissions, NOTE: based on the preliminary results without ILUC and DLUC effects). However, it contributes significantly to six impact categories: namely, land use (69 %), particulate matter (69 %), human toxicity (non-cancer) (29 %), terrestrial eutrophication (27 %), photochemical ozone formation (17 %), and marine eutrophication (16 %). It also has a 28 % contribution to abiotic depletion. For other impacts the contribution from sugarcane production are minor (less than 8 %). Particulate matter is caused by the direct emissions due to the cane waste burning during harvest. The pesticides used have an effect on the human toxicity (non-cancer) and abiotic depletion categories. Nitrogen oxide emissions due to application of fertilisers result in photochemical ozone formation, terrestrial and marine eutrophication.

The bioethanol production contributes to human toxicity (non-cancer) (20 %) caused by the zinc emissions to air from bagasse burning. It does not have a significant contribution in other impact categories (up to 8 %). This is mainly due to the use of the by-product bagasse for internal energy supply and the credits gained by its excess. The bio-MEG production shows a contribution of between 10-20 % to most of the impact categories.

The transportation requirements (ethanol from Brazil to India and Bio-MEG from India to Europe) contribute to marine eutrophication (18 %), terrestrial eutrophication (18 %), and photochemical ozone formation (18 %) due to the nitrogen oxides emissions to air. They also contribute to abiotic depletion (28 %) due to elements used in the production of lorries.

The environmental impact of bio-based PET bottles is dominated by the PTA production, polymerisation process and stretch blow moulding processes which are identical for petrochemical PET bottles. Together, their contribution to bio-based PET bottles ranges from 40 to 99 % in all impact categories except for particulate matter and land use.

The results of bio-based PET are further elaborated below. Detailed analysis is made to trace the origin of the impact in the major contributing processes for these impact categories and for impact categories where biomass production has a significant impact. Since the PTA production, polymerisation and stretch blow moulding processes are identical in both bio-based and petrochemical product systems, the contributors for the bio-MEG production processes are analysed (starting from sugarcane cultivation and harvest ending with transportation of bio-MEG to Europe).The most relevant impact categories that need a detailed interpretation are identified according to PEFCR guidance (i.e. cumulatively contributing to 80 % of the total environmental impact (excluding toxicity related impact categories) based on the normalised and weighted results (see section 3.4.1)). In descending order starting with the most relevant, they are: particulate matter, climate change, abiotic depletion (fossil fuels), water use and acidification.

- **Particulate Matter.** For particulate matter, 69 % of the contribution to PET bottle production comes from sugarcane cultivation and harvesting. 99 % of this originates from particulate emissions from waste burning. A sensitivity analysis is made in section 4.1 which shows that the effect of the waste burning practice phase-out on particulate matter and other impact categories.

- **Climate change.** For bio-based PET the biogenic carbon embedded in the product is calculated as 0.45 kg CO_2/kg, based on the molecular structure. When the biogenic carbon removal is considered, the cradle to factory gate climate change impact of bio-based PET bottle is 3.23 kg CO_2 eq./kg PET from cradle to factory gate. Again, we emphasize that this is a preliminary conclusion of carbon accounting without considering the effects of DLUC and ILUC. The PTA production, polymerisation and stretch blow moulding, together account for 80 % of the climate change impact. The remaining impacts are caused by the bio-MEG production (20 %). Nearly 60 % originates from ethanol production process itself. For the rest, transportation (of ethanol to India and bio-MEG to Europe) requirements also have significant contribution (accounting for about 30 % of the impacts). The process energy used in the bio-MEG process is coal-based electricity and steam produced in India which accounts for the high impact.

- **Abiotic depletion (fossil fuels).** The PTA production, polymerisation and stretch blow moulding, all together account for 86 % of the impacts. Among these three processes, PTA production is the major contributor (62% of total), followed by stretch blow moulding (14%) and polymerization (10%). For stretch blow moulding, 93% of the impact is caused by the production of the electricity consumed by the process. The processes involved in the production of sugarcane and converting it to bio-MEG appears to have a relatively insignificant contribution.

- **Water use.** The PTA production accounts for 48 % of the impacts, with 60 % for cooling water and the rest for water use in production. The moulding contributes 35 % of the impact due to cooling water used in the conversion of the plastic product. The processes involved in the production of sugarcane and converting it to bio-MEG are seen to have relatively insignificant contribution.

- **Acidification.** The Bio-MEG production process contributes to 18 % of acidification. The impacts originate from the sulfur dioxide and nitrogen oxides emissions to air from the production processes of steam (50 %), oxygen (24 %) and electricity (22 %) used in this process. For sugarcane cultivation and harvest (contributes 12 %), 77 % of the contribution comes from the direct nitrogen oxides and sulfur dioxide emissions to air from waste burning.

For this case study, additional analysis of impact categories where biomass production has a significant contribution is also detailed:

- **Human toxicity (non-cancer effects).** A major contribution to the human toxicity (non-cancer effects) impact category comes from sugarcane production (29 % of bio-based PET bottle) and ethanol production (20 % of bio-based PET bottle). For the sugarcane cultivation and harvest process, soil emissions from applying pesticides (arsenic compounds) contribute to 83 % of impacts. For the ethanol production process, 70 % of the impacts originate from zinc emissions to soil and 20 % zinc emissions to air from bagasse combustion.

- **Terrestrial and Marine Eutrophication & Photochemical ozone formation.** For the sugarcane cultivation and harvesting process (contributes 27 % to terrestrial eutrophication, 16 % to marine eutrophication, 17 % photochemical ozone formation), the impacts mainly (>70 %) come from nitrogen oxides air emissions from the fertiliser application and waste burning. For ethanol production process (contributes 6 %), the impacts (>98 %) come from nitrogen oxides emissions from bagasse burning. Nitrogen oxides emissions caused by the transportation requirements (of ethanol to India and bio-MEG to Europe) also have a significant contribution (approximately 18 %).

3.1.2 Cradle-to-gate environmental impacts of bio-based alternative system 1: Bio-based PET bottle from EU crop mix

For this alternative hypothetical product system, ethanol is produced from European crops instead of Brazilian sugarcane (baseline, status-quo). The cradle-to-gate LCA results from the assessment of one functional unit of bio-based PET bottle from EU crops are presented in Table 64.

Table 64. Cradle-to-factory gate LCA results of bio-based PET bottle (alternative 1: from EU crops) for 1 functional unit (1 kg of PET bottle (excluding DLUC and ILUC effects)).

Impact category	Unit	Bio-based PET (alternative 1)
Climate change	kg CO_2 eq.	3.31
Ozone depletion	kg CFC-11 eq.	1.05E-05
Human toxicity, non-cancer effects	CTUh	1.58E-06
Human toxicity, cancer effects	CTUh	1.38E-07
Particulate matter	kg PM2.5 eq.	2.55E-03
Ionizing radiation HH	kBq U235 eq.	6.32E-01
Photochemical ozone formation	kg NMVOC eq.	1.22E-02
Acidification	molc H+ eq.	3.03E-02
Terrestrial eutrophication	molc N eq.	6.94E-02
Freshwater eutrophication	kg P eq.	1.40E-03
Marine eutrophication	kg N eq.	1.08E-02
Freshwater ecotoxicity	CTUe	2.44E+01
Land use	kg C deficit	9.16
Water use	m^3	5.70
Abiotic depletion	kg Sb eq.	7.20E-07
Abiotic depletion (fossil fuels)	MJ	78.0
NREU	MJ	86.3

The breakdown of process contribution can be found in Figure 38. The impacts of bio-based PET (alternative 1: from EU crops) are broken down into the main unit processes described in the inventory analysis. From ethanol production onward (bio-MEG production, polymerization and stretch blow moulding) the processes are the same as the baseline product system. Therefore, only the interpretation for EU crops production and ethanol production processes are described here.

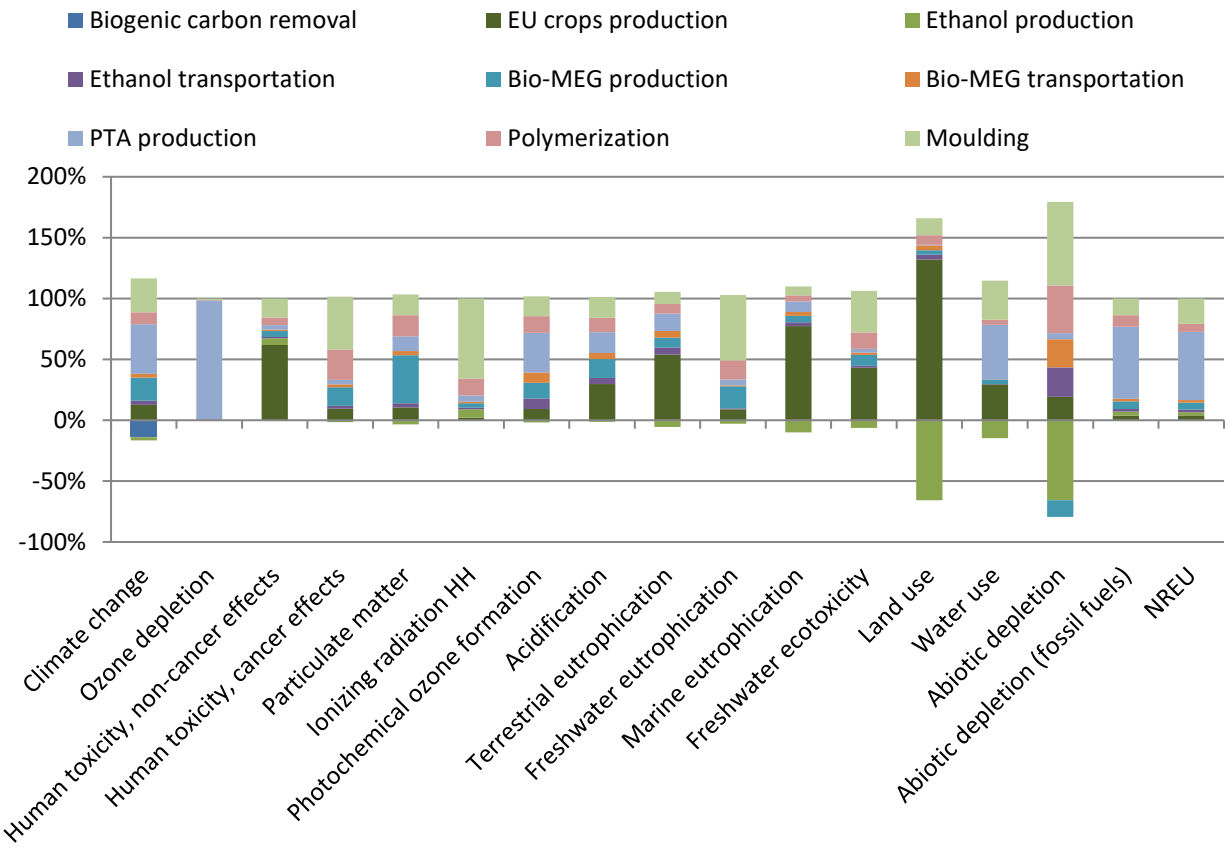


Figure 38. Breakdown of the cradle-to-factory gate results for bio-based PET bottles (alternative 1: from EU crops), 1 functional unit, excluding DLUC and ILUC effects.

Production of EU crops has low contribution to NREU and climate change (based on the preliminary results without ILUC and DLUC effects). However, it shows high contribution to human toxicity (non-cancer effects) (62 %), terrestrial eutrophication (54 %), marine eutrophication (77 %) and land use impact categories. The biogenic carbon stored in the bottle gives a 14 % saving in climate change impacts similar to the baseline system. Human toxicity (non-cancer) effects are caused by the emissions from pesticides to agricultural soil. Terrestrial eutrophication is due to direct field emissions of ammonia and nitrogen oxides emissions due to fertiliser application. Marine eutrophication is due to nitrate emissions to water and nitrogen oxides emissions from fertiliser application. EU crops production also has a significant contribution in the freshwater ecotoxicity, acidification and water use categories.

The bioethanol production process shows overall savings in the impact categories. This is mainly due to the by-product of DDGS from ethanol production from wheat and maize, which enables avoided animal feed production. The co-products are displaced with a mix of marginal feed ingredients (soybean meal, maize and palm oil) with the same standardised feed unit. Major savings are seen in the land use and abiotic depletion (due to avoided pesticides) categories, of about 66 % each.

3.2.1 Cradle-to-gate environmental impacts of bio-based alternative system 2: Bio-based PET bottle from wheat straw

For this alternative hypothetical product system, ethanol is produced from European wheat straw instead of Brazilian sugarcane (baseline, status-quo). The cradle-to-gate LCA results from the assessment of one functional unit of bio-based PET bottle from European wheat straw are presented in Table 65.

Table 65. Cradle-to-factory gate LCA results of bio-based PET bottle (alternative 1: from EU crops) for 1 functional unit (1 kg of PET bottle (excluding DLUC and ILUC effects)).

Impact category	Unit	Bio-based PET (alternative 2)
Climate change	kg CO_2 eq.	3.11
Ozone depletion	kg CFC-11 eq.	1.04E-05
Human toxicity, non-cancer effects	CTUh	5.58E-07
Human toxicity, cancer effects	CTUh	1.33E-07
Particulate matter	kg PM2.5 eq.	2.45E-03
Ionizing radiation HH	kBq U235 eq.	6.11E-01
Photochemical ozone formation	kg NMVOC eq.	1.16E-02
Acidification	molc H+eq.	2.27E-02
Terrestrial eutrophication	molc N eq.	3.69E-02
Freshwater eutrophication	kg P eq.	1.39E-03
Marine eutrophication	kg N eq.	3.67E-03
Freshwater ecotoxicity	CTUe	1.65E+01
Land use	kg C deficit	3.35
Water use	m^3	4.95
Abiotic depletion	kg Sb eq.	2.37E-06
Abiotic depletion (fossil fuels)	MJ	74.2
NREU	MJ	82.9

The breakdown of process contribution can be found in Figure 39. The impacts of bio-based PET (alternative 2: from EU wheat straw) are broken down into the main unit processes described in the inventory analysis. From ethanol production onward (bio-MEG production, polymerization and stretch blow moulding) the processes are the same as the baseline product system. Therefore, only the contributions coming from wheat straw and the ethanol production from it are described below.

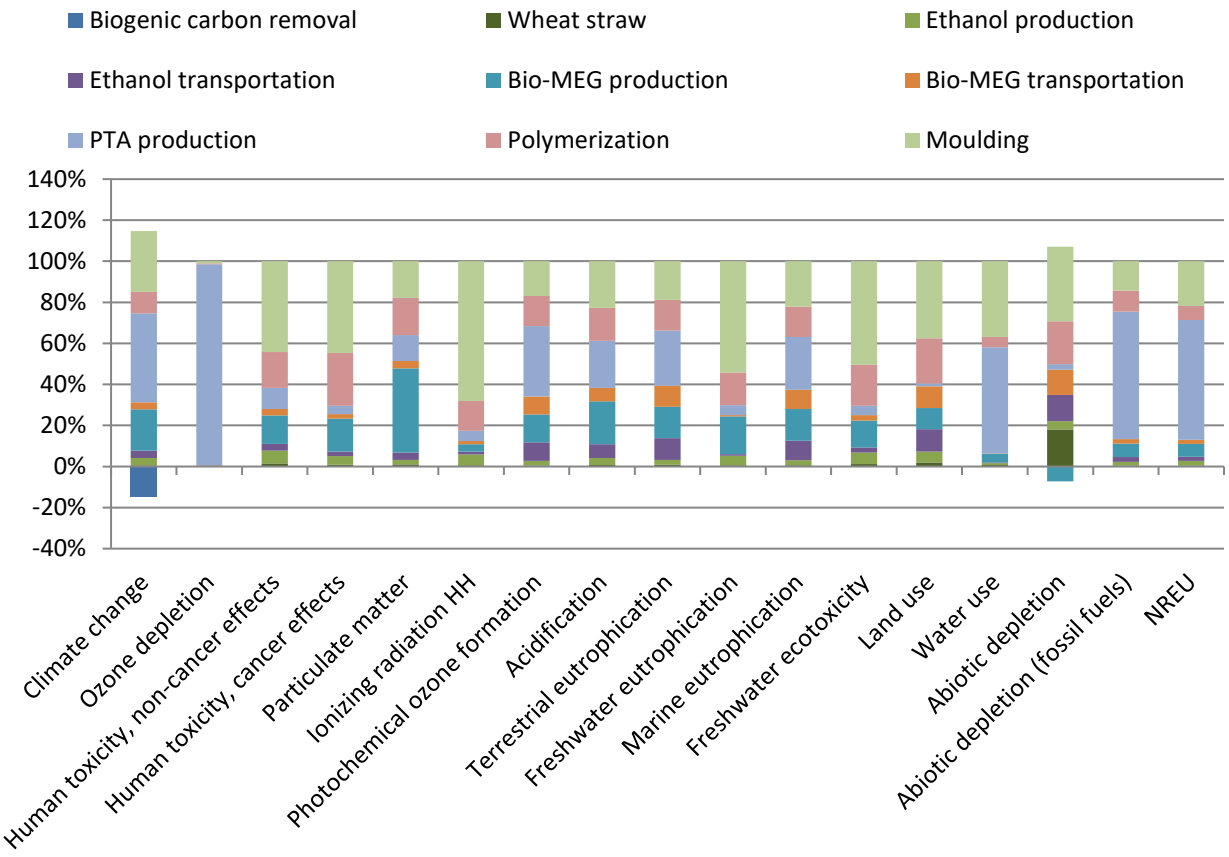


Figure 39. Breakdown of the cradle-to-factory gate results for bio-based PET bottles (alternative 2: from EU wheat straw), 1 functional unit

EU wheat straw has a negligible impact in all categories except for abiotic depletion where it has 18 % contribution. The contribution to abiotic depletion is due to the use of mineral fertilisers that are modelled to replace the nutritional value of soil with the removal of straw. The lack of impact for the remaining categories is because there are no biomass production related impacts. These would have resulted in significant contribution to the land use, water use and eutrophication categories which are not present here.

Also, the ethanol production process is seen not to have a significant contribution in any impact categories (maximum 6 %). This is mainly due to the energy requirement of this system being satisfied with by-products (lignin) internally.

3.2 Cradle-to-gate environmental impacts of the petrochemical reference system

The cradle-to-factory gate LCA results for one functional unit (=1 kg) of petrochemical PET bottle is presented in Table 66. This is calculated by adding the impact of moulding with the impact of bottle-grade plastic granulate production (given in Chapter 6 *Investigation of generic LCA data for fossil-based plastics*, Table 40 and section 2.1.4, Table 59) per FU of 1 kg. Since fossil based and bio-based PET are chemically identical, the same inventory data is used for calculating the impact of moulding (given in Table 55).

Based on Chapter 6 Ranges for environmental impacts from production of fossil-based plastics, 9 out of 17 impact categories were found suitable for petrochemical PET interpretation, namely: climate change, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication, marine eutrophication, water use, abiotic depletion (fossil fuels) and NREU. This is because comparison of different datasets revealed large differences for the other

impact categories. The data used is based on PlasticsEurope since it provides the most up-to-date eco-profile for European fossil-based plastics. However, it does not conform with ILCD requirements in its presentation of inventory details or impact assessment results. It does not distinguish between emissions to fresh water and to seawater, and it is therefore not possible to calculate the impacts on human health and ecosystems as well as fresh water eutrophication, because the receiving compartment is an important element in determining the fate and effect of an emission. Further details can be found in Table 40, where the environmental impact of 1 kg of PP and the proposed ranges of variation within each impact category are shown.

Table 66. Cradle-to-gate LCA results of petrochemical PET bottle for 1 FU (=1 kg).

Impact category	Unit	Total
Climate change	kg CO_2 eq.	3.16E+00
Particulate matter	kg PM2.5 eq.	1.54E-03
Photochemical ozone formation	kg NMVOC eq.	9.50E-03
Acidification	molc H+ eq.	1.67E-02
Terrestrial eutrophication	molc N eq.	2.70E-02
Marine eutrophication	kg N eq.	2.76E-03
Water use	m^3	4.95E+00
Abiotic depletion (fossil fuels)	MJ	8.01E+01
Non-renewable energy use. (NREU)	MJ	8.91E+01

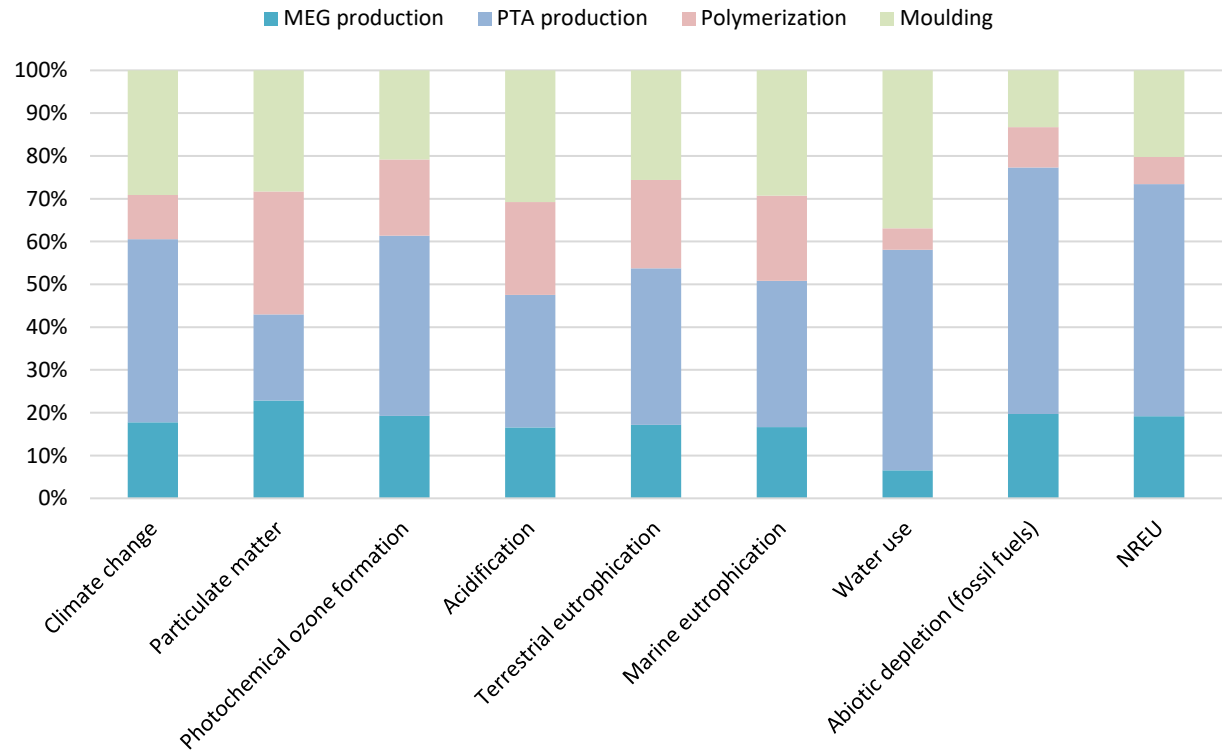


Figure 40. Breakdown of the cradle-to-gate LCA results for petrochemical PET bottle.

In Figure 40, the impacts of petrochemical PET bottles are broken down into the main unit processes of MEG production, PTA production, polymerisation, and moulding. It is seen that

PTA production is the major contributing process in categories – namely, climate change (43 %), water use (52 %), abiotic depletion (fossil fuels) (58 %) and NREU (54 %). In petrochemical ozone formation the contribution is also significant at 42 %. It also contributes significantly in other categories (the impact ranges from 20 % to 37 %).

Stretch blow moulding which has electricity use as the major activity, has a significant contribution in all categories, especially water use (37 %). The moulding process contributes to remaining categories in the range of 20-31 % (except for abiotic depletion (fossil fuels), where the contribution is 13 %).

MEG-production and polymerisation show contributions in the range of 7-23 % and 5-29 %, respectively.

3.3 End of Life results and interpretation of the beverage bottles

The results for the EoL impact assessment modelled in EASETECH are presented in this paragraph. The presentation includes (1) the mass and energy flows of all product systems in all EoL technologies, (2) the LCIA results for each product system (3) a comparison between product systems, and (4) the EoL mix.

In Annex 4 further results are presented, the contribution analysis in Figure 216 ff. and weighted results in Table 225 ff. The contribution analysis for each EoL technology is presented in Annex 4 and analysed under each product system.

3.3.1 Mass and energy flows

The mass and energy flows are presented in Table 67 and Table 68. The results correspond to the circular footprint formula in PEFCR guidance draft version 6.3 (European Commission, 2017), and in addition the paragraph looks at the carbon flows. The flows correspond to those presented in the flow charts for each technology in the approach and methodology chapter. The only difference between the product systems in the mass and energy flows for the beverage bottles is the distribution between biogenic and fossil carbon.

The tables show that 73 % of the EoL-reference flow (the collected plastic for recycling) is recycled, and that recycling results in 60 % of the plastic substituting virgin plastic for both product systems (calculated based on a sorting and technology efficiency as well as a market response, considering the decrease in quality of the plastic - see the methodology for further explanation). For incineration and landfill there is no material recovery.

For the incineration technology 31 % of the energy is recovered as heat/electricity. Plastic rejected in the sorting facility at the recycling plant is sent for incineration. For this flow 7.5 % of the energy input is recovered.

In the landfill scenario all the carbon, both biogenic and fossil, is assumed to be stored in the landfill.

In the incineration scenario the carbon is mainly emitted as CO_2 air emissions. Only 0.1 % of the carbon ends up in the bottom ash. In the recycling scenario the carbon is embedded in the recycled plastic.

Table 67. Material and energy flow for EoL of 1 kg 30 % bio-PET beverage bottles with 100 g contamination. Box letters refer to flow charts in the approach and methodology chapter (Figure 8 to Figure 13).

30 % BIO-PET							
Techno-logy	Box	Process	Mass kg	Bio carbon kg	Fossil carbon kg	Gas m^3 CH_4	Energy substitu-tion MJ
Material	I	Material input	1.10E+00	2.04E-01	4.77E-01		2.44E+01
Recycling	O	Recycled plastic	8.04E-01	1.54E-01	3.60E-01		
	S	Substituted plastic	6.51E-01	1.25E-01	2.92E-01		
	R, E, H	Rejects	2.97E-01	5.00E-02	1.17E-01		1.84E+00
Incineration w/wo energy recovery *	E, H	Energy production	1.10E+00	2.04E-01	4.77E-01		7.53E+00
	R1	Fly Ash	2.85E-03	0.00E+00	0.00E+00		
	R2	Bottom ash	9.17E-03	2.04E-04	4.77E-04		
	D	Direct emissions		2.04E-01	4.76E-01		
Landfill	L	Leachate	2.20E+00	0.00E+00	0.00E+00		
	G, E	Landfill gas				0.00E+00	0.00E+00
	CS	Storage in landfill	1.10E+00	2.04E-01	4.77E-01		

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied with 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

Table 68. Material and energy flow for EoL of 1 kg petrochemical PET beverage bottles with 100 g contamination. Box letters refer to flow charts in the approach and methodology chapter (Figure 8 to Figure 13).

PET						
Technology	Box	Process	Mass kg	Fossil carbon kg	Gas m^3 CH_4	Energy substitu-tion MJ
Material	I	Material input	1.10E+00	6.81E-01		2.44E+01
Recycling	O	Recycled plastic	8.04E-01	5.14E-01		
	S	Substituted plastic	6.51E-01	4.16E-01		
	R, E, H	Rejects	2.97E-01	1.67E-01		1.84E+00
Incineration w/wo energy recovery*	E, H	Energy production	1.10E+00	6.81E-01		7.53E+00
	R1	Fly Ash	2.85E-03	0.00E+00		
	R2	Bottom ash	9.17E-03	6.81E-04		
	D	Direct emissions		6.80E-01		
Landfill	L	Leachate	2.20E+00	0.00E+00		
	G, E	Landfill gas			0.00E+00	0.00E+00
	CS	Storage in landfill	1.10E+00	6.81E-01		

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied with 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

3.3.2 LCIA for product systems

The impact assessment results are presented per the EoL-reference flow, i.e. 1 kg plastic and 100 g of contamination for each EoL technologies (assuming 100 % disposal via that technology), see Table 68.

The overall LCIA results for both the bio-based PET and petrochemical PET are presented in Table 69 for all EoL technologies. The results are commented upon and further explained in the following sections.

Table 69. Total results of treating 1 kg plastic (incl. 100 g contamination) for all impact categories. Highest impact within each impact category is red and lowest impact green. The intended EoL option is marked with a bold box.

Impact category (Unit)	Bio-based PET (30 %)			Petrochemical PET		
	Recycling	Incinera-tion	Land-filling	Recycling	Incinera-tion	Land-filling
Climate change – fossil (kg CO_2 eq.)	-7.25E-01	1.37E+00	1.36E-02	-5.41E-01	2.11E+00	1.36E-02
Climate change – biogenic (kg CO_2 eq.)	1.14E-01	6.83E-01	9.84E-04	-6.90E-02	-6.50E-02	9.84E-04
Climate change – total (kg CO_2 eq.)	-6.10E-01	2.05E+00	1.46E-02	-6.10E-01	2.05E+00	1.46E-02
Ozone depletion (kg CFC-11 eq.)	-6.85E-06	-3.73E-08	2.80E-10	-6.85E-06	-3.73E-08	2.80E-10
Human toxicity, non-cancer effects (CTUh)	-6.12E-08	-1.12E-08	3.88E-09	-6.12E-08	-1.12E-08	3.88E-09
Human toxicity, cancer effects (CTUh)	1.13E-09	-2.00E-09	5.83E-11	1.13E-09	-2.00E-09	5.84E-11
Particulate matter (kg PM2.5 eq.)	-2.44E-04	-6.29E-05	3.01E-06	-2.44E-04	-6.29E-05	3.01E-06
Ionizing radiation HH (kBq U235 eq.)	-1.78E-02	5.17E-02	2.58E-04	-1.78E-02	5.17E-02	2.58E-04
Photochemical ozone formation (kg NMVOC eq.)	-3.29E-03	8.58E-04	4.44E-05	-3.29E-03	8.58E-04	4.44E-05
Acidification (molc H+ eq.)	-3.58E-03	1.57E-03	4.71E-05	-3.58E-03	1.57E-03	4.71E-05
Terrestrial eutrophication (molc N eq.)	-6.74E-03	5.48E-03	1.80E-04	-6.74E-03	5.48E-03	1.80E-04
Freshwater eutrophication (kg P eq.)	-7.11E-05	1.07E-05	2.93E-07	-7.11E-05	1.07E-05	2.93E-07
Marine eutrophication (kg N eq.)	-6.53E-04	4.43E-04	5.92E-05	-6.53E-04	4.43E-04	5.92E-05
Freshwater ecotoxicity (CTUe)	7.57E-01	-1.36E-01	8.94E-03	7.57E-01	-1.36E-01	8.94E-03
Land use (kg C deficit)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use (m^3)	-2.89E-01	-1.86E-01	1.36E-02	-2.89E-01	-1.86E-01	1.36E-02
Abiotic depletion (kg Sb eq.)	-5.76E-08	-2.54E-07	3.19E-09	-5.76E-08	-2.54E-07	3.19E-09
Abiotic depletion	-3.74E+01	-7.06E+00	2.02E-01	-3.74E+01	-7.06E+00	2.02E-01

(fossil fuels) (MJ)						
NREU (MJ)	-3.68E+01	-8.11E+00	1.85E-01	-3.68E+01	-8.11E+00	1.85E-01

3.3.3 LCIA for baseline and reference results: PET

The results indicate that recycling is the preferred EoL technology for PET beverage bottles as the majority of impact categories perform better (green colour) than incineration and landfilling. Exceptions are the impact categories of human toxicity (cancer effects), freshwater ecotoxicity and abiotic depletion, where incineration performs better.

Based on the results presented in Table 69, it is difficult to rank the environmental performance of incineration and landfilling of PET bottles.

Further details on the individual EoL options for the PET beverage bottles are presented below, looking into specific impact categories and contribution of impacts from different processes and elemental emissions.

3.3.3.1 Further discussion of the intended scenario (recycling)

In the recycling scenario, the substitution of PET contributes to the largest part of the savings in all impact categories. Of the actual impacts in this scenario, the plastic recycling plant contributes significantly to the impacts due to consumption of electricity and process stream from natural gas.

Incineration of rejects in the recycling scenarios also contributes significantly to some impact categories. In the recycling scenarios, 24.5 % of the plastic is incinerated and the emissions originate both from direct emissions from waste incineration as well as treatment of bottom ash and fly ash. Direct emissions from incineration also constitute a significant share of climate change impacts.

3.3.3.2 Contribution to climate change impacts

For the climate change impact category, the three EoL technologies for PET (biogenic and fossil carbon) are compared in Figure 41, where the contribution of grouped processes can be observed.

For the recycling scenario, substitution of PET has the largest climate change savings, while incineration of plastic causes the largest climate change impacts due to the direct emissions.

For the incineration scenario, the direct emissions cause the largest impacts, while energy substitution causes climate change savings. Bottom ash and fly ash contribute to the largest part of the freshwater ecotoxicity (no data shown in the figure).

Landfilling of PET has very little impact on climate change, which was also observed in other impact categories such as acidification, freshwater eutrophication and ionizing radiation. As PET is non-degradable, no landfill gas is generated, limiting the environmental impacts from the landfilling. The very low climate change impact from landfill EoL scenario (not visible in the figure) is from construction and operation of the landfill and transportation of the waste.

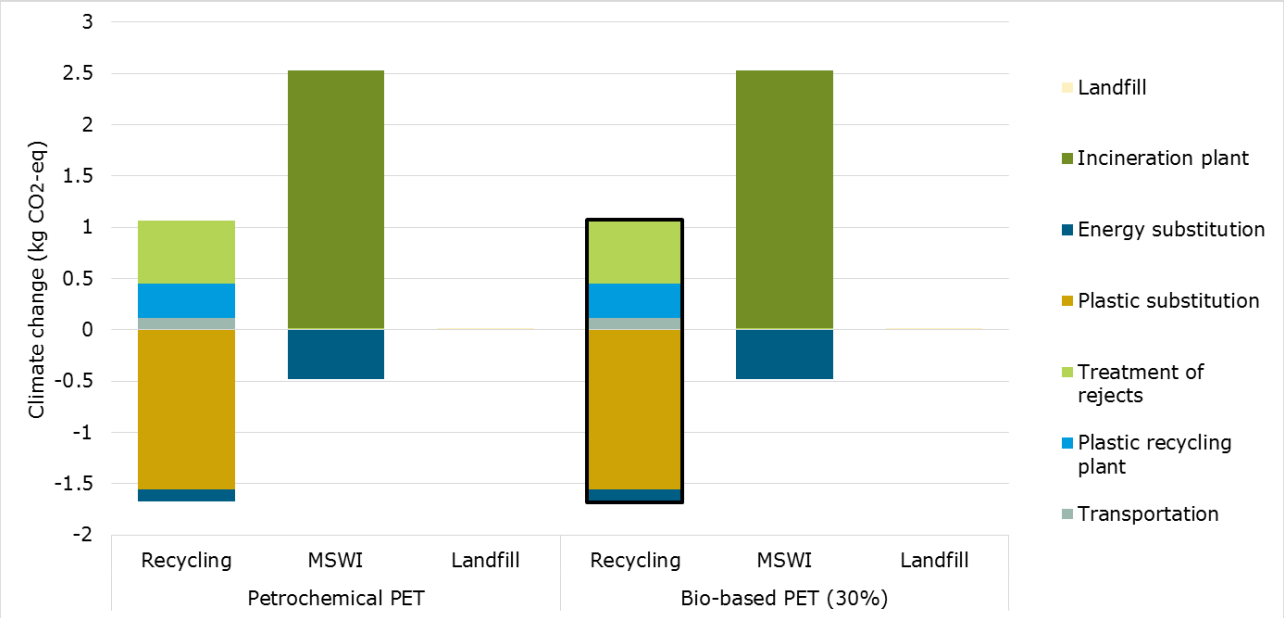


Figure 41. Climate change for treating 1 kg PET bottles (incl. 100 g contamination) in different EoL technologies. The intended EoL option is marked with a black box. See Annex 4 for details of the content of the grouped processes. Results for incineration are presented with and without energy recovery. The results without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amounts of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

3.3.3.3 Contribution analysis

The contribution analysis reflects the relative impact of each process within each EoL technology for all product systems. The contribution analysis figures are presented in Annex 4.

The results of the contribution analysis for the two product systems are very similar, the only difference being the contribution of biogenic and fossil climate change. The contribution analysis of the intended EoL technology of the beverage bottles (recycling) shows that the savings in impacts from plastic substitution have the largest contribution in 16 of 19 impact categories on the overall LCIA results. It is only in the impact categories of freshwater ecotoxicity and human toxicity, non-cancer, that the treatment of rejects has a larger contribution due to emission of metals to water and soil. Plastic recycling is responsible for 38 % of the contribution in water use, from washing the plastic, and the highest contribution in abiotic depletion (35 %). Moreover, transportation has a small contribution in all categories, the highest being 20 % in human toxicity (non-cancer).

For landfilling of PET beverage bottles only three processes contribute to the overall LCIA results: Construction and operation, leachate treatment and transport. Transport has the largest contribution in impact categories such as NREU, abiotic depletion (fossil fuels), terrestrial eutrophication, acidification, photochemical ozone formation human toxicity, non-cancer and fossil climate change. The leachate treatment has the largest contribution in freshwater ecotoxicity (51 %), mostly from metals emitted to soil and water and marine eutrophication (74 %) due to ammonium emitted to surface water.

Looking into the contribution analysis for incineration of PET beverage bottles, direct emissions from incineration as well as electricity and heat substitution constitute the largest part of the

overall LCIA results. The direct emissions from incineration have the largest contribution to total climate change impact potentials (more biogenic climate change impact for the bio-based PET and fossil PET). Moreover, the direct emissions contribute a large impact on photochemical ozone formation, terrestrial and marine eutrophication. In the impact categories of NREU (74 %), abiotic depletion (fossil fuels) (61 %), freshwater eutrophication (63 %) and ionizing radiation (79 %) the heat substitution has the highest contribution. Electricity substitution contributes most in human toxicity, both with and without cancer effects (79 % and 66 %), particulate matter (67 %), water use (68 %) and abiotic depletion (78 %).

3.3.4 Weighted results

The result of the weighting of the EoL technologies is showed in Figure 42 (numerical results are presented in Annex 4). For methodology and choice of normalization and weighing factors, please see the Approach and methodology chapter, section 5.2.3 "Use of normalised and weighted results to determine the preference of EoL option" and the values in Table 30.

According to the weighted results, recycling of PET beverage bottles is best, followed by landfilling and then incineration. The same results apply for both product systems.

There is a large saving in the abiotic depletion (fossil fuels) when recycling from substituting PET. Furthermore, there are considerable contributions from ozone depletion, which stems from heat and electricity substitution from incinerating the rejects of the recycling plant. The largest contribution to the weighted results is the climate change impact category for incineration, which stems from direct emissions at the incineration plant. For landfilling the largest contribution is from climate change and abiotic depletion of fossil fuels, which is due to transportation and the operation of the landfill.

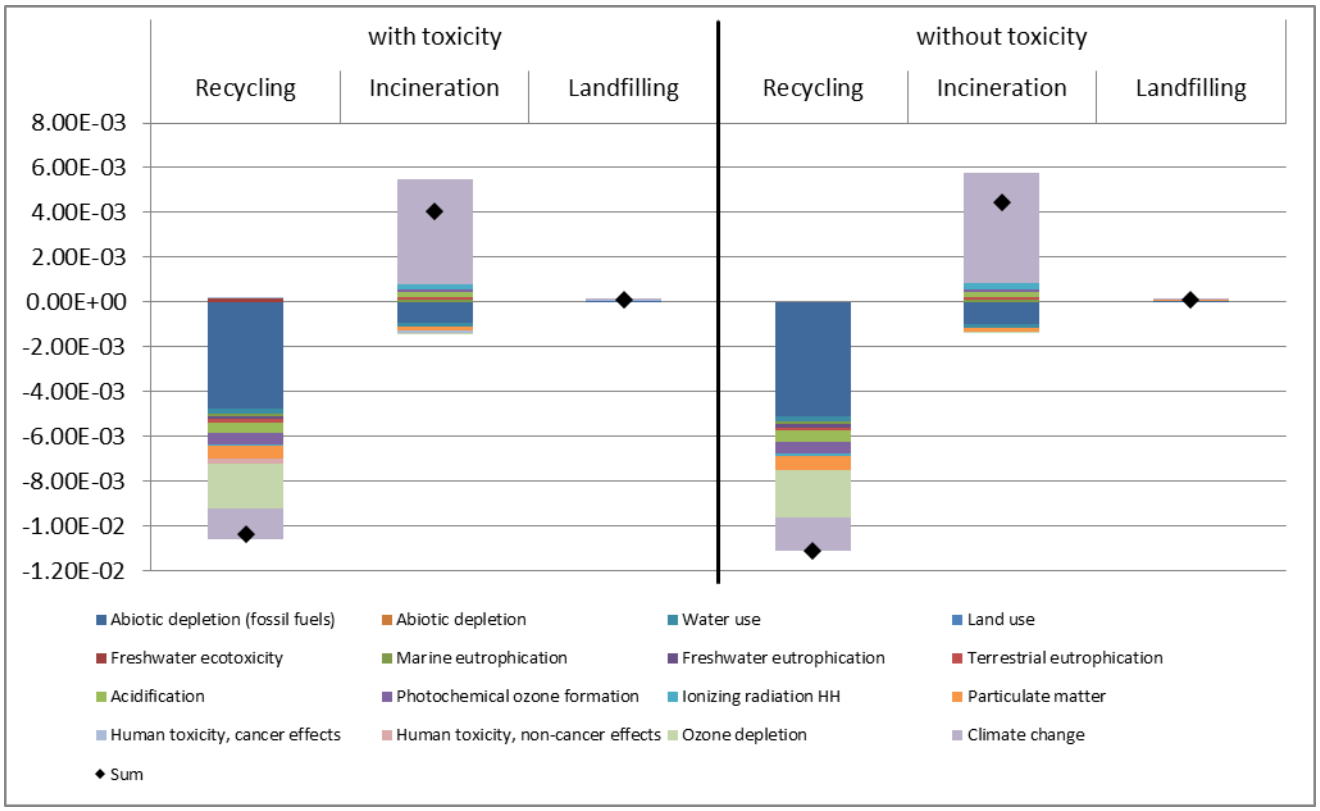


Figure 42. Weighted EoL results for the baseline product system, bio-based PET beverage bottles per 1 kg of plastic. Numerical results can be observed in Annex 4.

3.3.5 End of life mixed results

The LCIA results were calculated with the chosen EU mix of EoL technologies for PET plastic being 60 % is recycled, 20 % incinerated and 20 % landfilled. The results are presented in Table 70.

The total climate change impact of the EoL mix was calculated to 0.0465 kg CO_2-eq per kg plastic. As previously shown for the individual EoL technologies, the results of the EoL mix show the same results (except for a different distribution between biogenic and fossil carbon) for the bio-based and fossil PET beverage bottles.

Table 70. LCIA for EoL mix of the beverage bottles for all product systems per functional unit (100 bottles – 1 kg of plastic).

Impact category	Unit	Bio-based PET	Petrochemical PET
Plastics recycling	%	60	60
MSWI with and energy recovery	%	20	20
Landfilling	%	20	20
Climate change – fossil	kg CO_2 eq.	-1.59E-01	1.01E-01
Climate change - biogenic	kg CO_2 eq.	2.05E-01	-5.42E-02
Climate change - total	kg CO_2 eq.	4.65E-02	4.65E-02
Ozone depletion	kg CFC-11 eq.	-4.11E-06	-4.11E-06
Human toxicity, non-cancer effects	CTUh	-3.82E-08	-3.82E-08
Human toxicity, cancer effects	CTUh	2.89E-10	2.89E-10
Particulate matter	kg PM2.5 eq.	-1.58E-04	-1.58E-04
Ionizing radiation HH	kBq U235 eq.	-3.05E-04	-3.05E-04
Photochemical ozone formation	kg NMVOC eq.	-1.80E-03	-1.80E-03
Acidification	molc H+ eq.	-1.82E-03	-1.82E-03
Terrestrial eutrophication	molc N eq.	-2.91E-03	-2.91E-03
Freshwater eutrophication	kg P eq.	-4.05E-05	-4.05E-05
Marine eutrophication	kg N eq.	-2.91E-04	-2.91E-04
Freshwater ecotoxicity	CTUe	4.29E-01	4.29E-01
Land use	kg C deficit	0.00E+00	0.00E+00
Water use	m^3	-2.08E-01	-2.08E-01
Abiotic depletion	kg Sb eq.	-8.47E-08	-8.47E-08
Abiotic depletion (fossil fuels)	MJ	-2.38E+01	-2.38E+01
NREU	MJ	-2.36E+01	-2.36E+01

3.4 Cradle-to-grave results and interpretation

3.4.1 Bio-Based baseline system (Bio-Based PET bottle from Brazilian sugarcane)

The cradle-to-grave results of bio-based PET bottle are given in Table 71. The breakdown of impacts for the baseline product system between biomass production, manufacturing and EoL is shown in Figure 43. The EoL mix is 60 % recycling, 20 % incineration and 20 % landfilling. Biomass production has minor impacts except for the particulate matter. At EoL stage savings are achieved in almost all impact categories owing to the plastic substitution with recycling. Major savings are observed for the ozone depletion, abiotic depletion (fossil fuels) and NREU categories. Increasing recycling can further enhance the benefits.

Table 71. Cradle-to-grave LCA results of bio-based PET bottle (baseline) for 1 functional unit (=1 kg) of PET bottles (excluding DLUC and ILUC effects).

Impact category	Unit	Biomass	Manufacturing	EoL
Climate change	kg CO_2 eq.	-0.25	3.48	0.05
Ozone depletion	kg CFC-11 eq.	1.56E-08	1.04E-05	-4.11E-06
Human toxicity, non-cancer effects	CTUh	2.95E-07	7.34E-07	-3.82E-08
Human toxicity, cancer effects	CTUh	8.00E-09	1.30E-07	2.89E-10
Particulate matter	kg PM2.5 eq.	6.85E-03	3.04E-03	-1.58E-04
Ionizing radiation	kBq U235 eq.	2.70E-03	5.80E-01	-3.05E-04
Photochemical ozone formation	kg NMVOC eq.	2.52E-03	1.26E-02	-1.80E-03
Acidification	molc H+ eq.	3.16E-03	2.29E-02	-1.82E-03
Terrestrial eutrophication	molc N eq.	1.49E-02	4.10E-02	-2.91E-03
Freshwater eutrophication	kg P eq.	2.07E-05	1.34E-03	-4.05E-05
Marine eutrophication	kg N eq.	7.83E-04	4.03E-03	-2.91E-04
Freshwater ecotoxicity	CTUe	1.37E+00	1.61E+01	4.29E-01
Land use	kg C deficit	7.43E+00	3.40E+00	0.00E+00
Water use	m^3	2.90E-02	5.27E+00	-2.08E-01
Abiotic depletion	kg Sb eq.	5.07E-07	1.30E-06	-8.47E-08
Abiotic depletion (fossil fuels)	MJ	1.27	72.81	-23.81
NREU	MJ	1.30	80.96	-23.65

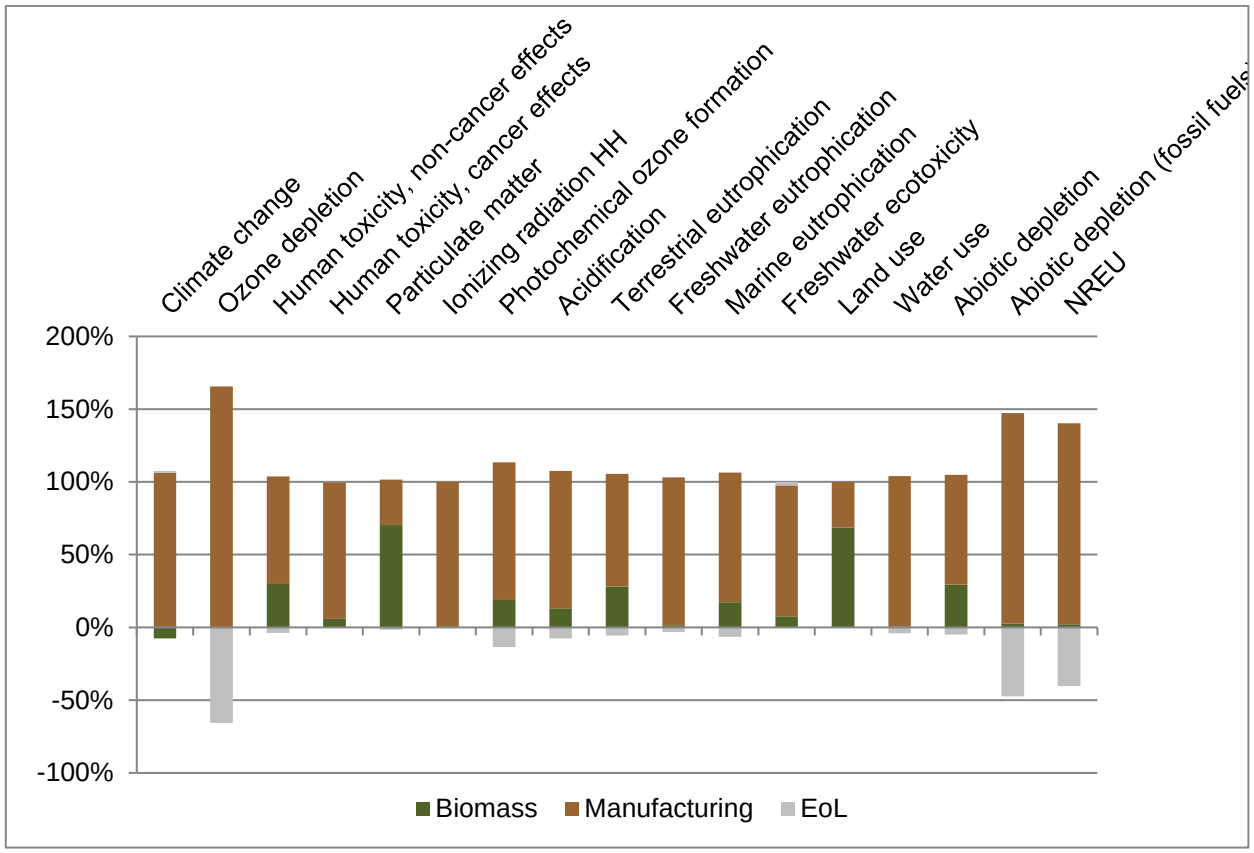
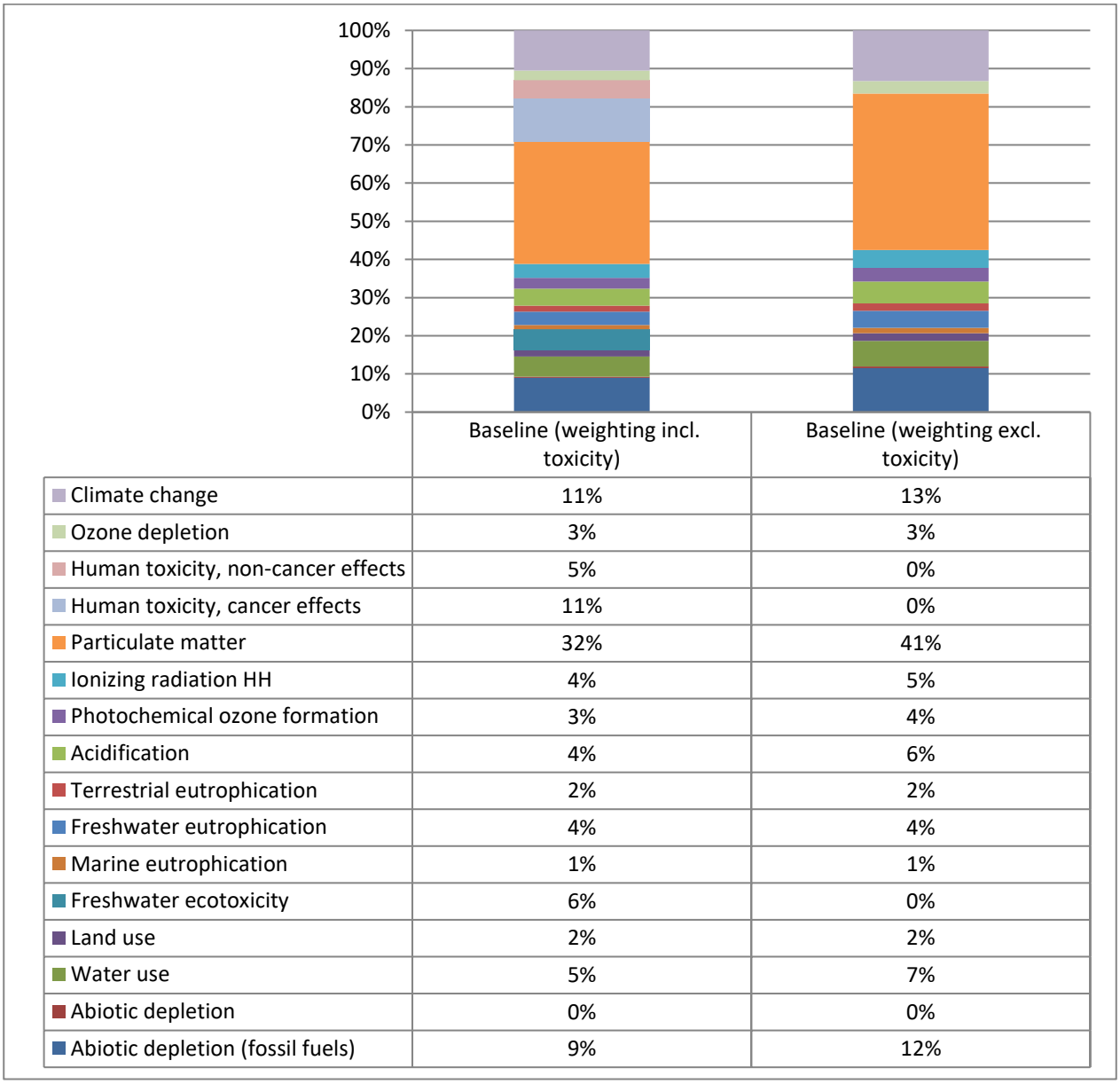


Figure 43. Breakdown of the potential impacts from bio-based PET bottles (baseline) across all impact categories, cradle-to-grave results (excluding DLUC and ILUC effects).

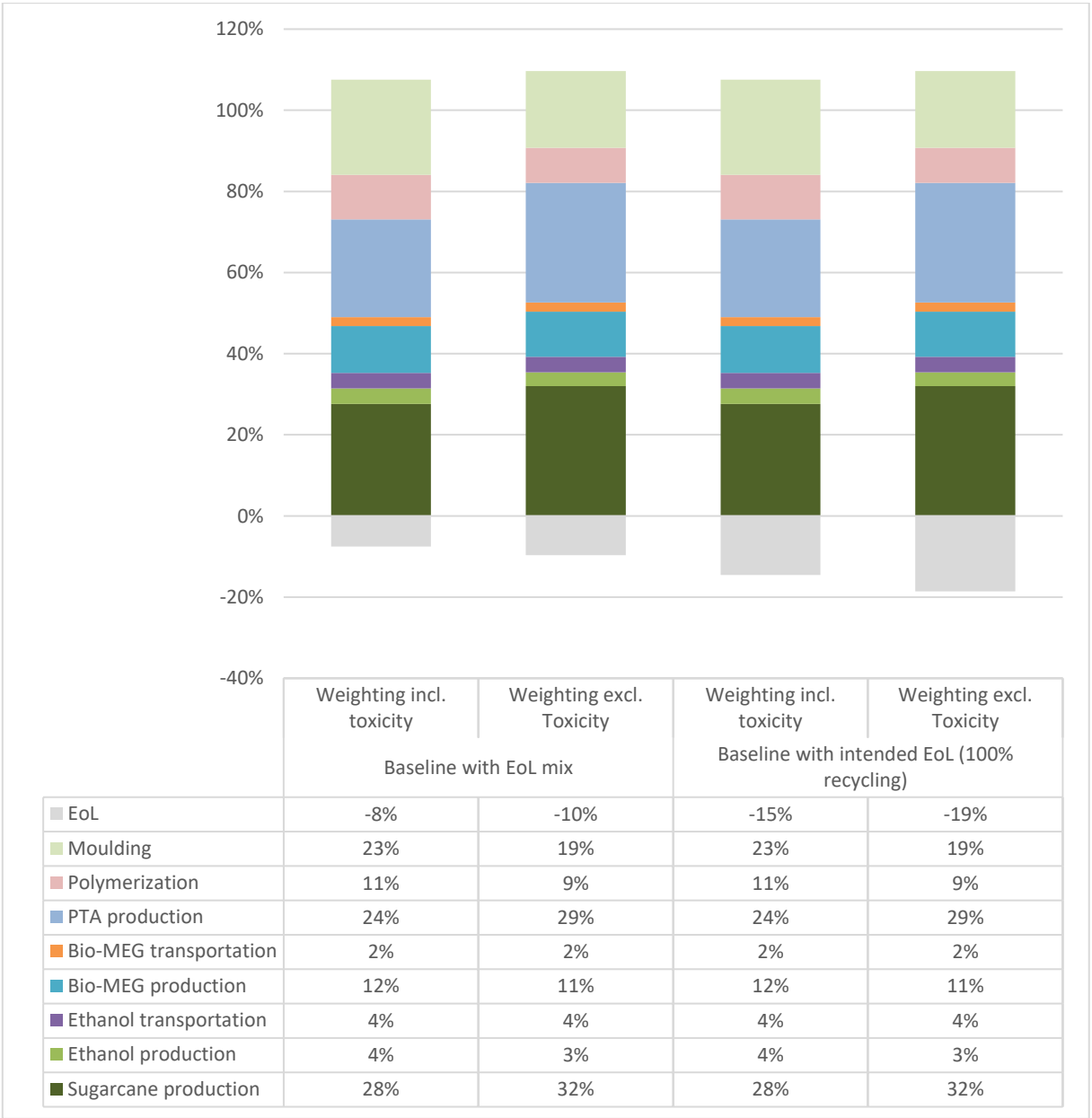
According to PEFCR guidance v 6.3, the most relevant impact categories are identified as all impact categories that cumulatively contribute to at least 80 % of the total environmental impacts (excluding toxicity related impact categories). Based on the normalised and weighted results shown in Figure 44, excluding toxicity-related impacts, the most relevant impact categories are: particulate matter (41 %), climate change (13 %), abiotic depletion (fossil fuels) (12 %), water use (7 %) and acidification (6 %) forming a cumulative 80 % contribution to total impacts. These impact categories are identified as the hot spots for this case study and detailed analysis is made for each of them in section 3.1.1. The rest of the impact categories contribute each up to 5 %.



	Baseline (weighting incl. toxicity)	Baseline (weighting excl. toxicity)
Climate change	11%	13%
Ozone depletion	3%	3%
Human toxicity, non-cancer effects	5%	0%
Human toxicity, cancer effects	11%	0%
Particulate matter	32%	41%
Ionizing radiation HH	4%	5%
Photochemical ozone formation	3%	4%
Acidification	4%	6%
Terrestrial eutrophication	2%	2%
Freshwater eutrophication	4%	4%
Marine eutrophication	1%	1%
Freshwater ecotoxicity	6%	0%
Land use	2%	2%
Water use	5%	7%
Abiotic depletion	0%	0%
Abiotic depletion (fossil fuels)	9%	12%

Figure 44. Contribution of the 16 PEF categories based on normalised and weighted results of bio-based PET bottles (baseline system), with and without toxicity categories.

The contributions of the life cycle stages and the different processes within the baseline system are shown in Figure 45. The most relevant life cycle stage is manufacturing (72/74 % cumulative contribution). The most relevant processes in this stage are PTA production (24/29 %), moulding (23/19 %) and bio-MEG production (12/11 %). Minor impacts (up to 5 %) are observed in ethanol production. The sugarcane production stage has a significant contribution (28/32 %) whereas the contribution of the transportation stage is low (6 %). The EoL stage provides savings of 8/10 % of the impacts with the EoL mix (60 % recycling, 20 % incineration and 20 % landfill). The effect of having 100 % recycling as EoL is also shown in the figure. This enables an additional 7-9 % reduction in the impacts. Section 3.1.1 describes in detail the contribution of the processes for the bio-based PET bottle system.



	Baseline with EoL mix		Baseline with intended EoL (100% recycling)	
	Weighting incl. toxicity	Weighting excl. Toxicity	Weighting incl. toxicity	Weighting excl. Toxicity
EoL	-8%	-10%	-15%	-19%
Moulding	23%	19%	23%	19%
Polymerization	11%	9%	11%	9%
PTA production	24%	29%	24%	29%
Bio-MEG transportation	2%	2%	2%	2%
Bio-MEG production	12%	11%	12%	11%
Ethanol transportation	4%	4%	4%	4%
Ethanol production	4%	3%	4%	3%
Sugarcane production	28%	32%	28%	32%

Figure 45. Contribution of the life cycle stages and processes based on normalised and weighted results of bio-based PET bottles (baseline system), with and without toxicity categories.

3.4.2 Petrochemical reference system

The cradle-to-grave LCA results for one functional unit (=1 kg) of petrochemical PET bottles are presented in Table 72. Based on Chapter 6 *Ranges for environmental impacts from production of fossil-based plastics*, 9 out of 17 impact categories were found suitable for petrochemical PET interpretation, namely: climate change, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication, marine eutrophication, water use, abiotic depletion (fossil fuels) and NREU. The breakdown of impacts between manufacturing and EoL is shown in Figure 46. The estimated EoL mix for petrochemical PET bottles is 60 % recycling, 20 % incineration and 20 % landfilling (see section 2.2).

Table 72. Cradle-to-grave LCA results of 1 functional unit petrochemical PET bottles.

Impact category	Unit	Manufacturing	EoL	Total
Climate change	kg CO_2 eq.	3.16E+00	4.65E-02	3.20E+00
Particulate matter	kg PM2.5 eq.	1.54E-03	-1.58E-04	1.38E-03
Photochemical ozone formation	kg NMVOC eq.	9.50E-03	-1.80E-03	7.71E-03
Acidification	molc H+ eq.	1.67E-02	-1.82E-03	1.49E-02
Terrestrial eutrophication	molc N eq.	2.70E-02	-2.91E-03	2.41E-02
Marine eutrophication	kg N eq.	2.76E-03	-2.91E-04	2.47E-03
Water use	m^3	4.95E+00	-2.08E-01	4.5E+00
Abiotic depletion (fossil fuels)	MJ	8.01E+01	-2.38E+01	5.63E+01
Non-renewable energy use. (NREU)	MJ	8.91E+01	-2.36E+01	6.55E+01

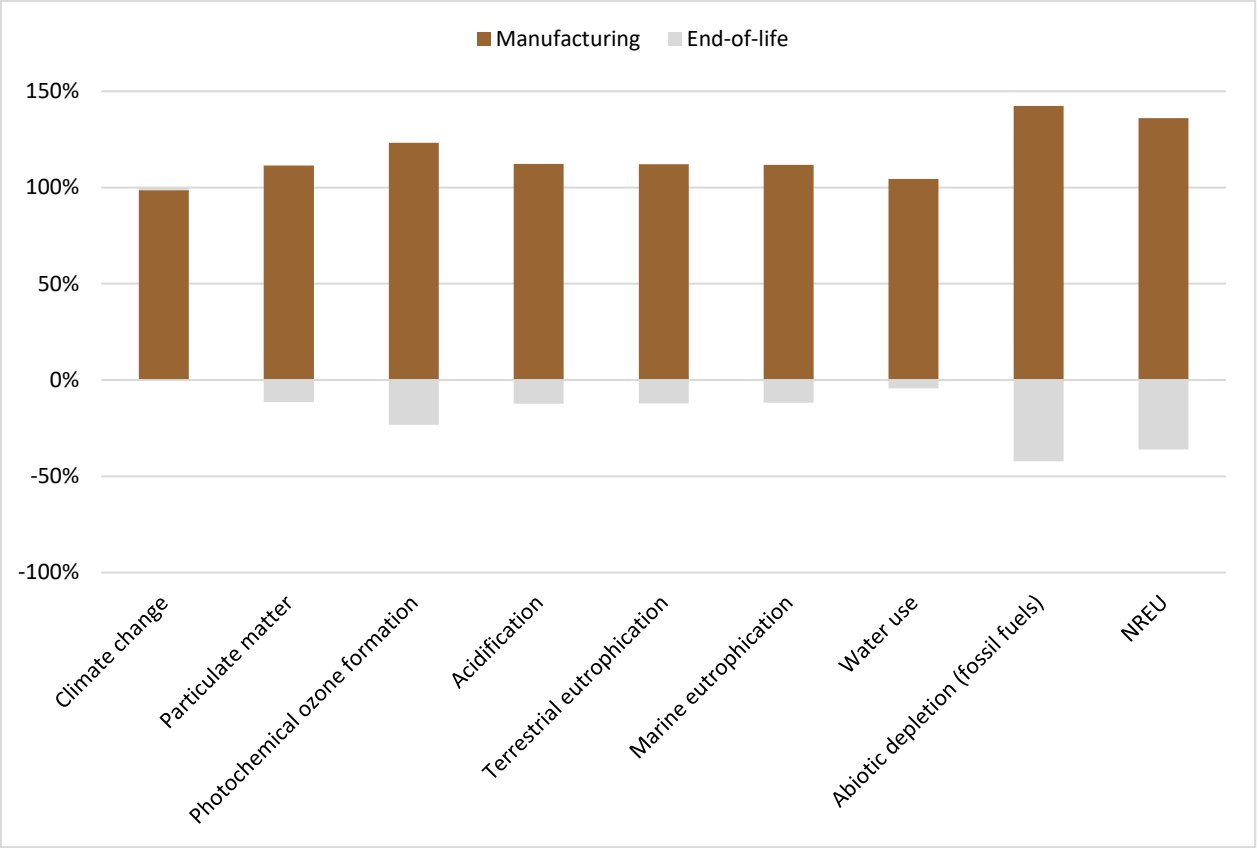
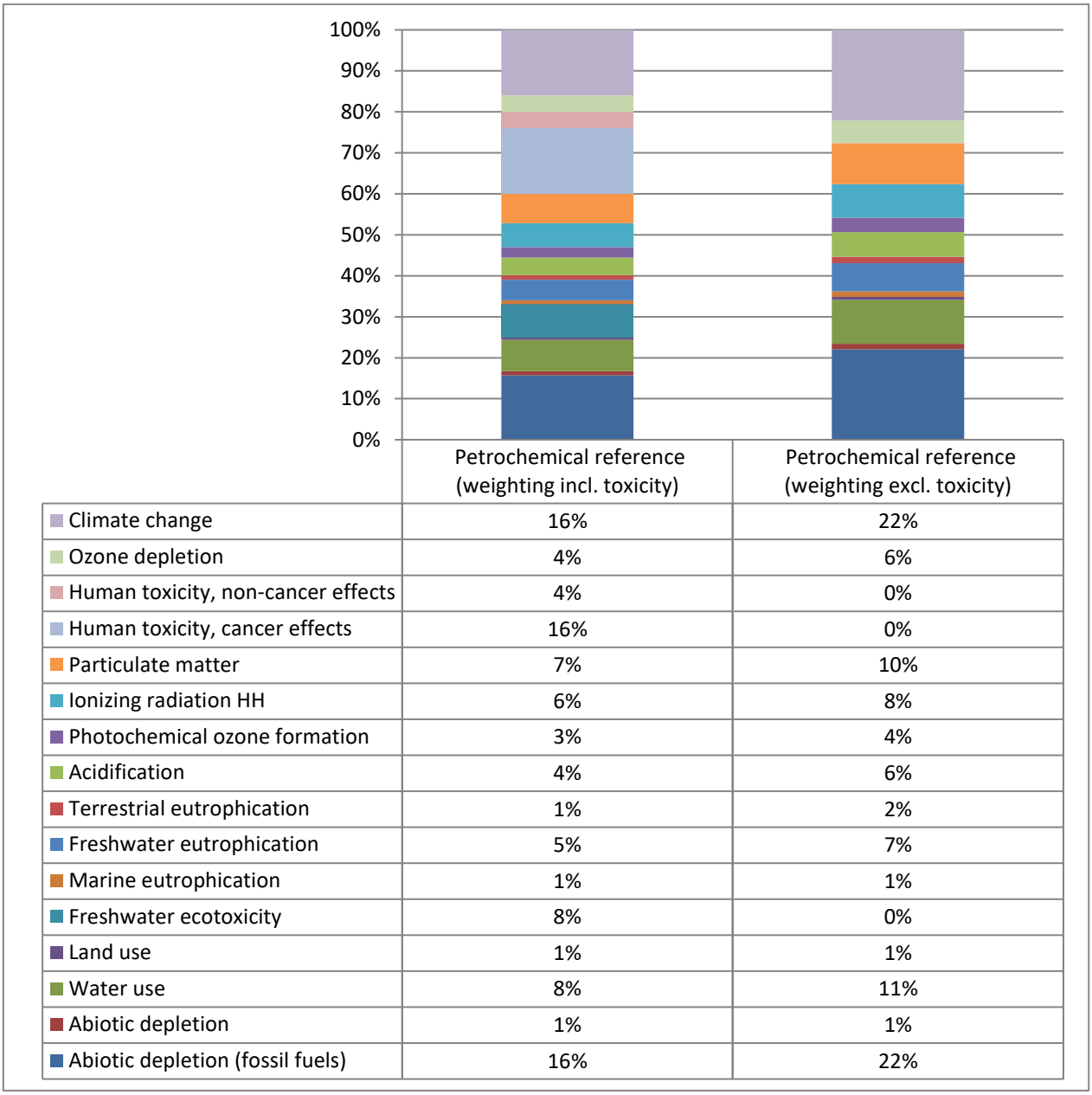


Figure 46. Breakdown of the potential impacts from petrochemical PET bottles across all impact categories, cradle-to-grave results.

It can be seen from the results (Figure 46) that the EoL stage provides savings in all nine impact categories. Significant savings are seen in the categories of photochemical ozone formation (23 %), abiotic depletion (fossil fuels) (42 %) and NREU (36 %). The savings within the EoL stage are due to the substitution of plastic with recycling and the avoided heat/electricity production with incineration.

Normalisation and weighting have been applied to the reference system (petrochemical PET bottle) to understand which among the assessed impact categories are more relevant for this application and what the main environmental hotspots are. To do this, all the impact categories needed to be considered in the calculation. Hence, the weighted results are subject to considerable uncertainties, as highlighted within *Chapter 6 Ranges for environmental impacts from production of fossil-based plastics*, therefore their numerical values must be considered only as indicative.

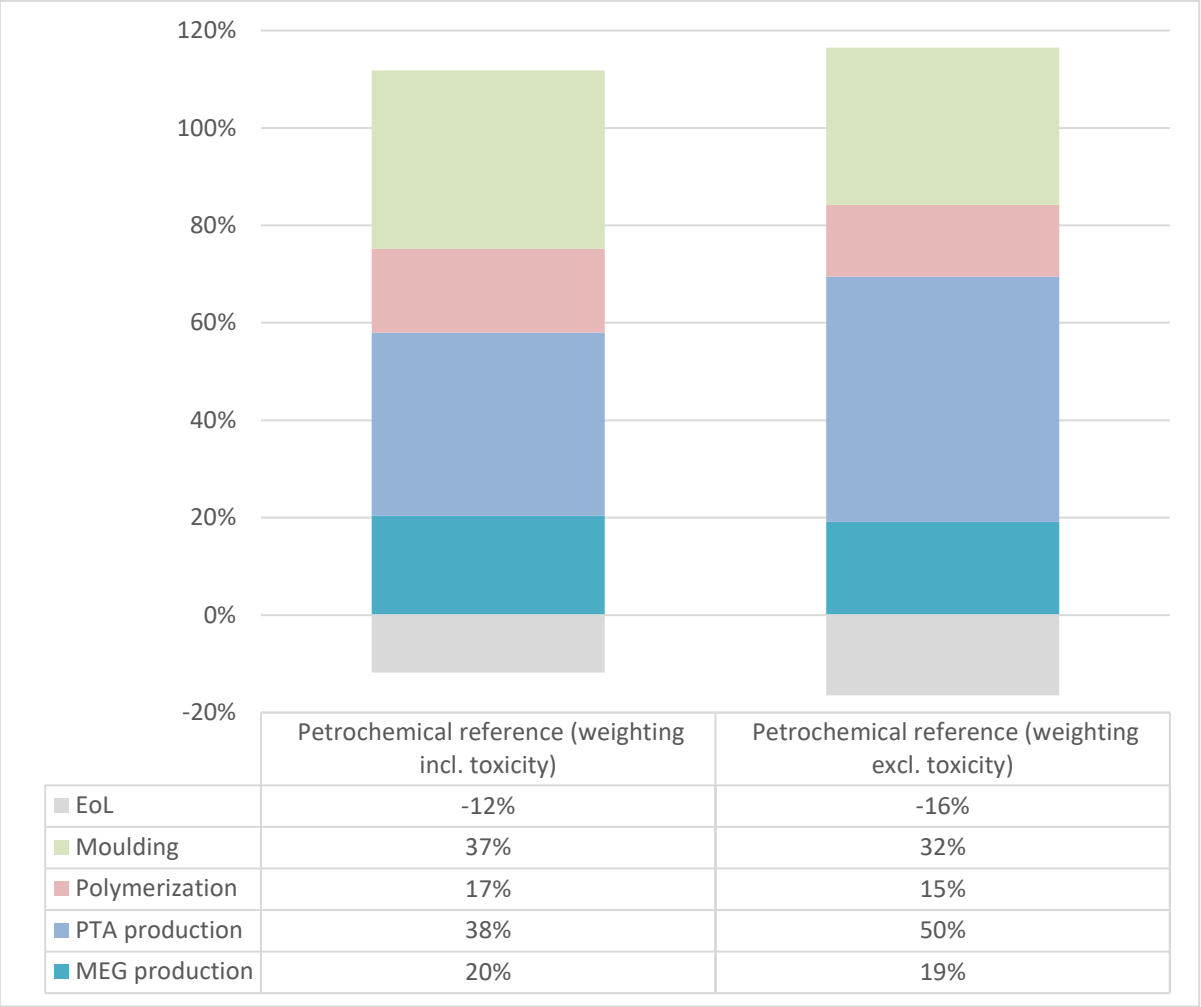
Based on the normalised and weighted results shown in Figure 47, excluding toxicity related impacts, the most relevant impact categories for the petrochemical PET bottles are: climate change (22 %), abiotic depletion (fossil fuels) (22 %), water use (11 %), particulate matter (10 %), ionizing radiation HH (8 %) and freshwater eutrophication (6 %) forming cumulatively an 80 % contribution to total impacts. The rest of the impact categories are less relevant with each contributing up to 6 %. It is seen that the top four most relevant categories are among the suitable impact categories identified in Chapter 6.



	Petrochemical reference (weighting incl. toxicity)	Petrochemical reference (weighting excl. toxicity)
Climate change	16%	22%
Ozone depletion	4%	6%
Human toxicity, non-cancer effects	4%	0%
Human toxicity, cancer effects	16%	0%
Particulate matter	7%	10%
Ionizing radiation HH	6%	8%
Photochemical ozone formation	3%	4%
Acidification	4%	6%
Terrestrial eutrophication	1%	2%
Freshwater eutrophication	5%	7%
Marine eutrophication	1%	1%
Freshwater ecotoxicity	8%	0%
Land use	1%	1%
Water use	8%	11%
Abiotic depletion	1%	1%
Abiotic depletion (fossil fuels)	16%	22%

Figure 47. Contribution of the 16 PEF categories based on normalised and weighted results of petrochemical PET bottles, with and without toxicity categories.

The contributions of the life cycle stages and the different processes within the petrochemical reference system are given in Figure 48 (based on normalisation and weighting of the results). The most relevant processes are PTA production (38/50 %) and stretch blow moulding (37/32 %). The EoL stage provides savings of 12/16 %.



	Petrochemical reference (weighting incl. toxicity)	Petrochemical reference (weighting excl. toxicity)
EoL	-12%	-16%
Moulding	37%	32%
Polymerization	17%	15%
PTA production	38%	50%
MEG production	20%	19%

Figure 48. Contribution of the life cycle stages and processes based on normalised and weighted results of petrochemical PET bottles, with and without toxicity categories.

3.4.3 Comparing the bio-based baseline system with bio-based alternative and petrochemical reference

Comparing the bio-based baseline system (from Brazilian sugarcane) with bio-based alternatives (from EU crops and wheat straw)

The comparison of cradle-to-grave results of bio-based PET bottle baseline system with alternative bio-based systems are given in Figure 49. To produce the PET bottles from cradle-to-gate, the processes involved from ethanol transportation onwards are identical for all three systems. The EoL stage of the three product systems are also the same. The only difference lies in the feedstock used and location. In the baseline system ethanol is produced in Brazil from sugarcane, in the alternative system 1 EU crops of wheat, maize and sugar beet are used and in the alternative system 2 EU wheat straw is used.

Alternative 1 (EU crops) performs the worst in all impact categories except for particulate matter, photochemical ozone formation, land use and abiotic depletion. Baseline (Brazilian sugarcane) performs the worst in these categories. The baseline system performs significantly worse in particulate matter which is due to sugarcane waste burning. For eight of the remaining impact categories their performances are similar (within 8 % difference). For acidification, human toxicity (non-cancer), freshwater ecotoxicity, terrestrial eutrophication and marine eutrophication, the baseline performs significantly better (>15 %) than alternative

1. Alternative 2 (EU wheat straw) performs the best overall. Only in the abiotic depletion impact category, does the alternative 1 product system show the lowest impacts. This is due to the high savings achieved in ethanol production from by-product credits for animal feed replacement.

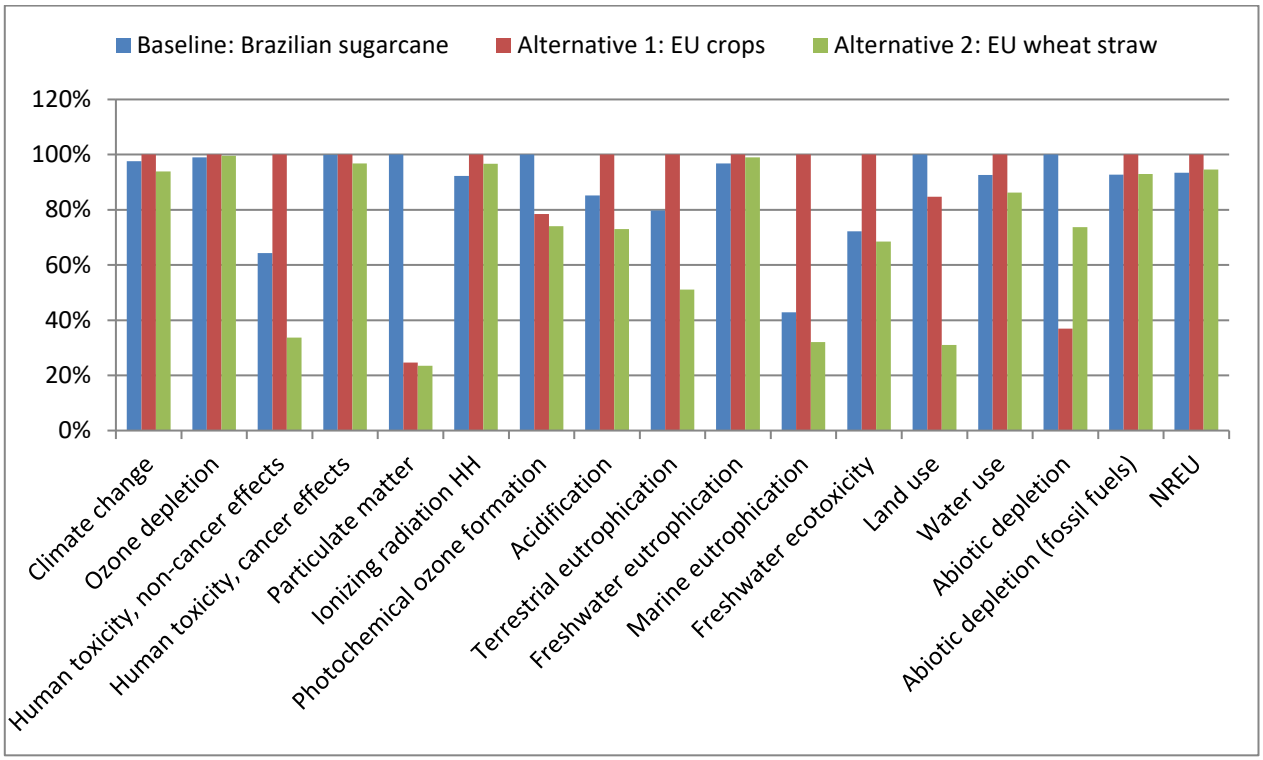


Figure 49. Comparing a bio-based PET bottle baseline (from Brazilian sugarcane) system with bio-based alternative systems (from EU crops and wheat straw), cradle-to-grave results (excluding DLUC and ILUC effects).

Comparing the bio-based systems with a petrochemical reference

The comparison of cradle-to-grave results of a bio-based PET baseline and alternatives with the petrochemical PET bottle per functional unit is provided in Figure 49. The bio-based PET bottle product systems perform similarly (within 5 % difference) in climate change (excluding DLUC and ILUC effects). In two categories (NREU and abiotic depletion (fossil fuels)) bio-based PET offers slightly lower impacts (up to 10 %) compared to the petrochemical bottles. For water use they perform similarly (within 15 % difference) especially with the alternative 2 product system. Bio-based PET bottles have higher impacts than the petrochemical PET bottles in the remaining five out of nine suitable impact categories: particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication, and marine eutrophication. Since the PTA production, polymerisation and stretch blow moulding processes are identical in both bio-based and petrochemical product systems, the differences come from biomass, ethanol and bio-MEG production and transport processes, namely; particulate matter (waste burning during harvest), photochemical ozone formation, terrestrial and marine eutrophication (direct emissions from fertiliser use, waste and bagasse burning and transportation) and acidification (bio-MEG production). However, the differences between petrochemical and bio-based PET bottle results for these impact categories are considerably decreased (from about 40 % worse to 10-20 % worse) if wheat straw is used as feedstock. The EoL stage is identical for all product systems since the PET bottle products are chemically identical. In Figure 50 the EU mix EoL scenario is used. If the intended EoL scenario (100 % recycling) would be used, the

impacts would be lowered by the same amount (about 10 % reduction in normalised and weighted results) and the comparison would not change.

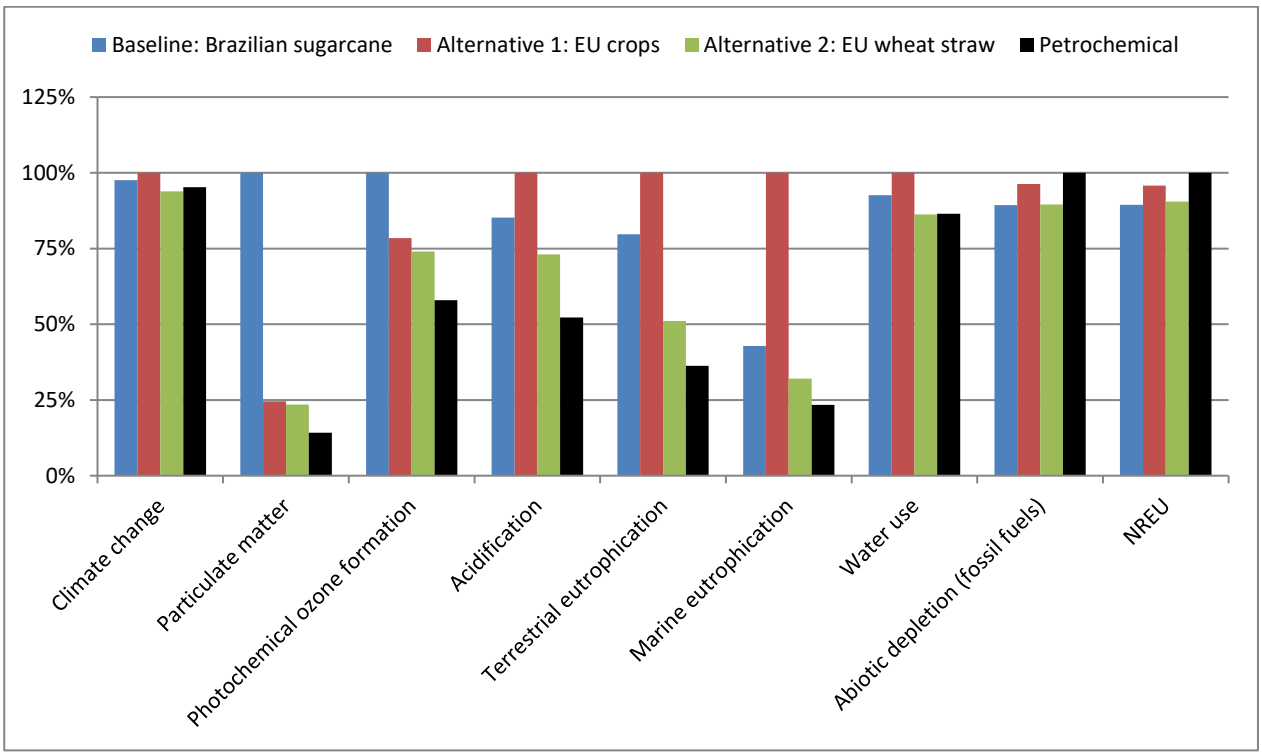


Figure 50. Comparing bio-based PET bottle baseline and alternatives with petrochemical PET bottle, cradle-to-grave results (excluding DLUC and ILUC effects).

4 Discussion

4.1 Sensitivity analysis

4.1.1 Phasing-out sugarcane waste burning

The calculations performed in the sensitivity analysis were made by removing the waste burning emissions in sugarcane cultivation and harvesting stage.

Waste burning phase-out has effects on six of the impact categories (climate change, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication and marine eutrophication). The change in the contribution of the process stages for the six impact categories are shown in Figure 51. Not surprisingly, a significant effect on particulate matter is observed (70 % reduction). This reduction makes bio-based PET have comparable results with petrochemical PET for the particulate matter category. For the other categories 2 % to 11 % reductions from the baseline are observed.

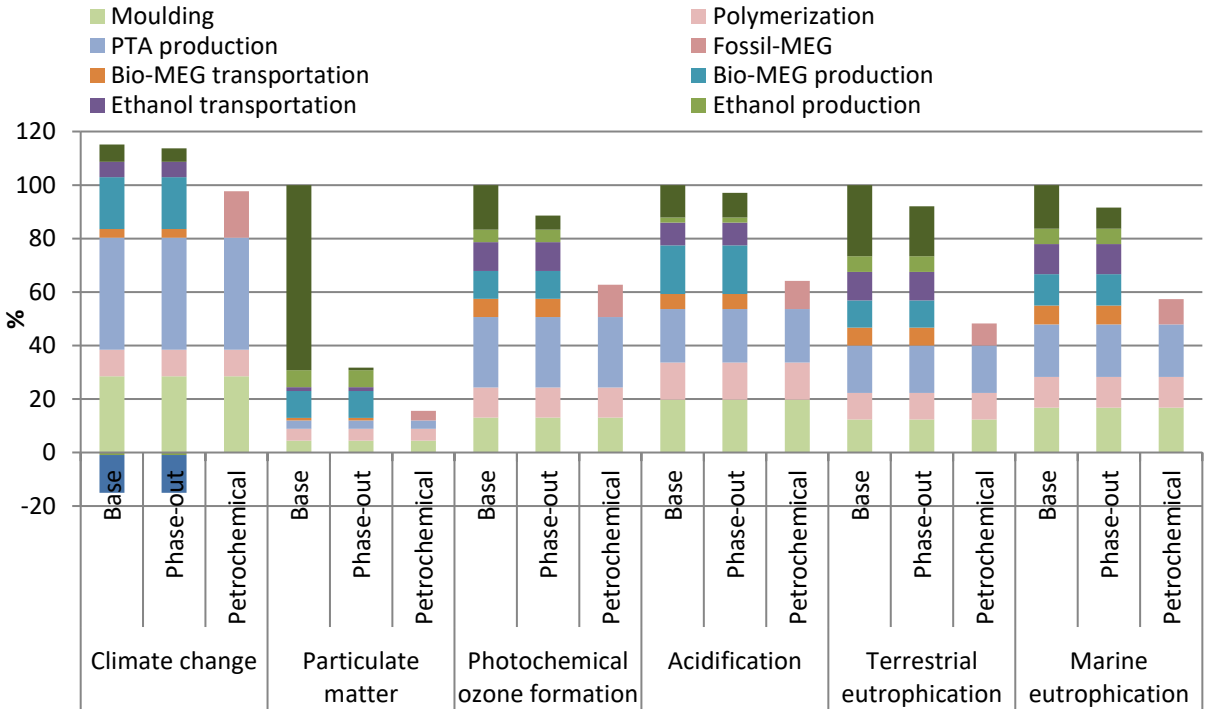


Figure 51. Effect of waste burning phase-out on the bio-based PET bottle results (Baseline).

4.1.2 Stretch blow moulding electricity consumption

Stretch blow moulding shows a considerable contribution in most of the impact categories. It should be noted that the stretch blow moulding process is the same for the bio-based and petrochemical PET. Therefore, the modification of this process has the same effect on both systems. The major impact of this process comes from the electricity used. As a base case value, a conservative 6.1 MJ_e/kg was taken for electricity consumption based on a literature survey (1.8-6.1 MJ_e) (CalRecycle, 2011). The lower end of the survey of 1.8 MJ_e/kg was used for the sensitivity analysis. In Table 73, the effect of a change in the electricity consumption of stretch blow moulding in the impact categories is shown. Significant reductions were observed for ionizing radiation (50 %), freshwater eutrophication (38 %), freshwater ecotoxicity (30 %), human toxicity (cancer effects) (29 %), climate change (19 %) and NREU (19 %).

Table 73. Effect of change in electricity consumption of stretch blow moulding process on the bio-based PET bottle results (Baseline).

Cradle to factory gate				
		Base electricity consumption	Minimum electricity consumption	% reduction
Climate change	kg CO_2 eq.	3.23	2.63	19
Ozone depletion	kg CFC-11 eq.	1.04E-05	1.03E-05	1
Human toxicity, non-cancer effects	CTUh	1.03E-06	8.74E-07	15
Human toxicity, cancer effects	CTUh	1.38E-07	9.78E-08	29
Particulate matter	kg PM2.5 eq.	9.89E-03	9.66E-03	2
Ionizing radiation HH	kBq U235 eq.	5.83E-01	2.93E-01	50
Photochemical ozone formation	kg NMVOC eq.	1.51E-02	1.39E-02	8
Acidification	molc H+ eq.	2.61E-02	2.27E-02	13
Terrestrial eutrophication	molc N eq.	5.59E-02	5.16E-02	8
Freshwater eutrophication	kg P eq.	1.37E-03	8.46E-04	38
Marine eutrophication	kg N eq.	4.81E-03	4.31E-03	10
Freshwater ecotoxicity	CTUe	1.75E+01	1.22E+01	30
Land use	kg C deficit	10.83	10.30	4
Water use	m^3	5.30	5.15	3
Abiotic depletion	kg Sb eq.	1.81E-06	1.53E-06	15
Abiotic depletion (fossil fuels)	MJ	74.09	67.10	9
NREU	MJ	82.26	66.45	19

4.1.3 End of life sensitivity analysis

The EoL sensitivity analysis was performed by increasing/decreasing the parameter sorting and technology efficiency of the recycling plant within limits found in the literature (see Table 74) for the bio-based product system. These values have a large range due to the fact that beverage bottles in some European countries are part of a refund system with very high purity for the waste stream, e.g. in Denmark. This differs from the ordinary heterogeneous plastic waste stream collected in households, where the quality is much lower. With a collection rate of 60 % the total recycling rate investigated is hence between 23 % and 56 % compared to former 45 %.

Table 74. Variation in input parameters in the sensitivity analysis.

Product system	Technology	Parameter changed	Original EoL (%)	Lower (%)	Higher (%)	References
Bio-based PET	Recycling	Sorting and technology efficiency	75.5	39	96	Dansk Retursystem, 2016, Pivnenko et al., 2015), and JRC, 2009

Figure 52 shows the sensitivity analysis, were the y-axis represents the percentage change in total impact characterization results compared with the original. When the percentage is negative, the results are lower than for the original and vice versa.

In general, the results show that an increase in recycling efficiency will give a better environmental performance. This appears logical as increasing the recycling efficiency will

cause an increase in the substituted PET which was previously shown to contribute significantly to the environmental savings for recycling compared to other EoL options.

There are two impact categories where the difference in percentage is above 400 %. The human toxicity (non-cancer) impact category is sensitive to the amount of reject incinerated, the heat generated and the bottom ash and fly ash treatment.

The freshwater ecotoxicity impact category is sensitive to substitution of PET. This difference would change the overall conclusion in the impact categories.

For human toxicity the lower sensitivity would give the highest impact of all EoL technologies (changing the best technology to incineration) and for the higher sensitivity scenario the lowest impacts (keeping the best technology as recycling).

For freshwater ecotoxicity the higher sensitivity analysis scenario would change the overall conclusion on which EoL technology performs best, making recycling perform best (formerly incineration performed best).

Abiotic depletion shows that lowering the recycling efficiency will positively affect abiotic depletion. This is because less heat is generated from incinerating rejects which contributes negatively to the abiotic depletion.

The impact category of climate change shows that the recycling efficiency parameter greatly influences the total climate change impact. Increasing the recycling efficiency will increase environmental savings from substituting virgin PET and decrease direct climate change emissions from incinerating the plastic rejects. Lowering the sorting and technology efficiency would change the conclusion on the best EoL technology with respect to climate change from recycling to landfilling.

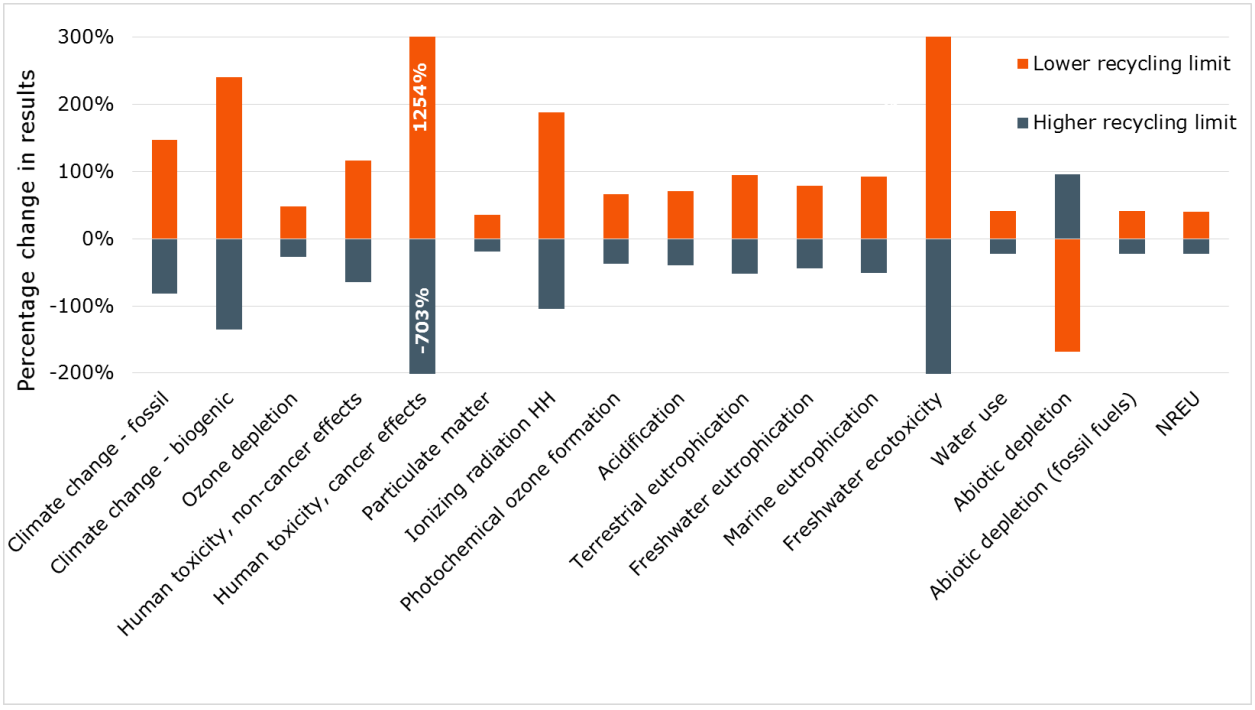


Figure 52. Sensitivity analysis for EoL recycling 30 % Bio-PET beverage bottles. The upper limit for sorting and technology efficiency was set to 96 % and the lower to 39 %.

The sensitivity of the recycling sorting and technology efficiency on the total cradle-to-grave results was also calculated and is presented in Table 75 for the seven selected comparable

impact categories. The cradle-to-grave climate change results increase by 25 % when the sorting and technology efficiency is decreased from 75.5 % to 39 %. The impact on abiotic depletion (fossil fuels) and NREU increases by 18 % and 15 % respectively when the recycling efficiency is lowered. The largest contribution in the recycling EoL technology is PET substitution for all seven impact categories. Even though the EoL share of the cradle-to-grave results is not high, recycling is 60 % of the EoL mix and the PET substitution clearly has the highest impact and therefore the cradle-to-grave results are to an extent affected by the change in sorting and technology efficiency.

Table 75. The change in percentage of higher and lower sorting and technology efficiency at the PLA recycling facility to the cradle-to-grave results of seven selected impact categories.

Impact category	Lower recycling (%)	Higher recycling (%)
Climate change	25	-14
Particulate matter	1	0
Photochemical ozone formation	10	-6
Acidification	6	-4
Terrestrial eutrophication	7	-4
Abiotic depletion (fossil fuels)	18	-10
NREU	15	-8

4.2 Discussion on End of Life uncertainties

The parameters analysed in the sensitivity analysis above and other parameters, which may be uncertain, e.g., estimations, assumptions, huge ranges in industry or EU member states, missing data, etc. are discussed in this section. Methodological shortcomings and additional research requirements are also presented.

The collection efficiency for recycling in the estimated EoL mix – This parameter differs between EU member states; some countries have a closed loop recycling system, e.g., a refund system with a high collection rate, while in other countries beverage bottles are collected in a co-mingled stream with lower collection rates.

The sensitivity analysis above showed that the technology and sorting efficiency for recycling of PET is sensitive to most impact categories, which changes the cradle-to-grave results by up to 25 % (see Table 75). This parameter is based on a number of reports but will vary due to different collection schemes. There is a substantial difference between collection in a co-mingled waste stream compared to for example a refund system with solely PET bottles.

The calculated quantity of biogenic carbon in the bio-based PET is 30 % (when MEG is biogenic). This parameter has a large influence on the composition of direct emissions from incineration in GWP, due to the different emission factors between fossil and biogenic carbon emissions. There is great development in this area, so this parameter is expected to increase in the coming years. Furthermore, it is technically possible to produce PTA from biomass, which makes it possible to produce 100 % bio-based PET using both bio-MEG and bio-PTA. In this scenario, the only difference with respect to EoL is that emissions originate from biogenic carbon instead of fossil carbon.

Apart from the above case study's specific uncertainties, several other general uncertainties that apply to most of the presented case studies are discussed in the Approach and Methodology section 3.5. These include uncertainties related to:

- the marginal energy technologies utilised

- the development of the EoL technologies (efficiency increase)
- the consumption of materials and energy at the EoL treatment plants
- littering, which is not modelled in this LCA

The collection rate as well as the sorting and technology efficiency for mechanical recycling are the most significant uncertainties.

5 Assessing non-PEF impact categories: taking into account ILUC and endpoint of GTP

5.1 Baseline – Ethanol from Brazilian sugarcane

5.1.1 Global Temperature Change Potential

The GTP impacts of baseline bio-based PET bottle from Brazilian sugarcane breakdown into the main unit processes described in section 2.1. The inventory analysis is shown in Figure 53. When biogenic carbon removal is considered, the cradle-to-gate GTP 100a is 3.25 kg CO_2 eq/FU. Observations for GTP are very similar to those of climate change described in section 3.1.1.

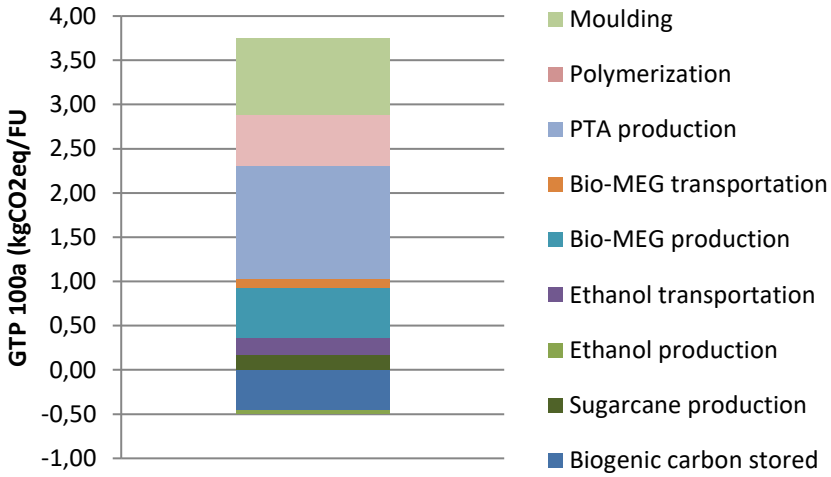


Figure 53. Break down of the cradle-to-gate GTP 100a results for bio-based PET bottle (baseline), 1 functional unit, excluding DLUC and ILUC effects.

5.1.2 Land Use Change emissions

Table 76 presents the cradle-to-grave characterised results broken down into feedstock (with biogenic carbon stored separately), manufacturing, EoL, DLUC and ILUC. As highlighted in Chapter 4, DLUC and ILUC cannot be summed up. DLUC is also presented as it is a "process" that should be considered in the LCA, according to the latest PEFCR Guidance v6.3 (European Commission, 2018). DLUC was calculated as specified in the methodology of the PEFCR Guidance v6.3, as detailed in Chapter 4. Figure 54 presents the characterised results for case study 1 with the inclusion of ILUC. Results with the inclusion of DLUC are not shown, as they are rather negligible (Table 76) and barely visible in an illustration such as Figure 54 (contributing only to climate change and making 1.3 % of the impact, when included). Again, it should be highlighted that the reason why DLUC only contributes to the climate change impact is due to the methodological choices of the PEFCR Guidance v6.3.

Table 76. Characterised cradle-to-grave results broken down per process, including DLUC and ILUC. Case study 1, baseline with Brazilian sugarcane ethanol.

		Biogenic carbon removal	dLUC	iLUC	Sugarcane production	Manufacturing	EoL
Climate change	kg CO_2 e	-4.58E-01	4.40E-02	2.60E-01	2.10E-01	3.48E+00	4.65E-02
Ozone depletion	kg CFC-11 e			5.43E-11	1.56E-08	1.04E-05	-4.11E-06
Human toxicity, non-cancer effects	CTUh			1.19E-10	2.95E-07	7.34E-07	-3.82E-08
Human toxicity, cancer effects	CTUh			1.12E-11	8.00E-09	1.30E-07	2.89E-10
Particulate matter	kg PM2.5 e			6.86E-06	6.85E-03	3.04E-03	-1.58E-04
Ionizing radiation HH	kBq U235 e			-4.74E-06	2.70E-03	5.80E-01	-3.05E-04
Photochemical ozone formation	kg NMVOC e			8.09E-04	2.52E-03	1.26E-02	-1.80E-03
Acidification	molc H+ e			1.62E-04	3.16E-03	2.29E-02	-1.82E-03
Terrestrial eutrophication	molc N e			9.09E-04	1.49E-02	4.10E-02	-2.91E-03
Freshwater eutrophication	kg P e			5.99E-07	2.07E-05	1.34E-03	-4.05E-05
Marine eutrophication	kg N e			9.66E-05	7.83E-04	4.03E-03	-2.91E-04
Freshwater ecotoxicity	CTUe			2.00E-03	1.37E+00	1.61E+01	4.29E-01
Land use	kg C deficit			7.47E-04	7.43E+00	3.40E+00	0.00E+00
Water use	m^3			8.01E-03	2.90E-02	5.27E+00	-2.08E-01
Abiotic depletion	kg Sb e			2.37E-09	5.07E-07	1.30E-06	-8.47E-08
Abiotic depletion (fossil fuels)	MJ			6.76E-03	1.27E+00	7.28E+01	-2.38E+01
NREU	MJ			6.46E-03	1.30E+00	8.10E+01	-2.36E+01

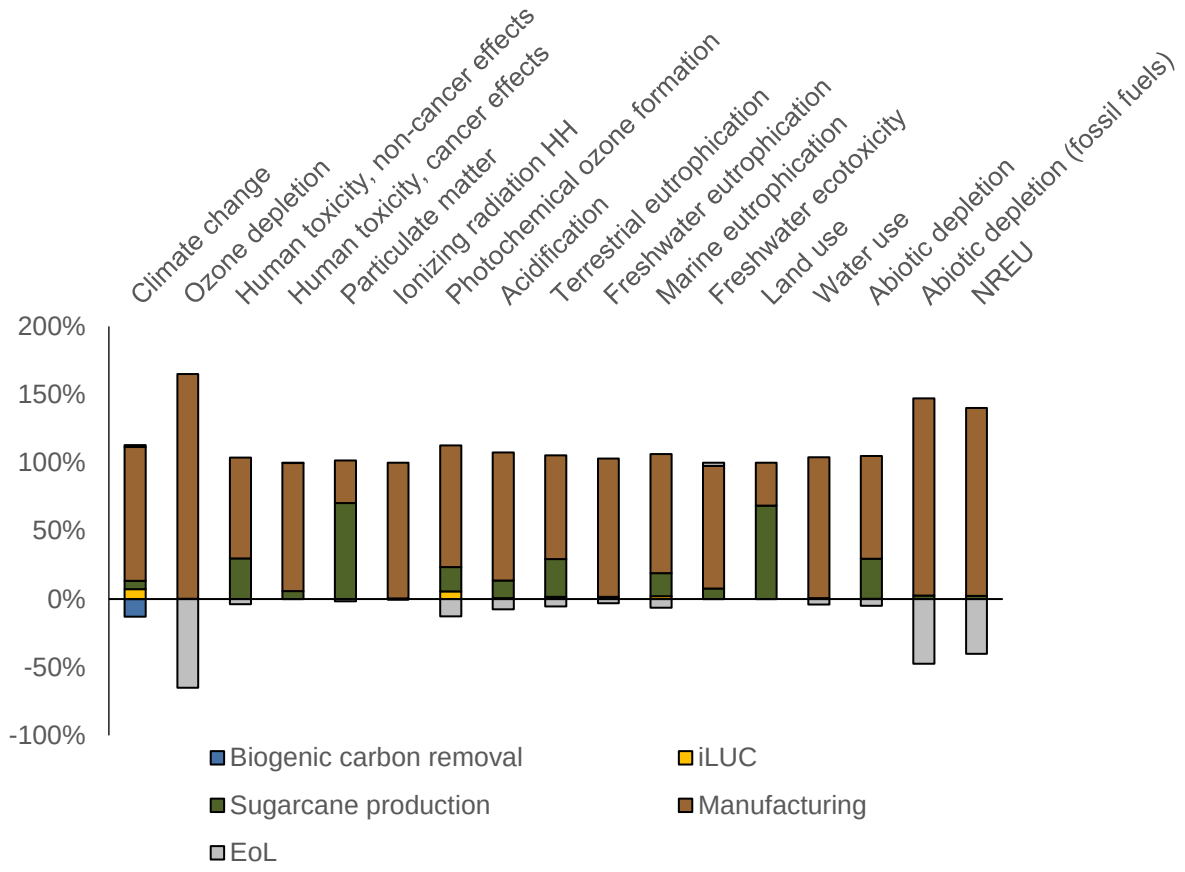


Figure 54. Relative characterised results broken down for all impact categories and including ILUC – case study 1, baseline with Brazilian sugarcane ethanol

As can be seen from Table 76 and to some extent from Figure 54, ILUC does contribute to all impact categories considered herein. However, the contribution of ILUC is rather negligible for most impact categories except climate change and photochemical ozone formation, two impacts dominated by land expansion (Figure 15). To some extent, ILUC also contributes to terrestrial and marine eutrophication (for these respective impacts, the ILUC contribution represents ca. 2 % of the overall impact). The ILUC contribution is below 1 % for all other impacts. In the case of climate change, the ILUC impact is essentially due to the CO_2 releases resulting from land clearing, and in the case of photochemical ozone formation, to carbon monoxide also released during land clearing (burning the cleared biomass; Figure 17).

In Table 76, the contribution of ILUC to the overall climate change impact is modest representing ca. 7 % of the total impact (including ILUC). On the other hand, it has about the same magnitude as the impact from producing the sugarcane itself (Figure 54; sugarcane cultivation represents 5.9 % of the climate change impact when ILUC is included, otherwise 6.4 %). LUC processes are dependent upon the amount of land needed. Hence, the greater the yield of a given crop is, the lower is the LUC. Similarly, the less (land-dependant) biomass is required for producing a given bio-based product, the lower the LUC. For this case study, an average yield of 22.7 Mg DM ha^{-1}, was used. Considering higher yields such as those reported by e.g. Seabra et al. (2011), namely 24.8 Mg DM ha^{-1}, could have lowered the ILUC (and DLUC) impact; this applies for all impact categories. Additional insights on the ILUC process (breakdown intensification/expansion, substances contributing to the climate change impact) are as presented for ILUC at the end of section 4.

5.2 Ethanol from EU crops mix

5.2.1 Global Temperature Change Potential

The GTP impacts of alternative bio-based PET bottle from EU crops breakdown into the main unit processes described in section 2.1. The inventory analysis is shown in Figure 55. When biogenic carbon removal is considered, the cradle-to-gate GTP 100a is 3.35 kg CO_2 eq/FU. Observations for GTP are very similar to those for climate change described in section 3.2.1.

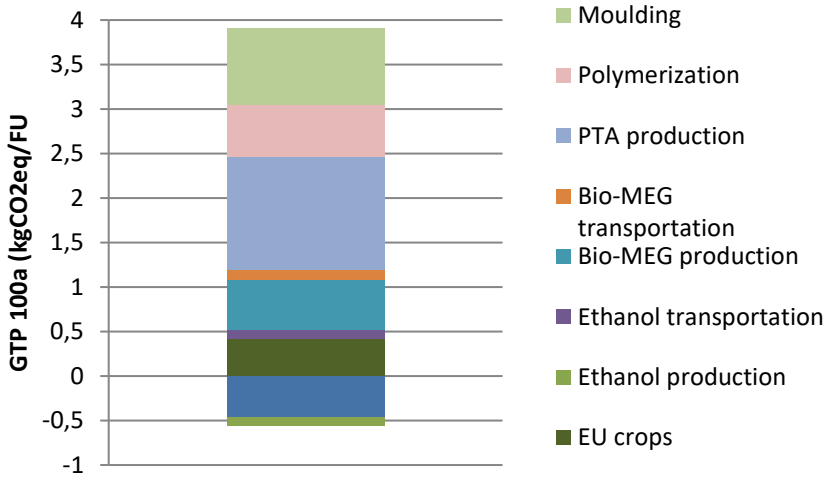


Figure 55. Break down of the cradle-to-gate GTP 100a results for bio-based PET bottle (alternative: EU crops), 1 functional unit, excluding DLUC and ILUC effects.

5.2.2 Land Use Change emissions

Table 77 presents the cradle-to-grave characterised results broken down into feedstock (with biogenic carbon stored separately), manufacturing, EoL, DLUC and ILUC. As highlighted in Chapter 4, DLUC and ILUC cannot be summed up. DLUC is also presented as it is a "process" that should be considered in the LCA, according to the latest PEFCR Guidance v6.3 (European Commission, 2018). DLUC was calculated as specified in the methodology of the PEFCR Guidance v6.3, as detailed in Chapter 4. Figure 56 presents the characterised results for case study 1 with the inclusion of ILUC.

Results with the inclusion of DLUC are not shown, as they are rather negligible (Table 77) and barely visible in an illustration such as Figure 56 (contributing to climate change only and accounting for 0.02 % of the impact, when included; this was 1.3 % in the baseline case). As explained in Chapter 4, DLUC is based on the 20-year history of crops in a given country. For the feedstock considered in this study, DLUC is only observed for a few crops, namely Brazilian sugarcane (used in the baseline of this case), German maize, French wheat and German wheat (the latter three all part of the EU average ethanol mix). Hence, this case study involves DLUC for both the baseline and the variant. The reason why DLUC is much more insignificant in the variant than in the baseline case is reflected in Table 25 of Chapter 4 (much less nature conversion area is considered, that area is from grassland rather than forestland and it has much lower GHG emission factors).

Table 77. Characterised cradle-to-grave results broken down per process, including DLUC and ILUC. Case study 1, alternative with ethanol stemming from EU crops.

	Biogenic carbon removal	dLUC	iLUC	EU crops	Manufacturing	EoL
Climate change (kg CO_2 e)	-4.60E-01	5.84E-04	5.64E-01	4.23E-01	3.35E+00	4.65E-02
Ozone depletion (kg CFC-11 e)			1.18E-10	1.12E-08	1.04E-05	-4.11E-06
Human toxicity, non-cancer effects (CTUh)			2.57E-10	9.75E-07	6.04E-07	-3.82E-08
Human toxicity, cancer effects (CTUh)			2.44E-11	1.32E-08	1.25E-07	2.89E-10
Particulate matter (kg PM2.5 e)			1.49E-05	2.70E-04	2.28E-03	-1.58E-04
Ionizing radiation HH (kBq U235 e)			-1.03E-05	1.36E-02	6.18E-01	-3.05E-04
Photochemical ozone formation (kg NMVOC e)			1.75E-03	1.13E-03	1.11E-02	-1.80E-03
Acidification (molc H+ e)			3.51E-04	8.98E-03	2.13E-02	-1.82E-03
Terrestrial eutrophication (molc N e)			1.97E-03	3.75E-02	3.19E-02	-2.91E-03
Freshwater eutrophication (kg P e)			1.30E-06	1.27E-04	1.28E-03	-4.05E-05
Marine eutrophication (kg N e)			2.09E-04	8.38E-03	2.47E-03	-2.91E-04
Freshwater ecotoxicity (CTUe)			4.35E-03	1.05E+01	1.39E+01	4.29E-01
Land use (kg C deficit)			1.62E-03	1.21E+01	-2.92E+00	0.00E+00
Water use (m^3)			1.74E-02	1.68E+00	4.02E+00	-2.08E-01
Abiotic depletion (kg Sb e)			5.13E-09	1.39E-07	5.81E-07	-8.47E-08
Abiotic depletion (fossil fuels) (MJ)			1.46E-02	2.93E+00	7.51E+01	-2.38E+01
NREU (MJ)			1.40E-02	3.32E+00	8.30E+01	-2.36E+01

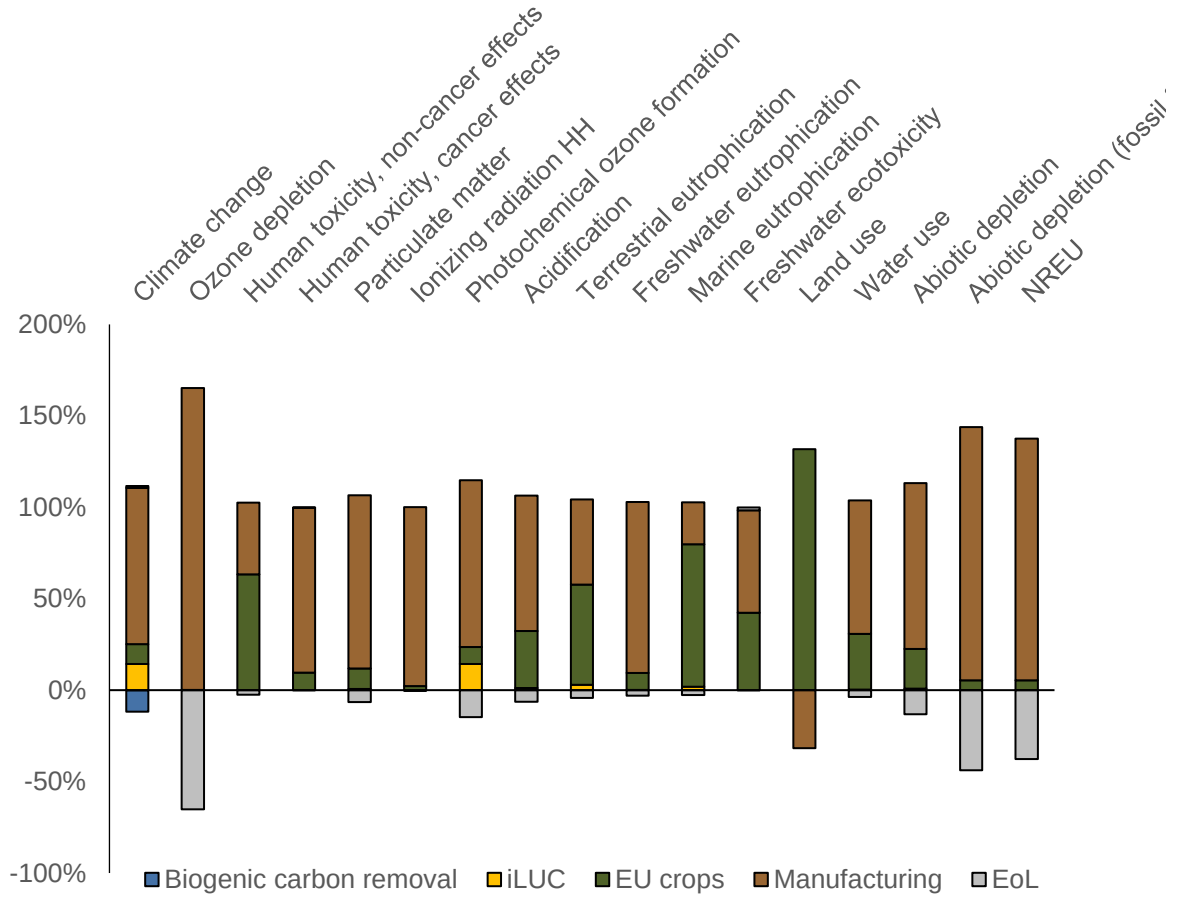


Figure 56. Relative characterised results broken down for all impact categories and including ILUC – case study 1, baseline with ethanol stemming from EU crops.

As can be seen from Table 77 and to some extent from Figure 56, ILUC does contribute to all impact categories considered herein, as in the baseline case. However, as for the baseline case, the contribution of ILUC is rather negligible for most impact categories except climate change and photochemical ozone formation, two impacts dominated by land expansion (Figure 15). In the case of climate change, as explained earlier, the ILUC impact is essentially due to the CO_2 release resulting from land clearing, and in the case of photochemical ozone formation, to carbon monoxide also released during land clearing (burning the cleared biomass; Figure 17).

As opposed to the baseline, however, there are more than just four impacts where the ILUC contribution represents more than 1 % of the impact. These are: climate change (14 %); and photochemical ozone formation (14 %), as mentioned, but also terrestrial eutrophication (3 %), marine eutrophication (2 %) and acidification (1.2 %). Both acidification, terrestrial- and marine eutrophication are essentially due to nitrogen oxides released during land clearing (more than 80 % of the impact; Figure 17).

In Table 75, the contribution of ILUC to the overall climate change impact is not as modest as in the baseline case, representing ca. 14 % of the total impact (versus 7 % in the baseline case, i.e. twice as much). On the other hand, it has about the same magnitude as the impact from producing the EU crops themselves (Figure 56; EU crop cultivation represents 11 % of the climate change impact when ILUC is included, otherwise 13 %). LUC processes are dependent upon the amount of land needed. Hence, the greater the yield of a given crop is, the lower is the LUC. This is essentially why this alternative PET scenario leads to a higher

ILUC than in the baseline case. In fact, the average yield considered for Brazilian sugarcane is 22.7 Mg DM ha^{-1}, while it is 11 Mg DM ha^{-1} for average EU ethanol crops (considering the share of wheat, maize and sugar beet in the ethanol mix, and the EU country mix where these crops stem from; Table 54).

6 Conclusions, limitations and recommendations

The environmental impacts of PET bottles are largely determined by the production of PTA and stretch blow moulding processes, regardless of the bio-based or the petrochemical origin of the polymer. The manufacturing phase accounts for nearly 75 % of the total impacts, with the highest contributor of PET polymer production, including bio-MEG, PTA and polymerisation. These three processes contribute to over 50 % of the total cradle-to-grave impacts, followed by the plastics conversion step (bottle blown moulding), contributing to 23 % of the total impacts. However, the production of biomass stage is also very influential in six impact categories namely: land use, particulate matter, human toxicity (non-cancer), terrestrial eutrophication, photochemical ozone formation, and marine eutrophication. Particulate matter emissions alone caused ca. 28 % of total cradle-to-grave impacts.

Based on the comparisons of weighted results of different EoL options for all 16 impact categories, recycling is identified as the preferred EoL option for PET bottles, followed by landfilling and incineration. The EoL stage has a relatively low relevance with 10 % contribution to the overall score using the EoL mix (60 % recycling, 20 % incineration and 20 % landfill). This is mainly due to the assumed 60 % recycling. If this rate could be increased to 100 % recycling, an additional 7-9 % reduction in the overall impacts could be achieved.

Bio-based PET (with 30 % bio-based content) offers limited environmental benefits:

- It is comparable with petrochemical PET in global warming potential (within 5 % difference) and water use (within 15 % difference).
- It offers a slight impact reduction (about 10 %) in abiotic depletion (fossil fuels) and non-renewable energy.
- It performs worse in the other five impact categories selected a being suitable for comparison namely; particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication, and marine eutrophication.
- Since the PTA production, polymerisation and stretch blow moulding processes are identical for both bio-based and petrochemical product systems, the differences come from biomass, ethanol and bio-MEG production and transport processes. Most of the impacts are caused by the agricultural phase, e.g. cane waste burning, direct emissions from fertiliser and pesticides application. The combustion of bagasse in ethanol production also has considerable effects. Transoceanic container transportation is an important contributor to marine and terrestrial eutrophication due to nitrogen oxides emissions. Sensitivity analysis shows that phasing out cane waste burning in the coming years in Brazil will substantially reduce bio-based PET's particulate matter impacts, although it still does not change the results of the comparison.

Two alternative hypothetical bio-based product systems have been considered. The only difference lies in the feedstock used to produce ethanol. The first feedstock alternative considered is the European crops market mix composed of maize, wheat and sugar beet. The second alternative feedstock considered is European wheat straw. From ethanol production onward (bio-MEG production, polymerization and stretch blow moulding) the processes are the same as the baseline product system (from Brazilian sugarcane, status-quo). The alternative system 1 performs worse than the baseline especially for acidification, human toxicity (non-cancer), freshwater ecotoxicity, terrestrial eutrophication and marine eutrophication categories. The alternative 2 performs the best and provides reduction of impacts compared to the baseline. Accordingly, it is seen that by switching to wheat straw as feedstock, the differences in results between bio-based and petrochemical PET bottle are considerably

decreased for the categories where bio-based was performing significantly worse than petrochemical PET bottle (from about 40 % worse to 10-20 % worse).

The impacts of DLUC are negligible. ILUC does contribute to all impact categories considered herein. However, the contribution of ILUC is rather negligible for most impact categories except climate change (7 % of the impact when the bio-based PET is based on Brazilian sugarcane and 14% of the impact when it is based on EU crops) and photochemical ozone formation (5.7 % of the impact when the bio-based PET is based on Brazilian sugarcane and 14% of the impact when it is based on EU crops), two impacts dominated by land expansion. In the case of climate change, the ILUC impact is essentially due to the CO_2 releases resulting from land clearing and in the case of photochemical ozone formation, to carbon monoxide also released during land clearing.

The weighted EoL results of both product systems, bio-based PET bottles and petrochemical PET bottles, show that recycling performs best environmentally followed by landfilling and incineration.

The EoL sensitivity analysis looking at the sorting and technology efficiency of recycling (the amount of the incoming plastic to the plastic facility, which is in fact recycled) varies around Europe (from 39 % to 96 %), which would affect the cradle-to-grave results by up to 25 % (looking at the seven selected impact categories), but not the overall conclusion that recycling performs best.

Littering is excluded from this case study due to lack of data and knowledge of the impact assessment models. The exact amount (and relative share) of plastics that are littered is uncertain for a specific application and will largely differ between countries. The impacts of littering and microplastics are currently not properly captured in the toxicity-related impact categories (ecotoxicity and human toxicity). As pointed out by Pant et al. (2018) fate, exposure, and effect modelling for macro- and micro-plastics (including additives) is still in its infancy. Physical impacts (e.g. entanglement; ingestion of larger plastic particles and their effects) and chemical impacts (e.g. due to microplastic formation, microplastics as carriers of other chemicals, unknown impact of additives) and biological impacts (e.g. microplastics as carriers of germs/alien species) are largely unknown and currently not included in LCA. Thus, the impacts of littering (e.g. leaving plastics in the soil or in a marine environment) are most likely underestimated in the current LCA for the non-degradable applications.

CASE STUDY 2: SINGLE-USE CLIPS

1 Goal and scope definition

1.1 Goal and background

The goal of this case study is to assess the environmental profile of bio-based clips production and compare it with the fossil-based counterpart. Traditionally, clips are made with polypropylene which is not biodegradable or compostable. Bio-based clips can offer the same functionality while being biodegradable and compostable. Clips are used to support the growth of crops like tomatoes, cucumbers and peppers. After the harvest, because clips are attached to the plants, they are difficult to remove. They are highly contaminated with plant residues, and therefore gathering from the field is very labour intensive and costly. Traditional clips that remain in field cause contamination. Bio-based biodegradable clips provide a valuable solution as they do not need to be removed. They can fully biodegrade and compost with the plant waste (Guerrini et al., 2017). They should be certified biodegradable and compostable according to the standard EN 13432.

Bio-based clips are produced from starch plastics. Starch plastics represent 10.3 % of the global production capacity of bio-based plastics (Aeschelmann et al., 2017). Europe has a solid position in the production and commercialization of starch blends with several companies being active in this sector, such as Novamont, Rodenburg, Biotec, FKuR and Kingfa (Aeschelmann et al., 2017). Application of starch plastics include flexible packaging, carrier bags, disposable tableware, agriculture and horticulture (mulch films, clips, pots) and loose-fill packaging foams. Carrier bags make up most of the consumption, whereas agricultural plastics have a very small market share (Kaeb et al., 2016). The demand for agricultural plastics at global scale is expected to increase 10 % per year until 2020 (Santagata et al., 2017).

Starch plastics are blends of starch with one or more polymers (fossil or bio-based polyesters). Plasticisers such as glycerine can be used to destructure native starch into thermoplastic starch for improved mechanical properties (Broeren et al., 2017a). Depending on the industrial application and required technical properties, thermoplastic starch is blended (reactively extruded) with complexing agents of fossil or bio-based polyesters such as polylactic acid (PLA), polybutyrate adipate-co-terephthalate (PBAT) and polyhydroxyalkanoates (PHAs), to produce polyester-complexed starch biopolymers (Broeren et al., 2017a). To achieve good adhesion between the polymers, compatibiliser additives are used. By blending starch with biodegradable polyesters, 100 % biodegradability can be attained.

1.2 Scope

The geographical scope is Europe for the purchase, use and disposal of the clips. However, the study considers all processes occurring outside Europe prior to purchase (e.g. feedstock cultivation and harvesting, conversion processes). For the baseline, the cultivation of maize and potato used for producing starch takes place in Europe. For the alternative feedstock, the potato waste is sourced from the European food industry. PLA used as copolyester is produced in the U.S. from maize that is cultivated in the U.S. Other copolyesters and additives are sourced from Europe. Production of the starch blend and moulding into clips take place in Europe.

The temporal scope is current production (2017-2018), with relevant developments foreseen for the near future (5-10 years). The technical scope is the range of commercially available technologies.

1.3 Function and functional unit

The function is to support the growth of crops like tomatoes, cucumbers and peppers.

The functional unit of this case study is defined as:

- 45 000 single-use clips used for horticultural purposes for one hectare of land

The petrochemical counterpart of bio-based clips is plastic clips made of polypropylene (PP). The average weight of a PP clip is 1.2 g and a bio-based clip is 1.6 g (Novamont, 2018). To fulfil the functional unit, 72 kg of bio-based clips and 54 kg of PP clips are needed.

1.4 Product systems

The baseline for the bio-based product system is polyester-complexed starch biopolymer clips produced from European crops (maize or potato). In the starch plastics industry, very often companies produce starch blends based on their own technologies. The environmental impacts are analysed based on each technology; here, company-specific data are used. Aggregated data are then constructed by making a weighted average based on the market share of companies. The market share of starch-based plastics from maize is two thirds, whereas the remaining one third of the market is composed of potato starch-based plastics.

An alternative bio-based product system is polyester-complexed starch biopolymer clips from potato waste from the European food industry. The reclaimed starch is currently used commercially in producing starch plastics. It is of interest to see the impacts if instead of producing starch from biomass, it is reclaimed from waste.

The reference petrochemical system is PP clips produced in Europe.

1.5 System boundaries

A cradle-to-grave approach is taken including the life cycle stages of feedstock production, manufacturing and EoL. The consumer use phase is excluded from the analysis which is same for both product systems and has negligible impact. Thus, the life cycle can be divided into cradle-to-gate and EoL.

2 Life cycle inventory analysis

2.1 Process description, data, assumptions and multifunctionality

2.1.1 Baseline: Bio-Based clips from maize/potato

For this process, data is obtained from the largest industrial starch-based bioplastic producers. The biodegradable and bio-based polymer grades used for clips application have the following properties. They are biodegradable thermoplastic materials made with maize/potato starch and biodegradable copolyesters based on proprietary technology. The copolyester can be bio-based, fossil based or a mix of both. For the potato starch-based plastic, one of the copolyesters is bio-based PLA. For the maize starch-based plastic, both bio-based (PLA) and fossil-based (proprietary) polyesters are used. Maize is sourced in Europe (from Italy). Potato is sourced in Europe. It is converted to starch and supplied to the company. Copolyester and other raw materials are produced in Europe from bio and fossil resources and supplied to the plastic producer. The starch-based bioplastic production consists of a combination (by means of extrusion) of the native maize/potato starch with the copolyesters and other raw materials (i.e. additives such as compatibilisers and plasticisers). Possible plastic scraps from the manufacturing process are directly recycled in the extrusion process. The thermoplastic granules (plastic pellets), ready to be processed into the final application, are then sent to other economic operators to be converted into clips through injection moulding.

The process flow diagram of the bio-based clips product system is shown in Figure 57 for maize-based clips and in Figure 58 for potato-based clips.

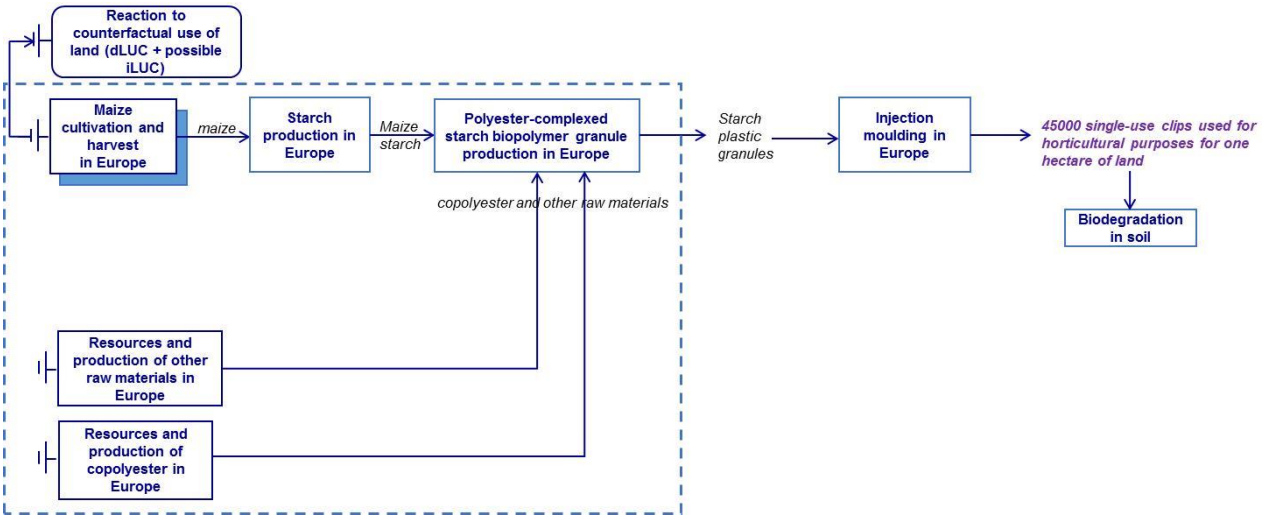


Figure 57. Process flow diagram of bio-based clips from maize product system

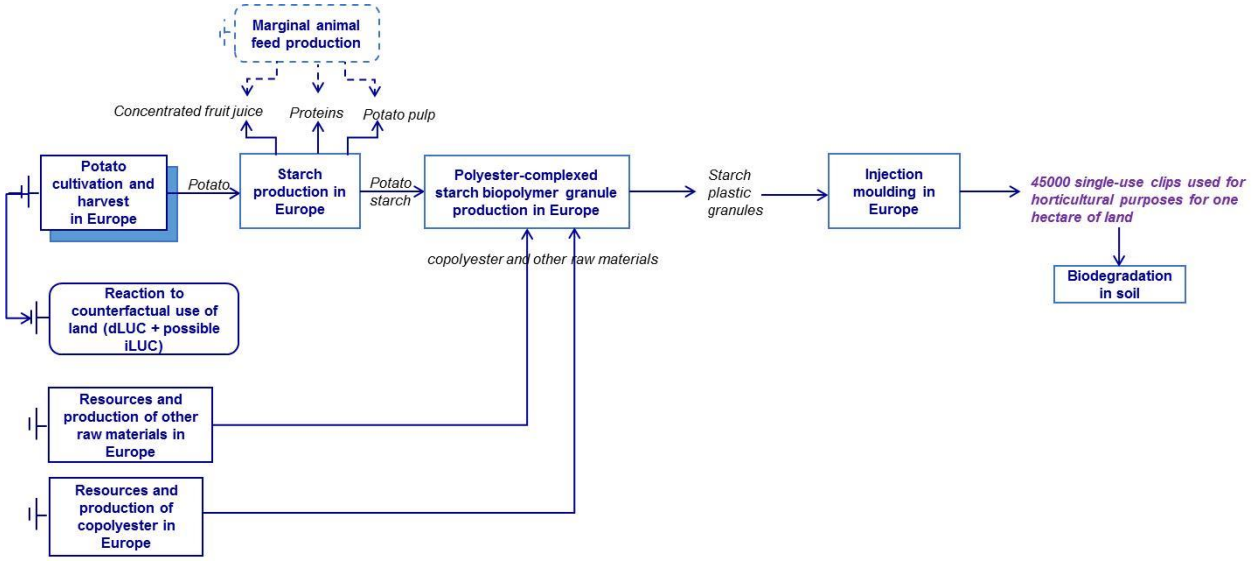


Figure 58. Process flow diagram of bio-based clips from potato product system

2.1.1.1 Bio-Based clips from maize

The maize-based starch plastic producer provided cradle-to-gate elementary flow data for their injection moulding grade plastic, which was compiled using confidential company-specific foreground data. The producing company informed us that for background data, the Ecoinvent 3.3 database was used. This data includes all the unit processes to produce the injection moulding grade granulates (shown with a dotted box in Figure 57). Therefore, it is not possible to show the breakdown of this product into unit processes (maize starch production, bio- and fossil-based copolyesters production, other raw materials (i.e. additives such as compatibilisers and plasticisers) production and granulate production). For the injection moulding of the starch plastic granulates to clips, the same data as for potato starch-based clips is used (refer to section 2.1.1.2).

To show the impact associated with maize cultivation and harvest separately, data for this section was requested and obtained from the company through personal communication (Novamont, 2018). This allows us to breakdown the impacts into biomass and manufacturing as shown aggregated with the potato-based starch plastic results in section 3.1.1. From the elemental composition of the polyester-complexed starch biopolymer provided by the company, biogenic carbon content is used to calculate the biogenic carbon embedded in the bio-based clips (1.16 kg CO_2 eq/kg maize starch-based polymer).

2.1.1.2 Bio-based clips from potato

For the potato-based starch plastic the processes involved can be divided into the following key unit processes:

2.1.1.2.1 Potato cultivation and harvest

Potatoes are sourced mainly from Germany. The Agrifootprint database is used for the inventory data of potato production in Germany. From the composition of the polyester-complexed starch biopolymer provided by the plastic producer, biogenic carbon content is used to calculate the biogenic carbon embedded in the bio-based clips (1.27 kg CO_2 eq./kg potato starch-based polymer).

2.1.1.2.2 Potato starch production

After harvesting, potatoes are cleaned and grinded into pulp. The starch slurry and juice are then separated. The juice is further processed to produce proteins and concentrated fruit juice. Starch slurry is refined and dried to produce native starch to be sent to granulate production. Concentrated fruit juice, proteins and pulp are obtained as co-products of the starch production process. The Agrifootprint database is used for the inventory data reflecting a typical potato wet milling process in Europe. Drying of the starch to the required moisture level is included in the inventory.

2.1.1.2.3 Potato starch transportation

The potato starch is transported to the granulate production site by lorry.

2.1.1.2.4 Copolyester (PLA) production

Bio-based copolyester PLA from maize is used as copolyester. Data is obtained from the PLA producer located in the US. Please refer to case study 3 for further information.

2.1.1.2.5 Copolyester (PLA) transportation

The copolyester is transported to the granulate production site by combination of lorry, bulk carrier and freight train.

2.1.1.2.6 Other raw materials production

Other renewable and non-renewable raw materials (compatibilisers, stabilisers) are produced in Europe. Data from the Ecoinvent 3.3 database is used for them.

2.1.1.2.7 Other raw materials transportation

Other raw materials used in granulate production are transported by lorry to the granulate production site.

2.1.1.2.8 Polyester-complexed starch biopolymer granulate production

Starch is combined with copolyester and other raw materials to get a thermoplastic biodegradable bio-based material (i.e. granulate) based on proprietary technology. The key activity-level data are the electricity consumption and waste treatments. Foreground data is based on Broeren et al. (2017a).

2.1.1.2.9 Granulate transportation

The starch-based biodegradable bioplastic granulate produced is transported to the converter site by lorry.

2.1.1.2.10 Injection moulding

Clips are produced through injection moulding of the starch-based granulates. With 1 kg of starch-based granulate, 0.994 kg of clips is attained. The key activity level data is electricity consumption which is 1.48 kWh/kg as taken from the Ecoinvent 3.3 database. The foreground data sources used for the baseline bio-based clips are summarised in Table 78. For background data, e.g. grid electricity and heat, other utilities and production of chemicals and materials, Ecoinvent 3.3 is used.

Table 78. Data used in the baseline bio-based clips.

Processes	Data sources, foreground	Comments
Maize cultivation and harvest	Novamont, 2018	Represents production of maize in Italy obtained from the supplier of maize-starch based bioplastic producer.
Maize-starch based granulate production	Confidential company specific foreground data	Includes all the unit processes for the production of the injection moulding grade granulates (maize starch production, copolyester and other raw material production, their transportation and granulate production).
Potato cultivation and harvest	Agri-footprint	Represents potato cultivation in Germany. Crop yields derived mainly from 2010-2014 FAO statistics (FAOSTAT, 2014)
Potato starch production	Agri-footprint	Represents a typical potato wet milling process in Europe, reference year of 2014
PLA (copolyester) production	Vink et al., 2015	with updated background data using Gabi database version 2018 (Personal communication with E. Vink). Based on data from industrial producer, see case study 3 for details
Other raw materials (i.e. compatibilsers and plasticisers) production	Broeren et al., 2017a & Ecoinvent 3.3	Represents average European processes, reference year 2016
Transportation (of potato starch, copolyester and other raw materials)	Broeren et al., 2017a	Represents average distances from the suppliers to the granulate production site. Within Europe transportation by lorry, from U.S. transportation by combination of lorry, bulk carrier and freight train
Potato-starch based granulate production	Broeren et al., 2017a	Represents average compounding extruder, key activity level data is electricity consumption: 1.9 MJ/kg granulate
Injection moulding	Ecoinvent 3.3	Reference year 2016 based on information from European companies, key activity level data is electricity consumption: 1.48 kWh/kg moulded plastic

2.1.1.2.11 Multifunctionality

The potato starch production process involves co-production of potato pulp, proteins and concentrated fruit juice. To avoid allocation, the system is expanded to include their animal feed replacement. The co-products are displaced with a mix of marginal feed ingredients with the same standardised feed unit. The marginal protein feed is taken as soybean meal, the marginal carbohydrate is taken as maize and the marginal oil is taken as palm oil, based on Tonini et al. (2016). These choices are based on detailed elaboration of recent demand trends and future projections (FAPRI, 2012; FAOSTAT, 2014).

2.1.2 Bio-Based alternative: Bio-Based clips from potato waste

The process flow diagram of the bio-based clips from the potato waste product system is shown in Figure 59. The only difference with the baseline potato-starch based product system is reclaimed potato starch is used instead of virgin starch. Therefore, the processes of potato cultivation and harvest and conversion to potato starch are not present in this product system

in comparison with the baseline product system. The subsequent processes to convert the reclaimed potato starch to the bio-based clips are identical with the baseline potato-based product system; details are provided in section 2.1.1.

Reclaimed starch is a by-product from the production of sliced potato products such as fries. Starch is reclaimed from the wastewater generated during potato processing. The waste stream is assumed to be free of any upstream environmental burdens (e.g. with the impacts associated with potato cultivation and processing). However, the subsequent evaporation of the wet starch cake to the required 18 % moisture content is taken into consideration and assigned to the reclaimed potato starch. The foreground data sources used for the reclaimed starch are given in Table 79.

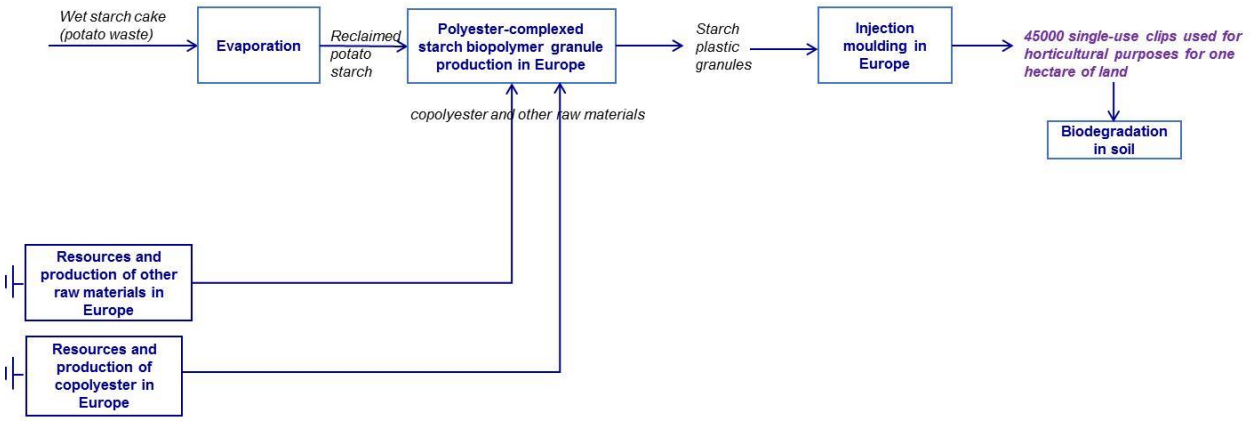


Figure 59. Process flow diagram of bio-based clips from potato waste product system

Table 79. Data used in the alternative bio-based clips.

Processes	Data sources, foreground	Comments
Reclaimed potato starch production	Broeren et al., 2017a	Energy required for evaporation of the wet starch cake to the required 18 % moisture content is 2.36 MJ/kg reclaimed starch

2.1.3 The reference systems: petrochemical PP

The reference petrochemical system is PP clips. PlasticsEurope data is taken as the reference since it provides the most up to date eco-profile based on European plastic producers. However, as explained in Chapter 6 (*Ranges for environmental impacts from production of fossil-based plastics*), PlasticsEurope data for petrochemical plastics are not transparent and do not comply with ILCD requirements. Differences between the available data from different data sources for PP are found to be large, and some impact categories were found not suitable (Section 6.1 in Approach and methodology). The data is provided in Table 35 for the suitable impact categories for 1 kg PP granulate production. For the transport of the granulates and injection moulding into clips, the same data is used as for the bio-based system.

2.2 End of Life description, data and assumptions

The single-use agricultural clips have different possible EoL options depending on the product system, as observed in Table 80.

The petrochemical PP single-use clips have two possible EoL options;

- **MSW Incineration with and without energy recovery**; a generic MSW incineration plant is assumed, which represents average EU waste incineration with and without energy recovery with an overall heat efficiency of 22 % and electrical efficiency of 9 % (CEWEP, 2013; Eurostat, 2017a). The process includes the direct emissions from waste incineration as well as indirect emissions from the production of the input materials, the combustion of fossil fuels as well as treatment of bottom ash and fly ash. The flowchart for MSW incineration is presented in Figure 9.

- **MSW landfilling**; the process includes the construction and operation of an average landfill site, direct emissions of landfill gas, landfill gas collection, flaring and recovery system, treatment of collected leachate and direct emissions from uncollected leachate. 29 % of the collected landfill gas is used to generate electricity (OpenLCA Nexus, 2015). The electricity conversion efficiency is 37 % (Christensen, 2011). The flowchart for MSW landfilling is presented in Figure 10.

The starch single-use clips have in addition the option of in-situ field-based biodegradation that includes avoided collection and direct emissions to air and soil. The clips are certified biodegradable and compostable according to the standard EN 13432. Furthermore, they are certified according to OK Biodegradable Soil by the Austrian certification institute TÜV. The flowchart for the in-situ field-based biodegradation is presented in Figure 11.

A major uncertainty in this case study is that a large share of the petrochemical clips tends to be left in the field and become a littering problem. It should also be noted that the likelihood of the biodegradable clips being incinerated or landfilled is questionable, due to difficulty of gathering the clips from the field after utilisation. Additionally, for the starch clips they are not easily gathered and could be biodegraded to a great extent.

After harvesting the crops, the single used clips must be gathered from the field, collected, transported and then disposed of in landfill or incinerated. This applies to both the petrochemical and bio-based clips. Gathering the clips requires energy consumption by sieving of soil with a sieving machine and a gathering machine. The gathering is estimated to consume 3 litres of diesel per EoL-reference flow of petrochemical plastic and 2 litres per EoL-reference flow of bio-based plastic (Lipasto 2014 & ELCD, n.d.). For the sieving it is estimated that the electricity consumption is 0.1468 kWh per EoL-reference flow for both product systems, based on Zaman et al., 2004. The gathering and sieving are added to the EoL technologies' impacts for incineration and landfilling, but not for in-situ as the clips will be left on field. The soil sieved from the clips is assumed to be brought back to the field, hence 1 kg of plastic and 100 g contamination is landfilled or incinerated. The details on the EoL mix are described at the end of this section.

Table 80. The EoL options, the intended EoL option and the estimated EoL mix.

Product system	EoL options modelled individually (100 % scenarios)	Intended EoL option	Estimated EoL collection mix of EU (based on statistics of waste treatments of waste plastics)
Bio-based starch plastics	Incineration Landfilling In-situ	In-situ	In-situ: 100 %
Petrochemical PP	Incineration Landfilling		Incineration: 56 % Landfilling: 44 %

The single-use clips have the following chemical composition (see Table 81), which is utilised as the input to the EoL LCA model. The full chemical composition is presented in Annex 1.

If the single use clips are to be gathered for treatment, a large portion of soil will be attached to the clips and thereby follow the plastic. The amount of contamination can be three to four times the weight of the plastic for mulch films (Plasticulture, 2018). In this study the contamination of soil was set to 3 kg per kg of plastic clips, as there is no specific data to find for contamination for clips. Hence, the EoL-reference flow for modelling the EoL for single-use clips is:

- 1 kg plastic + 100 g contamination + 3 kg soil

Table 81. Overview of the material composition of single-use clips (1 kg plastic + 100 g contamination).

Chemical component	Unit	Bio-based starch	Petrochemical PP
Water	% of total	0	9.5
TS (VS+ash)	% of total	100	90.5
VS	% of TS	99.99	97.6
C fossil	% of TS	20.63	77.1
C biogenic	% of TS	32.60	4.5
References		Novamont, 2018; Rodenburg, 2018	Götze et al., 2016

The average European EoL technologies are described in the approach and methodology chapter in the section regarding EoL. In that section, factors that do not depend on the specific product system are described, e.g. the energy efficiency of the incineration plant and the consumption of energy and materials in each technology. Furthermore, principles for substitution are described. This involves the energy produced in the EoL technologies is being substituted with marginal electricity and heat. In addition, to the description here, the contamination (soil) and additional soil being removed, when the agricultural products are gathered for collection and treatment, a depletion of the organic carbon from soil is modelled in the cases were the clips are incinerated or landfilled.

There are several product system dependent factors, which are presented in Table 82, for each possible EoL technology.

Table 82. Values for product system dependent factors for single-use clips.

EoL Technology	Product system dependent factor	Unit	Bio-based starch	Reference	Petrochemical PP	Reference
MSWI with and without energy recovery	Energy content	MJ/kg	21.8	Novamont, 2018	41.86	Götze et al., 2016
MSWI with and without energy recovery/ Landfilling	Diesel consumption for gathering	liter	2	Lipasto 2014; ELCD, n.d.	3	Lipasto 2014; ELCD, n.d.
MSWI with and without energy recovery/ Landfilling	Electricity use for sieving	Kwh	0.147	Zaman el al., 2004	0.147	Zaman el al., 2004
Landfilling	1st order decay rate for methane generation	1/s	0.032	Calculated based on data from Ecoinvent 3.4	0	Estimated
In-situ biodegradation in field	C Degradation (%)	% of C bio	100	Novamont, 2018		

In addition to modelling full (100 %) incineration and landfilling individually, a mix of these EoL scenarios was also modelled to represent the current situation in Europe. The distribution between incineration and landfilling of the petrochemical PP clips is based on European average distribution for plastics. The incineration and landfilling of the clips is based on general plastic waste in Europe (European Commission, 2018) the division between these EoL options is 44 % landfilling and 56 % incineration.

3 Life cycle impact assessment results and interpretation

3.1 Cradle-to-gate environmental impacts and interpretation, bio-based system(s)

3.1.1 Cradle-to-gate environmental impacts of Bio-Based baseline system: Bio-based clips from maize/potato

The aggregated cradle-to-factory gate LCA results for one functional unit of bio-based clips (72 kg) are presented in Table 83. This represents the aggregated results based on the market share of 2/3 for starch plastics from maize and 1/3 for starch plastics from potato as described in section 1.4.

Table 83. Cradle-to-gate LCA results of 1 functional unit bio-based clips (baseline system) (excluding DLUC and ILUC effects).

Impact category	Unit	Bio-based clips (market average)
Climate change	kg CO_2 eq.	1.09E+02
Ozone depletion	kg CFC-11 eq.	2.17E-05
Human toxicity, non-cancer effects	CTUh	7.68E-05
Human toxicity, cancer effects	CTUh	1.04E-05
Particulate matter	kg PM2.5 eq.	9.30E-02
Ionizing radiation, human health	kBq U235 eq.	3.60E+01
Photochemical ozone formation	kg NMVOC eq.	6.58E-01
Acidification	molc H+ eq.	1.37E+00
Terrestrial eutrophication	molc N eq.	2.69E+00
Freshwater eutrophication	kg P eq.	7.38E-02
Marine eutrophication	kg N eq.	4.02E-01
Freshwater ecotoxicity	CTUe	1.18E+03
Land use	kg C deficit	6.58E+02
Water use	m^3	1.05E+02
Abiotic depletion	kg Sb eq.	4.85E-04
Abiotic depletion (fossil fuels)	MJ	3.47E+03
NREU	MJ	4.10E+03

In Figure 60, the impacts of potato starch bio-based clips are broken down into the main unit processes described in the section 2.1 inventory analysis.

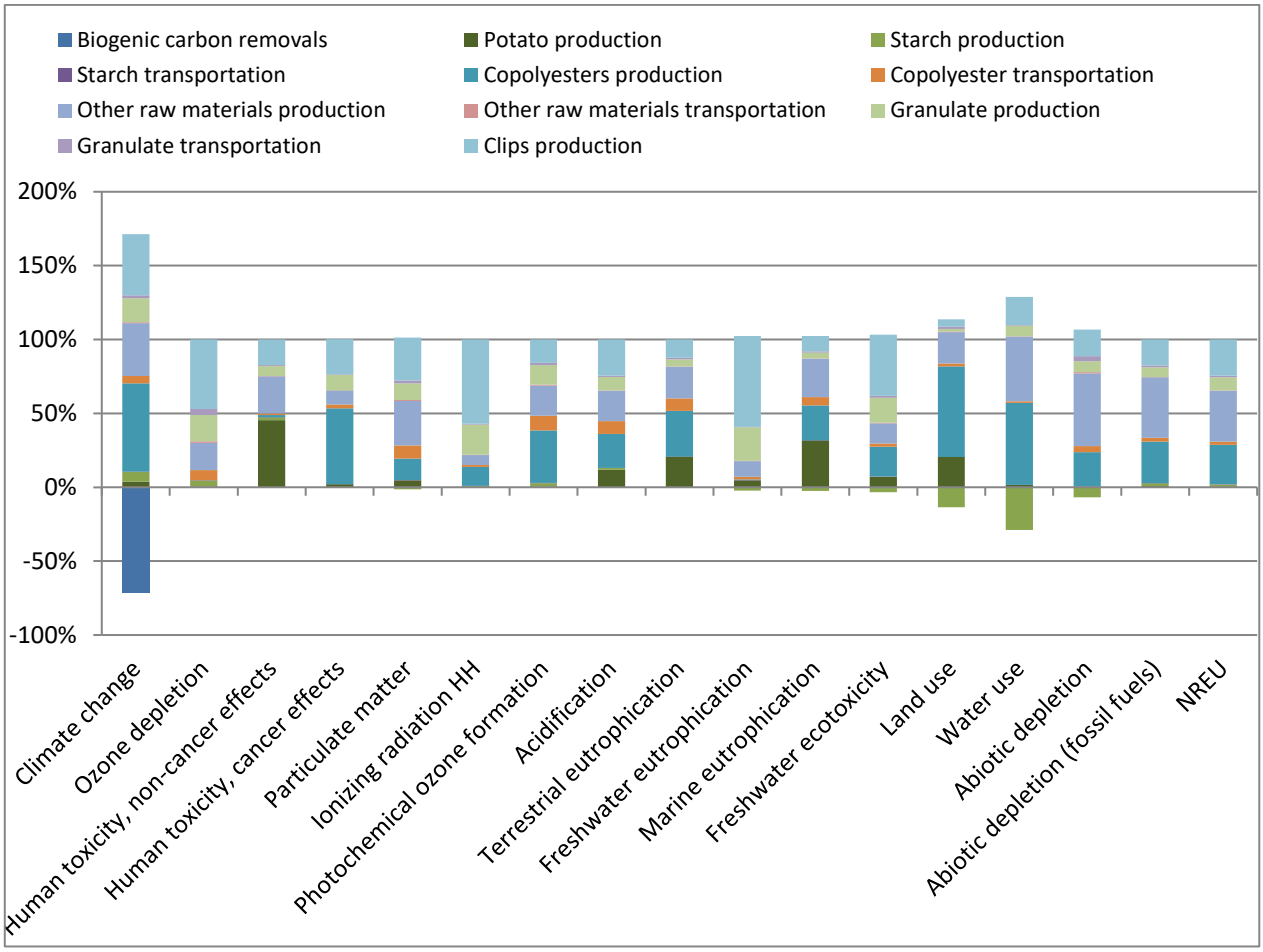


Figure 60. Break down of the cradle-to-gate LCA results for bio-based clips (baseline, from potato), excluding DLUC and ILUC effects.

Potato production contributes significantly to human toxicity (non-cancer effects) (45 %), marine eutrophication (32 %), terrestrial eutrophication (21 %) and land use (20 %). The contribution to marine eutrophication comes from direct nitrate emissions to water (90 %) from application of fertilisers and from crop residues. The contribution to terrestrial eutrophication is ammonia emissions to air (87 %) due to use of manure and fertiliser. For other impact categories, contribution from potato production is up to 11 %.

Starch production does not have a significant contribution to any category except for land use and water use for which it provides a 15-20 % saving due to the avoided animal feed production by the starch by-products.

The bio-based copolyester PLA from maize has a high contribution to climate change (60 %), human toxicity (cancer effects) (52 %), land use (60 %) and water use (55 %). It also has a significant contribution (about 27 %) to the abiotic depletion (fossil fuels) and NREU categories. Please refer to case study 3 for a detailed description and breakdown of the impacts for PLA from maize.

Other renewable and non-renewable raw materials used are seen to have a significant contribution to the abiotic depletion (49 %), abiotic depletion (fossil fuels) (41 %), NREU (35 %) and climate change (36 %) categories.

The contribution of transportation requirements (starch and copolyester transportation to granulate production, granulate transportation to clips production) to the environmental impact of bio-based clips are minor (total maximum 12 % contribution to impact categories).

Granulate production and clips production processes have parallel contribution to the impact categories. The key activity level data in both processes is electricity consumption, with a higher electricity requirement in clips production showing a higher contribution here. A significant contribution of these processes is seen for ozone depletion, ionizing radiation and freshwater eutrophication: combined they contribute to about 70 % of the environmental impact of clips in these categories.

The results for bio-based clips are further elaborated below. Following the PEFCR guidance the most relevant impact categories are identified as (in descending order starting with most relevant): abiotic depletion (fossil fuels), climate change, particulate matter, acidification, ionizing radiation, freshwater eutrophication and photochemical ozone formation. They cumulatively contribute to 83 % of the total environmental impact (excluding toxicity-related impact categories) based on the normalised and weighted results (see section 3.4.1). Detailed analysis is made to trace the origin of the impact in the major contributing processes for these impact categories and for impact categories where biomass production has significant impact.

The interpretation of the most relevant impact categories is here detailed:

- **Abiotic depletion (fossil fuels).** The production of other raw materials (i.e. compatibilers and plasticisers) account for 41 % of the impacts. This arises from the petrochemical additives used where 60 % of the impact is from crude oil and 35 % from natural gas). The other main contributions come from copolyester of PLA production (28 %). This is due to the energy requirement in production of lactic acid (please refer to case study 3 for a more detailed breakdown for PLA). Clips production (18 %) also has significant contribution due to the electricity used.

- **Climate change.** The copolyester PLA production accounts for 60 % of the climate change impact (Figure 60) mainly due to the heat and electricity requirements (refer to case study 3 for the breakdown of PLA from maize). Other raw materials production account for 36 % of the impact. The remaining contributions of unit processes to bio-based clips come from granulate and clips production processes (total 58 %), from the electricity consumption. The biogenic carbon stored gives a 72 % saving in climate change.

- **Particulate matter.** For particulate matter, cumulatively 40 % of the contribution comes from granulate production and clips production where key activity data is electricity used. 30 % of the contribution comes from other raw materials production. Copolyester of PLA production shows a 15 % contribution.

- **Acidification.** Clips production accounts for 24 % of the acidification impact due to the sulphur dioxide (77 %) and nitrogen oxides (21 %) emissions from the electricity consumption. The copolyester PLA production accounts for 23 % of the acidification impact due to the emissions caused by the production of the chemicals used in the lactic acid production process (refer to case study 3 for more details). A significant (21 %) contribution comes from the emissions (sulfur dioxide and nitrogen oxides) from the production of chemicals used in other raw materials production. Biomass production contributes 12 % to the acidification impact, due to direct ammonia emissions to air (83 %) from the use of manure and fertilisers.

- **Ionizing radiation.** For ionizing radiation, cumulatively 78 % of the contribution comes from granulate production, and clips production where the key activity data is electricity used. Copolyester of PLA production shows a 13 % contribution.

- **Freshwater eutrophication.** For freshwater eutrophication, cumulatively 84 % of the contribution comes from clips production (61 %) and granulate production (23 %), where the key activity data is electricity used.

- **Photochemical ozone formation.** The copolyester PLA production accounts for 36 % of photochemical ozone formation. This is due to the nitrogen oxides emissions from the cultivation of maize used for PLA and the air emissions from the PLA production process itself (please refer to case study 3 for more details). Other raw material production has a 20 % contribution due to nitrogen oxides and non-methane volatile organic compounds emissions caused by the production of the chemicals used. The remaining contribution mainly comes from clips production (16 %) and granulates production (13 %) where the key activity data is electricity used.

Additional analysis of impact categories where biomass production has a significant contribution reported here:

- **Terrestrial eutrophication.** For terrestrial eutrophication, 21 % of the contribution comes from potato production, 31 % from copolyester production and 22 % from other raw materials production. The impacts originate from direct ammonia emissions to air due to use of manure and fertilisers, and direct nitrous emissions to air.

- **Marine eutrophication.** For marine eutrophication, 32 % of the contribution comes from potato production, 24 % from copolyester production and 26 % from other raw materials production. 66 % of the impacts originate from direct nitrate emissions to water from application of fertilisers and from crop residues. 26 % originates from nitrous emissions to air.

- **Land use.** For land use, 60 % of the contribution comes from land occupation for maize production for PLA. 23 % of the contribution comes from potato production. Production of other renewable raw materials has a 21 % contribution to land use. Due to the avoided animal feed production by the starch by-products, a 15 % reduction in land use is seen in the starch production process.

3.2.1 Cradle-to-gate environmental impacts of bio-based alternative system: Bio-based clips from potato waste

The cradle-to-factory gate LCA results for one functional unit (72 kg) of bio-based clips for thealternative system is presented in Table 84.

Table 84. Cradle-to-gate LCA results of 1 functional unit bio-based clips (alternative system) (excluding DLUC and ILUC effects).

Impact category	Unit	Bio-based clips (alternative)
Climate change	kg CO_2 eq.	1.18E+02
Ozone depletion	kg CFC-11 eq.	1.26E-05
Human toxicity, non-cancer effects	CTUh	4.20E-05
Human toxicity, cancer effects	CTUh	1.45E-05
Particulate matter	kg PM2.5 eq.	6.97E-02
Ionizing radiation, human health	kBq U235 eq.	4.43E+01
Photochemical ozone formation	kg NMVOC eq.	6.71E-01
Acidification	molc H+ eq.	1.07E+00
Terrestrial eutrophication	molc N eq.	2.46E+00
Freshwater eutrophication	kg P eq.	7.26E-02
Marine eutrophication	kg N eq.	3.01E-01
Freshwater ecotoxicity	CTUe	1.09E+03
Land use	kg C deficit	8.13E+02
Water use	m^3	8.59E+01
Abiotic depletion	kg Sb eq.	1.42E-04
Abiotic depletion (fossil fuels)	MJ	3.44E+03
NREU	MJ	4.32E+03

In Figure 61, the impacts of potato waste-based starch bio-based clips are broken down into the main unit processes described in the section 2.1 inventory analysis.

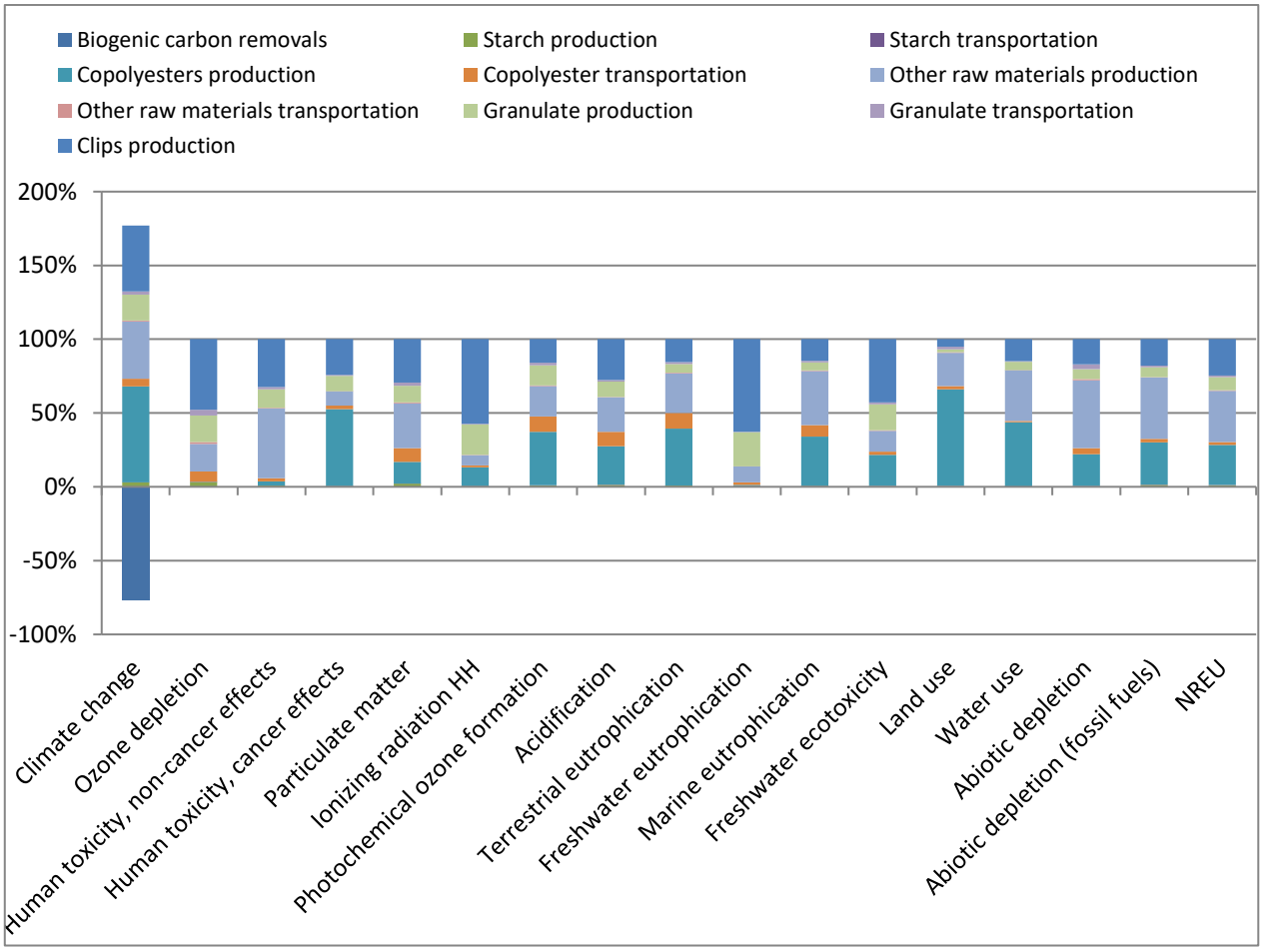


Figure 61. Breakdown of the cradle-to-gate LCA results for bio-based clips (alternative, from potato waste)

The breakdown is similar to the breakdown of the potato starch-based product system given in Figure 60. The difference lies in the fact that there is no impact coming from potato production. The savings achieved in starch production in the baseline system by avoided animal feed production with the by-products is not seen in this alternative system. The alternative system of production from potato waste performs slightly better (up to 20 %) than the virgin potato-based production. However, when comparing with the aggregated results of maize and potato based production (Figure 61), the alternative system is seen to perform worse in climate change (8 %), NREU (5 %), human toxicity (cancer effects) (40 %) and ionizing radiation (23 %) showing maize starch based starch performing better in these categories. Due to the aggregated data for the maize starch-based production we are not able to trace the origin of the difference.

3.2 Cradle-to-gate environmental impacts of the petrochemical reference system

The cradle-to-factory gate LCA results for 1 functional unit (54 kg) of petrochemical PP clips is presented in Table 85. This is calculated by multiplying the impact of plastic granulate production (given in Table 35 for 1 kg PP granulate) with 54 kg to convert per functional unit (FU). Impacts of granulate transportation and injection moulding into clips are added to this.

Based on Chapter 6 *Ranges for environmental impacts from production of fossil-based plastics*, 7 out of 17 impact categories were found suitable for PP, namely: climate change, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication, abiotic depletion (fossil fuels) and NREU (see chapter 6). This is because comparison of different datasets revealed large differences for the other impact categories. The data used is based on PlasticsEurope, since it provides the most up to date eco-profile for European fossil-based plastics. However, it does not conform with ILCD requirements in its presentation of inventory details or impact assessment results. It does not distinguish between emissions to fresh water and to seawater, and it is therefore not possible to calculate the impacts on human health and ecosystems as well as freshwater and marine eutrophication, because the receiving compartment is an important element in determining the fate and effect of an emission. Further details can be found in Table 35 where the environmental impact of 1 kg of PP and the proposed ranges of variation within each impact category are shown.

Table 85. Cradle-to-gate LCA results of petrochemical PP clips for 1 FU (=54 kg).

Impact category	Unit	Petrochemical PP clips
Climate change	kg CO_2 eq.	1.31E+02
Particulate matter	kg PM2.5 eq.	3.46E-02
Photochemical ozone formation	kg NMVOC eq.	3.86E-01
Acidification	molc H+ eq.	5.09E-01
Terrestrial eutrophication	molc N eq.	9.84E-01
Abiotic depletion (fossil fuels)	MJ	4.49E+03
Non-renewable energy use (NREU)	MJ	4.93E+03

In Figure 62, the impacts of petrochemical PP clips are broken down into the main unit processes of granulate production, transportation and clips production. It is seen that production of PP granulates shows the highest contribution in all impact categories selected as suitable for comparison. Its contribution ranges from 52 % to 89 %. The ecoprofile for the PP granulate comprises extraction and refining of crude oil and natural gas, fluid catalytic cracking of heavy hydrocarbons, steam cracking of naphtha into lower olefins, and polymerisation of the monomers into polypropylene. However, it is not possible to show the breakdown of the impact of granulate production into these unit processes due to in the lack of transparency of the PlasticsEurope data. The transportation is within Europe and shows negligible contribution to impacts. Injection moulding process shows significant a contribution (10-44 %). The main activity level data of this process is electricity consumption.

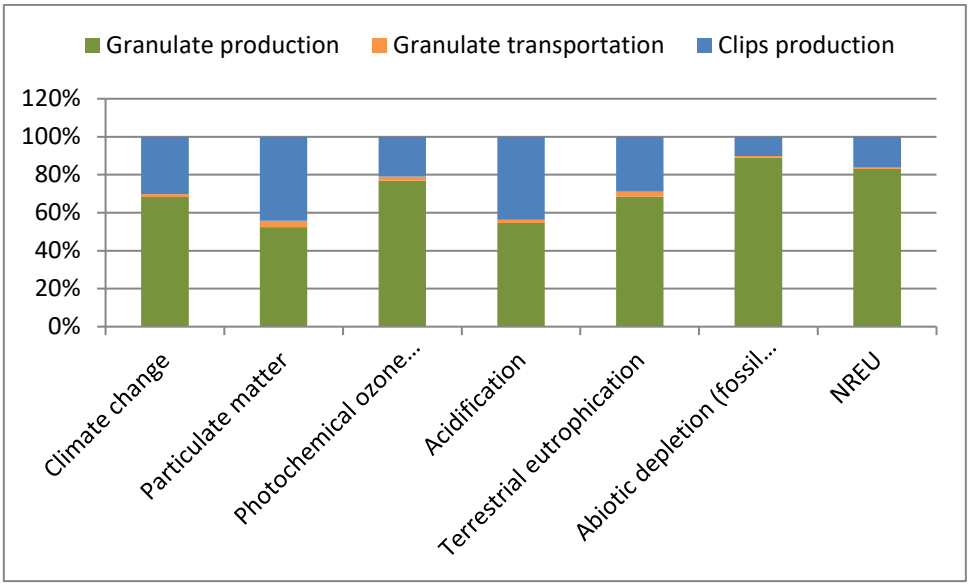


Figure 62. Breakdown of the cradle-to-gate LCA results for petrochemical PP clips.

3.3 End of life results and interpretation for single-use clips

The results for the EoL impact assessment modelled in EASETECH are presented in this paragraph. The presentation includes (1) mass and energy flows of all product systems in all technologies, (2) the LCIA results for each product system, (3) a comparison between product systems, and (4) the EoL mix.

In Annex 4 further results are presented, the contribution analysis is in Figure 226 ff. and the weighted results in Table 227 ff. The contribution analysis for each EoL technology is presented in Annex 4 and analysed under each product system.

3.3.1 Mass and energy flows

The mass and energy flows, as outputs from EASTECH, are presented in Table 86 and Table 87. These flows refer to the single use clips without the additional soil. The results correspond to the circular footprint formula in PEFCR guidance draft version 6.3 (European Commission, 2017). In addition, the paragraph looks at the carbon flows. The flows correspond to the flows observed in the flow charts for each technology in the approach and methodology chapter.

For the incineration technology 31 % of the energy in the material input is recovered as heat/electricity for both petrochemical PP and the bio-based starch. When landfilling the starch clips, the energy recovery is 7 % (methane formation by degradation in the landfill), while no energy is utilised from landfilling PP clips (as PP plastic does not degrade).

Landfilling the bio-based single-use clips permanently stores 61 % of the total mass (without soil) in the landfill and 30 % of the biogenic carbon; the rest is emitted due to biodegradation. For the petrochemical PP 100 % of the mass is stored in the landfill.

For the in-situ technology 10 % of the biogenic carbon is assumed to be stored permanently in the soil, the remainder is lost as air emissions. Of the total mass, 73 % is left in-situ.

Table 86. Material and energy flow for 1 kg starch with 100 g contamination. Box letters refer to flow charts in the approach and methodology chapter (Figure 8 to Figure 13).

Starch

Technology	Box	Process	Mass kg	Bio carbon kg	Fossil carbon kg	Gas m³ CH_4	Energy substitu-tion MJ
Material	I	Material input	1.10E+00	3.25E-01	2.27E-01		2.17E+01
Incineration w/wo energy recovery *	E, H	Energy pro-duction	1.10E+00	3.25E-01	2.27E-01		6.69E+00
	R1	Fly Ash	1.04E-01	0.00E+00	0.00E+00		
	R2	Bottom ash	2.95E-01	3.25E-04	2.05E-04		
	D	Direct emissions		3.25E-01	2.26E-01		
Landfill	L	Leachate	1.34E+00	0.00E+00	0.00E+00		
	G, E	Landfill gas		2.28E-01		2.47E-01	1.56E+00
	CS	Storage in landfill	6.68E-01	9.69E-02	2.05E-01		
In-situ	C1	Air emissions		2.93E-01			
	C2	Storage in soil	8.07E-01	3.21E-02			

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

Table 87. Material and energy flow for EoL of 1 kg fossil PP with 100 g contamination. Box letters refer to flow charts in the approach and methodology chapter (Figure 8 to Figure 13).

PP						
Technology	**Box**	**Process**	**Mass kg**	**Fossil carbon kg**	**Gas m³ CH_4**	**Energy substitution MJ**
Material	I	Material input	1.10E+00	8.11E-01		4.02E+01
Incineration w/wo energy recovery *	E, H	Energy production	1.10E+00	8.11E-01		1.24E+01
	R1	Fly Ash	2.65E-02	0.00E+00		
	R2	Bottom ash	6.54E-02	2.62E-06		
	D	Direct emissions		8.11E-01		
Landfill	L	Leachate	2.20E+00	0.00E+00		
	G, E	Landfill gas			0.00E+00	0.00E+00
	CS	Storage in landfill	1.10E+00	8.11E-01		

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

3.3.2 LCIA for product systems

The impact assessment results are presented per the EoL-reference flow, i.e. 1 kg plastic and 100 g of contamination and 3 kg of soil for each EoL technology (assuming 100 % disposal via that technology), see Table 80.

The overall LCIA results for both the petrochemical PP clips and the bio-based clips are presented in Table 88 for all EoL technologies. The results are commented upon and further explained in the following sections.

Table 88. Total results of treating 1 kg bio-based starch clips and 1 kg petrochemical PP clips (incl. 100 g contamination and 3 kg of soil) for all impact categories. The intended EoL option is marked with a bold box.

Impact category	Unit	Bio-based starch			Petrochemical PP	
		Incinera-tion	Land-filling	In-situ	Incinera-tion	Land-filling
Climate change - fossil	kg CO_2 eq.	2.00E+00	3.00E-02	-1.21E-01	1.30E+01	9.39E+00
Climate change - biogenic	kg CO_2 eq.	1.66E+00	4.06E+00	1.08E+00	4.42E-01	2.78E-01
Climate change - total	kg CO_2 eq.	9.92E+00	1.03E+01	9.54E-01	1.35E+01	9.67E+00
Ozone depletion	kg CFC-11 eq.	-2.89E-08	1.89E-06	-1.44E-12	-4.81E-08	1.19E-08
Human toxicity, non-cancer effects	CTUh	3.75E-07	6.82E-09	3.09E-08	2.14E-06	1.80E-06
Human toxicity, cancer effects	CTUh	2.02E-07	-2.98E-10	6.89E-09	2.29E-07	1.25E-08
Particulate matter	kg PM2.5 eq.	1.42E-04	-2.96E-05	1.38E-06	9.77E-04	9.31E-04
Ionizing radiation HH	kBq U235 eq.	4.95E-02	-3.39E-03	-5.68E-06	1.11E-01	2.58E-02
Photochemical ozone formation	kg NMVOC eq.	1.03E-02	1.24E-03	2.10E-04	4.82E-02	3.91E-02
Acidification	molc H+ eq.	1.06E-02	7.55E-05	1.52E-04	4.61E-02	3.63E-02
Terrestrial eutrophication	molc N eq.	4.82E-02	4.94E-04	8.96E-04	2.09E-01	1.66E-01
Freshwater eutrophication	kg P eq.	6.88E-04	-3.53E-06	1.39E-07	7.18E-04	1.70E-05
Marine eutrophication	kg N eq.	4.07E-03	1.91E-04	8.21E-05	1.83E-02	1.48E-02
Freshwater ecotoxicity	CTUe	3.52E+00	1.57E-02	9.40E-02	5.23E+00	1.01E+00
Land use	kg C deficit	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	m^3	1.06E+00	-3.62E-02	-3.07E-03	7.83E+00	7.18E+00
Abiotic depletion	kg Sb eq.	-1.51E-07	-6.54E-08	-1.25E-10	-3.44E-08	3.32E-07
Abiotic depletion (fossil fuels)	MJ	1.48E+01	-4.34E-02	-5.37E-02	1.32E+02	1.26E+02
NREU	MJ	1.36E+01	8.52E-02	-5.35E-02	1.29E+02	1.25E+02

The significance of the colour scale is the same as for all the product systems. This means that you can compare the colour horizontally across the table, but only comparing between the same impact category.

3.3.3 LCIA for baseline, Bio-Based clips

The characterised results for the EoL options of bio-based starch clips shows that landfilling and in-situ field-based biodegradation performs better than incineration (greener colours). It seems there is a slight preference for in-situ in several impact categories. The weighted results show that the in-situ technology has the best performance for bio-based clips followed by landfilling and incineration.

Further details on the individual EoL options for the bio-based starch clips are presented below, looking into specific impact categories and contribution of impacts from different processes and elemental emissions.

3.3.3.1 Intended EoL technology: In-situ of biodegradation of starch

The LCIA results for the bio-based clips show that in-situ biodegradation technology has the best performance of the product systems in most impact categories (see Table 88). For a few impact categories, the in-situ technology performs worse than the other EoL technologies (e.g. in human toxicity, cancer effects). Here, the largest contribution originates from metal emissions to soil and water from the 100 g of contamination consisting of garden waste, soil, stones and foreign objects.

The in-situ model for bio-based clips covers the avoidance of gathering of the clips from the field, contributing to savings in all impact categories. Furthermore, air emissions (CO_2 and to a lesser extent CH_4) from left-on-field application of the starch are included. As the clips are left in the soil, there is also no removal of soil, i.e. all negative impacts that are associated with soil removal in the other EoL scenarios are avoided.

Looking at the contribution analysis for the in-situ technology (see Annex 4), only two processes contribute to emissions: Avoided collection and use-on-land. The contribution analysis shows that the use-on-land emissions contribute significantly more to all impact categories than the avoided collection. The avoided collection has a contribution of 17 %-25 % in the impact categories of NREU, abiotic depletion, abiotic depletion (fossil), water use, ozone depletion and climate change (fossil).

3.3.3.2 Landfilling of starch

Landfilling of bio-based clips has very similar environmental impacts as the in-situ technology (see Table 88). Landfilling of the bio-based clips causes the largest impacts to the impact categories of climate change, ozone depletion and photochemical ozone formation. The landfill gas is the primary cause of these impacts.

The leachate treatment has the largest contribution to marine eutrophication due to ammonium, nitrate and nitrogen oxide emissions to water and air.

The electricity substitution produced from the collected landfill gas has the largest contribution in 11 out of 18 impact categories.

3.3.3.3 Incineration of starch

Incineration of the bio-based clips shows the best performance in two impact categories, and the worst in 14 (see Table 88). The direct emissions from incineration of the starch clips have the largest contribution to the climate change, terrestrial eutrophication, marine eutrophication and photochemical ozone formation.

Bottom ash treatment has the largest contribution to the human toxicity impacts, freshwater eutrophication and ecotoxicity.

The largest potential savings caused by the incineration technology comes from substitution of heat and electricity. The savings from substitution of heat contribute to the largest share in the impact categories of NREU, abiotic depletion (fossil fuels) and ozone depletion. The savings from electricity substitution contribute the most in water use, abiotic depletion and particulate matter. The treatment of an additional 3 kg of soil in the incineration plant contributes to a high impact in several impact categories (e.g. climate change, water use, NREU).

3.3.3.4 Weighted end of life results for the baseline product system (bio-based clips)

The characterised results for the bio-based clips have been normalised and then weighted, see Figure 63. The numerical results can be found in Annex 4. For methodology and choice of normalisation and weighing factors, please see the Approach and methodology chapter, section 5.2.3 "Use of normalised and weighted results to determine the preference of EoL option" and the values in Table 30.

The weighted results show, that the in-situ technology has the best performance for bio-based clips followed by landfilling and incineration. The largest contribution to the weighted results is the climate change impact category.

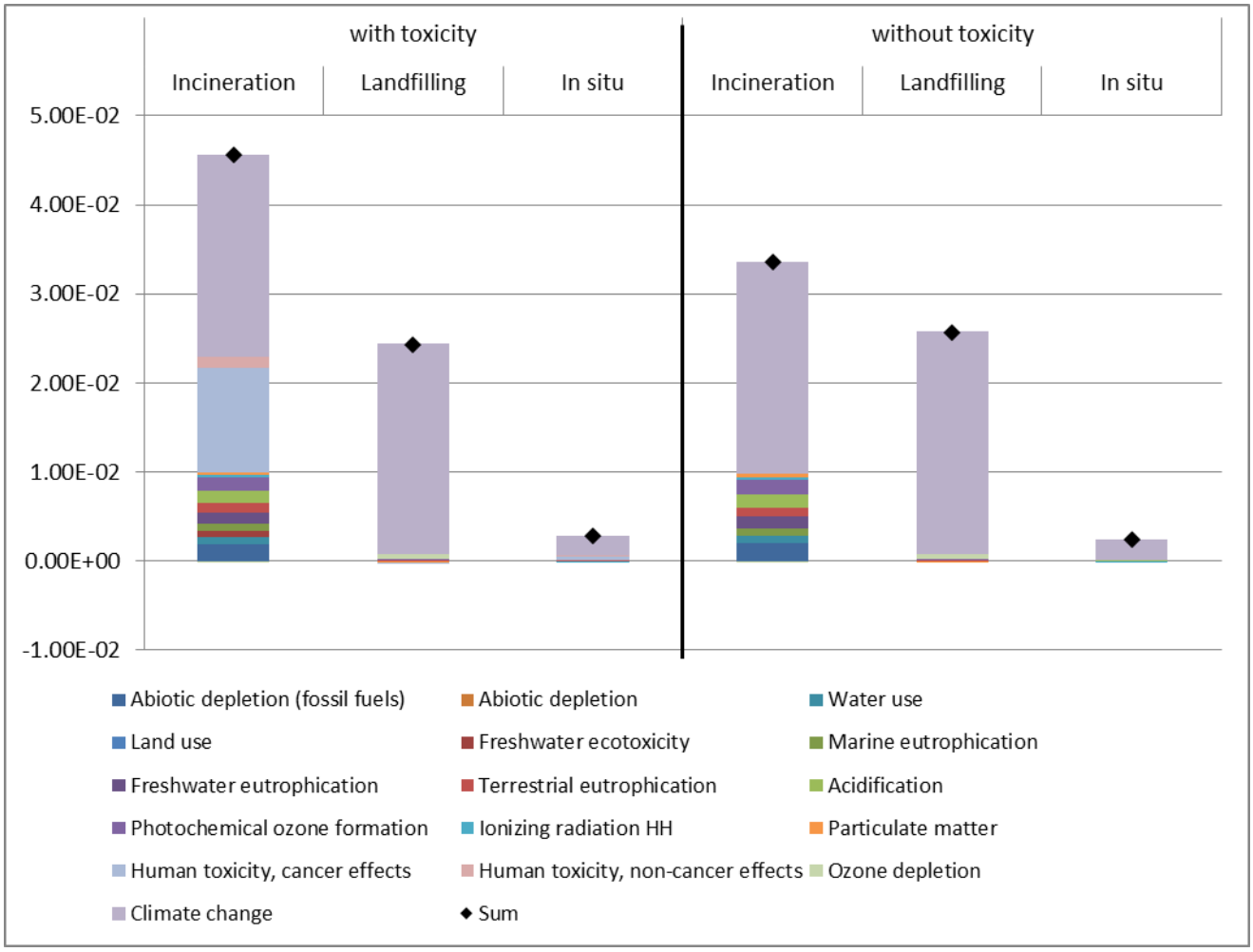


Figure 63. Weighted EoL results for the baseline product system, bio-based starch clips. The numerical figures can be found in Annex 4.

3.3.4 LCIA for the reference product system, petrochemical PP

The characterised results for the EoL options of the petrochemical clips show an overall preference of landfilling the PP clips in comparison with incineration.

Further details on the individual EoL options for the PP clips are presented below, looking into specific impact categories and contribution of impacts from different processes and elemental emissions.

3.3.4.1 Landfilling of PP

Overall the results in Table 88 show that landfilling of petrochemical PP single-use clips perform better in more impact categories compared to incineration.

Since the plastic is not degraded within the first 100 years of modelling, no landfill gas is generated, thus there is no energy recovery or emissions of landfill gas. This excludes the impact categories of abiotic depletion, ozone depletion and non-renewable energy use, where landfilling is not preferred.

The construction and operation of the landfill has the largest impact in the categories of climate change, ozone depletion, the human toxicity cancer effects, ionising radiation, particulate matter, freshwater eutrophication, water use and abiotic depletion.

Transportation to the landfill has a large impact on climate change, human toxicity (non-cancer), photochemical ozone formation, terrestrial eutrophication and NREU.

Leachate treatment has a substantial impact of 48 % in freshwater ecotoxicity and 64 % in marine eutrophication but contributes less than 20 % to the other impact categories. The marine eutrophication impact from leachate treatment mostly comes from ammonium that is released to surface water.

Gathering the clips from the field and sieving the soil contribute substantially (more than 60 % of the absolute contribution) to all impact categories, except ozone depletion.

3.3.4.2 Incineration of PP

Direct emissions from the incineration process contribute significantly to most impact categories, e.g. climate change, photochemical ozone formation, terrestrial and marine eutrophication.

Incineration of the plastic generates electricity and heat which contribute to savings of environmental impacts in most categories. For examples, heat substitution has the largest savings in the impact categories of NREU, abiotic depletion (fossil fuels), ozone depletion and the largest impact in the ionising radiation category.

Bottom ash treatment has the largest impact on human toxicity (cancer effects) and freshwater ecotoxicity. The large contribution in cancer effects is due to chromium emissions to water and soil.

3.3.5 Details on climate change

The impact assessment results are presented per the EoL-reference flow, i.e. 1 kg plastic, 100 g of contamination and 3 kg of soil for each EoL technology (assuming 100 % disposal via that technology), see Table 80.

The total climate change results of the different EoL technologies are presented in Figure 64. The figure shows that the preferred EoL technology regarding climate change impact for the bio-based starch clips is clearly the in-situ technology (0.95 kg CO_2-eq per EoL-reference flow). For the PP clips the preferred technology regarding climate change impact is landfilling (13.5 and 9.7 kg CO_2-eq per EoL-reference flow, respectively, for incineration and landfilling).

The gathering and sieving process has a significant contribution to climate change for all EoL technologies expect the in-situ technology. Treatment of 3 kg of soil and the direct emissions have significant contributions for the incineration technology. As the bio-based starch is biodegradable, and thus may generate landfill gas over time, the potential impacts from landfilling the bio-based clips are higher than for the PP clips. The transportation process includes the avoided collection in the in-situ technology.

The bio-based clips weigh 33 % more per FU. This is one reason why incineration and landfilling of the bio-based clips perform worse with respect to climate change impact than landfilling of the petrochemical PP clips, and are comparable to incineration of the petrochemical PP clips.

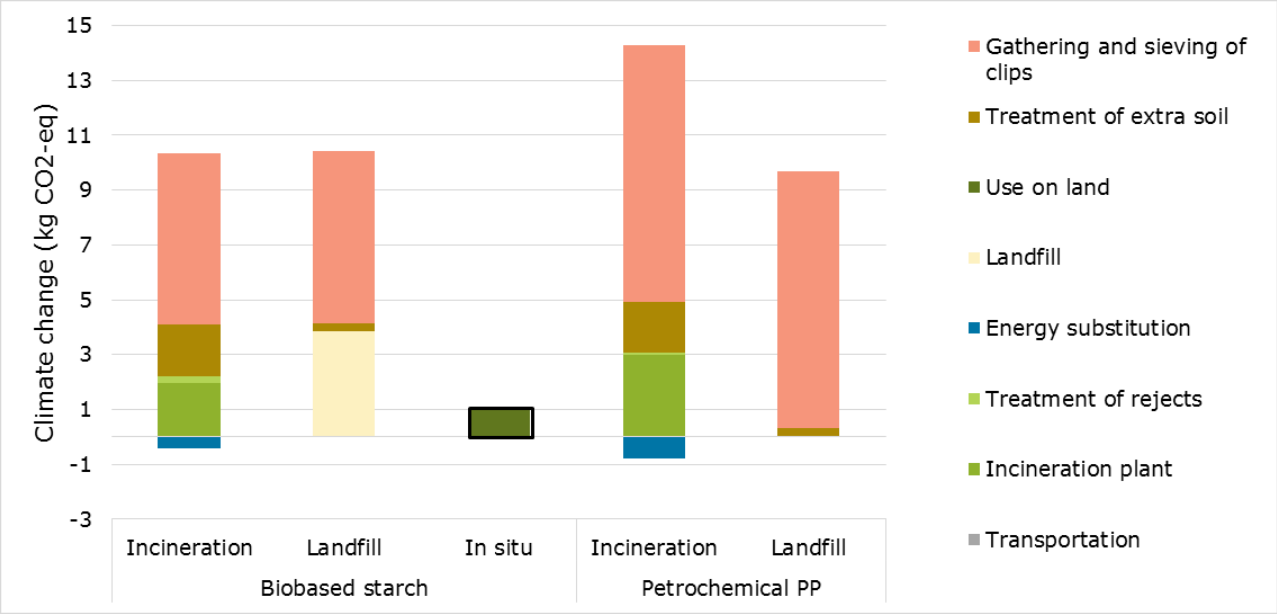


Figure 64. Climate change for treating 1 kg of single-use clips (incl. 100 g contamination and 3 kg of soil) in different EoL technologies. The intended EoL option is marked with a black box. See Annex 4 for details of the content of the grouped processes. The transportation process in the in-situ EoL options consists of avoided collection. The results without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied with 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

The gathering of clips and sieving of soil contributes 75 % on average to all impact categories (the contribution ranges from 0-99 %) for the landfill EoL of bio-based starch clips. The gathering and sieving has the highest contribution to 16 out of 18 impact categories. The treatment of soil in the landfill contributes 3 % on average to all impact categories (the contribution ranges from 0-7 % for the impact categories), of the EoL LCIA results on landfilling.

For the incineration EoL technology of bio-based starch, the gathering of clips and sieving of soil from the clips contributes 43 % on average for all impact categories (the contribution ranges from 1 to 74 %). The treatment of soil in incineration together with bio-based starch clips contributes to 33 % in average for all impact categories (ranging in contribution from 3-92 %). It is only in the impact categories of particulate matter and ionizing radiation where the gathering, sieving and treatment of soil does not contribute more than half of the impacts for incineration.

Gathering and sieving together with treatment of soil has a substantial influence on the climate change impact of EoL results for landfilling and incineration.

3.3.6 Comparing all EoL technologies in all the product systems with weighted results

The weighted results are shown per functional unit (i.e. different from the above results which are is for the EoL-reference flow of 1 kg plastic and 100 g contamination and 3 kg soil), in

Figure 65. This shows that the in-situ EoL option of the bio-based starch clips preforms the best environmentally, landfilling second best and incineration worst. In fact, all three EoL options for the bio-based clips are preferable compared to the petrochemical clips, even considering the higher weight of the bio-based clips compared to the petrochemical clips.

The largest contribution from incinerating the starch clips is from climate change and human toxicity (cancer effects), which respectively are from collection and sieving as well as the direct emissions at the incineration plant, and from the bottom ash treatment. Landfilling the bio-based and biodegradable starch have large impacts on climate change, which is from landfill gas direct emissions and flaring. The in situ EoL option also has its main contribution in the climate change impact category, which is from direct emissions.

Incinerating the petrochemical PP clips contributes to climate change, abiotic depletion of fossil fuels and human toxicity (cancer effects). These impacts originate from the collection of the clips and sieving of the soil, and for the latter impact category from the bottom ash treatment. For landfilling, the contributions are mainly from climate change and abiotic depletion (fossil fuels) due to collection and sieving.

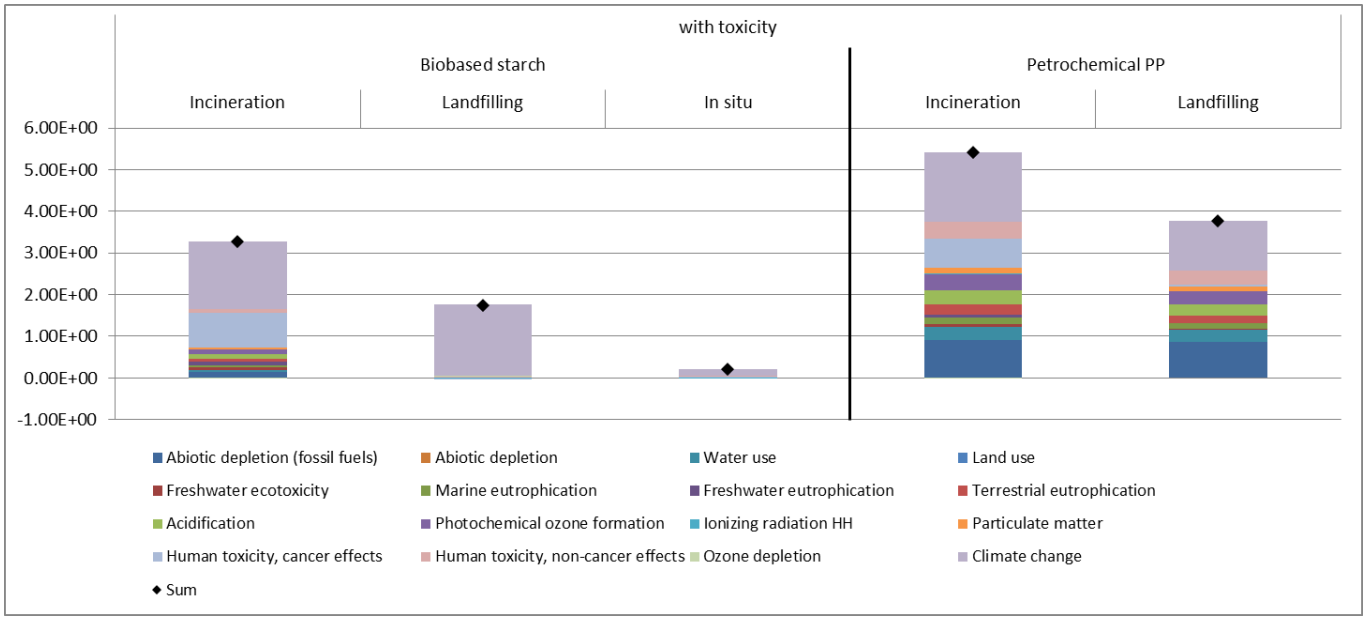


Figure 65. The weighted results with toxicity of the two products systems for single-use clips for all EoL options per functional unit. Numerical values can be found in Annex 4.

3.3.7 End of life mix results

The LCIA for all product systems for the estimated EoL mix per functional unit are presented in Table 89.

Comparing the two systems the bio-based starch product performs considerably better than the petrochemical PP product. The bio-based product will be left on the field and therefore the additional soil that is gathered with the clips is not removed. Gathering and sieving the clips and removing the additional soil that is attached to the clips contributes to a large share of the impacts. One impact category where the bio-based clips are less favourable is caused by larger savings in ozone depletion when substituting electricity and heat produced from incineration of petrochemical PP clips. Using bio-based starch clips with an in-situ EoL can save 568 kg CO_2 eq. per FU compared to the mix of incineration and landfilling of the petrochemical PP clips.

Moreover, 176 CTUe per FU can be saved in freshwater ecotoxicity and 6,870 MJ per FU of NREU by using the bio-based starch clips with an in-situ EoL technology.

Table 89. LCIA for EoL mix of the single use clips for all product systems per functional unit (45 000 clips covering 1 ha, i.e. 72 kg bio-based starch clips and 54 kg petrochemical PP clips).

Impact category	Unit	Bio-based starch	Petrocehmical PP
In-situ	%	100	
Incineration	%		56
Landfill	%		44
Climate change - fossil	kg CO_2 eq. / FU	-8.74E+00	6.17E+02
Climate change - biogenic	kg CO_2 eq. / FU	7.75E+01	2.00E+01
Climate change - total	kg CO_2 eq. / FU	6.87E+01	6.37E+02
Ozone depletion	kg CFC-11 eq. / FU	-1.04E-10	-1.17E-06
Human toxicity, non-cancer effects	CTUh / FU	2.23E-06	1.07E-04
Human toxicity, cancer effects	CTUh / FU	4.96E-07	7.21E-06
Particulate matter	kg PM2.5 eq. / FU	9.90E-05	5.17E-02
Ionizing radiation HH	kBq U235 eq. / FU	-4.09E-04	3.97E+00
Photochemical ozone formation	kg NMVOC eq. / FU	1.51E-02	2.39E+00
Acidification	molc H+ eq. / FU	1.09E-02	2.26E+00
Terrestrial eutrophication	molc N eq. / FU	6.45E-02	1.03E+01
Freshwater eutrophication	kg P eq. / FU	9.98E-06	2.21E-02
Marine eutrophication	kg N eq. / FU	5.91E-03	9.05E-01
Freshwater ecotoxicity	CTUe / FU	6.77E+00	1.82E+02
Land use	kg C deficit / FU	0.00E+00	0.00E+00
Water use	m^3 / FU	-2.21E-01	4.07E+02
Abiotic depletion	kg Sb eq. / FU	-9.00E-09	6.86E-06
Abiotic depletion (fossil fuels)	MJ / FU	-3.87E+00	6.97E+03
NREU	MJ / FU	-3.85E+00	6.87E+03

3.4 Cradle-to-grave environmental impacts and interpretation

3.4.1 Bio-Based baseline system

The aggregated cradle-to-grave LCA results from the assessment of one functional unit of bio-based clips (72 kg) is given in Table 90. The breakdown of impacts between biomass production, manufacturing and EoL is shown in Figure 66. For EoL of bio-based clips, 100 % biodegradation in soil is the intended and real scenario, and the values in Table 90 refer to that. The manufacturing phase dominates the impacts. Biomass production has significant contribution to water use, land use, human toxicity (non-cancer effects), marine eutrophication and terrestrial eutrophication. For the other impact categories, biomass production has a minor contribution. The EoL treatment of in-situ biodegradation is seen to have negligible impact in the overall cradle-to-grave results, except for climate change. The climate change impact of the EoL stage is due to the direct emissions from the degradation of the product on field.

Table 90. Cradle-to-grave LCA results of 1 functional unit bio-based clips baseline system (excluding DLUC and ILUC effects).

Impact category	Unit	Biomass	Manufactu ring	EoL	Total
Climate change	kg CO_2 eq.	-7.61E+01	1.85E+02	6.87E+01	1.78E+02
Ozone depletion	kg CFC-11 eq.	6.83E-07	2.11E-05	-1.04E-10	2.17E-05
Human toxicity, non-	CTUh	3.47E-05	4.21E-05	2.23E-06	7.90E-05

cancer effects					
Human toxicity, cancer effects	CTUh	5.07E-07	9.88E-06	4.96E-07	1.09E-05
Particulate matter	kg PM2.5 eq.	6.27E-03	8.68E-02	9.90E-05	9.31E-02
Ionizing radiation, human health	kBq U235 eq.	1.97E-01	3.58E+01	-4.09E-04	3.60E+01
Photochemical ozone formation	kg NMVOC eq.	3.56E-02	6.22E-01	1.51E-02	6.73E-01
Acidification	molc H+ eq.	1.58E-01	1.22E+00	1.09E-02	1.39E+00
Terrestrial eutrophication	molc N eq.	6.78E-01	2.01E+00	6.45E-02	2.75E+00
Freshwater eutrophication	kg P eq.	2.94E-03	7.08E-02	9.98E-06	7.38E-02
Marine eutrophication	kg N eq.	1.85E-01	2.18E-01	5.91E-03	4.08E-01
Freshwater ecotoxicity	CTUe	6.22E+01	1.11E+03	6.77E+00	1.18E+03
Land use	kg C deficit	2.41E+02	4.16E+02	0.00E+00	6.58E+02
Water use	m^3	6.79E+01	3.69E+01	-2.21E-01	1.05E+02
Abiotic depletion	kg Sb eq.	6.03E-05	4.24E-04	-9.00E-09	4.85E-04
Abiotic depletion (fossil fuels)	MJ	7.50E+01	3.40E+03	-3.87E+00	3.47E+03
NREU	MJ	7.83E+01	4.02E+03	-3.85E+00	4.09E+03

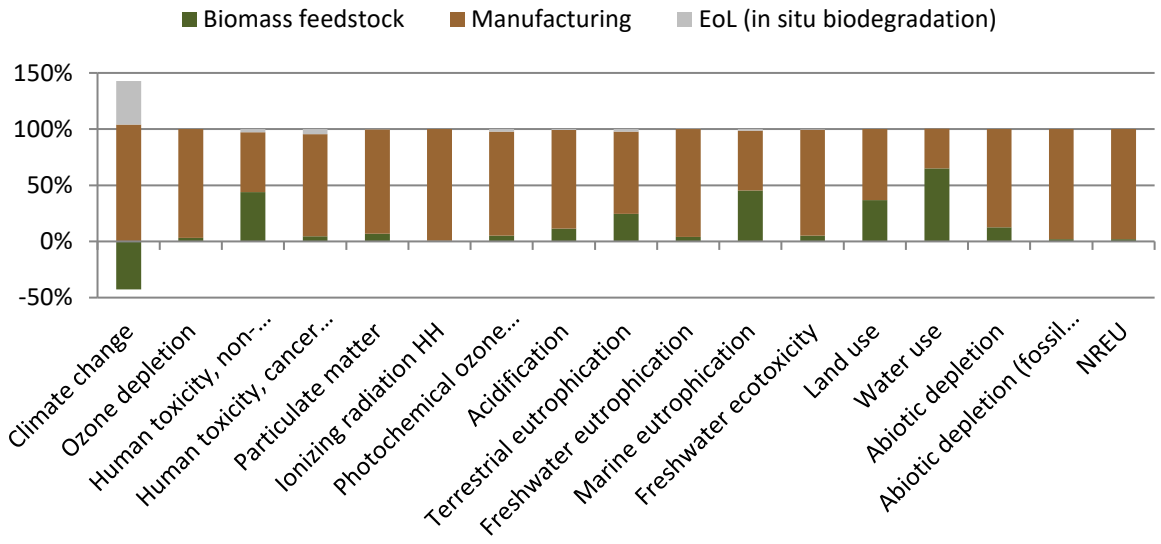
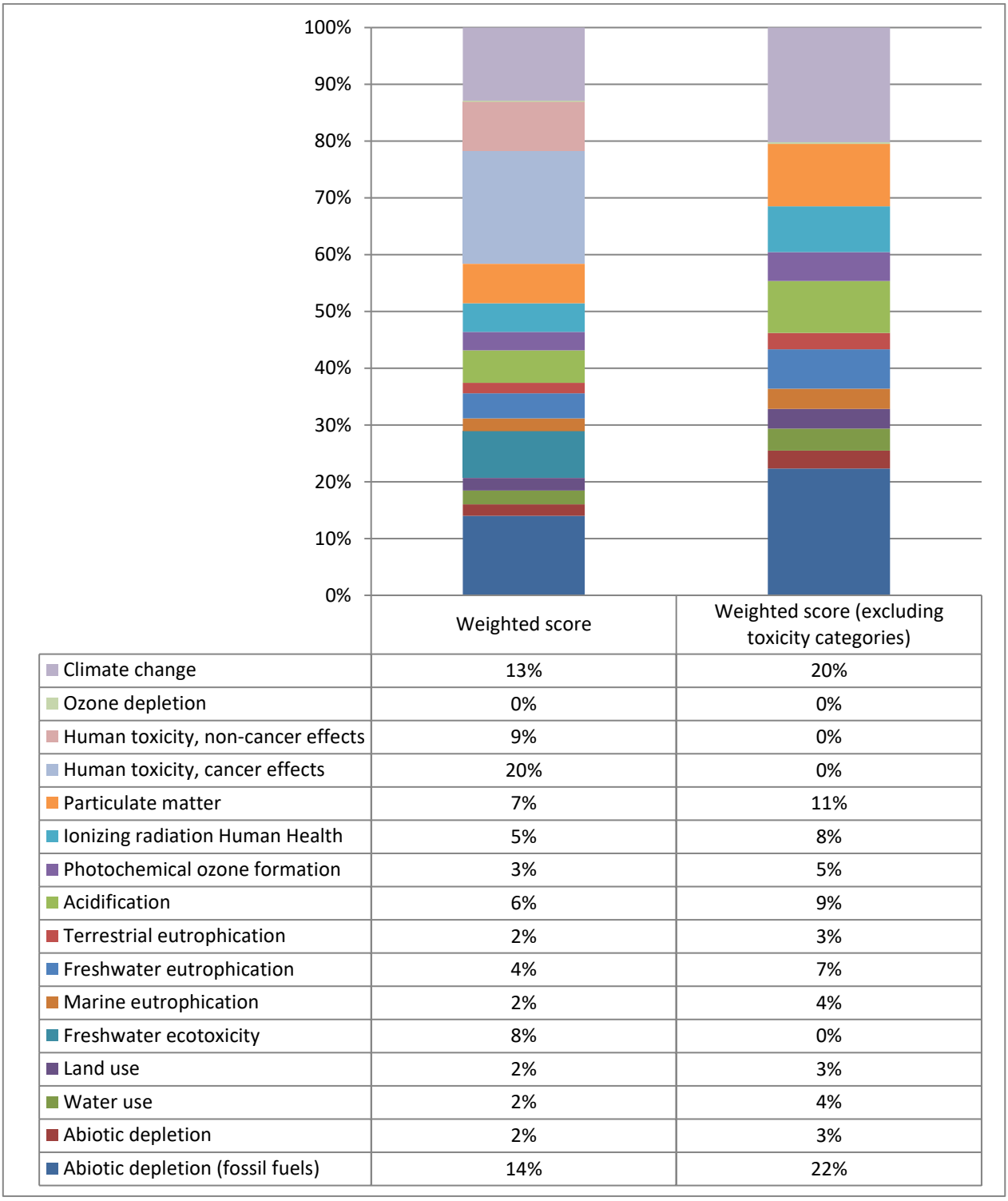


Figure 66. Breakdown of the potential impacts from the bio-based clips baseline system, cradle-to-grave results (excluding DLUC and ILUC effects).

According to PEFCR guidance v 6.3, the most relevant impact categories are identified as all impact categories that cumulatively contribute to at least 80 % of the total environmental impact (excluding toxicity related impact categories). Based on the normalised and weighted results shown in Figure 67, excluding toxicity related impacts, the most important impact categories are: abiotic depletion (fossil fuels) (22 %), climate change (20 %), particulate matter (11 %), acidification (9 %), ionizing radiation (8 %), freshwater eutrophication (7 %) and photochemical ozone formation (5 %), which accounts for a cumulative 83 % contribution to the total impacts. These impact categories are identified as the hot spots for this case study and detailed analysis is made for each of them in section 3.1.1. The rest of the impact categories each contribute up to 4 %.



	Weighted score	Weighted score (excluding toxicity categories)
Climate change	13%	20%
Ozone depletion	0%	0%
Human toxicity, non-cancer effects	9%	0%
Human toxicity, cancer effects	20%	0%
Particulate matter	7%	11%
Ionizing radiation Human Health	5%	8%
Photochemical ozone formation	3%	5%
Acidification	6%	9%
Terrestrial eutrophication	2%	3%
Freshwater eutrophication	4%	7%
Marine eutrophication	2%	4%
Freshwater ecotoxicity	8%	0%
Land use	2%	3%
Water use	2%	4%
Abiotic depletion	2%	3%
Abiotic depletion (fossil fuels)	14%	22%

Figure 67. Contribution of normalised and weighted results of bio-based clips (baseline system), with and without toxicity categories.

Contributions of the life cycle stages and the different processes within are shown in Figure 68. Please refer to section 3.1.1 for more detailed elaboration per impact category of the contributing processes for the potato starch-based system. For the maize starch-based process due to aggregated data it is not possible to have a break down into unit processes for the granulate production. Analysis is made on the elementary flows for the three most relevant

impact categories. For abiotic depletion (fossil fuels), 55 % of the contribution comes from natural gas, 23 % from crude oil and the rest from coal. For climate change, 80 % of the contribution comes from CO_2 (fossil) emissions, 14 % from methane (fossil) and 6 % from dinitrogen monoxide emissions. For particulate matter 50 % of the contribution comes from particulates and 44 % from sulphur dioxide emissions to air.

As shown in Figure 68, production of the clips is the most important life cycle stage, cumulatively contributing to 88 % (91 % without toxicity categories) of the impacts. The granulate production is the most significant process with about a 60 % contribution, followed by clips production (28 %). For biomass production the biogenic carbon sequestered results in very low overall impact for this stage. The EoL is also found to have low contribution (8 %) to overall impacts with biodegradation in soil showing low impacts to environment.

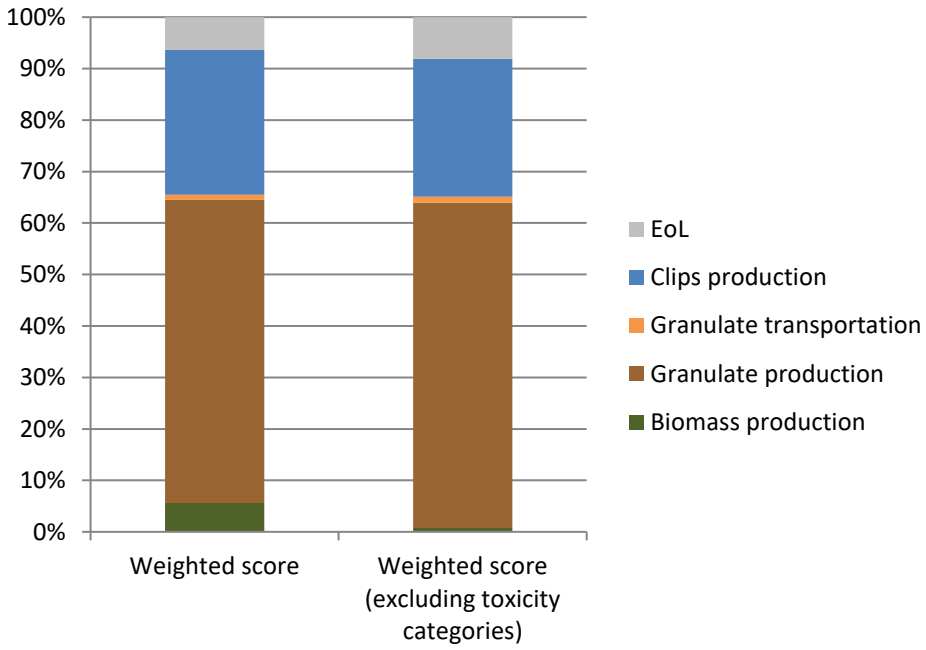


Figure 68. Contribution of the life cycle stages and processes based on normalised and weighted results of bio-based clips (baseline system), with and without toxicity categories.

3.4.2 Bio-Based alternative system (Bio-based clips from potato waste)

The cradle-to-grave LCA results from the assessment of one functional unit of bio-based clips (72 kg) of alternative reclaimed starch from potato waste-based system is given in Table 91. The breakdown of impacts between biomass production, manufacturing and EoL is shown in Figure 69. The EoL is 100 % biodegradation in soil. The manufacturing phase dominates the impacts. EoL and biomass feedstock have negligible impact in the overall cradle-to-grave results, except for climate change. The climate change impact of the EoL is due to the direct emissions from the degradation of the product on field. The climate change impact of the biomass feedstock is due to the biogenic carbon sequestered from the environment.

Table 91. Cradle-to-grave LCA results of 1 functional unit bio-based clips alternative system (excluding DLUC and ILUC effects).

Impact category	Unit	Biomass	Manufacturing	EoL	Total
Climate change	kg CO_2 eq.	-9.12E+01	2.09E+02	6.87E+01	1.87E+02
Ozone depletion	kg CFC-11 eq.	0.00E+00	1.26E-05	-1.04E-10	1.26E-05
Human toxicity, non-cancer effects	CTUh	0.00E+00	4.20E-05	2.23E-06	4.42E-05
Human toxicity, cancer effects	CTUh	0.00E+00	1.45E-05	4.96E-07	1.50E-05
Particulate matter	kg PM2.5 eq.	0.00E+00	6.97E-02	9.90E-05	6.98E-02
Ionizing radiation, human health	kBq U235 eq.	0.00E+00	4.43E+01	-4.09E-04	4.43E+01
Photochemical ozone formation	kg NMVOC eq.	0.00E+00	6.71E-01	1.51E-02	6.87E-01
Acidification	molc H+ eq.	0.00E+00	1.07E+00	1.09E-02	1.08E+00
Terrestrial eutrophication	molc N eq.	0.00E+00	2.46E+00	6.45E-02	2.52E+00
Freshwater eutrophication	kg P eq.	0.00E+00	7.26E-02	9.98E-06	7.26E-02
Marine eutrophication	kg N eq.	0.00E+00	3.01E-01	5.91E-03	3.07E-01
Freshwater ecotoxicity	CTUe	0.00E+00	1.09E+03	6.77E+00	1.10E+03
Land use	kg C deficit	0.00E+00	8.13E+02	0.00E+00	8.13E+02
Water use	m^3	0.00E+00	8.59E+01	-2.21E-01	8.57E+01
Abiotic depletion	kg Sb eq.	0.00E+00	1.42E-04	-9.00E-09	1.42E-04
Abiotic depletion (fossil fuels)	MJ	0.00E+00	3.44E+03	-3.87E+00	3.44E+03
NREU	MJ	0.00E+00	4.32E+03	-3.85E+00	4.32E+03

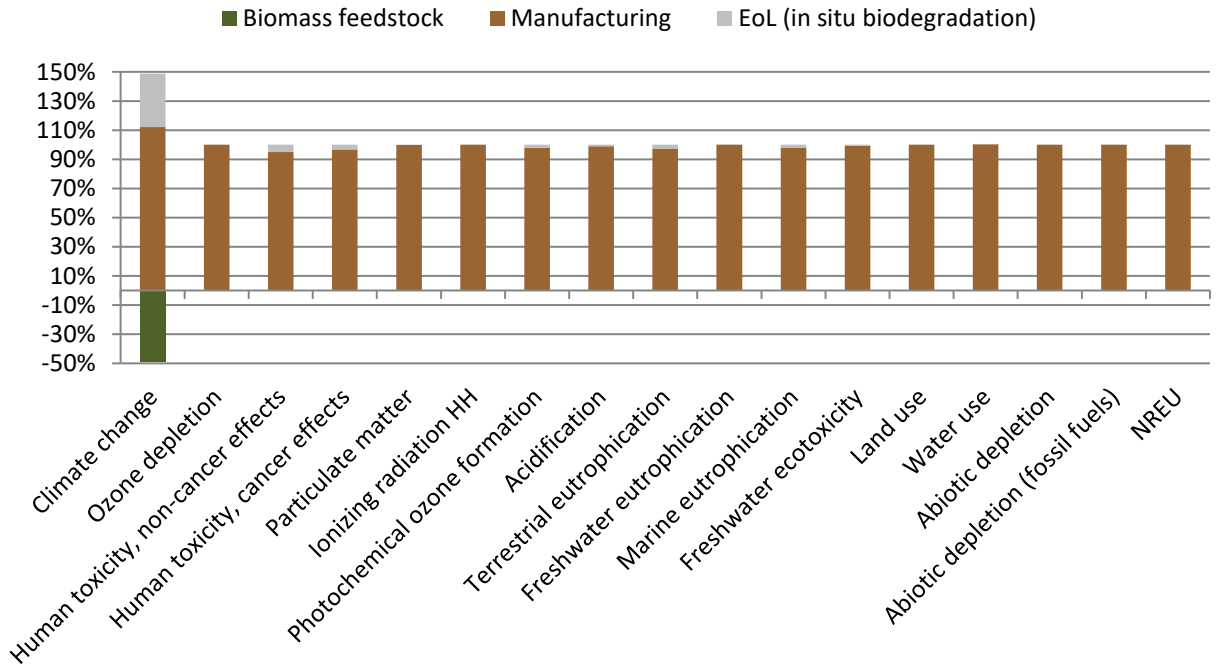


Figure 69. Breakdown of the potential impacts from the bio-based clips alternative system (reclaimed starch), cradle-to-grave results (excluding DLUC and ILUC effects).

3.4.3 Petrochemical reference system

The cradle-to-grave LCA results for one functional unit (54 kg) of petrochemical PP clips are presented in Table 92. The breakdown of impacts between manufacturing and EoL is shown in Figure 70. The estimated EoL mix for petrochemical PP clips is 56 % incineration and 44 %

landfilling (see section 2.2). It is seen that the high energy required for the modelled hypothetical gathering and sieving of clips at EoL have a very significant contribution (58-91 %) to the overall impacts of the clips.

Table 92. Cradle-to-grave LCA results of 1 functional unit petrochemical PP clips.

Impact category	Unit	Manufacturing	EoL	Total
Climate change	kg CO_2 eq.	1.31E+02	6.37E+02	7.69E+02
Particulate matter	kg PM2.5 eq.	3.46E-02	5.17E-02	8.63E-02
Photochemical ozone formation	kg NMVOC eq.	3.86E-01	2.39E+00	2.77E+00
Acidification	molc H+ eq.	5.09E-01	2.26E+00	2.76E+00
Terrestrial eutrophication	molc N eq.	9.84E-01	1.03E+01	1.13E+01
Abiotic depletion (fossil fuels)	MJ	4.49E+03	6.97E+03	1.15E+04
Non-renewable energy use (NREU)	MJ	4.93E+03	6.87E+03	1.18E+04

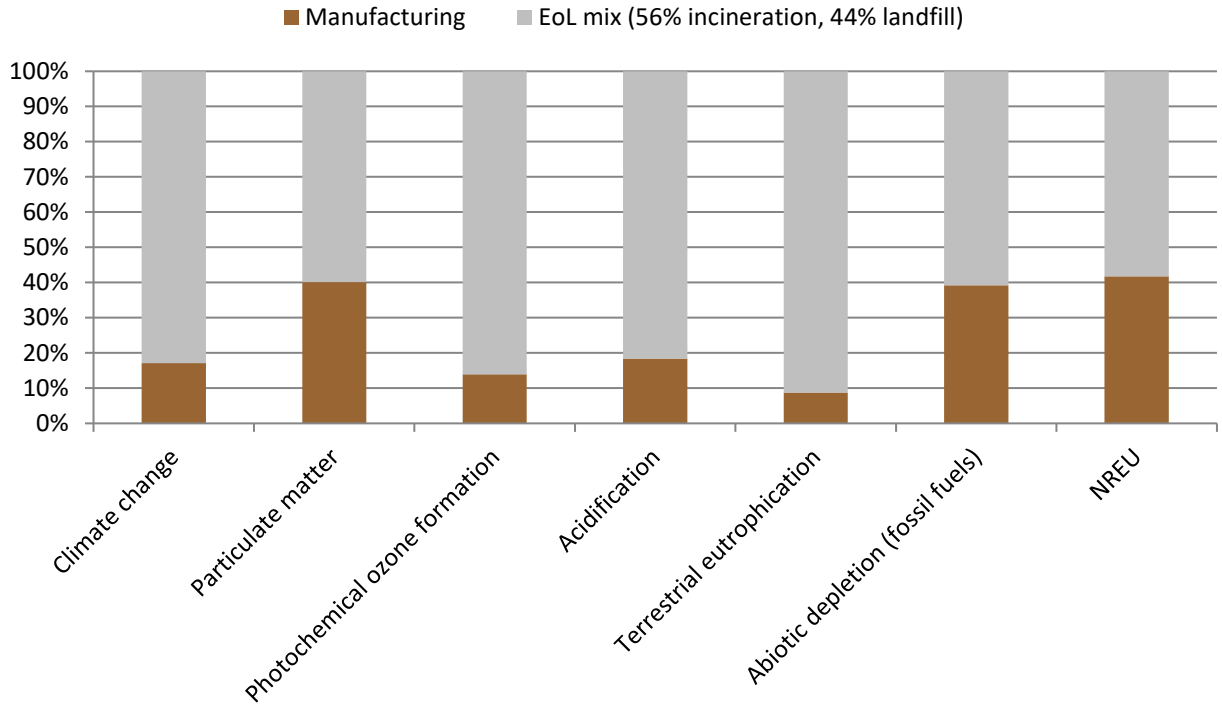
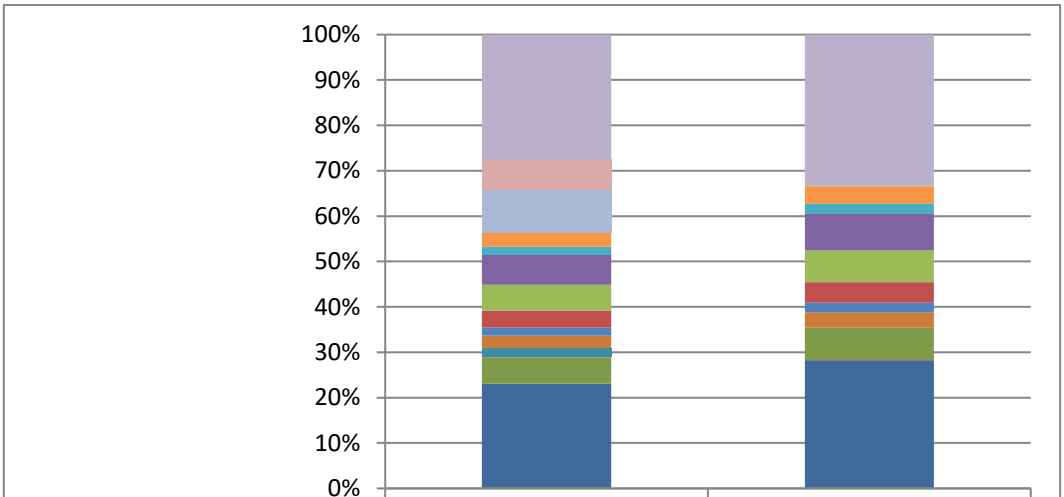


Figure 70. Breakdown of the potential impacts from the petrochemical PP clips for selected impact categories, cradle-to-grave results.

Normalisation and weighting have been applied to the reference system (petrochemical PP clips) to understand which among the assessed impact categories are more relevant for this application and what the main environmental hotspots are. To do this, all the impact categories need to be taken into the calculation. Hence, the weighted results are subject to considerable uncertainties, as highlighted in *Chapter 6 Ranges for environmental impacts from production of fossil-based plastics*, therefore their numerical values must be considered only as indicative.

Based on the normalised and weighted results shown in Figure 71, excluding toxicity-related impacts, the hotspot impact categories of petrochemical PP clips are: climate change (33 %), abiotic depletion (fossil fuels) (28 %), photochemical ozone formation (8 %), acidification (7 %) and water use (7 %) forming a cumulative 83 % contribution to total impacts. The rest of the impact categories are less relevant with each contributing up to 5 %. Note that the top four most relevant categories are among the suitable impact categories identified in Chapter 6.

Contributions of the life cycle stages and the different processes within them are given in Figure 71. The EoL is the most relevant life cycle stage, contributing to 74 % of the impacts due to the very high energy required for collecting and sieving the petrochemical clips. The manufacturing stage contributes a cumulative 25 % to the impacts, with 15 % of the contribution coming from granulate production and 10 % from clips production.



	Weighted score	Weighted score (excluding toxicity categories)
Climate change	28%	33%
Ozone depletion	0%	0%
Human toxicity, non-cancer effects	7%	0%
Human toxicity, cancer effects	9%	0%
Particulate matter	3%	4%
Ionizing radiation HH	2%	2%
Photochemical ozone formation	7%	8%
Acidification	6%	7%
Terrestrial eutrophication	4%	5%
Freshwater eutrophication	2%	2%
Marine eutrophication	3%	3%
Freshwater ecotoxicity	2%	0%
Land use	0%	0%
Water use	6%	7%
Abiotic depletion	0%	0%
Abiotic depletion (fossil fuels)	23%	28%

Figure 71. Contribution of normalised and weighted results of petrochemical PP clips with and without toxicity categories.

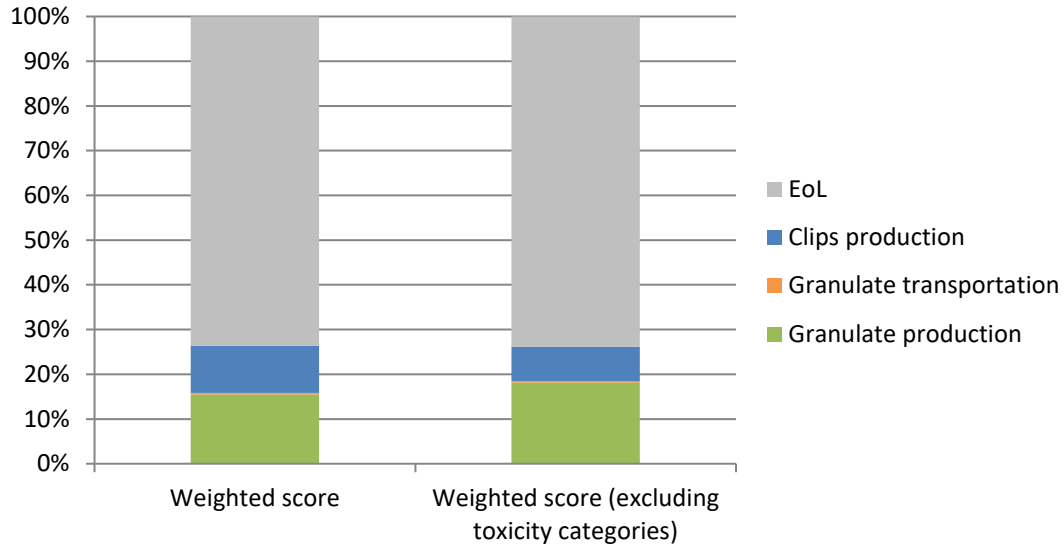


Figure 72. Contribution of the life cycle stages and processes based on normalised and weighted results of petrochemical PP clips, with and without toxicity categories.

3.4.4 Comparing the Bio-Based baseline system with bio-based alternative and petrochemical reference

3.4.4.1 Comparing the Bio-Based baseline system with bio-based alternative (from potato waste)

The comparison of cradle-to-grave results of the bio-based clips baseline system with the alternative system is given in Figure 73. The alternative system performs significantly better (>40 %) in three categories: ozone depletion, human toxicity (non-cancer effects) and abiotic depletion. It performs slightly better (up to 25 %) than the baseline in the particulate matter, acidification, marine eutrophication and water use impact categories. These differences are mainly because there is no biomass production in the alternative system, since reclaimed starch is used instead of virgin starch produced from biomass. Therefore, there is no requirement for fertiliser, pesticides and water use for biomass production. There is however, higher impact for land use in the alternative system. This is because in potato starch-based production there is a compounding of starch with 43 % bio-based PLA and use of other bio-based additives. This results in higher land use than maize starch-based production, where mainly fossil-based copolyester and a lower amount of bio-based copolyester is used. Maize starch based plastic forms most of the share in the aggregated results for the baseline system. Therefore, in comparison with the alternative system based on potato starch, lower land use is seen in the baseline system. For the impact categories of climate change, abiotic depletion (fossil fuels) and NREU, similar results are attained.

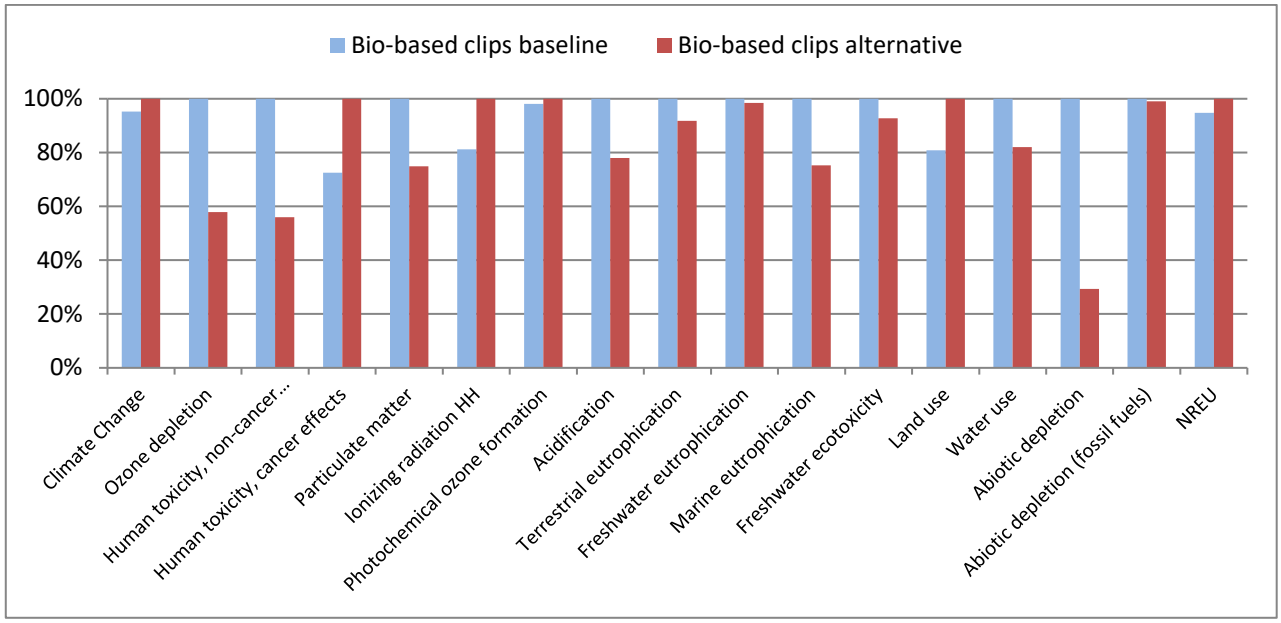


Figure 73. Comparing bio-based clips baseline (native maize and potato starch based) system with bio-based alternative (reclaimed starch from potato waste based) system, cradle-to-grave results (excluding DLUC and ILUC effects).

3.4.4.2 Comparing the Bio-Based baseline system with petrochemical reference

Figure 74 shows the cradle-to-grave comparison of bio-based (baseline) (72 kg/FU) and petrochemical clips (54 kg/FU). For bio-based clips, the EoL is biodegradation in soil which is both the real and intended EoL scenario. For petrochemical PP clips, the used clips are assumed to be collected and sent to landfill (44 %) and incineration (56 %). The bio-based clips perform significantly better (about 65-77 %) than the petrochemical counterparts in five out of seven suitable indicators (climate change, photochemical ozone formation, terrestrial eutrophication, abiotic depletion (fossil fuels) and NREU). For acidification, bio-based clips offer 50 % less impacts, and for particulate matter similar results are observed. EoL plays an important role in such a comparison due to the high impact from the hypothetical collection and waste management scenario of the petrochemical clips (see the break down of results in Section 3.4.3).

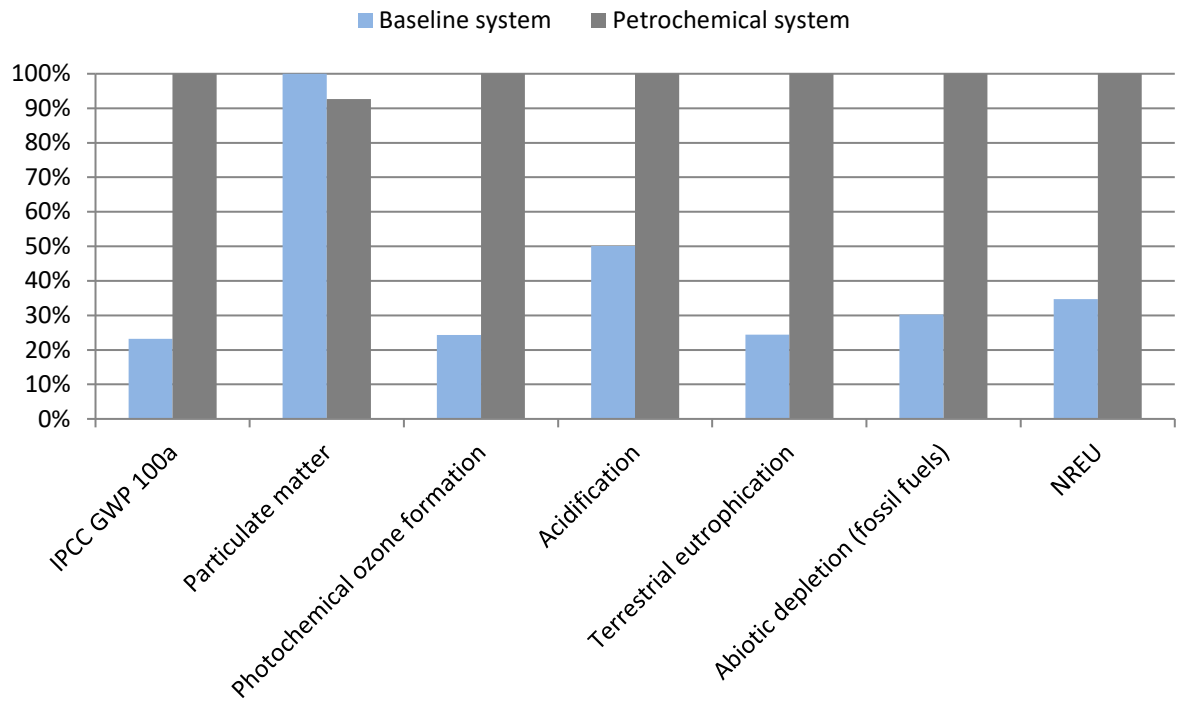


Figure 74. Comparing bio-based clips with petrochemical clips, cradle-to-grave results for selected indicators (see text) (excluding DLUC and ILUC effects).

4 Discussion

4.1 End of life sensitivity analysis

In this sensitivity analysis, a closer look is taken at the intended waste management technology for the bio-based product system. For this field application product, in-situ biodegradation was chosen as the intended technology. The main parameter varied is the degradation of carbon in the soil, which is dependent on the soil type. Lower storage of carbon in soil and higher emissions to air are observed in coarse sandy soil (EASETECH) compared to the modelled sandy loam soil. The variation in parameters is presented in Table 93. EASETECH furthermore assumes a distribution for heavy clay soil of 89.14 % CO_2 emission to air and 10.86 % C soil storage. As mentioned earlier, these figures are based on simulations in Daisy, a Danish deterministic agro-ecosystem model (Hansen et al., 2012)

Table 93. Variation in input parameters in the sensitivity analysis.

Product system	Technology	Parameter changed	Current	Higher	References
Starch	In-situ	Division between C storage and CO_2 emission to air from the degraded C	11.31 / 88.68 (sandy loam soil)	9.88 / 90.11 (coarse sandy soil)	EASETECH

The only impact categories affected by change in this distribution are the climate change categories. Figure 75 presents the sensitivity analysis where both original in-situ EoL treatment and the higher limit are presented in terms of climate change. The total climate change impact has a minimal increase of 3 % compared to the current model. Thus, even though there could be soil types in Europe with a higher or lower split of carbon distribution between soil and air, this will not change the overall conclusion when comparing the EoL technologies.

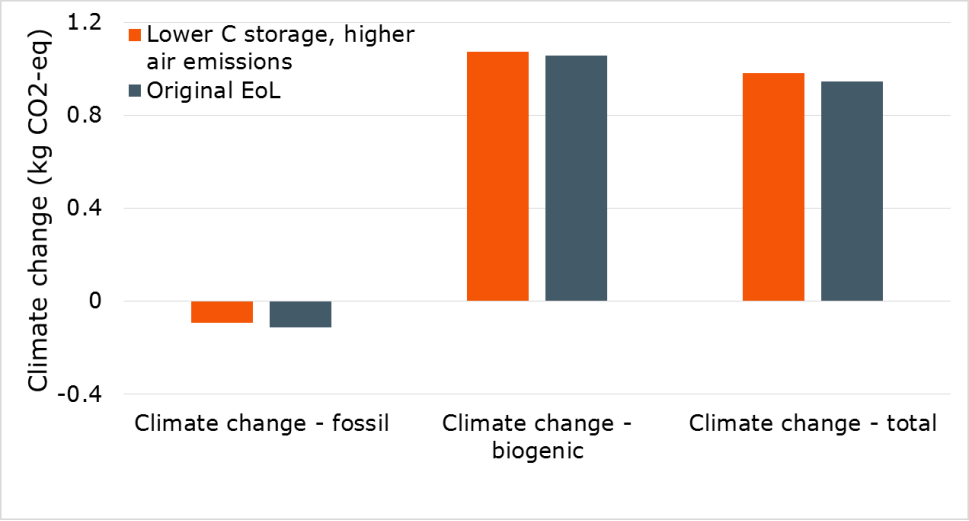


Figure 75. Sensitivity analysis for EoL single use clips.

This sensitivity analysis shows that changing the soil type would change the cradle-to-grave climate change results for the starch-based clips by less than 2 % for both a lower and a higher carbon degradation.

4.2 Discussion on End of Life uncertainties

The parameters analysed in the sensitivity analysis above and other parameters, which may be uncertain, i.e., estimations, assumptions, huge ranges in industry or EU member states, missing data, etc. are discussed in this section. Methodological shortcomings and additional research requirements are also presented.

In-situ field-based biodegradation is based on a Danish soil type (sandy loam soil), but the sensitivity analysis above shows that this parameter is not sensitive to the overall cradle-to-grave results (less than 2 % change and only in the climate change category, see Figure 75).

The modelling does not take other impacts e.g., microplastics in the soil into account. Temperature and soil moisture will also affect biodegradation, which have not been included in this investigation. Further research in these matters is relevant.

Energy consumption to gather and sieve the clips plus treatment of soil - The amount of energy used for gathering and sieving of clips is an uncertain factor that has a large influence on GWP for all EoL technologies other than in-situ. Literature values that are not specifically from gathering or sieving of clips are applied. This is because incineration and landfilling of the clips are most probably hypothetical scenarios, as the small clips are likely not collected. The values will differ based on the machinery used for gathering and machinery used for sieving.

Quantity of soil attached to the clips - The treatment of 3 kg of soil is based on values of contamination attached to mulch films from Plasticulture (2018). There are currently no values reported for contamination attached to clips used in agricultural practices. The quantity of contamination (soil), which needs to be gathered and sieved, comes with an uncertainty. The environmental impact for EoL technologies other than in-situ will likely be sensitive to this as the treatment of extra soil represents a large share of the overall environmental impact (see Annex 4 for contribution analysis). The quantity of soil attached to the clips when entering the EoL technology will depend on the collection and sieving treatment and could likely be lower than for mulch films, as the soil is more susceptible to falling off a small item such as a clip compared to a plastic film.

Littering is not modelled in this LCA, but is especially important for this product, as the clips are prone to being littered, as gathering of clips once they are spread in a field requires significant resources. Littering is further discussed in the Approach and Methodology chapter in section 3.1.4. Plastic left on a field will be more prone to becoming a littering problem than plastic treated within the waste management system.

Apart from the above case study's specific uncertainties, several other general uncertainties that apply to most of the presented case studies are discussed in the Approach and methodology in chapter 3.5., e.g., data quality, critical assumptions and limitations in End of Life modelling. These include uncertainties related to:

- the estimated EoL mix
- the marginal energy technologies utilised
- the development of the EoL technologies (efficiency increase)
- the consumption of materials and energy at the EoL treatment plants
- biodegradation in landfills
- the chemical composition of the biodegradable plastics.

The uncertainties that are most influential to the results are expected to be the assumptions on gathering and sieving of the soil from the clips, and the soil attached to the clips, as well as biodegradability and the rate of biodegradation of the starch clips in a landfill. The exclusion of

littering from this case study due to an inadequate methodology is a major sensitivity and means that the EoL mix assumed for the petrochemical clips is unrealistic. The biggest emphasis for further research needs to be focused on littering, but also biodegradation rates of biodegradable plastics in non-intended EoL technologies is limited and should also be addressed.

5 Assessing non-PEF impact categories: taking into account ILUC and endpoint of GTP

5.1 Global Temperature Change Potential

The GTP impacts of bio-based clips breakdown into the main unit processes described in section 2.1. The inventory analysis is shown in Figure 76. When the biogenic carbon removal is considered, the cradle-to-gate GTP 100a is 94.7 kg CO_2 eq./FU. Observations for GTP are very similar to those for the climate change impacts described in section 3.1.1.

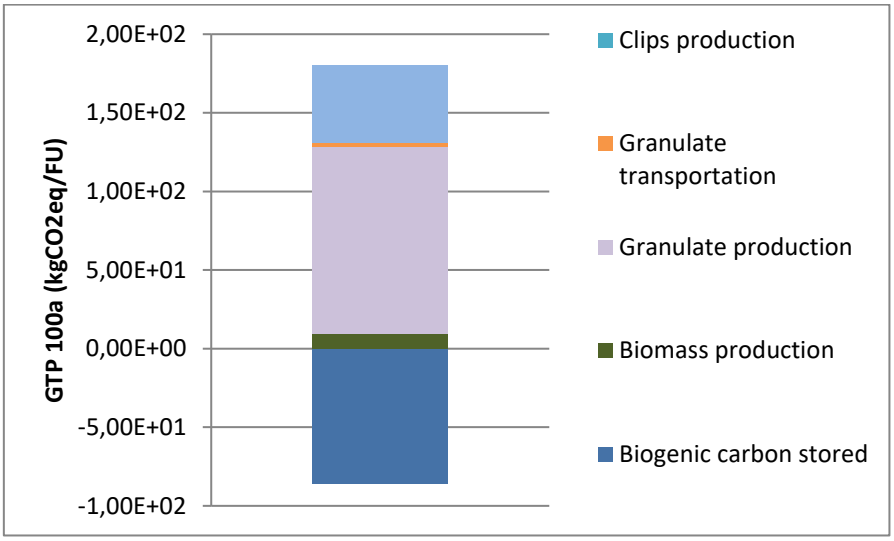


Figure 76. Breakdown of the cradle-to-gate GTP 100a results for bio-based clips, 1 functional unit, excluding DLUC and ILUC effects.

5.2 Land Use Change emissions

Table 94 presents the cradle-to-grave characterised results broken down into feedstock (for potato used for starch, maize used for starch and maize used for PLA; with biogenic carbon stored separately), manufacturing, EoL and ILUC (for potato used for starch; maize used for starch and maize used for PLA). As highlighted in Chapter 4, DLUC and ILUC cannot be summed up (it is whether DLUC or ILUC). This case study does not involve crops that imply DLUC (namely Brazilian sugarcane, German maize, French wheat and German wheat), based on the methodology from the PEFCR Guidance V6.3 presented in Chapter 4. Figure 77 presents the characterised results for case study 2 with the inclusion of ILUC.

Table 94. Characterised cradle-to-grave results broken down per process, including ILUC. Case study 2, baseline.

	Biogenic carbon removals	iLUC (potato for starch)	iLUC (maize IT for starch)	iLUC (maize for PLA)	Potato production (for starch)	Maize IT production (for starch)	Maize production (for PLA, US)	Manufacturing	EoL
Climate change (kg CO_2 e)	-8.60E+01	2.29E+00	7.29E+00	1.23E+01	1.64E+00	8.27E+00	5.09E+00	1.80E+02	6.87E+01
Ozone depletion (kg CFC-11 e)		4.78E-10	1.52E-09	2.56E-09	2.04E-08	6.62E-07	1.83E-10	2.11E-05	-1.04E-10
Human toxicity, non-cancer effects (CTUh)		1.04E-09	3.32E-09	2.05E-09	1.20E-05	2.28E-05	1.32E-07	4.20E-05	2.23E-06
Human toxicity, cancer effects (CTUh)		9.90E-11	3.15E-10	3.27E-10	9.74E-08	4.09E-07	5.90E-08	9.82E-06	4.96E-07
Particulate matter (kg PM2.5 e)		6.04E-05	1.92E-04	3.24E-04	1.11E-03	5.17E-03	1.85E-03	8.49E-02	9.90E-05
Ionizing radiation HH (kBq U235 e)		-4.17E-05	-1.33E-04	1.76E-04	2.49E-02	1.72E-01	5.24E-01	3.53E+01	-4.09E-04
Photochemical ozone formation (kg NMVOC e)		7.12E-03	2.27E-02	3.81E-02	1.30E-03	3.43E-02	1.01E-01	5.22E-01	1.51E-02
Acidification (molc H+ e)		1.43E-03	4.54E-03	7.65E-03	4.81E-02	1.10E-01	7.64E-02	1.14E+00	1.09E-02
Terrestrial eutrophication (molc N e)		8.00E-03	2.55E-02	4.29E-02	2.12E-01	4.66E-01	4.13E-01	1.59E+00	6.45E-02
Freshwater eutrophication (kg P e)		5.27E-06	1.68E-05	2.83E-05	1.18E-03	1.76E-03	2.56E-04	7.06E-02	9.98E-06
Marine eutrophication (kg N e)		8.50E-04	2.70E-03	4.55E-03	4.47E-02	1.40E-01	4.34E-02	1.74E-01	5.91E-03
Freshwater ecotoxicity (CTUe)		1.76E-02	5.61E-02	2.28E-02	2.68E+01	3.54E+01	9.73E+01	1.02E+03	6.77E+00
Land use (kg C deficit)		6.58E-03	2.09E-02	3.53E-02	5.93E+01	1.82E+02	3.37E+02	7.92E+01	0.00E+00
Water use (m^3)		7.05E-02	2.24E-01	3.78E-01	3.46E-01	6.75E+01	1.47E+01	2.21E+01	-2.21E-01
Abiotic depletion (kg Sb e)		2.08E-08	6.63E-08	1.12E-07	1.47E-07	6.01E-05	8.66E-06	4.16E-04	-9.00E-09
Abiotic depletion (fossil fuels) (MJ)		5.95E-02	1.89E-01	3.19E-01	8.02E+00	6.70E+01	5.97E+01	3.34E+03	-3.87E+00
NREU (MJ)		5.69E-02	1.81E-01	3.05E-01	8.74E+00	6.96E+01	6.25E+01	3.96E+03	-3.85E+00

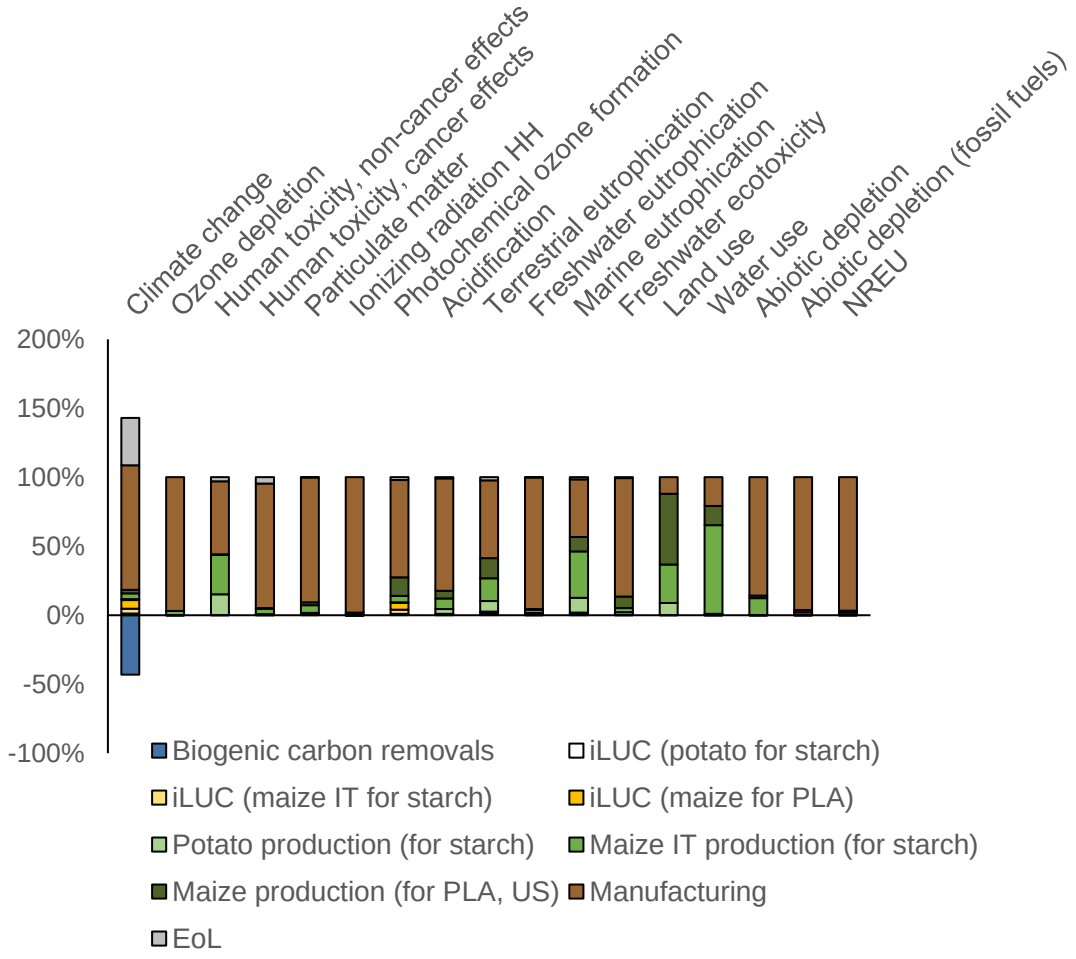


Figure 77. Relative characterised results broken down for all impact categories and including ILUC – case study 2, baseline.

As can be seen from Table 94 and to some extent from Figure 77, ILUC does contribute to all impact categories considered herein, as in case 1. However, as for case 1, the contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (11 % of the impact), photochemical ozone formation (9 % of the impact), terrestrial eutrophication (3 % of the impact) and marine eutrophication (1.7 % of the impact). All these impacts are dominated by land expansion (Figure 15). In the case of climate change, as explained earlier, the ILUC impact is essentially due to the CO_2 release resulting from land clearing, and in the case of photochemical ozone formation, to carbon monoxide also released during land clearing (burning the cleared biomass; Figure 17). Both acidification, terrestrial- and marine eutrophication are essentially due to nitrogen oxides released during land clearing (more than 80 % of the impact; Figure 17).

The contribution of ILUC to climate change has a magnitude slightly higher than the impact from producing the crops themselves (Figure 77; crop cultivation represents 8 % of the climate change impact when ILUC is included). ILUC is here shown in white to yellowish tones for all crops contributing to it (for the climate change impact, ILUC potato starch represents 1 %, ILUC maize starch 4 % and ILUC maize PLA 6 %). As it could be expected, these relative proportions among crops are similar for the impact "land use" (this magnitude is essentially a

function of the yield and the amount demanded per functional unit). Similarly, all crop cultivation processes are shown separately (greenish tones).

Although potato has a lower yield than maize, its ILUC impact is lower than for maize as less is used per functional unit. In absolute terms, the ILUC impact for climate change here is quite high compared to the one obtained for e.g. case 1 (Table 76 vs Table 94; this is also true compared to most other cases but case 5). This reflects the importance of having conversion processes requiring as little biomass as possible, so the climate change impacts from both the crop cultivation and the ILUC can be lower.

6 Conclusions, limitations and recommendations

The goal of this case study was to assess the environmental profile of bio-based clips production and to compare it with the fossil-based counterpart. For the EoL of petrochemical PP clips, gathering of clips, sieving and then disposal by landfilling (44 %) and incineration (56 %) is modelled. It should be noted that this is a hypothetical scenario and currently the petrochemical clips are left on field as litter. However, littering cannot be modelled adequately in LCA (see also the methodology chapter and the final conclusions chapter of this report). For starch-based bio-based clips, the intended and current EoL mix are both in-situ biodegradation in soil. Therefore, for the bio-based clips, there is no requirement for gathering the clips and sieving the soil which results in a major difference in the EoL results between bio-based and petrochemical clips (see section 3.4). The weighted results of each product system with EoL mixes show that the bio-based starch clips are preferred over petrochemical PP clips. The sum of the weighted impacts with toxicity for EoL in-situ of bio-based starch is 0.2, while for petrochemical PP with EoL mix incineration and landfill is 4.7. Furthermore, when the petrochemical clips are collected, the assumed removal of the soil together with the clips causes depletion of the soil's organic carbon, which adds to the EoL impacts of petrochemical clips.

An alternative bio-based system was considered where starch is reclaimed from potato waste instead of producing it from biomass (maize/potato). The difference between the potato starch-based system is that there are no biomass production impacts enabling slightly better (up to 25 % impact reduction) performance for the particulate matter, acidification, marine eutrophication and water use impact categories.

The contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (11 % of the impact), photochemical ozone formation (9 % of the impact), terrestrial eutrophication (3 % of the impact) and marine eutrophication (1.7 % of the impact). All these impacts are dominated by land expansion. In the case of climate change, the ILUC impact is essentially due to the CO_2 releases resulting from land clearing and in the case of photochemical ozone formation, to carbon monoxide also released during land clearing (burning the cleared biomass). Both acidification, terrestrial- and marine eutrophication are essentially due to nitrogen oxides released during land clearing (more than 80 % of the impact).

Looking at the comparison of the bio-based clips (baseline) with petrochemical PP clips from cradle-to-gate, bio-based clips were found to perform significantly worse than the petrochemical counterpart apart from the climate change, abiotic depletion (fossil fuels) and NREU categories where it provides slightly better results. However, based on the comparison of cradle-to-grave results per functional unit, bio-based clips perform better in all suitable impact categories. It performs significantly better (up to 77 %) than the petrochemical counterpart in climate change, photochemical ozone formation, acidification, terrestrial eutrophication, abiotic depletion (fossil fuels) and NREU. For particulate matter, it performs similarly to the petrochemical reference.

Littering is excluded from this case study due to lack of data and knowledge of the impact assessment models. The exact amount (and relative share) of plastics that are littered is uncertain for a specific application and will vary between different EU countries. The impacts of littering and microplastics are currently not properly captured in the toxicity-related impact categories (ecotoxicity and human toxicity). As pointed out by Pant et al. (2018) fate, exposure, and effect modelling for macro- and micro-plastics (including additives) is still in its infancy. Physical impacts (e.g. entanglement; ingestion of larger plastic particles and their effects) and chemical impacts (e.g. due to microplastic formation, microplastics as carrier of other chemicals, unknown impact of additives) and biological impacts (e.g. microplastics as carrier of germs/alien species) are largely unknown and currently not included in LCA. Thus,

the impacts of littering (i.e. leaving plastics in the soil or in a marine environment) are most likely underestimated in the current LCA for the non-degradable applications.

Ultimately, the negative cradle-to-grave performance is determined by the assumption that the petrochemical clips would be removed (including a significant amount of soil). It is questionable whether this EoL scenario is ultimately more desirable than leaving the fossil clips in the soil, but the long-term impacts of littering are also not well understood. Given the excellent in-situ EoL performance and relatively limited share of the cradle-to-gate impacts of the production of the bio-based clip compared to the overall impacts, the bio-based clips appear to be the overall preferred option.

CASE STUDY 3: SINGLE-USE CUPS FOR COLD DRINKS

1 Goal and scope definition

1.1 Goal and background

In this case study, the goal is to assess the potential environmental impacts of bio-based and petrochemical single-use drinking cups. The analysis is focused on cups suitable for cold beverages sold, used and disposed of in Europe. Among the most used types of cups currently on the market, PP, PET and PS cups can be considered the benchmark. Some bio-based alternative materials have been recently introduced for this application in the market. PLA is the most important bio-based material for single-use cups and it is the focus of the investigation of this chapter. A second bio-based material included in the analysis is bio-based PP which is one of the potential available alternatives.[50] Both the considered bio-based materials are 100 % bio-based but only PLA is biodegradable. There are many research routes leading to bio-based PP. The bio-based PP analysed in this case study is derived from used cooking oil; the polymer is identical with petrochemical PP and is not biodegradable.

1.1.1 Polylactide

PLA has emerged as an important bio-based plastic in the past decades. Among its technical characteristics, its biodegradability is an interesting property that has attracted much attention in developing single-use packaging products.

The technological assessment of the PLA production process corresponds to a TRL (technology readiness level) of 9 (successfully operating technology). This fits the scope of the study i.e. focusing on commercialised bio-based products (see the chapter named *Introduction, goal and scope*). Figure 78 shows the chemical structure of PLA. PLA is a 100 % bio-based polymer produced through polymerisation of monomer lactic acid or lactide. Through ring opening polymerisation of lactide, a ring structure formed by self-esterification of two lactic acid molecules is obtained. Monomer lactic acid or lactide can be obtained by fermentation of sugars. Commercially, the L-lactide route, which leads to Poly-L-lactide (PLLA) is the dominant route. The alternative route - the D-lactide- is used to deliver better thermal performance (PDLA).

[50] Neste's NExBTL technology triggered the production of first bio-based polypropylene in 2018. Neste has stated its strategy is to focus on develop, together with its partners, durable plastic applications. Therefore, the example of single-use cups for cold drinks is a theoretical example to showcase a bio-based product alternative derived from used cooking oil.

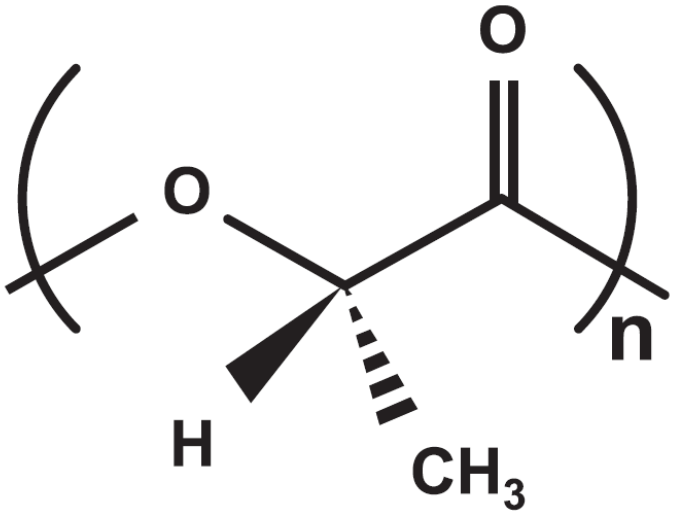


Figure 78. Molecular structure of PLA (Vink et al., 2015).

In 2018, industrial scale production of PLA (lactic acid) was offered by two key players: NatureWorks LLC, which has an annual capacity of 150 ktons in Nebraska, USA (Vink et al. 2015), and Corbion Purac/Total-Corbion PLA, which has a capacity of 75 ktons of lactide (and PLA from the second half of 2018) in Rayong, Thailand (Total-Corbion, 2018). The production capacity of PLA has increased by nearly 300 % compared to 2003 (based on data from 2003 reported by Shen et al. (Shen et al., 2010), representing one of the most rapidly growing bio-based plastics on the market. It is projected that PLA production will increase by more than 50 % by 2021 compared to 2017 (EuropeanBioplastics, 2017). The attention is on PLA available on the current and near future European market. Moreover, a possible future PLA produced in Europe from European maize is also assessed to estimate the potential environmental benefits deriving from the reduction of the transportation distance and possible disadvantages caused by the change of location of the biomass cultivation.

1.1.2 Bio-based polypropylene from used cooking oil (UCO)

Bio-based PP has the same chemical structure and characteristics as petrochemical PP, but it is 100 % bio-based. Due to its chemical structure, bio-based PP belongs to the group of bio-based plastics which are not biodegradable.

There are many biochemical and thermochemical routes under investigation to derive PP from bio-based feedstocks. These routes are often linked with bio-diesel, renewable-diesel and ethanol production technologies. Currently, three routes are being investigated by the industry, each using different feedstocks and applying different conversion technologies.

- **From sugar or starch crops e.g. sugarcane, sugar beet, maize or wheat.** Through biochemical sugar fermentation ethanol can be produced. Ethanol becomes the intermediate product from which ethylene can be derived. The process that causes this conversion is dehydration, which together with dimerization allows the production of ethylene and butane-1. These two chemicals can be converted into propylene through metathesis. Braskem and Novozymes are developing a large-scale production of bio-based PP from sugarcane (Niaounakis, 2015).

- **From biomass gasification** (solid biomass eventually torrefied e.g. agricultural wastes). Gasification produces syngas that is treated in a dehydration process (of propanol/butanol) or that feed a methanol synthesis unit. In the first case through

dehydration and metathesis propylene is produced similarly to the previous route. In the second case the methanol is converted to dimethyl-ether (Gay et al., 2011).

- **From vegetable oil.** Bio-based naphtha can be derived from used cooking oil (or vegetable oil) through hydro-treatment. Propylene can be obtained from steam cracking. Neste with its partners is developing this production on an industrial scale (Neste, 2018).

In this project, it is possible to analyse the third route, bio-based -PP derived from vegetable used cooking oil) which has industrial support. The commercial scale system pilot is at demonstration stage therefore the TRL can be assessed as near to 8.[51]

1.1.3 Petrochemical reference: PP and PET

Today, most single-use cups are made from polypropylene (PP), polystyrene (PS) and polyethylene terephthalate (PET). In this study petrochemical PP and PET cups are selected as the references because both are widely used to contain cold beverages, which is the main application of PLA cups. PS cups are used more to contain hot beverages. A comparison of the environmental impact of PP and PET with bio-based plastics will be performed for the recommended impact categories identified for these materials (see the section Investigation of generic LCA data for fossil-based plastics).

1.2 Scope

Geographical scope: the final product is purchased and disposed of in Europe while the manufacturing phases and the supply chain could extend globally (see chapter *Introduction, goal and scope*). Today, PLA is produced at commercial scale by two producers:

- NatureWorks LCC, PLA granulates made from maize in Nebraska, USA.
- Total-Corbion PLA, PLA granulates made from sugarcane in Thailand.

The weighted average (based on production capacities) of the two production routes are taken as the default baseline for the bio-based PLA cups. See further detailed description in the "Product Systems" and "Life cycle inventory analyses" sections of this chapter.

Temporal scope: current production (2017-2018) with relevant developments foreseen for the short-time future (5-10 years) (see Chapter 2 "Approach and methodology" of the report).

Technological scope: PLA production of the commercialised technologies (level 9 TRL) is assessed using company-specific data. For bio-based PP made from used cooking oil, the TRL is around 8 –(validated prototype). The technology has not been fully commercialised. Primary data are obtained from Neste.

1.3 Function and functional unit

The primary function of single-use cups is to contain liquids. The functional unit is defined as:

- 1,000 single-use drinking cups used for cold beverages, each with a volume of 200 ml

[51] It should be noted that the research and development of bio-based PP made from used cooking oil by Neste focuses on durable application rather than single-use application (Personal communication with Neste, 2018). We nevertheless include this material because the concept of utilisation of waste is of particular interest.

The term 'cold' beverage means a liquid with a temperature ranging from 0°C to 20°C. For cold beverage cups, PLLA is sufficient to fulfil the requested thermal and mechanical properties (see Table 95 showing the main mechanical and thermal properties of the investigated materials).

Among different sizes of the cups, 200-ml has been selected since it is a very common standard for this application based on our literature review.

The analysed volume of 200 ml corresponds to a typical height of the cups ranging from 9-10 cm (Garrido & Alvarez des Castillo, 2007). Based on the information obtained from a small market survey and interviewing some producers, cups of fixed volume, height and offering similar primary function (i.e. all for serving drinks at events and thus with enough stiffness) have different weights: between 2.9-3.5 g for PP, 4.1-4.7 g for PLA and 5.5- 6.4 g for PET.

Fixing the diameter to 75 mm (which represents a common dimension for the selected 200 ml capacity for all the investigated materials), the following weight for each cup has been defined: 3.88 g for PP (both petrochemical and bio-based), 4.59 g for PLA and 5.51 g for PET. The selection of these weights is based on the following considerations:

- The study aimes to compare different products that offer the same primary functionality for the consumer: for a drinking cup, that means the comparison must be based on the same size/content (fixed volume, height and diameter) and same strength/stiffness (these two properties vary with the thickness of the cups).

- The three materials have different mechanical and thermal properties that have to be taken into account in the comparison (Table 95). For petrochemical plastics the reported values are average values for thermoforming grades available on the market while for PLA they refer specifically to the Luminy® LX175 grade for thermoforming (Corbion, 2016). For the single-use cup application, stiffness (measured in Young's modulus) is the primary desired property for design.[52] Other properties like strength and barrier properties (i.e. oxygen permeation coefficient and water vapour coefficient) are not considered as the primary property to fulfil the function of this application.

- Based on the stiffness-constraint design, it is possible to determine the minimal weights of PET and PP cups based on the weight of the PLA cup, which is 4.59 g based on the market survey for 75 mm diameter cups. The corresponding weight for the PP and PET cups with the same stiffness can be derived by applying the so-called material substitution factor (MSF) developed by Ashby in 1999.

- MSFs are derived by comparing material indices (MI) of candidate materials to those of the reference material (see Equation 4). MIs were introduced by Ashby (1999), who developed a generic approach to estimate a material's performance in a particular mechanical function. This performance is expressed as a mathematical function of intrinsic material properties called an MI. For example, to minimise the mass of a beam with specified stiffness, length and height (with width as a free variable), the MI E/ρ should be maximised, where E is Young's modulus and ρ is density. The material with the highest MI value can perform the function at the lowest mass (by adapting the width of the beam used in this example). Thus: the higher the MI value, the better the material. The ratio of the MIs of a material and the reference material yields an MSF, i.e. the difference in the mass needed to perform identically in a given function

[52] When using a cup, we do not want the packaging to be too elastic but rather, it should keep the form of the packaging while the cup is held. This property of "less-easy-to-deform" is expressed in material science as 'stiffness'.

(Equation 4). Substitution factors exceeding 1.0 indicate that the functionally-equivalent product would be heavier than the reference.

$$MSF = \frac{m}{m_{ref}} = \frac{MI_{ref}}{MI}$$ Equation 4

Where:

MSF= material substitution factor of a material compared to the reference (ref), dimensionless

m = mass (for a given function), kg

MI = material index (for a given function)

The MI for stiffness-limited panels is suggested by Ashby (1999), shown in Equation 5:

Stiffness-constrained design: $MI_{stiffness} = \frac{E^{1/3}}{\rho}$ Equation 5

Where E is the flexural modulus (Young's modulus) in Giga-Pascal (GPa), ρ is the density in m^3/kg. By fixing the weight of PLA cup to 4.59 grams (based on the market survey), the weights of PET cup and PP cup can be calculated based on the stiffness-constrained design by applying both Equation 4 and Equation 5: 5.51 grams and 3.88 grams, respectively. PET is the heaviest option because it has a high density and relatively low Young's modulus compared to PLA and PP. PP is the lightest option; although its Young's modulus is not as high as that of PLA, the MI of PP is still the highest due to the low density of the material.

Table 95. Comparison of the selected properties of the analysed materials for single-use cups.

Material (grams per cup)	Weight per funct. unit (kg)	Glass trans. T (°C)	Melt-ing T (°C)	Young's Modulus (MPa) ISO 527	Tensile strength (MPa) ISO 527	Bio-C (%)	Density (g/cm^3)	Industrial compostability
PLA [1]	4.59[a]	55-60	190	3500	45	100	1.2	yes
PET [2]	5.51[b]	20-30	255	2200	55	0	1.3	no
PP [3]	3.88[b]	30-50	170	1800	40	0	0.9	no
Bio-based PP [3]	3.88[b]	30-50	170	1800	40	100	0.9	No

a weight based on market survey.
b weight calculated based on stiffness constrained design based on Material Indices developed by Ashby (1992). See further explaination in text.
Sources: [1] Luminy® LX175 grade for thermoforming (Corbion, 2016); [2] (Throne, 2008); [3] (MOLGROUP, 2017), (NATPET, 2018).

It can be noticed that not all the selected weights fall in the middle of the ranges observed by the market survey. In particular, the selected weight for PP cups falls outside the range. This can be a consequence of two reasons: the limited sample analysed in the market survey and the fact that the ovens, molds and trim used for PLA cups are not optimally designed to handle PLA but adapted to its manufacturing (Jamshidian et al., 2010). Specifically, PLA is thermoformed using ovens, molds and trims often adapted for PET or PS while the equipment used for PP is less optimally designed for PLA (Jamshidian et al., 2010). A sensitivity analysis on the variation of these weights has been included in the discussion section.

1.4 Product systems

Five product systems are analysed, three bio-based and two petrochemical counterparts:

- **Baseline:** PLA market average mix made from maize cultivated in the United States and sugarcane from Thailand. This product system reflects the status-quo of the PLA production according to the defined temporal scope.
- **Alternative 1.** PLA made from maize cultivated in Europe, fictional case. This scenario has the goal to show the environmental consequences of hypothetical production of PLA in Europe and from European maize.
- **Alternative 2**. Bio-based PP made from used cooking oil (based on patent EP3095844A1 (Hakola & Nyman, 2016).
- **Reference 1.** Petrochemical PP (see specific section in chapter *Investigation of generic LCA data for fossil-based plastics*).

- **Reference 2.** Petrochemical PET (see specific section in chapter *Investigation of generic LCA data for fossil-based plastics*).

1.5 System boundaries

A cradle-to-grave approach is taken including the life cycle stages of biomass feedstock production, transportation of intermediates (e.g. PLA granulates), manufacturing into end products (thermoforming), distribution of the final product and EoL. The investigated EoL options are plastic recycling, MSW incineration and landfilling for all the product systems (see section *EoL methodology* for overview of selected EoL options). Moreover, PLA cups have the additional options of industrial composting and Anaerobic digestion. The consumer use phase is excluded from the analysis because it is the same for all the product systems and it would add a negligible impact. Thus, the life cycle results will be divided into two stages: cradle-to-user and EoL.

2 Life cycle inventory analysis

2.1 Process description, data, assumptions and multifunctionality

2.1.1 Baseline. PLA using maize from United States and sugarcane from Thailand

The baseline case consists of two major technologies for the conversion of sugars into PLA. These two technologies are represented by NatureWorks and Total-Corbion PLA production processes. Based on the capacities of the two producers, in the LCI modelling, PLA is assumed to be 33 % by weight from Total-Corbion PLA and 67 % by weight from NatureWorks. This weighted average represents the 'average technology' of this baseline product system based on the current installed capacity (75 kton PLA/y and 150 kton PLA/y respectively for Total-Corbion PLA and NatureWorks.

The commercial names of NatureWorks and Corbion PLA are Ingeo® and Luminy®.The corresponding grades used for extrusion/thermoforming are respectively named 2002D and LX175.

The process flow diagrams of the baseline case are shown Figure 79 (NatureWorks's PLA) and Figure 80 (Total-Corbion PLA's PLA). The following key unit processes that are modelled in the foreground system are:

- Biomass cultivation and harvest;
- Lactic acid and PLA production, consisting of three sub-processes:
    - Starch/sugar milling;
    - Lactic acid production;
    - PLA production.
- Transportation of PLA polymer to Europe;
- Production of cups (Thermoforming);
- Distribution of the cups;
- Use phase and EoL.

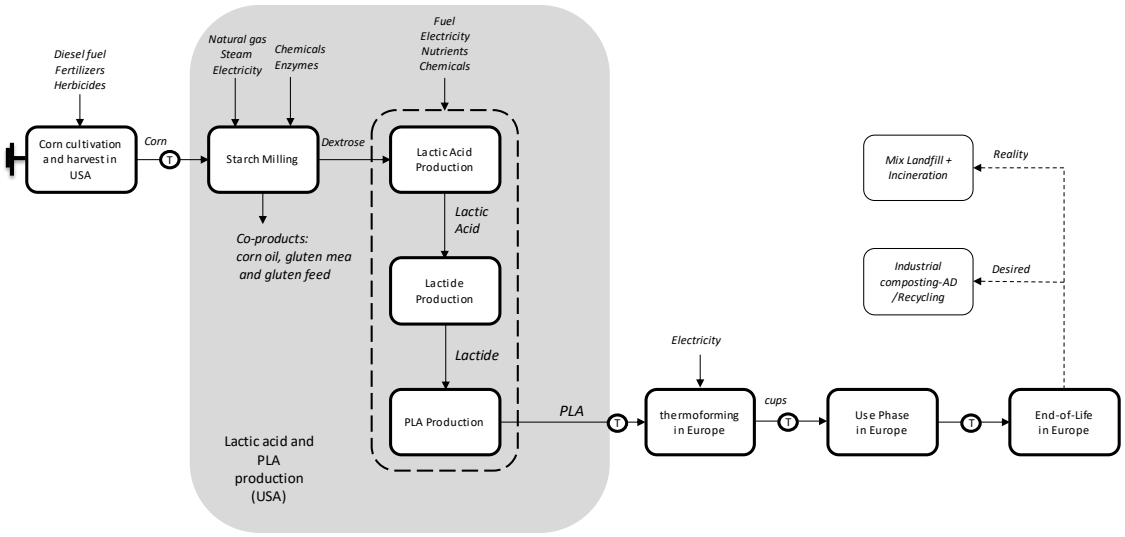


Figure 79. Process flow diagram of PLA cups production from US maize (reproduced based on Vink et al, 2015).

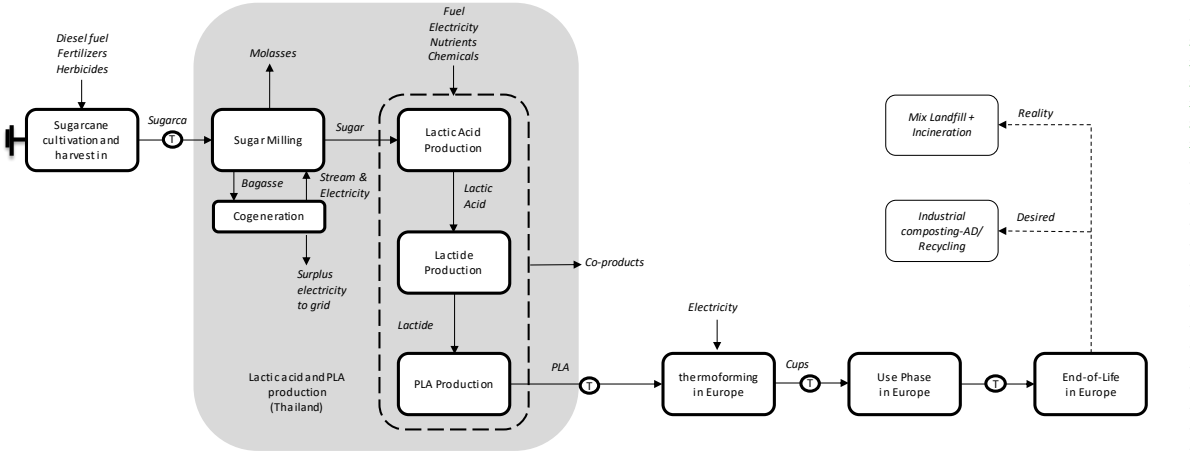


Figure 80. Process flow diagram of PLA cups production from sugarcane (reproduced based on Groot et al., 2010).

Table 96 shows a summary of the data sources used for the calculations for each stage of the cradle-to-user life cycle (see Figure 79 and Figure 80). Company specific data have been used whenever possible. When updated company data were not available, literature and LCA databases have been used.

Table 96. Summary of data sources for each phase of the cradle-to-user life cycle - baseline

Processes	Data sources	Comments
Maize cultivation	Primary company data & Vink et al., 2015	Represents site-specific data
Sugarcane cultivation	Primary company data & Agri-fooprint 3 database	Represents site-specific data whenever possible and average Thai sugarcane cultivation assumed for missing data.
PLA and lactic acid production	Primary company data	Represents the specific production technologies of the two producers.
Transportation of the polymer	Assumptions based on the actual locations of the production plants	Represents the actual distances existing between Europe and the producers
Thermoforming	Ecoinvent 3.3 & Suwamance et al. 2012	Represents thermoforming of cups occurring in Europe
Distribution of cups	Assumptions based on default distribution scenarios of PEFCR guidelines	Represents the distribution of cups from the European manufacturing plants where the thermoforming process occurs to the users.

2.1.1.1 Maize cultivation and harvest

The life-cycle of NatureWorks's PLA starts with the cultivation and harvesting of maize which take place in Nebraska and Iowa (USA) (Vink et al., 2015). The main inputs in this phase are agricultural chemicals (fertilisers, herbicides and pesticides), electricity and fuels for the farm activities (e.g. diesel used by the tractors), agricultural land and irrigation water. The maize production is included in the dataset *Gabi ts 8, Ingeo Polylactide (PLA) biopolymer production* and has been modelled by NatureWorks. The current LCI data from cradle to factory gate of 1 kg of PLA produced by NatureWorks represents the latest NatureWorks' eco-profile for PLA (Vink et al., 2015), with updated background data using Gabi database version 2018 (Personal communication with E. Vink).

The site-specific data for the maize used for NatureWorks PLA have been used in this part of the LCI. The black box impact of this maize has been provided by the company. The yield is assumed to be 10.28 ton/ha/yr for maize with 15 % moisture by weight (Vink et al., 2015).

2.1.1.2 Sugarcane cultivation and harvest

The sugarcane used for PLA production is cultivated in Thailand. The sugarcane cultivation and harvesting life-cycle stage has been modelled based on primary company data and included in the dataset for PLA provided by the company. *Sugar cane, at farm/TH Economic* from Agri-footprint 3 has been used as the main process to obtain the breakdown corresponding to the cultivation step. The water inventory, provided by the company in the sugarcane cultivation dataset, was based on the actual water needed in the local area where the main mills providing the sugarcane are located.

2.1.1.3 Lactic acid and PLA production unit process (Starch milling, Lactic acid production, PLA production)

Based on the different biomass and sugars, the conversion process of the sugar into PLA can start from dextrose (NatureWorks' technology) or sucrose (Corbion-Total's technology).

In the case of NatureWorks, the harvested maize is transported to the maize wet milling factory where dextrose is produced via enzymatic conversions. Only the starch content

(estimated to be 58 % of the mass content based on Vink et al. (Vink et al., 2015)) of maize will be hydrolyzed to dextrose. The co-products of the starch-milling and their percentages on a dry mass-basis are: maize oil (about 3 %), gluten meal (about 5 %) and gluten feed (about 25 %) (Vink et al., 2015). During dextrose production steam and electricity are needed and sourced locally (Nebraska, US infrastructure). In particular, steam is produced in natural gas fired steam boilers located at the NatureWorks' plant (Vink et al., 2015). To solve this case of multi-functionality, sub-division was performed by Vink et al. (Vink et al., 2015) in the NatureWorks' eco-profile. The unit process has been divided into 11 sub-processes (Vink et al. 2015) and, for each sub-process, where an allocation was needed a mass allocation (on a dry basis) was applied. Dextrose is sent to the fermentation unit to produce lactic acid which is the precursor of the polymer PLA. In lactic acid production - gypsum is produced as a by-product in small quantities. A credit was assigned through the substitution of primary production of gypsum from mining (Vink et al., 2015). The entire starch/sugar milling, lactic acid production and PLA polymerisation are in an integrated production site which is located in Blair (Nebraska). To produce each kilogram of PLA 1.57 kg of maize is required (Vink et al. 2015). *Gabi ts 8, Ingeo Polylactide (PLA) biopolymer production* has been used to model this unit process for NatureWorks' technology with updated background data from 2018. This LCI model is confidential.

At Total-Corbion PLA, the biomass used for PLA production is sugarcane. The harvested sugarcane is processed to produce raw sugar (Figure 80). The by-products of this process with common weight shares are: molasses (4.2 %w), filter cake (4.5 %w) and bagasse (34.0 %w) (Prueksakorn et al., 2014). The sugar milling of sugarcane requires heat and electricity which are entirely produced by burning bagasse in a combined heat and power (CHP) plant incorporated into the sugar mill. The surplus of electricity is sold in the Thai national grid (Groot et al., 2010). The sugar is then transported and processed into the lactic acid plant located next to the mill. During the lactic acid production process, gypsum and stillage are produced as by-products and are removed through filtration and sold (Groot et al., 2010). This gypsum has similar characteristics of mined gypsum while the stillage is used as animal feed due to its nutritional properties (system expansion with substitution of mined gypsum and animal feed has been applied). There are also other minor fermentation residues which are also purified and sold and used as soil conditioner. After lactic acid purification, PLA is produced through open ring polymerization. Primary data provided by the company based on the actual operation data has been used to model this unit process for Total-Corbion PLA's technology. The technology is the same as described by Groot et al. (2010). The updated LCI dataset is confidential.

2.1.1.4 PLA granulates transportation from US to Europe

The PLA granulates produced in Nebraska are transported to Europe to be converted into cups. The distances assumed in the modelling are:

- 2000 km by freight train from the PLA production site in Nebraska to a suitable harbour on the Atlantic coast of the United States (assumed to be New York). For this transport step, *Transport, freight train {US}* from Ecoinvent 3.3 has been used as dataset.

- 6000 km by transoceanic container ship from New York to an European harbour (assumed to be Rotterdam). The process named *Transport, freight, sea, transoceanic ship {GLO}* from Ecoinvent 3.3 was used as background data.

This is also what was assumed for single-use PLA cups in a previous comparative LCA of drinking cups (OVAM, 2006) and it has been confirmed by personal communication with Vink E.

2.1.1.5 PLA granulates transportation from Thailand to Europe

The PLA granulates produced in Rayong (Thailand) are transported to Europe to be converted into cups. The distances assumed in the modelling are:

- 200 km by freight train for transportation within Thailand. For this transport step, specific data based on the actual location of Corbio-Total are used. *Transport, freight train {US}* from Ecoinvent 3.3 has been used as the dataset.
- 17000 km by transoceanic container ship from Bangkok to a European harbour (assumed to be Rotterdam). The process named *Transport, freight, sea, transoceanic ship {GLO}* from Ecoinvent 3.3 has been used.

2.1.1.6 Production of single-use cups in Europe: thermoforming

The end-product, the cup, is manufactured via the process of thermoforming in two steps:

1 Sheet extrusion: the resin is melted and extruded into a sheet (extrusion of plastic sheet);

2 Cup forming: the sheet is then mechanically pressed into the shape of cups at the glass transition temperature (the part of the process more properly named 'thermoforming').

A so-called "inline extruder- thermoformer" process is assumed to be used for the baseline analysis. In an inline extruder, the extrusion of plastic sheets and thermoforming are integrated. This solution allows for electricity savings compared to a separate production line because the downstream thermoforming recovers part of the original heat from the extrusion, and therefore only limited preheat is needed (Florian, 1996). Depending on the material to be processed, the temperature and the time required by the process vary. The electricity consumption assumed for each material is shown in Table 97.

These values are based on the electricity use for the different materials derived from literature (Suwamance et al., 2012; RECIPE, 2007) and on the typical savings that inline thermoforming allows compared to a general thermoforming (estimated from Ecoinvent 3.3 data). The unit process of thermoforming *Extrusion of plastic sheets and thermoforming, inline {FR}| processing | Alloc Def, U* provided by Ecoinvent 3.3 was modified based on literature information on the specific electricity consumption. *Electricity, medium voltage {Europe without Switzerland}| market group for | Alloc Def, U* has been used as background data.

Table 97. Electricity consumption of extrusion of plastic sheets and thermoforming.

Material	PP	PLA	PET
Electricity consumption (kWh) inline	1.41	1.23	1.03
Electricity consumption (kWh) separated	1.48	1.35	1.13
Reference	Ecoinvent 3.3 modified based on RECIPE (RECIPE 2007)	Ecoinvent 3.3 modified based on Suwamance et al. (Suwamance et al., 2012)	Ecoinvent 3.3 modified based on RECIPE (RECIPE 2007)

PLA is thermoformed using ovens, molds and trims often adapted for PET or PS while the equipment used for PP is less optimally designed for PLA (Jamshidian et al., 2010).

2.1.1.7 Distribution of the single-use cups to the end user (within Europe)

For the final distribution of the product, it is assumed that the cups are distributed through a distribution centre.

According to previous similar studies (OVAM, 2006), a distance of 850 km between the producer of the cups and the distribution centre has been assumed. The process *Transport, freight, lorry >32 metric ton, EURO4 {RER}* from Ecoinvent 3.3 has been used as dataset.

Subsequently, the cups are transported to the user. The distance between the user and the distribution centre is assumed as 250 km (according to PEFCR's default recommendations). According to PEFCR guidelines, the process *Transport, freight, lorry 3.5-7.5 metric ton, EURO3 {RER}* has been chosen.

This set of data for the distribution of the cups has been implemented for all the product systems in this case study.

2.1.2 Alternative 1. PLA made from European maize

This system represents a fictional production of PLA cups by exploiting the potential impact of using European maize. This product system is largely based on NatureWorks' processes. Compared to NatureWorks's maize-PLA described above, two major changes have been applied for this alternative case:

- Maize cultivation and harvest in Europe. The European average maize is modelled based on Agri-footprint 3 data. The most important European maize producers are in Germany, France, Italy, Romania, Hungary, Spain, Poland and Bulgaria, in total these countries produces about 85 % of European maize per year (based on EUROSTAT average data in past five years from 2013 to 2017) (EUROSTAT, 2017).The contribution of each EU country to the average mix assumed for the modelled maize is shown in Table 98.

Table 98. Contribution of the EU countries in the modelled European maize accounting for 85 % of the EU maize production (EUROSTAT, 2017).

EU country of production	Share assumed in the model (%)
Germany	7.9
France	23.7
Italy	14.7
Romania	17.1
Hungary	12.5
Spain	13.0
Poland	6.4
Bulgaria	4.7

- PLA polymer transportation. Since this is a fictional supply chain, no real company data is available. We therefore used PEFCR guidance's default recommendations to model

the transportation. According to PEFCR guidelines, for all suppliers inside Europe, the following transportation distances and mode are assumed: 230 km by truck (>32 ton, EURO 4) (*Transport, freight, lorry >32 metric ton, EURO4 {RER}| transport, freight, lorry >32 metric ton, EURO4 | Alloc Def, U*), 280 km by train (*Transport, freight train {Europe without Switzerland}| market for | Alloc Def, U*) and 360 km by ship (*Transport, freight, inland waterways, barge {RER}| processing | Alloc Def, U*). All the foreground data of these three processes have been extracted from Ecoinvent 3.3.

It should be noted that because of a lack of transparency in the lactic acid and PLA production process, we assumed that NatureWorks's production can be used as a proxy for a fictional European lactic acid and PLA production. However, it is important to keep in mind when comparing the two, that the background data for Europe would be different from those for the US. Table 99 summarises all the sources of data used to model each life cycle stage.

Table 99. Summary of data sources for each phase of the cradle-to-user life cycle – alternative 1.

Processes	Data sources	Comments
Maize cultivation	Primary company data & Agri-fooprint 3 database	Represents European average data
PLA and lactic acid production	Primary company data	Represents the specific production technologies of one current producer of PLA that uses the same route in another country.
Transportation of the polymer	Assumptions based on default transportation scenarios of PEFCR guidelines	Represents the distribution of cups within Europe.
Thermoforming	Ecoinvent 3.3 & Suwamance et al. 2012	Represents thermoforming of cups occurring in Europe
Distribution of cups	Assumptions based on default distribution scenarios of PEFCR guidelines	Represents the distribution of cups from the European manufacturing plants where the thermoforming process occurs to the users.

2.1.3 Alternative 2: Bio-based PP made from UCO

The second alternative product system refers to bio-based PP derived from used cooking oil. This is a TRL 8 technology with a successfully operating prototype (Neste, 2018). The flow diagram describing this product system is shown in Figure 81.

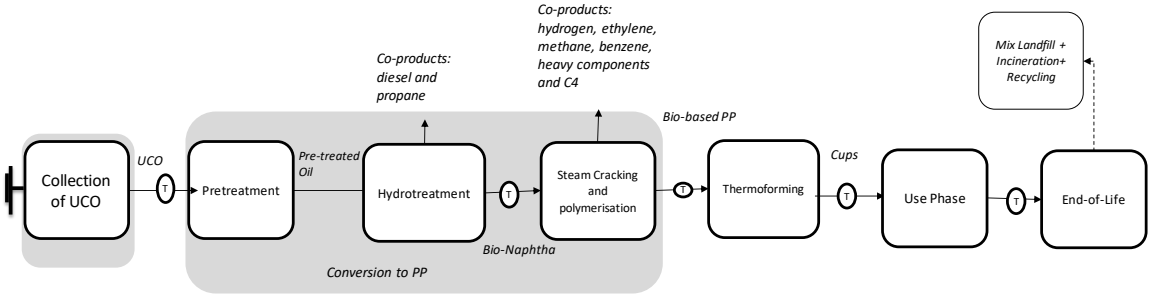


Figure 81. Process flow diagram of bio-based PP cups from used cooking oil (production site is in Europe).

The first step which is identified is the collection of UCO. The collection of UCO occurs locally, close to where the refinery plant is located.[53] The UCO is collected by trucks or vans from restaurants, commercial and private waste, and local waste collection centers.

At the refinery, the collected crude UCO is first sent to the pre-treatment process where solid impurities are filtered out. The process requires water, chemicals, heating and cooling which are locally sourced. The waste water of this operation is then treated in an on-site waste water treatment plant.

The pre-treated oil is ready to be upgraded via hydrotreatment by adding high pressure hydrogen. The final outputs of the hydro-treatment step are a hydro-treated vegetable oil (i.e. renewable diesel), propane, bio-based naphtha (refer to as 'bio-naphtha' from here on) and a large amount of water. This bio-naphtha is the feedstock to produce bio-based polypropylene. The propane is sold as a by-product. The main input of the process, hydrogen, is sourced from refinery hydrogen produced via steam reforming of natural gas.

The bio-naphtha is then transported to the cracking plant. The assumed distance is based on site-specific data and is kept confidential.

The steam cracking of bio-naphtha is very similar to that of petrochemical naphtha; the steam cracking of naphtha delivers two main products, propylene and ethylene, and several by-products. Ethylene represents the highest mass fraction among the cracked gases compared to propylene. The fuel burnt in the cracking unit is LPG. Additional energy is provided by steam which is also largely used for dilution. The by-products produced are heat, hydrogen, methane and other light gases, benzene and heavier products. Exergy allocation is performed for this multi-output process to consider the quality of the energy outputs.[54]

The last step of the chain is the polymerization of the propylene and it is the same as the polymerization of the petrochemical PP.

Energy allocation is applied as a partitioning approach to solve the multifunctionality of hydrotreatment. The use of a partitioning approach to solve this case of multifunctionality is not fully consistent with the section *Procedure of multi-functionality processes* in Chapter 2 "Approach and methodology". However, applying substitution when the focus is a non-dominant by-product (i.e. bio-naphtha and bio-propylene in this case study) could lead to misleading results because all credits will be assigned to a minor stream of the production, where the output of the technology is to optimise the main product - renewable diesel. The importance of the co-products' physical significance in the choice of allocation method is not sufficiently covered by the ISO guidance (Sandin et al., 2015), nor does the latest PEFCR guidance provides this instruction. According to the ILCD handbook, the dominant co-product to which all the credits are assigned should 'contribute more than 50 % to the combined market value of all co-functions of the analysed multifunctional process or system'. In the case of steam cracking co-production, for exampel ethylene is produced with a higher yield compared to propylene therefore this last requirement is not properly complied with for propylene which is the investigated product. For the base case, we choose the recommendation from RED to perform the allocation based on energy/exergy content since all by- and co-products are energy products. A discussion of this allocation methodological choice is also included in annex *Multi-functionality bio-based PP cups/packaging films product system*.

[53] The specific site is analysed in this LCA, but the location is kept confidential.

[54] This is a common allocation method when heat is one of the co-products (e.g. this has been applied by Edwards et al. (Edwards et al., 2017)) when the quality, not only the quantity, of the energy needs to be addressed.

Biogenic carbon removal is considered for the cradle-to-user phase according to the RED II proposal because a vegetable waste is used, and biogenic carbon will be stored in the product. This amount of carbon will then be released into the atmosphere during EoL as carbon emission if the carbon is fully oxidised (e.g. in a MSWI).

The summary of the data sources for the unit processes highlighted in Figure 81 is reported in Table 100.

Table 100. Summary of data sources for each phase of the cradle-to-user life cycle-Alternative 2

Processes	Data sources	Comments
UCO collection, pre-treatment and hydrotreatment	Primary company data; Nikander, 2008; IFEU, 2006; JEC, 2014.	Represents site-specific data
Transportation of the Naphtha and steam cracking	Primary company data & Karimzadeh et al., 2015	Represents site-specific data whenever possible
Polymerisation	PlasticsEurope- Industry 2.0 database (PlasticEurope, 2013)	Represents European polymerisation
Transportation of the polymer	Assumptions based on default transportation scenarios of PEFCR guidelines	Represents the transportation of the polymer from the biorefinery to the plants where the thermoforming process occurs
Thermoforming	Ecoinvent 3.3, RECIPE 2007 and Suwamance et al. 2012	Represents thermoforming of cups occurring in Europe
Distribution of cups	Assumptions based on default distribution scenarios of PEFCR guidelines	Represents the distribution of cups from the European manufacturing plants where the thermoforming process occurs to the users.

2.1.4 Reference 1. Petrochemical PP

The flow diagram of the petrochemical PP cups system can be found in Figure 82. As with all the conventional plastics, PP derives from steam cracking of naphtha, which is a multi-output unit where propylene is one of the co-products. By means of propylene polymerisation, polypropylene resin is obtained, and it is ready for film extrusion. The dataset selected for the baseline is retrieved from PlasticsEurope eco-profile (see section 6.1.1 *Databases used for the comparison*). However, as explained in Chapter 6 (*Ranges for environmental impacts from production of fossil-based plastics*), PlasticsEurope data for petrochemical plastics are not transparent and do not comply with ILCD requirements. Differences between the available data from different data sources for PP are found to be large and some impact categories were found to be not suitable (see section 6.1 in Approach and methodology). The main data sources used to model the unit processes shown in Figure 82 are reported in Table 101.

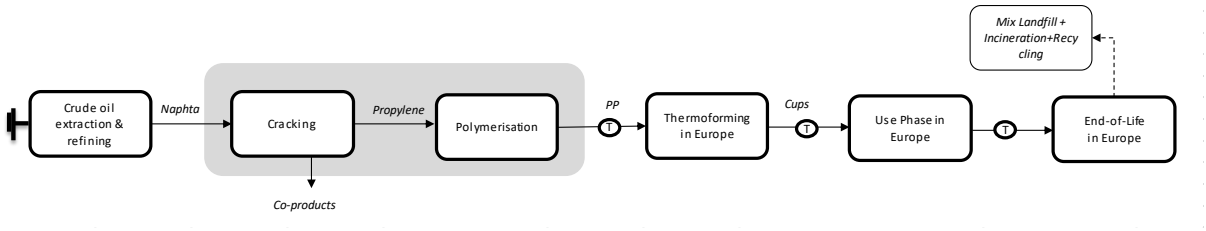


Figure 82. Process flow diagram of PP cups production from crude oil.

Table 101. Summary of data sources for each phase of the cradle-to-user life cycle- Petrochemical reference 1.

Processes	Data sources, foreground	Comments
PP production (fossil)	PlasticsEurope, 2017 & Industry data 2.0	Represents European average production of petrochemical PP
Transportation of the polymer	Assumptions based on default distribution scenarios of PEFCR guidelines	Represents the transportation of the polymer from the European refineries to the European manufacturing plants where the cups are produced
Thermoforming	Ecoinvent 3.3 & RECIPE 2007	Represents thermoforming of cups occurring in Europe
Distribution of cups	Assumptions based on default distribution scenarios of PEFCR guidelines	Represents the distribution of cups from the European manufacturing plants where the thermoforming process occurs to the users.

2.1.5 Reference 2. Petrochemical PET

Figure 83 shows all the production steps of PET cups. The PET used for cups is in an amorphous state which allows it to be transparent. PET is derived from two intermediate products obtained from the cracking of naphtha: monoethylene glycol (MEG) and purified terephthalic acid (PTA). They are the inputs of the esterification process which gives an intermediate product that becomes amorphous PET after polymerization. Amorphous PET is modelled according to what done for PET bottle grade, as explained in the *Case study 1: Beverage bottles* and Ranges for environmental impacts from production of fossil-based plastics. For fossil-based MEG, Ecoinvent 3.3 *Ethylene glycol {RER}| production | Alloc Def, U* is used. The climate change and NREU impact categories are updated with the breakdown given in the latest PlasticsEurope PET eco-profile (PlasticsEurope, 2017). Fossil-based PTA production is instead based on PlasticsEurope's PTA eco-profile (PlasticsEurope, 2014). The reference PET is made of 0.32 kg MEG/kg PET and 0.86 kg PTA/kg PET according to stoichiometry (see section 2.1.4 of case study 1- *The fossil reference systems: petrochemical PET*). The compiled data for PET is compared with the available data from different data sources for PET in Chapter 6. Differences are found to be large and some impact categories were found not applicable. Table 102 summarises the data sources used for the main unit processes and intermediates products shown in Figure 83.

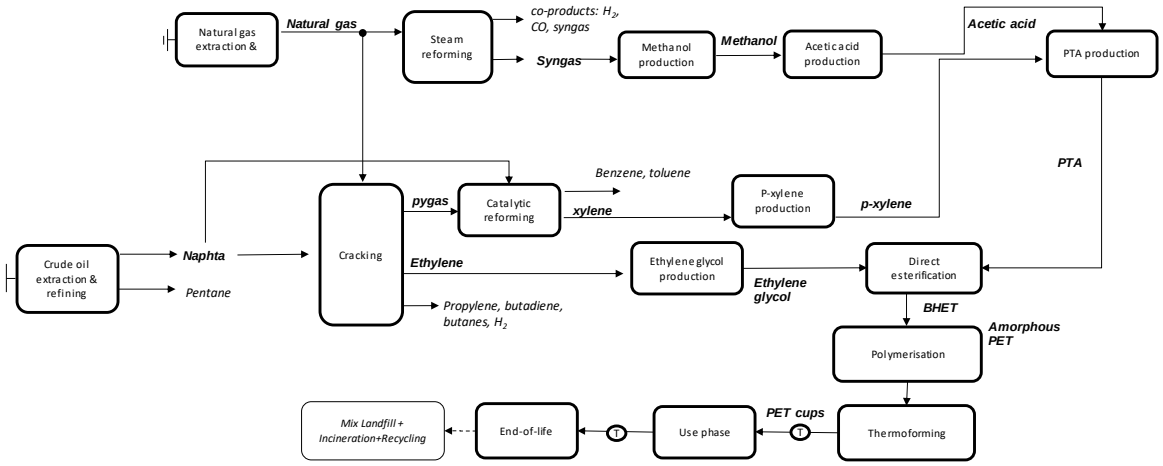


Figure 83. Process flow diagram of PET cups production from crude oil.

Table 102. Summary of data sources for each phase of the cradle-to-user life cycle-Petrochemical reference 2.

Processes	Data sources	Comments
Petrochemical-MEG production	Ecoinvent 3.3	Ethylene glycol {RER}| production | Alloc Def, U
PTA production (fossil)	PlasticsEurope, 2017 & Industry data 2.0	Represents European average, reference year of 2015
Transportation of the polymer	Assumptions based on default distribution scenarios of PEFCR guidelines	Represents the transportation of the polymer from the European refineries to the European manufacturing plants where the cups are produced
Thermoforming	Ecoinvent 3.3 & RECIPE 2007	Represents thermoforming of cups occurring in Europe
Distribution of cups	Assumptions based on default distribution scenarios of PEFCR guidelines	Represents the distribution of cups from the European manufacturing plants where the thermoforming process occurs to the users.

2.2 End of Life description, data and assumptions

The single-use cups have different possible EoL options depending on the product system, as observed in Table 103. The average technologies are explained in detail in the Approach and methodology chapter (Section 3.4 Average End of Life technologies).

For cups made from PP, both bio-based and petrochemical, and the petrochemical PET, three possible EoL options are considered:

- **Mechanical plastic recycling**: the process includes the energy and material requirements for the transportation to the facility, sorting, cleaning and recycling processes. The recycled PET is assumed to substitute virgin PET production and PP is assumed to substitute virgin PP. The rejects from the recycling process are sent to

incineration (both with and without energy recovery) and includes transportation. The flowchart for plastic recycling is presented in Figure 8.

- **MSW Incineration with and without energy recovery**: a generic MSW incineration plant is assumed, which represents average EU waste incineration with and without energy recovery with an overall heat efficiency of 22 % and electrical efficiency of 9 % (CEWEP, 2013; Eurostat, 2017a). The process includes the direct emissions from waste incineration as well as indirect emissions from the production of the input materials, the combustion of fossil fuels as well as treatment of bottom ash and fly ash. The flowchart for MSW incineration is presented in Figure 9.

- **MSW landfilling**: the process includes the construction and operation of an average landfill site, direct emissions of landfill gas, landfill gas collection, flaring and recovery system, treatment of collected leachate and direct emissions from uncollected leachate. 29 % of the collected landfill gas is used to generate electricity (OpenLCA Nexus, 2015). The electricity conversion efficiency is 37 % (Christensen, 2011). The flowchart for MSW landfilling is presented in Figure 10.

The biodegradable PLA cups in addition have options of:

- **Industrial composting**: includes indirect emissions from material and energy consumption, direct emissions from the plant and use on land, fertiliser substitution from compost and rejects sent to incineration (30 % of the plastic and 5 % of the organic waste). The flowchart for industrial composting is presented in Figure 12.

- **Anaerobic digestion**: mix of technologies modelled (combined with and without post-maturation (aerobic), which includes indirect emissions from material and energy consumption, pre-treatment (rejects sent to incineration, 30 % of the plastic and 5 % of the organic waste), electricity, heat and fuel generation from biogas, direct emissions from the plant and use on land, and fertiliser substitution from compost and digestate. The flowchart for anaerobic digestion is presented in Figure 13.

The intended EoL technologies for the PLA cups are recycling and industrial composting. It should be noted that recycling of PLA is still on a theoretical level, as there are no commercial plants recycling PLA today. For the alternative bio-based PP the intended technology is recycling.

The "EoL mix of EU" refers to mixed waste management based on the available statistics in Europe on plastic waste treatment. The current collection for treatment of plastic waste is 30 % recycling, 39 % incineration and 31 % landfilling in Europe (European Commission, 2018). This is assumed to be applicable for the PET and PP cups, also for the bio-based PP.

For the EoL mix for the PLA cups, it is estimated that Europeans can sort the same amount of PLA cups, which they today are able to sort for recycling of the commercial plastics, hence 30 %. This was then equally divided between the two intended technologies for PLA; industrial composting and recycling, hence 15 % each. The EoL technologies modelled and the EoL mix are presented in Table 103.

The EoL mix of EU is used to model the product systems to create a most likely "status-quo" of waste management of a given product.

Table 103. The EoL options, the intended EoL option and the estimated EoL mix.

Product system	EoL options modelled individually (100 % scenarios)	Intended EoL option	Estimated EoL collection mix of EU (based on statistics of waste treatments of waste plastics)
Bio-based PLA	Recycling Incineration Landfilling Industrial composting Anaerobic digestion	50 % recycling and 50 % industrial composting	Recycling: 15 % Industrial composting: 15 % Incineration: 39 % Landfilling: 31 %
Bio-based PP	Recycling Incineration Landfilling	Recycling	Recycling: 30 % Incineration: 39 % Landfilling: 31 %
Petrochemical PP	Recycling Incineration Landfilling		Recycling: 30 % Incineration: 39 % Landfilling: 31 %
Petrochemical PET	Recycling Incineration Landfilling		Recycling: 30 % Incineration: 39 % Landfilling: 31 %

The average European EoL technologies are described in the Approach and methodology chapter, (Section 3.4 Average End of Life technologies). In that section, factors that do not depend on the specific product system are described, e.g. the energy efficiency of the incineration plant and the consumption of energy and materials for each technology. Furthermore, principles for substitution are described. This involves the energy produced in the EoL technologies being substituted with marginal electricity and heat, as well as the recycling substitution methodology.

In addition, there are a few product system dependent factors, which are presented in Table 104, for each possible EoL technology depending on the product system. The recycled plastic is substituted with fossil PET for both the PLA product system and the petrochemical PET, whereas for both the PP product systems PP is substituted with petrochemical PP. The recycling rate is calculated by multiplying the collection rate and the sorting and technology efficiency. The sorting and technology efficiency was deemed to be the same for all pathways. In Table 8 higher efficiencies are observed for PET, but these will only be relevant for the bottles (cPET – high crystallinity), whereas the cups are made of aPET (amorphous), which does not have a higher recycling rate than polyolefins, hence increasing the sorting and technology efficency for PET in case study 3 would not be reliable.

The recycled PLA will substitute PET. This is chosen as the study looks at the limited time frame of the next 5-10 years. In this time, recycling of PLA is not expected to become commercially viable, due to low amounts of PLA on the market. Hence PLA will for a good time replace PET, as this will be the marginal plastic. The PLA could also replace PP, and the implications of this are reflected in the results.

Table 104. Values for product system dependent factors for cups.

EoL Technology	Product system dependent factor	Unit	Bio-based PLA	Reference	Bio-based and petrochemical PP	Reference	Petro-chemical PET	Reference
Plastics recycling	Sorting and technology efficiency (amount collected for recycling which is recycled)	% of input	70	Estimated based on Plastic Recyclers Europe, 2017, Replast, n.d., SWEREC, 2007, Franklin Associates, 2011, Giugliano et al., 2011 and Dall et al., 2003	70	Estimated based on Plastic Recyclers Europe, 2017, Replast, n.d., SWEREC, 2007, Franklin Associates, 2011, Giugliano et al., 2011 and Dall et al., 2003	70	Estimated based on Plastic Recyclers Europe, 2017, Replast, n.d., SWEREC, 2007, Franklin Associates, 2011, Giugliano et al., 2011 and Dall et al., 2003
MSWI with and without energy recovery	Energy content	MJ/kg	18.72	Boonmee et al., 2016; Pradhan et al., 2010; Hermann et al., 2010; Krause and Townsend., 2016; Kolstad et al., 2012	37.84	Götze et al., 2016 (Modified C)	22.91	Götze et al., 2016
Landfilling	1st order decay rate for methane generation	1/s	0.011	Calculated based on (Kolstad et al., 2012)	0	Estimated	0	Estimated
Industrial composting	VS Degradation (%)	% of VS	80	Pradhan et al., 2010				
	C Degradation (%)	% of C bio	57.1	Pradhan et al., 2010				
Anaerobic digestion	Anaerobically biodegradable biogenic carbon	% of bio C	47	Pradhan et al., 2010				

The single-use cups have the following material composition, see Table 105, which is used as the composition of the EoL-reference flows to the EoL LCA model. The full chemical composition is presented in Annex 1. The intermediate EoL-reference flow for analysing the impact of the EOL of the cups is:

- 1 kg of plastic and 100 g contamination

The contamination consists of organic waste – see further details in the Approach and methodology chapter, section 3.3.2 Contamination composition.

Table 105. Overview of the material composition of single-use cups (1 kg plastic + 100 g contamination).

Chemical component	Unit	Bio-based PLA	Bio-based PP	Petrochemical PP	Petrochemical PET
Water	% of total	6.6	3.3	3.3	3.3
TS (VS+ash)[a]	% of total	93.4	96.8	96.8	96.8
VS	% of TS	99.4	94.0	94.0	99.0
C fossil	% of TS	0.2	0.0	77.6	64.0
C biogenic	% of TS	50.0	77.6	0	0.0
References		Boonmee et al., 2016; Pradhan et al., 2010; Hermann et al., 2010; Krause and Townsend., 2016; Kolstad et al., 2012	Götze et al., 2016 (Modified C)	Götze et al., 2016	Götze et al., 2016

a TS: total solids, VS: volatile solids.

3 Life cycle impact assessment results and interpretation

3.1 Cradle-to-user environmental impacts and interpretation, bio-based systems

3.1.1 Cradle-to-user environmental impacts of Bio-Based baseline system: PLA

Table 106 and Figure 84 show the cradle-to-user environmental impacts of the PLA cups (the baseline system).

Table 106. Cradle-to-user environmental impact - PLA (baseline).

Impact category	Unit	PLA baseline (1 FU of cups)	PLA baseline (1 kg of cups)
Climate change	kg CO_2 eq.	8.63	1.88
Ozone depletion	kg CFC-11 eq.	1.04E-06	2.26E-07
Human toxicity, non-cancer effects	CTUh	3.35E-06	7.31E-07
Human toxicity, cancer effects	CTUh	8.81E-07	1.92E-07
Particulate matter	kg PM2.5 eq.	6.49E-03	1.41E-03
Ionizing radiation HH	kBq U235eq.	1.22	0.27
Photochemical ozone formation	kg NMVOCeq.	6.08E-02	1.33E-02
Acidification	molc H+eq.	9.90E-02	2.16E-02
Terrestrial eutrophication	molc Neq.	2.46E-01	5.35E-02
Freshwater eutrophication	kg Peq.	9.64E-04	2.10E-04
Marine eutrophication	kg Neq.	4.02E-02	8.76E-03
Freshwater ecotoxicity	CTUe	71.04	15.48
Land use	kg C deficit	91.98	20.05
Water use	m^3	6.07	1.32
Abiotic depletion	kg Sb eq.	1.55E-05	3.38E-06
Abiotic depletion (fossil fuels)	MJ	212.28	46.26
NREU	MJ	257.86	56.19

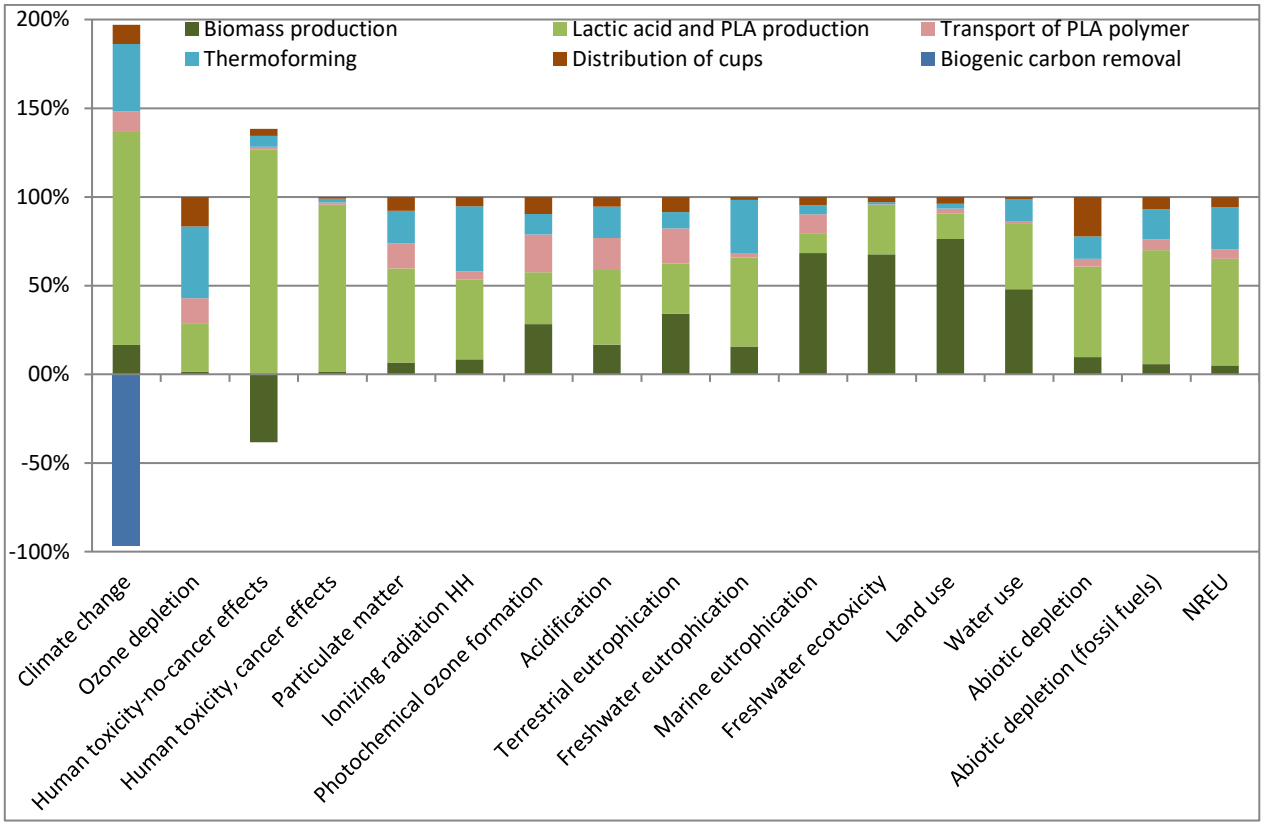


Figure 84. Breakdown of the cradle-to-user environmental impact of cups made from PLA (baseline).

Based on the results from cradle-to-user, we observed the following environmental impact contributions:

- The biomass production (US maize and Thai sugarcane) is the dominant (more than 60 % of the cradle-to-user environmental impact) source of environmental burden for marine eutrophication, freshwater ecotoxicity, land use and water use. The negative impact on human toxicity-(non-cancer effects) is a consequence of Thai sugarcane that absorbs zinc from the soil (more details are provided in the interpretation of this impact category). The cultivation step has a negligible impact on ozone depletion, human toxicity-cancer effects and abiotic depletion-(fossil fuels).

- The lactic acid and PLA production is the main cause of climate change, human toxicity (both cancer and non-cancer effects), particulate matter, ionisation radiation HH, acidification, freshwater eutrophication, abiotic depletion and abiotic depletion (fossil fuels), and NREU. It has a minor impact (about 10 %) on marine eutrophication.

- The transportation of PLA granulates -from Nebraska or Rayong to European plastics conversion companies- contributes significantly to ozone depletion, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication and marine eutrophication. For climate change and abiotic depletion of fossil fuels (NREU), the transportation of the PLA granulates plays a relatively minor role.

- Thermoforming (including the extrusion of plastic sheets) of PLA plays an important role in all of the impact categories. This is mainly caused by electricity use. It has a significant contribution (20-50 %) to the cradle-to-user environmental impacts of ozone

depletion, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication and marine eutrophication.

- The final distribution of cups has almost the same impact as the transportation of the granulates, with high impacts on abiotic depletion and ozone layer depletion due to the emissions caused by road transportation.

Based on the cradle-to-grave weighted results (see section 3.4 of this case study chapter), the most important impact categories (i.e. the ones that account for 80 % of the total impacts) are summarised as the following:

- If toxicity impacts are excluded, six categories cover approximately 79 % of the total impacts. In order of relevancy, these are climate change, abiotic depletion (fossil fuels), particulate matter, acidification, land use and photochemical ozone formation.

- If toxicity impacts are included in the weighting: all three toxicity categories appear to have a significant contribution: human toxicity (cancer effects), freshwater ecotoxicity and human toxicity (non-cancer effects).

According to the PEFCR guideline, the impact categories that cumulatively account for the 80 % of the total environmental impact (excluding toxicity related impact categories) have to be interpreted. As aforementioned, this leads to a detailed interpretation of six impact categories (climate change, abiotic depletion (fossil fuels), particulate matter, acidification, land use and photochemical ozone formation).

- **Climate change**: The cradle-to-user GHG emissions per functional unit of PLA cups has been estimated to be 8.63 kg CO_2 eq. or 1.88 kg CO_2 eq./kg. The gross GHG emissions (by excluding biogenic carbon removal) originate from:

    - 61 % from the production of lactic acid and PLA;

    - 21 % from thermoforming;

    - 8 % from biomass cultivation; and

    - the remaining 10 % by the two transportation steps (pellets (5 %) and cups (5 %)).

As aforementioned, the lactic acid and PLA production unit process is the unit process with the biggest environmental impact (61 %). The GHG emissions due to lactic acid and PLA production are mainly carbon dioxide of fossil origins (87 %).This unit process can be subdivided into four main sub-processes which are starch/sugar milling, lactic acid production, lactide production and polymerisation (see Figure 85 which provides further detail of the breakdown). In the production of lactic acid and PLA, lactic acid production has the highest contribution (60 %), followed by lactide production (25 %), polymerisation (9 %) and sugar/starch milling (6 %). The most important activities which lead to the climate change impacts in lactic acid production are the production and combustion of fuels (e.g. the natural gas burnt for producing steam), electricity and the production of chemicals (mainly lime and sulfuric acid). For lactide production, the use of natural gas and electricity are the main contributors to GHG emissions.

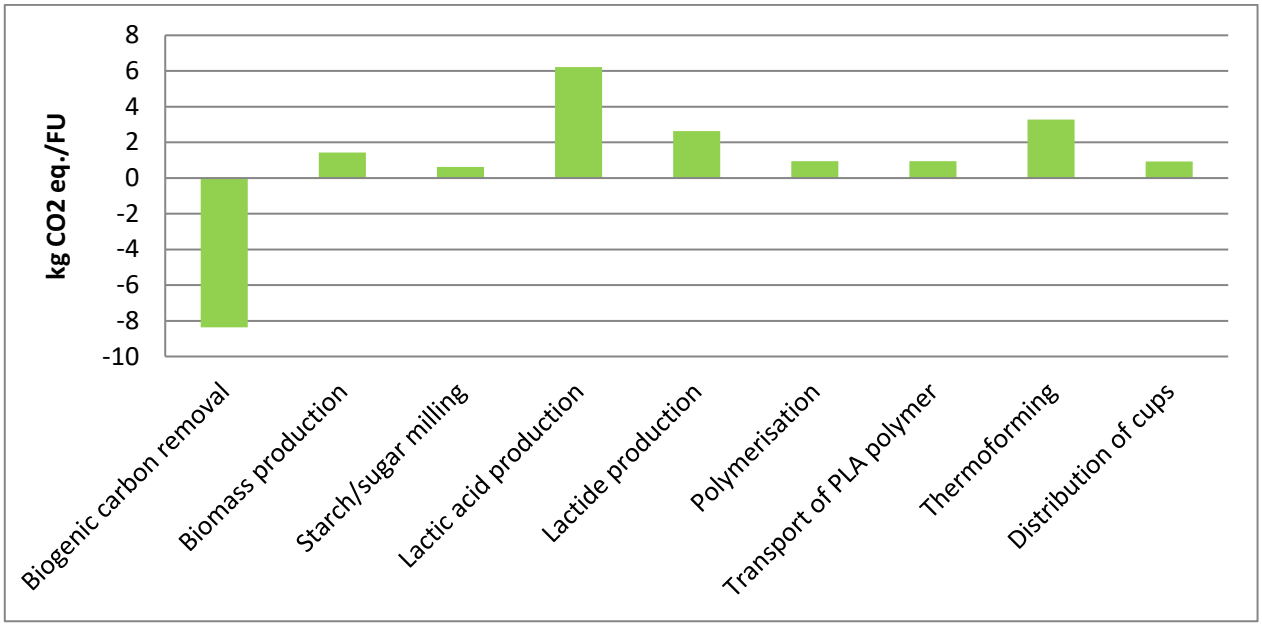


Figure 85. Cradle-to-user breakdown of climate change impact potentials for PLA(baseline) cups on functional unit basis: total Net GHG emissions (including biogenic carbon removal): 8.6 kg CO_2 eq., total Gross GHG emissions (excluding biogenic carbon removal): 17.0 kg CO_2 eq.

Regarding the GHG emissions of thermoforming, 91 % of these are the result of electricity consumption (*Electricity, medium voltage {Europe without Switzerland}* of Ecoinvent 3.3). The emissions of European mix electricity impacting the most on climate change are carbon dioxide of fossil origins (about 90 %) and methane of fossil origins (9 %)

Biomass cultivation contributes only 8 % of the total gross cradle-to-user GHG emissions without considering biogenic carbon removal. If biogenic carbon is considered, the biomass cultivation has a negative value for climate change impact (-8.37 kg CO_2 eq./FU). Note that this sequestered biogenic carbon would be released to the atmosphere again if it is fully or partially oxidised at the EoL waste management stage.

- **Abiotic depletion of fossil fuels (and similarly NREU).** These are two similar indicators. NREU is widely reported in the literature. As with to the climate change impact, abiotic depletion (fossil fuels) and NREU is dominated by lactic acid and PLA production; it accounts for ca. 66 % of total cradle-to-user impacts. Within this 66 % of the impacts, the major contributions are (see also Figure 86): lactic acid fermentation and downstream separation (61 %), lactide production (27 %), polymerisation (8 %) and starch/sugar milling: (4 %). Thermoforming of cups accounts for 26 % of the total NREU and 98 % of that is caused by the electricity used. The biomass cultivation impacts accounts for only 3 %, while the transportation only accounts for 11 %.

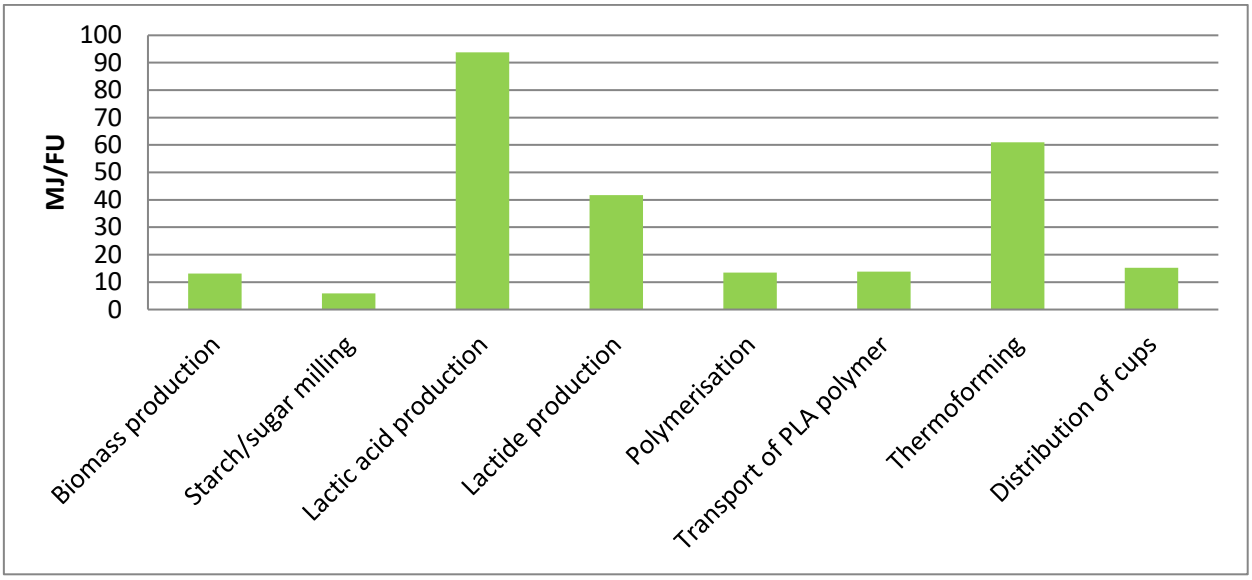


Figure 86. Cradle-to-user NREU breakdown for lactic acid and PLA production (MJ per FU).

- **Particulate matter.** For particulate matter, the lactic acid and PLA production is the step with the major cradle-to-user impact (52 %). The particulate emissions of this life-cycle stage come almost entirely from the lactic acid production while the lactide production and Polymerisation step have very minor contribution. The particulate matter of lactic acid and PLA unit process is released as particulates <2.5 % um (70 %), sulphur dioxide (26 %) and particulates <10 um (5 %). As regards the sugar-milling of PLA from Thai sugarcane, the bagasse burning can be identified as the main source of particulates. The second major contributor to particulate matter release is the thermoforming stage which causes 20 % of the total. Ninety-seven percent of the particulate emissions of thermoforming are a consequence of the production of the electricity used. The particulate matter caused by electricity production is 55 % due to sulphur dioxide emissions, 42 % particulates <2.5 um and nitrogen oxides 3 %. The two transportation steps are also relevant emitters of particulates and they account for 22 % of the share. The impact of the transportation stages is the consequence of particulates (<2.5 um) emitted (75 %) and sulphur dioxide released (16 %). As regards the cultivation of maize and sugarcane, it is responsible for only 6 %.

- **Acidification.** The life-cycle stage with the highest contribution (41 %) is the production of lactic acid and PLA. This is almost entirely due to the fermentation of the starch/sugar into lactic acid while starch/sugar milling and lactide production have minor impacts. The acidification caused by the lactic acid and PLA production is mainly due to the emissions of sulphur dioxide (78 %) and nitrogen oxides (about 10 %) caused by the production of the chemicals used during the conversion process. Nineteen percent of the impact is instead associated with the thermoforming process (96 % of which is caused by the electricity used). The acidification caused by the production of the electricity used in thermoforming is due to emissions of sulphur dioxide (77 %) and nitrogen oxides (21 %). Eighteen percent of the acidification is caused by the transportation of PLA granulates by container ships due to the SO_2 emissions caused by combustion of diesel, while only 5 % is caused by the transportation of the final product (road transportation). Sixteen percent of the acidification is due to the biomass production. During the cultivation stage, the direct emissions due to fertiliser use accounts for 59 %, while another 10 % are direct

emissions from the use of manure. The indirect emissions account for 25 % and are mainly due to the production of the electricity used and diesel burnt in machinery.

- **Land use.** The land use impact is dominated by the biomass production phase accounting for 76 % of the total cradle-to-user impact. In particular, the land use- accounted with the carbon deficit method- results are slightly higher in the case of sugarcane compared to maize per kg of PLA. The lactic acid and PLA production also has a considerable share of 14 %, due to substances that are burnt and used during the processing. Each of the transportation stages and thermoforming are responsible for only about 3 % of the impact.

- **Photochemical Ozone formation.** Photochemical ozone formation is generated mainly during three life cycle stages: lactic acid and PLA production (29 %), biomass production (28 %) and the transportation of polymer (21 %). The photochemical ozone formation related to the lactic acid and PLA production is mainly caused by emissions of nitrogen oxides (59 %), NMVOC (29 %) and sulphur dioxide (9 %). The photochemical ozone formation caused by the biomass production is instead mainly due to emissions of nitrogen oxides (81 %). Thermoforming accounts another 12 %, and 88 % is due to the production of the electricity used.

Several other impact categories which have been found to be relevant for biomass production are also interpreted here.

- **Water use.** Water use does not appear to be in the top 80 % of impact categories as summarised in Section 3.4. It is responsible for only 2-3 % of the total cradle-to-grave impact. Nevertheless, water is an important resource for bio-based products therefore we briefly explain where the water use comes from: 48 % of the water use impact is from the biomass feedstock cultivation while the 37 % is from the production of the PLA polymer. Thermoforming also has a non-negligible relevance (12 %). Regarding the lactic acid and PLA production unit process, the consumption of water is mainly due to lactic acid production (51 %) and sugar milling (31 %).

- **Human toxicity with and without cancer effects, plus freshwater ecotoxicity.** Human toxicity without cancer effects is almost totally produced during lactic acid and PLA production (92 %). This is caused mainly by emissions of zinc to soil (75 %) and mercury (7 %) and zinc (5 %) to air. A negative impact during biomass production is observed in the figure. This is almost entirely caused by the zinc absorbed from the soil by sugarcane. Heavy metals are often added to synthetic fertilisers to facilitate the absorption of the nutrients for crops. They are also often found in animal manure (especially zinc).[55] The sugarcane cultivation in Thailand uses both synthetic fertilisers and animal manure (Durlinger et al., 2017). These metals are either leached into the soil or absorbed by the biomass. When there is more zinc absorbed by the biomass than is leached, a negative emission appears in the LCI model (Durlinger et al., 2017). However, the zinc absorbed by the biomass will eventually end up in the environment if the biomass is burned for energy (e.g. bagasse), returned to the soil (fermentation distillage applied to the soil) or removed from the production lines and ending up in

[55] The amount of zinc and copper are found very high in pig manure because they are often added in the feed for animal health reasons (Durlinger et al., 2017).

waste (water) treatments. These emission pathways could potentially explain the high contribution of zinc during the lactic acid and PLA production.[56]

Human toxicity with cancer effects is also caused almost entirely by lactic acid and PLA production (94 % of the total cradle-to-user impact). This is mainly caused by emissions of chromium to water (50 %), to air (20 %) and to soil (20 %). Freshwater ecotoxicity comes more from biomass production (68 %) than from lactic acid and PLA production (28 %). The emissions that cause freshwater ecotoxicity are mainly: atrazine (herbicide) to soil (31 %), chlorpyrifos (pesticide) to soil (28 %), chlorpyrifos to water (18 %) and metolachlor (herbicide) to soil (11 %).

3.1.2 Cradle-to-user environmental impacts of Bio-Based alternative system 1. PLA from European maize

Table 107 shows the results of this hypothetical route of PLA made from EU maize (see description in Section 2.1.2) and compared with the bio-based baseline system (results described in Section 3.1.1). By switching the feedstock of PLA from US maize to locally sourced EU maize, out of 16 impact categories, 12 categories have increased impacts. The four impact categories with decreased impacts (i.e. ozone layer depletion, particulate matter, photochemical ozone formation and abiotic depletion (mineral and metal)) reflect the benefits from the reduced length of logistics chains.

The breakdown of the impacts based on life cycle stages shown in Figure 84 reveals that most of these increased impacts are associated with the mixed EU maize cultivation. The reduced impacts of (transoceanic) transportation are apparently not sufficient to compensate the increased impacts originating from EU maize cultivation. One of the main reason is the lower yield of European maize (7.23 ton/ha/yr for maize 15 % moisture estimated from Agri-footprint 3) compared to maize cultivated in Nebraska/Iowa (10.28 ton/ha/yr for maize with 15 % moisture by weight (Vink et al., 2015)). A higher yield corresponds to a more efficent use of land and consequently a lower use of resources per kg of crop for exampel fertilisers. Nevertheless, it should be noticed that the modelled maize represents the European average production. The productivity of maize varies widely among European countries ranging from 4.06 (Romania) to 11.06 ton/ha/yr (Spain). In consequence, significantly different results would be obtained if e.g. only the Spanish production were considered[57].

[56] Due to lack of transparency of the industrial data, the exact zinc balance through the entire life cycle cannot be fully reconstructed.

[57] For the remaining EU countries: 9.88 ton/ha/yr in Germany, 9.26 ton/ha/yr in Italy, 9.22 ton/ha/yr in France, 6.67 ton/ha/yr in Poland, 6.04 ton/ha/yr in Hungary and 5.91 in Bulgaria. Based on FAO (2016).

Table 107. Cradle-to-user environmental impacts (16 PEFCR categories + NREU) per functional unit -PLA cups made from European maize, compared with the results of the baseline PLA cups.

Impact category	Unit	PLA based on EU maize	PLA baseline	Increase (+) or decrease (-) of impacts (%)
Climate change	kg CO_2 eq.	11.55	8.63	34
Ozone depletion	kg CFC-11eq	8.46E-07	1.04E-06	-18
Human toxicity, non-cancer effects	CTUh	6.67E-06	3.35E-06	99
Human toxicity, cancer effects	CTUh	1.32E-06	8.81E-07	50
Particulate matter	kg PM2.5eq	6.30E-03	6.49E-03	-3
Ionizing radiation HH	kBq U235eq	1.54	1.22	27
Photochemical ozone formation	kg NMVOCeq	4.39E-02	6.08E-02	-28
Acidification	molc H+eq	1.40E-01	9.90E-02	41
Terrestrial eutrophication	molc Neq	4.42E-01	2.46E-01	80
Freshwater eutrophication	kg Peq	1.56E-03	9.64E-04	61
Marine eutrophication	kg Neq	8.11E-02	4.02E-02	102
Freshwater ecotoxicity	CTUe	121.13	71.04	71
Land use	kg C deficit	115.34	91.98	25
Water use	m3	41.94	6.07	591
Abiotic depletion	kg Sbeq	1.02E-05	1.55E-05	-34
Abiotic depletion (fossil fuels)	MJ	230.89	212.28	9
NREU	MJ	292.05	257.86	13

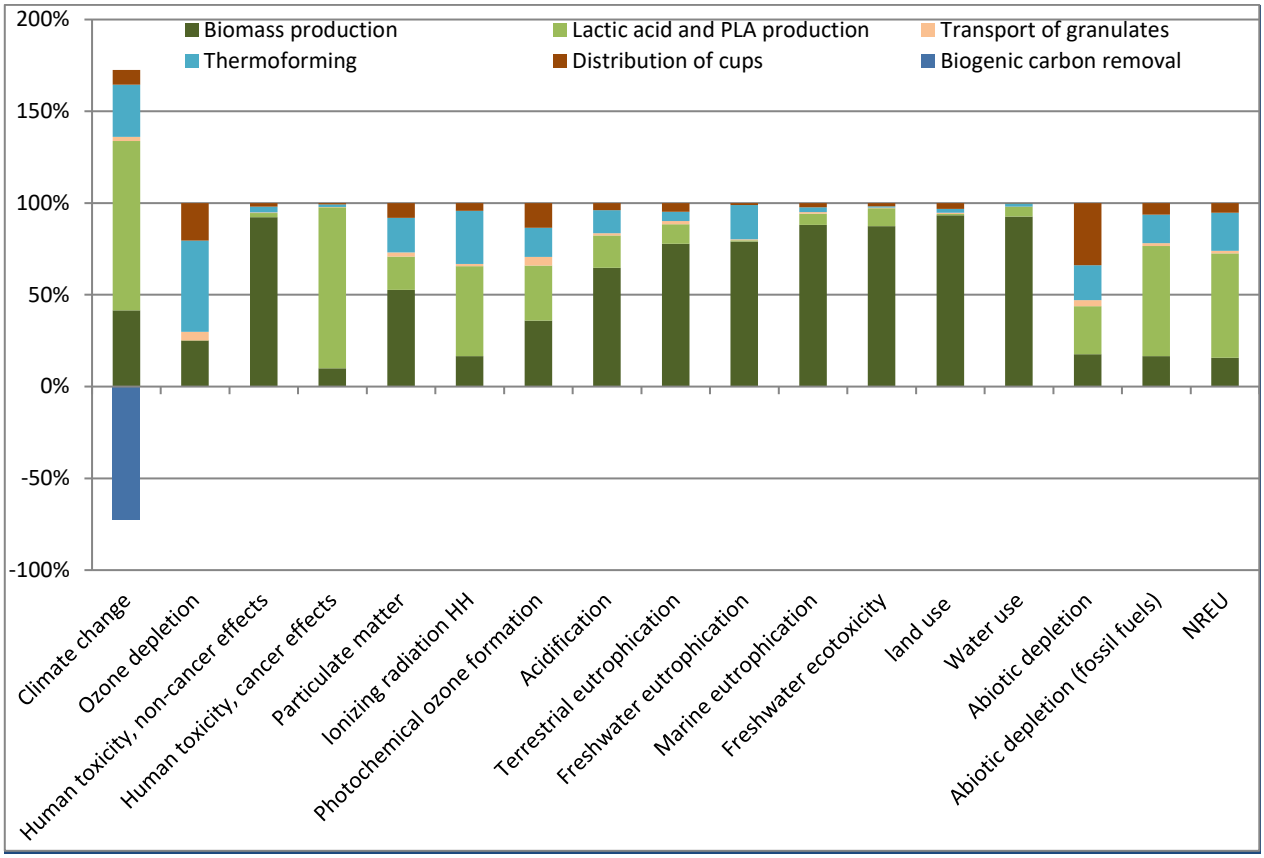


Figure 87. Breakdown of cradle-to-user environmental impacts of bio-based alternative system 1 (Bio-based PLA from European maize), 16 PEFCR impact categories + NREU.

The impact categories showing major differences (e.g. more than ±30 % changes in impacts) between this alternative route and the baseline route in Table 107 are further interpreted below.

- **Climate change.** The cradle-to-user GHG emissions are assessed to be 11.55 kg CO_2 eq. on a functional unit basis which is about 34 % higher compared to the baseline. The climate change impact potentials of this product system come from the European maize cultivation (24 %), the conversion of maize into PLA (53 %), the transportation of the pellets (1 %), the manufacturing of the cups (18 %) and from the final distribution (5 %). The impact of the lactic acid and PLA production unit process in terms of climate change does not vary significantly (less than 2 %). In this case, what is much higher is the impact of the biomass production because of the more carbon intensive maize production (0.98 kg CO_2 eq./kg PLA for average European maize) compared to the average maize of Nebraska and Iowa- Thai sugarcane (0.29 kg CO_2 eq./kg PLA). This is caused by the more carbon intensive maize production (0.63 kg CO_2 eq. for average European maize) compared to maize cultivated in Nebraska and Iowa (0.17 kg CO_2 eq. /kg maize (based on Vink et al., 2015)). Among the EU modelled countries, the ones with the most carbon intensive maize production are Romania (0.91 kg CO_2 eq/kg maize) and Poland (0.75 kg CO_2 eq./kg maize). These last two countries account for 20.3 % and 8.2 % respectively of the European maize production which is modelled: in terms of climate change impact, they represent 26 % and 8 % respectively of the total. The carbon intensities of maize production in other EU countries like Germany and France are approximately half of those of Romania, and their carbon intensities are similar to the average maize cultivated in US. Moreover, the maize from Iowa/Nebraska

which is used by NatureWorks for PLA production has an impact which is 30 % lower on climate change compared to average cultivation in the US. In terms of emissions, the maize cultivation in Poland and Romania produces 2-3 times more carbon dioxide of fossil origins (about 60 % of the climate change impact) and 2-4 times more emissions of methane (about 6 % of the climate change impact) compared to France. The emissions of dinitrogen monoxides (20-40 % of the impact of climate change) are instead released in more similar quantities. The carbon dioxide emissions of fossil origins come from diesel burnt in agricultural machineries (70-80 %); emissions of methane come from the production of the electricity used (40-60 %) and from the diesel burnt in machinery (20-30 %).

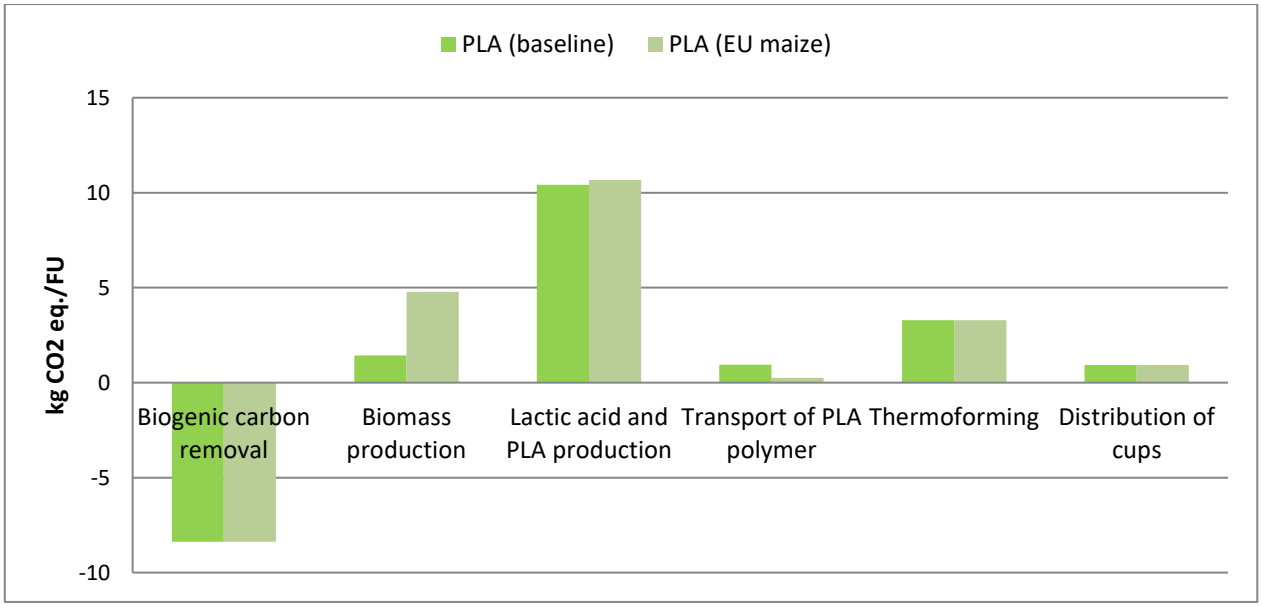


Figure 88. Cradle-to-user breakdown of climate change impact potentials for PLA (EU maize) cups on functional unit basis and comparison with the baseline product system.

- **Acidification.** Acidification per PLA cup is in total 41 % higher when PLA is produced from European maize instead of from the mix of Nebraska/Iowa maize and Thai sugarcane. This is mainly because the acidification from European maize cultivation is 79 % higher compared to the Nebraska/Iowa maize used for PLA production (baseline). The acidification caused by European maize is caused by emissions of ammonia (70 %), emissions of sulphur dioxide (18 %) and nitrogen oxides (8 %). The emissions of ammonia are mainly direct emissions due to the use of fertilisers (65 %) and manure (25 %) while the emissions of sulphur dioxide are mainly indirect emissions deriving from the use of electricity (50 %) and diesel burnt in machinery (30 %).

- **Terrestrial eutrophication.** For the alternative system 1, the feedstock cultivation is responsible for 77 % of the impact, while the conversion of maize into PLA accounts for 11 %. Compared to the baseline, terrestrial eutrophication increases by 80 % per cup, despite an 85 % reduction in the impact caused by the transportation of the polymer. The main reason behind is that the impact of the biomass production increases by three times. The cause is twofold:

    - In this case study, the biomass feedstock is only maize while in the baseline case it is a mix of maize (67 %) and sugarcane (33 %). The cultivation of the amount of sugarcane needed for making 1 kg of PLA had half the impact

compared to the cultivation of the amount of US maize needed for making 1 kg of PLA.

- The impact caused by European cultivation of maize is 70 % higher compared to Nebraska/Iowa maize cultivation. The terrestrial eutrophication of the European maize is due to emissions of ammonia (84 %) and emissions of nitrogen oxides (12 %).

- **Marine eutrophication.** Marine eutrophication increases by 100 % when PLA is produced from European maize compared to import it from US. This is because the impact of the feedstock cultivation increases by 1.6 times. Marine eutrophication caused by European maize cultivation is due to emissions of nitrate to water (90 %) which is caused by fertilisers (60 %), nitrate emissions from crop residues (30 %) and use of manure (7 %).

- **Water use.** In the alternative system 1, the total water use is about six times higher compared to the baseline. More than 90 % of the water needed is required by the cultivation step for this product system. The water use from the biomass cultivation (maize from the US and sugarcane from Thailand) in the baseline analysis is 2.9 m^3 per functional unit whereas in this alternative route the EU mixed maize requires 39 m^3. There are two reasons for this significant increase in water use. Firstly, the PLA is produced only from maize and not from the mixed maize/sugarcane; the Thai sugarcane needed per kg of PLA requires 60 % less water compared to the amount of Nebraska/Iowa maize needed for 1 kg of PLA. Secondly, average European maize requires 10 times more water compared to maize cultivated in Nebraska/Iowa. It should be noticed that the water use accounted for with the characterization factors of the AWARE methodology for maize production varies widely in Europe. For example, the water use impact of maize cultivation is 0.02 m3 in Germany and 31.7 m3 in Spain per kg of maize, although the absolute water withdrawals for irrigation in these two countries are not so substantial because in Spain water is a more constrained resource than in Germany.

- **Land use.** Ninety-three percent of the land use impact is caused by the feedstock production (European maize). For European maize the land use carries a carbon deficit ranging between 8.9 (Spain) and 24.1 (Romania) kg C deficit per kg of maize: the land use impact varies widely depending on the country where the maize is produced. For the European average maize, it results in 14.0 kg C deficit per kg of maize. The land use for the PLA of the alternative system 1 is 25 % higher compared to the PLA of the baseline product system. The main reason behind this is that Nebraska/Iowa maize has a 23 % lower carbon deficit compared to the average European maize.

- **Abiotic depletion.** Abiotic depletion for PLA produced under the conditions of the alternative product system 1 decreases by 34 % compared to the baseline PLA. This is due to the fact that the abiotic depletion of the lactic acid to PLA production unit process decreases by 66 %, due to the absence of the CHP burning bagasse. Nevertheless, there is an increase of 20 % in the impact related to biomass cultivation. The abiotic depletion due to European maize cultivation is caused by the consumption of lead (90 %) which is caused by the diesel used in the machinery (93 %).

3.1.3 Cradle-to-user environmental impacts of Bio-Based alternative system 2: Bio-based PP

Table 108 shows the cradle-to-user environmental impacts of bio-based PP cups made from used cooking oil (UCO). The results are compared with the baseline PLA cups on a functional unit basis. It can be seen that bio-based PP offers substantial impact reduction in nearly all 16 impact categories (the exception is ozone layer depletion, for which the impact decrease is

minor). The cause is twofold. Firstly, the PP is a lighter material compared to PLA. The weight of PP cups per functional unit is about 18 % lower than the weight of PLA cups. This leads to lower impacts that are directly associated with mass, e.g., transportation and thermoforming. This can also be seen in Figure 89.

Table 108. Cradle-to-user environmental impacts (16 PEFCR categories + NREU) of per *functional unit* bio-based PP cups made from used cooking oil, compared with the results of the baseline PLA cups. Functional unit: bio-based PP cups: 3.88 kg, PLA cups: 4.59 kg.

Impact category	Unit	bio-based PP	PLA baseline	Increase (+) or decrease (-) in impacts
Climate change	kg CO_2 eq.	-5.04	8.63	-154 %
Ozone depletion	kg CFC-11eq	9.91E-07	1.03E-06	-2 %
Human toxicity, non-cancer effects	CTUh	4.11E-07	3.35E-06	-87 %
Human toxicity, cancer effects	CTUh	5.02E-08	8.81E-07	-94 %
Particulate matter	kg PM2.5eq	2.40E-03	0.0065	-62 %
Ionizing radiation HH	kBq U235eq	0.75	1.22	-38 %
Photochemical ozone formation	kg NMVOCeq	2.01E-02	0.061	-65 %
Acidification	molc H+eq	3.35E-02	0.10	-65 %
Terrestrial eutrophication	molc Neq	6.62E-02	0.25	-72 %
Freshwater eutrophication	kg Peq	3.60E-04	0.001	-62 %
Marine eutrophication	kg Neq	6.02E-03	0.040	-84 %
Freshwater ecotoxicity	CTUe	3.71	71.0	-94 %
Land use	kg C deficit	10.17	92.0	-89 %
Water use	m3	3.73	6.07	-36 %
Abiotic depletion	kg Sbeq	5.95E-06	1.55E-05	-61 %
Abiotic depletion (fossil fuels)	MJ	89.47	212	-50 %
NREU	MJ	119.94	258	-47 %

Secondly, the impacts from the production of the material, bio-based PP, represent a relatively small share compared to the share of the impacts from the production of lactic acid and PLA in the baseline. The collection of used cooking oil has an almost negligible contribution (see Figure 89). The relatively high contribution of the production of bio-based PP to water use (see Figure 89) is caused mainly by the electricity consumption during the polymerisation step, which is similar, in absolute terms, to the polymerisation of PLA and petrochemical PP.

The conversion of UCO into PP is also not energy or carbon intensive compared to the production of lactic acid and PLA. Comparing the results of these two technologies, it can be seen that if a thermochemical conversion is carried out efficiently, it offers much lower impacts compared to a biochemical process (e.g. fermentation) in which a large amount of water needs to be used in the system.

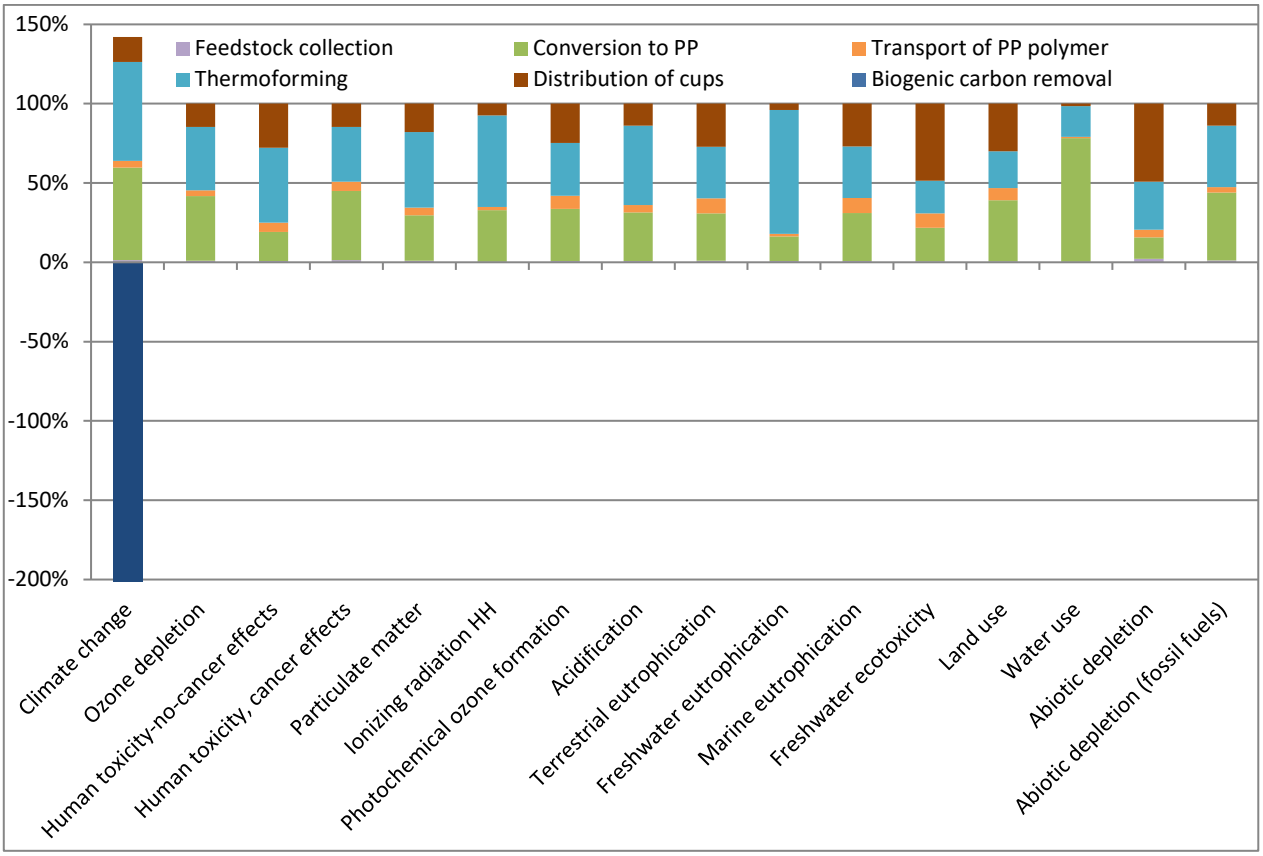


Figure 89. Breakdown of cradle-to-user environmental impacts of bio-based alternative 2 (Bio-based PP cups made from UCO), results for 16 PEFCR categories + NREU.

With a deeper level of detail and for the main impact categories, the main causes of environmental impact are here detailed:

- **Climate change.** The total climate change is caused by two main life cycle stages: the conversion of UCO into PP (44 %) and the thermoforming process (42 %). The rest of the climate change impact is caused mainly by the distribution of the cups (10 %). Regarding the conversion into PP, the impact is caused by the steam cracking (51 %), by polymerisation (23 %) and by hydrotreatment (including transportation of bio-naphta) (21 %). The impact of steam cracking is caused primarily (85 %) by emissions of carbon dioxide of fossil origins (these are made up of 30 % direct emissions, 54 % indirect emissions for the generation of the steam used, and 8 % indirect emissions due to LPG production). The climate change impact of hydrotreatment is caused primarily (86 %) by the production of the hydrogen used. The impact of thermoforming is caused by the production of the electricity used (93 %).

- **Particulate matter.** The emissions of particulate matter are caused by thermoforming (46 %), conversion into PP (31 %) and distribution of cups (17 %). The impact of thermoforming comes primarily (97 %) from the electricity used. The impact of the conversion stage is distributed as follows: 49 % caused by steam cracking, 28 % by polymerisation and 15 % by hydrotreatment (including transportation of naphtha). The impact of hydrotreatment comes from the production of the hydrogen used (65 %) and fro the transportation of the naphtha to the stream cracking unit (22 %).

- **Photochemical ozone formation.** The impact in terms of photochemical ozone formation is divided as follows: 36 % caused by the conversion into PP, 31 % by thermoforming and 23 % by the distribution of the cups. The impact of the conversion process is caused by the polymerisation step (37 %), by hydrotreatment (28 %) and another 28 % from steam cracking. The impact of hydrotreatment due primarily to the production of the hydrogen used (86 %). For the steam cracking, 57 % of the photochemical ozone formation is a consequence of the production of the steam used, while the 42 % comes from the production of the LPG.

- **Acidification.** The contribution to the total acidification impact of each life cycle stage is as follows: 48 % from thermoforming, 32 % from conversion into PP, 13 from % distrubution of cups, 5 % from transport of PP polymer and 1 % from the UCO collection. The majority (98 %) of the acidification caused by thermoforming is a consequence of the production of the electricity used. The impact of the conversion stage is mainly due to three sub-unit processes: steam cracking (39 %), polymerisation (35 %) and hydrotreatment (20 %). The impact of steam cracking is caused by the production of the steam used (69 %) and by the production of the LPG used (31 %). The impact of hydrotreatment is caused mainly by the production of the hydrogen used (81 %).

- **Terrestrial eutrophication.** The life cycle stages that contribute the most are the conversion into PP (32 %), the thermoforming (31 %) and the distribution of the cups (26 %). The transportation of the PP polymer represents a share of 9 % while the UCO collection accounts for less than 1 %. Among the processes which belong to the conversion stage, the main contributors are: hydrotreatment (37 %), polymerisation (30 %) and steam cracking (24 %). The majority (87 %) of the impact related to hydrotreatment is due to the production of the hydrogen used. The impact of the steam cracking is caused by the production of steam (63 %) and by production of LPG burnt (37 %).

- **Marine eutrophication.** The shares of the life cycle impact breakdown for this impact category are very similar to those related to terrestrial eutrophication. The life cycle stages that contribute the most are the conversion into PP (33 %), the thermoforming (31 %) and the distribution of the cups (26 %). The impact of the conversion stage is structured as follows: 37 % from hydrotreatment (86 % due to the production of the hydrogen used), 24 % from steam cracking (63 % due to steam production and 37 % due to LPG production) and 31 % from polymerisation. The majority (97 %) of the impact of thermoforming is due to the production of the electricity used.

- **Land use.** Land use impact is caused by the conversion to PP (40 %), by the distribution of cups (29 %), by thermoforming (23 %) and by the transportation of the PP polymer (7 %). The land use of the conversion step comes almost entirely (82 %) from the steam cracking unit process and this is caused by LPG production (71 %) and by steam production (29 %). The majority (98 %) of the impact of thermoforming is a consequence of the production of the electricity used.

- **Water use.** Two life cycle stages cover almost the total water use: the conversion to PP (79 %) and the thermoforming process (19 %). The polymerisation step is entirely responsible for the water use (almost 100 %) of the conversion step. The water use of thermoforming is primarily caused by the electricity production (97 %).

- **Abiotic depletion.** Abiotic depletion is mainly caused by the distribution of the cups (48 %), thermoforming (30 %) and the conversion into PP (15 %). The abiotic depletion of the distribution step is mainly (83 %) caused by the transportation from the distribution centre to the final user where a lorry with a small capacity is used. The

electricity production accounts for 72 % of the impact of thermoforming. For the conversion step, 39 % of the abiotic depletion is generated by polymerisation, 22 % by steam cracking (44 % due to steam production and 66 % by LPG production) and 18 % by the pre-treatment.

- **Abiotic depletion-fossil fuels.** The breakdown for this impact category is structured as follows: 51 % from conversion into PP, 33 % from thermoforming and 12 % from distribution of the cups. The steam cracking (58 %) and polymerisation (27 %) processes are the two main contributors of non-renewable resources used during the conversion stage while the hydrotreatment accounts for another 11 %. The impact of the steam cracking is due to steam production (52 %) and LPG production (47 %). The majority (98 %) of the abiotic depletion (fossil fuels) of the thermoforming stage is due to the electricity production.

3.2 Cradle-to-user results of petrochemical reference systems

3.2.1 Reference 1 petrochemical PP

With the same level of breakdown showed for PLA cups, in this section the results for PP cups are collected (see Table 109). In Figure 90 it is possible to identify the impact of each stage on the life cycle. Based on Chapter 6 *Ranges for environmental impacts from production of fossil-based plastics*, 7 out of 17 impact categories were found suitable for petrochemical PP interpretation, namely: climate change, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication, abiotic depletion (fossil fuels) and NREU. The main reasons for the exclusion of most of the assessed impact categories of the PlasticsEurope ecoprofile for PP are due to large differences identified among the impact assessment results provided by the main LCA databases, non-compliance with several ILCD requirements, and a lack of transparency in the allocation approach. Moreover, impact categories like marine and freshwater ecotoxicity are considered not reliable because there is no distinction within the PlasticsEurope dataset between emissions to fresh water and to seawater. Further details can be found in Table 35 where the environmental impact of 1 kg of PP and the proposed ranges of variation within each impact category are shown.

Table 109. Cradle-to-user environmental impacts (16 PEFCR categories + NREU) of per functional unit for petrochemical PP cups.

Impact category	Unit	Petrochemical PP (ref.1)
Climate change	kg CO_2 eq.	10.95
Particulate matter	kg PM2.5 eq.	3.07E-03
Photochemical ozone formation	kg NMVOC eq.	3.59E-02
Acidification	molc H+ eq.	4.41E-02
Terrestrial eutrophication	molc N eq.	9.69E-02
Abiotic depletion (fossil fuels)	MJ	353.86
NREU	MJ	386.16

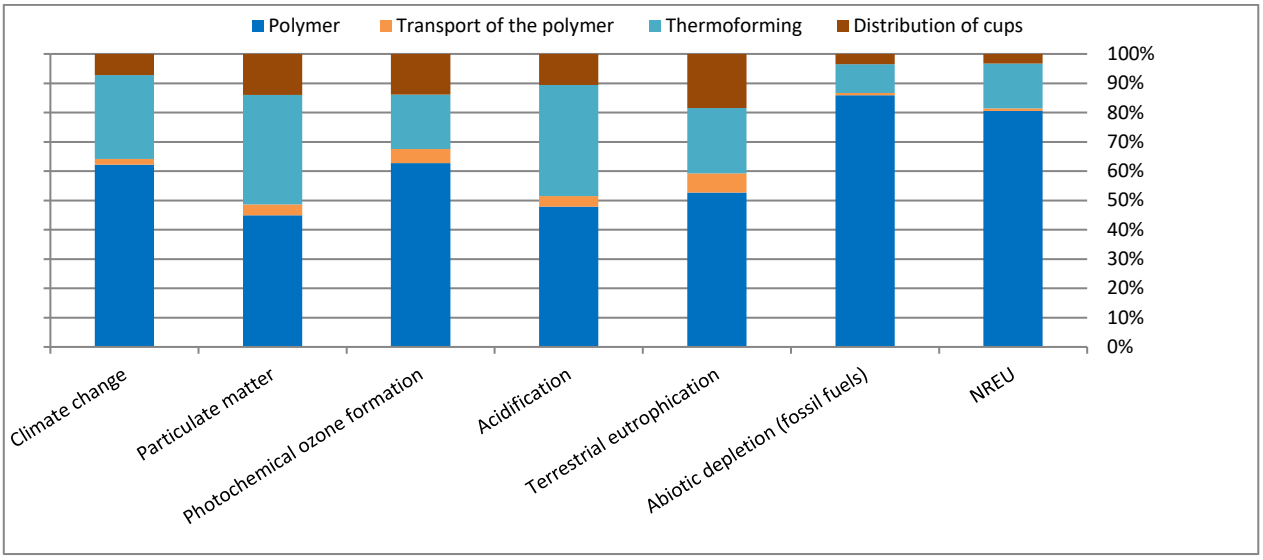


Figure 90. Breakdown of the cradle-to-user results for reference system 1 (PP cups), 16 PEFCR impact categories + NREU.

It is possible to notice that for the impact categories recommended for comparison:

- The production of PP granulates has the highest impact in terms of climate change, photochemical ozone formation, terrestrial eutrophication, marine eutrophication, water use, abiotic depletion (fossil fuels) (or NREU). Its impact is also very relevant for particulate matter and acidification.
- The transport of polymer granulates, assumed to occur inside Europe, does not have an important contribution.
- Thermoforming impacts significantly in all the impact categories therefore it can be perceived as the second source of environmental impact of PP cups.
- The distribution of PP cups also has a non-negligible impact, with the exception of abiotic depletion (fossil fuels) and NREU.

3.2.2 Reference 2 Petrochemical PET

Figure 90 above summarises the environmental impacts of PET cups, breaking them down into the four macro-stages, namely, material production, transport of granulates, thermoforming and distribution of cups. Table 110 summarises the cradle-to-user results per functional unit for this product system. Based on Chapter 6 *Ranges for environmental impacts from production of fossil-based plastics*, 9 out of 17 impact categories were found suitable for petrochemical PET interpretation, namely: climate change, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication, marine eutrophication, water use, abiotic depletion (fossil fuels) and NREU. This is because comparison of different datasets revealed large differences for the other impact categories. The data used is based on PlasticsEurope since it provides the most up-to-date eco-profile for European fossil-based plastics. However, it does not conform with ILCD requirements in its presentation of inventory details or impact assessment results. It does not distinguish between emissions to fresh water and to seawater, and it is therefore not possible to calculate the impacts on human health and ecosystems as well as fresh water eutrophication, because the receiving compartment is an important element in determining the fate and effect of an emission. Further details can be found in Table 40 where the environmental impact of 1 kg of PET and the proposed ranges of variation within each impact category are shown.

Table 110. Cradle-to-user environmental impacts (16 PEFCR categories + NREU) of per functional unit for petrochemical PET cups.

Impact category	Unit	Petrochemical PET (ref.2)
Climate change	kg CO_2 eq.	16.01
Particulate matter	kg PM2.5 eq.	6.61E-03
Photochemical ozone formation	kg NMVOCeq.	5.29E-02
Acidification	molc H+eq.	7.84E-02
Terrestrial eutrophication	molc Neq.	1.48E-01
Marine eutrophication	kg Neq.	1.39E-02
Water use	m^3	18.02
Abiotic depletion (fossil fuels)	MJ	431.36
NREU	MJ	463.58

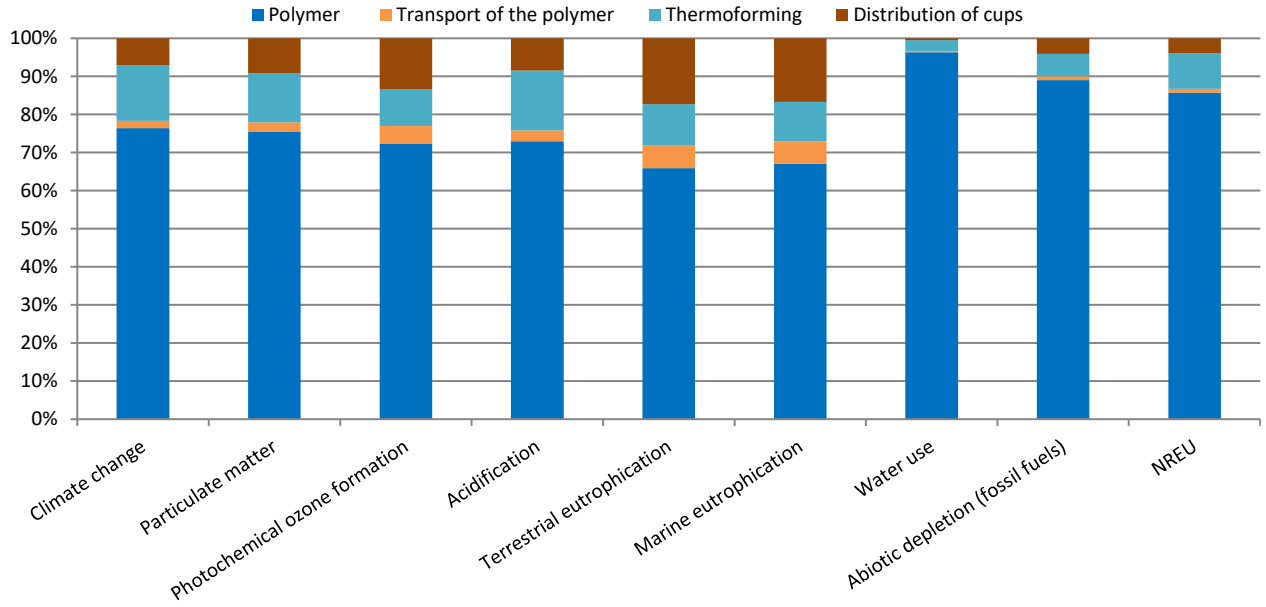


Figure 91. Breakdown of the cradle-to-user results for reference 2 (PET).

As it is possible to notice from Figure 91, the majority of all impacts comes from the production of the PET granulates. This covers more than the 55 % of the environmental impact in all the impact categories. The impact of the material is more than 90 % when referring to ozone depletion. The transport of granulates is negligible while thermoforming and distribution of cups present approximately similar shares in each impact category. Notably, the distribution of cups together with thermoforming represents 25 % of the climate change impact.

3.3 Results of end of life of single use cups

The results for the EoL impact assessment are presented in this paragraph. Firstly, the mass and energy flows of all product systems in all technologies are presented, secondly the LCIA results for each product system, thirdly a comparison between product systems, and lastly the EoL mix. In Annex 4 further results are presented, the contribution analysis is given in Figure 236 ff. and weighted results in Table 225 ff..

3.3.1 Mass and energy flows

The mass and energy flows of EoL waste management for the four product systems covered in this case study (PLA, PP and PET) are presented in Table 111 to Table 114. The results correspond to the circular footprint formula in PEFCR guidance draft version 6.3 (European Commission, 2017), and in addition the section looks at the carbon flows. The flows correspond to the flows observed in the flow charts for each technology in the approach and methodology chapter of the full report (section 3.4 Average End of Life technologies). The following refers to the EoL-reference flow, i.e. 1 kg plastic and 100 g contamination.

It is observed that when recycling, 57 % of the material input is substituted, for all product systems (calculated based on sorting and technology efficiency as well as a market response, considering the decrease in quality of the plastic - see the methodology for further explanation). For the biological treatments of PLA respectively 94 % (1.04 kg) and 70 % (0.77 kg) of the input waste goes into the composting and anaerobic digestion plants. The remainder of the input is rejected in the pre-sorting facility and sent for incineration. The output is 0.22 kg compost in the industrial composting. For anaerobic digestion it is 0.085 kg compost and 6.9 kg digestate which is utilised in agricultural fields as a substitute for fertilisers. Composting and anaerobically digesting the PLA recovers respectively 30 % and 38 % of the nutrients (in terms of NPK fertilisers).

Regarding energy, 31 % of the energy content of the EoL-reference flow is utilised in the incineration for all product systems (corresponding to the energy recovery rates for heat and electricity, respectively 22 % and 9 %). When landfilling, 5 % of the energy content is recovered from the PLA, while none of the energy is recovered for the PET and bio-based PP. This is due to the fact that the non-biodegradable plastics do not create landfill gas which can be recovered for energy production. Incineration of residues from the composting process and the recycling process obtains 9 % of the energy content of the waste input. Anaerobic digestion of PLA recovers only 4 % of the energy content of PLA (electricity, heat and fuel), and in addition 9 % of the energy from the incineration of rejects from the AD-plant, hence in total 13 % of the energy is recovered.

Looking at the carbon balance when landfilling cups, 58 % of the biogenic carbon and 100 % of the fossil carbon (from contamination) is stored in the landfill for the PLA product system, while 100 % of the carbon is stored in the landfill for the other two product systems. In the biological treatments, respectively 32 % and 30 % of the carbon is utilised for use on land, in the industrial composting scenario and the anaerobic digestion scenario.

Table 111. Material and energy flow for EoL of 1 kg PLA with 100 g contamination. Box letter refer to flow charts in the approach and methodology chapter (Figure 8 to Figure 13).

PLA								
Technology	**Box**	**Process**	**Mass kg**	**Bio carbon kg**	**Fossil carbon kg**	**Gas m^3 CH_4**	**Energy substitution MJ**	**Fertiliser substitution kg NPK**
Material	I	Material input	1.10E+00	5.10E-01	1.79E-04		1.95E+01	1.65E-03
Recycling	O	Recycled plastic	7.70E-01	3.93E-01	0.00E+00			
	S	Substituted plastic	6.24E-01	3.18E-01	0.00E+00			
	R, E, H	Rejects	3.30E-01	1.17E-01	1.79E-04		1.76E+00	
Incineration w/wo energy recovery *	E, H	Energy production	1.10E+00	5.10E-01	1.79E-04		5.88E+00	
	R1	Fly Ash	5.23E-04	0.00E+00	0.00E+00			
	R2	Bottom ash	2.51E-03	5.10E-04	1.79E-07			
	D	Direct emissions		5.10E-01	1.79E-04			
Landfill	L	Leachate	1.40E+00	0.00E+00	0.00E+00			
	G, E	Landfill gas		2.12E-01		2.00E-01	1.03E+00	
	CS	Storage in landfill	6.99E-01	2.98E-01	1.79E-04			
Compost	O, S	Compost	2.27E-01	1.63E-01	1.70E-04			5.02E-04
	R, E, H	Rejects	6.26E-02	6.38E-02	2.66E-06		1.72E+00	
Anaerobic digestion	B, E1, H1, F	Biogas		2.20E-02		2.59E-02	8.73E-01	
	C, S	Compost	8.50E-02	2.34E-02	1.93E-05			7.37E-05
	D, S	Digestate	6.90E+00	2.76E-01	1.06E-04			5.56E-04
	R, E2, H2	Rejects	3.30E-01	1.53E-01	5.36E-05		1.76E+00	

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

Table 112. Material and energy flow for EoL of 1 kg bio-based PP with 100 g contamination. Box letter refer to flow charts in the approach and methodology chapter.

Bio-based PP						
Techno-logy	**Box**	**Process**	**Mass kg**	**Bio carbon kg**	**Gas m^3 CH_4**	**Energy substitution MJ**
Material	I	Material input	1.10E+00	8.26E-01		4.03E+01
Recycling	O	Recycled plastic	7.70E-01	5.78E-01		
	S	Substituted plastic	6.24E-01	4.68E-01		
	R, E, H	Rejects	3.30E-01	1.90E-03		3.74E+00
Incineration w/wo energy recovery *	E, H	Energy production	1.10E+00	8.26E-01		1.25E+01
	R1	Fly Ash	1.66E-02	0.00E+00		
	R2	Bottom ash	4.83E-02	8.26E-04		
	D	Direct emissions		8.25E-01		
Landfill	L	Leachate	2.20E+00	0.00E+00		
	G, E	Landfill gas			0.00E+00	0.00E+00
	CS	Storage in landfill	1.10E+00	8.26E-01		

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

Table 113. Material and energy flow for EoL of 1 kg petrochemical PP with 100 g contamination. Box letter refer to flow charts in the approach and methodology chapter.

Petrochemical PP						
Technology	Box	Process	Mass kg	Fossil carbon kg	Gas m^3 CH_4	Energy substitution MJ
Material	I	Material input	1.10E+00	8.26E-01		4.03E+01
Recycling	O	Recycled plastic	7.70E-01	5.78E-01		
	S	Substituted plastic	6.24E-01	4.68E-01		
	R, E, H	Rejects	3.30E-01	1.90E-03		3.74E+00
Incineration w/wo energy recovery *	E, H	Energy production	1.10E+00	8.26E-01		1.25E+01
	R1	Fly Ash	1.66E-02	0.00E+00		
	R2	Bottom ash	4.83E-02	8.26E-04		
	D	Direct emissions		8.25E-01		
Landfill	L	Leachate	2.20E+00	0.00E+00		
	G, E	Landfill gas			0.00E+00	0.00E+00

	CS	Storage in landfill	1.10E+00	8.26E-01		

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

Table 114. Material and energy flow for EoL of 1 kg petrochemical PET with 100 g contamination. Box letter refer to flow charts in the approach and methodology chapter (Figure 8 to Figure 13).

Petrochemical PET						
Technology	**Box**	**Process**	**Mass kg**	**Fossil carbon kg**	**Gas m^3 CH_4**	**Energy substitution MJ**
Material	I	Material input	1.10E+00	6.81E-01		2.44E+01
Recycling	O	Recycled plastic	7.70E-01	4.77E-01		
	S	Substituted plastic	6.24E-01	3.86E-01		
	R, E, H	Rejects	3.30E-01	1.57E-03		2.26E+00
Incineration w/wo energy recovery *	E, H	Energy production	1.10E+00	6.81E-01		7.53E+00
	R1	Fly Ash	2.85E-03	0.00E+00		
	R2	Bottom ash	9.17E-03	6.81E-04		
	D	Direct emissions		6.80E-01		
Landfill	L	Leachate	2.20E+00	0.00E+00		
	G, E	Landfill gas			0.00E+00	0.00E+00
	CS	Storage in landfill	1.10E+00	6.81E-01		

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

3.3.2 Baseline system and alternative 1: Bio-Based PLA

The impact assessment results for all the EoL technologies are presented in Table 115 for the PLA product system. These results are given per the EoL-reference flow, i.e. 1 kg plastic and 100 g of contamination for each EoL technologies (assuming 100 % disposal via that technology).

Recycling has the highest number of impact categories with the lowest impacts for the bio-based PLA cups. There is no clear second ranking for the EoL options of the PLA cups on the characterised results. The weighted results show, both with and without toxicity, that the recycling technology is the most favourable choice, followed by industrial composting, anaerobic digestion, incineration and lastly landfilling (see Figure 92).

Further details on the individual EoL options for the bio-based PLA cups are presented below, looking into specific impact categories and the contribution of impacts from different processes and elemental emissions.

Table 115. Total results of treating 1 kg PLA (incl. 100 g impurities) for all impact categories. Highest impact within each category is red and lowest impact green. The intended EoL option is marked with a bold box.

Impact category (Unit)	Bio-based PLA				
	Recycling	Incineration	Landfilling	Industrial composting	Anaerobic digestion
Climate change – fossil (kg CO_2 eq.)	-1.06E+00	-2.94E-01	3.35E-03	-4.45E-02	-1.75E-01
Climate change - biogenic (kg CO_2 eq.)	4.96E-01	1.82E+00	3.09E+00	1.58E-01	6.32E-01
Climate change - total (kg CO_2 eq.)	-5.62E-01	1.52E+00	3.10E+00	1.13E-01	4.58E-01
Ozone depletion (kg CFC-11 eq.)	-6.35E-06	-2.88E-08	1.82E-06	-6.15E-09	-1.09E-08
Human toxicity, non-cancer effects (CTUh)	-6.53E-08	-7.97E-09	-1.06E-09	3.80E-08	4.64E-08
Human toxicity, cancer effects (CTUh)	-6.17E-09	-1.23E-09	-3.01E-10	-2.38E-09	-2.13E-09
Particulate matter (kg PM2.5 eq.)	-2.20E-04	-4.38E-05	-2.38E-05	-1.04E-06	-1.15E-05
Ionizing radiation HH (kBq U235 eq.)	-1.59E-02	4.05E-02	-3.21E-03	1.56E-02	1.64E-02
Photochemical ozone formation (kg NMVOC eq.)	-2.95E-03	8.70E-04	9.20E-04	2.53E-04	6.20E-04
Acidification (molc H+ eq.)	-3.14E-03	1.43E-03	-4.05E-05	6.96E-04	1.05E-03
Terrestrial eutrophication (molc N eq.)	-5.59E-03	5.29E-03	3.65E-05	2.27E-03	4.18E-03
Freshwater eutrophication (kg P eq.)	-5.36E-05	7.29E-05	-3.11E-06	1.92E-05	2.26E-05
Marine eutrophication (kg N eq.)	-5.65E-04	4.24E-04	3.30E-05	4.76E-04	4.60E-04
Freshwater ecotoxicity (CTUe)	-2.30E+00	-1.01E-01	-3.77E-03	-2.36E-02	-1.49E-02
Land use (kg C deficit)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use (m^3)	-2.55E-01	-1.42E-01	-5.71E-02	5.19E-02	4.00E-02
Abiotic depletion (kg Sb eq.)	-5.20E-08	-1.96E-07	-4.68E-08	-5.42E-08	-1.22E-07
Abiotic depletion (fossil fuels) (MJ)	-3.45E+01	-5.48E+00	-3.73E-01	-9.96E-01	-8.49E-01
NREU (MJ)	-3.40E+01	-6.31E+00	-2.31E-01	-1.45E+00	-1.24E+00

The significance of the colour scale is the same as for all the other product systems (Table 116 and Table 117). The colors can be compared horizontally across all tables, but only comparing between the same impact category.

3.3.2.1 Intended EoL technology: Plastic recycling of PLA

Based on the LCIA results presented in Table 115, recycling has the lowest impact for PLA in all the impact categories, except for abiotic depletion, compared to the other EoL technologies.

The largest savings in all impact categories in the recycling technology are from substitution of PET, which overall also has the largest contribution to all impact categories (absolute percentage). The indirect emissions from the recycling plant have the highest impact in most categories, except in the impact category of climate change where the largest emission of CO_2-eq. comes from the treatment of rejects, followed by indirect emissions from the recycling plant. For human toxicity (non-cancer effects) and marine eutrophication, transportation contributes most to the impacts on recycling PLA. For ionizing radiation heat substitution also has a major impact.

3.3.2.2 Intended EoL technology: Industrial Composting of PLA

Based solely on the climate change impact, the second most preferred EoL technology for PLA, is industrial composting with an impact of 0.113 kg CO_2-eq. Industrial composting however does not perform best in any impact categories.

The use of compost contributes most to the climate change impact, but the impact almost balances out with the fertiliser substitution. Industrial composting furthermore performs second best in human toxicity (cancer effects) and photochemical ozone formation, where respectively fertiliser substitution and emission from the composting plant have the largest contribution (in absolute figures). The EoL technology performs worst in three impact categories. The contribution analysis for the intended EoL technology, industrial composting, for the PLA food packaging film shows that the contribution to the LCIA results is quite evenly distributed between the processes depending on the impact category. The direct emissions from the composting facility contribute to a large share in the three eutrophication impact categories as well as acidification, petrochemical ozone formation and climate change fossil. The heat substitution from incinerating rejects contributes to large savings in the impact categories of NREU, abiotic depletion (fossil fuels) and ozone depletion, but has a large impact in the category ionizing radiation. The use on land of the compost leads to a large contribution to the impact categories of biogenic climate change, human toxicity (non-cancer) and freshwater ecotoxicity. Fertiliser substitution contributes to large savings in human toxicity (cancer effects), biogenic climate change and freshwater ecotoxicity.

3.3.2.3 Landfilling of PLA

Landfilling shows high impacts in many impact categories compared with the other EoL technologies. In terms of climate change impact landfilling performs worst with CO_2 emissions of 3.1 kg CO_2 eq. due to landfill gas combustion (flaring), direct emissions of methane from the gas upgrading, and direct greenhouse gas emissions to the environment. The direct emissions from landfill gas combustion contributes to a large share of the biogenic climate change, the total climate change, ozone depletion and photochemical ozone formation. In the same four categories, the landfill gas that is not collected and oxidised in covers also contributes a large share. In all other impact categories, it is the electricity substitution that contributes the largest share ranging from 27 % to 95 % of the absolute impact.

3.3.2.4 Anaerobic digestion of PLA

Looking into the contribution of the processes for the EoL technology anaerobic digestion, the contribution is evenly distributed between several processes. The substitution of heat and electricity from the incineration of rejects contributes to a large share in the categories NREU, abiotic depletion (fossil fuels), ionising radiation, particulate matter and ozone depletion. The total energy recovery (i.e. additionally covering the electricity, heat and fuel produced from the biogas) contributes from 59 % to 94 % of the absolute impacts of these impact categories. The fertiliser substitution gives high savings in human toxicity cancer effects and climate change - biogenic. Treatment of rejects and use on land of compost contribute the most to biogenic climate change impacts, human toxicity (non-cancer) and freshwater ecotoxicity.

Anaerobic digestion performs worst in photochemical ozone formation due to emissions (NO_X, CH_4, CO, MNVOC) from the digestate and the post-composting of the compost.

3.3.2.5 Incineration of PLA

Incineration performs best in abiotic depletion because of savings in electricity substitution (78 % of the absolute impact). Incineration performs second best in seven impact categories, but worst in four impact categories. Heat substitution both acts as an impact and a saving depending on the impact category. The largest contributor to impacts from incineration of PLA single-use cups are the direct emissions from the incineration, especially in biogenic climate change, photochemical ozone formation, acidification, terrestrial and marine eutrophication. The largest savings in the impacts categories of water use (68 % of absolute impact), human toxicity (64 and 66 % of absolute impact), particulate matter (65 % of absolute impact) and abiotic depletion (78 % of absolute impact) come from electricity substitution. From heat substitution the contribution analysis shows that the largest savings are in climate change – fossil (63 % of absolute impact), freshwater ecotoxicity (56 % of absolute impact), NREU (74 % of absolute impact) and abiotic depletion (fossil fuels) (61 % of absolute impact).

3.3.2.6 Weighted end of life results for the baseline and alternative 1 Bio-Based PLA

The weighted results for the baseline EoL technologies are shown in Figure 92, and the numerical results can be found in Annex 4. For methodology and choice of normalisation and weighing factors, please see the Approach and methodology chapter, section 5.2.3 "Use of normalised and weighted results to determine the preference of EoL option" and the values in Table 30.

The weighted results show, both with and without toxicity, that the recycling technology is the most favourable choice, followed by industrial composting, anaerobic digestion, incineration and lastly landfilling.

The largest contribution to the weighted results is the climate change impact category in most of the EoL technologies. The savings in the recycling scenarios are distributed between several impact categories, primarily human toxicity (cancer effects), which is due to savings when PLA substitutes for virgin PET.

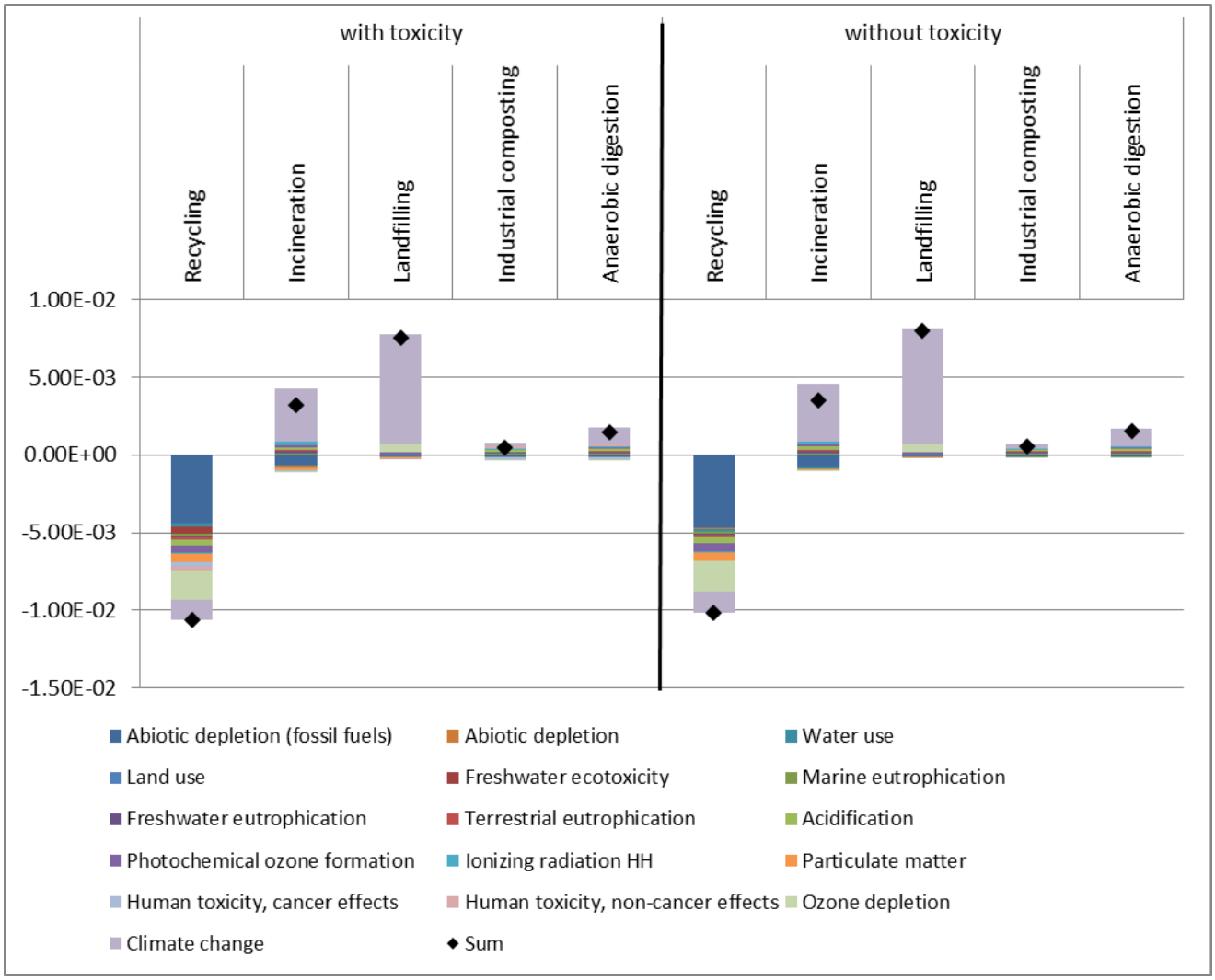


Figure 92. Weighted EoL results for the baseline and alternative 1 product system, bio-based PLA cups.

3.3.3 Alternative 2 and reference 1: Bio-Based PP and petrochemical PP

The EoL results for the alternative bio-based PP and the reference petrochemical PP are presented per EoL technology here. The total LCIA results for the bio-based PP and petrochemical PP product systems for all the technologies are presented in Table 116.

Incineration of the bio-based PP cups results in a climate change impact of 2.3 CO_2-eq compared to 0.046 kg CO_2-eq for recycling and 0.016 kg CO_2-eq for landfilling, presented in Figure 93. For the remaining impact categories and overall, there is not a clear preferable EoL option for the PP cups, but a vague preference of recycling and more clearly the least preferred EoL option is incineration.

Further details on the individual EoL options for the PP cups are presented below, looking into specific impact categories and contribution of impacts from different processes and elemental emissions.

Table 116. Total results of treating 1 kg bio-based PP and Petrochemical PP (incl. 100 g contamination) for all impact categories. Highest impact within each category is red and lowest impact green. The intended EoL option is marked with a bold box.

Impact category (Unit)	Bio-based PP			Petrochemical PP		
	Recycling	Incineration	Landfilling	Recycling	Incineration	Landfilling
Climate change - fossil (kg CO_2 eq.)	-8.05E-01	-6.19E-01	1.51E-02	1.03E-01	2.41E+00	1.51E-02
Climate change - biogenic (kg CO_2 eq.)	8.51E-01	2.92E+00	4.54E-04	-5.66E-02	-1.07E-01	4.54E-04
Climate change - total (kg CO_2 eq.)	4.63E-02	2.30E+00	1.55E-02	4.63E-02	2.30E+00	1.55E-02
Ozone depletion (kg CFC-11 eq.)	-2.78E-08	-6.26E-08	4.59E-10	-2.78E-08	-6.26E-08	4.59E-10
Human toxicity, non-cancer effects (CTUh)	3.51E-08	5.01E-08	3.89E-09	3.51E-08	5.01E-08	3.89E-09
Human toxicity, cancer effects (CTUh)	4.12E-08	1.46E-07	7.44E-11	4.12E-08	1.46E-07	7.44E-11
Particulate matter (kg PM2.5 eq.)	-1.75E-04	-1.18E-04	2.98E-06	-1.75E-04	-1.18E-04	2.98E-06
Ionizing radiation HH (kBq U235 eq.)	1.05E-02	8.50E-02	6.74E-04	1.05E-02	8.50E-02	6.74E-04
Photochemical ozone formation (kg NMVOC eq.)	-2.35E-03	8.98E-04	4.83E-05	-2.35E-03	8.98E-04	4.83E-05
Acidification (molc H+ eq.)	-1.33E-03	2.04E-03	6.25E-05	-1.33E-03	2.04E-03	6.25E-05
Terrestrial eutrophication (molc N eq.)	-2.49E-03	6.40E-03	1.98E-04	-2.49E-03	6.40E-03	1.98E-04
Freshwater eutrophication (kg P eq.)	5.19E-05	2.38E-04	5.34E-07	5.19E-05	2.38E-04	5.34E-07
Marine eutrophication (kg N eq.)	-2.68E-04	5.30E-04	6.06E-05	-2.68E-04	5.30E-04	6.06E-05
Freshwater ecotoxicity (CTUe)	5.51E-01	3.26E+00	9.39E-03	5.51E-01	3.26E+00	9.39E-03
Land use (kg C deficit)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use (m^3)	3.25E-01	-3.03E-01	2.27E-02	3.25E-01	-3.03E-01	2.27E-02
Abiotic depletion (kg Sb eq.)	-1.18E-07	-4.26E-07	2.58E-09	-1.18E-07	-4.26E-07	2.58E-09
Abiotic depletion (fossil fuels) (MJ)	-3.25E+01	-1.15E+01	2.35E-01	-3.25E+01	-1.15E+01	2.35E-01
NREU (MJ)	-3.34E+01	-1.32E+01	1.97E-01	-3.34E+01	-1.32E+01	1.97E-01

The colour scale has the same significance as in the other product systems (Table 115 and Table 117). This means that you can compare the colour horizontally across all tables, but only comparing between the same impact category.

3.3.3.1 Recycling of PP

Recycling the PP single-use cups will give the best environmental performance in seven impact categories, almost exclusively due to savings from substituting PP. Recycling performs worst in the water use impact category, with the largest contribution from washing at the plastic recycling plant. The contribution analysis for recycling of the bio-based and petrochemical PP single-use cups shows that the plastic substitution is the largest contributor in all impact categories except climate change biogenic, both human toxicity categories, freshwater ecotoxicity, freshwater eutrophication, water use and abiotic depletion. Transportation has the largest contribution in human toxicity (non-cancer effects) with 36 % of the absolute impact. Treatment of rejects has the largest contribution to the LCIA results in freshwater ecotoxicity, freshwater eutrophication and human toxicity (cancer effects).

3.3.3.2 Landfilling of PP

Landfilling the PP shows the lowest impact in six impact categories, but at the same time the highest impact in five categories. As both bio-based PP and petrochemical PP are not biodegradable there is no landfill gas generated, resulting in limited impacts on the environment in some impact categories. This contrasts with the other EoL technologies, which have only limited impacts in these impact categories, leaving the landfilling to perform better. The minor impacts come from the construction and operation of the landfill, transportation and leachate treatment. The construction and operation of the landfilling facility has the largest impact in the categories of climate change (biogenic), ozone depletion, human toxicity (cancer effects), ionising radiation, particulate matter, freshwater eutrophication, water use and abiotic depletion. Transportation has a large impact on climate change (fossil), human toxicity non-cancer, photochemical ozone formation, terrestrial eutrophication and NREU. Leachate treatment has a substantial impact of 48 % in freshwater ecotoxicity and 64 % in marine eutrophication but contributes to less than 20 % in other impact categories. The marine eutrophication impact from leachate treatment mostly comes from ammonium that is released to surface water.

3.3.3.3 Incineration of PP

Incineration performs best in only three impact categories (ozone depletion, water use and abiotic depletion) and worst in ten impact categories. The large savings in water use come from substitution of electricity. The good performance for abiotic depletion comes from both heat and electricity substitution. The largest contributor to the fossil climate change impact of incineration are the direct air emissions from the incineration plant. The bottom ash treatment has the largest impact on human toxicity (cancer effects), freshwater eutrophication and freshwater ecotoxicity. The large contribution in cancer effects is due to chromium emissions to water and soil and the freshwater eutrophication comes from phosphate emissions to water and soil. The fly ash treatment has the largest contribution in the human toxicity (non-cancer) impact category. The heat substitution has the largest savings in the impact categories NREU, abiotic depletion (fossil fuels), and ozone depletion and the largest impact in the ionising radiation category. The direct emissions have a large impact on the climate change impact (biogenic for the bio-based and fossil for the petrochemical PP), photochemical ozone formation, terrestrial and marine eutrophication.

3.3.4 Reference 2: Petrochemical PET

The total LCIA results for the petrochemical PET product system for all the EoL technologies are presented in Table 117.

The results for the petrochemical PET cups are similar to the PP cups, though recycling shows lower impacts. Incineration still seems to be the least favourable choice for waste management of the PET cups.

Further details on the individual EoL options for the PET cups are presented below, looking into specific impact categories and the contribution of impacts from different processes and elemental emissions.

Table 117. Total results of treating 1 kg petrochemical PET (incl. 100 g contamination) for all impact categories. Highest impact within each category is red and lowest impact green.

Impact category (unit)	Unit	Petrochemical PET		
		Recycling	Incineration	Landfilling
Climate change - fossil		-3.36E-01	2.11E+00	1.53E-02
Climate change - biogenic	kg CO_2 eq.	-6.86E-02	-6.50E-02	4.54E-04
Climate change - total	kg CO_2 eq.	-4.05E-01	2.05E+00	1.57E-02
Ozone depletion	kg CO_2 eq.	-6.35E-06	-3.73E-08	4.59E-10
Human toxicity, non-cancer effects	kg CFC-11 eq.	-5.01E-08	4.27E-08	3.89E-09
Human toxicity, cancer effects	CTUh	3.28E-09	3.03E-08	7.44E-11
Particulate matter	CTUh	-2.26E-04	-6.29E-05	2.99E-06
Ionizing radiation HH	kg PM2.5 eq.	-1.25E-02	5.17E-02	6.74E-04
Photochemical ozone formation	kBq U235 eq.	-2.95E-03	8.58E-04	4.85E-05
Acidification	kg NMVOC eq.	-3.10E-03	1.57E-03	6.26E-05
Terrestrial eutrophication	molc H+ eq.	-5.53E-03	5.48E-03	1.99E-04
Freshwater eutrophication	molc N eq.	-6.26E-05	4.27E-05	4.73E-07
Marine eutrophication	kg P eq.	-5.59E-04	4.43E-04	1.71E-05
Freshwater ecotoxicity	kg N eq.	1.66E+00	1.31E+01	8.52E-03
Land use	CTUe	0.00E+00	0.00E+00	0.00E+00
Water use	kg C deficit	-2.68E-01	-1.86E-01	2.27E-02
Abiotic depletion	m^3	-6.93E-08	-2.54E-07	2.58E-09
Abiotic depletion (fossil fuels)	kg Sb eq.	-3.50E+01	-7.06E+00	2.37E-01
NREU	MJ	-3.45E+01	-8.11E+00	1.98E-01

The significance of the colour scale is the same as for all the other product systems (Table 115 and Table 116). The colours can be compared horizontally across all tables, but only comparing between the same impact category.

3.3.4.1 Recycling of PET

The results of the LCIA shows that recycling has the lowest impact in 13 impact categories with significant savings from substituting PET. The contribution analysis shows that the plastic substitution remains the largest contributor in all impact categories except for human toxicity (cancer effects), freshwater ecotoxicity and abiotic depletion. In the freshwater ecotoxicity and the human toxicity (cancer effects) categories, the treatment of rejects contributes to largest impact.

3.3.4.2 Landfilling of PET

Landfilling performs best in human toxicity (cancer effects) and freshwater ecotoxicity for the PET cups. In six impact categories landfilling has the highest impact compared to other EoL technologies. PET is not biodegradable meaning no landfill gas is generated, resulting in limited impacts on the environment. The minor impacts come from the construction and operation of the landfill, transportation and leachate treatment. The only difference compared to landfilling PP is the leachate composition, due to a minor difference in the chemical composition of PP and

PET (and their contamination). The construction and operation of the landfilling facility has the largest impact on ozone depletion, human toxicity (cancer effects), ionising radiation, water use and abiotic depletion. Transportation has a large impact on climate change (fossil), human toxicity (non-cancer), photochemical ozone formation, terrestrial eutrophication, marine eutrophication and NREU. Leachate treatment has a substantial impact of 42 % in the freshwater ecotoxicity impact category, due to metal emissions in soil and water, but contributes 10 % or less in other impact categories.

3.3.4.3 Incineration of PET

Incineration has the largest climate change impact from the direct burning of the cups, and moreover has the highest impact in in nine other impact categories. Incineration performs best only in abiotic depletion, mostly from substitution of heat. Contribution analysis of the incineration of PET shows that the electricity substitution contributes the most to the savings in the impact categories of climate change from biogenic carbon, particulate matter, water use and abiotic depletion. The heat substitution has the largest savings in the impact categories NREU, abiotic depletion (fossil fuels) and the largest impact in the ionising radiation category. The bottom ash treatment has the largest impact on human toxicity cancer effects and 43 % and 34 % respectively on freshwater eutrophication and freshwater ecotoxicity. The direct emissions have a large impact on climate change from fossil carbon, photochemical ozone formation, and terrestrial and marine eutrophication.

3.3.5 Details on climate change

The following section compares all four product systems and their individual EoL technologies. The comparison of climate change impacts is presented in Figure 93. Furthermore, refer to Table 115 to Table 117, for the other impact categories.

Comparing the impact of all product systems and all EoL options on climate change, recycling of PLA cups performs best with a saving of 0.56 kg CO_2 eq., followed by recycling of PET with a saving of 0.41 kg CO_2 eq. (see Figure 93).

In all scenarios other than recycling of PLA and PET, there is a net emission of CO_2 eq. Looking at overall climate change ipacts for all product systems, landfilling of the non-degradable plastic performs third best (0.02 kg CO_2 eq). Then comes recycling of the bio-based PP and the petrochemical PP (0.05 kg CO_2 eq.).

Overall results for the petrochemical plastics suggest that landfill is environmentally friendlier than most of the other technologies, because the plastic is assumed not to degrade within 100 years. Nonetheless the plastic resource is lost, which is also observed in the fact that landfilling of petrochemical PP performs worst in the impact categories of abiotic depletion, abiotic depletion (fossil fuels) and NREU.

The biological treatments of PLA emit respectively 0.16 kg CO_2 eq. and 0.63 kg CO_2 eq. for industrial composting and anaerobic digestion. For the biological treatments of PLA there are several limitations of the technologies. For industrial composting, as mentioned before the emissions from use on land almost cancels out the savings from fertiliser substitution. For anaerobic digestion; first of all, the PLA only degrades under wet thermophilic conditions in AD and only, a low percentage of all anaerobic digestion plants, and therefore the largest part of the PLA entering anaerobic digestion plants will have no methane yield. Secondly, the methane potential in the PLA is low compared to other organic waste, resulting in a low methane yield and gas production. Thirdly, a large share of the input is sorted out in pre-treatment, because the plants are designed to sort out impurities and cannot distinguish between petrochemical plastic cups and biodegradable cups. This gives the fertiliser substitution in anaerobic digesting PLA a larger share of the climate change benefits.

Landfilling is the worst performing EoL technology for PLA in terms of climate change with a climate change impact of 3.1 kg CO_2-eq. Landfill gas, both not collected, collected, and flared, has the largest impact on climate change and on ozone formation. The results stress the importance of keeping biodegradable PLA plastic from entering landfills.

Overall, the largest climate change saving is from PLA plastic recycling and the largest climate change emission is from landfilling PLA. At the present time the technology for recycling PLA is not available on a commercial scale, and therefore this scenario is not realistic in the near future. Moreover, the recycled PLA was substituted with PET, which gives high saving in environmental impacts. At present the recycled PLA would most likely substitute PET and therefore the recycling EoL was modelled in this way. The PLA could also replace PP and would in that case give a smaller environmental benefit since the production of PET is more environmental costly compared to PP. In the long term, the recycled PLA would replace PLA and hence the saved emissions would decrease (approximately halved when comparing results for climate change for cradle-to-user for PET and PLA, but this is outside of the temporal scope of this study). Recycling PET performs better than the recycling of PP, this is due to higher impacts from production of PET compared to PP, hence saving more when recycling PET.

Note that the weight of the functional unit also plays a role when comparing the product systems, which is not accounted for in Figure 93.

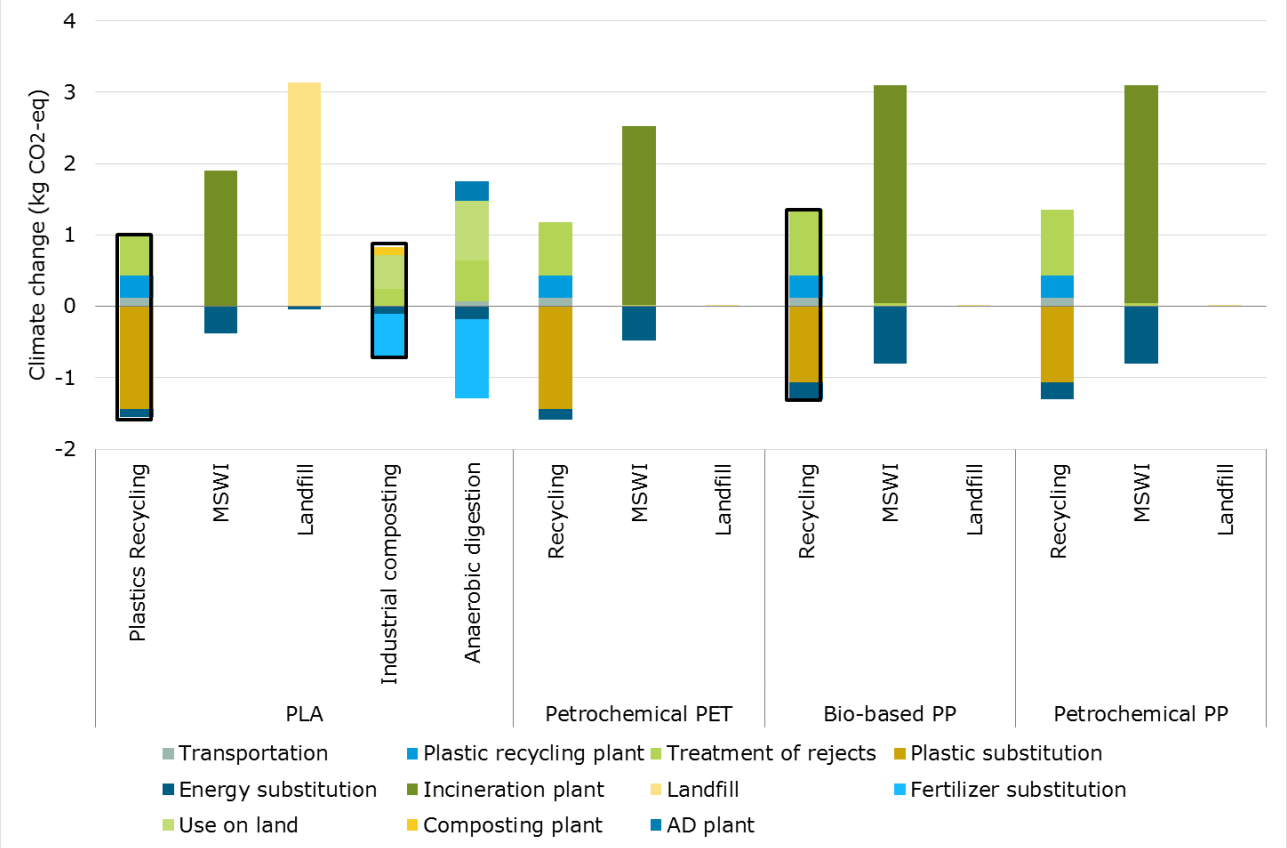


Figure 93. Climate change for treating 1 kg single-use cups (incl. 100 g contamination) in different EoL technologies. The intended EoL option is marked with a black box. See Annex 4 for details of the content of the grouped processes. The results without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

3.3.6 Comparing all EoL technologies in all the product systems with weigted results

Figure 94 presents the weighted results per FU (i.e. different from the above figure, where results are shown for the EoL-reference flow of 1 kg plastic and 100 g contamination) for all EoL technologies and for all product systems. Conclusively, recycling is the best performing EoL technology within each product system. Comparing within the PLA product system the ranking from best to worst performing EoL technology is recycling, industrial composting, anaerobic digestion, incineration and lastly landfilling. The ranking is the same for the product systems of bio-based PP, petrochemical PP and petrochemical PET, with recycling the most preferred option, followed by landfilling and then incineration.

For the bio-based PLA cups the following impact categories contribute predominantly to the weighted results. For recycling abiotic depletion of fossil fuels originating from savings from plastic substitution. Incinerating the PLA contributes especially to climate change due to direct emissions at the incineration plant. The large contribution to climate change of landfilling of PLA stems from landfill gas being emitted directly and flared.

For the PP product systems recycling makes its main contribution to abiotic depletion, which is also true for the PET. The savings are from plastic substitution. The PP product systems also make a large contribution to human toxicity (cancer effects), which come from incineration of the rejects. When incinerating the PP and PET plastics the main contributions are to climate change and abiotic depletion of fossil fuels, which are from respectively direct emissions at the incineration plant and heat substitution. There is for the PET product system also a contribution to freshwater ecotoxicity due to fly and bottom ash treatment. For landfilling the contribution is mainly to climate change which originate from transportation and landfill construction and operation.

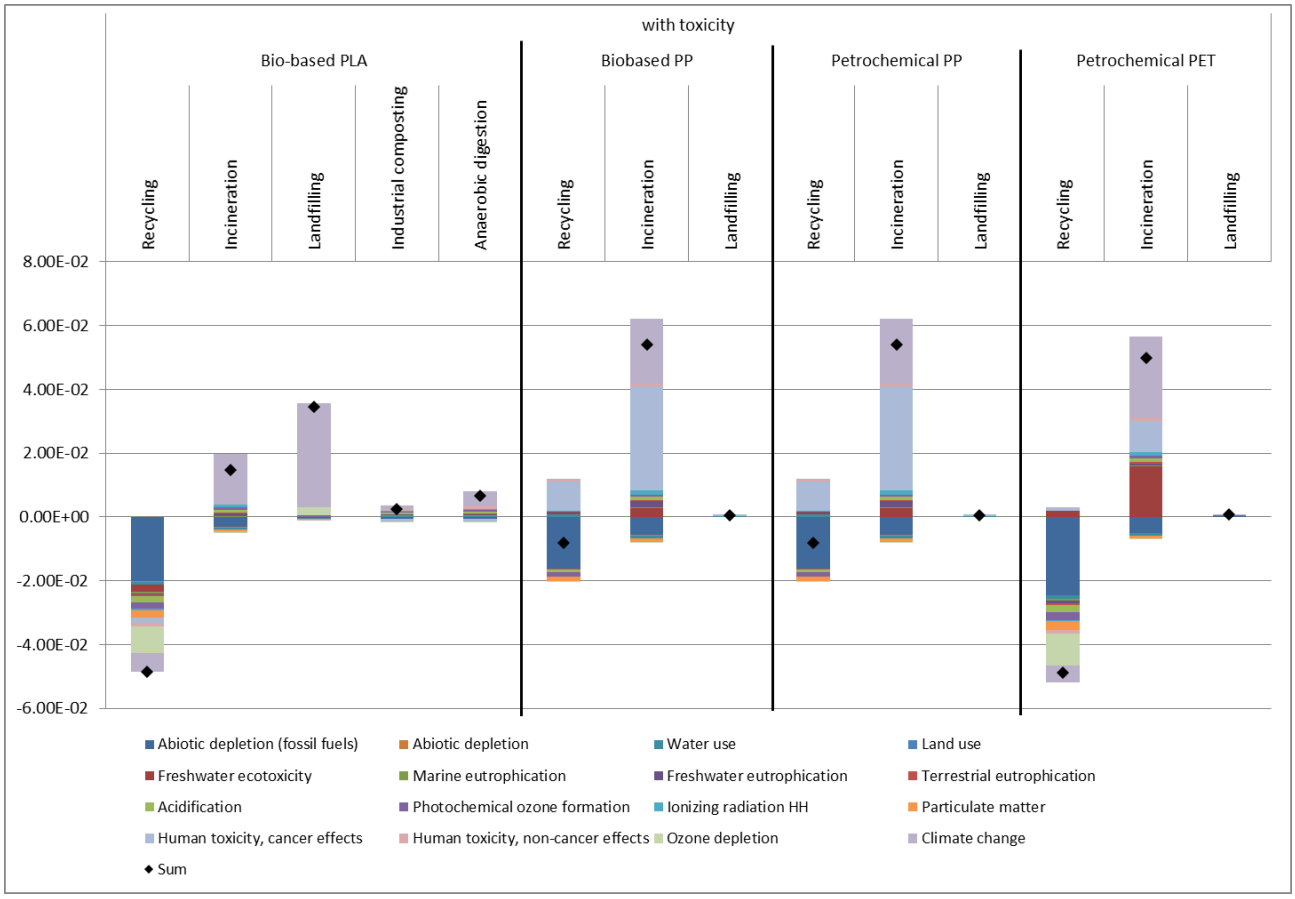


Figure 94. Weighted results with toxicity of the products systems for cups for each EoL option per functional unit. Numerical values can be observed in Annex 4.

3.3.7 End of life mix results

The LCIA for all product systems for the estimated EoL per functional unit are presented in Table 118. Overall the petrochemical PET system is preferred in most impact categories and next the bio-based PLA, although these are the heavier alternatives, i.e. utilises more material per FU. However, looking solely at climate change impact the PP product systems are preferable. The bio-based PLA does not perform better due to a large share of landfilling, that has a low environmental performance and industrial composting which does not perform as well as recycling.

Comparing the two bio-based alternatives the PLA performs better than the bio-based PP, except in climate change and abiotic depletion. This is due to higher recycling for the bio-based PP and therefore higher substitution of petrochemical plastic. The bio-based PLA performs best in four impact categories, the PP in three and the petrochemical PET in 10 impact categories.

Table 118. LCIA for EoL mix of the single-use cups for all product systems per functional unit (1 000 cups, or 3.12 kg of PLA, PP or PET).

Impact category	Unit	Bio-based PLA	Bio-based PP	Petrochemical PP	Petrochemical PET
Recycling	%	15			
Industrial composting	%	15	30	30	30
Incineration	%	39	39	39	39
Landfill	%	31	31	31	31
Climate change - fossil	kg CO_2 eq. / FU	-1.28E+00	-1.86E+00	3.78E+00	4.01E+00
Climate change - biogenic	kg CO_2 eq. / FU	8.11E+00	5.41E+00	-2.28E-01	-2.52E-01
Climate change - total	kg CO_2 eq. / FU	6.82E+00	3.55E+00	3.55E+00	3.76E+00
Ozone depletion	kg CFC-11 eq. / FU	-1.84E-06	-1.27E-07	-1.27E-07	-1.06E-05
Human toxicity, non-cancer effects	CTUh / FU	-3.46E-08	1.21E-07	1.21E-07	1.56E-08
Human toxicity, cancer effects	CTUh / FU	-8.53E-09	2.69E-07	2.69E-07	7.06E-08
Particulate matter	kg PM2.5 eq. / FU	-2.64E-04	-3.80E-04	-3.80E-04	-5.03E-04
Ionizing radiation HH	kBq U235 eq. / FU	6.77E-02	1.42E-01	1.42E-01	9.15E-02
Photochemical ozone formation	kg NMVOC eq. / FU	1.01E-03	-1.32E-03	-1.32E-03	-2.96E-03
Acidification	molc H+ eq. / FU	8.25E-04	1.61E-03	1.61E-03	-1.65E-03
Terrestrial eutrophication	molc N eq. / FU	7.24E-03	7.03E-03	7.03E-03	2.98E-03
Freshwater eutrophication	kg P eq. / FU	1.03E-04	4.20E-04	4.20E-04	-1.11E-05
Marine eutrophication	kg N eq. / FU	7.44E-04	5.63E-04	5.63E-04	5.66E-05
Freshwater ecotoxicity	CTUe / FU	-1.78E+00	5.58E+00	5.58E+00	3.09E+01
Land use	kg C deficit / FU	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	m^3 / FU	-4.75E-01	-5.38E-02	-5.38E-02	-8.04E-01
Abiotic depletion	kg Sb eq. / FU	-4.91E-07	-7.79E-07	-7.79E-07	-6.56E-07
Abiotic depletion (fossil fuels)	MJ / FU	-3.48E+01	-5.50E+01	-5.50E+01	-7.27E+01
NREU	MJ / FU	-3.60E+01	-5.87E+01	-5.87E+01	-7.41E+01

3.4 Cradle-to-grave environmental impacts and interpretation

In this section, the cradle-to-grave results for the investigated product systems are reported.

3.4.1 Bio-Based baseline system: PLA

The cradle-to-grave results of PLA (baseline) cups are given in Table 119. The breakdown of impacts for the baseline product system between biomass production (including carbon removal), manufacturing (including transportation) and EoL (mix) is shown in Figure 95.

Table 119. Cradle-to-grave LCA results (16 PEFCR categories + NREU) of PLA cups (baseline) per functional unit (excluding DLUC and ILUC effects).

Impact category	Unit	Biomass	Manufacturing	EoL (EU mix)	Total
Climate change	kg CO_2 eq.	-6.95	15.58	6.82	15.45
Ozone depletion	kg CFC-11 eq.	1.37E-08	1.02E-06	-1.84E-06	-8.00E-07
Human toxicity, non-	CTUh	-1.29E-06	4.64E-06	-3.46E-08	3.32E-06

cancer effects					
Human toxicity, cancer effects	CTUh	1.41E-08	8.67E-07	-8.53E-09	8.72E-07
Particulate matter	kg PM2.5 eq.	4.28E-04	6.06E-03	-2.64E-04	6.22E-03
Ionizing radiation HH	kBq U235 eq.	1.02E-01	1.12E+00	6.77E-02	1.29E+00
Photochemical ozone formation	kg NMVOC eq.	1.72E-02	4.36E-02	1.01E-03	6.19E-02
Acidification	molc H+ eq.	1.65E-02	8.25E-02	8.25E-04	9.98E-02
Terrestrial eutrophication	molc N eq.	8.37E-02	1.62E-01	7.23E-03	2.53E-01
Freshwater eutrophication	kg P eq.	1.51E-04	8.13E-04	1.02E-04	1.07E-03
Marine eutrophication	kg N eq.	2.74E-02	1.28E-02	7.44E-04	4.09E-02
Freshwater ecotoxicity	CTUe	47.98	23.06	-1.78	69.26
Land use	kg C deficit	70.16	21.83	0.00	91.98
Water use	m^3	2.92	3.15	-0.48	5.59
Abiotic depletion	kg Sbeq	1.50E-06	1.40E-05	-4.91E-07	1.50E-05
Abiotic depletion (fossil fuels)	MJ	12.15	200.14	-34.80	177.48
NREU	MJ	13.08	244.78	-36.01	221.85

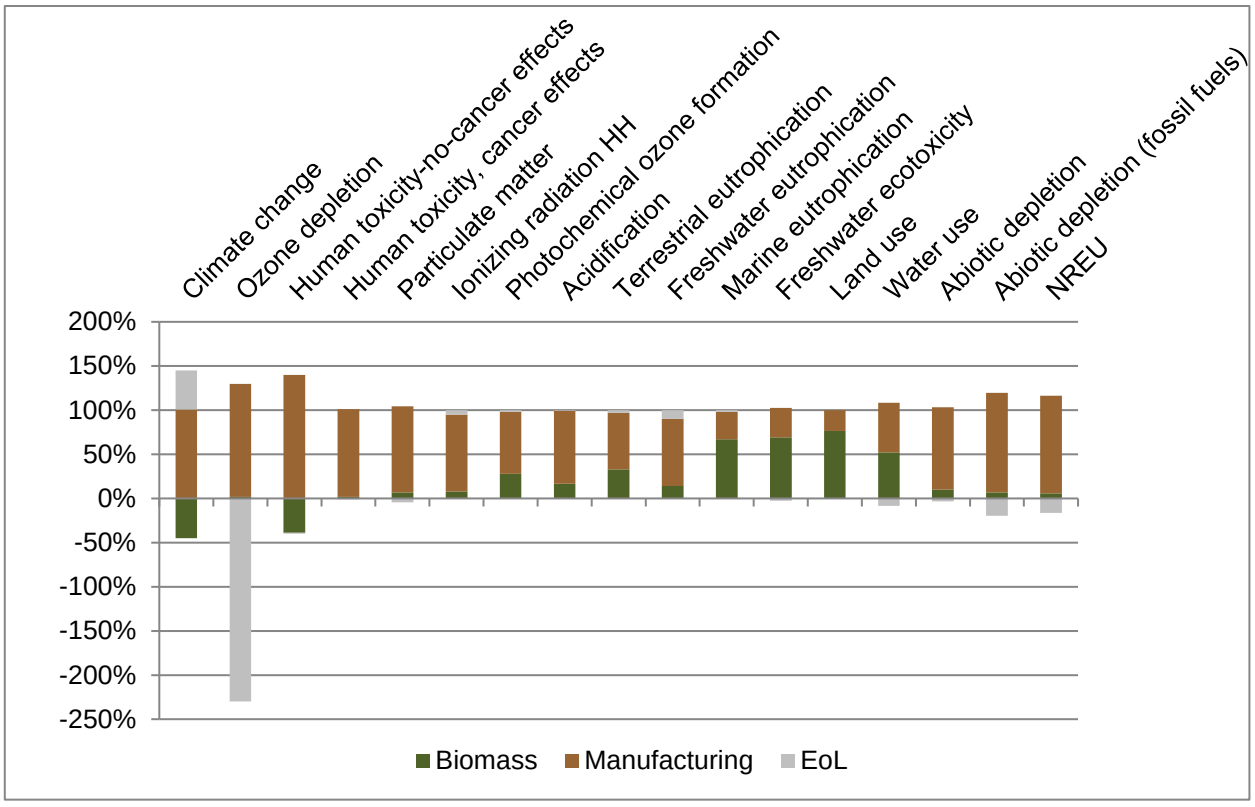
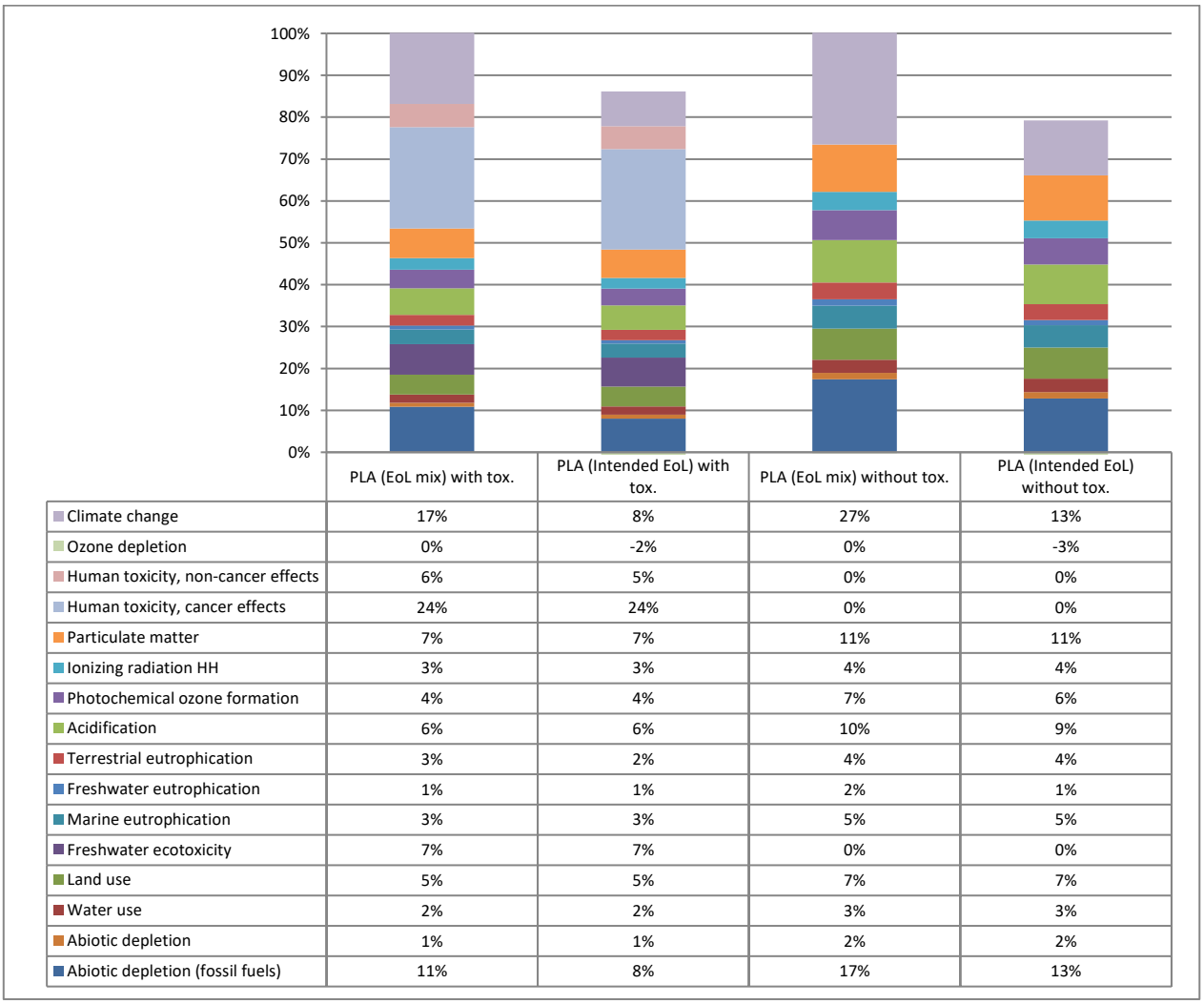


Figure 95. Breakdown of the cradle-to-grave environmental impacts from the PLA (baseline) cups across, 16 PEFCR impact categories + NREU (excluding DLUC and ILUC effects).

It is possible to notice that manufacturing (including transportation) is the major contributor to the environmental burden in all the impact categories except for marine eutrophication, freshwater ecotoxicity, land use and water use. Nevertheless, manufacturing also represents a significantly impacting life cycle stage in these last impact categories. Biomass production has generally a minor impact (less than 20 %). That threshold is overcome for photochemical ozone formation, terrestrial eutrophication, marine eutrophication, freshwater ecotoxicity, land and water use. At the EoL stage some savings are achieved in several impact categories owing

mainly to the plastic substitution with recycling and the heat and electricity produced during incineration. Major savings are observed for the ozone depletion and this environmental benefit is due to plastic substitution thanks to recycling. An increase in environmental impact due to EoL is significant for climate change and this is due to incineration and landfill.

Normalisation and weighting have been applied to the baseline system (PLA (baseline)) to identify the main hotspots of the aggregated impacts in a single score. Figure 96 shows the weighted results including or not including the toxicity impact categories (notice that the percentages for the case with intended EoL refer to 100 % as cradle-to-grave impact of the base case with EoL mix).



	PLA (EoL mix) with tox.	PLA (Intended EoL) with tox.	PLA (EoL mix) without tox.	PLA (Intended EoL) without tox.
Climate change	17%	8%	27%	13%
Ozone depletion	0%	-2%	0%	-3%
Human toxicity, non-cancer effects	6%	5%	0%	0%
Human toxicity, cancer effects	24%	24%	0%	0%
Particulate matter	7%	7%	11%	11%
Ionizing radiation HH	3%	3%	4%	4%
Photochemical ozone formation	4%	4%	7%	6%
Acidification	6%	6%	10%	9%
Terrestrial eutrophication	3%	2%	4%	4%
Freshwater eutrophication	1%	1%	2%	1%
Marine eutrophication	3%	3%	5%	5%
Freshwater ecotoxicity	7%	7%	0%	0%
Land use	5%	5%	7%	7%
Water use	2%	2%	3%	3%
Abiotic depletion	1%	1%	2%	2%
Abiotic depletion (fossil fuels)	11%	8%	17%	13%

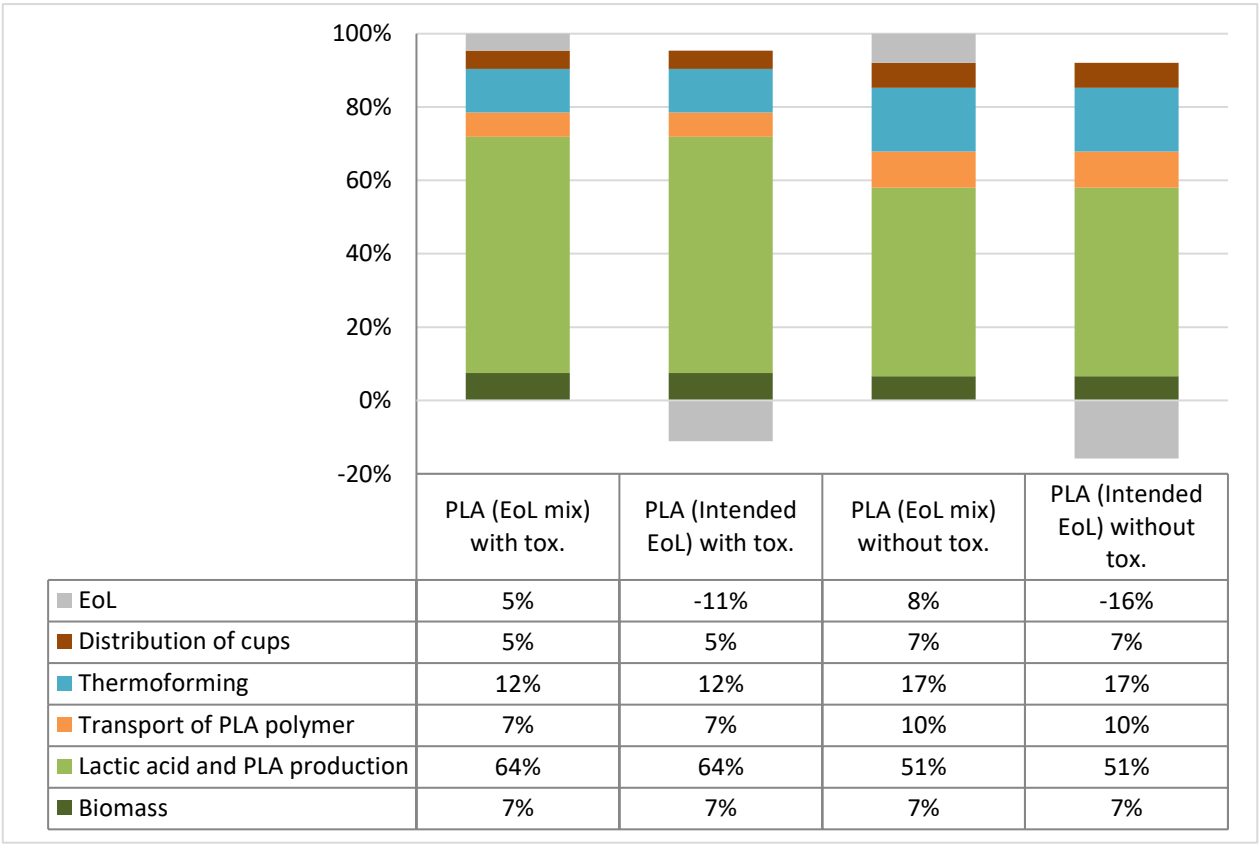
Figure 96. Weighted impact assessment result for PLA cups, cradle-to-grave baseline with (left) and without (right) toxicity categories

When the impact categories related to toxicity are included in the weighted results, for this product system the following impact categories contribute to 83 % of the total environmental impacts: human toxicity (cancer effects= (24 %), climate change (17 %) abiotic depletion of fossil fuels (11 %), particulate matter (7 %), freshwater ecotoxicity (7 %), human toxicity (non-cancer) (6 %), acidification (6 %) and land use (5 %).

From the assessment of the environmental impact excluding toxicity impact categories the following categories represent 79 % of the total environmental burden: climate change (27

%), abiotic depletion of fossil fuels (17 %), particulate matter (11 %), acidification (10 %), land use (7 %) and photochemical ozone formation (7 %).

Figure 97 shows the breakdown of the weighted single score results per life cycle stages. The manufacturing life cycle stage (including lactic acid and PLA production, thermoforming and two transportation services) is the main contributor of environmental burden, accounting for 85-87 % of the impact depending on wehter toxicity impact categories are included or not. When toxicity is included the impact of the chemical conversion from biomass to PLA has a higher share. Biomass production and EoL stages are the second or the third contributor, depending on the chosen weighting approach. When the toxicity impact is considered, biomass represents 7 % of the impact while EoL only 5 %. In contrast, without toxicity considerations, EoL has a slightly higher impact (8 %) compared to biomass (7 %). Among the processes of the manufacturing phase, the lactic acid and PLA production is the most impactful, contributing 51-64 % of the total cradle-to-grave environmental burden. Thermoforming contributes 12-17 % of the total cradle-to-grave impact and more than 90 % of it is caused by the production of electricity. The transportation of the polymer contributes to to 7-10 % of the total impacts representing the third biggest contributor of environmental impact, followed by EoL (5-8 %) and the distribution of the cups (5-7 %).



	PLA (EoL mix) with tox.	PLA (Intended EoL) with tox.	PLA (EoL mix) without tox.	PLA (Intended EoL) without tox.
EoL	5%	-11%	8%	-16%
Distribution of cups	5%	5%	7%	7%
Thermoforming	12%	12%	17%	17%
Transport of PLA polymer	7%	7%	10%	10%
Lactic acid and PLA production	64%	64%	51%	51%
Biomass	7%	7%	7%	7%

Figure 97. Breakdown of the weighted impact assessment result for PLA (baseline).

As highlighted by the weighted results, the EoL represents a share of only 5-8 % of the total impact for this case study. This impact can be further reduced by implementing the intended EoL share increases. For PLA, the intended EoL was identified as 50 % industrial composting/ 50 % recycling.

Table 120 shows the variation of the cradle-to-grave environmental impact for all the investigated impact categories when the intended EoL is implemented. The main reductions are observed for climate change (-51 %), ozone depletion (-1 593 %) and abiotic depletion of

fossil fuels (-26 %). The savings are achieved thanks to avoding the landfill and inceniration of PLA and thank to the benefits of virgin plastic substitution from recycling. Especially in the case of ozone depletion, among the assessed EoL options, landfill was the only option that provided an environmental burden instead of environmental savings: this led to that huge reduction of impact.

Overall, there is a reduction of 16-24 % of weighted impact by shifting from the default EoL mix to the intended EoL (see Figure 97).

Table 120. Cradle-to-grave LCA results (16 PEFCR categories + NREU) of PLA cups (baseline) with intended EoL per functional unit (excluding DLUC and ILUC effects).

Impact category	Unit	Intended EoL	Total with intended EoL	Variation of the total cradle-to-grave results with EoL mix (%)
Climate change	kg CO_2 eq.	-1.03	7.60	-51
Ozone depletion	kg CFC-11 eq.	-1.46E-05	-1.35E-05	-1 593
Human toxicity, non-cancer effects	CTUh	-6.28E-08	3.29E-06	-1
Human toxicity, cancer effects	CTUh	-1.96E-08	8.61E-07	-1
Particulate matter	kg PM2.5 eq.	-5.07E-04	5.98E-03	-4
Ionizing radiation HH	kBq U235 eq.	-7.22E-04	1.22E+00	-5
Photochemical ozone formation	kg NMVOC eq.	-6.19E-03	5.47E-02	-12
Acidification	molc H+ eq.	-5.61E-03	9.34E-02	-6
Terrestrial eutrophication	molc N eq.	-7.63E-03	2.38E-01	-6
Freshwater eutrophication	kg P eq.	-7.87E-05	8.86E-04	-17
Marine eutrophication	kg N eq.	-2.05E-04	4.00E-02	-2
Freshwater ecotoxicity	CTUe	-5.32	65.72	-5
Land use	kg C deficit	0.00E+00	9.20E+01	0
Water use	m^3	-0.47	5.60	0
Abiotic depletion	kg Sb eq.	-2.43E-07	1.53E-05	2
Abiotic depletion (fossil fuels)	MJ	-81.55	130.73	-26
NREU	MJ	-81.29	176.56	-20

3.4.2 Bio-Based alternative system 1: PLA (EU maize)

The cradle-to-grave results of PLA (EU maize) cups are given in Table 121. The breakdown of impacts for the baseline product system between biomass production (including carbon removal), manufacturing (including transportation) and EoL is shown in Figure 98.

Table 121. Cradle-to-grave LCA results of PLA cups (EU maize) on functional unit basis (excluding DLUC and ILUC effects).

Impact category	Unit	Biomass	Manufacturing	EoL	Total
Climate change	kg CO_2 eq.	-3.58	15.13	6.82	18.37
Ozone depletion	kg CFC-11 eq.	2.12E-07	6.34E-07	-1.84E-06	-9.91E-07
Human toxicity, non-cancer effects	CTUh	6.16E-06	5.13E-07	-3.46E-08	6.64E-06
Human toxicity, cancer effects	CTUh	1.31E-07	1.19E-06	-8.53E-09	1.32E-06
Particulate matter	kg PM2.5 eq.	3.33E-03	2.97E-03	-2.64E-04	6.04E-03
Ionizing radiation HH	kBq U235 eq.	2.57E-01	1.29E+00	6.77E-02	1.61E+00
Photochemical ozone formation	kg NMVOC eq.	1.58E-02	2.81E-02	1.01E-03	4.49E-02
Acidification	molc H+ eq.	9.05E-02	4.93E-02	8.25E-04	1.41E-01
Terrestrial eutrophication	molc N eq.	3.44E-01	9.78E-02	7.23E-03	4.49E-01
Freshwater eutrophication	kg P eq.	1.23E-03	3.25E-04	1.02E-04	1.66E-03
Marine eutrophication	kg Neq	7.15E-02	9.63E-03	7.44E-04	8.19E-02
Freshwater ecotoxicity	CTUe	105.91	15.22	-1.78	119.35
Land use	kg C deficit	107.64	7.70	0.00	115.34
Water use	m^3	38.88	3.06	-0.48	41.47
Abiotic depletion	kg Sbeq	1.80E-06	8.41E-06	-4.91E-07	9.72E-06
Abiotic depletion (fossil fuels)	MJ	38.39	192.50	-34.80	196.09
NREU	MJ	45.80	246.25	-36.01	256.05

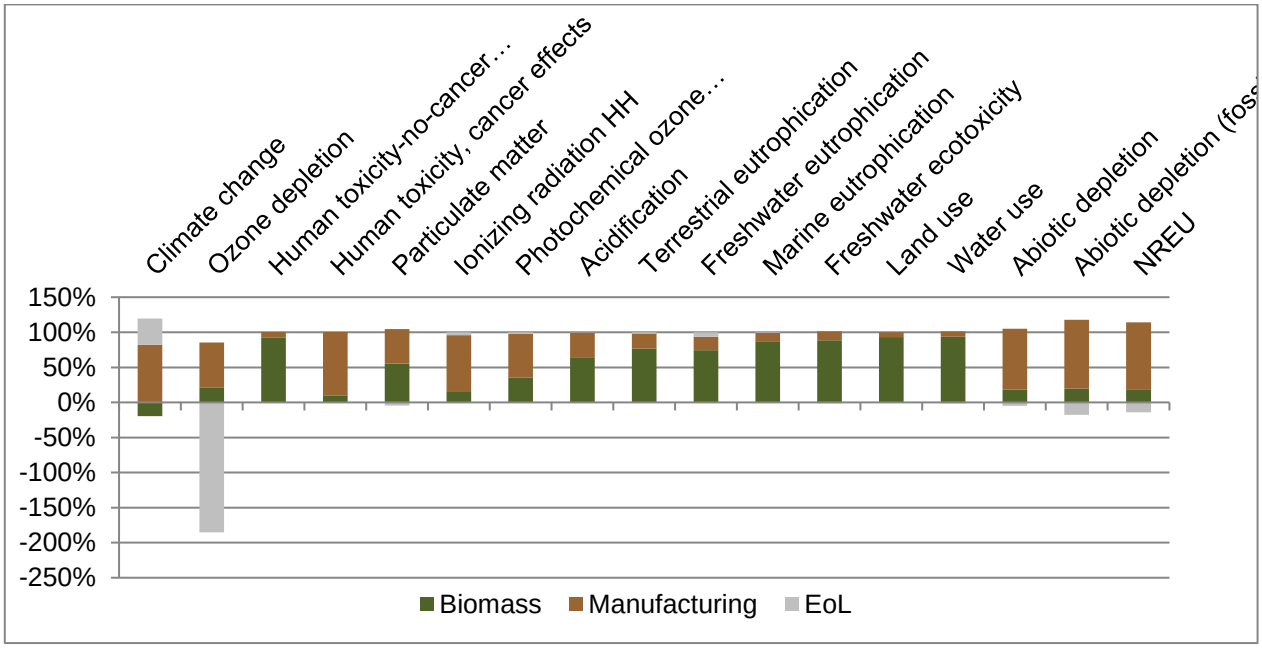


Figure 98. Breakdown of the potential impacts from the PLA (EU maize) cups across all impact categories, cradle-to-grave results (excluding DLUC and ILUC effects).

In this alternative product system (PLA cups from EU maize) biomass production no longer has a minor impact and in most of the impact categories contributes equally or even more

compared to manufacturing. At the EoL stage savings are achieved as for the baseline system, except for climate change where a significant impact is caused by incineration and landfill.

3.4.3 Bio-Based alternative system 2: Bio-based PP from UCO

The cradle-to-grave results of bio-based PP cups are given in Table 122. The breakdown of impacts for the baseline product system between feedstock (UCO collection and carbon removal), manufacturing (including transportation) and EoL is shown in Figure 99.

Table 122. Cradle-to-grave LCA results of bio-based PP (UCO) cups on functional unit basis (excluding DLUC and ILUC effects).

Impact category	Unit	Feedstock	Manufacturing	EoL	Total
Climate change	kg CO_2 eq.	-12.10	7.43	3.55	-1.12
Ozone depletion	kg CFC-11eq	1.19E-08	9.98E-07	-1.27E-07	8.83E-07
Human toxicity, non-cancer effects	CTUh	5.16E-09	4.28E-07	1.21E-07	5.55E-07
Human toxicity, cancer effects	CTUh	1.36E-09	5.01E-08	2.68E-07	3.20E-07
Particulate matter	kg PM2.5eq	3.67E-05	2.46E-03	-3.79E-04	2.12E-03
Ionizing radiation HH	kBq U235eq	4.87E-03	7.53E-01	1.42E-01	9.00E-01
Photochemical ozone formation	kg NMVOCeq	2.78E-04	2.11E-02	-1.32E-03	2.00E-02
Acidification	molc H+eq	4.17E-04	3.42E-02	1.61E-03	3.62E-02
Terrestrial eutrophication	molc Neq	1.01E-03	6.81E-02	7.02E-03	7.61E-02
Freshwater eutrophication	kg Peq	4.29E-06	3.65E-04	4.20E-04	7.90E-04
Marine eutrophication	kg Neq	8.96E-05	6.20E-03	5.63E-04	6.85E-03
Freshwater ecotoxicity	CTUe	0.03	4.07	5.58	9.68
Land use	kg C deficit	0.11	10.42	0.00	10.53
Water use	m3	0.01	3.86	-0.05	3.83
Abiotic depletion	kg Sbeq	1.67E-07	5.92E-06	-7.78E-07	5.31E-06
Abiotic depletion (fossil fuels)	MJ	1.25	103.85	-54.96	50.14
NREU	MJ	1.47	134.33	-58.66	77.14

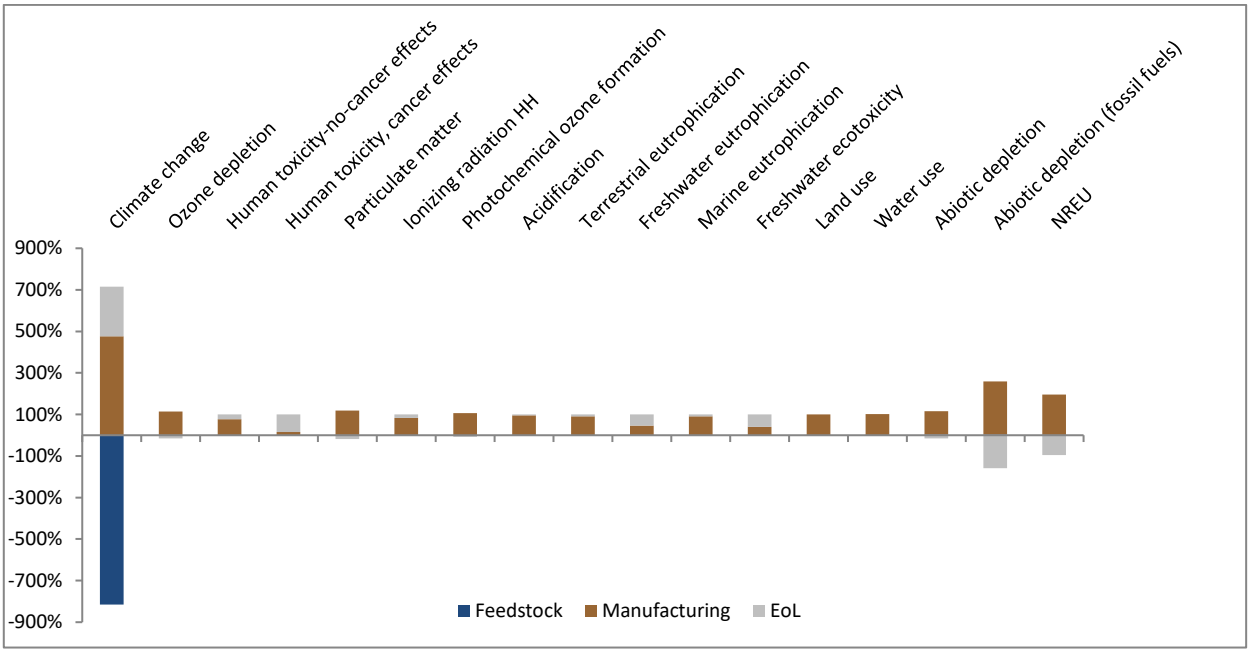


Figure 99. Breakdown of the potential impacts from the bio-based PP (UCO) cups across all impact categories, cradle-to-grave results (excluding DLUC and ILUC effects).

In this alternative product system (bio-based PP cups from UCO). The impact of the feedstock is negligible: it is a waste and therefore the only impact which is assigned is due to collection. The total impact in terms of climate change is negative and this is because of the carbon removal which for PP is higher compared to PLA due to higher carbon content. Manufacturing is the main source of environmental impacts, with the only exception of human toxicity (cancer effects), freshwater eutrophication and freshwater ecotoxicity.

During the EoL stage several relevant savings occur in terms of ozone depletion (mainly due to heat and electricity produced in MSWI), particulate matter (mainly due to plastic substitution of recycling), abiotic depletion (mainly thanks to electricity produced in MSWI), abiotic depletion of fossil fuels-and NREU- (mainly thanks to plastic recycling).

3.4.4 Petrochemical reference systems

In this section, the cradle-to-grave results of the two investigated petrochemical references are reported with the breakdown into manufacturing (granulates production, transportation stages and thermoforming) and EoL (Table 123 and Table 124).

Table 123. Cradle-to-grave LCA results of petrochemical PP cups on functional unit basis.

Impact category	Unit	Manufacturing	EoL mix	Total
Climate change	kg CO_2 eq.	10.95	3.55	14.50
Particulate matter	kg PM2.5eq	3.07E-03	-3.79E-04	2.69E-03
Photochemical ozone formation	kg NMVOCeq	3.59E-02	-1.32E-03	3.46E-02
Acidification	molc H^+eq	4.41E-02	1.61E-03	4.57E-02
Terrestrial eutrophication	molc Neq	9.69E-02	7.02E-03	1.04E-01
Abiotic depletion (fossil fuels)	MJ	353.86	-54.96	298.89
NREU	MJ	386.16	-58.66	327.50

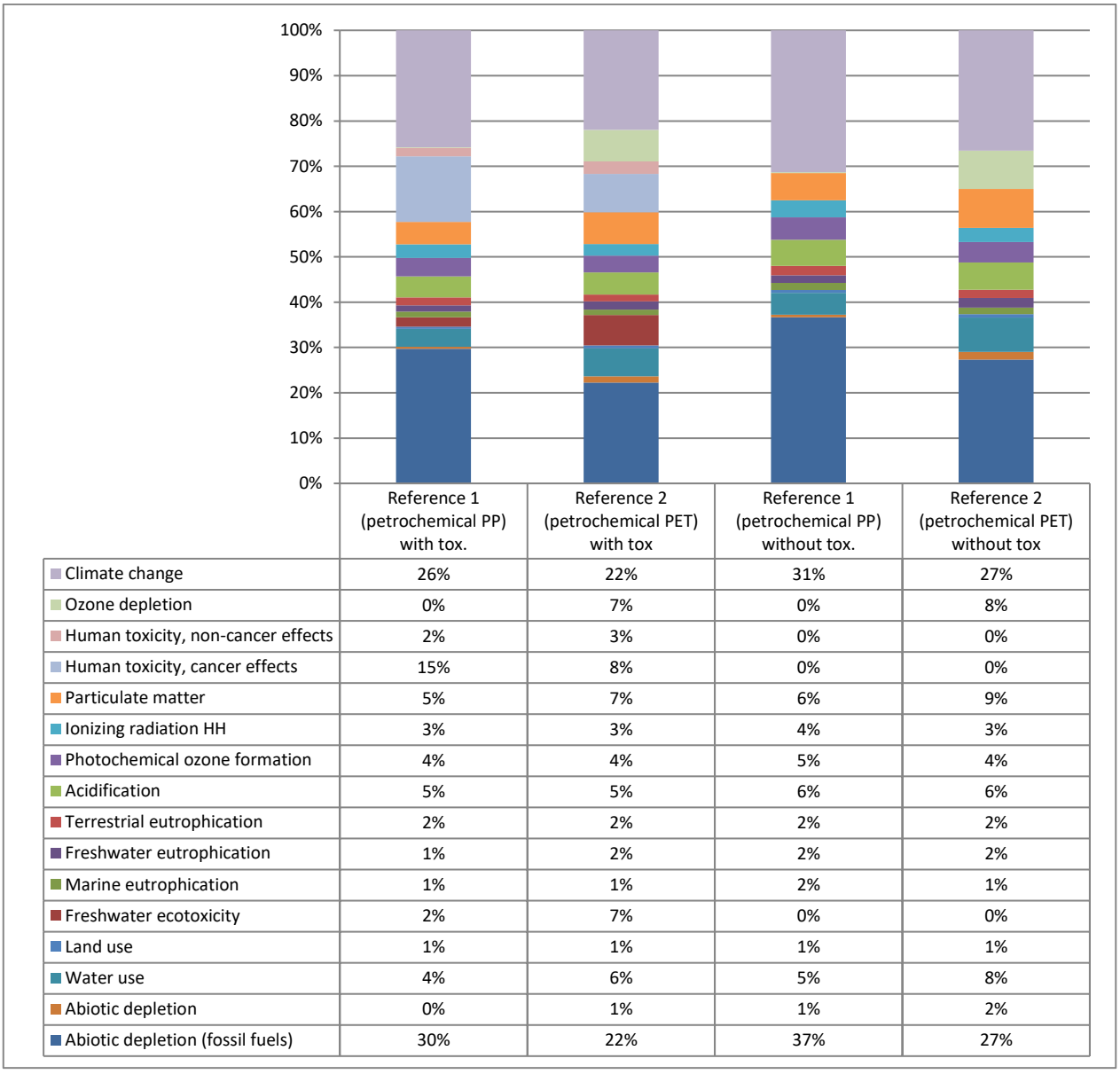
Table 124. Cradle-to-grave LCA results of petrochemical PET cups on functional unit basis.

Impact category	Unit	Manufacturing	EoL mix	Total
Climate change	kg CO_2 eq.	16.01	3.76	19.77
Particulate matter	kg PM2.5eq	6.61E-03	-5.04E-04	6.10E-03
Photochemical ozone formation	kg NMVOCeq	5.29E-02	-2.96E-03	4.99E-02
Acidification	molc H^+eq	7.84E-02	-1.65E-03	7.67E-02
Terrestrial eutrophication	molc Neq	1.48E-01	2.98E-03	1.51E-01
Marine eutrophication	kg Neq	1.39E-02	5.66E-05	1.39E-02
Water use	m^3	18.02	-0.80	17.22
Abiotic depletion (fossil fuels)	MJ	431.36	-72.71	358.65
NREU	MJ	463.58	-74.20	389.37

Normalisation and weighting have been applied to the reference systems to understand which among the assessed impact categories are more relevant for this application and what the main environmental hotspots are. To do this, the impact coming from impact categories which were not judged reliable were also considered. Hence, the weighted results are affected by the considerable uncertainties highlighted in Chapter 6 (Investigation of generic LCA data for fossil-based plastics) and their numerical values must be considered only as indicative. Nevertheless, from Figure 100, abiotic depletion (fossil fuels) along with climate change are the main environmental damages of the petrochemical reference systems and they are both among the recommended impact categories.

- For the PP product system, the main impact of the cups is in terms of abiotic depletion of fossil fuels (30-37 %), climate change (26-31 %). When toxicity is accounted for, human toxicity with cancer effects becomes the third most relevant typology of impact with a 15 % share and it is followed by particulate matter and acidification (both with a 5 % share), water use and photochemical ozone formation (both with a 4 % share of impact). When toxicity is not considered, particulate matter and acidification are the third most important impacts with a 6 % share each and they are followed by water use and photochemical ozone formation with a share of 5 %.

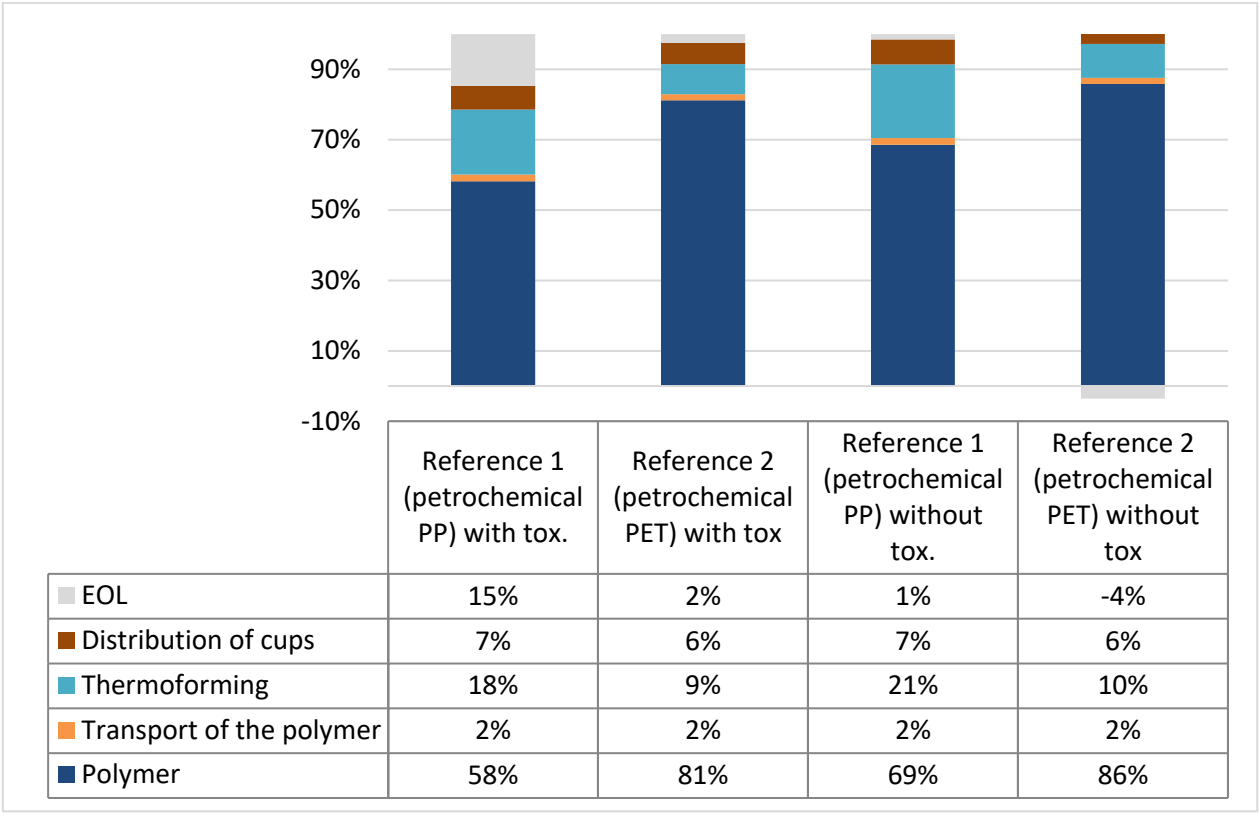
- For the second reference system (PET), also for this material abiotic depletion of fossil fuels is the first typology of impact with a 22-27 % share and it is followed by climate change with 22-31 %. When toxicity is considered, human toxicity with cancer effects covers 8 % of the total impact, freshwater ecotoxicity 7 % and human toxicity without cancer effects only 3 %. Excluding toxicity, particulate matter represents 8 % of the total impact and it is followed by ozone depletion (8 %), land use (8 %), acidification (6 %) and photochemical ozone formation (4 %).



	Reference 1 (petrochemical PP) with tox.	Reference 2 (petrochemical PET) with tox	Reference 1 (petrochemical PP) without tox.	Reference 2 (petrochemical PET) without tox
Climate change	26%	22%	31%	27%
Ozone depletion	0%	7%	0%	8%
Human toxicity, non-cancer effects	2%	3%	0%	0%
Human toxicity, cancer effects	15%	8%	0%	0%
Particulate matter	5%	7%	6%	9%
Ionizing radiation HH	3%	3%	4%	3%
Photochemical ozone formation	4%	4%	5%	4%
Acidification	5%	5%	6%	6%
Terrestrial eutrophication	2%	2%	2%	2%
Freshwater eutrophication	1%	2%	2%	2%
Marine eutrophication	1%	1%	2%	1%
Freshwater ecotoxicity	2%	7%	0%	0%
Land use	1%	1%	1%	1%
Water use	4%	6%	5%	8%
Abiotic depletion	0%	1%	1%	2%
Abiotic depletion (fossil fuels)	30%	22%	37%	27%

Figure 100. Cradle-to-grave weighted impact (per FU) for petrochemical references

Figure 101 shows the weight of each life-cycle stage on the total environmental impact. It is clear that for petrochemical cups most of the emissions are related to the production of the polymer: 58-69 % in the case of PP and 81-86 % in the case of PET. The thermoforming stage is the second contributor to environmental damage with 18-21 % of impact in the case of PP cups, and only 9-10 % in the case of PET cups. The distribution of the cups represents a 6-7 % share in the case of petrochemical cups. The EoL has a negligible share except for PP cups when toxicity is accounted for, where it represents a 15 % share mainly caused (12 % out of 15 %) by human toxicity with cancer effects.



	Reference 1 (petrochemical PP) with tox.	Reference 2 (petrochemical PET) with tox	Reference 1 (petrochemical PP) without tox.	Reference 2 (petrochemical PET) without tox
EOL	15%	2%	1%	-4%
Distribution of cups	7%	6%	7%	6%
Thermoforming	18%	9%	21%	10%
Transport of the polymer	2%	2%	2%	2%
Polymer	58%	81%	69%	86%

Figure 101. Breakdown of the weighted impact assessment results for petrochemical references.

3.4.5 Comparing the Bio-Based baseline system with bio-based alternatives and petrochemical reference

In this paragraph, the environmental performances of the cups reported in the previous sections are compared.

Firstly, the comparison of the life-cycle impact assessment results of all the cups is presented. After the analysis carried out in chapter 6, only 6 PEFCR impact categories were found to be appropriate for comparison for both the petrochemical materials (PP and PET): namely, climate change, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication and abiotic depletion of fossil fuels. In Figure 102, this comparison is therefore shown only for these impact categories. For all the remaining impact categories, the comparison is only carried out for the different bio-based alternatives and shown Figure 103). Secondly, the results for the baseline material managed with the intended EoL are also added to this comparative analysis.

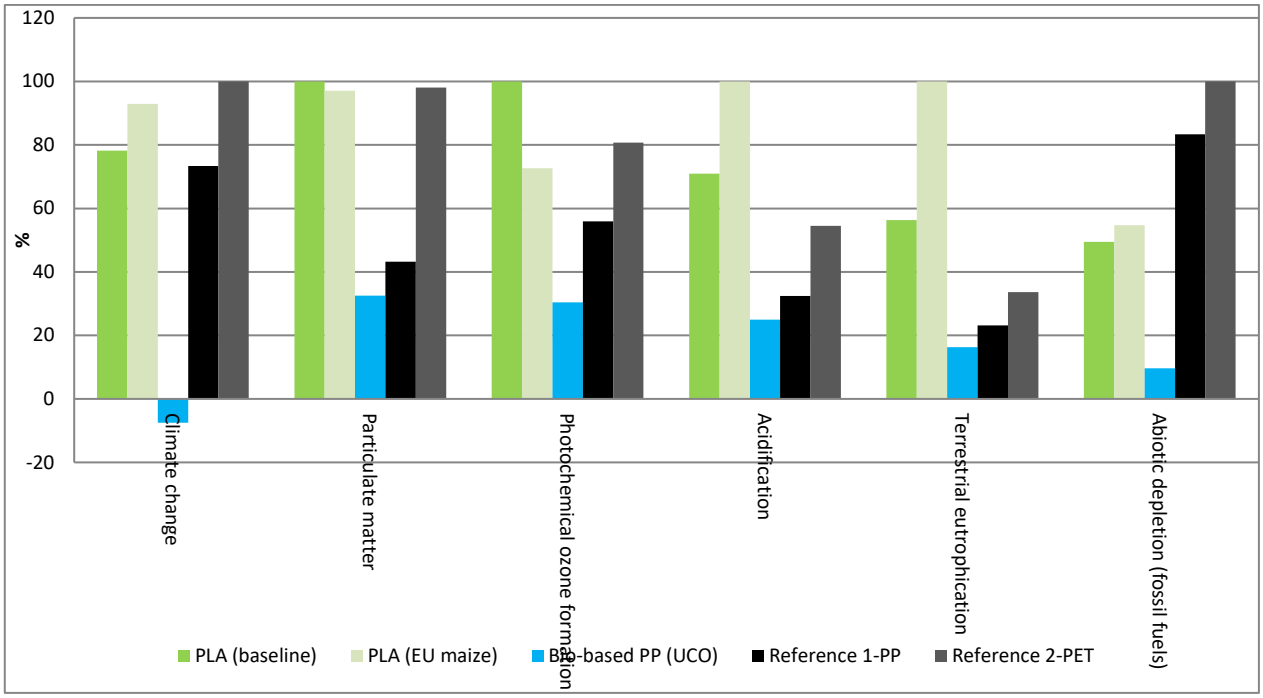


Figure 102. Comparing the cradle-to-grave environmental impact of 1 FU cups for the impact categories selected in chapter 6 (Investigation of generic LCA data for fossil-based plastics).

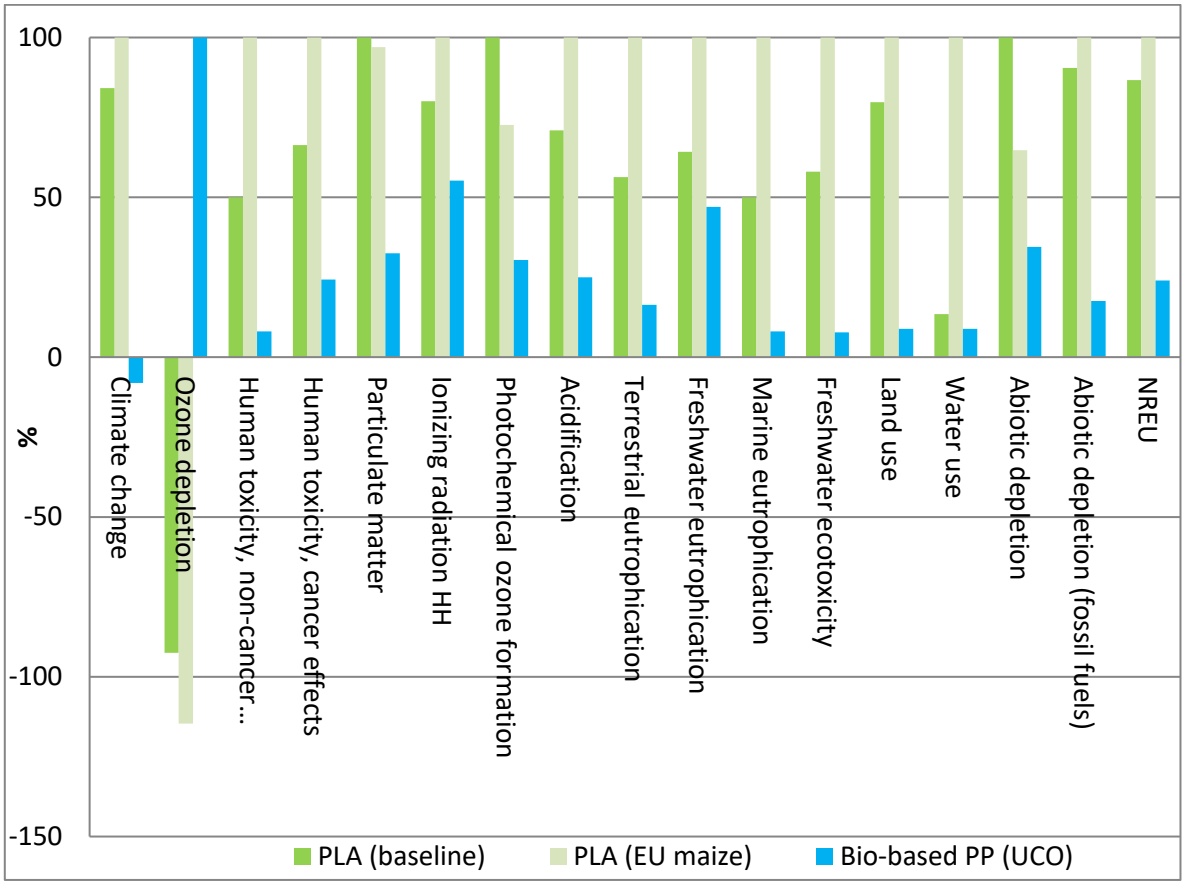


Figure 103. Comparing the cradle-to-grave environmental impact of 1 FU of bio-based single-use cups.

The following considerations emerge from Figure 102 and Figure 103:

- PLA (baseline) shows better environmental performance compared to both the two references (-41 % and -51 % respectively compared to PET and PP) only for abiotic depletion (fossil fuels). Additionally, PLA (baseline) offers significant benefits (more than 20 %) in terms of climate change compared to PET. Compared to PLA from European maize, PLA (baseline) performs generally better: out of 16 impact categories, it performs significantly worse only for photochemical ozone formation (+38 %) and abiotic depletion (+54 %).

- PLA (EU maize) -similarly to PLA (baseline) but with lower emissions savings - performs better than both the petrochemical references for abiotic depletion of fossil fuels (-34/-45 %). PLA (EU maize) is slightly better than PET in climate change (-7 %) -but with lower savings compared to PLA (baseline)- and for photochemical ozone formation (-10 %).

- Bio-based PP (alternative 2) shows the best performance in all six selected impact categories where both the petrochemical references can be compared. Moreover, bio-based PP performs better than bio-based PLA in all the PERFCR impact categories with the only exception of ozone depletion.

- Petrochemical PP shows significantly better environmental performance compared to PLA (baseline) in four out of six selected impact categories, namely: particulate matter (-57 %), photochemical ozone formation (-44 %), acidification (-54 %) and terrestrial eutrophication (-59 %). Moreover, PP has a slightly better climate change impact (-6 %) compared to PLA. All these impact categories with the only exception of climate change were the ones where petrochemical PP already showed the best performances in the cradle-to-user life-cycle stages. For climate change, the environmental performance of petrochemical PP cups is improved by the EoL and this is because of the higher percentage of PP cups recycled compared to PLA cups. Petrochemical PP shows better performances compared to PET for this application in all the six PEFCR impact categories under comparison.

- Petrochemical PET cups show significant environmental benefits compared to PLA (baseline) in photochemical ozone formation (-19 %), acidification (-23 %) and terrestrial eutrophication (-40 %). On the other hand, PET cups would lead to significant increases in climate change impacts (+28 % compared to PLA baseline) and non-renewable energy and resources use (+102 % compared to PLA baseline). Petrochemical PET shows worse performance compared to PP cups in this application.

The comparison between the baseline PLA and the two petrochemical references undergoes several changes when the intended EoL for PLA is also considered (50 % composting and 50 % recycling). In all the six impact categories of the comparison, PLA improves its environmental performance by shifting from the EoL mix to the intended EoL. From Figure 104, it can be seen that PLA cups would become the best option in terms of climate change (-48 % compared to petrochemical PP). Moreover, the baseline PLA with the intended EoL is the preferred option in terms of abiotic depletion of fossil fuels offering savings of -46 % compared to petrochemical PP managed with the EoL mix. On the other hand, PLA managed with the intended EoL has still significantly higher impacts in terms of particulate matter, photochemical ozone formation, acidification and terrestrial eutrophication especially compared to petrochemical PP.

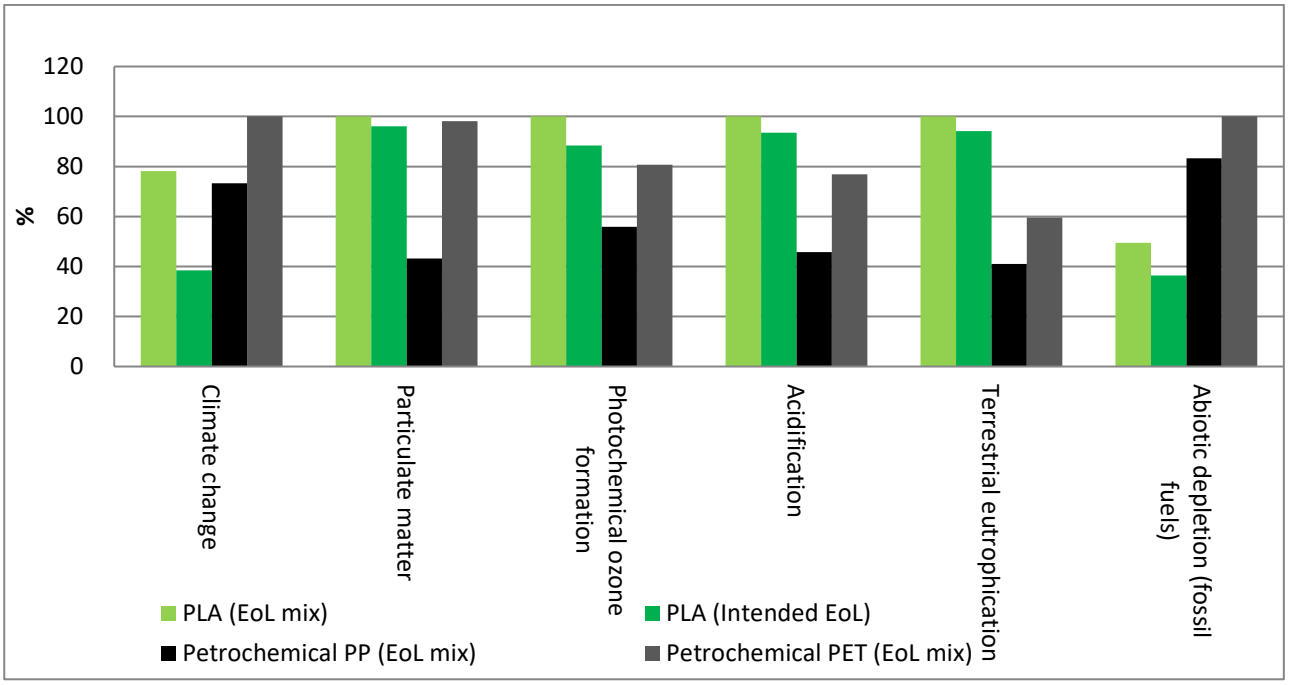


Figure 104. Comparing baseline and references cradle-to-grave environmental impact the impact categories selected in chapter 6 on FU basis.

4 Discussion

4.1 Sensitivity analysis for the weight assumed and the energy consumption of thermoforming process

A sensitivity analysis was performed to assess the variation of the impact assessment results due to the range of weights available on the market for these cups and the uncertainty of the electricity use during thermoforming.

- **The weight of the cups.** Based on the market ranges of weight for cups of 200 ml shown in the chapter *function and functional unit,* compared to the chosen nominal weight, it is possible to find cups with -25 % weight in the case of PP, -11/+2 % for PLA and +16 % for PET. This variation affects proportionally all the assessed environmental impacts. Therefore, the weight of the cup is one of most sensitive parameters to be chosen once the functional unit has been fixed in such a LCA comparison. Focusing on PLA cups, in case the weight decreased by 11 %, the final comparison based on the selected six impact categories will not be affected with the only exception of climate change where the impact of PLA cups would be much closer to the impact of petrochemical PP cups.

- **The amount of electricity needed for thermoforming.** For the nominal calculation, an inline thermoforming process has been selected and the electricity consumption assumed per kg of granulates is reported in Table 97. This is a conservative assumption from an environmental point of view compared to a non-inline thermoforming because this solution allows energy saving compared to the case of differently located extrusion and thermoforming. For a general thermoforming, based on Ecoinvent 3.3 and FRANKLIAN ASSOCIATES (2011), the electricity consumption can be 16-53 % higher compared to the assumed value for inline thermoforming. This figure is obtained for the electricity consumption of PP thermoforming, but it is assumed generally applicable to other resins. Based on company data, the assumed values for electricity use of inline thermoforming could also be 14 %, 23 % and 19 % lower respectively for PP, PET and PLA.

- Table 125 shows what materials and what impact categories are more affected by a variation of the electricity used during thermoforming according to the mentioned variations. It shows that climate change, particulate matter, acidification and NREU are the most affected impact categories. Among the considered materials the total impact of PET is the one least linked to electricity used by thermoforming while the impact of bio-based PP is the most affected. Focusing on PLA impact and on the final comparison for the six selected impact categories, the final conclusions are not affected by the chosen value for electricity consumption.

Table 125. Sensitivity analysis: variation of the cradle-to-grave environmental impact due to uncertainty on electricity used of thermoforming process for the 6 impact categories of the comparison.

Impact category	PLA (baseline)	PLA (EU maize)	Bio-based PP (UCO)	Ref.1 PP	Ref. 2 PET
Climate change	-4/11 %	-3/9 %	-9/35 %	-3/11 %	-3/7 %
Particulate matter	-3/10 %	-4/11 %	-6/25 %	-5/23 %	-3/7 %
Photochemical ozone formation	-2/6 %	-2/8 %	-4/15 %	-3/10 %	-2/5 %
Acidification	-3/9 %	-2/7 %	-7/26 %	-4/20 %	-4/8 %
Terrestrial eutrophication	-2/5 %	-1/3 %	-4/17 %	-3/11 %	-3/6 %
Abiotic depletion (fossil fuels)	-4/11 %	-3/10 %	-4/18 %	-1/6 %	-1/3 %

4.2 End of life sensitivity analysis

The sensitivity analysis was performed for the intended EoL options. One significant parameter has been chosen within each technology, which is likely to change due to technical improvements in both the plastic products and in the waste management technology. An overview of the variation in parameters is presented in Table 126.

- For the biodegradable PLA,recycling and industrial composting are the intended technologies. For industrial composting the parameter is the degradation percentage of the biogenic carbon in the product. For recycling the sorting and technology efficiency is varied.

- For the bio-based plastics that are not biodegradable, recycling was chosen as the intended technology. Here the sorting and technology efficiency is varied.

These parameter changes within a given technology were modelled because the average European technology in the EoL scenarios represent a rather broad set of technological variances. These changes will provide an insight into the variance within these technologies and depict a target for future waste scenarios.

Table 126. Variation in input parameters in the sensitivity analysis.

Product system	Technology	Parameter changed	Original EoL (%)	Lower (%)	Higher (%)	References
PLA	Industrial Composting	Biogenic carbon degradation	57.1	55	95	Hermann, 2011
PLA	Recycling	Sorting and technology efficiency	70	39	90.3	EASETECH
Bio-based PP	Recycling	Sorting and technology efficiency	70	39	90.3	EASETECH

4.2.1 Industrial composting of PLA

The sensitivity analysis was performed on the industrial composting technology. The parameter of biogenic carbon degradation was changed within a limit found in literature (see in Table 126). The lower limit was set to 55 % and the upper limit 95 % C degradation. The original degradation value for the product is 57.1 %. The lower limit is therefore relatively close to the original value.

Figure 105 presents the sensitivity analysis for the PLA cups, the x-axes shows the change in climate change compared to the original EoL. The only impact categories that are affected by the C degradation are the climate change impact categories. There is little difference from the lower limit and the original value, the change is a 5 % increase in climate change (both from biogenic and fossil). Using the higher C degradation limit will lead to a decrease in climate change of 0.15 kg CO_2 eq., which is an 85 % decrease in the climate change results compared to the original.

There are three main reasons for the changes in the climate change impact for biogenic carbon: 1) increasing the degradation percentages means that less of the biogenic carbon will be incinerated with the rejects and contribute to direct emissions of biogenic CO_2. 2) less biogenic carbon will be released to the atmosphere in form of the compost product that is spread on the land later; 3) more biogenic carbon is emitted directly from the composting plant when the biodegradation percentage is higher.

Considering the fossil CO_2, a lower C degradation will lead to a small saving of fossil CO_2 as there is less compost to be spread on land with a diesel consuming tractor. A higher C degradation will lead to less savings of fossil CO_2 when the compost is used on land, which is why the change is positive.

The lower and higher carbon degradation would not cause changes in the conclusion that industrial composting preforms second best in climate change for PLA.

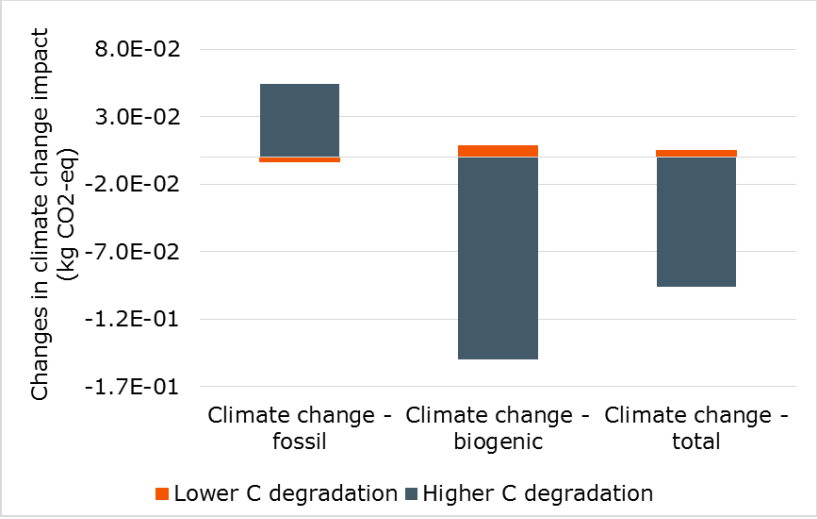


Figure 105. Sensitivity analysis for the parameter C degradation, for PLA single use cups (per EoL-reference flow, i.e. 1 kg PLA and 100 g contamination).

This sensitivity would decrease the cradle-to-grave climate change results when increasing the C degradation by less than 0.2 % and when decreasing C degradation, the cradle-to-grave results will increase by less than 0.01 % in the total climate change impact.

4.2.2 Recycling of PLA

A sensitivity analysis was also performed on the recycling efficiency parameter. The results of the analysis are presented in Figure 106, which shows the percentage change in total results compared with the original percentage. When the percentage is negative, the results are lower than for the original and vice versa. Applying the higher limit of the recycling efficiency of 90.3 % means that there are less rejects to incinerate. Increasing the recycling efficiency will both increase the environmental savings from substituting PET and have less direct emissions from the burning of the PLA. Thus, increasing the recycling efficiency will benefit all impact categories, except abiotic depletion. The abiotic depletion is higher when the recycling is increased, due to more resources used in the recycling plant when treating a higher amount, and less electricity generated when there are less rejects incinerated for energy generation.

The lower sorting and technology efficiency would change the conclusion for climate change impact, hence making industrial composting perform better than PLA recycling. For the following other impact categories, the prioritization of the EoL technology would change with this sensitivity analysis; ionizing radiation and freshwater eutrophication (lower recycling would cause landfilling to perform better) and abiotic depletion (with lower recycling, recycling would move to third place and perform better than industrial composting. For higher recycling, industrial composting would perform worst of all the EoL technologies).

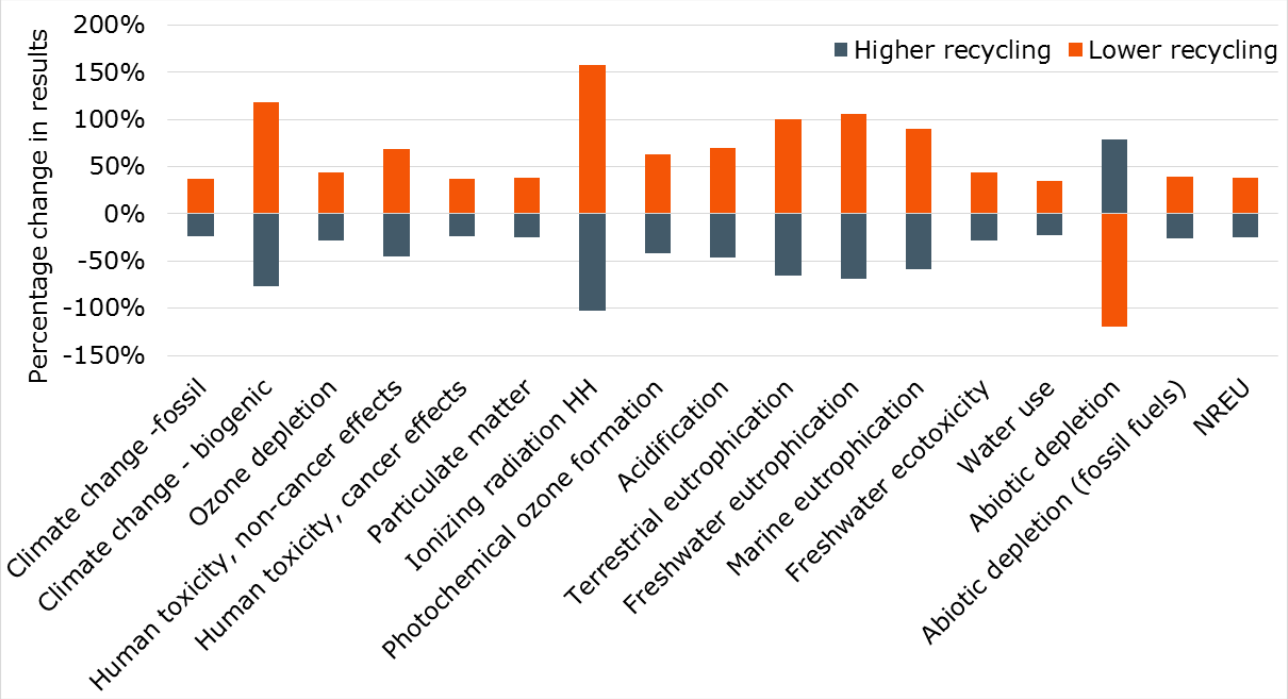


Figure 106. Sensitivity analysis for recycling of PLA single use cups on the EoL results for recycling PLA. Original recycling rate: 70 %, a higher recycling rate of 90.3 % and a lower recycling rate of 30 %.

The cradle-to-grave results change for the seven selected comparable impact categories, when changing the sorting and technology efficiency, can be seen in Table 127. In these seven categories, changing the recycling and sorting efficiency has a low impact on the overall cradle-to-grave results. Lowering the sorting and technology efficiency will result in an increase of total impacts from 1-5 %, where abiotic depletion (fossil fuels) has the largest percentage change. Increasing the technology efficiency will lead to a decrease of 1-3 % depending on the impact category.

Table 127. The change in percentage of higher and lower sorting and technology efficiency at the PLA recycling facility to the cradle-to-grave results of seven selected impact categories.

Impact category	Lower recycling	Higher recycling
Climate change	4 %	-2 %
Particulate matter	1 %	-1 %
Photochemical ozone formation	3 %	-2 %
Acidification	1 %	-1 %
Terrestrial eutrophication	1 %	-1 %
Abiotic depletion (fossil fuels)	5 %	-3 %
NREU	3 %	-2 %

4.2.3 Recycling of bio-based PP

A sensitivity analysis for the bio-based PP was performed on the parameter of sorting and technology efficiency. The percentage change in the total results when increasing or decreasing this parameter is presented in Figure 107. When the percentage is negative, the results are lower than for the original and vice versa. Increasing this parameter will lead to a higher amount of PP that can be substituted and at the same time a lower amount of rejects that are incinerated. Fourteen out of the 18 impact categories show lower potential impacts when the sorting and technology efficiency is increased. For most categories it is the increase or decrease in the amount of applicable substituted PP that has the biggest effect.

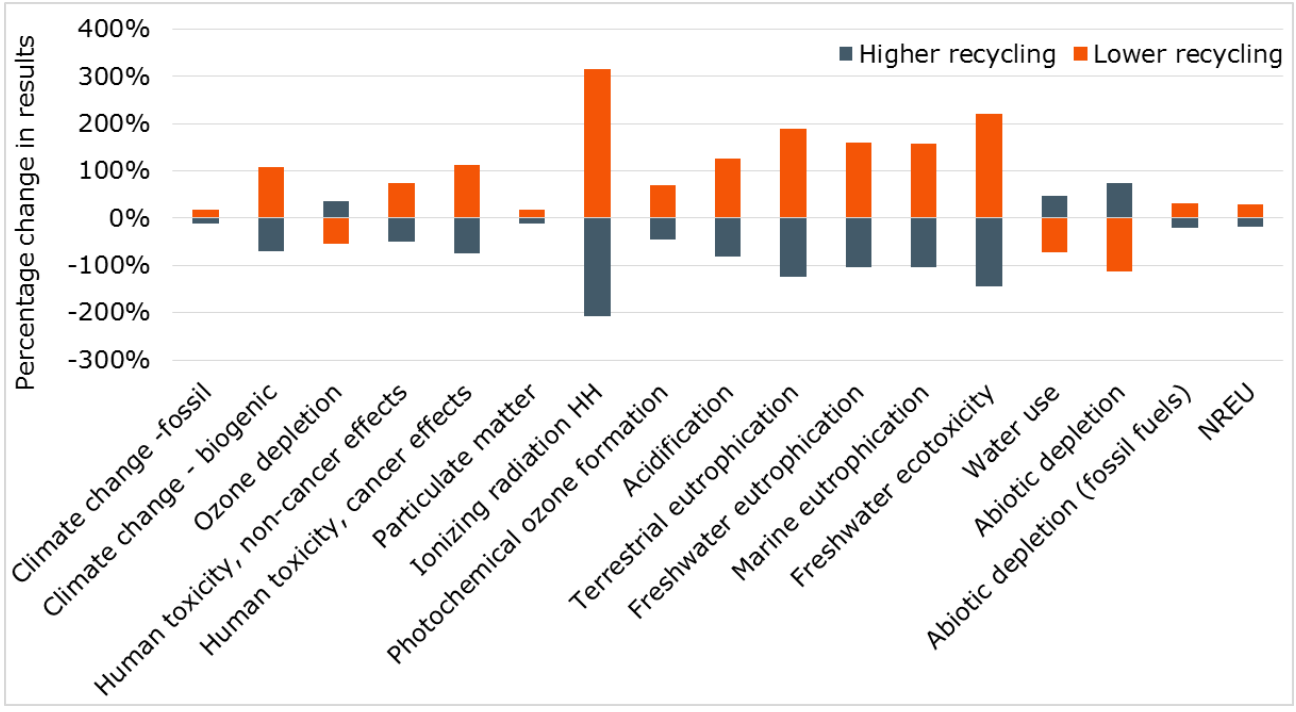


Figure 107. Sensitivity analysis for the parameter recycling efficiency, for bio-based PP single use cups on the results for EoL recycling of PP. The change percentage is compared to the original results for each category.

Ionising radiation is the most sensitive impact category with changes up to 315 %. Significant savings in ionizing radiation are achieved when substituting and increasing the efficiency. The second reason is that heat impacts ionising radiation and decreasing the amount of heat substituted lowers the potential impact of ionizing radiation. Freshwater ecotoxicity is also sensitive to this parameter, lowering the amount of rejects for incineration leads to a lower amount of fly ash and bottom ash which contribute to freshwater ecotoxicity. Water use is affected when increasing recycling, this is because water is used to clean the plastic at the recycling facility before it can be recycled. Observing the climate change impact, the changes in results were 108 % and -70 % respectively for the lower and higher limits of the parameter. The climate change impact increases with the increase of rejects sent to incineration, due to the direct emissions of the incineration.

The prioritisation of the EoL technologies performance for the PP product systems would change in many of the impact categories; climate change (recycling would perform best when recycling is at a higher rate), human toxicity, non-cancer effects (lower recycling would cause the incineration to perform better than recycling), ionizing radiation (recycling would perform better than landfilling when the efficiency is higher), acidification (with lower recycling, landfilling would perform better than recycling), terrestrial eutrophication (landfilling would perform better than recycling, when lowering the recycling plant's efficiency), freshwater eutrophication (higher efficiency would change recycling to be better performing than landfilling), marine eutrophication (landfilling would perform better than recycling with lower efficiency) and freshwater ecotoxicity (recycling would perform better than landfilling with a higher efficiency).

The sensitivity analysis for the cradle-to-grave results, when changing the sorting and technology efficiency for recycling of bio-based PP is presented in Table 128, for the seven selected comparable impact categories. The biomass carbon storage (-12.1) compared to the manufacturing and EoL (12.2) of the bio-based PP almost balance out, thus the cradle-to-grave variation is near zero. This influences the results of the sensitivity analysis, as a small change in the climate change impact in the EoL will appear to significantly influence the LCIA results when looking at the percentage change. The numerical change when increasing the recycling efficiency is -0.8 kg CO_2/FU. Both abiotic depletion and NREU are sensitive to changes in the recycling efficiency with a percentage change between -15 % and 23 %. Increasing the recycling efficiency will lead to a higher substitution rate which will save substantial impacts from NREU and abiotic depletion (fossil fuels). The results are sensitive to the recycling efficiency because plastic substitution has the largest contribution to all of the seven comparable impact categories, and the EoL life stage has a large share of the total impact for the bio-based PP.

Table 128. The change in percentage of higher and lower sorting and technology efficiency at the bio-based PP recycling facility to the cradle-to-grave results of seven selected impact categories.

Impact category	Lower recycling	Higher recycling
Climate change	109 %	-72 %
Particulate matter	2 %	-1 %
Photochemical ozone formation	9 %	-6 %
Acidification	5 %	-3 %
Terrestrial eutrophication	7 %	-5 %
Abiotic depletion (fossil fuels)	23 %	-15 %
NREU	15 %	-10 %

4.3 Estimate of EoL mix 2030

To discuss possible outcomes of policies like the *Action Plan for a Circular Economy* and the amended Directive 99/31/EC of 26 April 1999 on the landfill of waste, the following analysis of an EoL mix 2030 is conducted. In these policies several crucial targets for waste management are set, for example: reduction of municipal waste landfilled to 10 % and preparing for re-use and recycling to 65 % of total municipal waste, both by 2030.

These new targets are implemented in a EoL mix 2030 scenario, to investigate the consequences of the targets in the specific case study of cups. The analysis for the cups case study was chosen because this specific case study contains a large variety of polymers and EoL options for both the bio-based and petrochemical product systems.

For the bio-based PP, petrochemical PP and petrochemical PET the EoL mix is set to 65 % recycling (target for recycling of municipal waste by 2030) With the same sorting and technology efficiency as in the original scenarios of 70 %, this gives a collection rate for recycling of 92.8 %.

The remainder is divided into incineration and landfill with the same shares as previously (39 % for incineration and 31 % for landfilling). This will also fulfil the target for maximum landfilling of municipal waste by 2030 of 10 %. For the bio-based and biodegradable PLA the 92.8/65 % is divided equally into recycling and industrial composting, which applies in the same way as for the current EoL mix. The values applied are observed in Table 129.

It must be noted that this is a very uncertain estimate of the future EoL mix and is only performed to discuss possible impacts of the new legislative EU targets. Furthermore, this analysis does not account for any other future changes in the EoL technologies, like improvements of the technologies (e.g. the sorting and technology efficiency), or the changes in energy systems.

Table 129. Overview of the current EoL mix and the EoL mix for 2030 estimated based on the EU waste targets.

Product system	EoL mix	EoL collection mix 2030 Collection rate (recycling rate)
Bio-based PLA	Recycling: 15 % Industrial composting: 15 % Incineration 39 % Landfilling: 31 %	Recycling: 46.4 % (32.5 %) Industrial composting: 46.4 % (32.5 %) Incineration: 4.0 % Landfilling: 3.2 %
Bio-based PP Petrochemical PP Petrochemical PET	Recycling: 30 % Incineration 39 % Landfilling: 31 %	Recycling: 92.8 % (65 %) Incineration: 4.0 % Landfilling: 3.2 %

From Figure 108 it is observed that most impacts would decrease when applying the estimated EoL mix for 2030. For the PP product systems human toxicity (non-cancer effects) and water use would however increase. Human toxicity (non-cancer effects) increase for the PP product systems due to decreased incineration and hence less electricity substitution. The increased water use is due to increased recycling, which is utilised in the recycling facility. For all product systems abiotic depletion increase to a minor degree, again due to decrease in incineration and hence a decrease in substitution for electricity and heat production.

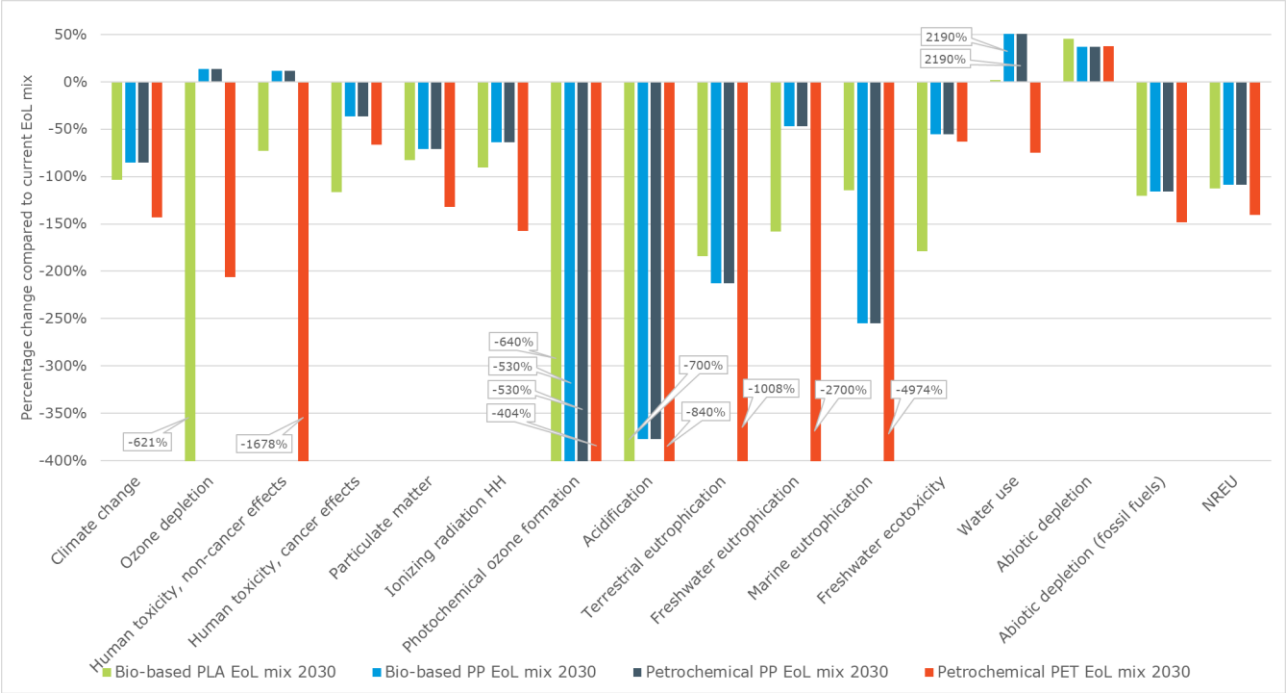


Figure 108. Changes to the environmental impacts of the EoL mix 2030 compared to the current EoL mix per functional unit. The change in percentage is compared to the original results, the EoL mix, for each impact category.

In Table 130 the impacts of applying the EoL mix 2035 to the cradle-to-grave results of the single-use cups are presented, which shows a substantial decrease in all seven selected impact categories (ranging from 4 % to over 250 % decrease of the overall impacts). The large percentage decrease in climate change impact for the bio-based PP compared to the other product systems is due to a low numerical value in the original scenario.

Table 130. The change in percentage when applying EoL collection mix 2030 for all product systems of the single-use cups to the cradle-to-grave results of seven selected impact categories.

Impact category	Bio-based PLA	Bio-based PP	Petrochemical PP	Petrochemical PET
Climate change	-46%	-270%	-21%	-27%
Particulate matter	-4%	-13%	-10%	-11%
Photochemical ozone formation	-10%	-35%	-20%	-24%
Acidification	-6%	-17%	-13%	-18%
Terrestrial eutrophication	-5%	-20%	-14%	-20%
Abiotic depletion (fossil fuels)	-24%	-127%	-21%	-30%
NREU	-18%	-83%	-19%	-27%

These results show that the directives revised and recently adopted by the European Parliament and the Council would decrease the overall impacts of all product systems, both bio-based and petrochemical. This implies that the overall conclusion does not change due to this change in EoL mix.

There lies a great uncertainty in applying a future EoL mix, which could develop in different ways than anticipated here. The development in PLA recycling is one uncertainty, where the split between industrial composting and recycling could lean more towards recycling, which

could possibly alter the overall conclusion making PLA more desirable. However, changes in energy systems and technological development in all life cycle stages will also change in the future, which could also alter the conclusion. Lastly, the very high collection rate needed to achieve the targeted recycling rate of 65 % seems unachievable. It is necessary to address the need for technical improvements in the sorting facility and the recycling facility as well as the quality of the waste input to achieve higher recycling rates.

4.4 Discussion on End of Life uncertainties

The parameters analysed in the sensitivity analysis above and other parameters, which may be uncertain, i.e., estimations, assumptions, huge ranges in industry or EU member states, missing data, etc. are discussed in this section. Methodological shortcomings and additional research requirements are also presented.

Sorting and technology efficiencies for recycling of all plastics - the sensitivity analysis above highlighted the effect of varying this parameter for biogenic PP. For bio-PP the sensitivity shows that this parameter has a large influence on the cradle-to-grave results: up to 109 % of the GWP and for the other impact categories up to 23 %, see Figure 105. It can be assumed that this parameter will also show a high sensitivity for petrochemical PP and PET, as the petrochemical PP is similar, and furthermore, it is sensitive to PET as the weighted results of PET recycling show a larger saving compared to PP (see Figure 94). The collection rate for PLA recycling is half that of PP and PET recycling, and therefore this parameter does not show a sensitivity towards the overall results with the EoL mix only increasing or decreasing the results by up to 5 % (see Figure 106 and Table 127).

PLA recycling – there is more than one uncertainty related to PLA recycling. Firstly, the technology is not commercialised yet, which is why there is no literature or knowledge regarding the sorting and technology efficiency of PLA recycling. PLA recycling is assumed to be like the other polymers (although it is not considered to have a major impact on the cradle-to-grave results) and the collection rate is inferred. Secondly, recycled PLA is set to substitute PET, but might as well substitute PP, and in the future PLA. The latter will imply a smaller environmental benefit when replacing PP and will be halved if the secondary PLA replaces virgin PLA (comparing results for climate change for cradle-to-user for PET and PLA, but this is evaluated to be outside the temporal scope of this study).

The quantity of rejects in industrial composting and AD is subject to uncertainties because the pre- and post-sorting of the processing plants have different technical set-ups. The quantity of reject of biodegradable plastic in the plants is also prone to change if more biodegradable plastic enters the waste stream. This was deemed to be outside of the temporal scope of this study.

Carbon biodegradation in industrial composting. This was analysed using a sensitivity analysis within the industrial composting above. The results from the analysis showed that this parameter will affect the GWP of industrial composting but will not alter the conclusion that industrial composting is the EoL technology that performs second best in terms of GWP for PLA (see Figure 105). Also, the VS degradation is uncertain for the industrial comparison, but this was not investigated with a sensitivity analysis. Adjusting the VS degradation is not assumed to have a major influence on the cradle-to-grave results.

The biodegradation of carbon is also uncertain in the technologies of landfilling and anaerobic digestion. This will again affect solely the GWP. These parameters depend e.g., on the biomass and additives within the material and will differ between types of PLA, but furthermore on the specific plants and the conditions in the landfill. The methane yield for PLA is therefore an uncertain factor. As the methane yield for bio-based plastics is relatively low for AD, and it has a low share contribution to the GWP of AD, this parameter is not expected to impact the results. On the other hand, the emissions of LFG of the landfill technology is the major impact for landfill, which is why this parameter is expected to have a high impact.

The quantity of biogenic carbon in PLA and PP. The PLA and PP studied in this case study are composed of 100 % biogenic carbon, although some PLA and PP will include fossil carbon from additives and co-polymers, and this will affect the environmental profile, especially in the incineration EoL technology for GWP, where different emission factors would apply for fossil and biogenic carbon.

Apart from the above case study's specific uncertainties, several other general uncertainties that apply to most of the presented case studies are discussed in the Approach and methodology in chapter 3.5., e.g., data quality, critical assumptions and limitations in End of Life modelling. These include uncertainties related to

- the estimated EoL mix
- the marginal energy technologies utilised
- the development of the EoL technologies (efficiency increases)
- the consumption of materials and energy at the EoL treatment plants
- littering, which is not modelled in this LCA
- the chemical composition of the biodegradable plastics.

The significant uncertainties are PLA recycling related to the EoL mix, sorting and technology for all recycling of all polymers as well as biodegradation of PLA in landfilling. There is a need for further research in the fields of biodegradation of PLA and the sorting and technology efficiency for PLA recycling.

5 Assessing non-PEF impact categories: taking into account ILUC and endpoint of GTP

5.1 Baseline Bio-Based PLA: Global Temperature change potential

As regards climate change assessed with Global Temperature change Potential (GTP)100a as indicator, it can be noticed that the cradle-to-user impact results are similar to climate change impacts in both absolute values and breakdown shares (Figure 109).

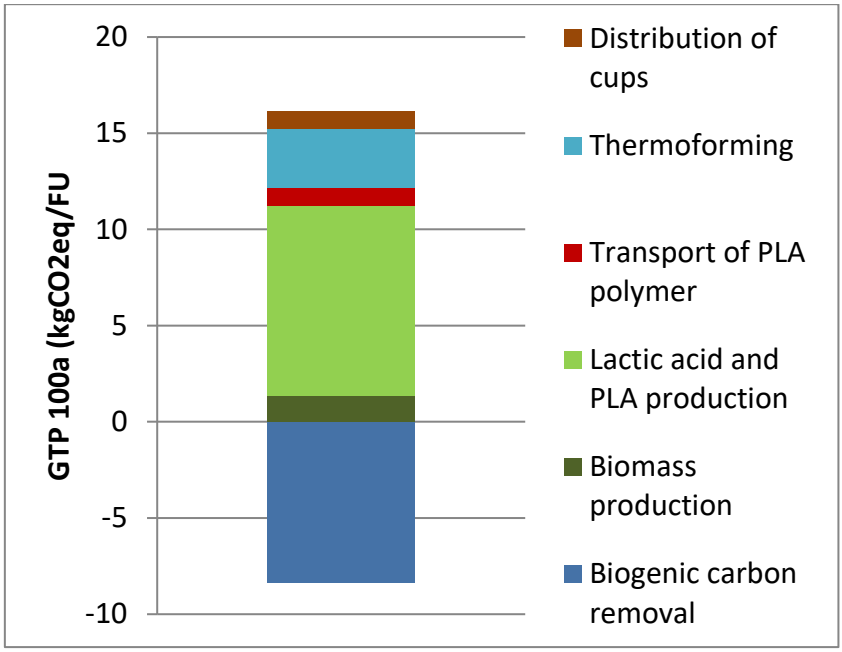


Figure 109. Breakdown of the cradle-to-gate GTP 100a results for baseline PLA cups, 1 functional unit, excluding DLUC and ILUC effects.

5.2 Baseline Bio-Based PLA: Land use change emissions

Table 131 presents the cradle-to-grave characterised results broken down into feedstock, manufacturing, EoL, and ILUC. As highlighted in Chapter 4, DLUC and ILUC cannot be summed up. This case does not involve crops that imply DLUC (namely Brazilian sugarcane, German maize, French wheat and German wheat), based on the methodology from the PEFCR Guidance V6.3 presented in Chapter 4. Figure 110 presents the characterised results for case 3 (baseline).

Table 131. Characterised cradle-to-grave results broken down per process, including ILUC. Case 3, baseline.

Impact category	Unit	Biomass production (incl. biogenic C)	Manufacturing	EoL (EU mix)	ILUC
Climate change	kg CO_2 eq.	-6.95	15.58	6.82	2.22
Ozone depletion	kg CFC-11eq	1.37E-08	1.02E-06	-1.84E-06	4.63E-10
Human toxicity, non-cancer effects	CTUh	-1.29E-06	4.64E-06	-3.46E-08	3.71E-10
Human toxicity, cancer effects	CTUh	1.41E-08	8.67E-07	-8.53E-09	5.90E-11
Particulate matter	kg PM2.5eq	4.28E-04	6.06E-03	-2.64E-04	5.85E-05
Ionizing radiation HH	kBq U235eq	1.02E-01	1.12E+00	6.77E-02	3.19E-05
Photochemical ozone formation	kg NMVOCeq	1.72E-02	4.36E-02	1.01E-03	6.89E-03
Acidification	molc H^+eq	1.65E-02	8.25E-02	8.25E-04	1.38E-03
Terrestrial eutrophication	molc Neq	8.37E-02	1.62E-01	7.23E-03	7.75E-03
Freshwater eutrophication	kg Peq	1.51E-04	8.13E-04	1.02E-04	5.11E-06
Marine eutrophication	kg Neq	2.74E-02	1.28E-02	7.44E-04	8.23E-04
Freshwater ecotoxicity	CTUe	47.98	23.06	-1.78	4.11E-03
Land use	kg C deficit	70.16	21.83	0.00	6.37E-03
Water use	m^3	2.92	3.15	-0.48	6.82E-02
Abiotic depletion	kg Sbeq	1.50E-06	1.40E-05	-4.91E-07	2.02E-08
Abiotic depletion (fossil fuels)	MJ	12.15	200.14	-34.80	5.76E-02
NREU	MJ	13.08	244.78	-36.01	5.51E-02

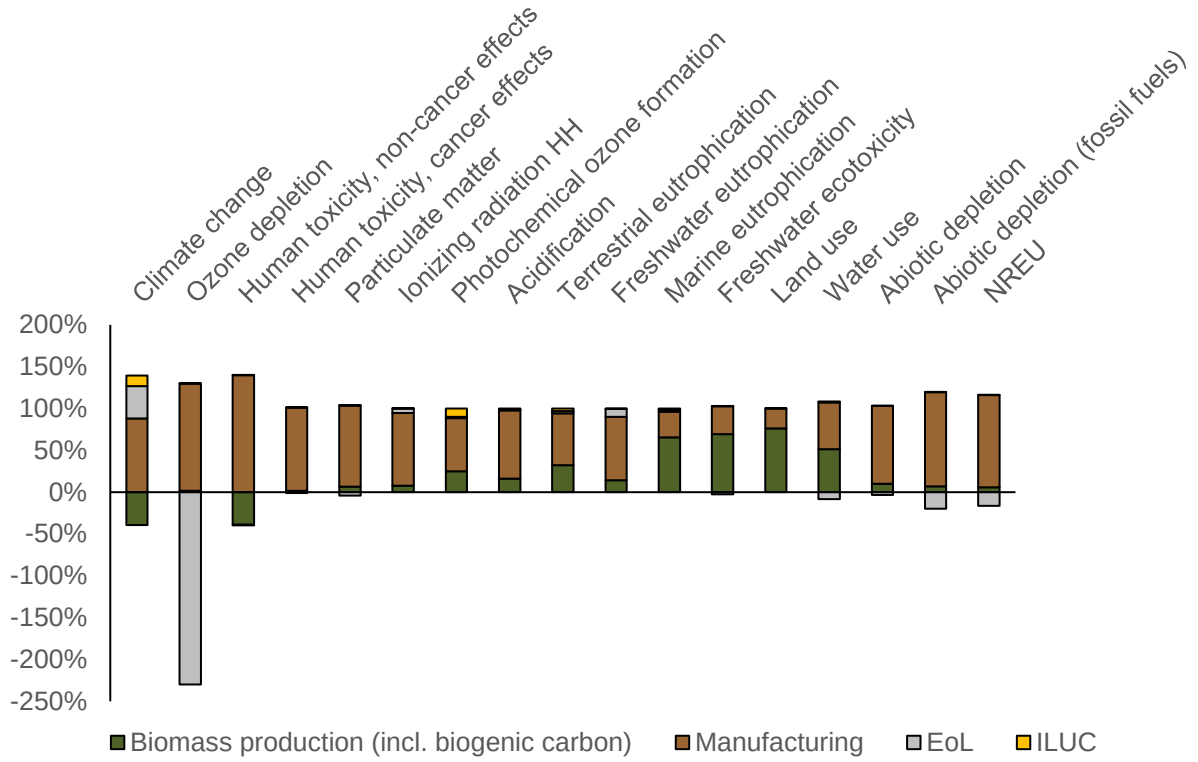


Figure 110. Relative characterised results broken down for all impact categories and including ILUC – case 3, baseline.

As it can be seen from Table 131 and to some extent from Figure 110, ILUC does contribute to all impact categories considered herein, as in previous cases. However, as for previous cases, the contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (13 % of the impact), photochemical ozone formation (10 % of the impact), terrestrial eutrophication (3 % of the impact), marine eutrophication (2 % of the impact), acidification (1.4 % of the impact) and water use (1.2 % of the impact). All these impacts are dominated by land expansion, except water use which is fully dominated by intensification (Figure 15). In the case of climate change, as explained earlier, the ILUC impact is essentially due to the CO_2 releases resulting from land clearing and in the case of photochemical ozone formation, to carbon monoxide also released during land clearing (burning the cleared biomass; Figure 17). Both acidification, terrestrial- and marine eutrophication are essentially due to nitrogen oxides released during land clearing (more than 80 % of the impact; Figure 17). The water use impact, on the other hand, is essentially reflecting the hydropower electricity demanded for the (additional) fertiliser production (intensification process), in particular phosphorus (diammonium phosphate).

In absolute terms, the ILUC impact for climate change here is quite high compared to the one obtained for e.g. case study 1 but is still lower than other cases (e.g. case study 2 and case study 5). This reflects the importance of having conversion processes requiring as little biomass as possible, so the climate change impacts from both the crop cultivation and the ILUC can be lower.

5.3 Alternative 1. PLA from EU maize: Global Temperature change potential

As regards climate change assessed with Global Temperature Change Potential (GTP)100a as indicator, it can be noticed that the cradle-to-user impact results are similar to the climate change impacts in both absolute values and breakdown shares (Figure 111).

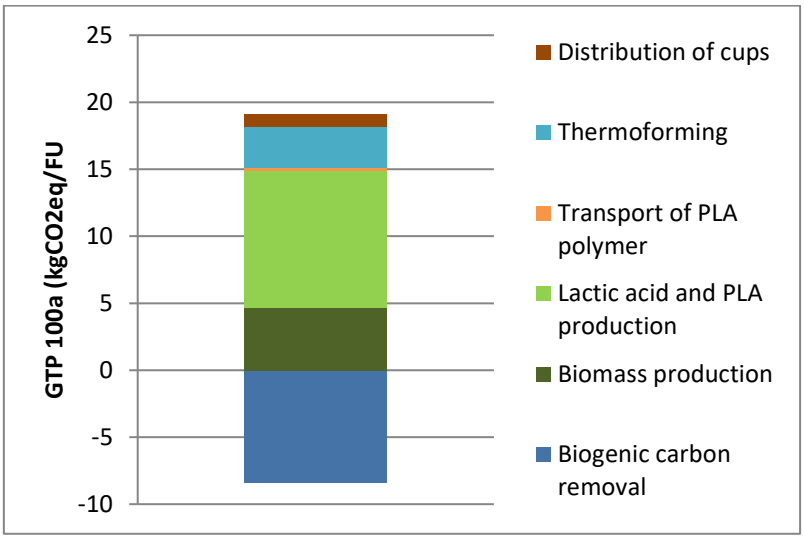


Figure 111. Breakdown of the cradle-to-gate GTP 100a results for EU PLA cups, 1 functional unit, excluding DLUC and ILUC effects.

5.3 Alternative 1. PLA from EU maize: Land use change emissions Land Use Change emissions

Table 132 presents the cradle-to-grave characterised results broken down into feedstock, manufacturing, EoL, and ILUC. As highlighted in Chapter 4, DLUC and ILUC cannot be summed up. DLUC is also presented as it is a "process" that should be considered in the LCA, according to the latest PEFCR Guidance v6.3 (European Commission, 2018). DLUC was calculated as specified in the methodology of the PEFCR Guidance v6.3, as detailed in Chapter 4. Figure 112 presents the characterised results for this variant of case 3 with the inclusion of ILUC.

Results with the inclusion of DLUC are not shown, as they are rather negligible (Table 132) and barely visible in an illustration as such Figure 112 (contributing to climate change only and making 0.003 % of the impact, when included). As explained in Chapter 4, DLUC is based on the 20-year history of crops in a given country. For the feedstock considered in this study, DLUC is only observed for a few crops, namely Brazilian sugarcane, German maize, French wheat and German wheat. Here, only one of these crops is involved (German maize). The reason why DLUC is insignificant here is reflected in Table 25 of Chapter 4 and is explained by the rather minor amount of German maize per functional unit. Again, it should be highlighted that the reason why DLUC only contributes to the climate change impact is due to the methodological choices of the PEFCR Guidance v6.3.

Table 132. Characterised cradle-to-grave results broken down per process, including ILUC. Case 3, alternative with EU maize.

Impact category	Unit	Biomass production (incl. biogenic C)	Manufacturing	EoL (EU mix)	ILUC	DLUC
Climate change	kg CO_2 eq.	-3.58E+00	1.51E+01	6.82E+00	4.34E+00	5.7E-04
Ozone depletion	kg CFC-11eq	2.12E-07	6.34E-07	-1.84E-06	9.05E-10	
Human toxicity, non-cancer effects	CTUh	6.16E-06	5.13E-07	-3.46E-08	7.26E-10	
Human toxicity, cancer effects	CTUh	1.31E-07	1.19E-06	-8.53E-09	1.16E-10	
Particulate matter	kg PM2.5eq	3.33E-03	2.97E-03	-2.64E-04	1.15E-04	
Ionizing radiation HH	kBq U235eq	2.57E-01	1.29E+00	6.77E-02	6.24E-05	
Photochemical ozone formation	kg NMVOCeq	1.58E-02	2.81E-02	1.01E-03	1.35E-02	
Acidification	molc H^+eq	9.05E-02	4.93E-02	8.25E-04	2.70E-03	
Terrestrial eutrophication	molc Neq	3.44E-01	9.78E-02	7.23E-03	1.52E-02	
Freshwater eutrophication	kg Peq	1.23E-03	3.25E-04	1.02E-04	1.00E-05	
Marine eutrophication	kg Neq	7.15E-02	9.63E-03	7.44E-04	1.61E-03	
Freshwater ecotoxicity	CTUe	1.06E+02	1.52E+01	-1.78E+00	8.05E-03	
Land use	kg C deficit	1.08E+02	7.70E+00	0.00E+00	1.25E-02	
Water use	m^3	3.89E+01	3.06E+00	-4.75E-01	1.34E-01	
Abiotic depletion	kg Sbeq	1.80E-06	8.41E-06	-4.91E-07	3.95E-08	
Abiotic depletion (fossil fuels)	MJ	3.84E+01	1.93E+02	-3.48E+01	1.13E-01	
NREU	MJ	4.58E+01	2.46E+02	-3.60E+01	1.08E-01	

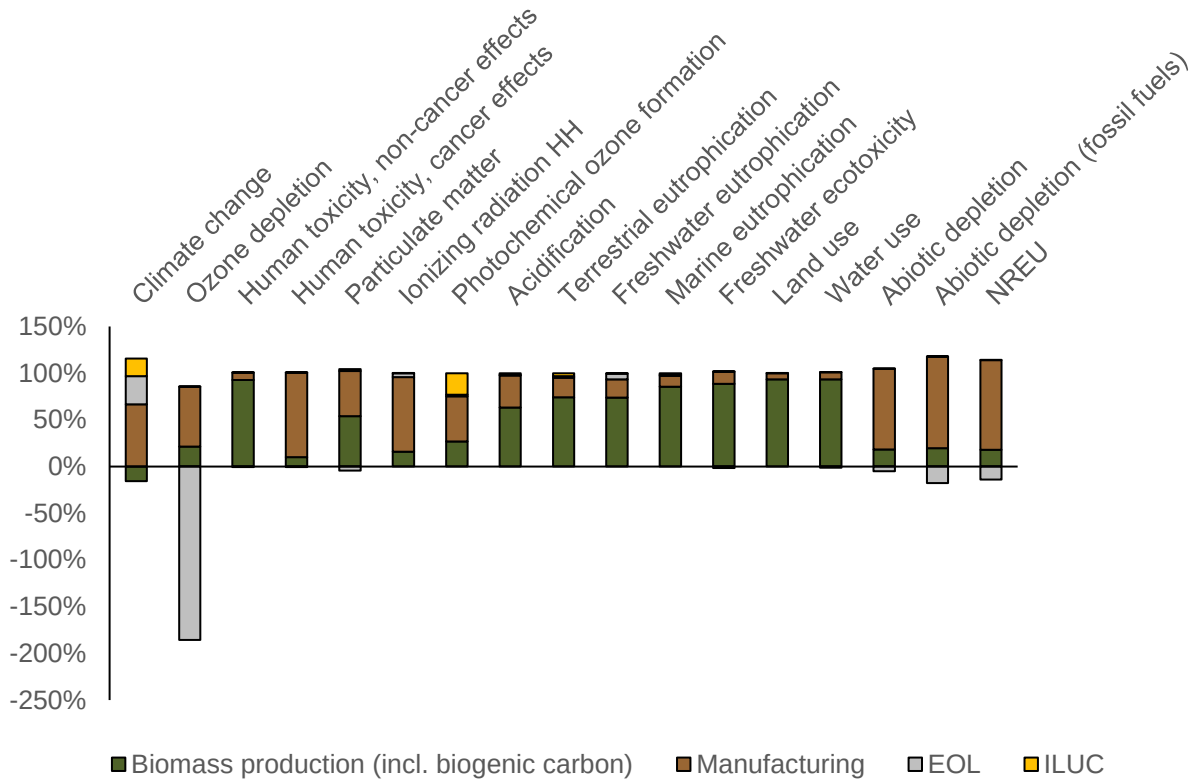


Figure 112. Relative characterised results broken down for all impact categories and including ILUC – case 3, alternative with EU maize.

As can be seen from Table 132 and to some extent from Figure 112, ILUC does contribute to all impact categories considered herein, as in previous case studies. However, as for previous cases, the contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (19 % of the impact), photochemical ozone formation (23 % of the impact), terrestrial euthrophication (3 % of the impact), marine eutrophication (2 % of the impact), acidification (2 % of the impact) and particulate matter (2 % of the impact). All these impacts are dominated by land expansion (Figure 15). In the case of climate change, as explained earlier, the ILUC impact is essentially due to the CO_2 releases resulting from land clearing and in the case of photochemical ozone formation, to carbon monoxide also released during land clearing (burning the cleared biomass; Figure 17). Acidification, terrestrial- and marine euthrophication are essentially due to nitrogen oxides released during land clearing (more than 80 % of the impact; Figure 17). The particulate matter impact is due to both the carbon monoxide and nitrogen oxides released during land clearing. It absolute terms, the ILUC climate change impact is not as high as for e.g. case study 2 or case study 5, but it is ca. twice as much the baseline impact.

6 Conclusions, limitations and recommendations

The cradle-to-grave environmental impact of different single-use cups has been assessed through the LCA methodology. PLA is one of the most important bio-based materials for single-use cups and it has been the main focus of the study. The current-near future commercial PLA has been considered as the baseline material under investigation and has been compared to two petrochemical references (PP and PET) and two alternative bio-based materials (PLA from EU maize and bio-based PP from used cooking oil).

For bio-based cups made of PLA, the most important environmental impacts are: climate change (27 %), abiotic depletion of fossil fuels (17 %), particulate matter (11 %) and acidification (10 %). When the impact categories related to toxicity are also considered, human toxicity (cancer effects) also becomes a significant impact (24 %). Independently whether toxicity impact categories are accounted for or not, the manufacturing life cycle stage is the main source of environmental burden (85-87 %) while biomass production and EoL have a minor impact. Energy consumption and chemicals used in the manufacturing process are responsible for the major parts of the impact. Within the manufacturing phase, the transportation of the polymer and the distribution of the final products also have a significant share ranging between 10 and 15 % of the total environmental burden.

A hypothetical production of PLA from European maize would not lead to environmental savings compared to both the current PLA available on the European market and the petrochemical counterparts.

Among the three bio-based product systems analysed, bio-based PP cups made from used cooking oil offer the best environmental performance and are the preferred choice, even if not biodegradable.

Out of the six impact categories identified as 'suitable for comparison' for petrochemical counterparts (See Chapter 6), from cradle-to-grave, PLA cups undoubtedly offer better environmental performance in terms of climate change -excluding land use change- and abiotic depletion of fossil fuels with benefits respectively of 22 % and 52 % compared to PET cups. PLA cups have instead higher impacts compared to PET cups in three impact categories strongly associated with agricultural production and lactic acid production, namely photochemical ozone formation (+24 %), acidification (+30 %) and terrestrial eutrophication (+68 %). Nevertheless, due to the importance of the first two impact categories in the weighted results, it is possible to conclude that PLA cups are more environmentally friendly compared to PET cups.

Compared to PP cups, PLA cups offer environmental benefits only in terms of abiotic depletion of fossil fuels (-41 %) while they have substantially higher environmental impact in particulate matter (+130 %), photochemical ozone formation (+79 %), acidification (+119 %) and terrestrial eutrophication (+143 %). The disadvantages of PLA compared to PP are mainly caused by the impact from lactic acid/PLA production. Petrochemical PP is the current competitor of PLA from an environmental point of view. For this application and in the near future, no overall environmental benefits are highlighted by the replacement of petrochemical PP cups with PLA cups.

If PLA cups could be managed through the intended EoL (50 % recycling and 50 % composting), they would offer substantial reduction in climate change impact (-48 %) compared to PP cups managed with the current EoL mix. On the other hand, PLA cups managed through the intended EoL option would still have significantly higher impact in terms of particulate matter, photochemical ozone formation, acidification and terrestrial eutrophication.

For petrochemical PP cups, significant environmental benefits are achieved in all the six impact categories by increasing the percentage of recycling as much as possible.

The weighted EoL results of the baseline product system, bio-based PLA cups, shows that recycling performs environmentally best followed by industrial composting, anaerobic digestion, incineration and landfilling. Looking at all four product systems the following option perform best to worst: recycling of petrochemical PET; recycling of bio-based PLA, recycling of petrochemical and bio-based PP; landfilling of bio-based PP; petrochemical PP and PET; industrial composting of bio-based PLA; anaerobic digestion of bio-based PLA; incineration of bio-based PLA; landfilling of bio-based PLA; incineration of petrochemical PET and lastly, incineration of bio-based PP and petrochemical PP.

Based on the weighted results with toxicity per FU for EoL mixes, the PLA is the preferred product system with regards to EoL. The PLA is followed by the petrochemical PET product system and the PP product system is the least preferred with regards to EoL. For the PLA EoL mix the climate change impact has the largest weighted impact. For the PP product systems the climate change impact together with abiotic depletion and human toxicity, cancer effects have the largest contribution to the weighted EoL mix results per FU. The PET product system shows the same three categories contributing the most and additionally freshwater ecotoxicity.

The EoL sensitivity analysis looking at the sorting and technology efficiency of the bio-based PP plastic in the recycling facility varies across Europe (from 55 % to 95 % of the plastic entering the plant which is actually recycled), which would affect the results by 1 to 109 % (looking at the seven selected impact categories), but not the overall conclusion that recycling, performs best. When applying the same type of sensitivity to the PLA product system (the sorting and technology efficiency varied from 39 % to 90 %), this leads to a low impact on the overall cradle-to-grave results (1-5 % looking at the seven selected impact categories).

This EoL sensitivity of the carbon degradation at the industrial composting facility would decrease the cradle-to-grave climate change results when increasing the C degradation by less than 0.2 % When decreasing C degradation the cradle-to-grave results will increase by a total GWP of less than 0.01 % (looking at the seven selected impact categories).

ILUC does contribute to all impact categories considered herein, as in previous cases. However, as for previous cases, the contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (13 - 19 % of the impact), photochemical ozone formation (10 - 23 % of the impact), terrestrial euthrophication (3 % of the impact), marine eutrophication (2 % of the impact), acidification (1.4 - 2 % of the impact), particulate matter (0.9 - 2 % of the impact) and water use (<1.2 % of the impact). All these impacts are dominated by land expansion. In absolute terms, the ILUC impact is, for climate change, twice as high for the variant with EU crops than in the baseline case.

Littering is excluded from this case study due to a lack of data and knowledge of the impact assessment models. The exact amount (and relative share) of plastics that are littered is uncertain for a specific application, and will vary widely between countries. The impacts of littering and microplastics are currently not properly captured in the toxicity-related impact categories (ecotoxicity and human toxicity). As pointed out by Pant et al. (2018) fate, exposure, and effect modelling for macro- and micro-plastics (incl. additives) is still in its infancy. Physical impacts (e.g. entanglement; ingestion of larger plastic particles and their effects) and chemical impacts (e.g. due to microplastic formation, microplastics as carrier of other chemicals, unknown impact of additives) and biological impacts (e.g. microplastics as carrier of germs/alien species) are largely unknown and currently not included in LCA. Thus, the impacts of littering (e.g. leaving plastics in the soil or in a marine environment) are most likely underestimated in the current LCA for the non-degradable applications.

CASE STUDY 4: SINGLE-USE CUTLERY

1 Goal and scope definition

1.1 Goal and background

The goal of this case study is to evaluate environmental impacts of a bio-based and compostable cutlery as an alternative to conventional fossil fuel-based plastic counterparts.

The disposable cutlery, generally made of polystyrene (PS), often ends up together with food waste from meals in general waste bins after use (Razza et al., 2009). The mixture of organic and plastic waste is not suitable for further recycling for economic reasons. The used plastics cutlery together with the food waste is then often disposed of in a landfill or incinerated.

Alternatively, single-use cutlery can be made from biodegradable plastics. In this case study, the used biodegradable plastics could be collected together with the food waste in the same waste stream and be disposed of by composting and/or anaerobic digestion.

Among bio-based and biodegradable polymers, polylactide acid (PLA) is an important material for the single-use cutlery application. PLA polymer is produced from lactic acid or lactide, which is a fermentation product from sugar or starch (Groot et al., 2010, Vink et al., 2015, Castro-Aguirre et al., 2016). The molecular structure of PLA is shown in Figure 113.

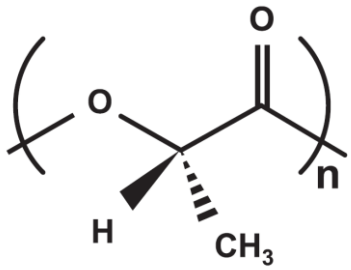


Figure 113. Molecular structure of PLA (Vink et al., 2015).

PLA can be produced in two forms - namely, poly L-lactic acid (PLLA) and poly D-lactic acid (PDLA) (Groot et al., 2010). Both forms have very similar environmental impacts, but different properties (Bie, 2018). PLLA in its amorphous form has a glass transition temperature (T_g) of 55 °C above which the polymer starts to soften (Manjure et al., 2016). However, PLLA/PDLA blends have higher heat resistance due to an increased crystallinity caused by PDLA. PLA blend with developed crystallinity can reach a T_g up to 110 °C (Product data sheet: Luminy L130, 2017), which is a sufficient temperature for application as cutlery. The PLA with increased crystallinity (and therefore improved thermal property) is called cPLA (crystalline PLA). In this case study, biodegradable cutlery is made from cPLA.

Currently, two main players – *NatureWorks LLC* and *Total Corbion* – dominate the PLA production market, with a combined market share over 90 % (Malinconico et al., 2017). The production technologies adopted by the two producers are analysed by this case study. *NatureWorks* is located in Blair, Nebraska, and *Total Corbion* is located in Rayong, Thailand. *NatureWorks* produces PLA (under the commercial name *Ingeo*) from dextrose extracted from maize cultivated in the US with an annual nameplate capacity of 150 ktons (Vink et al., 2015). *Total Corbion* produces PLA (under the commercial name *Luminy*) from raw sugar (sucrose) which is contained in sugarcane. *Total Corbion* is currently building a PLA polymerization plant to increase its production capacity up to 75 ktons p.a. of PLA granulates, which is expected to go into operation in the second half of 2018 (Corbion starts construction of 75kTpa PLA bioplastics plant [Press release], 2016). The plant is on the same site as the lactide production (75 ktons p.a. already in operation).

As was mentioned above this case study compares environmental impacts of two different cutlery systems – bio-based and fossil fuel-based. For the bio-based system, it is assumed that the PLA is sourced from a mixture of *Ingeo* and *Luminy*. The detailed scope is described below.

1.2 Scope (temporal, geographical, EoL)

The geographical scope of this case study refers to cutlery set (both bio- and fossil fuel-based) purchased, used and disposed of within Europe. The processes covering cradle to factory gate extend globally, depending on the specific supply chains.

For the bio-based cutlery, two production routes are considered. In the first route, the PLA granulates originate from maize cultivated and harvested in Nebraska and Iowa, USA, where the lactic acid and PLA production takes place too. The granulates are then transported to Southeast Asia to be converted into the end-product-cutlery. The cutlery is shipped to Europe for final distribution to end users.

In the second route, the lactic acid and PLA production are based on sugarcane cultivated in Thailand. The PLA granulates are then transported to China to be converted into cutlery and the final products are then shipped to the European market.

There is not a dominante cutlery supplier whith a majority market share in Europe as far as we investigated for this case study. The two specific supply chains using *NatureWorks* and *Total Corbion's* PLAs probably account for 10-25 % of the market share in Europe. Most cutlery suppliers in Europe we interviewed led us to manufacturers in Asia or Southeast Asia. We assume these are rather typical supply chains based on our interviews (i.e. using *NatureWorks* and *Total Corbion's* material with the compounding and cutlery production taking place somewhere in Asia).

For the fossil fuel-based cutlery, production of general purpose polystyrene (GPPS) followed by cutlery production is assumed to be in China. Manufactured cutlery is shipped to the European market.

An initial intention of this study was to investigate other bio-based cutlery from starch plastics and bamboo/wood-based materials. However, these materials were omitted from the case study due to the lack of industry willingness to support this project. Also, the project running time was too short to negotiate for a broader collaboration.

The used cutlery together with the food waste end up in an average waste management system in Europe. This infrastructure consists of incineration, landfilling and industrial composting for bio-based cutlery system, and incineration, landfilling and recycling for fossil fuel-based cutlery. More details on each EoL option (including the particular share for each option) are described in section 2.2. As mentioned in Chapter 3 "End of Life Inventory modelling", the impacts of littering are not assessed although in reality this could happen, as single-use cutlery is among the top plastic items found in litter on beaches (Werner et al., 2016).

1.3 Function and functional unit

The primary function of single-use cutlery is to serve warm meals; the functional unit of this study is defined as follows:

- 1.000 sets of single-use cutlery each consisting of a knife, a fork and a soup spoon ready to serve warm meals.

Based on the communication with industrial partners (names of the companies are confidential), the weights for the cPLA cutlery (with length of 165±5 mm) are estimated at 5.1 g, 4.1 g, and 4.4 g, for a spoon, a knife and a fork, respectively. These weights are used in the model of bio-based baseline product system (see Table 133).

Crystalline PLA (cPLA) cutlery with the same mould design is generally 35 % heavier by weight than PS cutlery (personal communication with industrial partners, confidential). Therefore, the weights for the PS cutlery are calculated at 3.8 g, 3 g and 3.3 g for a soup spoon, a knife and a fork, respectively. Table 133 shows an overview of individual cutlery weights, and the total weights per 1 functional unit which are used for calculations.

Table 134 shows the comparison of typical material properties of cPLA and PS.

Table 133. Overview of individual cutlery weights and weights per 1 functional unit.

	Unit	cPLA cutlery	PS cutlery
Spoon	g/piece	5.1	3.8
Knife	g/piece	4.1	3.0
Fork	g/piece	4.4	3.3
Total per 1 FU	kg/1.000 sets	13.6	10.1

Table 134. Comparison of typical physical and mechanical properties of cPLA and PS.

	Unit	cPLA [1]	PS [2]
Density	g/cm^3	1.24	1.05
Glass Transition Temperature	°C	100-110	82.8-94.9
Melting Temperature	°C	175	98.9-105.6
Young's Modulus*	MPa	3500	3399
Tensile Strength*	MPa	50	51.86
Bio-based content**	%	100	0
Biodegradability***	-	yes	no

Data sources: [1] (Product data sheet: Luminy L130, 2017); [2] (Product information: Polystyrol 148 H Q, 2016).

*Method: ISO 527-1.
**According to EN 16785-1 under certificate number DIC-00001.
***In compliance with EN 13432 standard.

1.4 Product systems

The bio-based baseline product systems are composed of a mixture of two production routes:

- cutlery produced in Southeast Asia using *NatureWorks'* maize-based PLA, representing 67 % of the share by weight, and

- cutlery produced in China using *Total Corbion's* sugarcane-based PLA, representing 33 % of the share by weight.

The calculation of the shares is based on the production capacity of PLA, i.e. 150 ktons p.a. for *NatureWorks* and 75 ktons p.a. for *Total Corbion*.

It is impossible to identify the real market shares of the two material producers for this specific application (a niche market). We therefore assume a so-called 'market mix' by assuming the weighted average of the production capacities of the two major PLA suppliers: namely, 67 % from *NatureWorks* and 33 % from *Total Corbion*.

The bio-based product system is compared with the reference system of fossil fuel-based PS cutlery. PS cutlery is produced in China from general purpose polystyrene whose production also occurs in China. The final product is shipped to Europe.

1.5 System boundaries

In this LCA study a cradle-to-grave approach is applied including the life cycle stages of feedstock production, transportation of intermediates, manufacturing (e.g. mixing of PLA material, injection moulding), distribution of the final product and the EoL. The consumer use phase is excluded from the assessment due to its negligible impact. Therefore, the results of the LCA are divided into two groups – cradle-to-user and EoL.

2 Life cycle inventory analysis

2.1 Process description, data, assumptions and multifunctionality

2.1.1 Baseline: Bio-Based cutlery from maize-based PLA (67 % by weight in the baseline)

Figure 114 shows the production route of maize-based cPLA. This production route represents 67 % of the environmental impact of the overall baseline system for the bio-based cutlery. The reference literature for this section is the article published by Vink et al. (2015). An overview of input data is shown in Table 135.

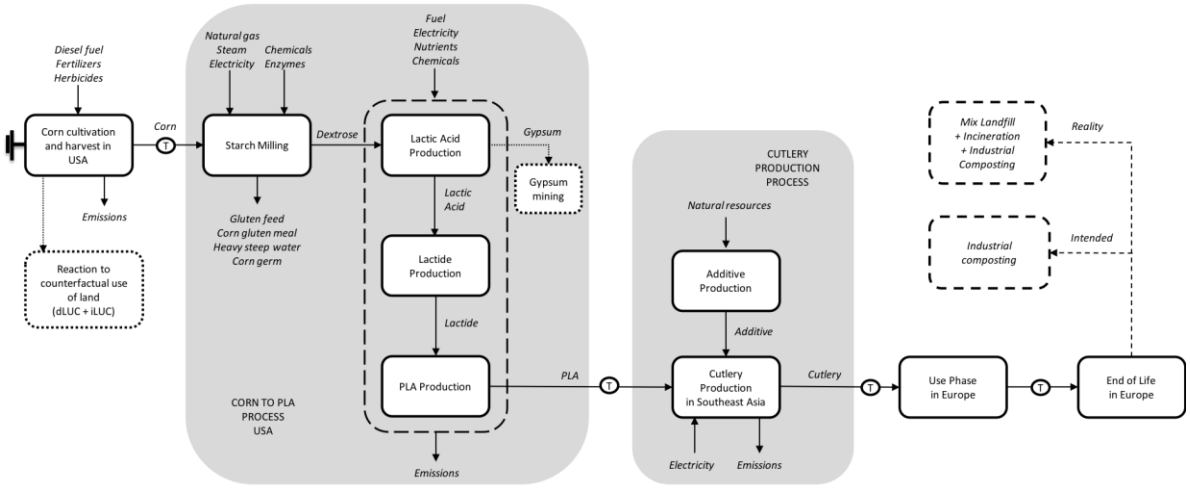


Figure 114. Process flow diagram of bio-based product system: cPLA cutlery production from maize (67 % by weight in the baseline).

Table 135. Sources of the key input data for maize-based cPLA cutlery model.

Production step	Dataset	Source of data
PLA production	Ingeo Polylactide (PLA) biopolymer production	NatureWorks' Eco-profile (2015), background data updated with Gabi 2008 database
Transportation of PLA	Transport, freight train (US) Transport, freight, sea, transoceanic ship (GLO) Transport, freight, lorry >32 metric ton, EURO4 (RER)	Ecoinvent v3.3 Modes and Distances based on confidential company data
Cutlery production	Injection Moulding (RER) Additive (material is confidential)	Confidential company data Incl. production of additive added to the final product (material and weight share are confidential) Ecoinvent v3.3 for background data
Distribution of cutlery	Transport, freight, sea, transoceanic ship (GLO) Transport, freight, lorry >32 metric ton, EURO4 (RER) Transport, freight, lorry 3.5-7.5 metric ton, EURO3 (RER)	Ecoinvent v3.3 Modes and Distances based on confidential company data and default recommendations from the PEF

2.1.1.1 Maize cultivation and harvest (in the US)

The life-cycle of *NatureWorks'* PLA starts with the maize cultivation in Nebraska and Iowa. The main inputs in this phase are agricultural chemicals (fertilisers, herbicides and pesticides), electricity and fuels for the farm activities (e.g. diesel used by the tractors), agricultural land and irrigation water. The maize production is included in the dataset *Gabi ts 8*, *Ingeo Polylactide (PLA) biopolymer production* and it has been modelled by *NatureWorks*. The current LCI data from cradle to factory gate of 1 kg of PLA polymer produced by *NatureWorks* represents the latest *NatureWorks'* eco-profile for PLA (Vink et al., 2015) with updated background data using Gabi database version 2018 (personal communication with E. Vink in Jan-June 2018). The site-specific data for the maize used for *NatureWorks* PLA have been used in this part of the LCI. The yield is assumed to be 10.28 ton/ha/yr for maize with 15 % moisture by weight (Vink et al., 2015).

2.1.1.2 Maize to PLA conversion (in the US)

The harvested maize is transported to the maize wet milling factory where dextrose is produced via enzymatic conversions. The starch content (estimated to be 58 % of the mass content based on Vink et al. (2015)) in the maize is hydrolysed to dextrose. The co-products of the starch-milling and their share of the dry mass-basis are: maize oil (about 3 %), gluten meal (about 5 %) and gluten feed (about 25 %) (Vink et al., 2015). During dextrose production steam and electricity are needed and sourced locally (Nebraska, US infrastructure). To solve this case of multi-functionality, sub-division was performed by Vink et al. (2015) in the *NatureWorks' eco-profile*. The unit process is divided into 11 sub-processes (Vink et al., 2015) and, for each sub-process, where an allocation is needed a mass allocation (on a dry basis) is applied. Dextrose is sent to the fermentation unit to produce lactic acid which is the precursor of the polymer PLA. For each kg of PLA, 1.57 kg of maize is required (Vink et al., 2015). In the downstream process of lactic acid production gypsum is produced as a by-product in small quantities. A credit was assigned to this co-production through the substitution of primary production of gypsum from mining (Vink et al., 2015). The entire starch/sugar milling, lactic acid production and PLA polymerisation are in an integrated production site which is located in Blair (Nebraska).

The LCI from maize cultivation to PLA production, modelled by *NatureWorks* reported in the latest eco-profiles of *Ingeo™* PLA*,* with the background data updated with Gabi 2018 datasets, is not publicly available.

2.1.1.3 Transportation of PLA granulates

Produced PLA polymer in granulate form is transported from Nebraska to New York. The distance of transport is estimated at 2000 km by cargo train (personal communication with E. Vink from *NatureWorks*). The granulates are then shipped to the cutlery manufacturer in Southeast Asia in containerships. The distance is estimated at 18500 km from the port of New York to a harbour in SE Asia (specific location is confidential). Truck transportation of 100 km is assumed from the harbour to the manufacturer site. The processes *Transport, freight train (US)*, *Transport, freight, sea, transoceanic ship (GLO)* and *Transport, freight, lorry >32 metric ton, EURO4 (RER)*, from *Ecoinvent v3.3,* are used in the model.

2.1.1.4 Cutlery production (including additive production) in SE Asia

After the polymer is shipped to the cutlery production site, it is mixed with additives (confidential information) and injection moulded into the final product – cutlery.[58] The production process includes three main processes: mixing, injection moulding and annealing. The key input for this production process is electricity. According to the interviewed cutlery producers, the production of cPLA cutlery requires 30 % higher electricity consumption than the production of PS cutlery. This is caused by longer cycle/batch time for cPLA cutlery in order to increase the crystallinity through the material during the annealing phase.

According to the producers, the whole cutlery processing is a dry procedure and there is neither water use nor waste water from the production. No solid waste is generated from this unit process as close-loop recycling is achieved. In this study, the mass efficiency of converting PLA granulates to cutlery is assumed to be 100 %.

The cutlery production is modelled by modifying the foreground data of the unit process "*Injection Moulding (RER)*" from Ecoinvent v3.3 in two respects:

- The foreground data of electricity consumption is updated based on primary data collected from the industry using local grid electricity (detailed information on the location is confidential); Based on the information from several cutlery producers, an average value is modelled for the baseline model.

- The construction of the injection moulding factory is excluded for two reasons: 1) construction of infrastructure is excluded in the system boundary (see the generic methodology described in Chapter 3), and 2) the construction of a generic chemical factory in Ecoinvent is not suitable to represent the injection moulding and cutlery production factory; the latter infrastructure needs to accommodate only physical conversions (not chemical).

- The environmental impact of the specific (and confidential) additive(s) is included. The background data of the production of the additive(s) production is based on *Ecoinvent v3.3*.

2.1.1.5 Shipping to Europe and distribution to end users

[58] Sometimes master batches are also produced. But this is not operated by the investigated supply chain.

After manufacturing the cutlery is shipped to Europe and distributed into the European market. The transportation between the cutlery production site in SE Asia and the distribution centre in Europe is estimated at 15000 km by ship and 1000 km by truck. Another 250 km of van transport is added to reflect the transport between distribution centre and the final user. The distances for truck and van transportation are estimated according to the PEFCR guidance v6.3 for the default distances. The processes *Transport, freight, sea, transoceanic ship (GLO)*, *Transport, freight, lorry >32 metric ton, EURO4 (RER)* and *Transport, freight, lorry 3.5-7.5 metric ton, EURO3 (RER)*, from *Ecoinvent v3.3*, are used in the model.

2.1.1.6 Multi-functionality

The PLA production has several multi-output unit processes. As explained in Chapter 2 "Life-cycle inventory modelling" in "Approach and methodology", for multioutput processes, allocation is avoided whenever possible by system expansion or subdivision. If allocation cannot be avoided, allocation is applied based on physical causality and/or economic outputs.

During the starch milling process co-production of several co-products occurs. These co-products are maize oil, gluten meal and gluten feed, in addition to starch (estimated as 58 % on a dry mass basis). Based on *NatureWorks* 2015's eco-profile, sub-division was performed (see also 2.1.1 of this case study report). The starch milling process is divided into 11 sub-processes. Based on this modelling choice, 1.57 kg of maize is required to each kg of produced PLA.

In lactide acid production, gypsum is produced as a co-product in small quantities from downstream separation process. A credit is assigned to co-production through the substitution of mined gypsum – a system expansion is applied.

2.1.2 Baseline: Bio-Based cutlery from sugarcane-based PLA (33 % by weight in the baseline)

This section describes the process for the sugarcane-based cPLA cutlery production (see Figure 115) and this route represents 33 % by weight of the bio-based baseline system. Sugarcane is cultivated in Thailand and the processes from cultivation to PLA production are based on *Total Corbion's* earlier publication Groot et al. (2010) and updated with the latest life cycle inventory data based on actual production data provided by *Total Corbion* (dataset is not publicly available). A summary of inventory data used for this system is presented in Table 136.

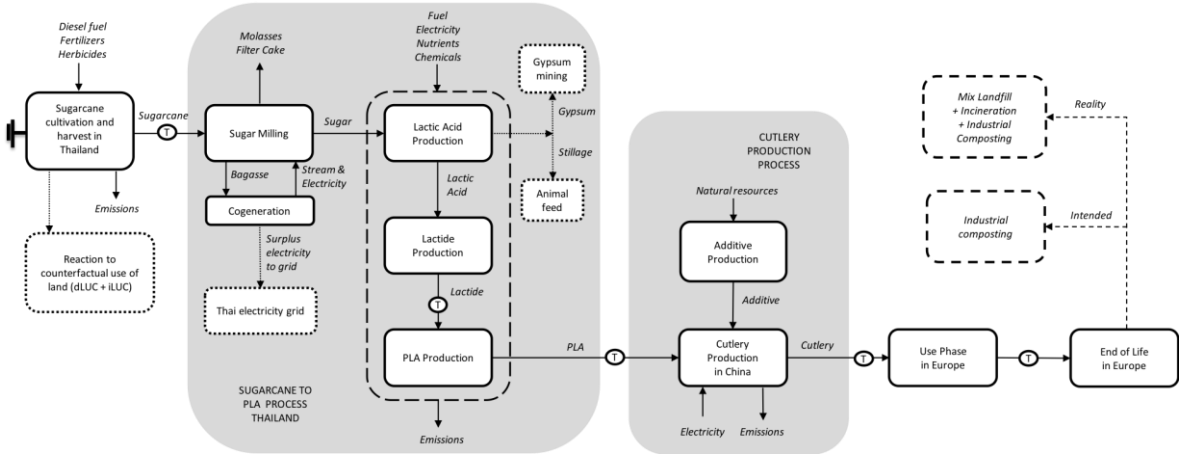


Figure 115. Process flow diagram of bio-based product system: cPLA cutlery production from sugarcane (33 % by weight in the baseline).

Table 136. Sources of input data for sugarcane-based cPLA cutlery model.

Production step	Dataset	Source of data
Sugarcane cultivation and PLA production	Confidential dataset	Groot et al. (2010) updated with the latest LCI provided by Total Corbion (Confidential)
Transportation of PLA	Transport, freight, sea, transoceanic ship (GLO) Transport, freight, lorry >32 metric ton, EURO4 (RER)	Ecoinvent v3.3 Modes and Distances based on confidential company data
Cutlery production	Injection Moulding (RER) Additive (confidential)	Foreground data is confidential company data including production of additive Ecoinvent v3.3 for background data
Distribution of cutlery	Transport, freight, sea, transoceanic ship (GLO) Transport, freight, lorry >32 metric ton, EURO4 (RER) Transport, freight, lorry 3.5-7.5 metric ton, EURO3 (RER)	Ecoinvent v3.3 Modes and Distances based on confidential company data in Southeast Asia. Default recommendations from the PEFCR guidance for the transportation services in Europe.

2.1.2.1 Sugarcane cultivation (in Thailand)

Sugarcane is cultivated in Thailand. Major activities involved in sugarcane cultivation consist of land preparation, use of fertilisers and pesticides, and harvesting. Although, harvesting is 90 % performed manually in Thailand (therefore no emissions from farm vehicles), farmers still practice a pre-harvest burning of sugarcane which make the final harvesting easier (Yuttitham et al., 2011, Prasara-A et al., 2016). Also, the majority of sugarcane residues is burnt in open fields (about 60 % in 2012) (Jenjariyakosoln et al., 2013). Dataset on sugarcane cultivation was provided by *Total Corbion* and is not publicly available.

2.1.2.2 Sugarcane to PLA conversion (in Thailand)

Harvested sugarcane is collected at a sugarmill where it is squeezed to extract a cane juice. The juice is evaporated, crystallised and centrifuged to make raw sugar. The co-products of

this process are molasses (4.2 %w), filter cake (4.5 %w) and bagasse (34.0 %w) (Prueksakorn et al., 2014). The sugar milling of sugarcane requires heat and electricity which are entirely produced by burning bagasse in a combined heat and power (CHP) plant incorporated into the sugar mill. The surplus of electricity is sold to the Thai national grid (Groot et al., 2010, Morão et al., 2018). The sugar is then transported and processed into the lactic acid plant located next to the mill. During the lactic acid production process, gypsum and stillage are produced as by-products and are removed through filtration and sold (Groot et al., 2010). This gypsum has similar characteristics to mined gypsum while the stillage is used as animal feed due to its nutritional properties – a system expansion is applied.

2.1.2.3 Transportation of PLA granulates

Produced PLA granulates are transported from Rayong to the cutlery producer in China. The distance for this transportation is estimated at 100 km by truck (distance between harbours and production facilities) and 2700 km by ship. The distances and mode of transportation are based on the specific supply chains investigated. Detailed locations are confidential. The processes *Transport, freight, sea, transoceanic ship (GLO)* and *Transport, freight, lorry >32 metric ton, EURO4 (RER),* from *Ecoinvent v3.3*, are used as the background data in the model.

2.1.2.4 Cutlery production (including additive production) in China

In this production route, the technology of cutlery production is the same as described in section 2.1.1.4. The cutlery production occurs in China therefore the process *Electricity, medium voltage (CN)* from *Ecoinvent v3.3* is used in customised dataset for injection moulding. The composition of cutlery (confidential) and the energy consumption (also confidential) are the same as those for the maize-based cPLA cutlery.

2.1.2.5 Shipping to Europe and distribution to end users

The distance between the factory in China and distribution centre in a European harbour is estimated at 18000 km by ship and 1000 km by truck. Another 250 km by van transportation is added to reflect the transportation between distribution centre and the final user in Europe. The distances for truck and van transport are estimated according to the PEFCR guidance v6.3 for the default distances. The processes *Transport, freight, sea, transoceanic ship (GLO)*, *Transport, freight, lorry >32 metric ton, EURO4 (RER)* and *Transport, freight, lorry 3.5-7.5 metric ton, EURO3 (RER),* from *Ecoinvent v3.3*, are used as the background data in the model.

2.1.2.6 Multi-functionality

Bagasse, one of the sugar production by-products, is burnt to generate electricity and heat which are used to run the sugarmill, with the surplus sold to the Thai national grid (Groot et al., 2010, Morão et al., 2018). The filter cake and stillage are returned back to the field as a soil conditioner, thus staying within the system boundaries (Morão et al., 2018).

In the lactide acid purification steps, gypsum is produced in small quantities therefore a credit is assigned to substitute gypsum made from primary mining.

2.1.3 Reference: Fossil fuel-based PS cutlery

Figure 116 shows the production of polystyrene cutlery. An overview of production steps and their source of data are given in Table 137.

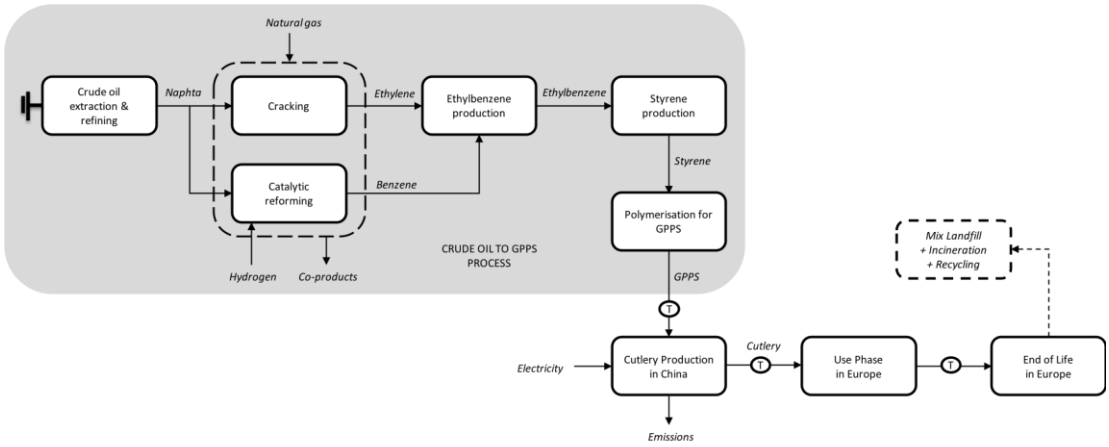


Figure 116. Process flow diagram of fossil fuel-based product system: PS cutlery production (adapted based on PlasticsEurope's eco-profile for GPPS (PlasticsEurope, 2012).

Table 137. Sources of input data for PS cutlery model.

Production step	Dataset	Source of data
PS production	General purpose polystyrene (GPPS)	PlasticsEurope 2010 Eco-profile updated with Gabi 2011 database[59]
Transportation of PS	Transport, freight, lorry >32 metric ton, EURO4 (RER)	Ecoinvent v3.3 Modes and Distances based on default recommendations from the PEF
Cutlery production	Injection Moulding (RER)	Confidential company data Ecoinvent v3.3 for background data
Distribution of cutlery	Transport, freight, sea, transoceanic ship (GLO) Transport, freight, lorry >32 metric ton, EURO4 (RER) Transport, freight, lorry 3.5-7.5 metric ton, EURO3 (RER)	Ecoinvent v3.3 Modes and Distances based on confidential company data and default recommendations from the PEF

2.1.3.1 Polystyrene production (China)

Cracking of naphtha forms many products, including ethylene and benzene which are converted to ethylbenzene by alkylation. Ethylbenzene is turned to styrene by dehydrogenation, followed by polymerization to polystyrene. A dataset for this process – PlasticsEurope eco-profile *General purpose polystyrene (GPPS)/EU-28* – is obtained from *Gabi ts 8* database.

[59] See Chapter 6 Ranges for environmental impacts from production of fossil-based plastics.

The exact source of PS cannot by identified, but based on an interview with cutlery producers (who process both PLA and PS cutleries) PS granulates are produced in China. The LCI data of average Chinese refineries is not available. Instead, we use PlasticsEurope's eco-profile to approximate the Chinese PS production. We are aware that that PlasticsEurope's eco-profiles are based on Western Europe's refinery and polymer production data, which represent the state-of-the-art PS production, and PS produced in China could have higher environmental impacts due to more carbon-intensive (e.g. coal-dependent) energy infrastructure in China[60]. However, since no transparency can be obtained from the available LCI data from either the PlasticsEurope's Eco-profiles or from a Chinese refinery/polystyrene producer, it is not possible to quantify the differences.

2.1.3.2 Transportation of PS granulates

The transport distance of PS granulates from production factory to cutlery manufacturer is assumed to be 1000 km by truck using the *Transport, freight, lorry >32 metric ton, EURO4 (RER)* process from the *Ecoinvent v3.3* database for the modelling.

2.1.3.3 PS cutlery production (China)

According to the cutlery producers interviewed, cPLA cutlery production consumes 30 % more energy than PS cutlery. The energy consumption, assumed in the inventory model, uses Chinese grid electricity as the background data.

In this product system, the cutlery production occurs in China therefore the process *Electricity, medium voltage (CN)* from *Ecoinvent v3.3* is used in modified dataset for injection moulding (the process of PS cutlery production is the same as for cPLA cutlery, see section 2.1.1.4) which represents the cutlery production. The exact composition of PS cutlery in this product system is confidential.

2.1.3.4 Shipping to Europe and distribution to end users

The distance between the factory in China and the European distribution centre is estimated at 1000 km by truck (from the Chinese factory to a Chinese harbour) and 18000 km by ship (from the Chinese harbour to a European harbour). Another 250 km of van transport are added to reflect the transport between distribution centre and the final user. The processes *Transport, freight, sea, transoceanic ship (GLO)*, *Transport, freight, lorry >32 metric ton, EURO4 (RER)* and *Transport, freight, lorry 3.5-7.5 metric ton, EURO3 (RER),* from *Ecoinvent v3.3,* are used in the model.

[60] According to IEA (2017), approximately 66.7 % of the total primary energy supply was based on combustion of coal in 2015 in China.

2.2 End of Life description, data, assumptions and multifunctionality

The single-use cutlery has different possible EoL options depending on the product system, as observed in Table 138. The petrochemical PS single-use cutlery has three possible EoL options:

- **Mechanical plastic recycling**: the process includes the energy and material requirements for the transportation to the facility, sorting, cleaning and recycling processes. The recycled PS is assumed to substitute virgin PS production. The rejects from the recycling process are sent to incineration (both with and without energy recovery) and includes transportation. The flowchart for plastic recycling is presented in Figure 8.

- **MSW Incineration with and without energy recovery**: a generic MSW incineration plant is assumed, which represents average EU waste incineration with and without energy recovery with an overall heat efficiency of 22 % and electrical efficiency of 9 % (CEWEP, 2013; Eurostat, 2017a). The process includes the direct emissions from waste incineration as well as indirect emissions from the production of the input materials, the combustion of fossil fuels as well as treatment of bottom ash and fly ash. The flowchart for MSW incineration is presented in Figure 9.

- **MSW landfilling**: the process includes the construction and operation of an average landfill site, direct emissions of landfill gas, landfill gas collection, flaring and recovery system, treatment of collected leachate and direct emissions from uncollected leachate. 29 % of the collected landfill gas is used to generate electricity (OpenLCA Nexus, 2015). The electricity conversion efficiency is 37 % (Christensen, 2011). The flowchart for MSW landfilling is presented in Figure 10.

The PLA mix cutlery has the following EoL options:

- **MSW Incineration with and without energy recovery**: a generic MSW incineration plant is assumed, which represents average EU waste incineration with and without energy recovery with an overall heat efficiency of 22 % and electrical efficiency of 9 % (CEWEP, 2013; Eurostat, 2017a). The process includes the direct emissions from waste incineration as well as indirect emissions from the production of the input materials, the combustion of fossil fuels as well as treatment of bottom ash and fly ash. The flowchart for MSW incineration is presented in Figure 9.

- **MSW landfilling**: the process includes the construction and operation of an average landfill site, direct emissions of landfill gas, landfill gas collection, flaring and recovery system, treatment of collected leachate and direct emissions from uncollected leachate. Twenty-nine percent of the collected landfill gas is used to generate electricity (OpenLCA Nexus, 2015). The electricity conversion efficiency is 37 % (Christensen, 2011). The flowchart for MSW landfilling is presented in Figure 10.

- **Industrial composting**: includes indirect emissions from material and energy consumption, direct emissions from the plant and use on land, fertiliser substitution from compost and rejects sent to incineration (30 % of plastic and 5 % of organic waste). The flowchart for industrial composting is presented in Figure 12.

- **Anaerobic digestion**: a mix of technologies is modelled (with and without post-maturation (aerobic)), which includes indirect emissions from material and energy consumption, pre-treatment (rejects sent to incineration - 30 % of plastic and 5 % of organic waste), electricity, heat and fuel generation from biogas, direct emissions from the plant and use on land, and fertiliser substitution from compost and digestate. The flowchart for anaerobic digestion is presented in Figure 13.

The intended EoL technology for the PLA mix cutlery is industrial composting.

In addition to modelling full (100 %) recycling, and incineration, landfilling, industrial composting and anaerobic digestion individually, a mix of these EoL scenarios was also modelled to represent the current situation in Europe. These figures are for the petrochemical EoL mix based on the available waste statistics in Europe on plastic waste treatment. The current collection for treatment of plastic waste is 30 % recycling, 39 % incineration and 31 % landfilling in Europe (European Commission, 2018). It was not possible within this study to collect data on the waste collection data of cutlery (neither PS or PLA).

The percentage of PS in packaging and other types of plastic materials is foreseen to be less in the future which can result in fewer available recycling facilities willing to accept PS. The sorting and technology efficiency was set to 25%, assuming that a large part of the PS is sorted out by the recycling facilities and sent to incineration.

For the EoL mix for the PLA cutlery it is estimated that Europeans sort the same amount of PLA cutlery for composting, which they today sort for recycling of commercial plastics, hence 30 %. The application of a general recycling percentage for bio-based as well as petrochemical plastic (based on how much plastic is currently collected for recycling in the EU) in the EoL technology mix may be questioned for especially this case study. However, keeping the share of the intended EoL technology constant between different product systems ensures that the comparison of bio-based and petrochemical products is not biased by assumed differences in the waste system. With the new targets of separate collection of organic waste, an increase in separate collection of organic waste is expected within the study's timeframe of the coming 5-10 years. Eurostat reports in 2014 indicate there were three countries that had no recycling of plastic as well as three countries that did not recycle organic waste, hence it was estimated in the original scenarios that an equal amount of EU members sort out organic waste and plastic waste (Eurostat, 2017c). To support this, a report by Copenhagen Resource Institute and BiPRO for the European Commission on *Assessment of separate collection schemes in the 28 capitals of the EU*, shows that on average the biowaste capture rate is 16 % while the plastic capture rate is 11 % (Seyring et al., 2015).

Table 138. The EoL options, their intended EoL option and the estimated EoL mix.

Product system	EoL options modelled individually (100 % scenarios)	Intended EoL option	Estimated EoL collection mix of EU (based on statistics of waste treatments of waste plastics)
Bio-based PLA mix	Incineration Landfilling Industrial composting Anaerobic digestion	Industrial composting	Industrial composting: 30 % Incineration: 39 % Landfilling: 31 %
Petrochemical PS	Recycling Incineration Landfilling		Recycling: 30 % Incineration: 39 % Landfilling: 31 %

The average European EoL technologies are described in the approach and methodology chapter in the section regarding EoL. In that section, factors that do not depend on the specific product system are described, e.g. the energy efficiency of the incineration plant and the consumption of energy and materials in each technology. Furthermore, principles for substitution are described. This involves the substitution of the energy produced in the EoL technologies with marginal electricity and heat, as well as the recycling substitution methodology.

In addition, there are several product system dependent factors, which are presented in Table 139, for each possible EoL technology. The recycled PS plastic is substituted with petrochemical PS.

The waste data on general plastics is not assumed to be applicable for the PS cutlery. EUCertplast (2018) states that fewer recycling facilities sort out PS, which is why the sorting and technology efficiency for PS is set to a lower rate than the other plastic polymers (PE, PP and PET). A sensitivity analysis was performed by increasing the amount of PS that is recycled (the sorting and technology efficiency as well as the collection rate), as recycling of PS can be difficult due to technical limitations today (Plastics Recyclers Europe, 2017). The recycling rate is calculated by multiplying the collection rate and the sorting and technology efficiency.

Table 139. Values for product system dependent factors for single-use cutlery.

EoL Technology	Product system dependent factor	Unit	Bio-based PLA mix	Reference	Petrochemical PS	Reference
Plastics recycling	Sorting and technology efficiency (amount collected for recycling which is recycled)	% of input			25 %	Estimated based on Dall et al., 2013 and Plastic Recyclers Europe, 2017
MSWI with and witout energy recovery	Energy content	MJ/kg	15.1	Boonmee et al., 2016; Pradhan et al., 2010; Hermann et al., 2010; Krause and Townsend., 2016; Kolstad et al., 2012 and Krupaka et al., 1985	36.0	Götze et al., 2016
Landfilling	1st oder decay rate for methane generation	1/s	0.011	Calculated based on (Kolstad et al., 2012)	0	Estimated
Industrial composition	VS Degradation (%)	% of VS	80 %	Pradhan et al., 2010		
	C Degradation (%)	% of C bio	57 %	Pradhan et al., 2010		
Anaerobic digestion	Anaerobically biodegradable biogenic carbon	% of C bio	47 %	Pradhan et al., 2010		

Disposing of cutlery will cause an increase in food leftovers that will enter the EoL technologies. For all technologies, except recycling of PS, the food leftovers will enter the same technology as the cutlery is modelled to enter. This is the case because for example,. if cutlery in one country or region will enter a waste stream which goes to incineration, the food leftovers will be assumed to enter the same stream. For recycling of the petrochemical cutlery,

a distribution for the food leftovers is assumed, based on Eurostat statistics of MSW; 53 % incineration and 47 % landfilling (Eurostat. 2017a).

The amount of extra food leftovers is calculated based on the weight of the cutlery and a study from Silvennoinen et al. (2015) and is also similar to the food leftovers utilised in an LCA by Razza et al. (2008). The study states that the amount of food leftovers ranges from 58-189 g/portion. A portion is in the current study defined as a plate and a set of cutlery. The different weight of the two product systems were considered and an attribution of the food leftovers to the plate as well. The number of portions per 1 kg of cutlery was calculated and then the range of food leftovers from 1 kg of plastic was calculated[61] to be

- 2.5 - 8.0 kg food leftovers, averaging 5.2 kg food leftovers / kg PLA mix; and
- 2.9 - 9.4 kg food leftovers, averaging 6.1 kg food leftovers / kg petrochemical PS.

The average was used for the scenarios, and the range was used for the sensitivity analysis of industrial composting of PLA.

The food leftovers consist of vegetable and animal food waste, in accordance with the functional unit. The chemical composition used to calculate the environmental performance was based on several waste characterisations (EASETECH). The chemical composition can be seen in Annex 1 Life cycle inventories for EoL.

The single-use cutlery has the following material composition, see Table 140, which is utilised as the input to the EoL LCA model. The full chemical composition is presented in Annex 1.

The EoL-reference flow of the EoL modelling of the single-use cutlery is

- PLA: 1 kg plastic + 100 g contamination + 5.2 kg food leftover
- PS: 1 kg plastic + 100 g contamination + 6.1 kg food leftover

Table 140. Material composition for single-use cutlery (1 kg plastic + 100 g contamination).

Chemical component	Unit	Bio-based PLA mix	Petrochemical PS
Water	% of total	6.6	9.5
TS (VS+ash)	% of total	92.5	90.5
VS	% of TS	80.5	97.9
C fossil	% of TS	0.2	80.9
C biogenic	% of TS	40.5	4.5
Reference		Boonmee et al., 2016; Pradhan et al., 2010; Hermann et al., 2010; Krause and Townsend., 2016; Kolstad et al., 2012	Götze et al., 2016

[61] The weight of a set of cutlery (spoon, knife and fork) is for the cPLA mix 13.6 g / set and PS 10.1 g / set. The weight of a different types of single-use plates (of petrochemical plastic, bio-based plastics and cardboard) was found to be around 10 g / plate. This is utilized to make a weight attribution between the cutlery and the plate for the food waste.

3 Life cycle impact assessment results and interpretation

3.1 Cradle-to-user environmental impacts and interpretation

3.1.1 Cradle-to-user environmental results and interpretation, Bio-Based baseline system

Overall results of cradle-to-user environmental impacts of the baseline system (bio-based cutlery) are show in Table 141. The table presents results for all 16 PEFCR impact categories and thecategory for non-renewable energy use (NREU). The breakdown of the life-cycle stages of the baseline system (bio-based cutlery) is presented in Figure 117.

Table 141. Cradle-to-user life cycle impact assessment results for the baseline (cutlery made from PLA, weighted average of two PLA sources, see text, excluding DLUC and ILUC effects).

Impact category	Unit	Total impact per 1 FU of cPLA cutlery (cradle-to-user)	Total impact per 1 kg of cPLA cutlery (cradle-to-user)
Climate change	kg CO_2 eq.	3.79E+01	2.79E+00
Ozone depletion	kg CFC-11eq	3.19E-06	2.35E-07
Human toxicity, non-cancer effects	CTUh	8.07E-06	5.93E-07
Human toxicity, cancer effects	CTUh	2.04E-06	1.50E-07
Particulate matter	kg PM2.5eq	3.43E-02	2.52E-03
Ionizing radiation HH	kBq U235eq	2.27E+00	1.67E-01
Photochemical ozone formation	kg NMVOCeq	2.23E-01	1.64E-02
Acidification	molc H+eq	3.90E-01	2.87E-02
Terrestrial eutrophication	molc Neq	8.56E-01	6.30E-02
Freshwater eutrophication	kg Peq	2.35E-03	1.73E-04
Marine eutrophication	kg Neq	1.17E-01	8.63E-03
Freshwater ecotoxicity	CTUe	1.65E+02	1.21E+01
Land use	kg C deficit	2.31E+02	1.70E+01
Water use	m3	1.60E+01	1.17E+00
Abiotic depletion	kg Sbeq	4.31E-05	3.17E-06
Abiotic depletion (fossil fuels)	MJ	7.48E+02	5.50E+01
NREU	MJ	8.01E+02	5.89E+01

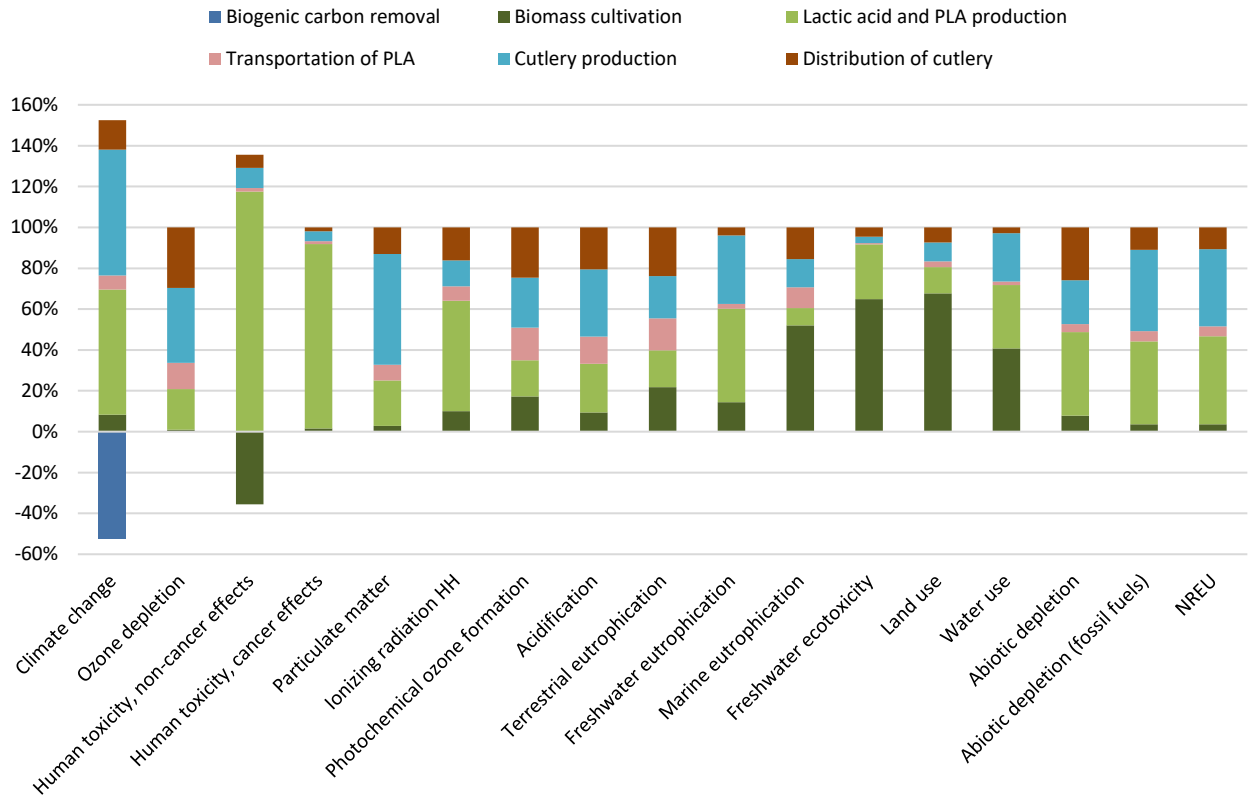


Figure 117. Breakdown of the cradle-to-user results for the baseline (cutlery made from PLA, weighted average of two PLA sources, see text, excluding DLUC and ILUC effects).

Based on the results from cradle-to-user impact assessment for the baseline, the following considerations can be highlighted for the impact of each life-cycle step:

- Biomass cultivation (mix of maize from the US and sugarcane from Thailand) is the main environmental burden in three categories with an overall cradle-to-user impact of more than 50 % – namely, marine eutrophication, freshwater ecotoxicity, and land use. A significant impact of this life-cycle stage is also observed in the category of water use where the impact reaches 41 %. The negative impact on human toxicity (non-cancer effects) is caused by the zinc absorbed from the soil by sugarcane during its growing. This zinc uptake becomes higher when the sugarcane is cultivated in contaminated areas. The highest zinc accumulation is in the root of the sugarcane, this is followed by bagasse, underground steam, leaves and juice (Sampanpanish & Tantitheerasak, 2015). A small impact (less than 5 %) of the cultivation is observed in the following five categories: ozone depletion, human toxicity (cancer effects), particulate matter, abiotic depletion (fossil fuels) and non-renewable energy use.

- The lactic acid and PLA production is the main cause of impact in the categories of climate change, human toxicity (both cancer and non-cancer) and ionizing radiation HH. Significant impact of this life-cycle step are also observed in four other categories – freshwater eutrophication, abiotic depletion, abiotic depletion (fossil fuels) and non-renewable energy use. In the category of water use the lactic acid and PLA production life-cycle step contributes 31 % of the total cradle-to-user water use impact.

- Transport of PLA granulates, which includes also transoceanic shipping, has a higher impact (10-20 %) in the categories of ozone depletion, photochemical ozone depletion,

acidification, and terrestrial and marine eutrophication. In the remaining categories the impact is less than 10 %.

- Cutlery production causes the main environmental burden in the categories of climate change and particulate matter (41 % and 54 %, respectively; excluding biogenic carbon removal). A significant impact (30-50 %) is also observed for ozone depletion, acidification, freshwater eutrophication, abiotic depletion (fossil fuels) and non-renewable energy use. Cutlery production has higher impacts in the remaining categories too. The environmental burden in this step primarily originates from production of electricity which is used during the cutlery production.

- Environmental impacts of cutlery distribution from Asian producers to European end users are mainly caused by emissions from transoceanic shipping. A higher burden from this life-cycle step can be observed for ozone depletion (30 %), photochemical ozone formation (25 %), acidification (21 %), terrestrial (24 %) and marine (16 %) eutrophication and abiotic depletion (26 %). Small impacts (less than 3 %) are observed for human toxicity (cancer effects) and water use.

According to the PEFCR guideline, the impact categories that cumulatively contribute to 80 % of the total environmental impact (excluding toxicity related impact categories) need to be interpreted in detail. Based on the results from weighting (see section 3.3.1), the following eight categories cover comulatively more than 80 % of the total cradle-to-grave impact: climate change, abiotic depletion (fossil fuels), particulate matter, acidification, photochemical ozone depletion and freshwater eutrophication.

These categories are discussed more in detail in following paragraphs in the order from the highest to the lowest contribution to the overall impact:

- **Climate change.** The cradle-to-user impact of climate change (including biogenic carbon removal) is estimated to be 2.79 kg CO_2 eq. per kg of cPLA cutlery[62] (see Table 141). If biogenic carbon removal is not accounted for, the gross GHG emissions are 4.25 kg CO_2 eq./kg of cPLA cutlery. To better explain the carbon balance in the climate change category, Figure 118 shows the breakdown of GHG emissions. Most of the gross GHG emissions (more than 80 %, excluding biogenic carbon removal) come from two life-cycle stages – cutlery production (41 % of the total gross GHG emissions) and lactic acid and PLA production (40 %) while transportation outside of Europe (both PLA granulates from US and Thailand and PLA cutlery from China and Southeast Asia) accounts in total for 9 %. The main GHG from the cutlery production step (i.e. converting PLA granules into cutlery via injection moulding) comes from electricity production from coal-based power generation in China and Southeast Asia which is used during this process. The impact of electricity from this step accounts for 34 % of total gross GHG emissions (or 84 % of GHG emissions from the cutlery production) while fossil carbon dioxide is the major GHG causing the climate change impact. The lactic acid and PLA production (40 % of the total gross cradle-to-user GHG emissions) consist of several steps which are sugar/starch milling, lactic acid production, lactide production and polymerisation. The GHG emissions from these steps (87 % of GHG emissions are fossil carbon dioxide) are mainly related to the production and use of chemicals (mainly

[62] Note that this temporally 'stored' biogenic carbon will return to the atmosphere in the EoL waste management phase if the product is fully oxidised. If the product is partially oxidised (e.g. into CH_4 instead of CO_2, or becomes soil carbon instead of being oxidised), the carbon balance needs to be re-calculated. This step is performed in the EOL sections (2.2 and 3.2) of this case study report.

lime and sulfuric acid) and to the production and use of electricity and heat. Within the lactic acid and PLA production step, the lactic acid production has the highest contribution to the climate change impact (24 % of the total gross cradle-to-user GHG emissions), followed by lactide production (10 % of the total gross cradle-to-user GHG emissions). Starch/sugar milling and polymerisation are relatively less carbon intensive processes (6 % combined of the total gross cradle-to-user GHG emissions).

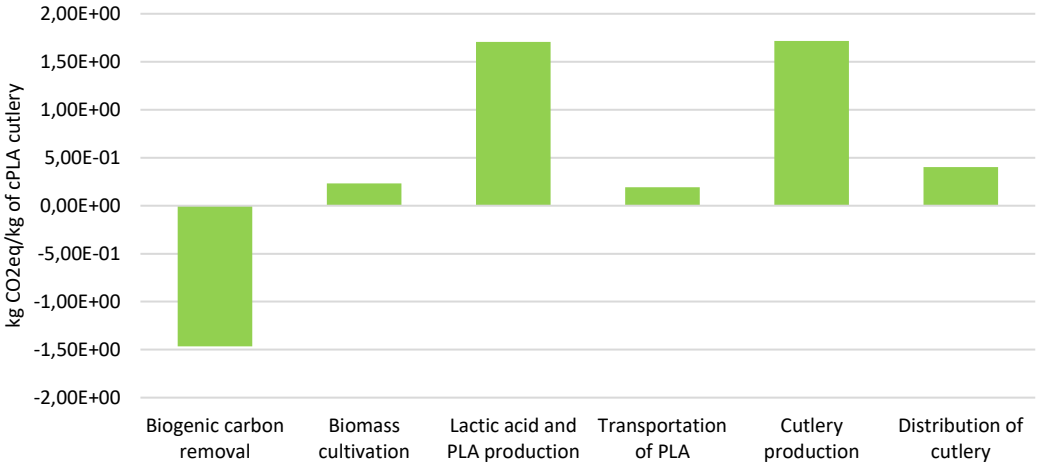


Figure 118. Breakdown of the cradle-to-user climate change results for the baseline (1 kg of cutlery made from PLA, weighted average of two PLA sources, see text, excluding DLUC and ILUC effects).

- **Abiotic depletion (fossil fuels).** Abiotic depletion (fossil fuels) is mainly caused by lactic acid and PLA production in the US and Thailand (43 %) and cutlery production in Asia (38 %) due to consumption of electricity and heat generated by using fossil fuels (coal and natural gas), but also due to the consumption of crude oil for the production of organic solvents. The depletion of natural gas, coal and crude oil due to the lactic acid and PLA production is 53 %, 26 % and 15 %, respectively. The depletion of coal, natural gas and crude oil due to the cutlery production is 44 %, 37 % and 18 %, respectively.

- **Particulate matter.** In the particulate matter impact category, the major contributors to total impact are the following life-cycle steps: cutlery production (54 %), lactic acid and PLA production (22 %), and distribution of cutlery (13 %). The particulate matter impact from cutlery production is allocated to electricity consumption from Chinese and Southeast Asian energy infrastructure. The electricity is primary obtained by coal burning which is the main source (68 % of the cutlery production or 37 % of the total cradle-to-user impact (68 %*54 %=37 %)) of released particulate matter (<2.5 μm) and sulphur dioxide with a contribution of 30 % out of 54 % (or 16 % of the total cradle-to-user impact (30 %*54 %=16 %)) contributed from the cutlery production. The particulate matter from lactic acid and PLA life-cycle step is released as particulates (<2.5 μm) (70 %) and sulphur dioxide emissions (26 %). The particulate matter impact of the distribution of cutlery is caused by SO_2 (47 %) and particulate matter (<2.5 μm) (45 %). More than 87 % of SO_2 emissions occur during transoceanic shipping. The emissions of particulate matter (<2.5 μm) are divided between ship, truck and van transport (see sections 2.1.1.5 and 2.1.2.5) approximately by shares of 40 %, 20 % and 40 %, respectively. The overall particulate matter impact of transportation to Europe (both PLA granulates from US and Thailand and PLA cutlery from China and Southeast Asia) accounts in total for 16 %.

- **Acidification.** The life-cycle stages with highest contribution are: cutlery production (33 %), lactic acid and PLA production (24 %), cutlery distribution (21 %) and PLA transportation (13 %). Emissions from transportation of both PLA granulates and cPLA cutlery represents together 33 %. The two key air emissions from cutlery production are SO_2 (79 % of the impact of cutlery production) and NO_x (20 %). These emissions occur mostly during the coal burning for electricity production. Almost the entire impact of lactic acid and PLA production is due to the fermentation process while starch/sugar milling and lactide production have minor impacts. Sulphur dioxide (78 % of the impact of lactic acid and PLA production) and nitrogen oxides (about 10 % of the impact of lactic acid and PLA production) emitted from the production of the chemicals (sulphuric acid) are the main contributor of the acidification impact from the fermentation process. The transportation of PLA granulates from PLA producers to cutlery producers (i.e. from the US to Southeast Asia and from Thailand to China) has a significant impact in this category due to combustion of fuels in vehicles in order to overcome long distances between locations of the production steps in the life-cycle. The impact is caused by SO_2 (approx. 60 % of the impact of transporation of PLA granulates) and NO_X (approx. 40 % of the impact of transporation PLA granulates) mostly due to the transoceanic shipping. The overall acidification impact of transportation from abroad to Europe (both PLA granulates from US and Thailand and PLA cutlery from China and Southeast Asia) accounts in total for 29 %.

- **Photochemical ozone formation.** The following life-cycle stages contributes to this impact category the most: distribution of cutlery (25 %), cutlery production (24 %), lactide acid and PLA production (18 %) and biomass production (17 %). Most of the impact from cutlery distribution is caused by NO_x emissions (86 %) occurring during transoceanic shipping (67 %) and van transport (22 %). Production and use of electricity in the the cutlery production step is the main contributor causing almost 24 % of the environmental impact in this category. The impact is caused by the realease of NO_x and SO_2 emissions during power production with contributions of 69 % and 13 %, respectively. The photochemical ozone formation related to the lactic acid and PLA production (18 %) is mainly caused by emissions of nitrogen oxides (59 %), NMVOC (29 %) and sulphur dioxide (9 %). The majority of NO_x emissions (84 %) from biomass production, which cause the impact in the photochemical ozone formation category, are from energy obtained by combusting diesel in agricultural machinery during maize cultivation. The overall photochemical ozone formation impact of transportation from abroad to Europe (both PLA granulates from US and Thailand and PLA cutlery from China and Southeast Asia) accounts in total for 32 %.

- **Freshwater eutrophication.** Freshwater eutrophication is mainly caused by the lactic acid and PLA production (46 %), the cutlery production (33 %) and the biomass cultivation (14 %). Freshwater eutrophication related to the lactic acid and PLA production is mainly caused by emissions of phosphate (60 %) and phosphorus (20 %) in water. Environmental burdens from cutlery production in this category are 99 % caused by phosphate emissions in water due to the electricity production in this life-cycle stage. Phosphate emissions in water are also the main burden from the biomass cultivation (95 %) – 88 % is allocated to the use of fertilisers and pesticides during the maize cultivation. The overall freshwater eutrophication impact of transportation from abroad to Europe (both PLA granulates from US and Thailand and PLA cutlery from China and Southeast Asia) accounts in total for 4 %.

A detailed interpretation for the human toxicity impact categories is giver here:

- **Human toxicity (non-cancer and cancer effects).** More than 90 % of the impact in both human toxicity categories occurrs during lactic acid and PLA production. A negative impact during the biomass production is observed in the figure. This is almost

entirely caused by the zinc absorbed from the soil by sugarcane. Heavy metals are often added to synthetic fertilisers to facilitate the absorption of the nutrients for crops. They are also often found in animal manure (especially zinc).[63] The sugarcan cultivation in Thailand uses both synthetic fertilisers and animal manure (Durlinger et al., 2017). These metals are either leached into the soil or absorbed by the biomass. When there is more zinc absorbed by the biomass than is leached, a negative emission appears in the LCI model (Durlinger et al., 2017). However, the zinc absorbed by the biomass will eventually end up in the environment if the biomass is burned for energy (e.g. bagasse), returned to the soil (residue distillage applying to the soil) or removed from the production lines and ending up in waste (water) treatments. These zinc emissions could potentially explain the high contribution of zinc during the lactic acid and PLA production. [64] Non-cancer effects are mainly caused by emissions of zinc to soil (75 %) and mercury to air (7 %). Cancer effects occur due to the emissions of chromium to water (50 %), air (20 %) and soil (20 %).

3.1.2 Cradle-to-user environmental impacts of the petrochemical reference system (PS)

The overall results of cradle-to-user environmental impacts of the petrochemical reference system (fossil fuel-based cutlery) are shown in Table 142. Based on Chapter 6 *Ranges for environmental impacts from production of fossil-based plastics*, 9 out of 17 impact categories were found suitable for petrochemical PS interpretation – namely, climate change, particulate matter, photochemical ozone formation, terrestrial eutrophication, freshwater eutrophication, marine eutrophication, water use, abiotic depletion (fossil fuels) and NREU. Further details can be found in Table 34 where the environmental impact of 1 kg of PS and the proposed ranges of variation within each impact category are shown (see Approach and methodology section 6.2.2, Table 34). The main reasons for the exclusion of most of the assessed impact categories of PlasticsEurope's eco-profile for PS are due to large differences identified among the impact assessment results provided by the main LCA databases, non-compliance with several ILCD requirements and a lack of transparency in the allocation approach. The breakdown of the life-cycle stages of the reference system is presented in Figure 119.

Table 142. Cradle-to-user life cycle impact assessment results for the reference (cutlery made from PS, excluding DLUC and ILUC effects).

Impact category	Unit	Total impact per 1 FU of PS cutlery (cradle-to-user)	Total impact per 1 kg of PS cutlery (cradle-to-user)
Climate change	kg CO_2 eq.	4.50E+01	4.46E+00
Particulate matter	kg PM2.5 eq	2.29E-02	2.27E-03
Photochemical ozone formation	kg NMVOC eq	1.42E-01	1.40E-02
Terrestrial eutrophication	molc N eq	4.75E-01	4.71E-02
Freshwater eutrophication	kg P eq	4.24E-04	4.20E-05

[63] The amount of zinc and copper are found very high in pig manure because they are often added in the feed for animal health reasons (Durlinger et al., 2017).

[64] Due to lack of transparency of the industrial data, the exact zinc balance through the entire life cycle cannot be fully reconstructed.

Marine eutrophication	kg N eq	4.33E-02	4.29E-03
Water use	m^3	1.02E+01	1.01E+00
Abiotic depletion (fossil fuels)	MJ	1.02E+03	1.01E+02
NREU	MJ	1.04E+03	1.03E+02

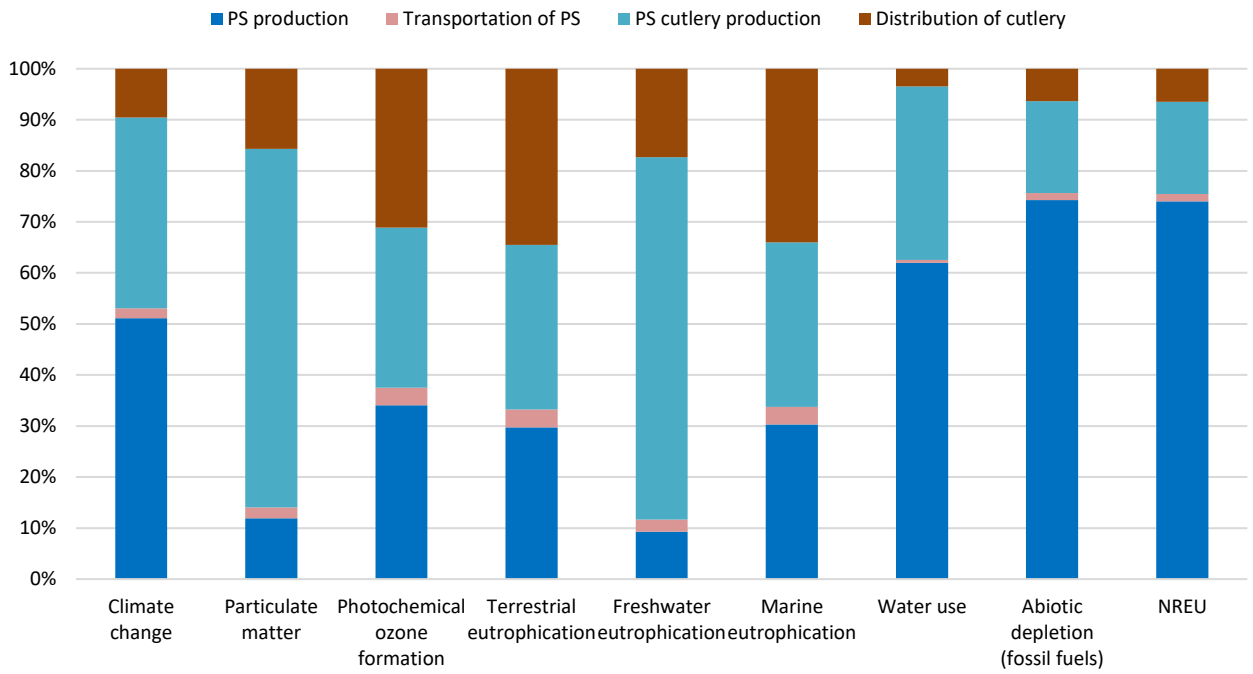


Figure 119. Breakdown of the cradle-to-user results for the reference (cutlery made from PS).

Based on the results from cradle-to-user impact assessment for the reference, the following considerations can be highlighted for the impact of each life-cycle step:

- Polystyrene production is a major contributor to the impact categories of climate change (51 %), water use (62 %), abiotic depletion (fossil fuels) (74 %) and non-renewable energy use (74 %). These impacts are caused mainly by the use of water, natural gas and crude oil during the PS production.

- Transportation of PS granulates has a low impact across all impact categories (less than 10 %).

- The cutlery production life-cycle stage has significant impact in the categories of freshwater eutrophication (71 %) and particulate matter (70 %). The majority of impact comes as the result of electricity consumption from coal-dependant energy infrastructure in China. Cutlery production has a higher impact in the remaining categories too, ranging from 18 % to 42 %.

- Distribution of PS cutlery has a non-negligible contribution to three categories: namely, terrestrial eutrophication (35 %), marine eutrophication (34 %) and photochemical ozone depletion (31 %). In the remaining categories, the impact is lower than 20 %. The impact coming from this life-cycle step is due to the releasing of exhaust gases by

vehicles during transport over long distances between the production and the use stage.

3.2 Results of the end of life of single use cutlery

The results for the EoL impact assessment modelled in EASETECH are presented in this paragraph. Firstly, the mass and energy flows of all product systems in all technologies are presented, secondly the LCIA results for each product system, thirdly a comparison between product systems, and lastly the EoL mix. In Annex 4 further results are presented, the contribution analysis is in Figure 261 ff. and weighted results are in Table 237 ff.

3.2.1 Mass and energy flows

The mass and energy flows, as outputs from EASTECH, are presented in Table 143 and Table 144. The results, correspond to the circular footprint formula in PEFCR guidance draft version 6.3 (European Commission, 2017), and in addition the paragraph looks at the carbon flows. The flows correspond to those observed in the flow charts for each technology in the approach and methodology chapter.

For industrial composting of the bio-based cPLA cutlery and food leftovers, 14 % of the mass goes to the compost, whereas 1 % ends up as rejects for incineration. The compost recovers 31 % of the elements which substitute fertilisers (N, P and K). When the cPLA and the food waste is anaerobically digested 4 % becomes compost and more than 200 % of the input mass is digestate (due to the addition of water in the process). This recovers 51 % of N, P and K for use as fertiliser. Rejects consist of 30 % of the plastics and 5 % of the food waste.

Regarding the reference petrochemical PS, 25 % of the collected PS for recycling is recycled and 20 % is substituted (calculated based on sorting and technology efficiency as well as market response, considering the decrease in quality of the plastic - see the methodology for further explanation). Including the food waste that enters the recycling waste stream with the cutlery, only 4 % of the total mass is recycled and 3 % is substituted. The food waste that is rejected from the recycling, is distributed between incineration (53 %) and landfilling (47 %).

Landfilling the cPLA cutlery means that 77 % of the mass is stored in the ground while 82 % is stored for the PS cutlery. The cPLA cutlery will degrade in the landfill together with the food waste but the PS cutlery will not.

The energy utilisation for the EoL-incineration of cPLA cutlery including food waste is 25 % and 26 % for the PS cutlery. 10 % of the energy is gained when landfilling the bio-based cLPA cutlery and 6 % for the petrochemical PS cutlery. In the biological technologies for the cPLA cutlery, 33 % of the energy is recovered in the AD through biogas utilisation, and additionally 4 % from incineration of the rejects. The energy recovery from incineration of rejects from the composting facility is 3 %.

Looking at the carbon flow, 99.9 % becomes air emissions in the incineration plant for both product systems. In the landfill 34 % of the biogenic carbon is stored for the cLPA and 21 % of the biogenic carbon from the food waste following the PS cutlery. In the industrial composting of cPLA, 73 % of biogenic carbon ends up in the compost and in the AD 50 % of the biogenic carbon is found in the compost and digestate, which is spread in fields.

Table 143. Material and energy flow for EoL of 1 kg cPLA mix with 100 g contamination and 5.23 of food waste.

Bio-based cPLA mix including food waste								
Technology	**Box**	**Process**	**Mass (kg)**	**Bio carbon (kg)**	**Fossil carbon (kg)**	**Gas (m^3 CH4)**	**Energy substitution**	**Fertiliser substit**

							(MJ)	ution (kg NPK)
Material	I	Material input	6.33E+00	1.15E+00	8.67E-03		4.59E+01	8.02E-02
Incineration w/wo energy recovery *	E, H	Energy production	6.33E+00	1.15E+00	8.67E-03		1.13E+01	
	R1	Fly Ash	2.49E-02	0.00E+00	0.00E+00			
	R2	Bottom ash	7.23E-02	7.90E-04	8.50E-06			
	D	Direct emissions		1.15E+00	8.67E-03			
Landfill	L	Leachate	9.80E+00	0.00E+00	0.00E+00			
	G, E	Landfill gas		7.55E-01		8.04E-01	4.47E+00	
	CS	Storage in landfill	4.90E+00	3.95E-01	8.64E-03			
Compost	O, S	Compost	8.53E-01	8.36E-01	8.24E-03			2.46E-02
	R, E, H	Rejects	5.39E-02	2.45E-01	1.29E-04		1.40E+00	
Anaerobic digestion	B, E1, H1, F	Biogas		3.58E-01		4.47E-01	1.51E+01	
	C, S	Compost	2.59E-01	6.77E-02	1.63E-03			6.25E-03
	D, S	Digestate	1.36E+01	5.11E-01	6.56E-03			3.45E-02
	R, E2, H2	Rejects	5.92E-01	1.60E-01	4.78E-03		1.76E+00	

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

Table 144. Material and energy flow for EoL of 1 kg PS with 100 g contamination and 6.14 kg of food waste.

Petrochemical PS including food waste							
Technology	**Box**	**Process**	**Mass kg**	**Bio carbon kg**	**Fossil carbon kg**	**Gas m³ CH4**	**Energy substitution MJ**
Material	I	Material input	7.24E+00	8.68E-01	2.04E+00		7.39E+01
Recycling	O	Recycled plastic	2.75E-01	0.00E+00	2.21E-01		
	S	Substituted plastic	2.23E-01	0.00E+00	1.79E-01		
	R, E, H	Rejects	6.97E+00	8.68E-01	1.81E+00		1.09E+01
Incineration w/wo energy recovery *	E, H	Energy production	7.24E+00	8.68E-01	2.04E+00		1.95E+01
	R1	Fly Ash	5.78E-02	0.00E+00	0.00E+00		

	R2	Bottom ash	1.67E-01	8.68E-04	1.75E-03		
	D	Direct emissions		8.67E-01	2.03E+00		
Landfill	L	Leachate	1.19E+01	0.00E+00	0.00E+00		
	G, E	Landfill gas		6.85E-01		7.53E-01	4.26E+00
	CS	Storage in landfill	5.95E+00	1.83E-01	8.95E-01		

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied with 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

3.2.2 Baseline: Bio-Based cPLA

The impact assessment results are presented per the EoL-reference flow, i.e. 1 kg plastic, 100 g of contamination and 5.2 kg food leftovers. For the cPLA cutlery the results are presented in Table 145 for each of the EoL technologies (assuming 100 % disposal via that technology). For the bio-based cPLA cutlery no clear conclusion can be made for ranking the EoL options. Thus, normalizing and weighing the characterised results to rank the EoL options were performed on the cPLA cutlery including food leftovers. The weighted results show, both with and without toxicity, that the industrial composting technology is best, secondly anaerobic digestion, thirdly incineration and lastly landfilling.

Further details on the individual EoL options for the bio-based cPLA cutlery are presented below, looking into specific impact categories and contribution of impacts from different processes and elemental emissions.

Table 145. Total results of treating 1 kg bio-based cPLA incl. food waste (incl. 100 g contamination and 5.2 kg food leftovers) for all impact categories. Highest impact within each category is red and lowest impact green.

Impact category (Unit)	Bio-based cPLA incl. food waste			
	Incineration	Landfilling	Industrial composting	Anaerobic digestion
Climate change - fossil (kg CO_2 eq)	-2.34E-01	2.37E-02	5.53E-01	-2.31E-01
Climate change - biogenic (kg CO_2 eq)	4.12E+00	1.34E+01	-3.41E-01	1.47E+00
Climate change - total (kg CO_2 eq)	3.88E+00	1.34E+01	2.11E-01	1.24E+00
Ozone depletion (kg CFC-11 eq)	-5.02E-08	6.67E-06	3.91E-09	-3.14E-08
Human toxicity, non-cancer effects (CTUh)	6.96E-08	6.29E-10	1.71E-06	1.18E-06
Human toxicity, cancer effects (CTUh)	1.33E-08	-1.21E-09	-1.11E-07	-8.09E-08
Particulate matter (kg PM2.5 eq)	3.03E-05	-9.92E-05	7.16E-05	1.21E-04
Ionizing radiation HH (kBq U235 eq)	8.08E-02	-1.30E-02	3.06E-02	3.92E-02
Photochemical ozone formation (kg NMVOC eq)	6.40E-03	4.08E-03	1.13E-03	2.40E-03
Acidification (molc H+ eq)	7,56E-03	-9.07E-05	4.76E-03	1.35E-02
Terrestrial eutrophication (molc N eq)	3.31E-02	4.41E-04	1.94E-02	6.06E-02
Freshwater eutrophication (kg P eq)	3.15E-03	-1.27E-05	1.23E-04	9.29E-04
Marine eutrophication (kg N eq)	2.64E-03	2.42E-04	3.79E-03	1.03E-02
Freshwater ecotoxicity (CTUe)	1.00E+00	-1.76E-03	9.88E-02	2.02E-01
Land use (kg C deficit)	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use (m^3)	-1.32E-02	-2.15E-01	4.66E-01	2.27E-01
Abiotic depletion (kg Sb eq)	-3.33E-07	-1.99E-07	-6.42E-07	-2.07E-06
Abiotic depletion (fossil fuels) (MJ)	-6.46E+00	-1.28E+00	2.01E+00	-5.25E-01
NREU (MJ)	-8.26E+00	-7.24E-01	6.29E-01	-1.74E+00

The significance of the colour scale is the same as for all the other product systems (Table 145 and Table 146). This means that you can compare the colour horizontally across all tables, but only comparing between the same impact category.

3.2.2.1 Intended EoL technology: Industrial composting of cPLA cutlery

The food leftovers will be composted at an industrial composting facility together with the cPLA cutlery. Industrial composting is a preferred treatment for recycling organic material. Comparing industrial composting with other EoL technologies, it has the lowest impact in climate change, human toxicity (cancer effects) and photochemical ozone formation. This EoL technology performs worst in the impact categories NREU and abiotic depletion (fossil fuels), as there is no energy recovery of the waste, only consumption. The largest savings come from

the avoidance of production of fertilisers. For human toxicity (cancer effects) the largest savings are from avoidance of introduction of heavy metals from conventional fertiliser, especially chromium from potassium fertiliser.

The impacts from industrial composting of the cPLA are primarily from treating the additional food leftovers, and not the cPLA. From the cPLA there are both savings and emissions, depending on the impact category, from the heat substitution, which is from incineration of the rejects. The emissions from use on land (i.e. direct emissions of GHG when the compost is spread on the field - primarily CO_2, but also methane and other substances like N_2O) is balanced out to a great extent by a saving in fertiliser substitution. Emissions from the composting facility also contribute with direct emissions in most impact categories.

3.2.2.2 Incineration of cPLA cutlery

The incineration of cPLA cutlery with food leftovers performs best in three impact categories (ozone depletion, abiotic depletion (fossil fuels) and NREU, compared with the other EoL options. The good performance in the energy categories is from the substitution of heat and electricity, contributing 56 % and 54 % of the absolute impact. The largest contributor to impacts from incineration of cPLA cutlery is the direct emissions from the incineration, especially in climate change impacts from biogenic carbon, photochemical ozone formation, acidification, terrestrial and marine eutrophication. The largest savings in the impacts categories of water use, human toxicity, particulate matter and abiotic depletion come from the electricity substitution. From heat substitution the contribution analysis shows the largest savings from climate change - fossil, freshwater ecotoxicity, NREU and abiotic depletion (fossil fuels).

3.2.2.3 Landfilling of cPLA cutlery

The LCIA for the cPLA cutlery shows that landfilling is the preferred EoL technology, looking at the overall colour scheme (Table 145). The impact categories, where landfilling perform best are human toxicity (non-cancer effects), particulate matter, ionizing radiation, acidification, the three eutrophication categories, freshwater ecotoxicity and water use. The reason for the good performance is primarily due to savings when substituting electricity (produced from the landfill gas) combined with few environmental impacts from landfill construction and operation, compared to the other EoL technologies. The exception is marine eutrophication, where there are negative impacts from leachate treatment (electricity consumption). The electricity substitution contributes most to the following impact categories; human toxicity, non-cancer effects (85 %), acidification (74 %) and freshwater ecotoxicity (96 %). In abiotic depletion it contributes 52 %, and for NREU 57 %.

Landfilling shows the highest impacts in three impact categories. This is due to a very limited energy recovery when landfilling, whereas the other EoL technologies offer substitution of energy and fertilisers. The landfill gas that is collected and combusted or not collected but oxidised under cover, contributes to the largest impact in the impact categories of total climate change, ozone depletion and photochemical ozone formation. The leachate treatment has the highest contribution in the impact category marine eutrophication from ammonium emissions to water and nitrogen oxides to air. Though the electricity generation from the landfill gas is low compared to the incineration, the substitution of electricity has the largest contribution in 13 out of 18 impact categories.

3.2.2.4 Anaerobic digestion of cPLA cutlery

Anaerobic digestion has the lowest impact compared to the other EoL technologies in the impact category abiotic depletion. Impacts arise from leakage of methane gas and from use of digestate on land. The additional food waste accounts for a large saving in fertiliser and energy substitution. The largest contribution overall to the anaerobic digestion is fertiliser substitution

and direct emissions from use on land, which in many impact categories almost balance each other. Other impacts stems from the incineration of rejects. In ionizing radiation heat substitution especially contributes to the impacts due to carbon-14 emissions to air. The energy substitution contributes to respectively 56 % and 52 % of the absolute impacts of abiotic depletion (fossil fuels) and NREU, primarily from the incineration of rejects of the cPLA cutlery.

3.2.2.5 Weighted end of life results for the baseline, cPLA cutlery

The results of weighting the EoL technologies are shown in Figure 120, and the numerical results are presented in Annex 4. For methodology and choice of normalization and weighting factors, please see the Approach and methodology chapter, section 5.2.3 "Use normalised and weighted results to determine the preference of EoL option" and the values in Table 30. Normalisation factors (NF) and weighting factors used by this study.

The weighted results for the baseline product system show, both with and without toxicity, that the industrial composting technology is best, secondly anaerobic digestion, thirdly incineration and lastly landfilling.

The largest contribution to the weighted results is the global warming potential impact category in incineration and landfilling. For the biological treatments (industrial composting and anaerobic digestion) other impact categories are dominant. For the results with toxicity the abiotic depletion categories contribute most. Without toxicity acidification and marine eutrophication are more dominant, as well as global warming potential. The negative impact from biological recycling is in abiotic depletion (fossil fuels), which is due to savings from heat and electricity substitution.

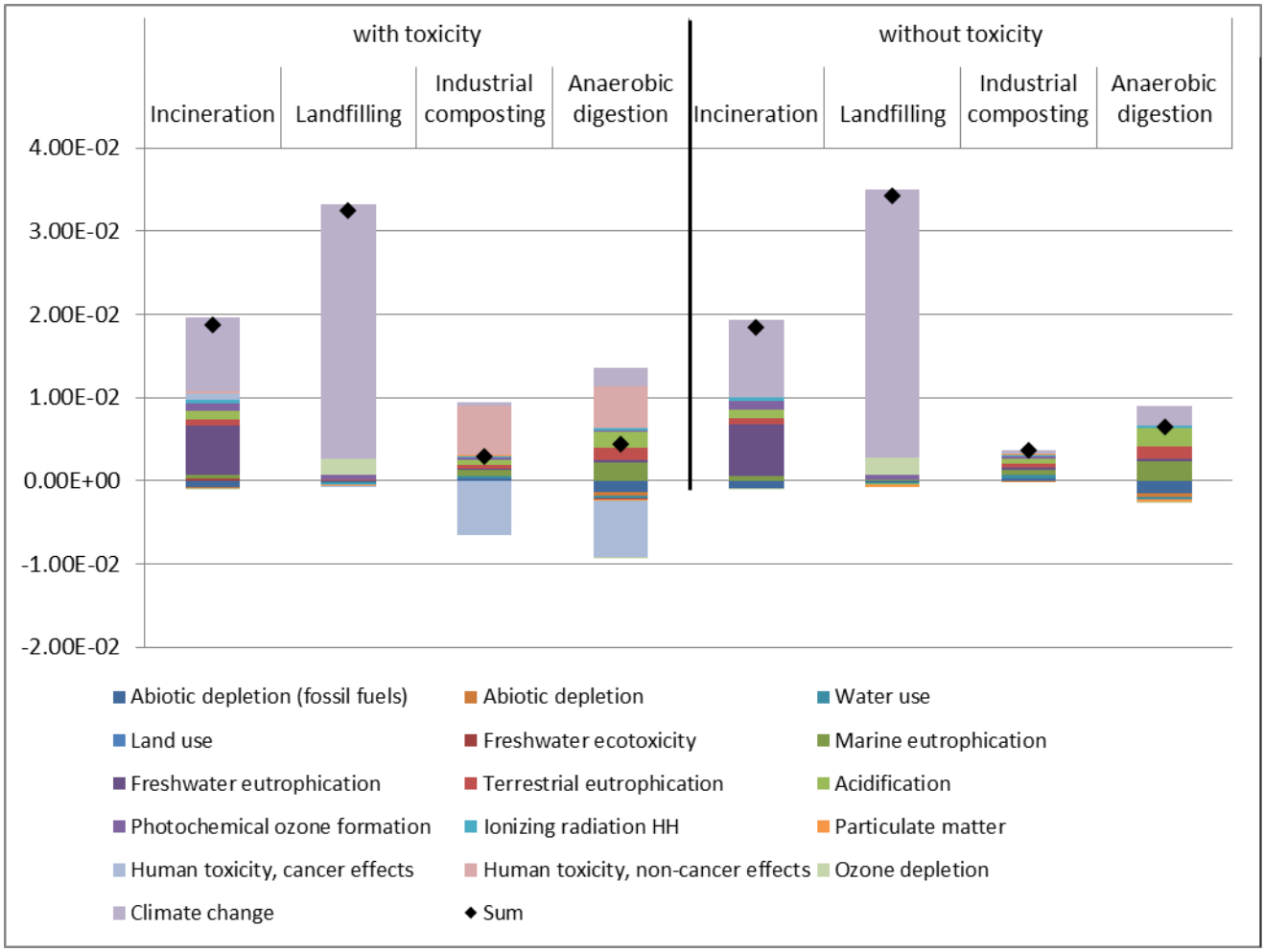


Figure 120. Weighted EoL results for the baseline product system, bio-based PLA cutlery including food leftovers. Numerical values can be observed in Annex 4.

3.2.3 Reference: Petrochemical PS

The results given in Table 146 are for 1 kg plastic and 100 g of contamination and 6.1 kg of food leftovers for the full EoL technologies (100 %). The food leftovers in the recycling scenario are assumed to be incinerated (53 %) and landfilled (47 %).

Looking at the LCIA results for PS cutlery including food leftovers there is no clear ranking for the three EoL technologies.

Further details on the individual EoL options for the PS cutlery are presented below, looking into specific impact categories and the contribution of impacts from different processes and elemental emissions.

Table 146. Total results of treating 1 kg petrochemical PS incl. food waste (incl. 100 g contamination and 6.1 kg food leftovers) for all impact categories. Highest impact within each category is red and lowest impact green.

Impact category (Unit)	Petrochemical PS incl. food waste		
	Recycling	Incineration	Landfilling
Climate change - fossil (kg CO_2 eq)	1.49E+00	2.53E+00	3.65E-02
Climate change - biogenic (kg CO_2 eq)	7.56E+00	3.01E+00	1.27E+01
Climate change - total (kg CO_2 eq)	9.05E+00	5.55E+00	1.28E+01
Ozone depletion (kg CFC-11 eq)	2.82E-06	-9.15E-08	6.09E-06
Human toxicity, non-cancer effects (CTUh)	1.54E-07	1.54E-07	4.81E-09
Human toxicity, cancer effects (CTUh)	1.57E-08	3.14E-08	-1.07E-09
Particulate matter (kg PM2.5 eq)	-3.52E-04	-5.05E-05	-9.13E-05
Ionizing radiation HH (kBq U235 eq)	1.04E-01	1.37E-01	-1.16E-02
Photochemical ozone formation (kg NMVOC eq)	4.34E-03	6.81E-03	3.95E-03
Acidification (molc H^+ eq)	3.28E-03	8.76E-03	-1.89E-05
Terrestrial eutrophication (molc N eq)	1.94E-02	3.66E-02	6.37E-04
Freshwater eutrophication (kg P eq)	2.04E-03	3.77E-03	-1.15E-05
Marine eutrophication (kg N eq)	1.61E-03	2.92E-03	3.00E-04
Freshwater ecotoxicity (CTUe)	3.13E+00	4.83E+00	8.88E-03
Land use (kg C deficit)	0.00E+00	0.00E+00	0,00E+00
Water use (m^3)	-4.80E-01	-2.69E-01	-1.80E-01
Abiotic depletion kg Sb eq	-3.80E-07	-6.16E-07	-1.87E-07
Abiotic depletion (fossil fuels) (MJ)	-2.23E+01	-1.51E+01	-9.65E-01
NREU (MJ)	-2.44E+01	-1.80E+01	-4.74E-01
The significance of the colour scale is the same as for all the other product systems (Table 145). This means that you can compare the colour horizontally across all tables, but only comparing between the same impact category.			

3.2.3.1 Recycling of PS cutlery

Recycling of the PS cutlery performs best in four impact categories (particulate matter, acidification, water use, abiotic depletion (fossil fuels) and NREU), compared to the other EoL

options with food leftovers. The food leftovers end up as rejects within the recycling facility and are distributed between incineration (53 %) and landfilling (47 %). The energy recovery from incineration/landfilling of the rejects and the food leftovers, as well as the substitution of PS, has a substantial contribution to the savings in most impact categories. In ionizing radation the heat substitution contributes to impacts. For marine eutrophocation a large share of the impacts come from leachate treatment due to the landfilling of the food leftovers and from direct emissions from the incineration plant.

3.2.3.2 Incineration of PS cutlery

Incineration has the highest impact in 9 out of 16 impact categories, including the ecotoxicity and eutrophication categories. Incineration of PS cutlery has the lowest impact in three categories: climate change, ozone depletion and abiotic depletion from substitution of heat and electricity.

The largest contribution from incineration of PS and the food leftovers stems from electricity and heat substitution, as well as direct emissions from the incineration plant. For freshwater eutrophication and human toxicity (cancer effects) a large share of the impacts come from bottom ash treatment and utilization. Fly ash treatment and utilization primarily contribute to human toxicity (non-cancer effects) and freshwater ecotoxicity.

3.2.3.3 Landfilling of PS cutlery

The LCIA results show that landfilling has the lowest impact in 9 out of 16 impact categories. The landfill performs best in human toxicity (both types), ionizing radiation, freshwater ecotoxicity and the eutrophication categories compared with the other EoL technologies. The landfill performs worst in NREU, abiotic depletion, and water use due to low energy substitution compared with the other EoL tehcnologies. The contribution stems primarily from electricity substitution from the recovered landfill gas. For the impact categories climate change, ozone depletion and photochemical ozone formation, a large share of the impacts stem from un-collected landfill gas. Transportation contributes to a large share of human toxicity (non-cancer effects), which is due to emissions of zinc to air and soil.

3.2.4 Details on climate change impacts

This section looks at both product systems of the cutlery case with food leftover, and all EoL technologies. For illustration the climate change impact is presented in Figure 121, with the contribution between processes in each of the technologies. The treatment of food leftovers is presented in a separate combined process (i.e. it includes the whole EoL phase for the food leftover treatment). Note that this section still compares per EoL-reference flow, i.e. 1 kg plastic, 100 g contamination and food leftover. Looking at the functional unit, we find that the PLA weighs 35 % more than the petrochemical PS cutlery.

Looking solely at the climate change impact, the industrial composting of PLA performs best (0.2 kg CO_2eq. / 1 kg PLA). Second best is anaerobic digestion of PLA (1.2 kg CO_2 eq. / kg PLA). These two EoL technologies are preferred, also when looking at the results per functional unit. Landfilling of PLA is clearly the worst performing pathway, looking at climate change, with an emission of 13.4 kg of CO_2eq. per kg PLA. This is again true when looking at the results per functional unit.

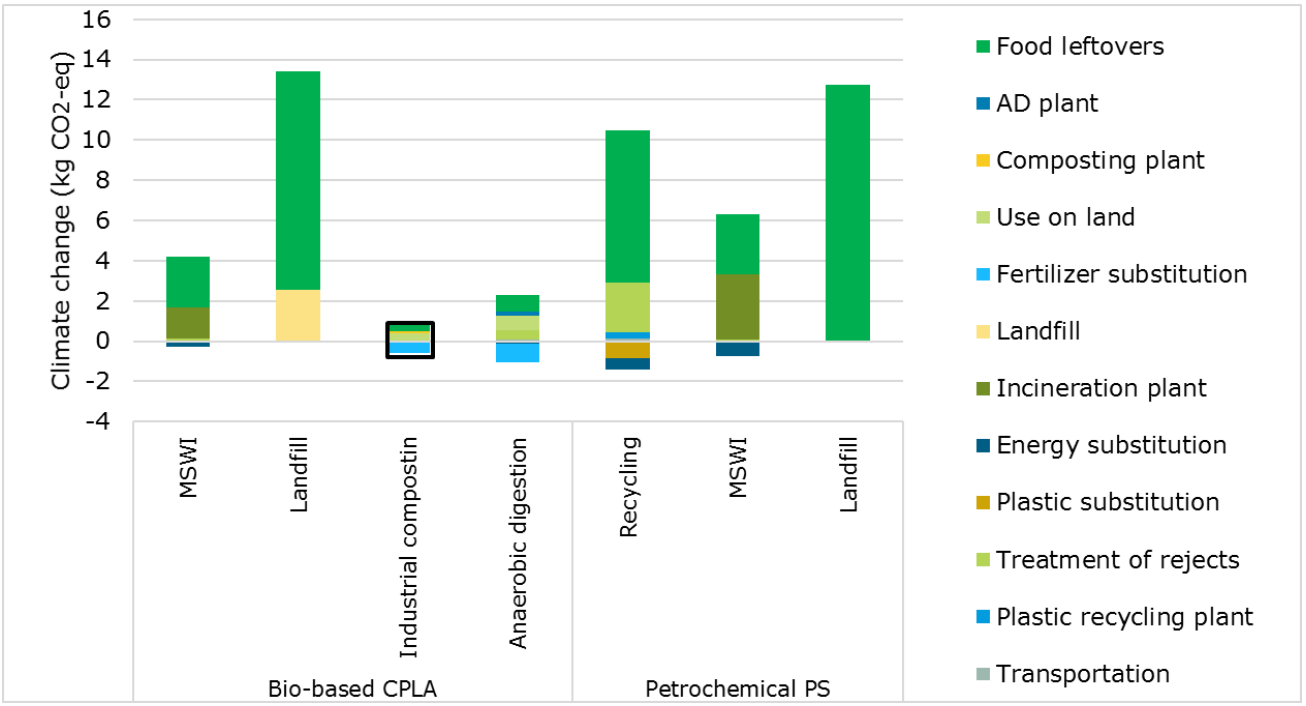


Figure 121. Climate change for treating 1 kg single-use cutlery (incl. 100 g contamination and 5.2-6.1 kg food leftovers) in different EoL technologies. The intended EoL option is marked with a black box. See Annex 4 for details of the content of the grouped processes. The results without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

3.2.5 Comparing all EoL technologies in all the product systems with weighted results

The weighted results with toxicity for both product systems and all EoL technologies are presented in Figure 122. The weighted reults show that comparing all EoL technologies for both product systems, industrial composting of cPLA is the preferred scenario. Anaerobic digestion of cPLA is slightly higher than the industrial composting. Looking into the petrochemical PS cutlery there is a minimal difference between the normalised values. The incineration performs slightly better than recycling of PS including food leftovers. If one looks at incineration solely with energy recovery (i.e. excluding plants without energy recovery), this technology would likely perform better than the other EoL technologies. The recycling rate for PS is relatively low, and most of the PS is sorted out and incinerated together with the food leftovers.

Climate change and photochemical ozone formation are dominant in the weighted results of incineration of the cPLA cutlery, which is due to direct emisisons from incinerating (mainly CO_2 and nitrogen oxides respectively). Climate change impacts arise from landfilling the cPLA due to landfill gas directly emitted or flared. For industrial composting and anaerobic digestion, the emissions in human toxicity (non-cancer effects) and savings in human toxicity (cancer effects) stem from respectively use on land and fertiliser substitution.

For the petrochemical PS cutlery, the recycling contributes mainly to climate change and abiotic depletion, due respectively to incineration and landfilling of the food leftovers and plastic substitution. Incineration of PS cutlery contribute to climate change and freshwater eutrophication due respectively to direct emissions of the incineration plant and treatment of fly and bottom ash. Landfilling PS cutlery and the food leftovers contribute mainly to climate

change, where the impacts originate from landfill gas being directly emitted and flared (the gas comes from degradation of the food leftovers).

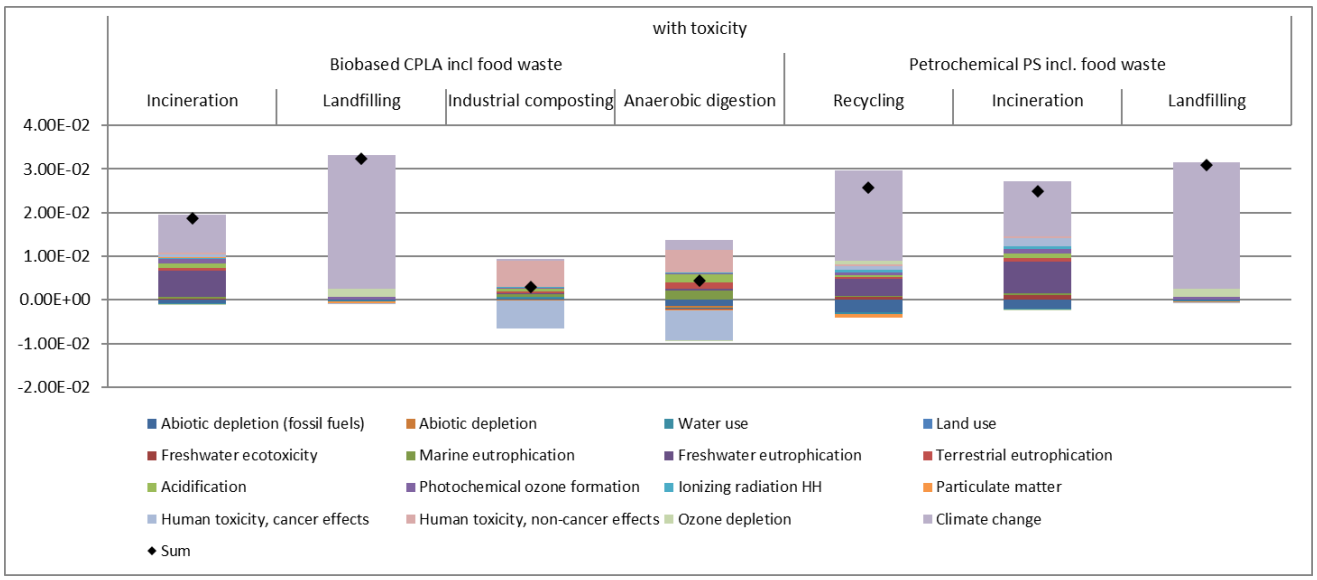


Figure 122. Weighted results with toxicity for the single-use cutlery including food leftovers for the individual EoL options per functional unit. Numerical values are found in Annex 4.

3.2.6 End of life mix results

The LCIA for all product systems for the estimated EoL per functional unit (i.e. different from the above results, which are for the EoL-reference flow of 1 kg plastic and 100 g contamination and 5.2-6.1 kg food leftovers) are presented in Table 147. The food leftovers rejected from recycling will be distributed with 53 % incinerated and 47 % landfilled. Looking solely at climate change, the bio-based cPLA product system is preferable, as the bio-based cPLA cutlery has much lower climate change impacts from industrial composting and incineration.

The EoL mix results per functional unit shows that, for the bio-based cPLA, there are seven impact categories where cPLA performs better and 11 impact categories were the petrochemical PS cutlery performs better. It is dependent on the impact category which product is more favourable.

Table 147. LCIA for EoL mix of the cutlery for all product systems (1 kg plastic + 100 g contamination + 5.2-6.1 kg food leftovers) per functional unit (1,000 pairs of cutlery).

Impact category	Unit	Bio-based cPLA	Petrochemical PS
Industrial composting	%	30	
Recycling	%		30
Incineration	%	39	39
Landfill	%	31	31
Climate change - fossil	kg CO_2 eq / FU	1.11E+00	1.46E+01
Climate change - biogenic	kg CO_2 eq / FU	7.68E+01	7.46E+01
Climate change - total	kg CO_2 eq / FU	7.79E+01	8.92E+01
Ozone depletion	kg CFC-11 eq / FU	2.79E-05	2.73E-05
Human toxicity, non-cancer effects	CTUh / FU	7.34E-06	1.09E-06
Human toxicity, cancer effects	CTUh / FU	-3.87E-07	1.68E-07
Particulate matter	kg PM2.5 eq / FU	3.48E-05	-1.55E-03
Ionizing radiation HH	kBq U235 eq / FU	4.99E-01	8.21E-01
Photochemical ozone formation	kg NMVOC eq / FU	5.57E-02	5.23E-02
Acidification	molc H+ eq / FU	5.91E-02	4.44E-02
Terrestrial eutrophication	molc N eq / FU	2.57E-01	2.05E-01
Freshwater eutrophication	kg P eq / FU	1.72E-02	2.10E-02
Marine eutrophication	kg N eq / FU	3.05E-02	1.73E-02
Freshwater ecotoxicity	CTUe / FU	5.72E+00	2.85E+01
Land use	kg C deficit / FU	0.00E+00	0.00E+00
Water use	m^3 / FU	9.25E-01	-3.08E+00
Abiotic depletion	kg Sb eq / FU	-5.23E-06	-4,17E-06
Abiotic depletion (fossil fuels)	MJ / FU	-3.15E+01	-1.30E+02
NREU	MJ / FU	-4.43E+01	-1.46E+02

3.3 Cradle-to-grave environmental impacts and interpretation

3.3.1 Bio-Based baseline system

Overall results for the cradle-to-grave environmental impacts of the bio-based baseline system (cPLA cutlery) are shown in Table 148. The table presents results for all 16 PEFCR impact categories and the category for non-renewable energy use (NREU). The breakdown of the baseline, shown in Figure 123, is divided into aggregated life-cycle stages – biomass production (incl. biogenic carbon removal), manufacturing of the cPLA cutlery (including PLA production, transportation of PLA granulates and distribution of cutlery) and the EoL mix. The results are presented without taking into account effects on ILUC or DLUC.

Table 148. Cradle-to-grave life cycle impact assessment results for the baseline (cutlery made from PLA, weighted average of two PLA sources, see text, EoL mix, excluding DLUC and ILUC effects) per 1 functional unit.

Impact category	Unit	Biomass production	Manufacturing	EoL mix	Total impact per 1 FU of cPLA cutlery (cradle-to-grave, EoL mix)
Climate change	kg CO_2 eq.	-1.67E+01	5.47E+01	7.79E+01	1.16E+02
Ozone depletion	kg CFC-11 eq.	3.05E-08	3.16E-06	2.79E-05	3.10E-05
Human toxicity, non-cancer effects	CTUh	-2.87E-06	1.09E-05	7.34E-06	1.54E-05
Human toxicity, cancer effects	CTUh	3.15E-08	2.01E-06	-3.87E-07	1.65E-06
Particulate matter	kg PM2.5 eq	9.54E-04	3.34E-02	3.48E-05	3.44E-02
Ionizing radiation HH	kBq U235 eq	2.27E-01	2.04E+00	4.99E-01	2.77E+00
Photochemical ozone formation	kg NMVOC eq	3.84E-02	1.84E-01	5.57E-02	2.79E-01
Acidification	molc H+ eq	3.67E-02	3.53E-01	5.91E-02	4.49E-01
Terrestrial eutrophication	molc N eq	1.86E-01	6.70E-01	2.57E-01	1.11E+00
Freshwater eutrophication	kg P eq	3.37E-04	2.01E-03	1.72E-02	1.95E-02
Marine eutrophication	kg N eq	6.11E-02	5.63E-02	3.05E-02	1.48E-01
Freshwater ecotoxicity	CTUe	1.07E+02	5.80E+01	5.72E+00	1.71E+02
Land use	kg C deficit	1.56E+02	7.48E+01	0.00E+00	2.31E+02
Water use	m3	6.50E+00	9.47E+00	9.25E-01	1.69E+01
Abiotic depletion	kg Sb eq	3.35E-06	3.98E-05	-5.23E-06	379E-05
Abiotic depletion (fossil fuels)	MJ	2.71E+01	7.21E+02	-3.15E+01	7.17E+02
NREU	MJ	2.92E+01	7.72E+02	-4.43E+01	7.57E+02

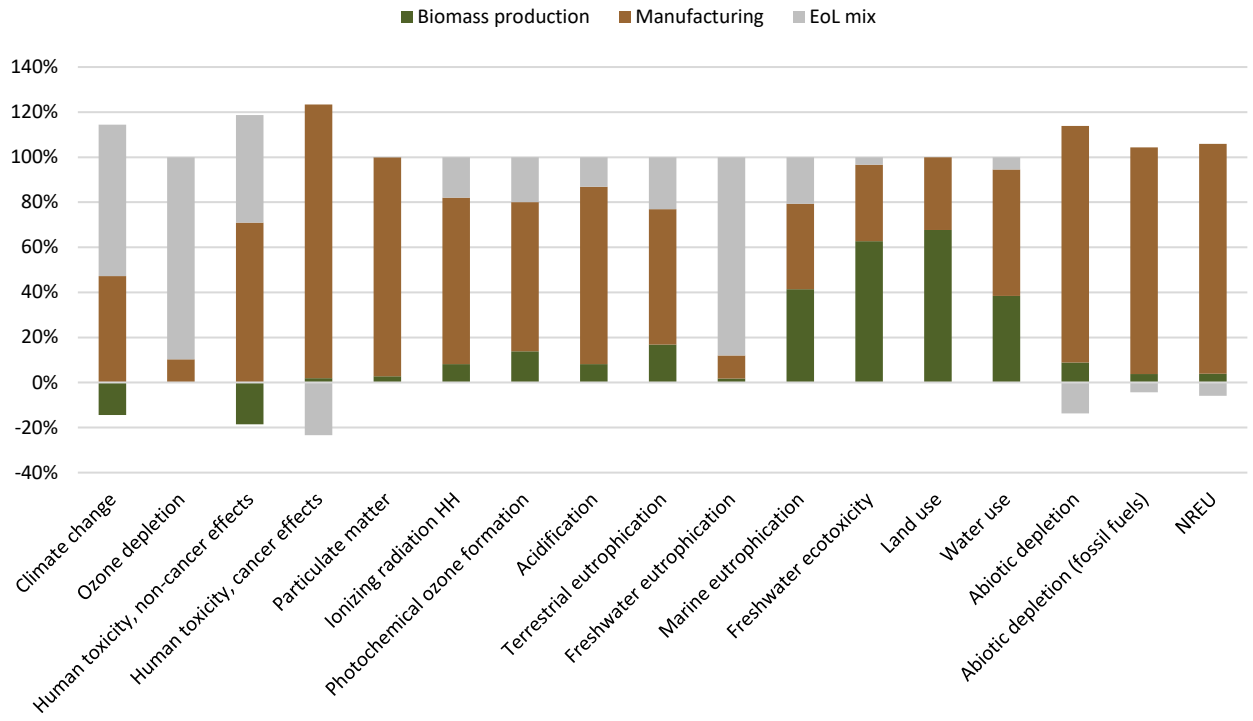


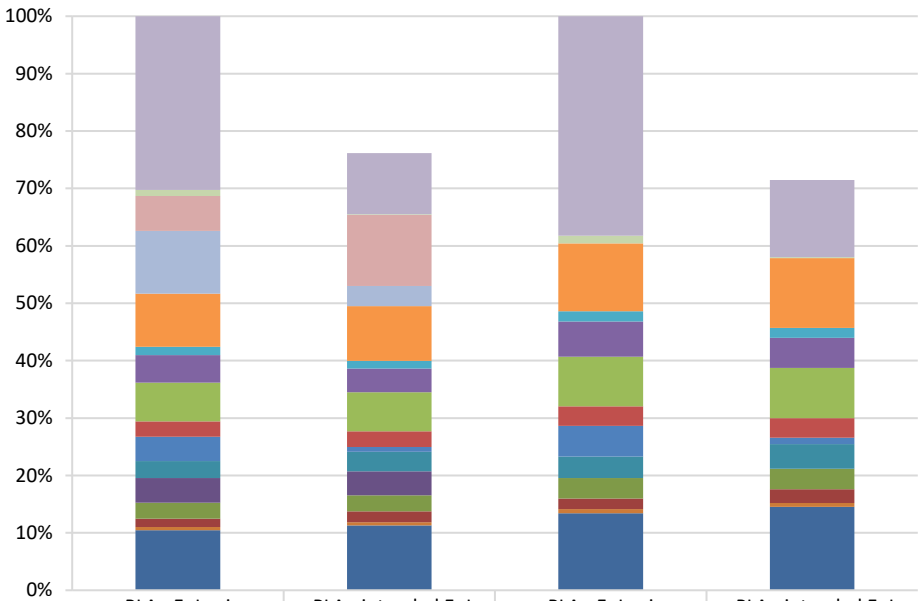
Figure 123. Breakdown of the cradle-to-grave results for the baseline (cutlery made from PLA, weighted average of two PLA sources, see text, EoL mix, without ILUC and DLUC effects).

As observed in Figure 123, at the EoL stage (EoL mix) savings are achieved in the categories of human toxicity (cancer effects), abiotic depletion, abiotic depletion (fossil fuels) and non-renewable energy use. These savings are mostly due to the energy recovery from waste incineration. However, the EoL stage (EoL mix) has a huge contribution to climate change, ozone depletion and freshwater eutrophication. A significant contribution is also observed in human toxicity (non-cancer effects).

The biomass production life-cycle stage has negative impacts in climate change and human toxicity (non-cancer effects). The negative impact on climate change is due to biogenic carbon removal. On the other hand, a high contribution from the biomass production stage is observed in the category of land use, and the categories of marine eutrophication, freshwater ecotoxicity and water use (caused by using irrigation water and fertilisers and pesticides during cultivation).

Manufacturing of cutlery (including transport of intermediate and final products) is the main environmental burden in 11 out of 17 assessed impact categories. Detailed interpretation of single life-cycle stages can be found in section 3.1.

To identify hot spots of the bio-based baseline system, normalisation and weighting of cradle-to-grave results has been performed. Detailed description of this procedure, including normalisation and weighting factors, can be found in the chapter "*Approach and Methodology*". By the weighting of EoL options (see section 3.2.2.5), industrial composting was found to be the EoL option with the lowest environmental impact. The results for EoL mix (see text) and intended waste management for bio-based cutlery – 100 % industrial composting – have been compared. The weighting of the baseline for both the EoL mix and intended EoL (with and without toxicity, respectively) for 16 impact categories is presented in Figure 124.



	cPLA - EoL mix (weighting incl. toxicity)	cPLA - intended EoL (weighting incl. toxicity)	cPLA - EoL mix (weighting excl. toxicity)	cPLA - intended EoL (weighting excl. toxicity)
Climate change	30%	11%	38%	13%
Ozone depletion	1%	0%	1%	0%
Human toxicity, non-cancer effects	6%	12%	0%	0%
Human toxicity, cancer effects	11%	4%	0%	0%
Particulate matter	9%	10%	12%	12%
Ionizing radiation HH	1%	1%	2%	2%
Photochemical ozone formation	5%	4%	6%	5%
Acidification	7%	7%	9%	9%
Terrestrial eutrophication	3%	3%	3%	3%
Freshwater eutrophication	4%	1%	5%	1%
Marine eutrophication	3%	3%	4%	4%
Freshwater ecotoxicity	4%	4%	0%	0%
Land use	3%	3%	4%	4%
Water use	1%	2%	2%	2%
Abiotic depletion	1%	1%	1%	1%
Abiotic depletion (fossil fuels)	10%	11%	13%	15%

Figure 124. Normalised and weighted cradle-to-grave results for the baseline (impact categories) of 1 FU cutlery.

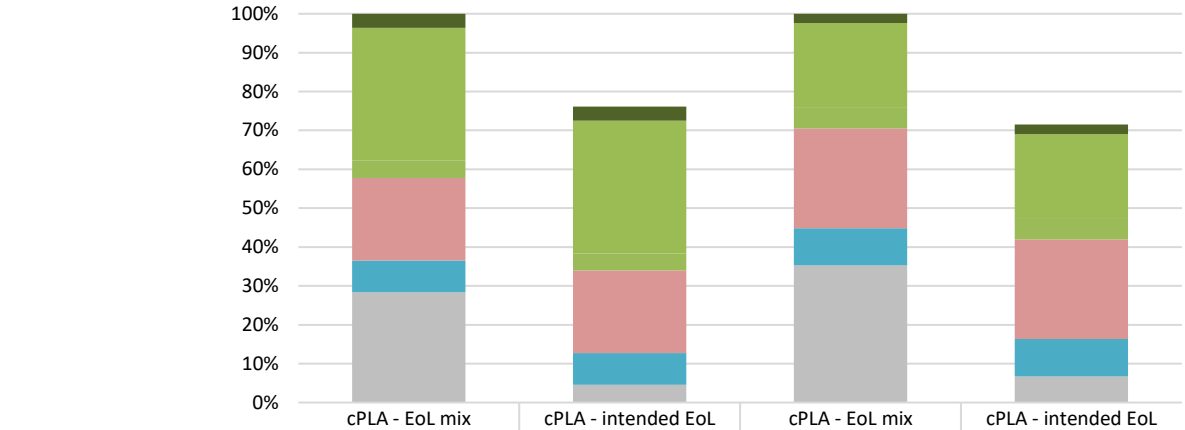
As shown in Figure 124, weighting has been carried out for two cases – one which includes toxicity, and the other without. Categories with a cumulative environmental impact of 80 % and more are identified as the hot spots:

- **Weighting including toxicity (EoL mix):** climate change (30 %), human toxicity (cancer effects) (11 %), abiotic depletion (fossil fuels) (10 %), particulate matter (9 %), acidification (7 %), human toxicity (non-cancer effects) (6 %), photochemical ozone formation (5 %) and freshwater eutrophication (4 %),

- **Weighting excluding toxicity (EoL mix):** climate change (38 %), abiotic depletion (fossil fuels) (13 %), particulate matter (12 %), acidification (9 %), petrochemical ozone formation (6 %) and freshwater eutrophication (5 %).

A detailed description of the identified impact categories is provided in section 3.1.

The weighting of impact categories showed a wide range of categories covering 80 % of the total impact while climate change and abiotic depletion (fossil fuels) cover together 40-51 % of the total impacts (for the EoL mix). To better understand and identify hot spots of environmental burden, a weighting for life-cycle stages has been carried out. The results are presented in Figure 125.



	cPLA - EoL mix (weighting incl. toxicity)	cPLA - intended EoL (weighting incl. toxicity)	cPLA - EoL mix (weighting excl. toxicity)	cPLA - intended EoL (weighting excl. toxicity)
Biomass cultivation	4%	4%	2%	2%
Lactic acid and PLA production	34%	34%	22%	22%
Transportation of PLA	4%	4%	5%	5%
Cutlery production	21%	21%	26%	26%
Distribution of cutlery	8%	8%	10%	10%
EOL	28%	5%	35%	7%

Figure 125. Normalised and weighted cradle-to-grave results for the baseline (life-cycle stages, EoL mix and intended EoL).

Based on the results from weighting of the life-cycle stages, the following stages are identified as the hot spots covering together more than 80 % of the overall impact:

- **Weighting including toxicity (EoL mix):** lactic acid and PLA production (34 %), EoL (28 %) and cutlery production (21 %),
- **Weighting excluding toxicity (EoL mix):** EoL (35 %), cutlery production (26 %) and lactic acid and PLA production (22 %).

It can be observed from the results that the biomass cultivation has a negligible impact on the overall score. Also, a minor impact (less than 10 %) is assigned to transportation of PLA granulates and distribution of the cutlery. The main environmental burden originates from three life-cycle stages, namely: lactic acid and PLA production, EoL and cutlery production. A major activity within these stages is the energy production, causing the release of carbon dioxide and particulate matter (<2.5 μm) emissions, but also causing the depletion of fossil fuels (especially coal and natural gas). A detailed description of impacts due to the life-cycle stages can be found in section 3.1.

The relatively high impact of the EoL stage (EoL mix) should be evaluated for further conclusions. A description of the EoL results can be found in section 3.2. It can be observed from the results shown in Figure 124 and Figure 125 that if the intended EoL was applied, the overall impact per weighted results would be lower by 24 % (including toxicity) or 29 %

(excluding toxicity) compared to the baseline with EoL mix. Moreover, the impact in the climate change category is significantly decreased by the intended EoL – from 30 % to 11 % in the weighted results including toxicity, and from 38 % to 13 % in the weighted results excluding toxicity.

3.3.2 Petrochemical reference system

The overall cradle-to-grave environmental impacts of the petrochemical reference system (PS cutlery) are shown in Table 149. Based on Chapter 6 *Ranges for environmental impacts from production of fossil-based plastics*, 9 out of 17 impact categories/indicators were found suitable for petrochemical PS interpretation – namely: climate change, particulate matter, photochemical ozone formation, terrestrial eutrophication, freshwater eutrophication, marine eutrophication, water use, abiotic depletion (fossil fuels) and NREU. The main reasons for the exclusion of most of the assessed impact categories of PlasticsEurope's eco-profile for PS are due to large differences identified among the impact assessment results provided by the main LCA databases, non-compliance with several ILCD requirements, and a lack of transparency in the allocation approach. The breakdown of the reference product system, shown in Table 149, is divided into aggregated life-cycle stages – manufacturing of the PS cutlery (including also PS production, transportation of PS granulates and distribution of cutleries) and the EoL The results are presented without the effects of ILUC or DLUC.

Table 149. Cradle-to-grave life cycle impact assessment results for the reference (cutlery made from PS, EoL mix).

Impact category	Unit	Manufacturing	EoL mix	Total impact per 1 FU of PS cutlery (cradle-to-grave, EoL mix)
Climate change	kg CO_2 eq.	4.50E+01	8.92E+01	1.34E+02
Particulate matter	kg PM2.5 eq	2.29E-02	-1.55E-03	2.14E-02
Photochemical ozone formation	kg NMVOC eq	1.42E-01	5.23E-02	1.94E-01
Terrestrial eutrophication	molc N eq	4.75E-01	2.05E-01	6.80E-01
Freshwater eutrophication	kg P eq	4.24E-04	2.10E-02	2.14E-02
Marine eutrophication	kg N eq	4.33E-02	1.73E-02	6.06E-02
Water use	m3	1.02E+01	-3.08E+00	7.14E+00
Abiotic depletion (fossil fuels)	MJ	1.02E+03	-1.30E+02	8.86E+02
NREU	MJ	1.04E+03	-1.46E+02	8.91E+02

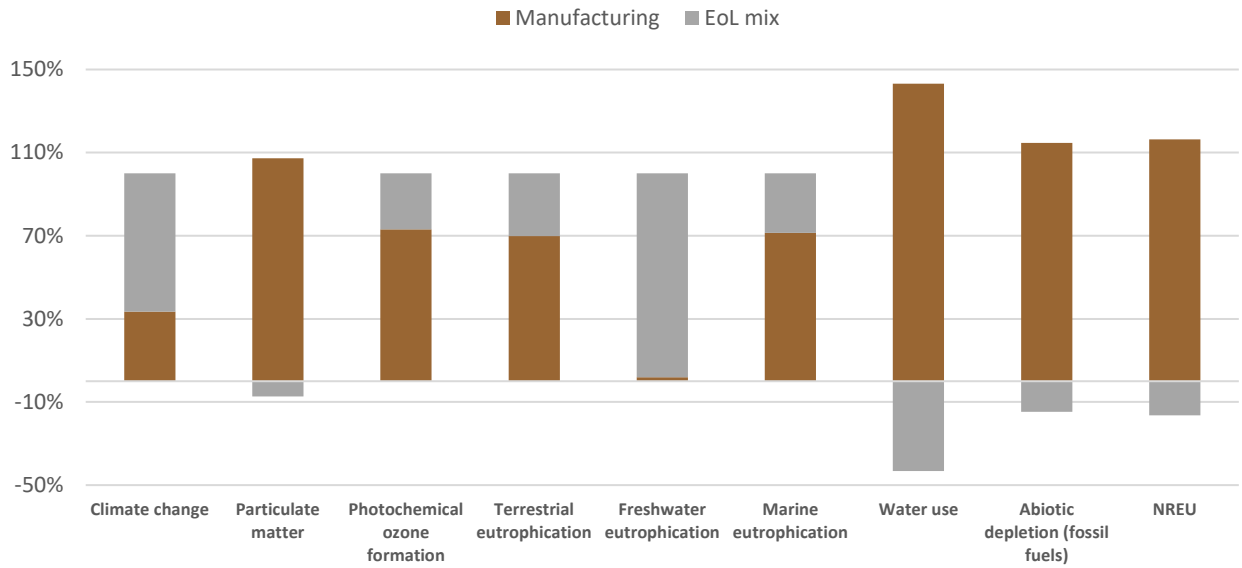
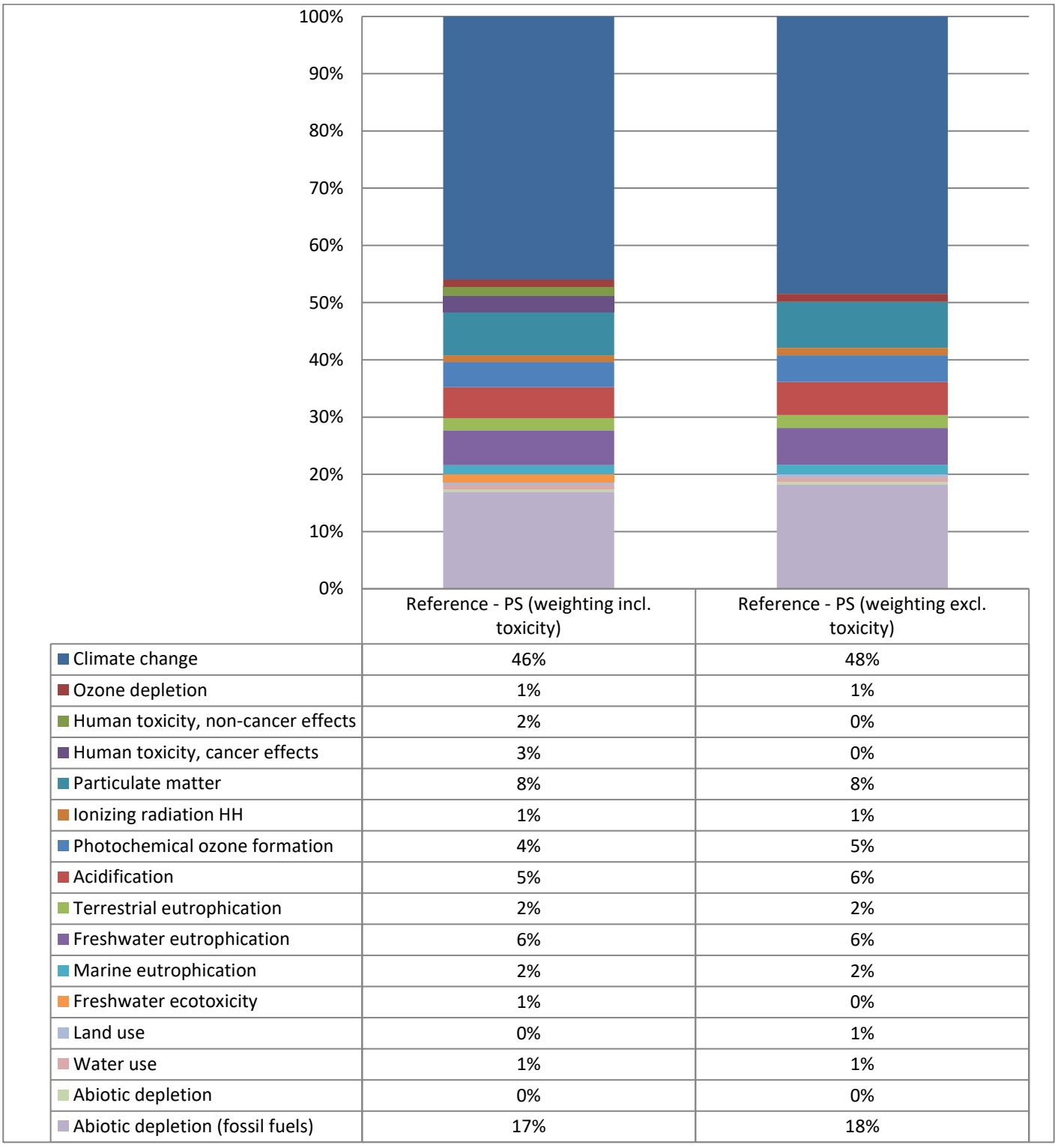


Figure 126. Breakdown of the cradle-to-grave results for the reference (cutlery made from PS).

As observed from Figure 126, a high EoL saving for the petrochemical reference system is achieved in the category of water use. Minor savings from the EoL stage are achieved in the categories of particulate matter, abiotic depletion (fossil fuels) and non-renewable energy use (due to the energy recovery from incineration and the recycling of material). A major contribution from the EoL stage is observed for climate change and freshwater eutrophication. In the remaining impact categories, the major contribution is caused by the life-cycle stage of cutlery manufacturing.

Normalisation and weighting have been applied to the reference systems to understand which among the assessed impact categories are more relevant for this application and what the main environmental hotspots are. To do this, the impact coming from impact categories which were not judged reliable were also taken into account. As the weighted results are affected by the considerable uncertainties highlighted in Chapter 6, their numerical values have to be considered only as indicative. Detailed description of this procedure, including normalisation and weighting factors, can be found in the chapter "*Approach and Methodology*". Weighting of the reference for impact categories is presented in Figure 127.



	Reference - PS (weighting incl. toxicity)	Reference - PS (weighting excl. toxicity)
Climate change	46%	48%
Ozone depletion	1%	1%
Human toxicity, non-cancer effects	2%	0%
Human toxicity, cancer effects	3%	0%
Particulate matter	8%	8%
Ionizing radiation HH	1%	1%
Photochemical ozone formation	4%	5%
Acidification	5%	6%
Terrestrial eutrophication	2%	2%
Freshwater eutrophication	6%	6%
Marine eutrophication	2%	2%
Freshwater ecotoxicity	1%	0%
Land use	0%	1%
Water use	1%	1%
Abiotic depletion	0%	0%
Abiotic depletion (fossil fuels)	17%	18%

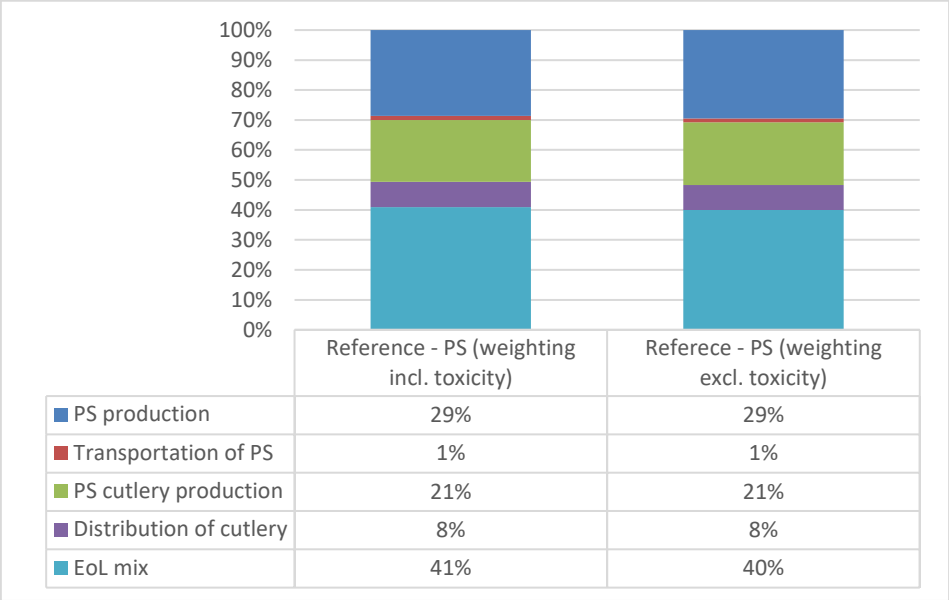
Figure 127. Weighted cradle-to-grave results for the reference (impact categories, EoL mix).

As shown in Figure 127, weighting has been carried out for two cases – one, which includes toxicity, and the other without. Categories with a cumulative environmental impact of at least 80 % are identified as the hot spots:

- **Weighting including toxicity:** climate change (46 %), abiotic depletion (fossil fuels) (17 %), particulate matter (8 %), freshwater eutrophication (6 %) and acidification (5 %),

- **Weighting excluding toxicity:** climate change (48 %), abiotic depletion (fossil fuels) (18 %), particulate matter (8 %) and freshwater eutrophication (6 %).

From the results, the climate change category visibly dominates (45-49 %) in the overall weighting, compared to the other categories. To better understand where these emissions are coming from, and to identify hot spots for life-cycle stages, a weighting for the life-cycle stages has been carried out. The results are presented in Figure 128.



	Reference - PS (weighting incl. toxicity)	Referece - PS (weighting excl. toxicity)
PS production	29%	29%
Transportation of PS	1%	1%
PS cutlery production	21%	21%
Distribution of cutlery	8%	8%
EoL mix	41%	40%

Figure 128. Weighted cradle-to-grave results for the reference (life-cycle stages).

Based on the results from weighting of the life-cycle stages, the following stages are identified as the hot spots (life-cycle stages with a cumulative impact over 80 %):

- **Weighting including toxicity:** EoL (41 %), PS production (29 %) and PS cutlery production (21 %),

- **Weighting excluding toxicity:** EoL (40 %), PS production (29 %) and PS cutlery production (21 %).

A main activity within the identified hotspots is the energy production (i.e. energy needed for PS and PS cutlery production). Released carbon dioxide and particulate matter (<2.5 μm) emissions, but also abiotic depletion of fossil fuels (coal and natural gas for energy production, crude oil for PS production) cause the main environmental burden from the overall impact.

3.3.3 Comparing the Bio-Based baseline with petrochemical reference

The cradle-to-grave environmental impacts of bio-based cPLA cutlery (baseline) and fossil fuel-based PS cutlery (reference) per one functional unit are compared in Figure 129, where only the recommended environmental categories are presented. Further information about the methodology behind this selection can be found in Chapter 6 *Ranges for environmental impacts from production of fossil fuel-based plastics.* Among the 16 PEFCR impact categories, only eight categories are appropriate for a comparison of both product systems in this case study. The results per impact category of the intended EoL for the baseline system (cPLA

cutlery) are also presented and compared with the EoL mix of both the baseline and the reference systems.

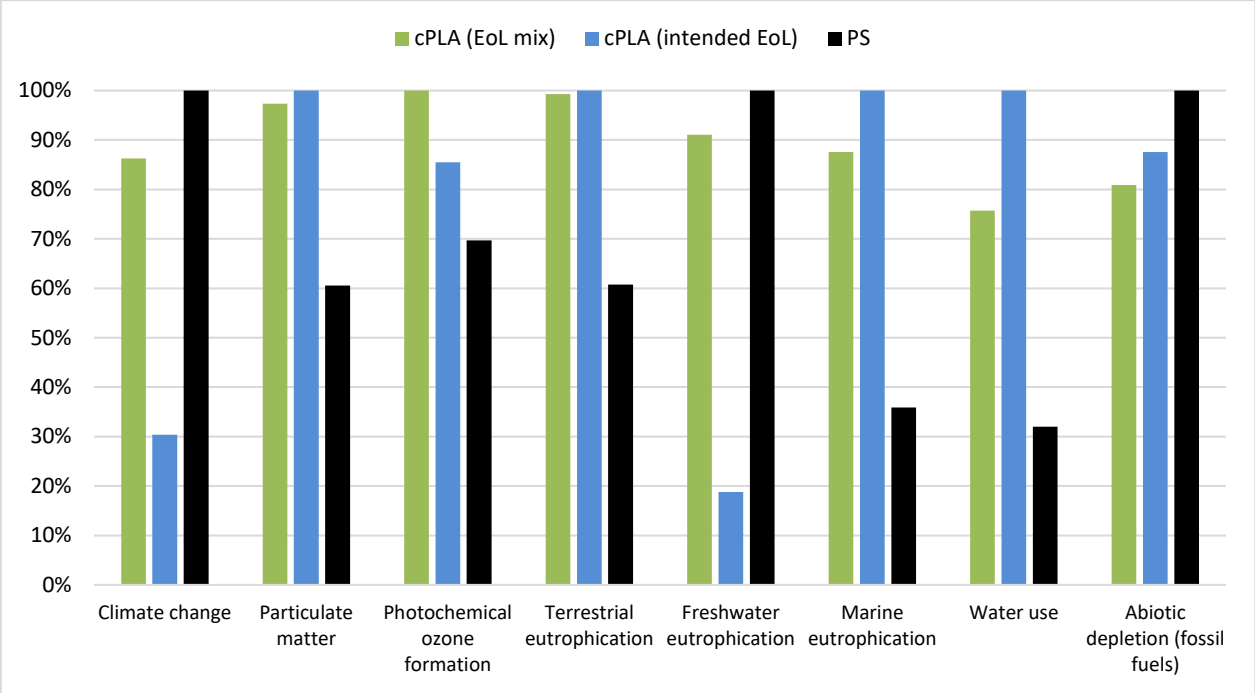


Figure 129. Comparison of bio-based cPLA cutlery (baseline, both EoL mix and intended EoL) with petrochemical PS cutlery (reference) for 1 functional unit, cradle-to-grave LCA results without DLUC and ILUC for the climate change impact category.

Based on the results from cradle-to-grave assessment of the baseline and reference product systems given in Figure 129, bio-based cPLA cutlery (EoL mix) has higher impacts than the fossil fuel-based PS cutlery in five out of eight categories. These five impact categories are: particulate matter, photochemical ozone formation, terrestrial eutrophication, marine eutrophication and water use. Significantly better performance by the petrochemical system (impact more than 50 % lower compared to the bio-based system with EoL mix) is observed for marine eutrophication and water use.

In the remaining three categories - namely climate change, freshwater eutrophication and abiotic depletion (fossil fuels), is bio-based cPLA cutlery with EoL mix (the baseline) performs better. However, the results for both product systems in these categories are comparable as the difference between them lies in the range from 5 % to 20 %. Thus, there is not a category where cPLA cutlery performs significantly better over PS cutlery.

Significant savings (more than 50 %) for climate change and for freshwater eutrophication can be observed from the results when comparing the baseline and the reference to the indented waste management.

Similar results for the bio-based baseline of the EoL mix and the intended EoL are obtained in four categories: particulate matter, terrestrial eutrophication, abiotic depletion (fossil fuels) and non-renewable energy use (not shown in the graph but similar to AD fossil fuels). In these categories, the intended EoL performs slightly worse, but by no more than 10 %.

The worst performance for the intended EoL compared to the EoL mix of the baseline system is observed in the marine eutrophication category (the baseline has 12 % lower impact) and the water use category (the baseline has 24 % lower impact).

4 Discussion

4.1 Sensitivity analysis of electricity consumption of cutlery production

By interviewing several cutlery producers a range of different electricity consumptions per 1 kg of manufactured cutlery was found. The highest and the lowest value of the electricity consumption range varies by ±33 % compared to the baseline. For the baseline, an average value was chosen. To evaluate how the final environmental impact might vary, a sensitivity analysis of electricity consumption of cutlery production has been carried out. The results are shown in Table 150.

Table 150. Sensitivity analysis of electricity consumption of cutlery production – a difference compared to the baseline ((+) higher range limit of electricity consumption, (-) lower range limit of electricity consumption).

Impact category	Unit	Difference
Climate change	kg CO_2 eq.	± 6 %
Ozone depletion	kg CFC-11 eq	± 0 %
Human toxicity, non-cancer effects	CTUh	± 1 %
Human toxicity, cancer effects	CTUh	± 1 %
Particulate matter	kg PM2.5 eq	± 16 %
Ionizing radiation HH	kBq U235 eq	± 1 %
Photochemical ozone formation	kg NMVOC eq	± 5 %
Acidification	molc H^+ eq	± 8 %
Terrestrial eutrophication	molc N eq	± 5 %
Freshwater eutrophication	kg P eq	± 1 %
Marine eutrophication	kg N eq	± 3 %
Freshwater ecotoxicity	CTUe	± 1 %
Land use	kg C deficit	± 2 %
Water use	m^3	± 4 %
Abiotic depletion	kg Sb eq	± 3 %
Abiotic depletion (fossil fuels)	MJ	± 10 %
NREU	MJ	± 9 %

The results show that changing the parameter of electricity consumption by ±33 % in the cutlery production step does not have a significant impact on the overall score – in almost all impact categories the changes are less than ±10 %, compared to the baseline. The only category where the change of impact is higher (±16 %) is particulate matter, which is primarily caused by the particulate matter (<2.5 μm) emitted during the electricity production.

4.2 End of life sensitivity analysis

The sensitivity analysis was performed for the intended EoL option, industrial composting of PLA mix. The significant parameter chosen, which is likely to change due to technical improvements in both the plastic products and in the waste management technology, is the degradation percentage of the biogenic carbon in the product. These parameter changes within a given technology whereas modelled because the average European technology in the EoL scenarios represents a rather broad set of technological variances. This change will provide an

insight into the variance within these technologies and depict a target for future waste scenarios. The parameter variation is presented in Table 151.

Furthermore, the amount of food leftovers which are composted have been analysed, and the range can be observed in Table 151, which is based on the previously described ranges. Lastly, a lower sorting and technology efficiency when recycling PS was investigated.

Table 151. Variation in input parameters in the sensitivity analysis.

Product system	Technology	Parameter changed	Original EoL	Lower	Higher	References
PLA mix	Industrial Composting	Biogenic carbon degradation	57.1	55	95	Hermann, 2011
PLA mix	Industrial composting	Amount of food leftover (kg per kg plastic)	5.2	2.5	8	Silvennoinen et al., 2015
PS	Recycling	Sorting and technology efficiency (%)	70	25	-	EUCertplast, 2018
PLA mix	EoL collection mix	Collection to industrial composting	30%	5%		

4.2.1 Industrial composting bio-based cPLA

The sensitivity analysis was performed on the industrial composting technology. The parameter of biogenic carbon degradation was changed within a limit found in literature (see in Table 150). The lower limit was set to 55 % and the upper limit 95 % C degradation. The original value for the product is 57.1 %. The lower limit is therefore relatively close to the original value. Table 151 presents the sensitivity analysis for the bio-based cPLA cutlery including food leftovers. Figure 130 shows the percentage change in climate change compared with the original EoL (PLA plus the additional food leftovers). When the percentage is negative, the results are lower than for the original and vice versa.

The only impact categories that are affected by the C degradation are the climate change categories. There is a 3 % difference in the total climate change impact between the lower limit and the original value. Using the higher C degradation limit will lead to a decrease in climate change impact of 0.12 kg CO_2-eq, which is a decrease of 45 % in the climate change results compared to the original. There are three main reasons for the changes in the climate impact for biogenic carbon;

- Increasing the degradation percentages means that less of the biogenic carbon will be incinerated with the rejects and contribute to direct emissions of biogenic CO_2.
- Less biogenic carbon will be released to the atmosphere in the form of the compost product that is spread on the land later.
- More biogenic carbon is emitted directly from the composting plant when the biodegradation percentage is higher.

Considering the fossil CO_2, a lower C degradation will lead to a small saving of fossil CO_2 as there is less compost to be spread on land with a diesel consuming tractor. A higher C degradation will lead to less savings of fossil CO_2 when the compost is used on land, which is why the change is positive.

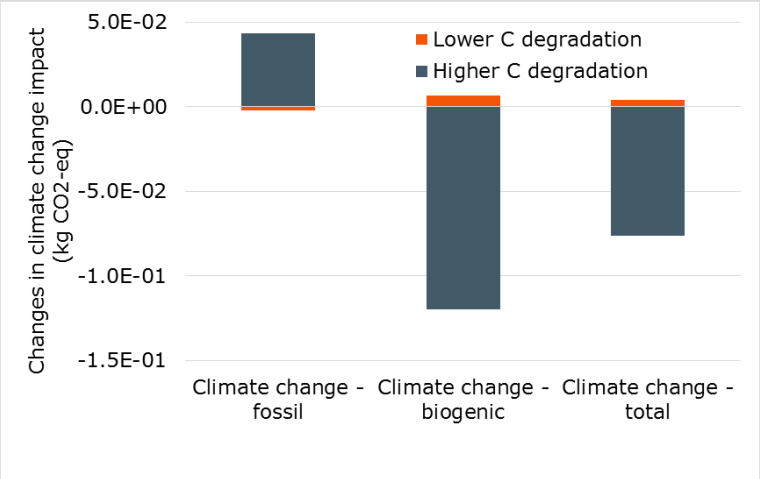


Figure 130. Sensitivity analysis for the parameter C degradation, for bio-based cPLA cutlery.

This sensitivity would decrease the cradle-to-grave climate change results when increasing the C degradation by less than 0.01 % and when decreasing C degradation, the cradle-to-grave results will increase the total climate change impact by less than 1 %. The sensitivity analysis shows that altering this parameter would not change the priorisation between the product systems and their individual EoL technologies' performance for climate change.

4.2.2 Varying the amount of food leftovers within the baseline product system

The amount of food leftovers incoming to the waste streams together with the cutlery is an uncertain parameter, the sensitivity of this parameter is assessed below. This implies that the EoL reference flow changes with including food leftovers, as a range of values were found in literature.

Figure 131 presents the percentage change in total results compared with the original (PLA plus the additional food leftovers). When the percentage is negative, the results are lower than for the original and vice versa. The original value for food leftovers is 5.2 kg/kg PLA, the lower limit is 2.5 kg/kg PLA and the higher is 8.0 kg/kg PLA.

The sensitivity analysis shows that the additional food leftovers have a substantial influence on the overall results. The percentage change is equal for both values, but always in opposite directions, as the original value was adopted as the average of the higher and lower limit. Overall the influence of additional food leftovers for industrial composting with PLA cutlery will increase the impact in most impact categories. This includes global warming potential, human toxicity (cancer effects) and abiotic depletion.

The impact category climate change (biogenic) shows a decrease when the amount of food leftovers is increased. More food leftovers will create a larger amount of compost that will replace conventional fertilisers and therefore give higher savings in terms of fertiliser substitution. The total climate change impact will however be negatively affected by the increase in food waste, as direct emissions increase, transportation increases and energy and resource use at the composting plant increases.

Composting more food leftovers decreases the impact on abiotic depletion and human toxicity – cancer. This is mostly due to the substitution achieved from fertiliser substitution. The non-

renewable energy use is the most sensitive impact category to changes in the amount of food leftovers.

The energy use in the composting plant is largely dependent on the amount of waste treated influencing categories such as ozone depletion, ionizing radiation, acidification and terrestrial eutrophication.

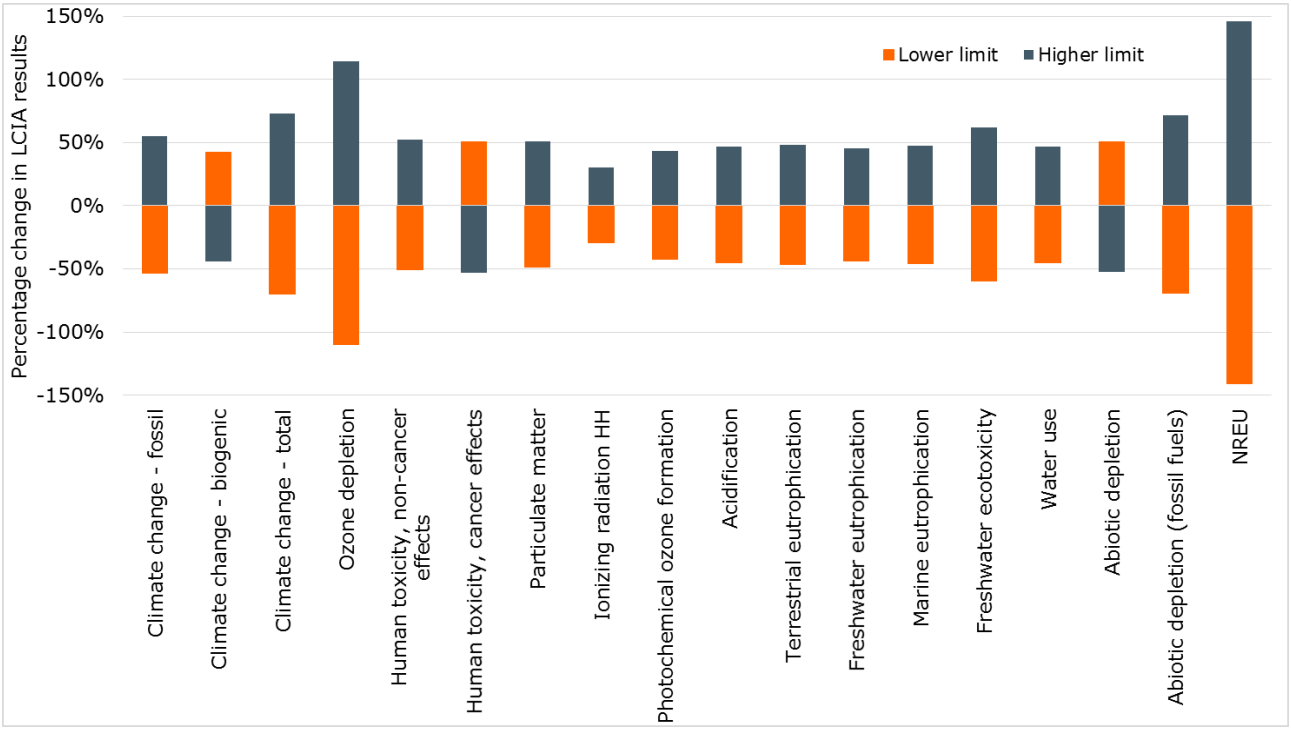


Figure 131. Sensitivity analysis for additional food leftovers for industrial composting of cPLA cutlery.

Table 152 shows the sensitivity for the cradle-to-grave results when the amount of food leftover is changed, for the seven selected comparable impact categories. The change in amount of food leftovers was only performed on the industrial composting technology while for the other EoL technologies in the EoL mix the food leftover amount was kept to 5.2 kg/kg PLA. It is though reasonable that the food leftovers would also increase in other EoL technologies.

Changing the amount of food leftovers in industrial composting has a low sensitivity (under 5 %) on the cradle-to-grave results in the impact categories climate change, particulate matter, photochemical ozone formation, abiotic depletion (fossil fuels) and NREU. Terrestrial eutrophication is the most sensitive of the selected impact categories to changes in the food leftovers, with a 10 % change. By changing the amount of food leftovers from 5.2 to 8 kg/kg PLA in industrial composting the LCIA cradle-to-grave results will increase by 6 % in the impact category acidification. The composting facility contributes to the largest amount of terrestrial eutrophication and acidification, when increasing/decreasing the amount of waste treated in the composting facility will cause a change in the amount of resources needed in the facility (e.g. electricity, diesel).

Table 152. The change in percentage of higher and lower amount of food leftovers in the industrial composting to the cradle-to-grave results of seven selected impact categories.

Impact category	Lower amount of food leftovers	Higher amount of food leftovers
Climate change	-2 %	2 %
Particulate matter	-1 %	1 %
Photochemical ozone formation	-2 %	2 %
Acidification	-6 %	6 %
Terrestrial eutrophication	-10 %	10 %
Abiotic depletion (fossil fuels)	-3 %	3 %
NREU	-2 %	2 %

4.2.3 Higher sorting and technology efficiency of recycling

PS plastic from households can be problematic to recycle due to technical limitations (Plastic Recyclers Europe, 2017) and furthermore not all European recycling facilities sort out PS today (EUCertplast, 2018). Furthermore, plastic recycling is affected by the market. A sensitivity analysis is carried out with higher recycling of PS, which includes a higher sorting and technology efficiency as the technical capacity is likely to improve in the future, alongside a higher collection rate of the plastic. In total the recycling rate is increased from 7.5 % (30 % collection rate x 25 % sorting and technology efficiency) to 50 % (71 % collection rate x 70 % sorting and technology efficiency).

The results of the sensitivity analysis are presented in Figure 132. Lowering this parameter leads to a decrease in the recycling of PS, substitution of PS and a higher amount of PS to incineration. The results show that overall the impacts increase when the sorting and technology efficiency is lowered. The impact categories show an increase from 55 % to 635 % in impacts when lowering this parameter to 45 %, indicating that this is a sensitive parameter in the recycling EoL technology for PS. Three impact categories (climate change – biogenic, ozone depletion and abiotic depletion) show a decrease in impact when lowering this parameter. Lowering the recycling sorting and technology efficiency will increase the fraction that is incinerated.

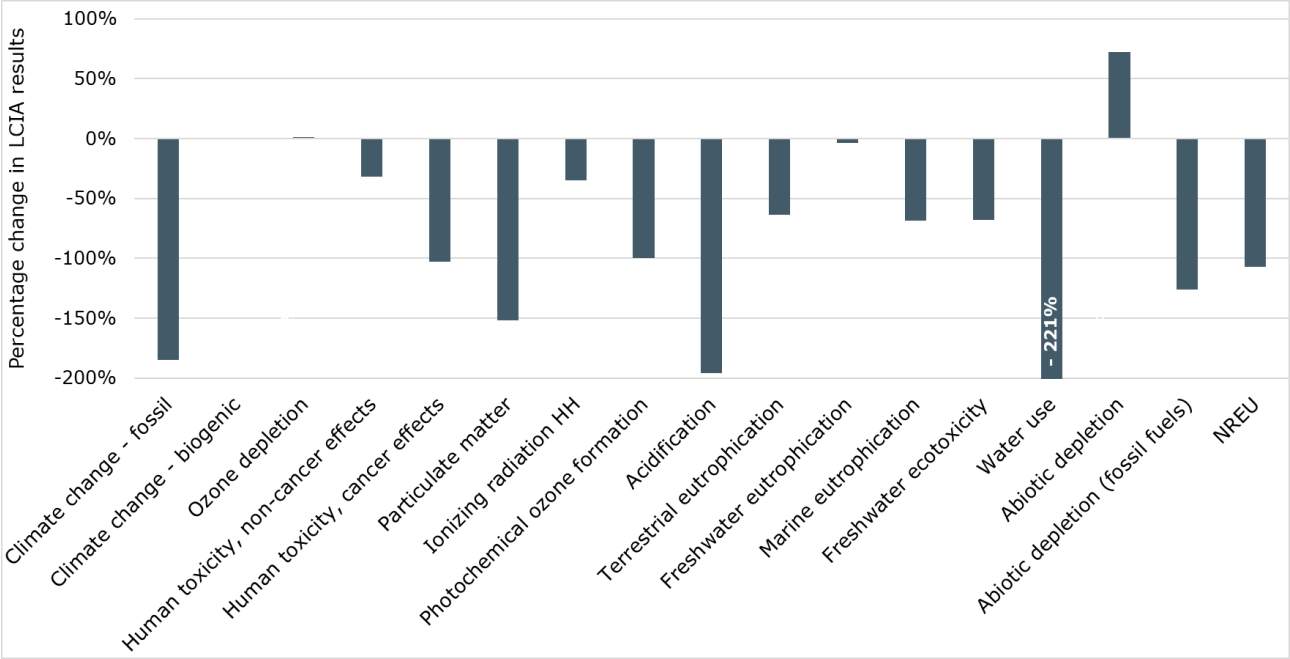


Figure 132 Sensitivity analysis for increasing the sorting and technology efficiency of recycling PS cutlery.

The sensitivity of this parameter on the overall cradle-to-grave LCIA results with EoL mix are presented in Table 153. The sensitivity is above 5 % for all impact categories except acidification and all the comparable categories show an increase in impact when lowering this parameter. From the sensitivity analysis it is evident that the recycling sorting and technology for PS is a sensitive parameter.

Table 153. The change in percentage of a higher collection rate as well as a higher sorting and technology efficiency in recycling of PS cutlery compared to the cradle-to-grave results of seven selected impact categories.

Impact category	Unit	Higher collection and recycling
Climate change	kg CO_2 eq.	-28 %
Particulate matter	kg PM2.5eq	-693 %
Photochemical ozone formation	kg NMVOCeq	-62 %
Acidification	molc H+eq	-85 %
Terrestrial eutrophication	molc Neq	-45 %
Abiotic depletion (fossil fuels)	MJ	855 %
NREU	MJ	546 %

4.2.4 Lower EoL collection mix for industrial composting of PLA

The percentage of cPLA cutlery collected for industrial composting is an uncertain factor, as there is no data on how much biogenic PLA is sent to industrial composting. It is questionable if the collection rate for recycling is as high as for petrochemical counterparts, when a widespread possibility for all EU member states to sort out organic waste is missing and there is a question as to whether the citizens would sort the PLA cutlery as organic waste.

A sensitivity analysis for lowering the collection of cPLA cutlery to industrial composting is performed according to Table 154. The remaining collection percentage is divided to incineration and landfill according to the European plastic share to each EoL technology (European Commission, 2018).

Table 154. Sensitivity analysis for lowering share of EoL mix collected for industrial composting

	Original EoL collection mix for cPLA	Sensitivity EoL collection mix
Industrial Composting	30 %	5 %
Incineration	39 %	53 %
Landfill	31 %	42 %

The sensitivity when changing the EoL collection mix with lower industrial composting is presented in Figure 133, with the percentage change in the LCLIA results. The GWP increases by 34 % when lowering the collection amount to industrial composting. There are eight impact categories were the impact decreases with this change in EoL collection mix and 11 categories were the impact increases. For particulate matter and for water use, the impact decreases by over 200 %. This change has very little effect on acidification (3 %) and terrestrial eutrophication (1 %). It therefore depends on the impact category, how sensitive the EoL mix is towards lowering the industrial composting share.

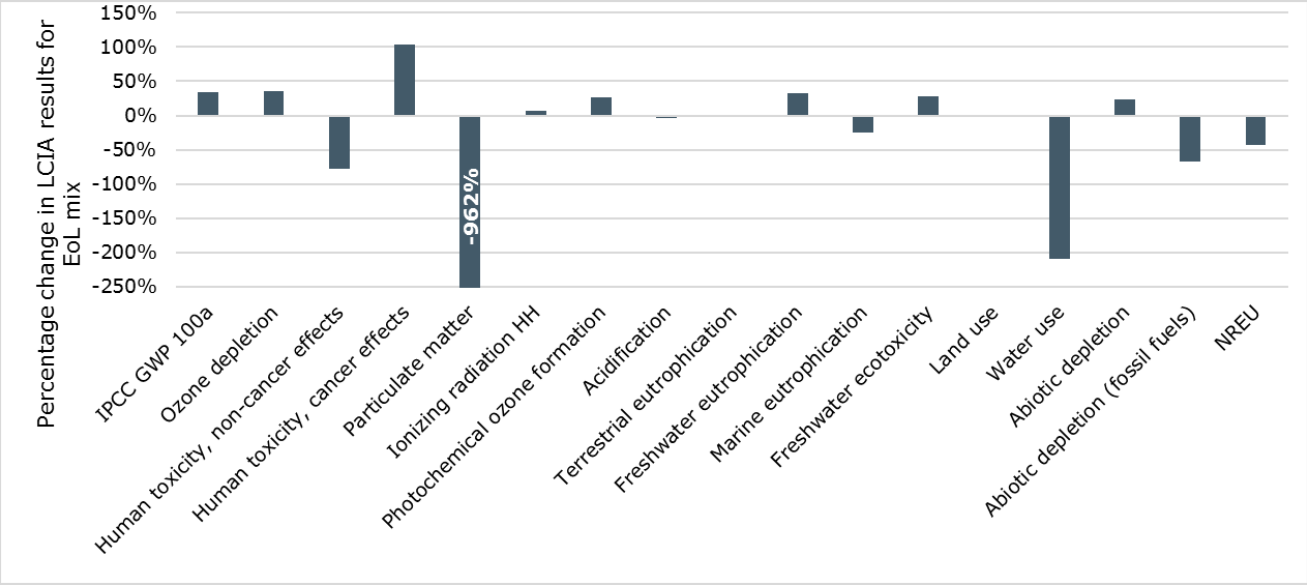


Figure 133 Sensitivity analysis for EoL collection mix with lower industrial composting.

The sensitivity of this change on the overall cradle-to-grave results is presented in Table 155. Changing the industrial composting share in the EoL collection mix to 5% increases the cradle-to-grave GWP results by 23 %. For the other six impact categories selected the percentage change to the cradle-to-grave results are 5 % or lower. The particulate matter was very sensitive when looking solely to the LCIA from the EoL but is not sensitive when the other life cycle stages are included.

Table 155. The change to the cradle-to-grave results of seven selected impact categories when lowering the share of industrial composting in the EoL collection mix.

Impact category	Unit	Higher collection and recycling
Climate change	kg CO_2 eq.	23%
Particulate matter	kg PM2.5eq	-1%
Photochemical ozone formation	kg NMVOCeq	5%
Acidification	molc H+eq	0%
Terrestrial eutrophication	molc Neq	0%
Abiotic depletion (fossil fuels)	MJ	-3%
NREU	MJ	-3%

4.3 Discussion on End of Life uncertainties

The parameters analysed in the sensitivity analysis above and other parameters, which may be uncertain, i.e., estimations, assumptions, huge ranges in industry or EU member states, missing data, etc. are discussed in this section. Methodological shortcomings and additional research requirements are also presented.

Including food leftovers - the cutlery would in some cases not include leftover food, thus the food leftovers could enter a different waste stream than the cutlery. This condition is not accounted for in this study. In this study the food leftovers are assumed to enter the same waste stream as the cutlery, except in the case of the mechanical recycling waste stream, where the food leftovers are shared between landfilling and incineration.

The quantity of food leftovers entering the waste stream with the cutlery is derived from uncertain data with a large margin of uncertainty, as data on how people dispose of their food leftovers is not readily available. Including the food leftovers affects the LCIA both negatively and positively, depending on the impact category. Including the food leftovers in the assessment generally favours the bio-based and biodegradable product system due to the benefits of additional organic waste for appropriate biological treatment. The sensitivity analysis of varying the quantity of food leftovers shows that this parameter is not highly sensitive towards the cradle to grave results, except for terrestrial eutrophication with a change in LCIA of up to 10 % (see Table 152).

Furthermore, the composition of the food leftovers is a sensitivity, as this varies depending on several factors. As the food leftovers have a significant share of all impacts in all EoL technologies (see contribution analysis in Annex 4), this is an assumed to be of major influence on the EoL results.

The collection and recycling rates of PS for recycling are very uncertain. For example, Plastic Recyclers Europe (2017) state that PS collected from households is most often landfilled or incinerated. This is despite initiatives that work on more efficient and less costly recycling processes for PS. Furthermore, the PS of cutlery might have a lower efficiency than other PS products, as the items are small, which could prove a sorting problem in the recycling process. Furthermore, people may be less likely to put dirty cutlery into the plastic recycling bin, which puts in question the collection rate.

The sensitivity analysis of increasing the sorting and technology efficiency shows that an increased efficiency representing other polymers will have a large influence on the LCIA results, both positive and negative, due to a change from energy substitution to plastic

substitution. Table 153 shows that impact categories have decreased up to almost 700 % and increased impacts in other impact categories down to 850 %.

The EoL mix on collection rate for PLA for industrial composting - The percentage of cPLA cutlery collected for industrial composting is an uncertain factor, as described in the sensitivity analysis above. The sensitivity analysis lowering the collection rate of cPLA cutlery to industrial composting shows a change to the overall cradle-to-grave results, presented in Table 155. Cradle-to-grave GWP results increase by 23 % and the other six impact categories increase by less than 5 %.

The quantity of rejects in industrial composting and AD is subject to uncertainties because the pre-sorting and post-sorting in the processing plants have different technical set-ups. The quantity of reject of biodegradable plastic in the plants is also prone to change if more biodegradable plastic enters the waste stream. This is outside of the temporal scope of this study.

Carbon biodegradation in industrial composting. This was analysed with a sensitivity analysis within the industrial composting above. The results from the analysis showed that this parameter will affect the GWP of industrial composting but will not alter the conclusion that industrial composting is the EoL technology that performs second best in terms of GWP for PLA (see Figure 105). Also, VS degradation is uncertain for the industrial comparison, but this was not investigated with a sensitivity analysis. Adjusting the VS degradation is not assumed to have a major influence on the cradle-to-grave results.

The biodegradation of carbon is also uncertain in the technologies of landfilling and anaerobic digestion. This will again affect solely the GWP. These parameters depend, for example, on the biomass and additives within the material and will differ between different types of PLA, but will also differ depending on the specific plant and the conditions in the landfill. The methane yield for PLA is therefore an uncertain factor. As the methane yield for bio-based plastics is relatively low for AD, and because it has a low share contribution to the GWP of AD, is not anticipated that this parameter affects the results. On the other hand, direct LFG emissions are a major impact for landfills, which is why the parameter is expected to have a high impact on results.

Biogenic carbon content in the PLA - The PLA studied in this case study is composed of 100 % biogenic carbon, although some PLA will include fossil carbon from additives and co-polymers, and this will affect the environmental profile, especially in the incineration EoL technology for GWP, where this would have different emission factors between fossil and biogenic carbon emissions.

Apart from the above case study's specific uncertainties, several other general uncertainties that apply to most of the presented case studies are discussed in the Approach and methodology in chapter 3.5., e.g., data quality, critical assumptions and limitations in End of Life modelling. These include uncertainties related to:

- the estimated EoL mix
- the marginal energy technologies utilised
- the development of the EoL technologies (efficiency increases)
- the consumption of materials and energy at the EoL treatment plants
- littering, which is not modelled in this LCA.

Biodegradation of PLA in landfilling, the collection and recycling rates of PS and the composition of the food leftovers are the significant uncertainties.

5 Assessing non-PEF impact categories: taking into account ILUC and endpoint of GTP

5.1 GTP: Global Temperature change potential

A breakdown of the Global Temperature Change Potential (GTP) impacts of bio-based baseline (cPLA cutlery) into the main processes (described in section 2.1.1) is shown in Figure 134.

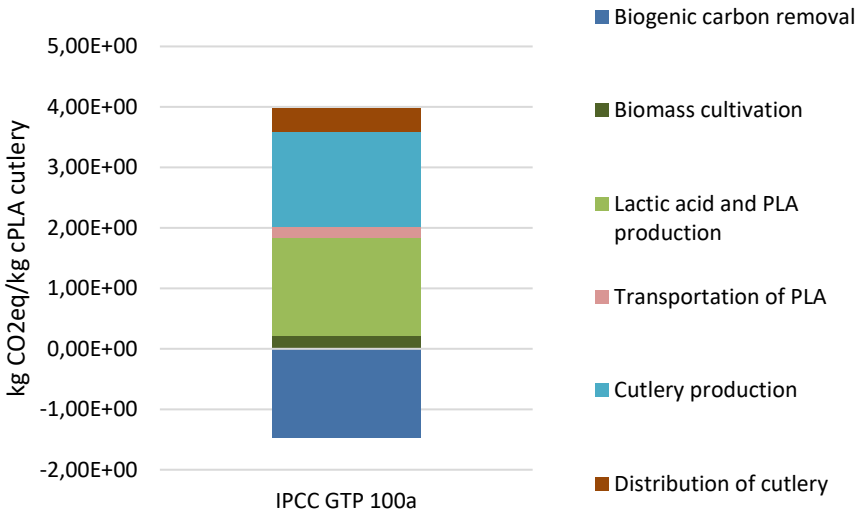


Figure 134. Breakdown of the cradle-to-user GTP 100a results, 1 functional unit, excluding DLUC and ILUC effects.

Observations for GTP are very similar to those for climate change described in section 3.1 – the production of lactic acid and PLA production together with the cutlery production represents the highest impact in the category. When the biogenic carbon removal is considered, the cradle-to-user GTP result is 2.52 kg CO_2 eq./kg cPLA cutlery (climate change result is 2.79 kg CO_2 eq./kg cPLA cutlery).

5.2 Land Use Change emissions

Table 163 presents the cradle-to-grave characterised results broken down into feedstock, manufacturing, EoL, and ILUC. As highlighted in Chapter 4, DLUC and ILUC cannot be summed up. This case study does not involve crops that imply DLUC (namely Brazilian sugarcane, German maize, French wheat and German wheat), based on the methodology from the PEFCR Guidance V6.3 presented in Chapter 4. Figure 135 presents the characterised results for case study 4 (baseline).

Table 156. Characterised cradle-to-grave results broken down per process, including ILUC. Case study 4. Bio-based baseline and EU mix EoL scenario

Impact category	Unit	Biomass production (incl. biogenic C)	Manufacturing	EoL (EU mix)	ILUC
Climate change	kg CO_2 eq.	-17	55	78	6
Ozone depletion	kg CFC-11eq	3.05E-08	3.16E-06	2.79E-05	1.27E-09
Human toxicity, non-cancer effects	CTUh	-2.87E-06	1.09E-05	7.34E-06	1.02E-09
Human toxicity, cancer effects	CTUh	3.15E-08	2.01E-06	-3.87E-07	1.62E-10
Particulate matter	kg PM2.5eq	9.54E-04	3.34E-02	3.48E-05	1.61E-04
Ionizing radiation human health	kBq U235eq	2.27E-01	2.04E+00	4.99E-01	8.76E-05
Photochemical ozone formation	kg NMVOCeq	3.84E-02	1.84E-01	5.57E-02	1.89E-02
Acidification	molc H+eq	3.67E-02	3.53E-01	5.91E-02	3.80E-03
Terrestrial eutrophication	molc Neq	1.86E-01	6.70E-01	2.57E-01	2.13E-02
Freshwater eutrophication	kg Peq	3.37E-04	2.01E-03	1.72E-02	1.40E-05
Marine eutrophication	kg Neq	6.11E-02	5.63E-02	3.05E-02	2.26E-03
Freshwater ecotoxicity	CTUe	1.07E+02	5.80E+01	5.72E+00	1.13E-02
Land use	kg C deficit	1.56E+02	7.48E+01	0.00E+00	1.75E-02
Water use	m3	6.50E+00	9.47E+00	9.25E-01	1.88E-01
Abiotic depletion	kg Sbeq	3.35E-06	3.98E-05	-5.23E-06	5.55E-08
Abiotic depletion (fossil fuels)	MJ	27	721	-31	1.58E-01
NREU	MJ	29	772	-44	1.51E-01

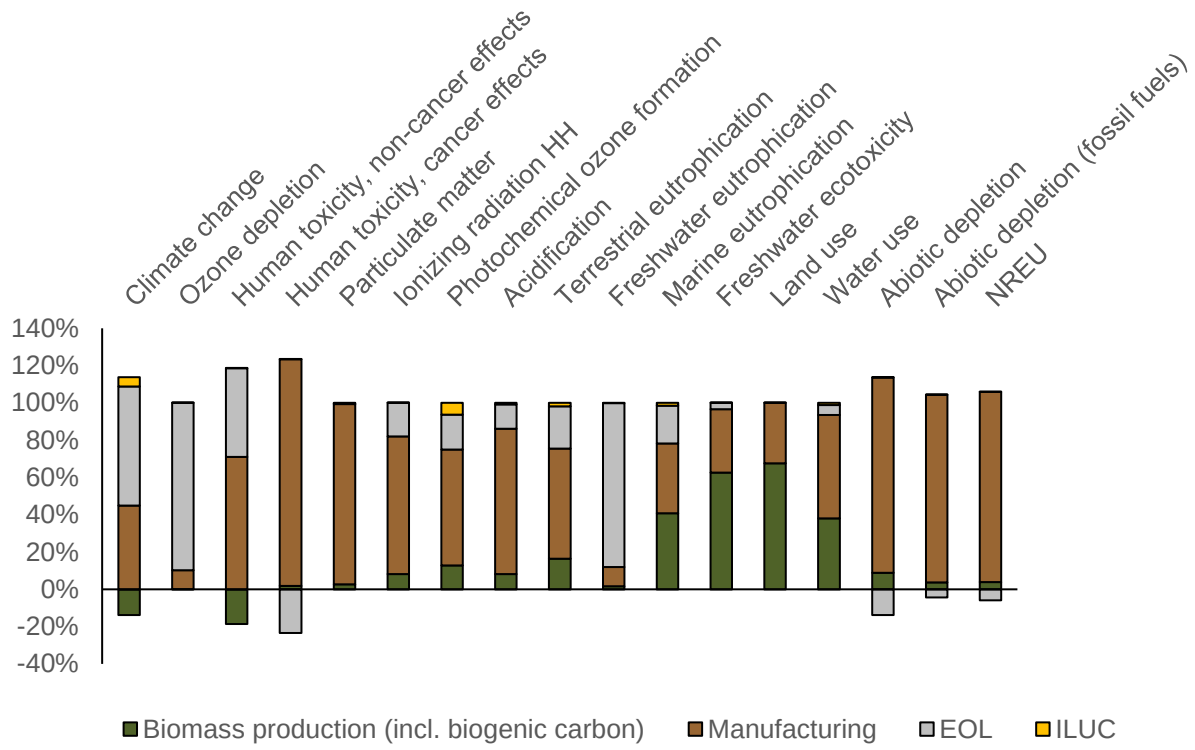


Figure 135. Relative characterised results broken down for all impact categories and including ILUC – case study 4.

As can be seen from Table 156 and to some extent from Figure 135, ILUC does contribute to all impact categories considered herein, as in previous cases. However, as for previous cases, the contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (5 % of the impact), photochemical ozone formation (6 % of the impact), terrestrial eutrophication (2 % of the impact), marine eutrophication (2 % of the impact), and water use (1.1 % of the impact). All these impacts are dominated by land expansion except for water use which is fully dominated by intensification (Figure 15). In the case of climate change, as explained earlier, the ILUC impact is essentially due to the CO_2 releases resulting from land clearing and in the case of photochemical ozone formation, to carbon monoxide also released during land clearing (burning the cleared biomass; Figure 17). Both terrestrial- and marine eutrophication are essentially due to nitrogen oxides released during land clearing (more than 80 % of the impact; Figure 17). The water use impact, on the other hand, essentially reflects the hydropower electricity demanded for the (additional) fertiliser production (intensification process), in particular phosphorus (diammonium phosphate).

In absolute terms, the ILUC impact for climate change here is quite high compared to the one obtained for e.g. case 1 but is still lower than the one of other cases (e.g. case study 2 or case study 5). This reflects the importance of having conversion processes requiring as little biomass as possible, so the climate change impacts from both the crop cultivation and the ILUC can be lower.

6 Conclusions, limitations and recommendations

For the bio-based cutlery made from crystalline PLA, the most important environmental impacts are identified as climate change and abiotic depletion (fossil fuels), together covering 40 % or 51 % of the cradle-to-grave environmental impacts with or without toxicity categories, respectively. Other important categories are human toxicity (cancer effects), particulate matter and acidification. Impacts are mainly associated with process energy and chemicals during the manufacturing phase, as well as the direct emissions during the EoL phase.

The manufacturing phase has the highest contribution of all life cycle stages. Lactic acid and PLA production, together with the cutlery production, accounts for half of the cradle-to-grave total environmental impacts (55 % or 48 % of the total impacts with or without toxicity categories, respectively). Energy consumption and chemicals used in the manufacturing process are responsible for the major parts of the impacts. Note that in the manufacturing phase, the process energy (both heat and electricity) is highly dependent on the fossil fuels origin based on the status-quo production. The electricity used in cutlery production in China and Southeast Asia is also carbon-intensive.

With a global supply chain, the impacts from the transportation services (for polymers and the final products) are not negligible. They account for 12 % or 15 % of the total impacts with or without toxicity categories, respectively.

The impacts from biomass production, however, are less significant (4 % or 2 % of the total impacts with or without toxicity categories). Even though one would expect more resources from land and water are required by the bio-based products, the land use and water use impacts are rather limited compared to the impacts from the manufacturing phase.

The EoL phase accounts for about one-third of the total impact (28 % or 35 % with or without toxicity categories). The status-quo mixed EoL of the PLA cutlery, consisting of incineration (39 %), landfilling (31 %) and industrial composting (30 %), contributes to 70 % of the total climate change impact (without DLUC and ILUC effects), 90 % of ozone depletion (mainly caused by landfilling), 90 % of freshwater eutrophication (mainly caused by incineration). The most preferable choice of EoL for PLA cutlery is industrial composting followed by anaerobic digestion, incineration and landfilling (ordered from the most preferable to the least).

After normalising and weighting the LCIA results the bio-based the cPLA cutlery shows a better performance for GWP, however for the other 5 comparable impact categories the PS cutlery performs slightly better. The difference between the weighted results is small and taking into consideration the uncertainties of this study, the conclusion is that there is no preference for a product system from an EoL perspective. The choice of EoL option hence plays an important role in the decreasing of overall impact and it is largely connected to climate change. If the most preferable EoL, industrial composting, is modelled as the "intended EoL" instead of the status-quo mix, the cradle-to-grave environmental impacts would decrease by 25 % or by 30 % with or without toxicity categories, respectively, while a nearly threefold decrease in climate change was observed.

Changing the carbon degradation to a lower degradation value found in literature, gives a 3 % higher climate change impact from industrial composting. The sensitivity analysis showed that this parameter will not lead to a change in the prioritation of the product systems. The parameter of sorting and technology efficiency for PS recycling is sensitive to the overall cradle-to-gate results, with an increase in the seven comparable impact categories between 1.5 % and 10.5 %. Increasing the amount of food leftovers entering the waste stream together with the cutlery from 5.3 to 8 kg, leads to an increase in the cradle-to-grave LCIA results between 1 and 10 %, depending on the impact category. Furthermore, lowering the sorting and technology efficiency of the recycling of petrochemical PS from 70 % to 25 % due

to low recyclability and the fact that not all recycling plants sort out PS, would decrease the cradle-to-grave results from 1.5 % up to 10.5 % for the seven comparable impact categories.

The feedstock involved for bio-based cutlery imply no DLUC, based on the PEFCR Guidance v6.3 methodology. The contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (5 % of the impact), photochemical ozone formation (6 % of the impact), terrestrial euthrophication (2 % of the impact), marine eutrophication (2 % of the impact), and water use (1.1 % of the impact). All these impacts are dominated by land expansion except for water use which is fully dominated by intensification.

Eight impact categories out of 16 are identified as 'suitable for comparison' between the bio-based and the petrochemical systems (See Chapter 6). From cradle-to-grave on a functional unit basis, the baseline cPLA cutleries offer marginal savings (up to 10 %) for climate change (without DLUC and ILUC effects), freshwater eutrophication and abiotic depletion (fossil fuels) compared to PS cutlery. The relatively higher impacts of PLA cutlery are partly attributed to the heavier weight of the PLA cutlery (13.6 kg/FU) compared to the PS cutlery (10.1 kg/FU). It should be stressed here that the impact of PLA is highly sensitive to the EoL choices. If all PLA cutlery is disposed of in industrial composting, the GHG emission savings could reach up to 70 % compared to the PS cutlery.

For the remaining five impact categories the PLA cutlery performs worse than the PS counterparts. Most of the impacts in these categories are associated with biomass production and energy consumption during lactic acid and cutlery production, e.g. water use from the cultivation phase, marine and terrestrial eutrophication are caused by fertiliser use in the biomass cultivation phase, particulate matter is mainly contributed by particulate matter (<2.5 um) and sulphur dioxide emissions from cutlery and lactic acid production.

A large part of the photochemical ozone depletion is caused by the transportation services, especially the distribution of cutlery to Europe by transoceanic shipping. Even if the transportation modes and distances are assumed to be the same for PLA and PS cutlery; PLA has a significantly higher impact (40 % higher compared to PS) due to the heavier weight of PLA.

It should also be noted that due to lack of data, the production of PS in China is approximated by the PlasticsEurope's Eco-profiles of PS, which represent the average Western European PS production. This approximation may underestimate the impact of PS cutlery in comparison because the Chinese energy infrastructure is much more carbon intensive than the Western European one. We consider this as a drawback of the case study, even though we consider that the major part of the impact of PS is captured in the assessment. This approach has been taken I order to maintain the consistency of using PlasticsEurope's Eco-profiles as the benchmark for comparison, as for all other case studies.

Littering is excluded from this case study due to lack of data and knowledge of the impact assessment models. The exact amount (and relative share) of plastics that are littered is uncertain for a specific application and will differ widely between countries. The impacts of littering and microplastics are currently not properly captured in the toxicity-related impact categories (ecotoxicity and human toxicity). As pointed out by Pant et al. (2018) fate, exposure, and effect modelling for macro- and micro-plastics (including additives) is still in its infancy. Physical impacts (e.g. entanglement; ingestion of larger plastic particles and their effects) and chemical impacts (e.g. due to microplastic formation, microplastics as carrier of other chemicals, unknown impact of additives) and biological impacts (e.g. microplastics as carrier of germs/alien species) are largely unknown and currently not included in LCA. Thus, the impacts of littering (e.g. leaving plastics in the soil or in a marine environment) are most likely underestimated in the current LCA for the non-degradable applications.

In conclusion, there is no clear winner between PLA and PS cutlery since the products were compared only for eight categories. There is a huge potential for impact savings if the EoL of PLA cutlery is appropriately managed.

CASE STUDY 5: AGRICULTURAL MULCHING FILMS

1 Goal and scope definition

1.1 Goal and background

The goal of this case study is to assess the environmental profile of bio-based mulch film production and compare it with the fossil-based counterpart. Traditionally, mulch films are made with low-density polyethylene which is not biodegradable. Fossil-based mulch films need to be removed and disposed at the end of crop cycle which is labour-intensive. The collected plastic is heavily contaminated with soil, plant waste etc. making recycling difficult. It is also of reduced quality due to tearing (Niaounakis, 2015). Typically, this film is landfilled or incinerated. Occasionally, the fossil-based films are not properly collected and disposed of, causing environmental concerns (Razza and Cerutti, 2017). A possible solution to this specific issue is to use bio-based biodegradable mulch films which do not need to be removed and disposed of. They achieve the same functionality as the traditional mulch films (Razza and Cerutti, 2017).

Bio-based mulch films are produced from starch plastics. Starch plastics represent 10.3 % of the global production capacity of bio-based plastics (Aeschelmann et al., 2017). Europe has a solid position in the production and commercialization of starch blends with several companies being active in this sector, such as Novamont, Rodenburg, Biotec, FKuR and Kingfa (Aeschelmann et al., 2017). Application of starch plastics include flexible packaging, carrier bags, disposable tableware, agriculture and horticulture (mulch films, clips, pots) and loose-fill packaging foams. Carrier bags make up the majority of the consumption, whereas agricultural mulch films have a very small market share (Kaeb et al., 2016). The demand for agricultural plastics at global scale is expected to increase 10 % per year until 2020 (Santagata et al., 2017). Stricter regulations for disposal of petroleum-based mulch films to avoid littering is increasing the demand for biodegradable films (Santagata et al., 2017). Therefore, biodegradable mulch films have a high future market potential since they can decompose completely in the soil.

The specific characteristics and properties for a mulch-film to be called biodegradable are defined by the standard EN 17033. The biodegradable mulching films need to be tested to show biodegradability based on this standard. Similarly, fossil based mulching films that are intended to be removed after use should be consistent with the standard EN 13655 that specifies the requirements related to dimensional, mechanical, optical and thermal characteristics of thermoplastic films for mulching applications in agriculture and horticulture.

Starch plastics are blends of starch with one or more polymers (petrochemical or bio-based polyesters). Plasticisers such as glycerine can be used to destructure native starch into thermoplastic starch for improved mechanical properties (Broeren et al., 2017a). Depending on the industrial application and required technical properties, thermoplastic starch is blended (reactively extruded) with complexing agents of petreochemical or bio-based polyesters such as polylactic acid (PLA), polybutyrate adipate-co-terephthalate (PBAT) and polyhydroxyalkanoates (PHAs), to produce polyester-complexed starch biopolymers (Broeren et al., 2017a). To achieve good adhesion between the polymers, compatibiliser additives are used. By blending starch with biodegradable polyesters, 100 % biodegradable mulch films can be obtained.

1.2 Scope

As explained in the Chapter "Approach and methodology", it is defined that the product (mulch film) is purchased, used and disposed of in Europe. The supply chains in the manufacturing phases could go outside Europe prior to purchase (e.g. feedstock cultivation and harvesting, conversion processes).

The bio-based and biodegradable mulch films studied are made from maize and potato cultivated in Europe. PLA used as copolyester is produced in the US from maize that is cultivated in the US. Other copolyesters and additives are sourced from Europe. Production of the starch blend and film extrusion into mulch films also take place in Europe.

The temporal scope is current production (2017-2018) with relevant developments foreseen for the near future (5-10 years). The technical scope is a range of commercially available technologies.

1.3 Function and functional unit

The function is to provide field mulching. Mulching is a worldwide agricultural practice where soil is covered to increase crop yield, conserve moisture, control weeds, reduce the use of pesticides and shorten time to harvest (Razza and Cerutti, 2017). The crop is planted by cutting holes in the film (Niaounakis, 2015).

The functional unit of this case is defined as:

- Providing field mulching for one hectare of land (i.e. average 6.000 m^2 of mulch film) in Europe for four to five months.

This functionality and time are suitable for fruit and vegetable crops such as melon, courgette, lettuce and sweet pepper. The petrochemical counterpart of bio-based and biodegradable mulch film is plastic mulch film made of low-density polyethylene (LDPE) which is not biodegradable. The main characteristics of the two mulch film products are given in Table 157. This data reflects the current commercial products in Europe obtained with personal communication (Novamont, 2018). Biodegradable mulch films are thinner and have higher density.

Table 157. Characteristics of the analysed bio-based and petrochemical mulch films.

Characteristics	Bio-based mulch film	LDPE mulch film
Thickness [µm]	12	35
Density [kg/m^3]	1,270	900
Weight per FU (kg)	91.4	189

Accordingly, to fulfil the defined functional unit (6000 m^2 of mulch film), 91.4 kg of bio-based mulch film and 189 kg of LDPE mulch film is required.

1.4 Product systems

The bio-based product system is polyester-complexed starch biopolymer mulch film from European crops (maize or potato). In the starch plastics industry, very often companies produce starch blends based on their own technologies. The environmental impacts are analysed based on each technology; here, company-specific data are used. Aggregated data are then constructed by making a weighted average based on the market share of companies. The market share of starch-based plastics from maize is two thirds, whereas the remaining one third of the market is composed of potato starch-based plastics.

The reference petrochemical system is LDPE mulch film produced in Europe.

1.5 System boundaries

A cradle-to-grave approach is taken including the life cycle stages of feedstock production, manufacturing and EoL. The consumer use phase is excluded from the analysis which is the same for both product systems and has negligible impact. Thus, the life cycle can be divided into cradle-to-gate and EoL.

2 Life cycle inventory analysis

2.1 Process description, data, assumptions and multifunctionality

2.1.1 Baseline: Bio-Based mulch films from maize/potato

For this process, data is obtained from the largest industrial starch-based bioplastic producers, one producer based on maize starch and another based on potato starch. The biodegradable and bio-based polymer grades used for mulch film application have the following properties. They are biodegradable thermoplastic materials made with maize/potato starch and biodegradable copolyesters based on proprietary technology. The copolyester can be bio-based, fossil based or a mix of both. For the potato-based starch plastic, the copolymesters consist of both bio-based (PLA) and fossil-based (PBAT); the potatoes are cultivated in Germany (Broeren et al. 2017a). For the maize-based starch plastic, the biomass is cultivated in Italy and the fossil-based copolyester (proprietary technology, detailed composition is confidential) is used. It is converted to starch and supplied to the granulate production site. Copolyester and other raw materials are produced in Europe from the bio-based and fossil feedstocks and supplied to the plastic producer. The production of starch plastics consists of blending and extruding the native maize/potato starch, copolyesters and additives such as compatibilisers and plasticisers. Possible plastic scraps from the manufacturing process are directly recycled in the extrusion process. The outcome of the blend extrusion is the thermoplastic granules (plastic pellets), which are ready to be processed into the final product (mulch films) via blown film extrusion. Both starch granules production (i.e. converting the native starch into plastics granules) and film blown extrusion (i.e. converting granules into films) take place in Europe.

The process flow diagram of the bio-based mulch film product system is shown in Figure 136 for maize-based mulch film and in Figure 137 for potato-based mulch film.

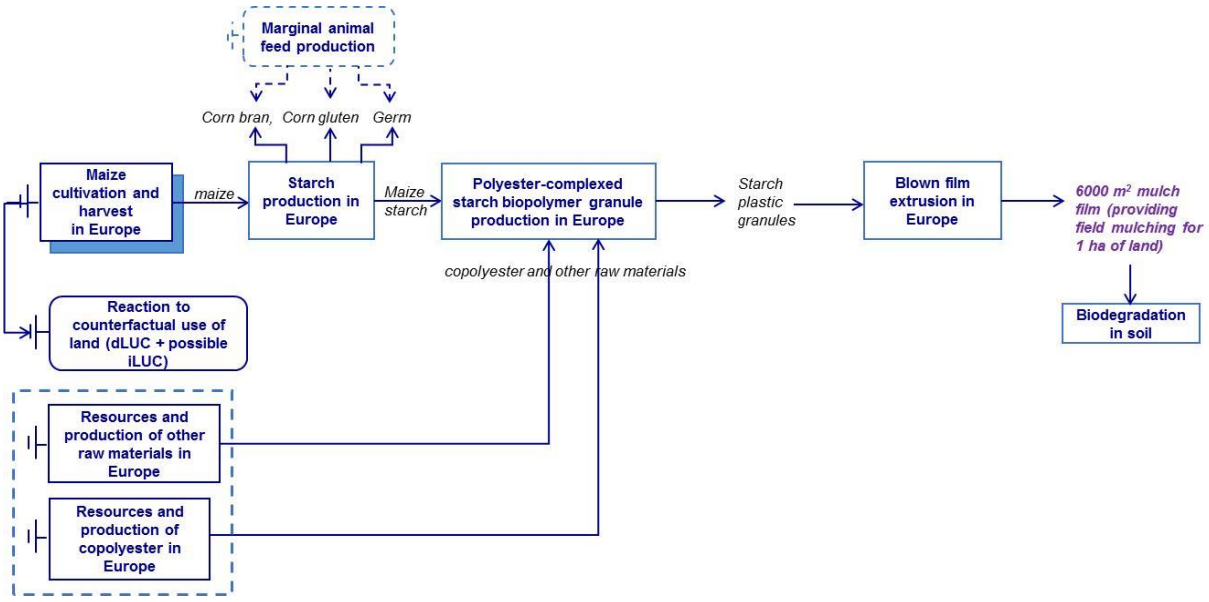


Figure 136. Process flow diagram of bio-based mulch film from maize product system.

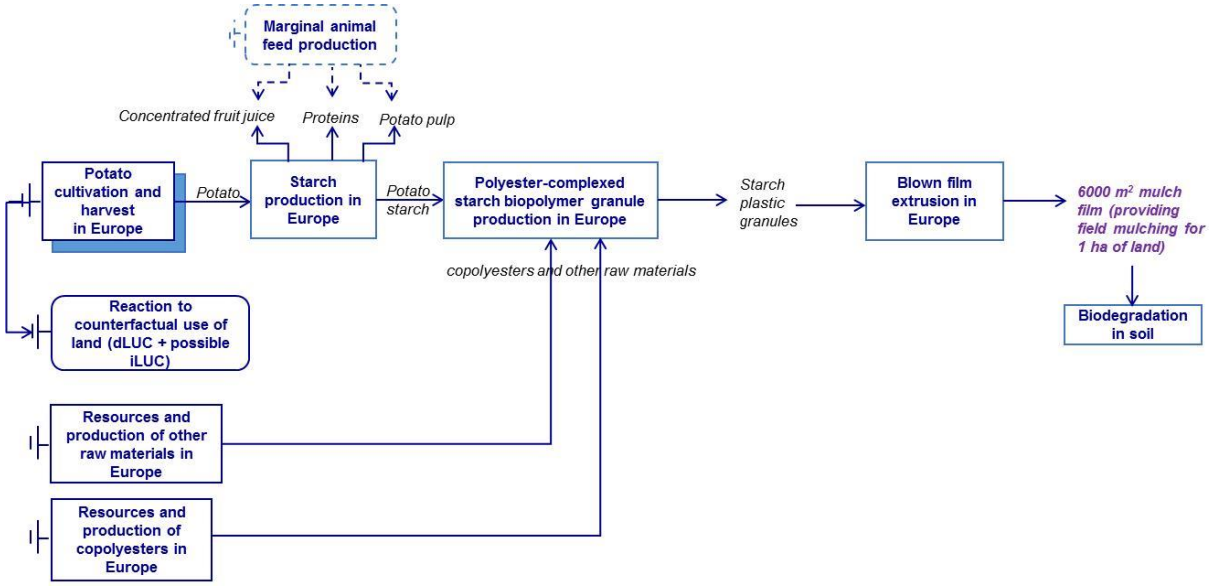


Figure 137. Process flow diagram of bio-based mulch film from potato product system

2.1.1.1 Bio-Based mulch film from maize

The maize-based starch plastic producer recommended to use Ecoinvent 3.3 data *Polyester-complexed starch biopolymer {RER}| production | Alloc Rec, U* to calculate the environmental profile of their mulch film grade biodegradable plastic (Novamont, 2018). To show the breakdown into key unit processes, specific information was attained through personal communication with the producing company.

The processes involved can be divided into the following key unit processes:

2.1.1.1.1 Maize cultivation and harvest

The starch-based bioplastic producer provided specific inventory data, in terms of maize yield, use of fertilisers, emissions etc. from their supplier sourcing maize from Italy. Drying of the maize grain for further processing is included in the inventory data. From the elemental composition of the polyester-complexed starch biopolymer provided by the company (Novamont, 2018), biogenic carbon content is used to calculate the biogenic carbon embedded in the bio-based mulch film (0.41 kg CO_2/kg starch-based polymer).

2.1.1.1.2 Maize starch production

The producing company also provided foreground data for maize starch production representative of its supply. The data reflects maize starch produced in Italy in a wet milling process. The starch is extracted and then refined. Maize bran, maize gluten and germ are co-products of the starch extraction process. These are used as animal feeds.

2.1.1.1.3 Maize starch transportation

The maize starch is transported to the granulate production site by lorry. The starch-based bioplastic producer provided the specific transportation distance between their production site and the supplier.

2.1.1.1.4 Copolyester and other natural and renewable raw materials (i.e. additives such as compatibilisers and plasticisers) production

The environmental burdens of the copolyester have been approximated with naphtha, natural gas and coal inputs provided within the Ecoinvent 3.3 Polyester-complexed starch biopolymer {RER}| production | Alloc Rec, U dataset. The specific information regarding the type of the copolyester is not provided. The breakdown between copolyester and the other natural and renewable raw materials is not available, but were informed that the majority of the impacts derive from the copolyester, with other raw materials contributing marginally. Naphtha and natural gas inputs are used both for material production and as fuel for heat production. Coal is used for additional heat production.

2.1.1.1.5 Copolyester and other raw materials transportation

The copolyester and the other raw materials produced are transported to the granulate production site by lorry. The producing company provided the specific transportation distance between their production site and the supplier.

2.1.1.1.6 Polyester-complexed starch biopolymer granulate production

Starch is combined with copolyester and other natural and renewable raw materials to get a thermoplastic biodegradable bio-based material (i.e. granulate) based on proprietary technology. The impacts of granule production are those associated to the electricity consumption and waste treatments within Ecoinvent 3.3 *Polyester-complexed starch biopolymer {RER}| production | Alloc Rec, U* dataset. The "gate to gate" electricity consumption in this process was corrected from 0.555 kWh/kg to 0.402 kWh/kg based on a publicly available Environmental Product Declaration (Novamont, 2004). The water consumption and wastewater treatment were not provided in this dataset, but they were requested and attained from the producing company and included in the inventory.

2.1.1.1.7 Granulate transportation

The starch-based biodegradable bioplastic granulate produced is transported to the converter site by lorry. The company provided the average approximate transportation distance between their production site and the mulch film producers.

2.1.1.1.8 Blown film extrusion

Mulch films are produced through blown film extrusion of the starch-based granulates. The key activity level data is electricity consumption. The electricity consumption is 0.4 kWh/kg (Razza et al., 2010).

The data sources used for the bio-based mulch film from maize are summarised in Table 161. For background data, e.g. grid electricity and heat, other utilities and production of chemicals and materials, Ecoinvent 3.3 is used.

Table 158. Data used in the bio-based mulch films from maize.

Processes	Data sources	Comments
Maize cultivation and harvest	Novamont, 2018	Represents production of maize in Italy attained from the supplier of maize-starch based bioplastic producer.
Maize starch production	Novamont, 2018	Represents production of maize starch of the supplier of maize-starch based bioplastic producer.
Copolyester (fossil-based) and other natural and renewable raw materials production	Ecoinvent 3.3	Naphtha, natural gas and coal inputs provided within Ecoinvent 3.3 *Polyester-complexed starch biopolymer {RER}| production | Alloc Rec, U* dataset
Transportation (of maize starch, copolyester and other raw materials)	Novamont, 2018	Represents distances from the suppliers to the granulate production site. All transportation is within Europe by lorry.
Maize-starch based granulate production	Ecoinvent 3.3 & Novamont, 2004	Electricity consumption and waste treatments within *Ecoinvent 3.3 Polyester-complexed starch biopolymer {RER}| production | Alloc Rec, U* dataset. Electricity consumption corrected from 0.555 kWh/kg to 0.402 kWh/kg granulate based on Novamont, 2004.
Granulate trasportation	Novamont, 2018	Represents average distance from granulate production site to extrusion site. Transportation is within Europe by lorry.
Blown film extrusion	Razza et al., 2010	Key activity level data is electricity consumption: 0.4 kWh/kg extruded plastic.

2.1.1.1.9 Multifunctionality

The maize starch production process involves co-production of maize bran, maize gluten and germ. To avoid allocation, the system is expanded to include their animal feed replacement. The co-products are displaced with a mix of marginal feed ingredients with the same standardised feed unit. The marginal protein feed is taken as soybean meal, the marginal carbohydrate is taken as maize and the marginal oil is taken as palm oil based on Tonini et al. (2016). These choices are based on detailed elaboration of recent demand trends and future projections (FAPRI, 2012; FAOSTAT, 2014).

2.1.1.2 Bio-Based mulch film from potato

For the potato-based starch plastic the processes involved can be divided into the following key unit processes:

2.1.1.2.1 Potato cultivation and harvest

Potatoes are sourced mainly from Germany. The Agrifootprint database is used for the inventory data of potato production in Germany. From the composition of the polyester-complexed starch biopolymer provided by the plastic producer, biogenic carbon content is used to calculate the biogenic carbon embedded in the bio-based mulch films (1.09 kg CO_2 eq./kg potato starch-based polymer).

2.1.1.2.2 Potato starch production

After harvesting potatoes are cleaned and grinded into pulp. The starch slurry and juice are then separated. The juice is further processed to produce proteins and concentrated fruit juice. Starch slurry is refined and dried to produce native starch to be sent to granulate production. Concentrated fruit juice, proteins and pulp are attained as co-products of starch production process. The Agrifootprint database is used for the inventory data reflecting a typical potato wet milling process in Europe. Drying of the starch to the required moisture level is included in the inventory.

2.1.1.2.3 Potato starch transportation

The potato starch is transported to the granulate production site by lorry.

2.1.1.2.4 Copolyesters production

Bio-based PLA from maize and petrochemical polybutylene adipate-co-terephthalate (PBAT) are used as copolyesters. For PLA data is obtained from the PLA producer located in the US. Please refer to case study 3 for further information. For PBAT data was not available in databases. It was modelled using stoichiometric amounts of 1,4-butanediol, adipic acid and purified terephthalic acid (PTA). For polymerization of the monomers to PBAT it was taken to be equal to PET polymerization. The data from case study 1 on beverage bottles was used for PTA and polymerization. The data for 1,4-butanediol and adipic acid was attained from the Ecoinvent 3.3 database.

2.1.1.2.5 Copolyesters transportation

The copolyesters are transported to the granulate production site by a combination of lorry, bulk carrier and freight train.

2.1.1.2.6 Other raw materials (i.e. additives such as compatibilisers and plasticisers) production

Other renewable and non-renewable raw materials (additives) are produced in Europe. Data from the Ecoinvent 3.3 database is used for them.

2.1.1.2.7 Other raw materials transportation

Other raw materials used in granulate production are transported by lorry to the granulate production site.

2.1.1.2.8 Polyester-complexed starch biopolymer granulate production

Starch is combined with copolyesters and other raw materials to get a thermoplastic biodegradable bio-based material (i.e. granulate) based on proprietary technology. The key activity level data are the electricity consumption and waste treatments. Foreground data is based on Broeren et al. (2017a).

2.1.1.2.9 Granulate transportation

The starch-based biodegradable bioplastic granulate produced is transported to the converter site by lorry.

2.1.1.2.10 Blown film extrusion

Mulch films are produced through blown film extrusion of the starch-based granulates. The same data for the maize-based production is used (section 2.1.1.1).

The data sources used for the bio-based mulch film from potato are summarised in Table 159. For granulate transportation and blown film extrusion processes, refer to Table 161. For

background data, e.g. grid electricity and heat, other utilities and production of chemicals and materials, Ecoinvent 3.3 is used.

Table 159. Data used in the bio-based mulch films from potato.

Processes	Data sources	Comments
Potato cultivation and harvest	Agri-footprint	Represents potato cultivation in Germany. Crop yields derived mainly from 2010-2014 FAO statistics (FAOSTAT, 2014)
Potato starch production	Agri-footprint	Represents a typical potato wet milling process in Europe, reference year of 2014
Copolyesters production	For PLA: Vink et al., 2015 For PBAT: Broeren et al., 2017a	PLA based on data from industrial producer, see case study 3 for details with updated background data using Gabi database version 2018 (Personal communication with E. Vink). PBAT based on modelling done by Broeren et al., 2017a
Other raw materials (i.e. compatibilsers and plasticisers) production	Broeren et al., 2017a & Ecoinvent 3.3	Represents average European processes, reference year 2016
Transportation (of potato starch, copolyesters and other raw materials)	Broeren et al., 2017a	Represents average distances from the suppliers to the granulate production site. Within Europe transportation by lorry, from U.S. transportation by combination of lorry, bulk carrier and freight train
Potato-starch based granulate production	Broeren et al., 2017a	Represents average compounding extruder, key activity level data is electricity consumption: 1.9 MJ/kg granulate

2.1.1.2.11 Multifunctionality

The potato-starch production process involves co-production of potato pulp, proteins and concentrated fruit juice. To avoid allocation, the system is expanded to include their animal feed replacement. The co-products are displaced with a mix of marginal feed ingredients with the same standardised feed unit. The marginal protein feed is taken as soybean meal, the marginal carbohydrate is taken as maize and the marginal oil is taken as palm oil based on Tonini et al. (2016). These choices are based on detailed elaboration of recent demand trends and future projections (FAPRI, 2012; FAOSTAT, 2014).

2.1.2 The reference system: petrochemical LDPE

The reference petrochemical system is LDPE mulch film. PlasticsEurope released in 2016 the latest eco-profile for petrochemical LDPE. This data is taken as the reference since it provides the most up to date eco-profile based on European plastic producers. However, as explained in Chapter 6 (Ranges for environmental impacts from production of fossil-based plastics), PlasticsEurope data for petrochemical plastics are not transparent and do not comply with ILCD requirements. Differences between the available data from different data sources are found to be large and some impact categories were found not suitable (Chapter 6 Ranges for environmental impacts from production of fossil-based plastics). The data is provided in Table 37 for the suitable impact categories for 1 kg LDPE granulate production. For the transportation of the granulates and blown film extrusion into mulch films, the same data is used as for the bio-based system.

2.2 End of Life description, data and assumptions

The mulching films have different possible EoL options depending on the product system, as observed in Table 160.

The petrochemical LDPE mulching film has the three following EoL options:

- **Mechanical plastic recycling**: the process includes the energy and material requirements for transportation to the facility, sorting, cleaning and recycling processes. The recycled LDPE is assumed to substitute virgin LDPE production. The rejects from the recycling process are sent to incineration (both with and without energy recovery) and includes transportation. The flowchart for plastic recycling is presented in Figure 8.

- **MSW incineration with and without energy recovery**: a generic MSW incineration plant is assumed, which represents average EU waste incineration with and without energy recovery with an overall heat efficiency of 22 % and electrical efficiency of 9 % (CEWEP, 2013; Eurostat, 2017a). The process includes the direct emissions from waste incineration as well as indirect emissions from the production of the input materials, the combustion of fossil fuels as well as treatment of bottom ash and fly ash. The flowchart for MSW incineration is presented in Figure 9.

- **MSW landfilling**: the process includes the construction and operation of an average landfill site, direct emissions of landfill gas, landfill gas collection, flaring and recovery system, treatment of collected leachate and direct emissions from uncollected leachate. 29 % of the collected landfill gas is used to generate electricity (OpenLCA Nexus, 2015). The electricity conversion efficiency is 37 % (Christensen, 2011). The flowchart for MSW landfilling is presented in Figure 10.

The starch mulching film has the following options;

- **MSW Incineration with and without energy recovery**: a generic MSW incineration plant is assumed, which represents the average EU waste incineration with and without energy recovery with an overall heat efficiency of 22 % and electrical efficiency of 9 % (CEWEP, 2013; Eurostat, 2017a). The process includes the direct emissions from waste incineration as well as indirect emissions from the production of the input materials, the combustion of fossil fuels as well as treatment of bottom ash and fly ash. The flowchart for MSW incineration is presented in Figure 9.

- **MSW landfilling**: the process includes the construction and operation of an average landfill site, direct emissions of landfill gas, landfill gas collection, flaring and recovery systems, treatment of collected leachate and direct emissions from uncollected leachate. 29 % of the collected landfill gas is used to generate electricity (OpenLCA Nexus, 2015). The electricity conversion efficiency is 37 % (Christensen, 2011). The flowchart for MSW landfilling is presented in Figure 10.

- **In-situ field-based biodegradation**: this includes avoided collection and direct emissions to air and soil. The flowchart for the in-situ field-based biodegradation is presented in Figure 11.

A major uncertainty in this case is the fact that Plastics Recyclers Europe estimate the current collection rate at around 60 % for mulching film (PRE, 2017). This means that a large share of the petrochemical mulching film is prone to be left in the field. This LCA looks at a thick LDPE film, which is possible to collect for either recycling, incineration or landfilling. Thinner LDPE

mulching film are likely to tear with sunlight and weather effects, making it difficult to gather for proper treatment. This also refers to the new European plastic strategy on LDPE mulch films (European Commission, 2018).

The recycling of mulch film is possible and in some European countries, e.g. Germany and Ireland, there is a significant amount being recycled due to proper collection system for the film being in place (PRE, 2017).

It should also be noted that the likelihood of the biodegradable starch mulching film being incinerated or landfilled is questionable, due to difficulty gathering the film from the field after utilisation, as the film would have biodegraded to a great extent.

Apart from modelling full (100 %) recycling, incineration, landfilling and in-situ individually, a mix of these EoL scenarios was also modelled to represent the current situation in Europe. The European mix of EoL technologies have been estimated based on several references (PRE, 2017; IFFPG, 2018; RIGK, 2018; EuPC, 2018) which state a recycling rate of more than 70 % for agricultural films in Ireland, 5 400 tonnes of agricultural films being recycled in Germany and a production of LDPE films in Europe of 86 000 tons. Including the fact that the likelihood of recycling is higher in e.g. green houses compared to the functional unit of this study, the rate of collection for recycling of mulch film is set to 5 %. Consultic (2014) report a recycling rate of 28 % for agricultural plastic in general. The incineration and landfilling of the remaining mulch film is based on general plastic waste in Europe (European Commission, 2018) the division between these EoL possibilities is set at 42 % landfilling and 53 % incineration.

For the bio-based mulch film it is assumed that 100 % will be in-situ field biodegraded.

The "EoL mix of EU" is used to model the product systems to create a most likely "status-quo" of waste management for a given product.

Table 160. The EoL options, their intended EoL option and the estimated EoL mix.

Product system	EoL options modelled individually (100 % scenarios)	Intended EoL option	Estimated EoL collection mix of EU (based on statistics of waste treatments of waste plastics)
Bio-based starch plastic	Incineration Landfilling In-situ	In-situ	In-situ: 100 %
Petrochemical LDPE	Recycling Incineration Landfilling		Recycling: 5 % Incineration: 53 % Landfilling: 42 %

The average European EoL technologies are described in the approach and methodology chapter in the section regarding EoL. In that section, factors that do not depend on the specific product system are described, e.g. the energy efficiency of the incineration plant and the consumption of energy and materials in each technology. Furthermore, principles for substitution are described. This involves that the energy produced in the EoL technologies is substituted with marginal electricity and heat, as well as the recycling substitution methodology.

In addition to the description here, the contamination (soil) and additional soil being removed, when the agricultural products are gathering for treatment, depletion of the organic carbon from soil is modelled when the mulch film is recycled, incinerated or landfilled.

Several product system dependent factors are presented in Table 161, for each possible EoL technology. The recycled LDPE mulch film plastic is substituted with petrochemical PE (PlasticsEurope, 2013). The recycling rate is calculated by multiplying the collection rate and the sorting and technology efficiency.

Table 161. Values for product system dependent factors.

EoL Technology	Product system dependent factor	Unit	Bio-based starch	Reference	Petrochemical LDPE	Reference
Plastics recycling	Sorting and technology efficiency (amount collected for recycling which is recycled)	% of input			37	Briassoulis et., al, 2013.
MSWI with and without energy recovery	Energy content	MJ/kg	23.6	Ecoinvent 3.4	41.45	Götze et al., 2016
Landfilling	1st order decay rate for methane generation	1/s	0.0075	Calculated based on data from Ecoinvent 3.4	0	Estimated
In-situ biodegradation in field	C degradation (%)	% of C bio	100	Ecoinvent 3.4		
Energy content is in lower heating value (LHV)						

The mulching films have the following material compositions, see Table 162, which is utilised as the input to the EoL LCA model. The full chemical composition is presented in Annex 1.

Table 162. Material composition of mulching film (1 kg plastic + 100 g contamination).

Chemical component	Unit	Bio-based starch	Petrochemical LDPE
Water	% of total	3.2	5.9
TS (VS+ash)	% of total	91.3	88.7
VS	% of TS	91.4	90.2
C fossil	% of TS	39.5	76.3
C biogenic	% of TS	17.7	0.3
Reference		Ecoinvent 3.4	Götze et al., 2016
TS: total solids, VS: volatile solids.			

If the mulching film is gathered from the field for treatment, a large portion of soil will be attached to the mulching film and thereby follow the plastic to recycling, incineration or landfilling. The removal of soil that occurs when gathering the mulching has been accounted for in the EoL treatment. The amount of extra soil is 3 kg per kg of plastic mulch film and based on Plasticulture (2018), which has reported that mulch film has a contamination that is 3 to 4 times the weight of the mulch film. Hence the EoL-reference flow for modelling the EoL for mulch film is:

- 1 kg plastic + 100 g contamination + 3 kg soil

The contamination consists of soil – see further details in Approach and methodology chapter, section 3.3.2 Contamination composition.

3 Life cycle impact assessment results and interpretation

3.1 Cradle-to-gate environmental impacts and interpretation, Bio-Based system(s)

The aggregated cradle-to-gate LCA results from the assessment of one functional unit of bio-based mulch film (91.4 kg) are presented in Table 163. This represents the aggregated results based on the market share of 2/3 for starch plastics from maize and 1/3 for starch plastics from potato as described in section 1.4.

Table 163. Cradle-to-factory gate LCA results of 1 functional unit bio-based mulch film (91.4 kg) (excluding DLUC and ILUC effects).

Impact category	Unit	Bio-based mulch film
Climate change	kg CO_2 eq.	1.52E+02
Ozone depletion	kg CFC-11eq	5.55E-05
Human toxicity, non-cancer effects	CTUh	4.22E-05
Human toxicity, cancer effects	CTUh	1.03E-05
Particulate matter	kg PM2.5eq	1.02E-01
Ionizing radiation, human health	kBq U235eq	3.38E+01
Photochemical ozone formation	kg NMVOCeq	6.76E-01
Acidification	molc H+eq	1.25E+00
Terrestrial eutrophication	molc Neq	2.35E+00
Freshwater eutrophication	kg Peq	6.11E-02
Marine eutrophication	kg Neq	4.44E-01
Freshwater ecotoxicity	CTUe	1.04E+03
Land use	kg C deficit	8.24E+02
Water use	m3	1.57E+02
Abiotic depletion	kg Sbeq	2.28E-04
Abiotic depletion (fossil fuels)	MJ	4.64E+03
NREU	MJ	5.14E+03

In Figure 138, the impacts of bio-based mulch film are broken down into the main unit processes described in section 2.1 inventory analysis. For the maize-based polymer it is not possible to show the impacts of copolyesters and other raw materials separately. Therefore, they are combined into one category for the aggregated results with the potato-based polymer.

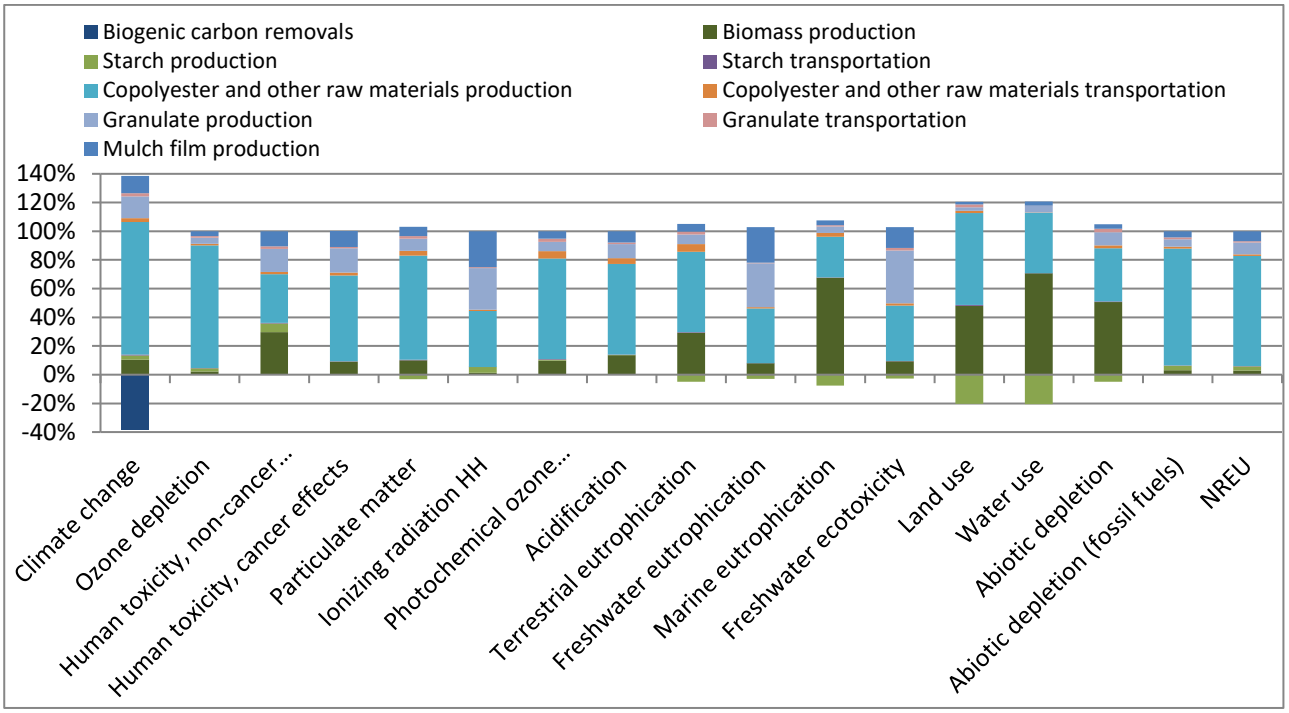


Figure 138. Breakdown of the cradle-to-gate environmental impact results for bio-based mulch films, excluding DLUC and ILUC effects.

Biomass (maize/potato) production contributes significantly to terrestrial eutrophication (29 %), marine eutrophication (68 %), human toxicity (non-cancer effects) (30 %), land use (48 %), water use (71 %) and abiotic depletion (51 %). The high contribution to water use is mainly due to irrigation. The contribution to marine and terrestrial eutrophication comes from direct nitrous emissions caused by the application of fertilisers. The pesticides production process causes the contribution of maize production to abiotic depletion. For other impact categories, contribution from maize production is up to 14 %.

Starch production does not have a significant contribution to any category except for land use and water use, for which it provides a 20 % saving. Due to the avoided animal feed production from the starch by-products, savings are also seen in the eutrophication and abiotic depletion categories.

The environmental impact of bio-based mulch films is dominated by copolyester and other raw material production. Their contribution ranges from 34 % to 86 % in all impact categories except for marine eutrophication (28 %). This is caused dominantly by the petrochemical copolyesters and raw materials and their impacts arise from the naphtha, natural gas and coal used in the production. For the bio-based copolyester of PLA, the detailed breakdown into unit processes can be seen in case study 3. This bio-based polyester shows major contribution in land use category (44 % of the total impacts from copolyester and other raw material production).

The contribution of transportation requirements (starch and copolyester transportation to granulate production, granulate transportation to mulch film production) to the environmental impact of bio-based mulch film are minor (maximum 8 % contribution to impact categories).

Granulate production and mulch-film production processes have a similar contribution to the impact categories. Their contribution is mostly in the range of 3 to 30 %. The key activity level data in both processes is electricity consumption.

The results for bio-based mulch film are further elaborated below. Following the PEFCR guidance the most relevant impact categories are identified as (in descending order starting with most relevant): abiotic depletion (fossil fuels), climate change, particulate matter, acidification, ionizing radiation, freshwater eutrophication and photochemical ozone formation. They cumulatively contribute to 83 % of the total environmental impact (excluding toxicity related impact categories) based on the normalised and weighted results (see section 3.4.1). Detailed analysis is made to trace the origin of the impact in the major contributing processes for these impact categories and for impact categories where biomass production has significant impact.

The most relevant impact categories are interpreted as the following:

- **Abiotic depletion (fossil fuels).** The majority (82 %) of the contribution to abiotic depletion (fossil fuels) comes from copolyester and other raw material production (i.e. additives). 61 % of the aggregated impacts are caused by the maize starch-based plastic which uses petrochemical copolyester and the impacts arise from naphtha (46 % of the 61 %) and natural gas (39 % of the 61 %). 39 % of the aggregated impacts are caused by the potato starch-based plastic where more than one third (37 %) of this 39 % comes from PBAT production (of which 41 % caused by adipic acid, 32 % is caused by 1,4 butanediol production), 30 % from PLA production and 33 % from other raw materials.

- **Climate change.** The aggregate biogenic carbon embedded in the product is calculated as 0.6 kg CO_2/kg bio-based starch plastic. When the biogenic carbon removal is considered, the cradle to factory gate climate change impact of bio-based mulch film is 1.66 kg CO_2 eq./kg or 152 kg CO_2 eq./FU. Again, it is emphasized that this is a preliminary conclusion of GHG accounting without considering the effects of DLUC and ILUC. The copolyester production accounts for 92 % of the climate change impact (Figure 138). About 40 % of the impacts come from the petrochemical copolyester used in the maize starch-based polymer (where heat produced from coal contributes 62 % of this impact). The bio-based PLA and petrochemical PBAT each contribute 24 % to the total impact. For PBAT, 56 % of the impacts come from adipic acid production and 31 % from 1,4 butanediol. The remaining contributions of unit processes to bio-based mulch film, almost equally come from the biomass production, granulate production and mulch film production processes (about 10-15 % each). For biomass production, 38 % of the impact is based on the diesel use in machinery, 26 % from direct field emissions from fertilisers and urea decomposition and 10 % from ammonium nitrate chemical fertiliser use. The climate change of granulate and mulch film production processes are caused by electricity consumption. The biogenic carbon stored in the film provides a 39 % saving in climate change impacts.

- **Particulate matter.** For particulate matter, 73 % of the contribution comes from copolyester and other raw material production. 60 % of the aggregated impacts are caused by the maize starch-based plastic which uses petrochemical copolyester and the impacts arise from coal used for heat (68 %) and naphtha used (20 %). 40 % of the aggregated impacts are caused by the potato starch-based plastic where 70 % of the impact comes from PBAT production (40 % caused by adipic acid, 44 % caused by 1,4 butanediol production), 15 % from PLA production and 15 % from other raw materials.

- **Ionizing radiation.** Ionizing radiation is caused mainly by copolyester and other raw material production (39%). This is followed by the production of the granulate contributing to the total impact with a share of 29% and the production of the mulching film (25%).

- **Freshwater eutrophication.** The main life cycle stages that cause the eutrophication of freshwater are: copolyester and other raw material production (38%), the production of granulates (30%) and the mulching film production (25%).

- **Photochemical ozone formation.** Copolyester and other raw material production is responsible for 70% of the total photochemical ozone formation. Another 10% of impact comes from the production of the biomass used as feedstock.

The interpretation of several other impact categories which are important for the biomass production phase of the investigated bio-based material are detailed here.

- **Marine eutrophication.** For marine eutrophication, 68 % of the contribution comes from biomass production. The majority (88 %) of this impact originates from direct nitrate emissions to water from application of fertilisers and from crop residues.

- **Land use.** For land use, 48 % of the contribution comes from land occupation for biomass (maize/potato) production for starch. 64 % of the contribution comes from copolyester production. 44 % of the impact is due to bio-based PLA production and the land use of the maize for that. The rest originates from the land use for onshore drilling for naphtha for the petrochemical copolyesters. Due to the avoided animal feed production by the starch by-products, a 20 % reduction in land use is seen in the starch production process.

- **Water use.** Biomass production contributes to 60 % of the gross water use impact[65], of which mainly (97 %) due to the irrigation requirements of maize. Due to the avoided animal feed production by the starch by-products, 18 % of the gross water use impacts can be avoided. 35 % of the gross impacts come from copolyester and other raw material production, of which the potato-based starch plastics dominate the impact (91 %). The copolyesters in the potato-based starch plastics play the most important role: 58 % of the 91 % is from PBAT, 27 % from PLA and 15 % from other raw materials such as additives.

- **Abiotic depletion.** Biomass production contributes 51 % of abiotic depletion. The impacts mainly (>70 %) are based on the production of pesticides. Copolyester and other raw materials production contributes to 37 %. The majority (86 %) of the impacts are caused by the potato starch-based plastic where 59 % of the impact comes from PBAT production (55 % caused by adipic acid, 40 % caused by 1,4 butanediol production), 19 % from PLA production and 22 % from other raw materials.

3.2 Cradle-to-gate environmental impacts of the petrochemical reference system

The cradle-to-factory gate LCA results for one functional unit of petrochemical LDPE mulch film (189 kg/FU) is presented in Table 164. This is calculated by multiplying the impact of plastic granulates production (given in Table 37 for 1 kg LDPE granulate) with 189 kg to convert per FU. Impacts of granulate transportation and and blown film extrusion to mulch films are added to this.

[65] "Gross water use impact" refers to the water use impact without the credits received from avoided impacts from by-products of starch production.

Based on Chapter 6 *Ranges for environmental impacts from production of fossil-based plastics*, 8 out of 17 impact categories were found suitable for LDPE, namely: climate change, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication, marine eutrophication, abiotic depletion (fossil fuels) and NREU (see Chapter 6 *Investigation of generic LCA data for fossil-based plastics*). This is because comparison of different datasets revealed large differences for the other impact categories. Further details can be found in Table 37 where the environmental impact of 1 kg of LDPE and the proposed ranges of variation within each impact category are shown. The data used is based on PlasticsEurope since it provides the most up to date eco-profile for European fossil-based plastics. However, it does not conform with ILCD requirements in its presentation of inventory details or impact assessment results. Furthermore, it does not distinguish between emissions to fresh water and to seawater, and it is therefore not possible to calculate the impacts on human health and ecosystems alongside freshwater eutrophication, because the receiving compartment is an important element in determining the fate and effect of an emission.

Table 164. Cradle-to-gate LCA results of petrochemical LDPE mulch film for 1 FU (=189 kg).

Impact category	Unit	Petrochemical LDPE mulch films
Climate change	kg CO_2 eq.	4.05E+02
Particulate matter	kg PM2.5 eq	7.88E-02
Photochemical ozone formation	kg NMVOC eq	1.65E+00
Acidification	molc H+ eq	1.21E+00
Terrestrial eutrophication	molc N eq	2.88E+00
Marine eutrophication	kg N eq	2.74E-01
Abiotic depletion (fossil fuels)	MJ	1.50E+04
Non-renewable energy use (NREU)	MJ	1.59E+04

In Figure 139, the impacts of petrochemical LDPE mulch films are broken down into the main unit processes of granulate production, transportation and mulch film production. It is seen that production of LDPE granulate shows the highest contribution in all (selected as suitable for comparison) impact categories. Its contribution ranges from 78 to 96 % for all categories. The ecoprofile for the LDPE granulate comprises extraction and refining of crude oil and natural gas, steam cracking of hydrocarbons into lower olefins, and polymerisation of the monomers into polyethylene. However, it is not possible to show the breakdown of the impact of granulate production into these unit processes in the lack of transparancy of the PlasticsEurope data. The transportation is within Europe and shows negligible contribution to impacts. Blown film extrusion process shows a small (up to 17 %) contribution. The main activity level data of this process is electricity consumption.

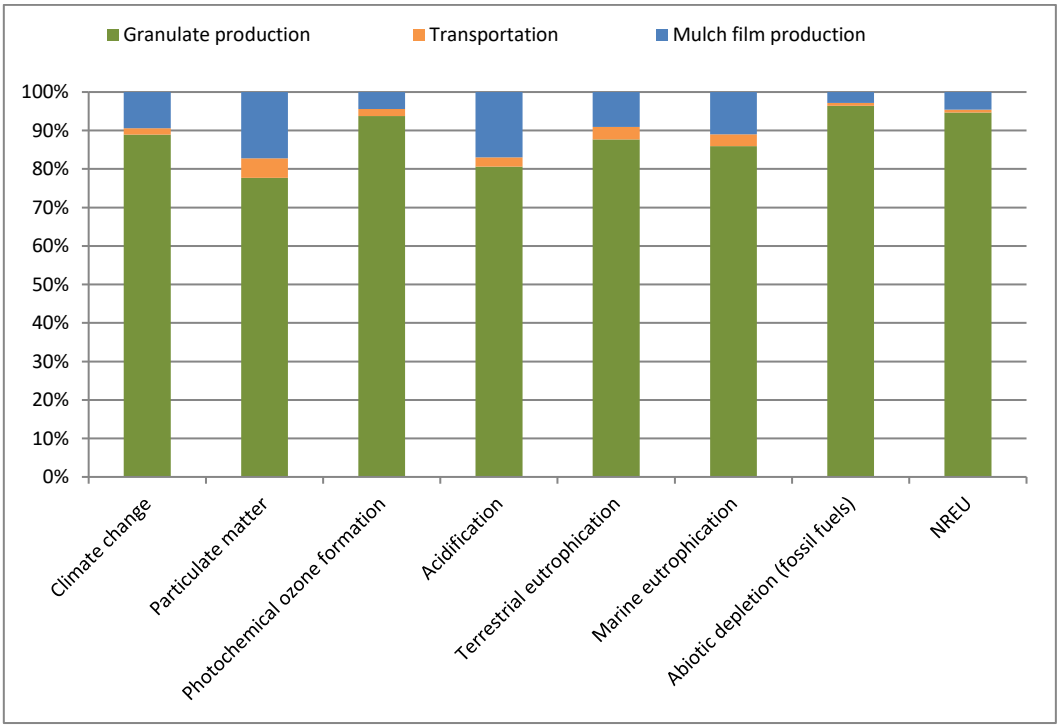


Figure 139. Breakdown of the cradle-to-gate LCA results for petrochemical LDPE mulch films.

3.3 End of life results and interpretation

The results for the EoL impact assessment modelled in EASETECH are presented in this paragraph. Firstly, the mass and energy flows of all product systems in all technologies are presented, the secondly the LCIA results for each product system, thirdly a comparison between product systems, and lastly the EoL mix. In Annex 4 further results are presented, contribution analysis is in Figure 275 ff. and weighted results are in Table 239 ff.

3.3.1 Mass and energy flows

In Table 165 and Table 166 the mass and energy flows of EoL waste management, as outputs from EASTECH, for the two product systems are presented. These mass flows correspond to the mulch film without the additional soil. These results correspond to the circular footprint formula in PEFCR guidance draft version 6.3 (European Commission, 2017), and in addition the paragraph looks at the carbon flows. The flows correspond to the flows observed in the flow charts for each technology in the approach and methodology chapter of the full report, section 3.4 Average End of Life technologies. Note that these flows do not contain the 3 kg of additional soil modelled as EoL-reference flow.

It is observed that 37 % of the plastic is recycled and 30 % substituted in the recycling technology for the petrochemical LDPE (calculated based on the sorting and technology efficiency as well as the market response, considering the decrease in quality of the plastic - see the methodology for further explanation). Landfilling the mulch film results in 87 % of the mass being stored for the bio-based starch and 100 % of the petrochemical LDPE. In the in-situ scenario, 90 % of the mass is stored in soil.

Different EoL scenarios result in varying energy production (recovery of the energy content of the waste input). In the incineration plant, 31 % of the energy content in the product is recovered in the incineration for the starch plastic and the petrochemical LDPE (the energy efficiency of the incineration plant). When landfilling 2 % of the energy content is recovered

from the starch, while none of the energy is recovered from the petrochemical LDPE. Incineration of residues from the LDPE recycling process recovers 19 % of the energy content of the waste input. The in-situ field application of starch and the landfilling of LDPE recovers none of the energy.

Looking at the carbon balance when landfilling mulching film, 56 % of the biogenic carbon and 84 % of the fossil carbon (from contamination) is stored for the starch product system, while 100 % of the carbon is stored in the landfill for the LDPE. When leaving the starch mulching film on the field, 10 % of the biogenic carbon is stored in the ground, while the rest results in air emissions (CO_2 and to a very small extent CH_4).

Table 165. Material and energy flow for EoL of 1 kg starch with 100 g contamination. Box letter refer to flow charts in the approach and methodology chapter. Box letter refer to flow charts in the approach and methodology chapter (Figure 8 to Figure 13). Excludes the 3 kg of additional soil.

Starch							
Technology	**Box**	**Process**	**Mass (kg)**	**Bio carbon (kg)**	**Fossil carbon (kg)**	**Gas (m^3 CH4)**	**Energy substitution (MJ)**
Material	I	Material input	1.10E+00	1.77E-01	4.67E-01		2.34E+01
Incineration w/wo energy recovery *	E, H	Energy production	1.10E+00	1.77E-01	4.67E-01		7.21E+00
	R1	Fly Ash	1.86E-02	0.00E+00	0.00E+00		
	R2	Bottom ash	5.40E-02	1.77E-04	3.90E-04		
	D	Direct emissions		1.77E-01	4.66E-01		
Landfill	L	Leachate	1.91E+00	0.00E+00	0.00E+00		
	G, E	Landfill gas		7.76E-02		1.07E-01	5.18E-01
	CS	Storage in landfill	9.53E-01	9.94E-02	3.90E-01		
In-situ	C1	Air emissions		1.60E-01			
	C2	Storage in soil	9.40E-01	1.75E-02			

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

Table 166. Material and energy flow for EoL of 1 kg petrochemical LDPE with 100 g contamination. Box letter refer to flow charts in the approach and methodology chapter. Box letter refer to flow charts in the approach and methodology chapter (Figure 8 to Figure 13). Excludes the 3 kg of additional soil.

LDPE						
Technology	**Box**	**Process**	**Mass kg**	**Fossil carbon kg**	**Gas m^3 CH4**	**Energy substitution MJ**
Material	I	Material input	1.10E+00	8.04E-01		3.98E+01
Recycling	O	Recycled plastic	4.07E-01	9.69E-04		
	S	Substituted plastic	3.30E-01	7.85E-04		
	R, E, H	Rejects	6.93E-01	8.03E-01		7.74E+00
Incineration w/wo energy	E, H	Energy production	1.10E+00	8.04E-01		1.23E+01

recovery *	R1	Fly Ash	2.85E-03	0.00E+00		
	R2	Bottom ash	9.17E-03	6.81E-04		
	D	Direct emissions		8.03E-01		
Landfill	L	Leachate	2.20E+00	0.00E+00		
	G, E	Landfill gas			0.00E+00	0.00E+00
	CS	Storage in landfill	1.10E+00	8.04E-01		

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

3.2.2 LCIA for product systems

The impact assessment results are presented per the EoL-reference flow, i.e. 1 kg plastic and 100 g of contamination and 3 kg of soil, for each of the EoL technologies (assuming 100 % disposal via that technology) in Table 167 for both for the starch and LDPE product systems.

Table 167. Total results of treating 1 kg bio-based starch and 1 kg petrochemical LDPE (incl. 100 g contamination and 3 kg of soil) for all impact categories. The intended EoL option is marked with a bold box.

Impact category (Unit)	Bio-based starch			Petrochemical LDPE		
	Incineration	Landfilling	In-situ	Recycling	Incineration	Landfilling
Climate change - fossil (kg CO_2 eq)	2.44E+00	4.83E-02	-6.76E-02	3.15E+00	3.68E+00	5.66E-02
Climate change - biogenic (kg CO_2 eq)	1.11E+00	1.88E+00	5.85E-01	4.67E-01	4.29E-01	2.63E-01
Climate change - total (kg CO_2 eq)	3.55E+00	1.93E+00	5.18E-01	3.61E+00	4.11E+00	3.20E-01
Ozone depletion (Kg CFC-11 eq)	-3.16E-08	6.70E-07	-1.44E-12	-2.60E-08	-5.77E-08	1.71E-09
Human toxicity, non-cancer effects (CTUh)	3.31E-07	1.20E-08	3.09E-08	5.01E-07	3.61E-07	1.45E-08
Human toxicity, cancer effects (CTUh)	2.02E-07	8.71E-11	6.89E-09	2.13E-07	2.22E-07	2.77E-10
Particulate matter (kg PM2.5 eq)	1.18E-04	-2.39E-06	1.38E-06	9.20E-05	5.82E-05	1.11E-05
Ionizing radiation HH (kBq U235 eq)	5.28E-02	5.58E-04	-5.68E-06	7.44E-02	8.71E-02	2.51E-03
Photochemical ozone formation (kg NMVOC eq)	9.35E-03	6.37E-04	2.10E-04	8.98E-03	9.27E-03	1.80E-04
Acidification (molc H^+ eq)	9.68E-03	1.80E-04	1.52E-04	9.52E-03	1.00E-02	2.33E-04
Terrestrial eutrophication (molc N eq)	4.40E-02	6.59E-04	8.96E-04	4.33E-02	4.44E-02	7.39E-04
Freshwater eutrophication (kg P eq)	6.88E-04	1.62E-07	1.39E-07	6.95E-04	7.03E-04	1.99E-06
Marine eutrophication (kg N eq)	3.70E-03	2.14E-04	8.21E-05	3.85E-03	3.74E-03	2.26E-04
Freshwater ecotoxicity (CTUe)	3.48E+00	2.86E-02	9.40E-02	4.00E+00	4.45E+00	3.50E-02
Land use (kg C deficit)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use (m^3)	8.80E-01	4.48E-02	-3.07E-03	2.26E+00	7.38E-01	8.49E-02
Abiotic depletion (kg Sb eq)	-1.75E-07	-1.52E-08	-1.25E-10	-8.33E-08	-3.53E-07	9.62E-09
Abiotic depletion (fossil fuels) (MJ)	1.14E+01	5.75E-01	-5.37E-02	6.90E+00	6.38E+00	8.80E-01
NREU (MJ)	1.01E+01	5.21E-01	-5.35E-02	3.77E+00	4.50E+00	7.36E-01

The highest impact within each category is red and the lowest impact green. The significance of the colour scale is the same as for all the product systems. This means that you can compare the colour across, but only comparing between the same impact category.

3.3.3 Baseline system results – starch mulching film

The overall conclusion for ranking the EoL options of starch mulching film is that both in-situ field-based biodegradation and landfilling perform well, but most often the in-situ performs better. The incineration of starch mulching film performs worst.

Further details on the individual EoL options for the starch mulch film are presented below, looking into specific impact categories and contribution of impacts from different processes and elemental emissions.

3.3.3.1 Intended EoL technology: In-situ biodegradation of starch mulching film

The LCIA results from the starch mulching film show that the In-situ EoL technology has the lowest environmental impacts compared with incineration and landfilling. The in-situ model covers the avoidance of collecting the waste, which contributes to savings in all impact categories and the emissions from left-on-field application of the starch. There is no removal of soil, i.e. no biological carbon removal, in this technology, further increasing the value of this technology. The in-situ performs best in nine impact categories out of 16, second best in two and worst in one impact category, abiotic depletion.

Looking at the contribution analysis for the in-situ EoL presented in Annex 4, there are only two processes that contribute to emissions, avoided collection and use-on-land. The contribution analysis shows that the use-on-land emissions clearly have a larger contribution in all impact categories than the avoided collection. The avoided collection has a contribution of 17 %-25 % in the impact categories of NREU, abiotic depletion, abiotic depletion (fossil), water use, ozone depletion and climate change fossil.

Further details on the individual EoL options for the starch plastic mulching film are presented below, looking into specific impact categories and contribution of impacts from different processes and elemental emissions.

3.3.3.2 Landfill of starch mulching film

As the starch is biodegradable and generates landfill gas over time, the potential impacts from landfilling starch compared to LDPE are higher. The construction and operation of the landfill causes the largest impact on water use and ionising radiation. The landfilling shows the best performance comparing the impacts from bio-based starch EoL in five out of 16 impact categories, second best in 10 and worst in one impact category (ozone depletion). For landfilling of the bio-based starch mulching film the largest contribution in the impact categories of climate change, ozone depletion and photochemical ozone formation come from landfill gas that is collected and combusted, or not collected but oxidised under cover. The leachate treatment has the largest contribution to marine eutrophication due to ammonium, nitrate and nitrogen oxide emissions to water and air. The electricity substitution from the collected methane has the largest contribution in 11 out of 18 impact categories.

3.3.3.3 Incineration of starch mulching film

For the incineration, the direct emissions from incinerating the starch mulching film have the largest contribution to climate change, terrestrial eutrophication, marine eutrophication and photochemical ozone formation. The heat substitution has the largest contribution in the form of savings in the impact categories NREU, abiotic depletion, freshwater eutrophication and ozone depletion but the largest contribution in the form of impact on ionising radiation. Electricity substitution saves most in the impact categories of water use, abiotic depletion, and

particulate matter. The treatment of an additional 3 kg of soil in the incineration plant contributes to a high impact in several categories (e.g. climate change, freshwater ecotoxicity, water use). In total the incineration has the highest impact in 14 impact categories.

3.3.3.4 Weighted end of life results for baseline – starch mulching film

The weighted EoL results are shown in Figure 140, and the numerical results can be found in Annex 4. For methodology and choice of normalisation and weighting factors, please see the Approach and methodology chapter, section 5.2.3 "Use normalised and weighted results to determine the preference of EoL option" and the values in Table 30. Normalisation factors (NF) and weighting factors used by this study.

The weighted results show, both with and without toxicity, that the in-situ technology is the best, secondly landfilling and thirdly incineration. The savings from the energy generation in incineration and landfilling are almost zero in the weighted results. Overall it is the climate change impact that has the largest contribution in the weighted results. However, for the incineration human toxicity (cancer effects) has the largest contribution (38 %) to the weighted results. For landfilling and in-situ biodegradation, the climate change impact contributes to 89 % and 66 % respectively of the total weighted impact with toxicity. The incineration shows weighted results with toxicity of 0.028, landfilling has 0.005 and the in-situ EoL 0.0018. It is clear from the weighted results that the intended EoL technology is the preferred option for agricultural mulching film. The in-situ technology is clearly the best suited EoL technology for the mulching film, and this technology requires little manpower, energy and creates low environmental impact compared with other EoL technologies. Due to the amount of soil that will enter the recycling and incineration facilities with the mulching film, these EoL technologies are not well suited for treating agricultural mulching film.

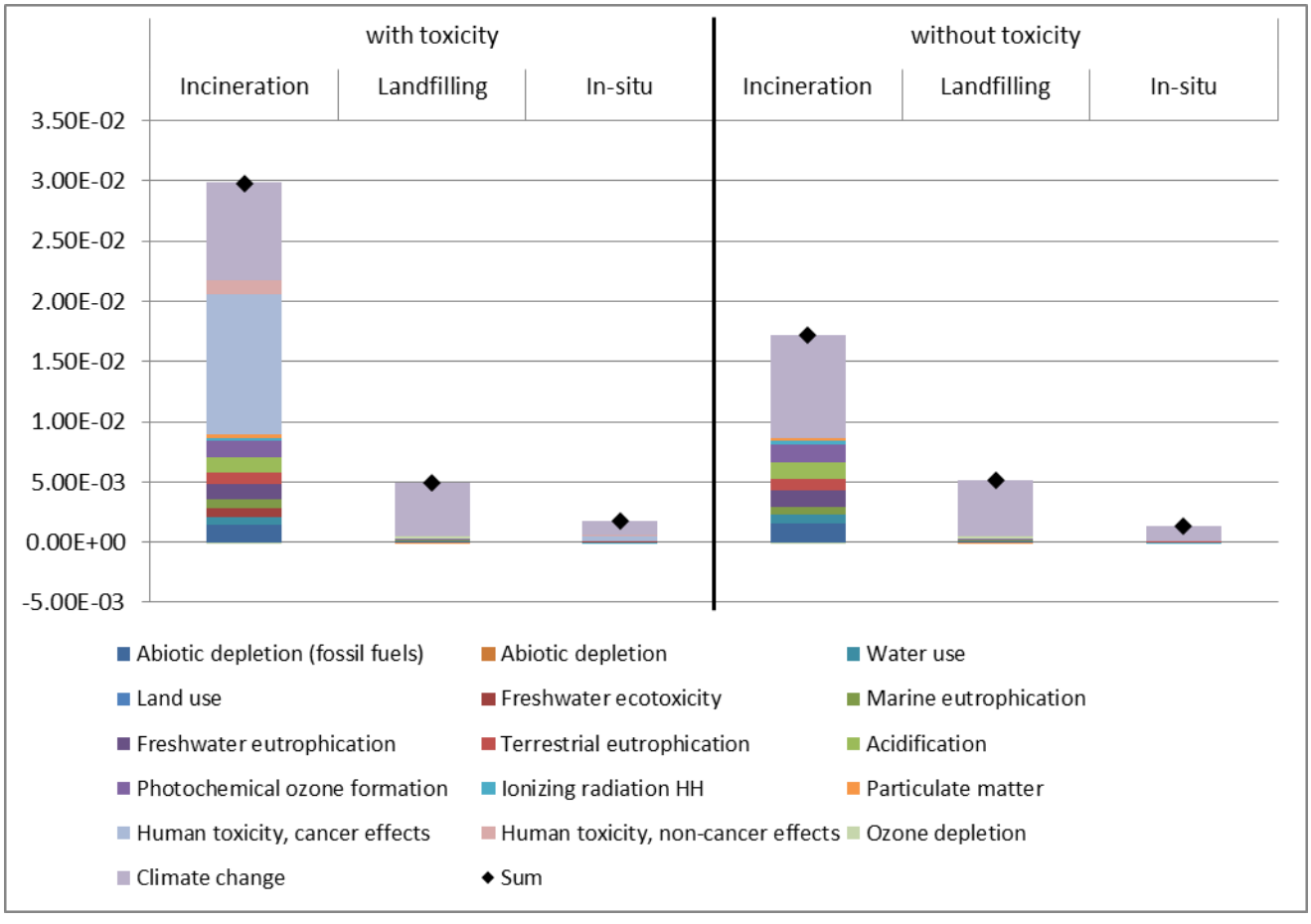


Figure 140. Weighted EoL results for the baseline product system, bio-based starch cutlery including food leftovers.

3.3.4 Reference system results – LDPE

The results of the reference system of LDPE for the mulching film was presented earlier in Table 167 and Table 168. Overall, landfilling has the lowest impact in most impact categories. Incineration performs well in a few impact categories, but the worst in many other impact categories. Recycling of the LDPE mulch film, which has a very low quality affects the environmental performance of the recycling.

Further details on the individual EoL options for the LDPE mulch film are presented below, looking into specific impact categories and the contribution of impacts from different processes and elemental emissions.

3.3.4.1 Landfilling of LDPE mulching film

The petrochemical LDPE mulching film does not degrade and therefore generates no landfill gas. Other impacts from landfill originate from removal of soil, construction and operation of the landfill, and leachate treatment. Landfilling performs the best in 14 out of 16 impact categories. The LCIA for the petrochemical LDPE film shows that the landfill has the lowest climate change impact of the three analysed technologies, with 0.32 kg CO_2-eq, presented in Figure 141. The contribution analysis of landfilling LDPE mulching film shows that the transport and the construction and operation of the landfills cause the largest contribution on the overall LCIA results. It is only in the impact category of marine eutrophication that the leachate treatment contributes the largest impact (64 %), due to emissions of ammonium and nitrate and nitrogen oxides in air and water. Overall, landfilling of petrochemical LDPE is the preferred option for most environmental impact categories. This is due to no degradation of the plastic in 100 years but excludes impact categories such as abiotic depletion and non-renewable energy use.

3.3.4.2 Recycling of LDPE mulching film

The technology efficiency at the recycling plant of LDPE is 37 % and therefore the remaining 63 % are rejected and enter a MSWI plant. The substitution ratio for the plastic substituted is only 33 % of each incoming 1 kg of plastic. Looking at the contribution analysis of the plastic recycling of petrochemical LDPE, there are several processes that have a large share of the contribution. The plastic substitution has the largest contribution in the impact categories of abiotic depletion and NREU, because a large amount of energy consumption is avoided when there is avoidance of virgin LDPE production. The treatment of rejects (e.g. direct and indirect emissions from the incineration plant and treatment of fly ash and bottom ash) has the largest contribution in the impact categories of climate change, human toxicity (cancer effects) and freshwater ecotoxicity. The electricity substitution from the incineration of rejects has the largest contribution in abiotic depletion, as substitution of electricity will result in savings in natural sources used in generating the marginal electricity in Europe. Moreover, the electricity substitution has the largest contribution in the impact category of particulate matter, originating from avoidance of particulate matter from power plants and other sources of electricity generation. Recycling the LDPE however occurs at a low rate due to low collection rate which is one of the major challenges with this scenario.

3.3.4.3 Incineration of LDPE mulching film

The direct emissions contribute to the largest part of the climate change impact from incineration. The treatment of the attached soil in the incineration plant contributes to the largest part of freshwater ecotoxicity. The incineration of petrochemical LDPE mulching film performs best in two impact categories, ozone depletion and abiotic depletion. Moreover, incineration performs second best in five impact categories and worst in 9. The contribution

analysis for the incineration of petrochemical LDPE mulching film shows that the substitution of heat and electricity has the largest contribution in most impact categories. The direct emissions from the incineration plant make the largest contribution to climate change, photochemical ozone formation, terrestrial and marine eutrophication. The treatment of bottom ash and fly ash has a large contribution in the impact categories of freshwater ecotoxicity and the human toxicity categories, mainly from emissions of arsenic and chromium. The substitution of electricity has a large contribution in the form of savings in the impact categories of water use, abiotic depletion and particulate matter.

3.3.5 Climate change details

The total climate change impact of the different EoL technologies and the product systems are presented in Figure 141.

It is evident that incineration is the worst performing technology regarding climate change for both the petrochemical and the bio-based mulching film. For the bio-based starch landfill generates 1.93 kg CO_2-eq., incineration 3.55 kg CO_2-eq. and in-situ 0.52 kg CO_2-eq. For the petrochemical LDPE mulching film, incineration has a climate change impact of 4.11 kg CO_2-eq, the recycling 3.61 kg CO_2-eq and the landfilling 0.32 kg CO_2-eq.

Comparing the climate change impact from incineration of the two product systems shows that the bio-based starch material has a lower impact than the petrochemical material in most impact categories.

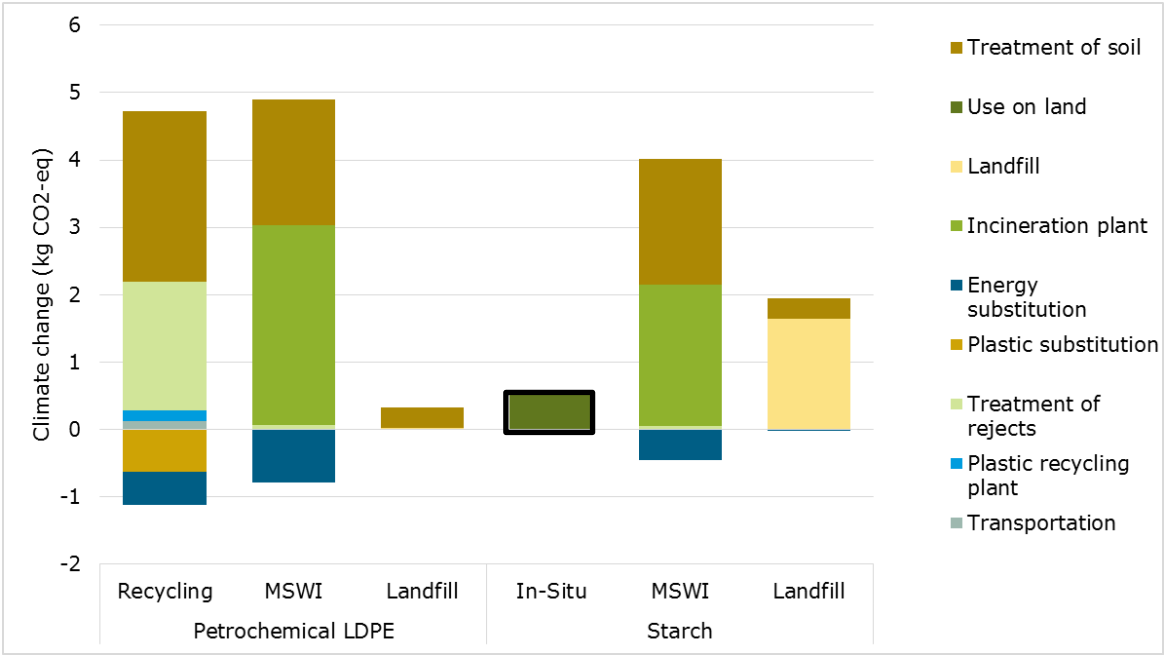


Figure 141. Climate change of treating 1 kg mulching film (incl. 100 g contamination and 3 kg of soil) in different EoL technologies. The intended EoL option is marked with a black box. See Annex 4 for details of the content of the grouped processes. The results without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

3.3.6 Comparison between the two product systems

The weighted results with toxicity for both product systems and all EoL technologies are presented in Figure 142 per functional unit (i.e. different from the above results which are for the EoL-reference flow of 1 kg plastic and 100 g contamination). From the figure it is noted that landfilling for LDPE mulching film and in-situ biodegradation of starch mulching film both have a low impact. The numerical values (available in Annex 4) show that landfilling of LDPE gives a slightly lower weighted score than the in-situ biodegradation of starch mulching film. For the bio-based starch mulching film, it is landfilling that is ranked second and incineration is the least preferable EoL technology. For the petrochemical LDPE mulching film, recycling and incineration have equally weighted results with toxicity. If one looks at incineration solely with energy recovery (i.e. excluding plants without energy recovery), this technology would likely perform better than the other EoL technologies.

The contribution to the impacts of incineration of the starch mulch film dominantly come from climate change and human toxicity (cancer effects), which originates respectively from direct emissions at the incineration plant and bottom ash treatment. Landfilling the bio-based mulch film contributes to climate change due to direct emissions of un-collected landfill gas and flaring of landfill gas. The in situ EoL option primarily contributes to climate change because of direct emissions.

The petrochemical LDPE mulch film makes a large contribution to climate change in all three EoL scenarios. For recycling and incineration, in addition to climate change, there is also a significant contribution to human toxicity (cancer effects). The climate change impacts are due to incineration of plastic (rejects), bottom ash treatment from landfilling of the soil and removal of soil. The human toxicity (cancer effects) impacts are due to bottom ash treatment of the soil. Landfilling the LDPE mulch film makes the largest contribution to climate change, which is due to removal of the soil.

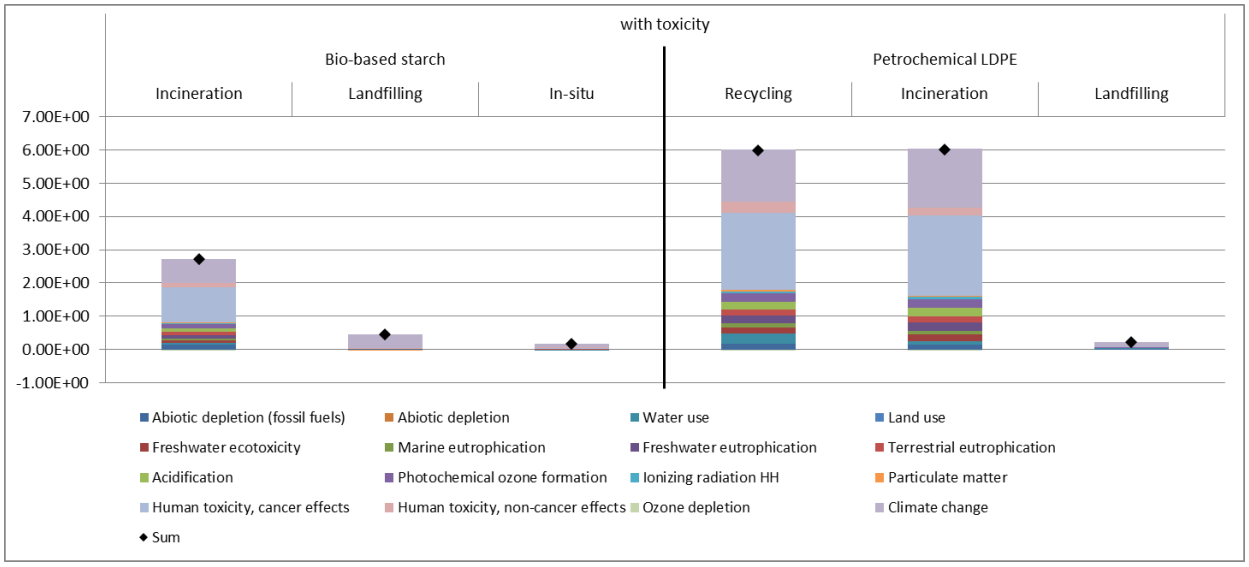


Figure 142. Weighted results with toxicity of the mulching film product systems with their EoL option per functional unit. Numerical values to be found in Annex 4.

3.3.7 End of life mix results

The LCIA for all product systems for the estimated EoL per functional unit are presented in Table 168. Comparing the two systems per FU, the bio-based starch product performs much better than the petrochemical LDPE product. There are only two impact categories were the petrochemical LDPE product has a lower impact. The bio-based product will be left on the field and therefore the additional soil that is attached to the mulch film is not removed. Treating the additional soil via recycling, incineration or landfill contributes to a large share of the impacts.

The two impact categories where the bio-based mulch film is less favourable, are caused by larger savings in ozone depletion and abiotic depletion when substituting electricity, heat or plastic material. The bio-based starch mulch film is half of the weight of the fossil LDPE mulch film, stressing the importance of comparison per functional unit.

Using a bio-based starch mulch film with an in-situ EoL can save 424 kg CO_2 compared to the mix of recycling, incineration and landfill of petrochemical LDPE mulch film. Moreover, 479 CTU of freshwater ecotoxicity and 103 m^3 of water can be saved by using bio-based starch mulch film with an in-situ EoL instead of petrochemical LDPE with the assumed EoL mix, just to mention a few examples. The intended EoL technology, in-situ, for the baseline starch system is therefore clearly the recommended EoL technology.

Table 168. LCIA for EoL mix of the mulch film for all product systems per functional unit (6.000 m^2 of mulch film).

Impact category	Unit	Bio-based starch	Petrocehmical LDPE
In-situ	%	100	
Recycling	%		5
Incineration	%		53
Landfill	%		42
Climate change - fossil	kg CO_2 eq / FU	-6.18E+00	4.03E+02
Climate change - biogenic	kg CO_2 eq / FU	5.35E+01	6.83E+01
Climate change - total	kg CO_2 eq / FU	4.73E+01	4.71E+02
Ozone depletion	kg CFC-11 eq / FU	-1.32E-10	-5.89E-06
Human toxicity, non-cancer effects	CTUh / FU	2.83E-06	4.21E-05
Human toxicity, cancer effects	CTUh / FU	6.30E-07	2.42E-05
Particulate matter	kg PM2.5 eq / FU	1.26E-04	7.59E-03
Ionizing radiation HH	kBq U235 eq / FU	-5.19E-04	9.62E+00
Photochemical ozone formation	kg NMVOC eq / FU	1.92E-02	1.03E+00
Acidification	molc H+ eq / FU	1.39E-02	1.12E+00
Terrestrial eutrophication	molc N eq / FU	8.19E-02	4.91E+00
Freshwater eutrophication	kg P eq / FU	1.27E-05	7.71E-02
Marine eutrophication	kg N eq / FU	7.50E-03	4.29E-01
Freshwater ecotoxicity	CTUe / FU	8.59E+00	4.87E+02
Land use	kg C deficit / FU	0.00E+00	0.00E+00
Water use	m^3 / FU	-2.81E-01	1.02E+02
Abiotic depletion	kg Sb eq / FU	-1.14E-08	-3.54E-05
Abiotic depletion (fossil fuels)	MJ / FU	-4.91E+00	7.74E+02
NREU	MJ / FU	-4.89E+00	5.45E+02

3.4 Cradle to grave results and interpretation

3.4.1 Bio-Based baseline system

The aggregated cradle-to-grave LCA results from the assessment of 1 functional unit of bio-based mulch film (91.4 kg) are given in Table 169. The breakdown of impacts between biomass production, manufacturing and EoL is shown in Figure 143. For EoL of bio-based mulch film 100 % in-situ biodegradation is the intended and the current EoL mix, and the values in Table 169 refer to that. The manufacturing phase (conversion of biomass to mulch films) dominates the impacts in most categories except for the categories of marine eutrophication, land use, water use and abiotic depletion. In these categories biomass production has a significant impact. Biomass production has a minor contribution for the other impact categories. The EoL (which is in-situ biodegradation in soil) has a negligible impact in

the overall cradle-to-grave results except for climate change, which is caused by the GHG emissions from the degradation of the product on field.

Table 169. Cradle-to-grave LCA results of 1 functional unit bio-based mulch film (excluding DLUC and ILUC effects).

Impact category	Unit	Biomass	Manufacturing	EoL	Total
Climate change	kg CO_2 eq.	-4.27E+01	1.95E+02	4.73E+01	1.99E+02
Ozone depletion	kg CFC-11eq	1.23E-06	5.43E-05	-1.32E-10	5.55E-05
Human toxicity, non-cancer effects	CTUh	1.25E-05	2.97E-05	2.83E-06	4.51E-05
Human toxicity, cancer effects	CTUh	9.36E-07	9.40E-06	6.30E-07	1.10E-05
Particulate matter	kg PM2.5eq	1.02E-02	9.18E-02	1.26E-04	1.02E-01
Ionizing radiation human health	kBq U235eq	4.15E-01	3.34E+01	-5.19E-04	3.38E+01
Photochemical ozone formation	kg NMVOCeq	6.62E-02	6.10E-01	1.92E-02	6.96E-01
Acidification	molc H^+eq	1.70E-01	1.08E+00	1.39E-02	1.27E+00
Terrestrial eutrophication	molc Neq	6.88E-01	1.66E+00	8.20E-02	2.43E+00
Freshwater eutrophication	kg Peq	4.88E-03	5.62E-02	1.27E-05	6.11E-02
Marine eutrophication	kg Neq	2.99E-01	1.44E-01	7.50E-03	4.51E-01
Freshwater ecotoxicity	CTUe	9.62E+01	9.43E+02	8.60E+00	1.05E+03
Land use	kg C deficit	3.96E+02	4.28E+02	0.00E+00	8.24E+02
Water use	m^3	1.11E+02	4.60E+01	-2.81E-01	1.56E+02
Abiotic depletion	kg Sbeq	1.15E-04	1.13E-04	-1.14E-08	2.28E-04
Abiotic depletion (fossil fuels)	MJ	1.41E+02	4.50E+03	-4.91E+00	4.63E+03
NREU	MJ	1.48E+02	4.99E+03	-4.89E+00	5.13E+03

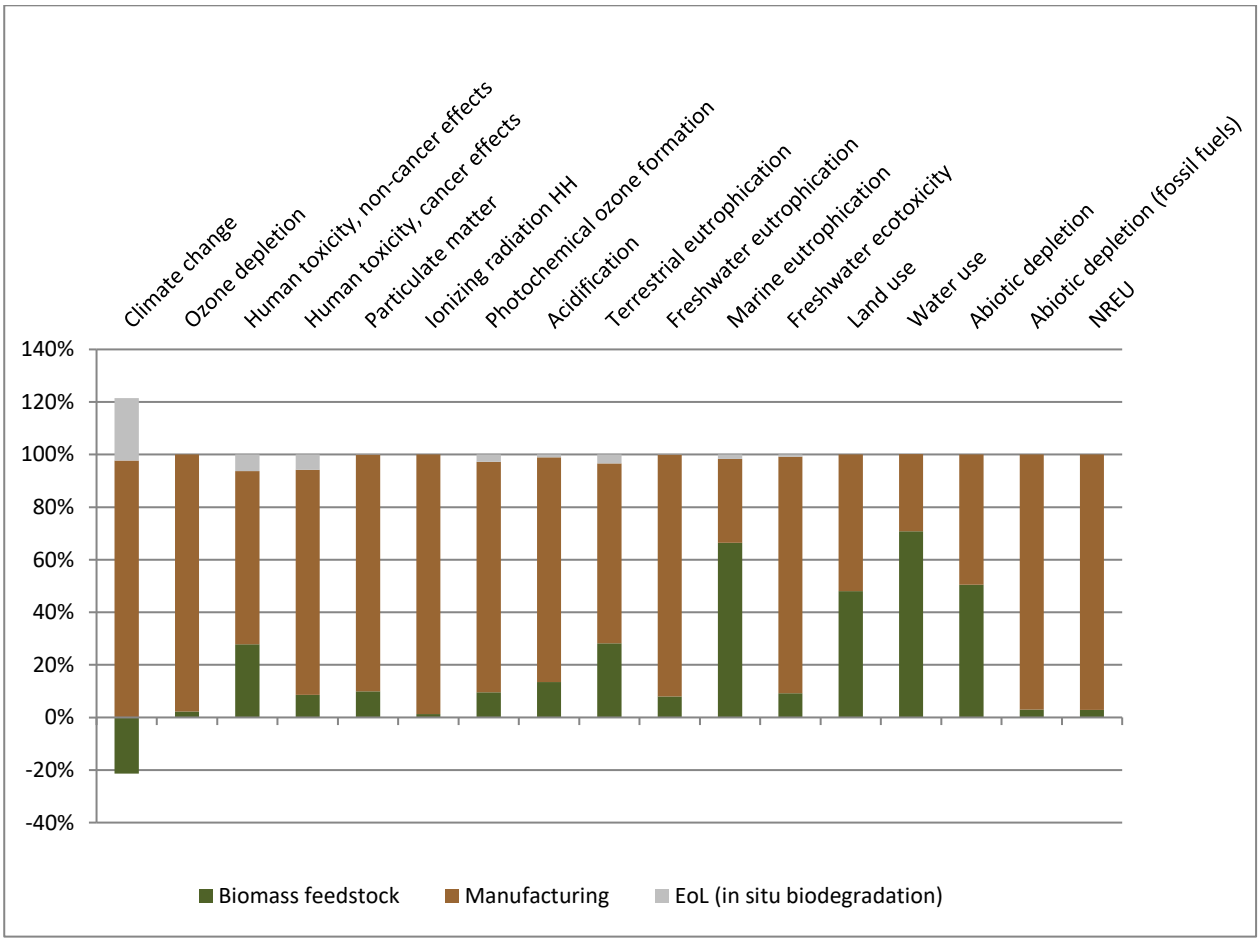
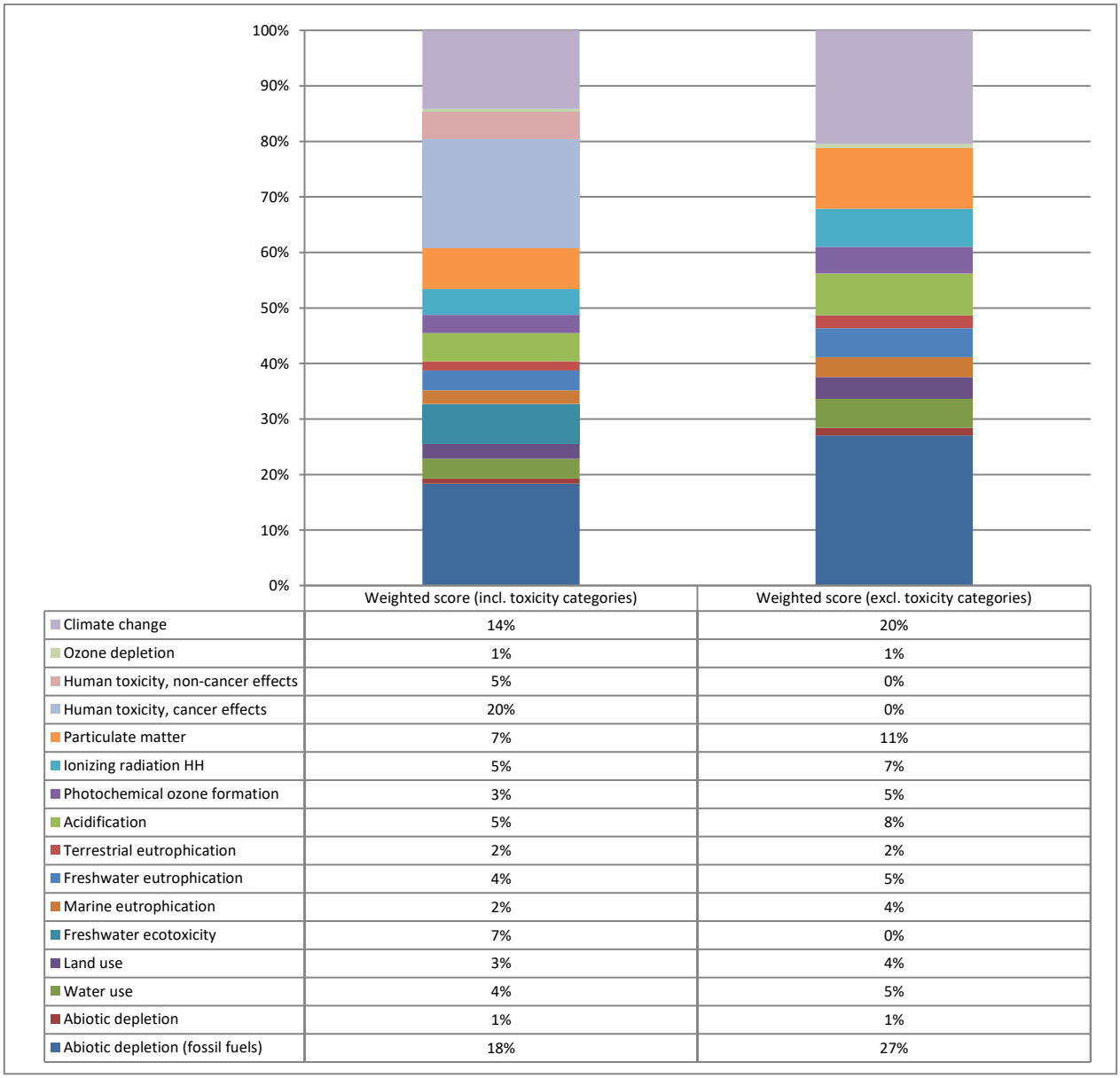


Figure 143. Breakdown of the potential impacts from the bio-based mulch films across all impact categories, cradle-to-grave results (excluding DLUC and ILUC effects).

According to PEFCR guidance v 6.3, the most relevant impact categories are identified as all impact categories that cumulatively contribute to at least 80 % of the total environmental impact (excluding toxicity related impact categories). Based on the normalised and weighted results shown in Figure 144, excluding toxicity related impacts, the most relevant impact categories are: abiotic depletion (fossil fuels) (27 %), climate change (20 %), particulate matter (11 %), acidification (8 %), ionizing radiation (7 %), freshwater eutrophication (5 %), and photochemical ozone formation (5 %) forming a cumulative 83 % contribution to total impacts. These impact categories are identified as the hot spots for this case study and detailed analysis is made for each of them in section 3.1.1. The rest of the impact categories contribute each up to 4 %.



	Weighted score (incl. toxicity categories)	Weighted score (excl. toxicity categories)
Climate change	14%	20%
Ozone depletion	1%	1%
Human toxicity, non-cancer effects	5%	0%
Human toxicity, cancer effects	20%	0%
Particulate matter	7%	11%
Ionizing radiation HH	5%	7%
Photochemical ozone formation	3%	5%
Acidification	5%	8%
Terrestrial eutrophication	2%	2%
Freshwater eutrophication	4%	5%
Marine eutrophication	2%	4%
Freshwater ecotoxicity	7%	0%
Land use	3%	4%
Water use	4%	5%
Abiotic depletion	1%	1%
Abiotic depletion (fossil fuels)	18%	27%

Figure 144. Contribution of the 16 PEF categories based on normalised and weighted results of bio-based mulch films, with and without toxicity categories

Contributions of the life cycle stages and the different processes within are shown in Figure 145. Please refer to section 3.1.1. for a more detailed elaboration per impact category of the contributing processes. As shown in Figure 145, the manufacturing phase (conversion of biomass to mulch films) is the most relevant life cycle stage cumulatively contributing to 85 % of the impacts. The copolyesters and other raw material production is the most significant process with about 60/65 % contribution followed by granulate production (13/10 %) and mulch film (blown film extrusion) (9/8 %) processes. For biomass production, the biogenic carbon sequestered results in low overall impact for this stage (11 %). The EoL is found to have a very low contribution (4 %) to overall impacts, with biodegradation in soil showing low impacts to the environment.

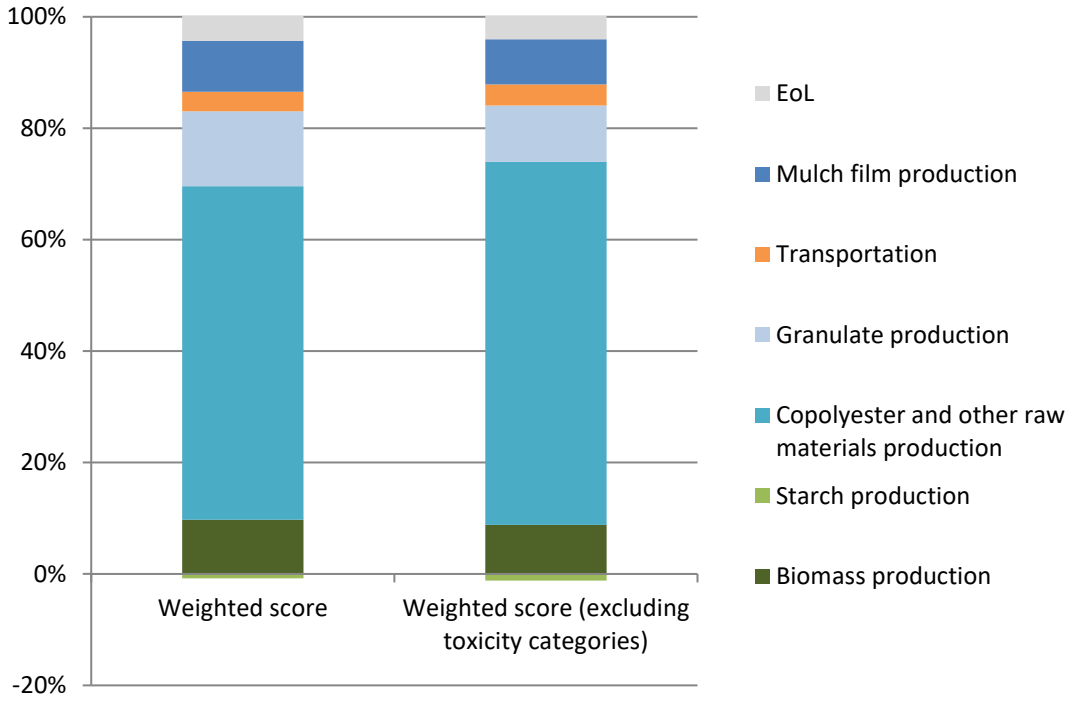


Figure 145. Contribution of the life cycle stages and processes based on normalised and weighted results of bio-based mulch films, with and without toxicity categories

3.4.2 Petrochemical reference system

The cradle-to-grave LCA results for 1 functional unit (189 kg) of petrochemical LDPE mulch film are presented in Table 170. The breakdown of impacts between manufacturing and EoL is shown in Figure 146. The estimated EoL mix for petrochemical LDPE mulch films is 5 % recycling, 53 % incineration and 42 % landfilling (see section 2.2). It is seen that the EoL makes a significant (about 60 %) contribution to climate change, terrestrial and marine eutrophication. For particulate matter, abiotic depletion (fossil fuels) and NREU the contribution of EoL is minor (up to 9 %).

Table 170. Cradle-to-grave LCA results of 1 functional unit petrochemical LDPE mulch films.

Impact category	Unit	Manufacturing	EoL	Total
Climate change	kg CO_2 eq.	4.05E+02	4.71E+02	8.76E+02
Particulate matter	kg PM2.5 eq	7.88E-02	7.59E-03	8.64E-02
Photochemical ozone formation	kg NMVOC eq	1.65E+00	1.03E+00	2.67E+00
Acidification	molc H^+ eq	1.21E+00	1.12E+00	2.32E+00
Terrestrial eutrophication	molc N eq	2.88E+00	4.91E+00	7.80E+00
Marine eutrophication	kg N eq	2.74E-01	4.29E-01	7.03E-01
Abiotic depletion (fossil fuels)	MJ	1.50E+04	7.74E+02	1.57E+04
Non-renewable energy use (NREU)	MJ	1.59E+04	5.45E+02	1.64E+04

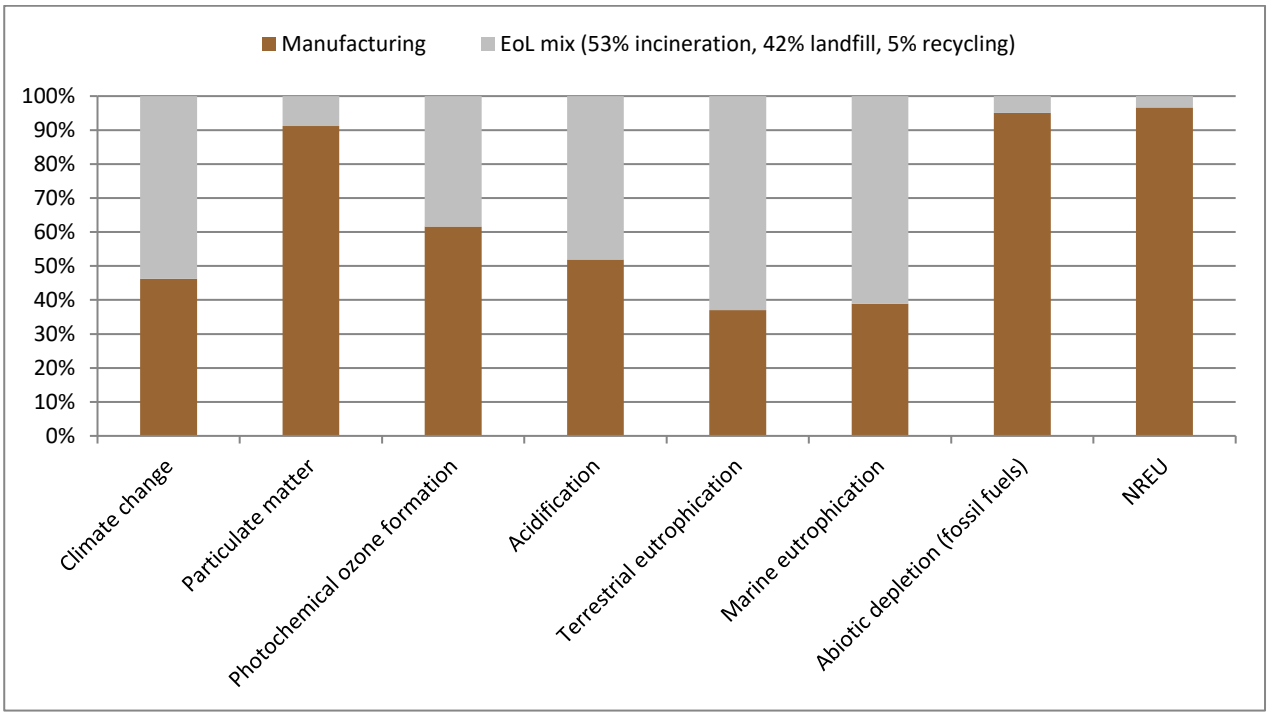
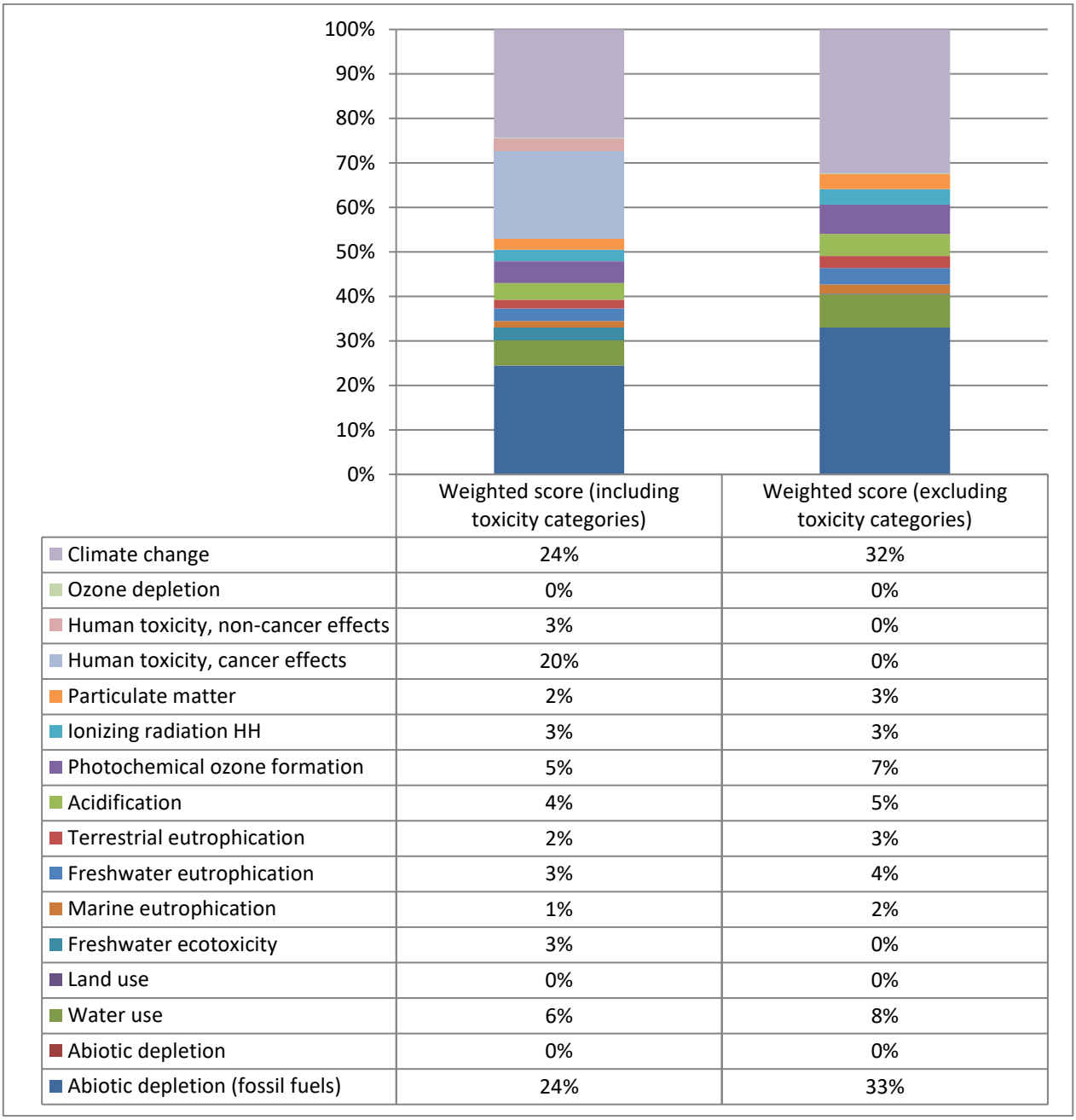


Figure 146. Breakdown of the cradle-to-gate LCA results for petrochemical LDPE mulch films.

Normalisation and weighting have been applied to the reference system (petrochemical LDPE mulch films) to understand which among the assessed impact categories are more relevant for this application and what the main environmental hotspots are. To do this, all the impact categories needed to be taken into the calculation. Hence, the weighted results are subject to considerable uncertainties, as highlighted in *Chapter 6 Ranges for environmental impacts from production of fossil-based plastics*, therefore their numerical values have to be considered only as indicative.

Based on the normalised and weighted results shown in Figure 147, excluding toxicity related impacts, the most relevant impact categories are: abiotic depletion (fossil fuels) (33 %), climate change (32 %), water use (8 %) and photochemical ozone formation (7 %) forming a cumulative 80 % contribution to total impacts. The rest of the impact categories are less relevant with each contributing up to 5 %. It is seen that the most relevant categories are among the suitable impact categories identified in Chapter 6, except for the water use category.

Contributions of the life cycle stages and the different processes within are given in Figure 148. Including the toxicity impact categories, manufacturing and EoL a have similar contribution. Excluding the toxicity categories, manufacturing is the most relevant life cycle stage contributing to a cumulative 69 % of the impacts with 62 % of the contribution coming from granulate production (extraction and refining of crude oil and natural gas, steam cracking of hydrocarbons into lower olefins, and polymerisation of the monomers into polyethylene).



	Weighted score (including toxicity categories)	Weighted score (excluding toxicity categories)
Climate change	24%	32%
Ozone depletion	0%	0%
Human toxicity, non-cancer effects	3%	0%
Human toxicity, cancer effects	20%	0%
Particulate matter	2%	3%
Ionizing radiation HH	3%	3%
Photochemical ozone formation	5%	7%
Acidification	4%	5%
Terrestrial eutrophication	2%	3%
Freshwater eutrophication	3%	4%
Marine eutrophication	1%	2%
Freshwater ecotoxicity	3%	0%
Land use	0%	0%
Water use	6%	8%
Abiotic depletion	0%	0%
Abiotic depletion (fossil fuels)	24%	33%

Figure 147. Contribution of the 16 PEF categories based on normalised and weighted results of petrochemical LDPE mulch films, with and without toxicity categories

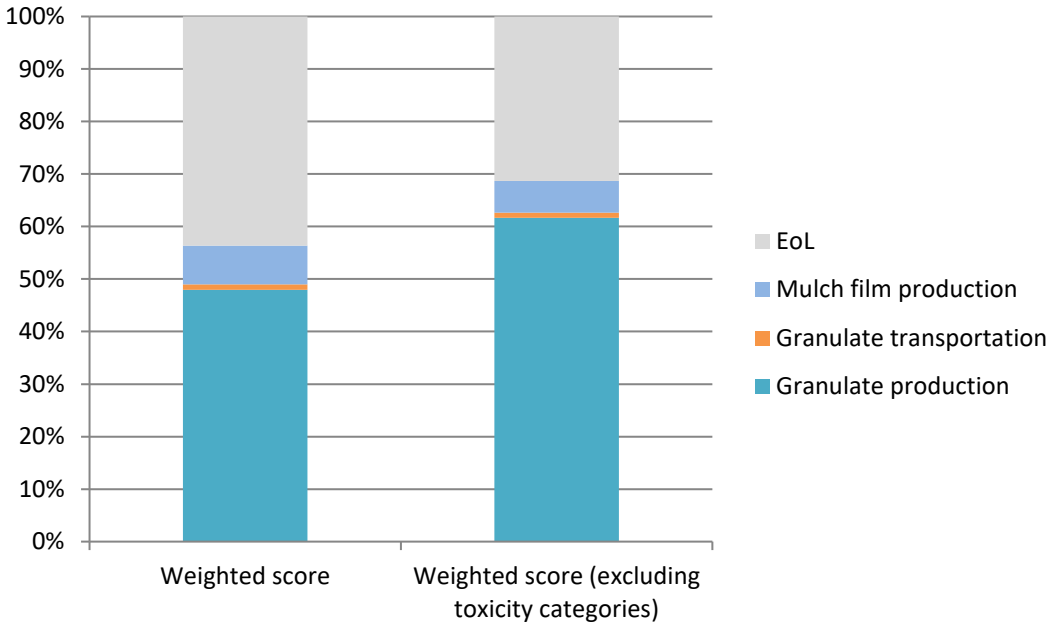


Figure 148. Contribution of the life cycle stages and processes based on normalised and weighted results of petrochemical LDPE mulch films, with and without toxicity categories

3.4.3 Comparing the Bio-Based baseline system with petrochemical reference

Based on the comparison of cradle-to-grave results per functional unit of bio-based mulch (91.4 kg/FU) and petrochemical mulch film (189 kg/FU) given in Figure 149, bio-based mulch films perform significantly better (about 70 %) than the petrochemical counterparts in five out of eight suitable impact categories (climate change, photochemical ozone formation, terrestrial eutrophication, abiotic depletion (fossil fuels) and NREU). For the acidification and marine eutrophication, bio-based mulch films perform about 40 % better. Only in the remaining category of particulate matter, bio-based mulch film performs 15 % worse than the petrochemical reference. For the bio-based mulch film this is influenced by the production of the petrochemical copolyester.

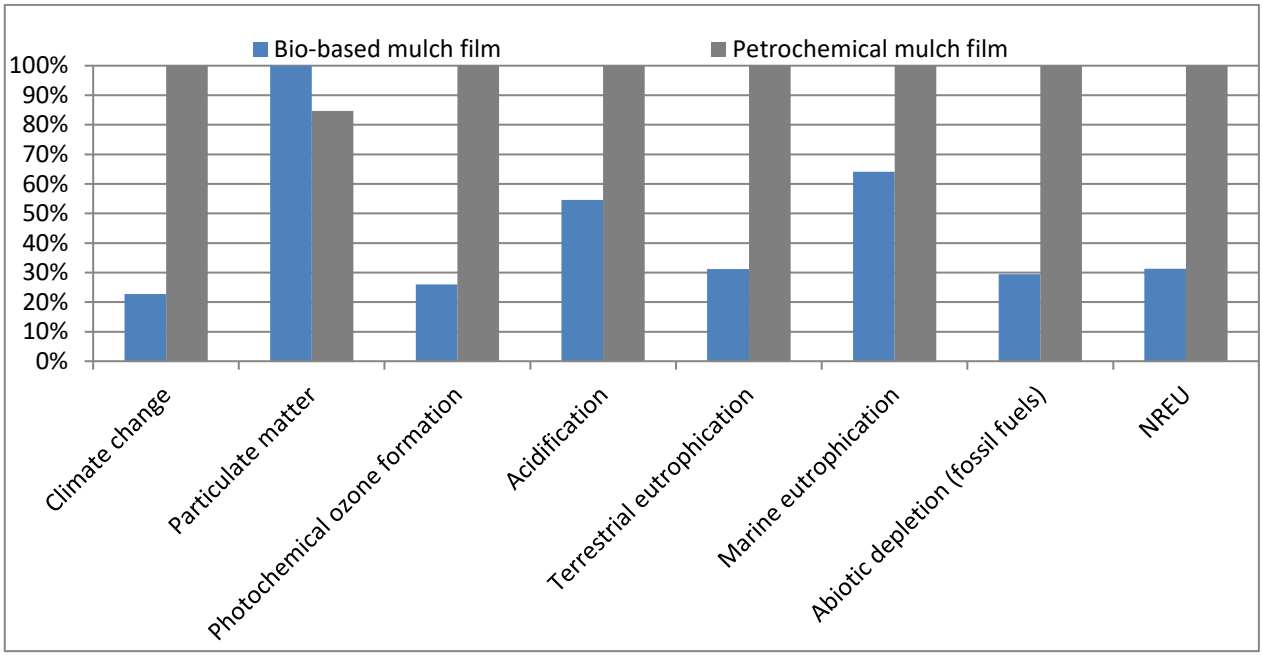


Figure 149. Comparing bio-based mulch films with petrochemical mulch-films, cradle-to-grave results (excluding DLUC and ILUC effects).

4 Discussion

4.1 End of life sensitivity analysis

The intended waste management technology for the bio-based product system is looked at more closely in this sensitivity analysis. For this field application product, in-situ biodegradation was chosen as the intended technology. The parameter is the degradation of carbon in the soil, which would be dependent on the soil type. The lower storage of carbon in soil and higher emissions to air are observed in coarse sandy soil (EASETECH) compared to the modelled sandy loam soil. The values used are presented in Table 171. EASETECH states a distribution for heavy clay soil of 89.13 % CO_2 emission to air and 10.86 % C soil storage. As mentioned previously, these figures are based on simulations in Daisy, a Danish deterministic agro-ecosystem model (Hansen et al., 2012).

Table 171. Variation in input parameters in the sensitivity analysis.

Product system	Technology	Parameter changed	Original EoL (%)	Higher (%)	References
Starch	In-situ	Division between C storage and CO_2 emission to air from the degraded C	11.31 / 88.68	9.88 / 90.11	EASETECH

The only impact category affected by change in this distribution are the climate change categories. Figure 150 presents the sensitivity analysis where both the original value and the higher limit are presented in terms of climate change. The additional climate change from modelling coarse sandy soil is minimal, 3 % of total climate change impacts compared to the original. Possibly there could be soil types in Europe with higher or lower split of carbon distribution between soil and air. This will not change the overall conclusion when comparing the EoL technologies, that in-situ biodegradation performs better (Table 148).

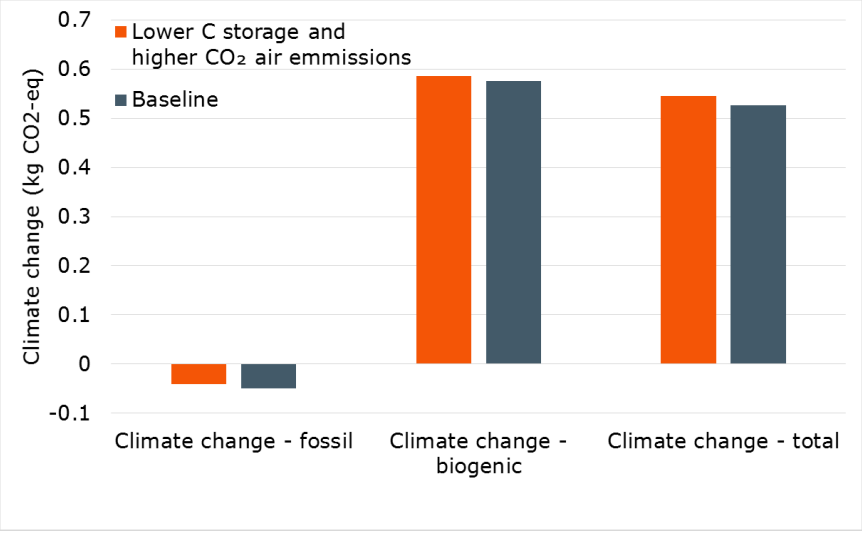


Figure 150. Sensitivity analysis for EoL mulching film for the In-situ.

This sensitivity would change the cradle-to-grave climate change results for the bio-based starch with less than 1 % for both a lower and a higher carbon degradation.

4.2 Discussion on End of Life uncertainties

The parameters analysed in the sensitivity analysis above and other parameters, which may be uncertain, i.e., estimations, assumptions, huge ranges in industry or EU member states, missing data, etc. are discussed in this section. Methodological shortcomings and additional research requirements are also presented.

In-situ field-based biodegradation is based on a Danish soil type (sandy loam soil), but the sensitivity analysis above shows that this parameter is not sensitive (less than 1 % change and only in climate change) to the overall cradle-to-grave results (see Figure 150).

The modelling does not take other impacts e.g., microplastics in the soil into account. Temperature and soil moisture will also affect biodegradation, which have not been included in this investigation. Further research on these matters is relevant.

Incineration and landfilling of biodegradable mulch films is questionable, as the film will possibly biodegrade or otherwise degrade, and therefore some starch plastic is not collected. Therefore, these technologies were not included in the EoL mix.

Quantity of soil attached to the mulch film. The treatment of 3 kg of soil per kg of plastic is based on values of contamination attached to mulch films from Plasticulture (2018) and carries an uncertainty. The environmental impact for EoL technologies other than in-situ will likely be sensitive to this as the treatment of extra soil has a large share of the overall environmental impact (see Annex 4 for contribution analysis). The treatment of soil has the largest influence on results for recycling and incineration, as the soil will cause increased workload for these EoL technologies.

Littering is not modelled in this LCA, but will be especially important for this product, as the mulch films are prone to being littered, as gathering used film from fields requires significant resources. Littering is further discussed in the Approach and Methodology chapter section 3.1.4. Plastic left in fields will be more prone to littering than plastic treated within the waste management system.

Possible energy consumption to gather the film is not accounted for. Gathering of the film by machine has not been accounted for in the modelling. In case study 2 the amount of energy used for gathering and sieving clips showed a large influence on GWP for all other EoL technologies than in-situ biodegradation. Inclusion of a specific energy consumption related to gathering of film in fields could be of significance but has not modelled. This would not alter the results that in-situ EoL performs best or that overall the starch EoL performs better than LDPE (based on the weighted results, see Figure 142 and EoL mix in Table 168).

Recycling of LDPE. In the EoL mix only 2% is actually recycled and, therefore, this parameter likely has low sensitivity towards the EoL mix. It should be noted that the sorting and technology efficiency is uncertain as this is only based on one scientific article and furthermore the collection rate is uncertain.

Apart from the above case study's specific uncertainties, several other general uncertainties that apply to most of the presented case studies are discussed in the Approach and methodology in chapter 3.5., e.g., data quality, critical assumptions and limitations in End of Life modelling. These include uncertainties related to:

- the estimated EoL mix
- the marginal energy technologies utilised
- the development of the EoL technologies (efficiency increases)

- the consumption of materials and energy at the EoL treatment plants
- biodegradation in landfills
- the chemical composition of the biodegradable plastics.

The quantity of soil attached to the film as well as biodegradability and rate of biodegradation of the starch in landfills are significant uncertainties.

Excluding littering from this case study due to inadequate methodology is a major sensitivity and means that the EoL mix assumed for the petrochemical film is unrealistic where littering is missing in the mix. The biggest emphasis for further research needs to be focused on littering: data pm biodegradation rates of biodegradable plastics in non-intended EoL technologies also is limited and should be addressed.

5 Assessing non-PEF impact categories: taking into account ILUC and endpoint of GTP

5.1 Global Temperature Change Potential

The GTP impacts of bio-based mulch film broken down into the main unit processes described in section the 2.1 inventory analysis are shown in Figure 151. When the biogenic carbon removal is taken into account, the cradle-to-gate GTP 100a is 137 kg CO_2 eq./FU. Observations for GTP are very similar to those of climate change impacts described in section 3.1.1.

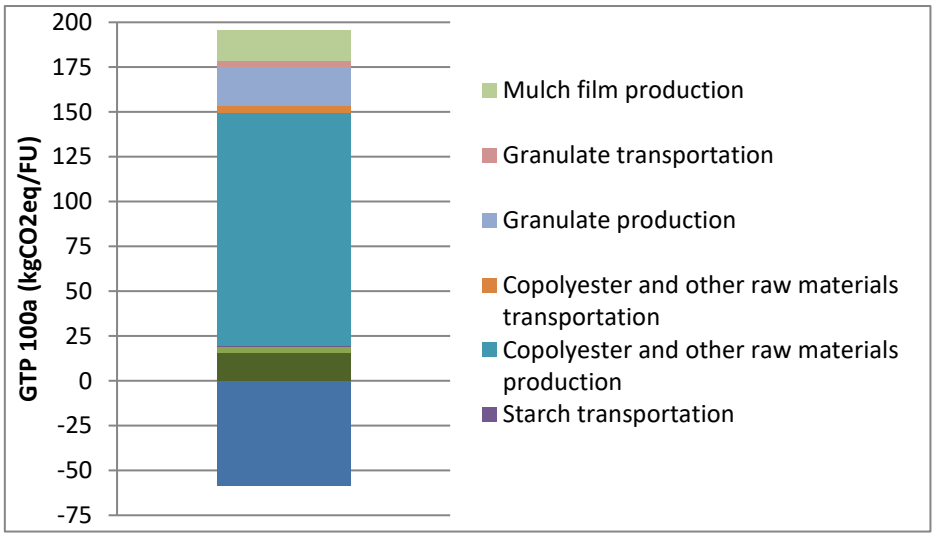


Figure 151. Breakdown of the cradle-to-gate GTP 100a results for bio-based mulch films, 1 functional unit, excluding DLUC and ILUC effects.

5.2 Land Use Change emissions

Table 172 presents the cradle-to-grave characterised results broken down into feedstock (for potato used for starch, maize used for starch and maize used for PLA; with biogenic carbon stored separately), manufacturing, EoL and ILUC (for potato used for starch; maize used for starch and maize used for PLA). As highlighted in Chapter 4, DLUC and ILUC cannot be summed up (it is whether DLUC or ILUC). This case does not involve crops that imply DLUC (namely Brazilian sugarcane, German maize, French wheat and German wheat), based on the methodology from the PEFCR Guidance V6.3 presented in Chapter 4. Figure 152 presents the characterised results for case study 5 with the inclusion of ILUC.

Table 172. Characterised cradle-to-grave results broken down per process, including ILUC. Case study 5, baseline.

	Biogenic carbon removals	iLUC (potato for starch)	iLUC (maize IT for starch)	iLUC (maize for PLA)	Potato production (for starch)	Maize IT production (for starch)	Maize production (for PLA, US)	Manufacturing	EoL
Climate change (kg CO_2 e)	-5.85E+01	1.92E+00	1.39E+01	8.42E+00	1.38E+00	1.45E+01	3.50E+00	1.91E+02	4.73E+01
Ozone depletion (kg CFC-11 e)		4.02E-10	2.89E-09	1.75E-09	1.72E-08	1.22E-06	1.25E-10	5.43E-05	-1.32E-10
Human toxicity, non-cancer effects (CTUh)		8.75E-10	6.30E-09	1.41E-09	1.01E-05	2.48E-06	9.07E-08	2.96E-05	2.83E-06
Human toxicity, cancer effects (CTUh)		8.31E-11	5.99E-10	2.25E-10	8.17E-08	8.55E-07	4.05E-08	9.36E-06	6.30E-07
Particulate matter (kg PM2.5 e)		5.07E-05	3.65E-04	2.22E-04	9.28E-04	9.24E-03	1.27E-03	9.05E-02	1.26E-04
Ionizing radiation HH (kBq U235 e)		-3.56E-05	-2.53E-04	1.21E-04	2.09E-02	3.94E-01	3.59E-01	3.30E+01	-5.19E-04
Photochemical ozone formation (kg NMVOC e)		5.97E-03	4.31E-02	2.62E-02	1.09E-03	6.51E-02	6.92E-02	5.41E-01	1.92E-02
Acidification (molc H+ e)		1.20E-03	8.64E-03	5.25E-03	4.04E-02	1.30E-01	5.25E-02	1.03E+00	1.39E-02
Terrestrial eutrophication (molc N e)		6.72E-03	4.85E-02	2.94E-02	1.78E-01	5.10E-01	2.84E-01	1.38E+00	8.20E-02
Freshwater eutrophication (kg P e)		4.42E-06	3.18E-05	1.94E-05	9.91E-04	3.89E-03	1.76E-04	5.60E-02	1.27E-05
Marine eutrophication (kg N e)		7.14E-04	5.15E-03	3.13E-03	3.75E-02	2.62E-01	2.98E-02	1.14E-01	7.50E-03
Freshwater ecotoxicity (CTUe)		1.48E-02	1.07E-01	1.56E-02	2.25E+01	7.37E+01	6.68E+01	8.77E+02	8.60E+00

Land use (kg C deficit)		5.53E-03	3.99E-02	2.42E-02	4.98E+01	3.46E+02	2.31E+02	1.96E+02	0.00E+00
Water use (m^3)		5.92E-02	4.27E-01	2.59E-01	2.91E-01	1.10E+02	1.01E+01	3.59E+01	-2.81E-01
Abiotic depletion (kg Sb e)		1.75E-08	1.26E-07	7.67E-08	1.24E-07	1.15E-04	5.94E-06	1.07E-04	-1.14E-08
Abiotic depletion (fossil fuels) (MJ)		4.99E-02	3.60E-01	2.19E-01	6.73E+00	1.35E+02	4.09E+01	4.46E+03	-4.91
NREU (MJ)		4.77E-02	3.44E-01	2.09E-01	7.33E+00	1.41E+02	4.29E+01	4.94E+03	-4.89

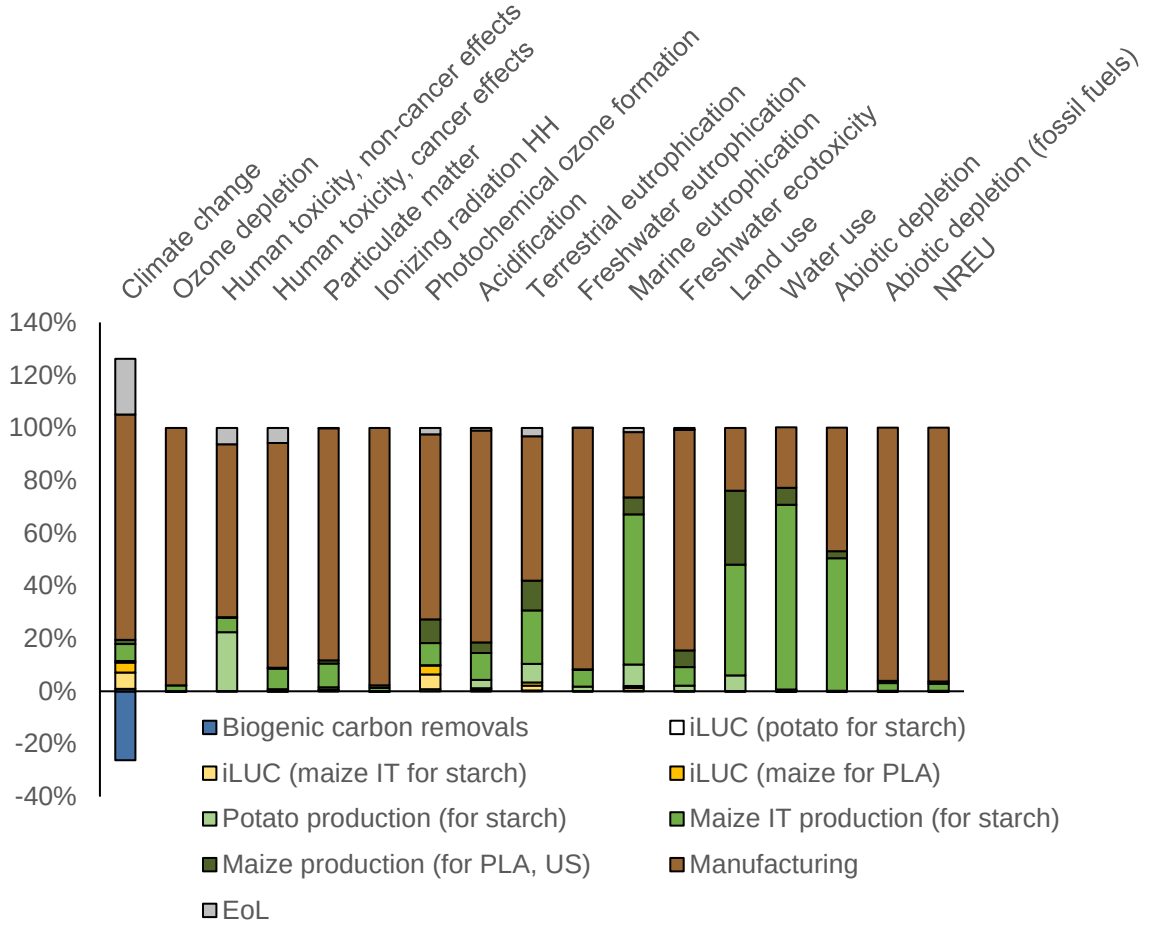


Figure 152. Relative characterised results broken down for all impact categories and including ILUC – case study 5, baseline.

As can be seen from Table 172 and to some extent from Figure 152, ILUC does contribute to all impact categories considered herein, as in previous cases. However, as for previous cases, the contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (11 % of the impact), photochemical ozone formation (10 % of the impact), terrestrial eutrophication (3 % of the impact) and marine eutrophication (1.8 % of the impact). All these impacts are dominated by land expansion (Figure 15). In the case of climate change, as earlier explained, the ILUC impact is essentially due to the CO_2 releases resulting from land clearing and in the case of photochemical ozone formation, to carbon monoxide also released during land clearing (burning the cleared biomass; Figure 17). Both acidification, terrestrial- and marine eutrophication are essentially due to nitrogen oxides released during land clearing (more than 80 % of the impact; Figure 17).

The contribution of ILUC to climate change has a magnitude slightly higher than the impact from producing the crops themselves (Figure 152; crop cultivation represents 9 % of the climate change impact when ILUC is included). As in case study 2, ILUC is here shown in white to yellowish tones for all crops contributing to it (for the climate change impact, ILUC potato starch represents 1 %, ILUC maize starch 6 % and ILUC maize PLA 4 %). Similarly, all crop cultivation processes are shown separately (greenish tones).

Although potato has a lower yield than maize, its ILUC impact is lower than for maize as less is used per functional unit. In absolute terms, the ILUC impact for climate change here is quite high compared to the one obtained for all other cases (except case 2). This reflects the importance of having conversion processes requiring as little biomass as possible, so the climate change impacts from both the crop cultivation and the ILUC can be lower.

6 Conclusions, limitations and recommendations

The goal of this case study was to assess the environmental profile of bio-based mulch film production and compare it with the fossil-based counterpart. Based on the results shown in section 3 of this case study, it is concluded that bio-based mulching films perform significantly better than the petrochemical mulching films on a cradle-to-grave basis for seven out of eight impact categories for which a comparison was possible. Especially for climate change, photochemical ozone formation, terrestrial eutrophication, abiotic depletion (fossil fuels) and NREU, the impacts of bio-based films are only 15-30 % of the petrochemical alternative. For the acidification and marine eutrophication, bio-based mulch films perform about 40 % better. For particulate matter emissions, the impacts of bio-based and petrochemical film are very similar. For the bio-based film, the manufacturing phase (conversion of biomass to mulch films) dominates the impacts in most categories except for the categories of marine eutrophication, land use, water use and abiotic depletion. In these categories, biomass production has a significant impact.

ILUC does contribute to all impact categories considered herein, as in previous cases. However, as for previous cases, the contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (11 % of the impact), photochemical ozone formation (10 % of the impact), terrestrial eutrophication (3 % of the impact) and marine eutrophication (1.8 % of the impact). All of these impacts are dominated by land expansion.

The weighted results of each EoL product system with their respective EoL mix show that the bio-based starch mulching film has a significantly lower environmental impact per FU. The weighted score with toxicity for the bio-based EoL mix is 0.16 and for the LDPE EoL mix it is 3.58. It can therefore be concluded that the bio-based product system for agricultural mulching film is the preferred choice.

There is though considerable uncertainty regarding the EoL of the mulching film in terms of the actual removal rates, degree of contamination, recyclability, shares that are being incinerated and landfilled across the EU etc., which need to be further investigated. Yet, these have limited impact on the overall results. The degree to which biogenic carbon is sequestered in the soil also does not impact the overall performance of the bio-based films. Furthermore, the petrochemical mulching film is prone to be littered (not collected and properly treated), which it has not been possible to address in this study.

Littering is excluded from this case study due to lack of data and knowledge of the impact assessment models. The exact amount (and relative share) of plastics that are littered is uncertain for a specific application and will differ widely between countries. The impacts of littering and microplastics are currently not properly captured in the toxicity-related impact categories (ecotoxicity and human toxicity). As pointed out by Pant et al. (2018) fate, exposure, and effect modelling for macro- and micro-plastics (incl. additives) is still in its infancy. Physical impacts (e.g. entanglement; ingestion of larger plastic particles and their effects) and chemical impacts (e.g. due to microplastic formation, microplastics as carrier of other chemicals, unknown impact of additives) and biological impacts (e.g. microplastics as carrier of germs/alien species) are largely unknown and currently not included in LCA. Thus, the impacts of littering (e.g. leaving plastics in the soil or in a marine environment) are most likely underestimated in the current LCA for the non-degradable applications.

Due to the problems related to assessing the impact of fossil films, the impacts on ozone depletion and all forms of toxicity could not be compared. As the long-term impacts of soil contamination with fossil film are also poorly understood, these impacts need to be investigated in more detail. Yet it is likely that these impacts are higher or at best equal to biodegradable films. In addition, when petrochemical film is removed, significant amounts of soil are removed with it, contributing to erosion. When this soil is either incinerated or

landfilled, it is permanently lost. Ultimately, it can be questioned whether these kind of impacts are adequately captured by the current set of standard impact categories.

Overall, it is concluded that bio-based mulching films offer significant environmental benefits in most impact categories that could be assessed, with little to no trade-offs.

CASE STUDY 6: FOOD PACKAGING FILMS

1 Goal and scope definition

1.1 Goal and background

In this case study, the goal is to assess the potential environmental impacts of bio-based and petrochemical packaging films. To have a coherent comparison, the final application of these packaging films have to be fixed in terms of performance. In this study, the analysis is focused on packaging films suitable to store vegetables for one week. Some bio-based alternative materials have been recently introduced for this application on the market. PLA is one of the most important bio-based materials for film packaging and it is the focus of the investigation in this chapter. A second bio-based material also included in the analysis is bio-based PP, which is a near-future emerging alternative. Both the considered bio-based materials are 100 % bio-based but only PLA is biodegradable. Among the different routes for bio-based PP production, the bio-based PP analysed in this case study is derived from used cooking oil. Among the petrochemical polymers, PP has been identified as the most common market benchmark for this application.

1.1.1 Polylactide

PLA has emerged as an important bio-based plastic in the past decades. Among its technical characteristics, its biodegradability is an interesting property that has attracted much attention in developing single-use packaging products.

The technological assessment of the PLA production process corresponds to a TRL (technology readiness level) of 9 (successfully operating technology). This fits the scope of the study i.e. focusing on commercialised bio-based products (see the chapter named *Introduction, goal and scope*). Figure 153 shows the chemical structure of PLA. PLA is a 100 % bio-based polymer produced through polymerisation of monomer lactic acid or lactide. Through ring opening polymerisation of lactide, a ring structure formed by self-esterification of two lactic acid molecules is obtained. Monomer lactic acid or lactide can be obtained by fermentation of sugars. Commercially, the L-lactide route, which leads to Poly-L-lactide (PLLA) is the dominant route. The alternative route - the D-lactide- is used to create a better thermal performance property (PDLA).

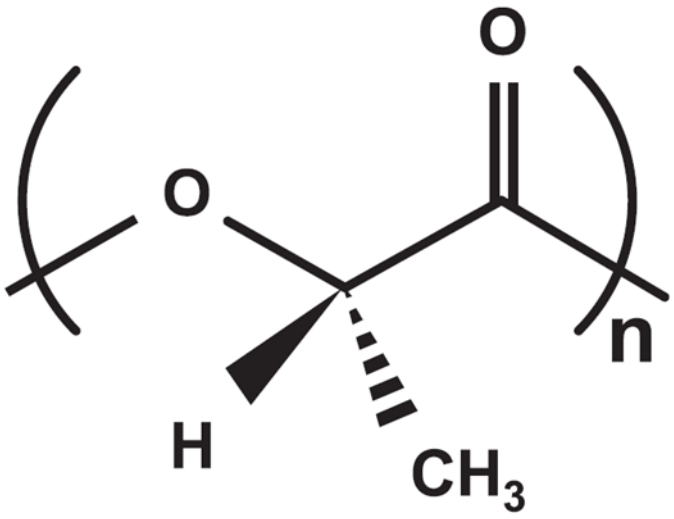


Figure 153. Molecular structure of PLA (Vink et al., 2015).

In 2018, industrial scale production of PLA (lactic acid) was offered by two key players: NatureWorks LLC, which has an annual capacity of 150 ktons in Nebraska, the the USA (Vink et al. 2015), and Corbion Purac/Total-Corbion PLA, which has a capacity of 75 ktons of lactide (and PLA from the second half of 2018) in Rayong, Thailand (Total-Corbion, 2018). The production capacity of PLA has increased by nearly 300 % compared to 2003 (based on data of 2003 reported by Shen et al. (Shen et al., 2010), representing one of the rapid growing bio-based plastics in the market. It is projected that PLA production will increase by more than 50 % by 2021 compared to 2017 (EuropeanBioplastics, 2017). The attention is on the PLA available on the current and near future European market. Moreover, a possible future PLA produced in Europe and from European maize is also assessed to estimate the potential environmental benefits deriving from the reduction of the transportation distance and possible disadvantages caused by the change of location of the biomass cultivation.

1.1.2 Bio-based polypropylene from used cooking oil (UCO)

Bio-based PP has the same chemical structure and characteristics as petrochemical PP but it is 100 % bio-based. Due to its chemical structure, bio-based PP belongs to the bio-based plastics which are not biodegradable.

There are many biochemical and thermochemical routes under investigation to derive PP from bio-based feedstocks. These routes are often linked with bio-diesel, renewable-diesel and ethanol production technologies. Currently, three routes are being investigated by the industry all using different feedstocks and applying different conversion technologies.

- **From sugar or starch crops e.g. sugarcane, sugar beet, maize or wheat.** Through biochemical sugar fermentation ethanol can be produced. Ethanol becomes the intermediate product from which ethylene can be derived. The process that causes this conversion is dehydration that together with dimerization allows the production of ethylene and butane-1. These two chemicals can be converted into propylene through metathesis. Braskem and Novozymes are developing a large-scale production of bio-based PP from sugarcane (Niaounakis, 2015).

- **From biomass gasification** (solid biomass eventually torrefied e.g. agricultural wastes). The gasification produces syngas that is treated in a dehydration process (of propanol/butanol) or that feeds a methanol synthesis unit. In the first case through dehydration and metathesis propylene is produced similarly to the previous route. In the second case the methanol is converted to dimethyl-ether (Gay et al., 2011).

- **From vegetable oil.** Bio-based naphtha can be derived from used cooking oil (or vegetable oil) through hydro-treatment. Propylene can be obtained from steam cracking. Neste with its partners is developing this production on an industrial scale (Neste, 2018).

In this project, it is possible to analyse the third route, bio-based -PP derived from vegetable used cooking oil) with the industrial support. The commercial scale pilot is at demonstration stage therefore the TRL can be assessed to be near 8.[66]

[66] It should be noted that the research and development of bio-based PP made from used cooking oil by Neste focuses primarily on durable applications rather than on single-use application (Personal communication with Neste, 2018). We nevertheless include this material because of the concept of utilisation of waste is of peculiar interest.

Geographical scope: the final product is purchased and disposed of in Europe while the manufacturing phases and the supply chain could extend globally (see chapter *Introduction, goal and scope*). Today, PLA is produced on a commercial scale by two producers:

- NatureWorks LCC, PLA granulates made from maize in Nebraska, USA.
- Total-Corbion PLA, PLA granulates made from sugarcane in Thailand.

The weighted average (based on production capacities) of the two production routes are taken as the default baseline for the bio-based PLA packaging films. See further detailed description in the section "product systems" and "life cycle inventory analysis" of this chapter.

Temporal scope: current production (2017-2018) with relevant developments foreseen for the short-time future (5-10 years) (see Chapter 2 "Approach and methodology" of the report).

Technological scope: PLA production of the commercialised technologies (9 level of TRL) are assessed using company-specific data. For bio-based PP made from used cooking oil, the TRL is around 8 -validated prototype. The technology has not been fully commercialised. Primary data are obtained from Neste.

1.2 Function and functional unit

The primary function is to provide packaging service for packaging fresh vegetables (e.g. bell pepper) for a short shelf life. The functional unit is defined as:

- 100 m^2 of film packaging for fresh vegetables for one week.

The primary function should include protecting of vegetables (i.e. with sufficient strength), keeping vegetables fresh for a short period of time (barrier properties of oxygen and water vapour) and the film should be transparent for customers to see through.

For this case study, only transparent films are considered and the material properties that are relevant for this application are: tensile strength, oxygen and water vapour permeation coefficients. Table 173 shows a comparison of the aforementioned properties between PLA and PP films. It can be seen that PLA is a heavier material; the tensile strength is better than PP, the oxygen barrier property is better than PP and the water vapour barrier property is much worse than PP. This makes PLA a strong film material for food packaging application with good oxygen protection properties while still allowing moisture to travel.

Table 173. Comparisons of intrinsic properties of PLA and PP bi-axially oriented film grades for food packaging application.

Material	Density (g/cm3)	Tensile strength (MPa) DIN EN ISO 527	Oxygen transmission rate (cc/m2/24hr) 23 °C, 0 % RH (25µm)	Water Vapour Transmission Rate (g/m2/24hr) 38-40°C, 90-100 % RH (25µm)
PLA [1]	1.2	45	1210	465
PP [2]	0.9	38	2526	6

Source: [1] (Bi-AX, 2018), (Corbion, 2016); [2] (EVAL, 2018), (MOLGROUP, 2017), (FILMAX, 2018).

In reality, packaging design is an iterative process requiring many trials and errors. It also has constraints other than material properties. For instance, very often new materials need to be

processed on an existing film extrusion machine with a similar setting as for processing the conventional materials. The existing plastics conversion infrastructure sometimes has a more determinative role than the calculated theoretical minimum for the final thickness and weight of PLA films.

Since the application is still a niche market, it is hard to identify a dominant design. We found contradictory arguments and designs for the same application.

Based on a small market survey conducted in this study, we did not find sufficient evidence for the PLA film packaging products which fit in line with the theoretical calculations based on the material substitution factors. Some film packaging companies (who produce film grade specifically for fresh vegetables) do offer PLA films with the same thickness as PP films for the same applications (Bi-AX, 2018). Moreover, via personal communication with the industrial experts, we understood that it is also possible to have the same area density (in m^2/kg) for both PLA and PP films for this application. Since PLA has a higher density than PP (see Table 173), this results in a thinner PLA film than a PP film. Nevertheless, a sensitivity analysis on the definition of the functional unit based on MSFs is also included in section 4.1.2.

In this study, the thickness of the PLA packaging film has been fixed to 25 micrometers which is a typical thickness for this application. Based on this thickness, the area density of PLA packaging films has been fixed to 32.1 m^2/kg according to a common commercial product (BI-AX, 2018). This area density corresponds to a one-sided heat sealable PLA film used to extend the shelf life of vegetables.

On a functional unit basis (100 m^2), this corresponds to 3.12 kg of PLA packaging film. The same weight is assumed also for 100 m^2 of PP films. This means that the PP film has a 36 µm thickness.

1.3 Product systems

Four product systems are analysed, three bio-based and one petrochemical counterpart:

- **Baseline:** PLA market average mix made from maize cultivated in the United States and sugarcane from Thailand. This product system reflects the status-quo of PLA production according to the defined temporal scope.

- **Alternative 1.** PLA made from maize cultivated in Europe, fictional case. This scenario has the goal to show the environmental consequences of a hypothetical production of PLA in Europe and from European maize.

- **Alternative 2**. Bio-based PP made from used cooking oil.

- **Reference 1.** Petrochemical PP.

1.4 System boundaries

A cradle-to-grave approach is applied including the life cycle stages of biomass feedstock production, transportation of intermediates (e.g. PLA granulates), manufacturing into end products (film extrusion), distribution of the final product and EoL. The investigated EoL options are plastic recycling, MSW incineration and landfilling for all the product systems. Moreover, PLA packaging films have the additional options of industrial composting and anaerobic digestion. The consumer use phase is excluded from the analysis because it is same for all the product systems and it would add a negligible impact. Thus, the life cycle results will be divided into two stages: cradle-to-user and EoL.

2 Life cycle inventory analysis

2.1 Process description, data, assumptions and multi-functionality

2.1.1 Baseline. PLA using maize from United States and sugarcane from Thailand

The baseline case consists of two major technologies for the conversion of sugars into PLA. These two technologies are represented by the NatureWorks and Total-Corbion production processes. Based on the capacities of the two producers, in the LCI modelling, PLA is assumed to be 33 % by weight from Total-Corbion and 67 % by weight from NatureWorks. This weighted average represents the 'average technology' of this baseline product system based on the current installed capacity (75 kton PLA/y and 150 kton PLA/y respectively for Total-Corbion and NatureWorks).

The commercial names of NatureWorks and Corbion® PLA are Ingeo® and Luminy®.The corresponding grades used for film extrusion are respectively named 4042D and LX175.

The process flow diagrams of the baseline case are shown in Figure 154 (NatureWorks's PLA) and Figure 155 (Total-Corbion's PLA). The following key unit processes are modelled in the foreground system:

- Biomass cultivation and harvest;
- Lactic acid and PLA production, consisting of three sub-processes;
- Starch/sugar milling;
- Lactic acid production;
- PLA production;
- Transportation of PLA polymer to Europe;
- Production of packaging films (film extrusion);
- Distribution of the packaging films;
- Use phase and EoL

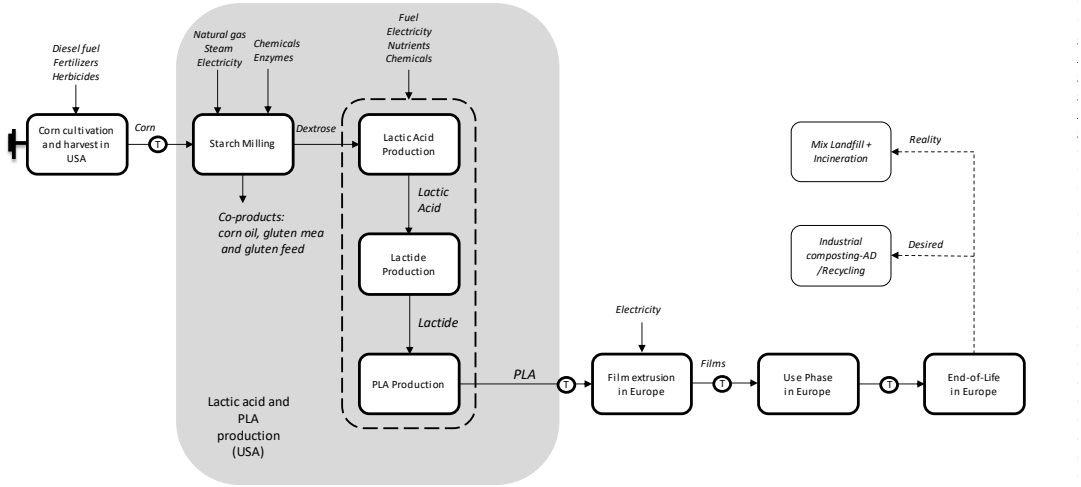


Figure 154. Process flow diagram of PLA films production from maize.

Table 174 shows a summary of the data sources used for the calculations for each stage of the cradle-to-user life cycle (see Figure 154 and Figure 155). Company specific data have been

used whenever possible. When recent company data were not available, literature and LCA databases have been used.

Table 174. Summary of data sources for each phase of the cradle-to-user life cycle - baseline

Processes	Data sources	Comments
Maize cultivation	Primary company data & Vink et al., 2015	Represents site-specific data
Sugarcane cultivation	Primary company data & Agri-fooprint 3 database	Represents site-specific data whenever possible and average Thai sugarcane cultivation assumed for missing data.
PLA and lactic acid production	Primary company data	Represents the specific production technologies of the two producers.
Transportation of the polymer	Assumptions based on the actual locations of the production plants	Represents the actual distances existing between Europe and the producers
Film extrusion	Ecoinvent 3.3	Represents film extrusion occurring in Europe
Distribution of films	Assumptions based on default distribution scenarios of PEFCR guidance	Represents the distribution of packaging films from the European manufacturing plants where the film extrusion process occurs and the users.

2.1.1.1 Maize cultivation and harvest

The life-cycle of NatureWorks's PLA starts with the cultivation and harvesting of maize which take place in Nebraska and Iowa (USA) (Vink et al., 2015). The main inputs in this phase are agricultural chemicals (fertilisers, herbicides and pesticides), electricity and fuels for the farm activities (e.g. diesel used by the tractors), agricultural land and irrigation water. The maize production is included in the dataset Gabi ts 8, Ingeo Polylactide (PLA) biopolymer production and has been modelled by NatureWorks. The current LCI data from cradle to factory gate of 1 kg of PLA produced by NatureWorks represents the latest NatureWorks' eco-profile for PLA (Vink et al., 2015) with updated background data using the Gabi database version 2018 (Personal communication with E. Vink).

The site-specific data for the maize used for NatureWorks PLA have been used in this part of the LCI. The black box impact of this maize has been provided by the company. The yield is assumed to be 10.28 ton/ha/yr for maize with 15 % moisture by weight (Vink et al., 2015).

2.1.1.2 Sugarcane cultivation and harvest

The sugarcane used for PLA production is cultivated in Thailand. The sugarcane cultivation and harvesting life-cycle stage has been modelled based on primary company data and included in the dataset for PLA provided by the company. *Sugar cane, at farm/TH Economic* from Agri-footprint 3 has been used as the main process to obtain the breakdown corresponding to the cultivation step. The water inventory, provided by the company in the sugarcane cultivation dataset, was based on the actual water needed in the local area where the main mills providing the sugarcane are located.

2.1.1.3 Lactic acid and PLA production

Based on the different biomass and sugars, the conversion process of the sugar into PLA can start from dextrose (NatureWorks' technology) or sucrose (Corbion-Total's technology).

In the case of NatureWorks, the harvested maize is transported to the maize wet milling factory where dextrose is produced via enzymatic conversions. Only the starch content

(estimated to be 58 % of the mass content based on Vink et al. (Vink et al., 2015)) of maize will be hydrolyzed to dextrose. The co-products of the starch-milling and their percentages on a dry mass-basis are: maize oil (about 3 %), gluten meal (about 5 %) and gluten feed (about 25 %) (Vink et al., 2015). During dextrose production steam and electricity are needed and sourced locally (Nebraska, US infrastructure). Steam is produced in natural gas fired steam boilers located at the NatureWorks' plant (Vink et al., 2015). To solve this case of multi-functionality, sub-division was performed by Vink et al. (Vink et al., 2015) in the NatureWorks' eco-profile. The unit process has been divided into 11 sub-processes (Vink et al. 2015) and, for each sub-process, where an allocation was needed a mass allocation (on dry basis) was applied. Dextrose is sent to the fermentation unit to produce lactic acid which is the precursor of the polymer PLA. In lactic acid production, gypsum is produced as a by-product in small quantities. A credit was assigned through the substitution of primary production of gypsum from mining (Vink et al., 2015). The entire starch/sugar milling, lactic acid production and PLA polymerisation are in an integrated production site which is located in Blair (Nebraska). 1.57 kg of maize is required to produce each kg of PLA (Vink et al. 2015). *Gabi ts 8, Ingeo Polylactide (PLA) biopolymer production* has been used to model this unit process for NatureWorks' technology with updated background data from 2018. This LCI model is confidential.

At Total-Corbion PLA, the biomass used for PLA production is sugarcane. The harvested sugarcane is processed to produce raw sugar (Figure 155). The by-products of this process with common weight shares are: molasses (4.2 %w), filter cake (4.5 %w) and bagasse (34.0 %w) (Prueksakorn et al., 2014). The sugar milling of sugarcane requires heat and electricity which are entirely produced by burning bagasse in a combined heat and power (CHP) plant incorporated into the sugar mill. The surplus of electricity is sold into the Thai national grid (Groot et al., 2010). The sugar is then transported and processed into the lactic acid plant located next to the mill. During the lactic acid production process, gypsum and stillage are produced as by-products and are removed through filtration and sold (Groot et al., 2010). This gypsum has similar characteristics to mined gypsum while the stillage is used as animal feed due to its nutritional properties (system expansion with substitution of mined gypsum and animal feed has been applied). There are also other minor fermentation residues which are also purified and sold and used as soil conditioner. After lactic acid purification, PLA is produced through open ring polymerization. Primary data provided by the company based on the actual operation data has been used to model this unit process for Total-Corbion PLA's technology. The technology is the same as described by Groot et al. (2010). The updated LCI dataset is confidential.

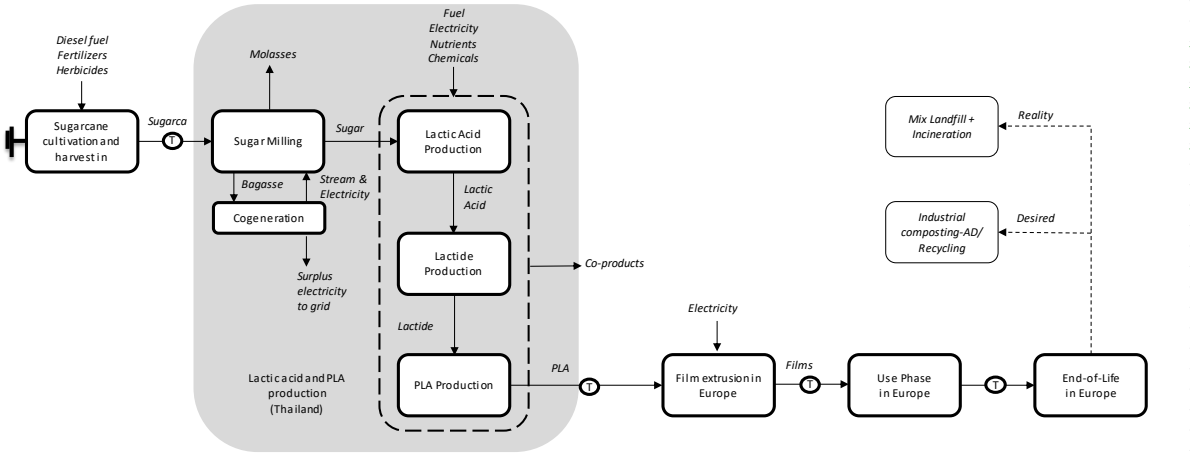


Figure 155. Process flow diagram of PLA packaging films production from sugarcane.

2.1.1.4 PLA granulates transportation from US (Nebraska) to Europe

The PLA granulates produced in Nebraska (US) are transported to Europe to be converted into packaging films. The distances assumed in the modelling are:

- 2000 km by freight train from Nebraska to a suitable harbour (assumed to be New York) on the Atlantic coast of the United States. For this transport step, specific data based on the actual location of NatureWorks are used. *Transport, freight train {US}* from Ecoinvent 3.3 has been used as dataset.

- 6000 km by transoceanic container ship from New York to an European harbour (assumed to be Rotterdam). The process named *Transport, freight, sea, transoceanic ship {GLO}* from Ecoinvent 3.3 has been used.

2.1.1.5 PLA granulates transportation from Thailand to Europe

The PLA granulates produced in Rayong (Thailand) are transported to Europe to be converted into packaging films. The distances assumed in the modelling are:

- 200 km by freight train for transportation within Thailand. For this transport step, specific data based on the actual location of Corbio-Total are used. *Transport, freight train {US}* from Ecoinvent 3.3 has been used as dataset.

- 17000 km by transoceanic container ship from Bangkok to an European harbour (assumed Rotterdam). The process named *Transport, freight, sea, transoceanic ship {GLO}* from Ecoinvent 3.3 has been used.

2.1.2 Production of packaging films: film extrusion

The extrusion of oriented plastic films can be performed through two different techniques: blown film extrusion or cast film extrusion. In both the techniques, the molten material is quenched and then reheated and stretched. PLA is only suitable for cast film extrusion (Jamshidian et al., 2010). This cast film extrusion process starts with the extrusion of the molten polymer through an extrusion die and then it is extruded and quenched on high-speed cooling rollers. The polymer is later reheated on other rolls to provide the desired axial orientation to the film. This orientation process is needed to improve the mechanical properties of the film. In this way, it is possible to improve the thermo-mechanical characteristics of PLA films to the same mechanical resistance of polypropylene films. As mentioned, PLA alone is not generally processed through film blowing due to its poor melt strength. Nevertheless, there is the possibility to improve the ratio of additives or the way they are introduced to increase the PLA's molecular weight and therefore to enlarge its range of processing applications (Mallet et al. 2013).

Extrusion, plastic film {RER}| production | Alloc Def, U) from Ecoinvent 3.2 has been used to model this process. It has been adjusted to reflect PP film extrusion. However, it has been used also for PLA film extrusion hence the same electricity consumption has been assumed for both the materials.

2.1.2.1 Distribution of the packaging films to end user (within Europe)

For the final distribution of the product, it is assumed that the films are distributed through a distribution centre and then to the final user at a vegetable packaging site.

According to previous similar studies (OVAM, 2006), a distance of 850 km between the producer of the films and the distribution centre has been assumed. The process *Transport, freight, lorry >32 metric ton, EURO4 {RER}* from Ecoinvent 3.3 has been used as the dataset.

Subsequently, the packaging films are transported to the user. The distance between the user and the distribution centre is assumed as 250 km (according to PEFCR). According to PEFCR guidance, the process *Transport, freight, lorry 3.5-7.5 metric ton, EURO3 {RER}* has been chosen.

This scenario for the distribution of the films has been implemented for all the product systems.

2.1.2.2 Alternative 1. PLA using maize from Europe

This system represents a fictional production of PLA packaging films by exploiting the potential impact of using European maize. This product system is largely based on NatureWorks' processes. Compared to NatureWorks's maize-PLA described above, two major changes have been applied for this alternative case:

- Maize cultivation and harvest in Europe. The European average maize is modelled based on Agri-footprint 3 data. The most important European maize producers are in Germany, France, Italy, Romania, Hungary, Spain, Poland and Bulgaria, in total these countries produce about 85 % of European maize per year (based on EUROSTAT average data from 2013 to 2017) (EUROSTAT, 2017).The contribution of each EU country to the average mix assumed for the modelled maize is shown in Table 175.

- PLA polymer transportation. Since this is a hypothetical supply chain, no real company data is available. We therefore used PEFCR's default recommendations to model the transportation. According to PEFCR guidance for all suppliers inside Europe, the following transportation distances and mode are assumed: 230 km by truck (>32 ton, EURO 4) (*Transport, freight, lorry >32 metric ton, EURO4 {RER}| transport, freight, lorry >32 metric ton, EURO4* | *Alloc Def, U*), 280 km by train (*Transport, freight train {Europe without Switzerland}| market for* | *Alloc Def, U*) and 360 km by ship (*Transport, freight, inland waterways, barge {RER}| processing* | *Alloc Def, U*). All the foreground data of these three processes have been extracted from Ecoinvent 3.3.

Table 175. Scoring of case studies.

EU country of production	Share assumed in the model
Germany	7.9 %
France	23.7 %
Italy	14.7 %
Romania	17.1 %
Hungary	12.5 %
Spain	13.0 %
Poland	6.4 %
Bulgaria	4.7 %

It should be noted that because of a lack of transparency in the lactic acid and PLA production process, we assumed that NatureWorks's production can be used as a proxy for a fictional European lactic acid and PLA production. Although it is important to keep in mind when comparing the two: the background data for Europe would be different from those for the US. Table 178 summarises all the main data sources used to model this product system.

Table 176. Summary of data sources for each phase of the cradle-to-user life cycle-Alternative 1.

Processes	Data sources	Comments
Maize cultivation	Primary company data & Agri-fooprint 3 database	Represents European average data
PLA and lactic acid production	Primary company data	Represents the specific production technologies of one current producer of PLA that uses the same route in another country.
Transportation of the polymer	Assumptions based on the actual locations of the production plants	Represents the actual distances existing between Europe and the producers
Film extrusion	Ecoinvent 3.3	Represents film extrusion occurring in Europe
Distribution of films	Assumptions based on default distribution scenarios of PEFCR guidance	Represents the distribution of packaging films from the European manufacturing plants where the film extrusion process occurs and the users.

2.1.2.3 Alternative 2. Bio-Based PP from cooking oil

The second alternative product system refers to bio-based PP derived from used cooking oil). This is a TRL 8 technology with a successfully operating prototype. The flowsheet describing this product system is shown in Figure 156.

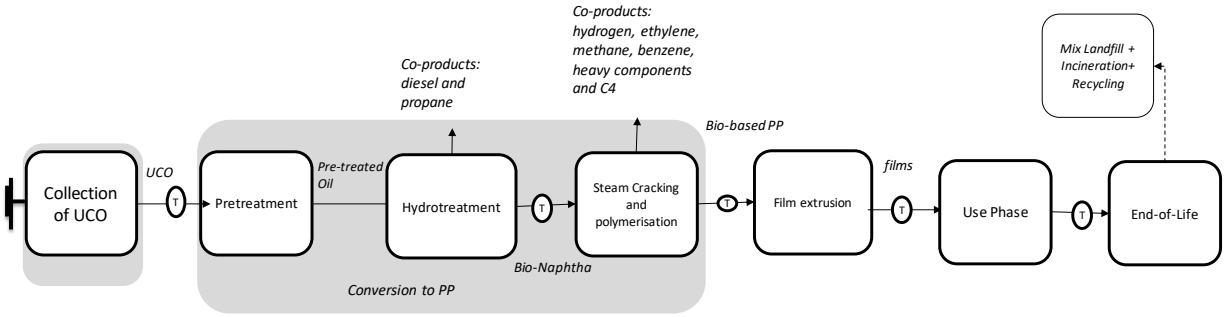


Figure 156. Flowsheet of bio-based PP packaging films obtained from UCO.

The first step which is identified is the collection of UCO. The collection of UCO occurs locally close to where the refinery plant is located[67]. The UCO is collected from the used oil and fats from restaurants, commercial and private waste and local waste collection centers by trucks or vans.

At the refinery, the collected crude UCO is first sent to the pre-treatment process where solid impurities are filtered out. The process requires water, chemicals, heating and cooling which are locally sourced. The waste water of this operation is then treated in an on-site waste water treatment plant.

The pre-treated oil is ready to be upgraded via hydrotreatment by adding high pressure hydrogen. The final output of the hydro-treatment step is a hydro-treated vegetable oil (i.e. renewable diesel), propane, bio-based naphtha (refer to as 'bio-naphtha' from here on) and a large amount of water. This bio-naphtha is the feedstock to produce bio-based polypropylene. The by-product propane is sold. The main input of the process, hydrogen, is sourced from refinery hydrogen produced via steam reforming of natural gas.

The bio-naphtha is then transported to the cracking plant. The assumed distance is based on site-specific data and is kept confidential.

The steam cracking of bio-naphtha is very similar to that of petrochemical naphtha; the steam cracking of naphtha delivers two main products, propylene and ethylene, and a number of by-products. Ethylene represents the highest mass fraction among the cracked gases compared to propylene. The fuel burnt in the cracking unit is LPG. Additional energy is provided by steam which is also largely used for dilution. The by-products produced are heat, hydrogen, methane and other light gases, benzene and heavier products. Exergy allocation is performed for this multi-output process to consider the quality of the energy outputs.[68]

The last step of the chain is the polymerization of the propylene and it is the same as the polymerization of the petrochemical PP.

Energy allocation is applied as a partitioning approach to solve the multifunctionality of hydrotreatment. The use of a partitioning approach to solve this case of multifunctionality is not fully consistent with the section *Procedure of multi-functionality processes* in the Chapter 2 "Approach and methodology". However, applying substitution when the focus is a non-

[67] The specific site is analysed in this LCA, but the location is kept confidential.

[68] This is a common allocation method when heat is one of the co-products (e.g. this has been applied by Edwards et al. (Edwards et al., 2017)) when the quality, not only the quantity, of the energy needs to be addressed.

dominant by-product (i.e. bio-naphtha and bio-propylene in this case) could lead to misleading results because all credits will be assigned to a minor stream of the production of which the output of the technology is to optimise the main product - renewable diesel. The importance of the co-products' physical significance in the choice of allocation method is not sufficiently guided by the ISO guidance (Sandin et al., 2015), neither does the latest PEFCR guidance provides this instruction. According to the ILCD handbook, the dominant co-product to which all the credits are assigned should 'contribute more than 50 % to the combined market value of all co-functions of the analysed multifunctional process or system'. In the case of steam cracking co-production, e.g. ethylene is produced with a higher yield compared to propylene therefore this last requirement is not properly complied with propylene which is the investigated product. For the base case, we choose the recommendation from RED to perform the allocation based on energy/exergy content since all by- and co-products are energy products. A discussion on this allocation methodological choice is also included in annex *Multi-functionality bio-based PP cups/packaging films product system*.

The biogenic carbon removal is considered for the cradle-to-user phase according to the RED II proposal because a vegetable waste is used, and biogenic carbon will be stored in the product. This amount of carbon will be then released into atmosphere during EoL as carbon emission if the carbon is fully oxidised (e.g. in a MSWI).

The summary of the data sources for the unit processes highlighted in Figure 156 is reported in Table 177.

Table 177. Summary of data sources for each phase of the cradle-to-user life cycle-Alternative 2.

Processes	Data sources	Comments
UCO collection, pre-treatment and hydrotreatment	Primary company data; Nikander, 2008; IFEU, 2006; JEC, 2014.	Represents site-specific data
Transportation of the Naphtha and steam cracking	Primary company data & Karimzadeh et al., 2015	Represents site-specific data whenever possible.
Polymerisation	PlasticsEurope- Industry 2.0 database (PlasticEurope, 2013)	Represents European polymerisation.
Transportation of the polymer	Assumptions based on default transportation scenarios of PEFCR guidance	Represents the transportation of the polymer from the biorefinery to the plants where the thermoforming process occurs
Film extrusion	Ecoinvent 3.3	Represents film extrusion occurring in Europe
Distribution of films	Assumptions based on default distribution scenarios of PEFCR guidance	Represents the distribution of packaging films from the European manufacturing plants where the film extrusion process occurs to the users.

2.1.2.4 Reference system: petrochemical PP

The flowsheet of petrochemical PP packaging films system can be found in Figure 157. As all the conventional plastics, PP derives from steam cracking of Naphtha, which is a multi-output unit where propylene is one of the co-products. By means of propylene polymerisation, polypropylene resin is obtained, and it is ready for film extrusion. The dataset selected for the baseline is retrieved from PlasticsEurope eco-profile. However, as explained in Chapter 6 (Ranges for environmental impacts from production of fossil-based plastics), PlasticsEurope

data for petrochemical plastics are not transparent and do not comply with ILCD requirements. Differences between the available data from different data sources for PP are found to be large and some impact categories were found not suitable, Chapter 6 Ranges for environmental impacts from production of fossil-based plastics (see section 6.1 in Approach and methodology).

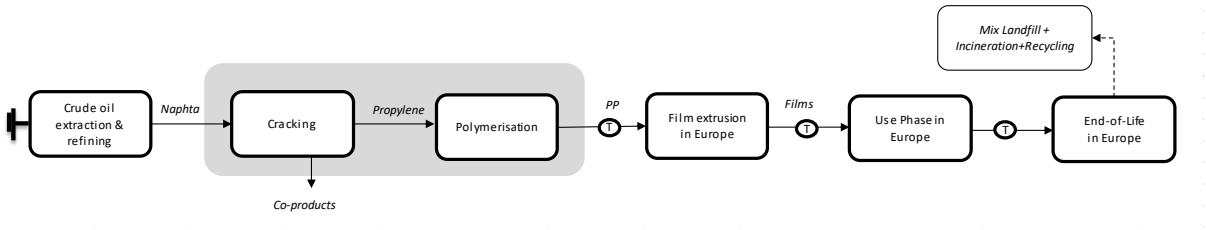


Figure 157 . Process flow diagram of PP packaging films production from crude oil

The main data sources used to model the unit processes shown in Figure 157 are reported in Table 178.

Table 178. Summary of data sources for each phase of the cradle-to-user life cycle-Petrochemical reference 1.

Processes	Data sources, foreground	Comments
PP production (fossil)	Plastics Europe, 2017 & Industry data 2.0	Represents European average production of petrochemical PP
Transportation of the polymer	Assumptions based on default distribution scenarios of PEFCR guidance	Represents the transportation of the polymer from the European refineries to the European manufacturing plants where the films are produced
Film extrusion	Ecoinvent 3.3	Represents film extrusion occurring in Europe
Distribution of films	Assumptions based on default distribution scenarios of PEFCR guidance	Represents the distribution of films from the European manufacturing plants where the extrusion process occurs to the users.

2.2 End of Life description, data, assumptions and multifunctionality

The food packaging films have different possible EoL options depending on product system, as observed in Table 179.

The petrochemical and bio-based PP food packaging films have three possible EoL options;

- **Mechanical plastic recycling**; the process includes the energy and material requirements for the transportation to the facility, sorting, cleaning and recycling processes. The recycled PP is assumed to substitute virgin PP production. The rejects from the recycling process are sent to incineration (both with and without energy recovery) and includes transportation. The flowchart for plastic recycling is presented in Figure 8.

- **MSW Incineration with and without energy recovery**; a generic MSW incineration plant is assumed, which represents average EU waste incineration with and without

energy recovery with an overall heat efficiency of 22 % and electrical efficiency of 9 % (CEWEP, 2013; Eurostat, 2017a). The process includes the direct emissions from waste incineration as well as indirect emissions from the production of the input materials, the combustion of fossil fuels as well as treatment of bottom ash and fly ash. The flowchart for MSW incineration is presented in Figure 9.

- **MSW landfilling**; the process includes the construction and operation of an average landfill site, direct emissions of landfill gas, landfill gas collection, flaring and recovery system, treatment of collected leachate and direct emissions from uncollected leachate. 29 % of the collected landfill gas is used to generate electricity (OpenLCA Nexus, 2015). The electricity conversion efficiency is 37 % (Christensen, 2011). The flowchart for MSW landfilling is presented in Figure 10.

Whereas the biodegradable PLA food packaging film in addition have;

- **Industrial composting**; includes indirect emissions from material and energy consumption, direct emissions from the plant and use on land, fertiliser substitution from compost and rejects sent to incineration (30 % of plastic and 5 % of organic waste). The flowchart for industrial composting is presented in Figure 12.

- **Anaerobic digestion**; mix of technologies modelled (combined with and without post-maturation (aerobic)), which includes indirect emissions from material and energy consumption, pre-treatment (rejects sent to incineration - 30 % of plastic and 5 % of organic waste), electricity, heat and fuel generation from biogas, direct emissions from the plant and use on land, and fertiliser substitution from compost and digestate. The flowchart for anaerobic digestion is presented in Figure 13.

The intended EoL technology for the PLA packaging film is industrial composting. For the alternative bio-based PP the intended technology is recycling.

In addition, to modelling full (100 %) for each technology, an EoL mix for Europe is modelled and is presented in Table 179. These figures are for the petrochemical EoL mix based on the available waste statistics in Europe on plastic waste treatment. The current collection for treatment of plastic waste is 30 % recycling, 39 % incineration and 31 % landfilling in Europe (European Commission, 2018). A report from Plastics recylers Europe (2015) reports the collection rate of PP films as 32 % for commercial and industrial waste and 18 % for household waste. These waste statistics is assumed to be applicable for the PP packaging film.

For the EoL mix for the PLA packaging film it is estimated that the Europeans can sort the same amount of PLA packaging film, which they today are able to sort for recycling of the commercial plastics, hence 30 %. This amount would then be sorted to the intended technology, industrial composting.

The application of a general recycling percentage for bio-based as well as petrochemical plastic (based on how much plastic is currently collected for recycling in the EU) in the EoL technology mix may be questioned for especially this case study. However, keeping the share of collection for recycling constant between different product systems ensures that the comparison of bio-based and petrochemical products is not biased by assumed differences in the waste system. With the new targets of separate collection of organic waste, an increase in separate collection of organic waste is expected within the study's timeframe of the coming 5-10 years.

Table 179. The EoL options, their intended EoL option and the estimated EoL mix.

Product system	EoL options modelled individually (100 % scenarios)	Intended EoL option	Estimated EoL collection mix of EU (based on statistics of waste treatments of waste plastics)

Bio-based PLA	Recycling Incineration Landfilling Industrial composting Anaerobic digestion	Industrial composting	Recycling: 15 % Industrial composting: 15 % Incineration: 39 % Landfilling: 31 %
Bio-based PP	Recycling Incineration Landfilling	Recycling	Recycling: 30 % Incineration: 39 % Landfilling: 31 %
Petrochemical PP	Recycling Incineration Landfilling		Recycling: 30 % Incineration: 39 % Landfilling: 31 %

The average European EoL technologies are described in the approach and methodology chapter in the section regarding EoL. In that section, factors that do not depend on the specific product system are described, e.g. the energy efficiency of the incineration plant and the consumption of energy and materials in each technology. Furthermore, principles for substitution are described. This involves that the energy produced in the EoL technologies is substituted with marginal electricity and heat, as well as the recycling substitution methodology. The recycling rate is calculated by multiplying the collection rate and the sorting and technology efficiency.

In addition, there are several product system dependent factors, which are presented in Table 180 for each possible EoL technology. The recycled plastic is substituted with petrochemical PP in all product systems. The recycled PLA will substitute PP. This is chosen as the study looks at the limited time frame of the next 5-10 years. In this time period, even recycling of PLA is not known to become commercially, due to low amounts of PLA. Hence PLA will for a good time replace PP, as this will be the marginal plastic. The PLA could also replace PLA, and the implications of this are discussed in the results.

Table 180. Values for product system dependent factors for packaging film.

EoL Technology	Product system dependent factor	Unit	Bio-based PLA	Reference	Bio-based PP	Petrochemical PP	Reference
Plastics recycling	Sorting and technology efficiency (amount collected for recycling which is recycled)	% of input	70	Estimated based on Plastic Recyclers Europe, 2017, Replast, n.d., SWEREC, 2007, Franklin Associates, 2011, Giugliano et al., 2011 and Dall et al., 2003	70	70	Estimated based on Plastic Recyclers Europe, 2017, Replast, n.d., SWEREC, 2007, Franklin Associates, 2011, Giugliano et al., 2011 and Dall et al., 2003
MSWI with and without energy recovery	Energy content[a]	MJ/kg	18.72	Boonmee et al., 2016; Pradhan et al., 2010; Hermann et al., 2010; Krause and Townsend., 2016; Kolstad et al., 2012	37.84	37.84	Götze et al., 2016 (Modified C)
Landfilling	1st order decay rate for methane generation	1/s	0.011	Calculated based on (Kolstad et al., 2012)	0	0	Estimated
Industrial composting	VS Degradation (%)	% of VS	80	Pradhan et al., 2010			
	C Degradation (%)	% of C bio	57.1	Pradhan et al., 2010			
Anaerobic digestion	Anaerobically biodegradable biogenic carbon	% of bio C	47	Pradhan et al., 2010			
Energy content is in lower heating value (LHV).							

The food packaging films have the following material compositions, see Table 181, which is used as the input to the EoL LCA model. The full chemical composition is presented in Annex 1. The EoL-reference flow is:

- PLA: 1 kg plastic, 100 g contamination and 15 kg food leftovers
- PP: 1 kg plastic, 100 g contamination and 15 kg food leftovers

The contamination consists of organic waste – see further details in Approach and methodology chapter, section 3.3.2 Contamination composition.

Table 181. Material composition of food packaging film (1 kg plastic + 100 g contamination).

Chemical component	Unit	Bio-based PLA	Bio-based PP	Petrochemical PP
Water	% of total	6.6	3	3
TS (VS+ash)	% of total	93.4	97	97
VS	% of TS	99.4	94	94
C fossil	% of TS	0.2	0	77.61
C biogenic	% of TS	50.0	77.61	0
Reference		Boonmee et al., 2016; Pradhan et al., 2010; Hermann et al., 2010; Krause and Townsend., 2016; Kolstad et al., 2012	Götze et al., 2016 (Modified C)	Götze et al., 2016

TS: total solids, VS: volatile solids.

The plastic packaging for vegetables can lead to a higher amount of vegetable food waste in the EoL technologies analysed. The amount in kg of vegetables that can be packed with 100 m^2 of plastic packaging varies a lot depending on the type of vegetable packed. E.g. there will be a large difference between the amount (weight) comparing potatoes and salad which could be wrapped in the same amount of packaging. It is assumed that 300 g of vegetables need 0.12 m² packaging based on Manolopoulou et al. (2012), which is assumed to cover an average for all vegetables.

A study performed by FAO in 2011 showed that retailers waste 10 % of vegetables and consumers have a 19 % vegetable waste rate. Therefore, it can be estimated that 16 % of the packed vegetables will directly end up in the waste stream. The amount of wasted vegetables per kg plastic is hence estimated to be 15 kg vegetables/kg plastic[69], as both types have the same weight per FU.

[69] The vegetables are calculated to have a density that is 26.7 times higher than plastic. The vegetable waste that is thrown out per 0.12 m^2 plastic packaging film holding 100 grams of vegetables is 4.3 kg/kg plastic. It is estimated that a packaging film at 0.12 m^2 on average holds 350 grams of vegetables and thereby the average amount of vegetables thrown out per kg of plastic is 15 kg.

3 Life cycle impact assessment results and interpretation

3.1 Cradle-to-user environmental impacts of the Bio-Based systems

3.1.1 Baseline system Bio-Based PLA

Table 182 and Figure 158 show the cradle-to-user environmental impacts of the PLA films for the baseline product system.

Table 182. Cradle-to-user environmental impacts of 1 kg and 100 m^2 PLA films, 16 PEFCR impact categories + NREU (results without the effects of land use changes).

Impact category	Unit	PLA baseline (1 FU of films)	PLA baseline (1 kg of films)
Climate change	kg CO_2 eq.	5.11	1.64
Ozone depletion	kg CFC-11eq	5.61E-07	1.80E-07
Human toxicity, non-cancer effects	CTUh	2.27E-06	7.27E-07
Human toxicity, cancer effects	CTUh	5.93E-07	1.90E-07
Particulate matter	kg PM2.5eq	4.23E-03	1.35E-03
Ionizing radiation HH	kBq U235eq	0.77	0.25
Photochemical ozone formation	kg NMVOCeq	3.96E-02	1.27E-02
Acidification	molc H+eq	6.26E-02	2.01E-02
Terrestrial eutrophication	molc Neq	1.62E-01	5.18E-02
Freshwater eutrophication	kg Peq	5.67E-04	1.82E-04
Marine eutrophication	kg Neq	2.68E-02	8.59E-03
Freshwater ecotoxicity	CTUe	48.22	15.45
Land use	kg C deficit	63.61	20.39
Water use	m3	6.03	1.93
Abiotic depletion	kg Sbeq	2.50E-05	8.00E-06
Abiotic depletion (fossil fuels)	MJ	136.02	43.60
NREU	MJ	160.50	51.44

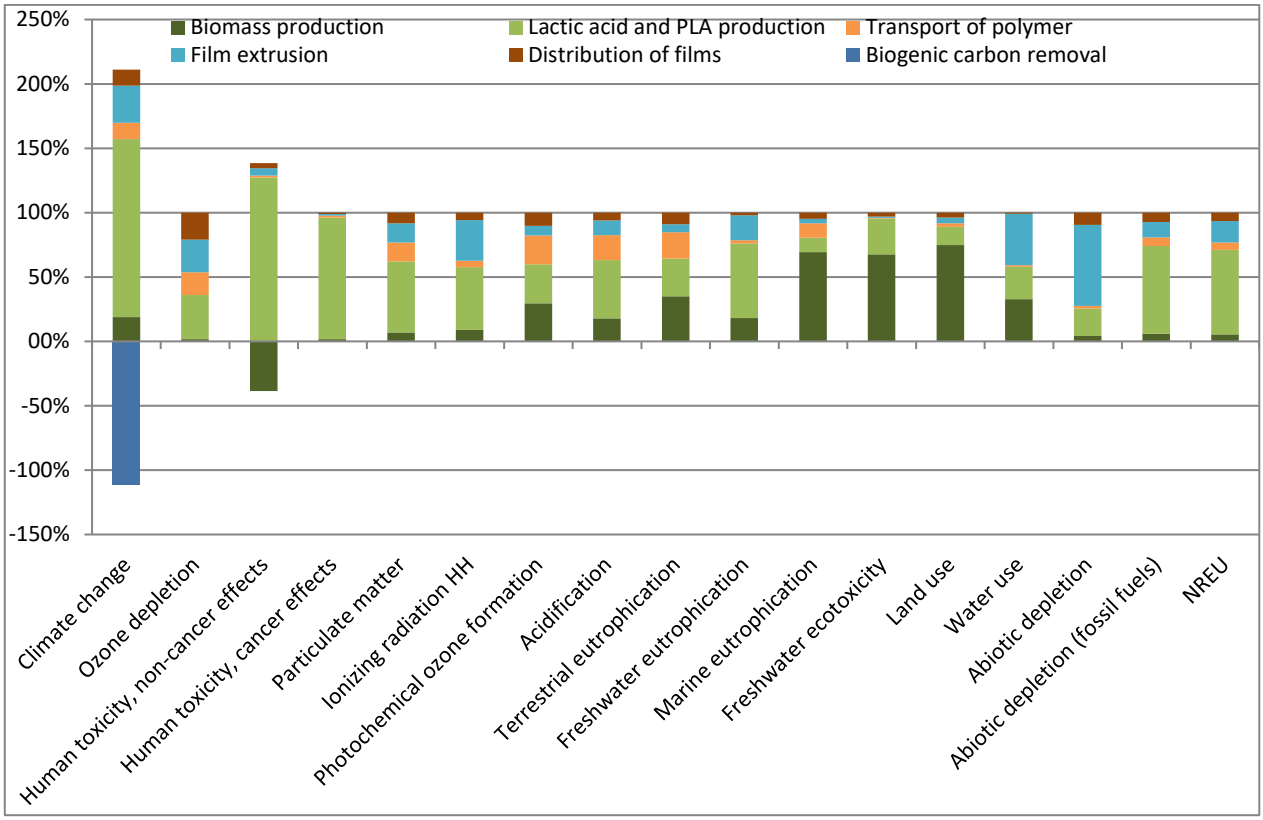


Figure 158. Breakdown of the cradle-to-user environmental impact of packaging films made from PLA (baseline).

Based on the results from cradle-to-user, the following considerations can be highlighted for the impact of each life-cycle step:

- The biomass production (US maize and Thai sugarcane) is the dominant (more than 60 % of the cradle-to-user environmental impact) source of environmental burden for marine eutrophication, freshwater ecotoxicity and land use. The negative impact on human toxicity (non-cancer effects) is a consequence of Thai sugarcane that absorbs zinc from the soil (more details are provided in the interpretation of this impact category behind). The cultivation step has a negligible impact on ozone depletion, human toxicity-cancer effects, abiotic depletion and abiotic depletion-fossil fuels.

- The lactic acid and PLA production among the five identified life cycle stages of the breakdown, is the main cause of climate change, human toxicity (both cancer and non-cancer effects), particulate matter, ionization radiation HH, acidification, freshwater eutrophication, abiotic depletion and abiotic depletion (fossil fuels), and NREU. Instead, it has a minor impact (about 10 %) to marine eutrophication.

- The transport of PLA granulates -from Nebraska or Rayong to European plastics companies- has a significant impact only for ozone depletion, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication and marine eutrophication. It is negligible for human toxicity (both cancer and non-cancer effects), freshwater eutrophication and ecotoxicity, land use and water use. For climate change and abiotic depletion of fossil fuels (NREU), the transportation of the PLA granulates plays also a very minor role for climate change and NREU.

- The film extrusion of PLA plays an important role in all the impact categories. This is mainly caused by the use of electricity. It represents a very significant environmental burden of the chain in the following: ozone depletion, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication and marine eutrophication.

- The final distribution of packaging films has almost the same impact as the transportation of granulates, with exclusion of abiotic depletion where it has an about triple impact.

Based on the cradle-to-grave weighted results (see section 3.4 of this case study chapter), the most important impact categories (that cover 80 % of the total impacts) are summarised as the following:

- If toxicity impacts are excluded, six categories cover about the 80 % of the total impacts. These are climate change, abiotic depletion (fossil fuels), acidification, freshwater eutrophication, particulate matter and photochemical ozone formation.

- If toxicity impacts are included in the weighting: all three toxicity categories appear to have significant contribution: human toxicity (cancer effects), freshwater ecotoxicity and human toxicity (non-cancer effects).

The detailed interpretations for the main impact categories and other relevant ones, are provided in the following section for cradle-to-user impacts. According to the PEFCR guidance (version 6.3) the impact categories that cumulatively contribute to 80 % of the total environmental impact (excluding toxicity related impact categories) have to be interpreted in detail. In this case, as aforementioned, this leads to a detailed interpretation of six impact categories (climate change, abiotic depletion of fossil fuels, acidification, freshwater eutrophication, particulate matter and photochemical ozone formation).

- **Climate change.** The climate change impact per FU of PLA has been estimated to be 5.11 kg CO_2 eq. The biogenic carbon removal is also shown, and it is derived from the chemical composition of PLA which has 50 % carbon content by weight (this amounts to 1.82 kg CO_2 eq./kg PLA). The considered life-cycle stages contribute to this value as show in Figure 159. The climate change impact of PLA films comes from the production of lactic acid from sugar/starch (39 %), from lactide production from lactic acid (17 %), from film extrusion (15 5), from biomass cultivation (9 %) and from the transportation steps (12 %, of which pellets (6 %) and packaging films (6 %)). The most important activities which lead to the climate change impact in lactic acid production are the production and use of fuels (e.g. the natural gas burnt for producing steam), electricity and chemicals (mainly lime and sulfuric acid). Regarding lactide production, only the use of natural gas and electricity apply. The GHG emissions due to lactic acid and PLA production are mainly carbon dioxide of fossil origins (87 %).

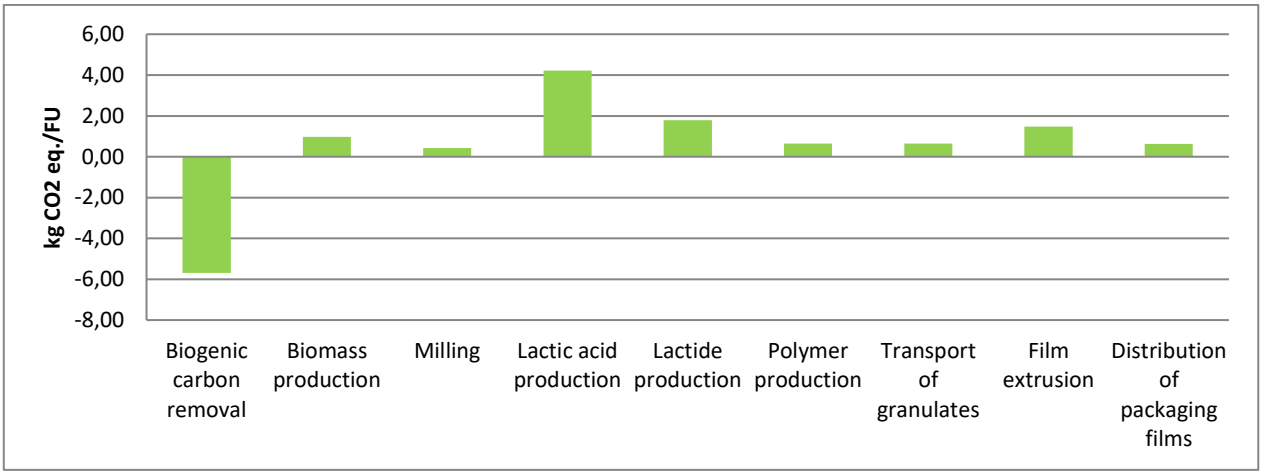


Figure 159. Cradle-to-user breakdown of climate change impacts for PLA(baseline) packaging films on functional unit basis (for both PLA and PP films: 3.12 kg).

As regards the GHG emissions of film extrusion, they are primarily (70 %) the result of electricity consumption (Electricity, medium voltage {Europe without Switzerland} of Ecoinvent 3.3) while another 12 % is a consequence of the plastic waste which is lost during processing. The emissions of European mix electricity impacting the most on climate change are carbon dioxide of fossil origins (about 90 %) and methane of fossil origins (9 %).

- **Abiotic depletion (fossil fuels) and NREU.** NREU (and strictly linked abiotic depletion related to fossil resources) is mainly caused by lactic acid and PLA production (68 %). The 61 % of this last impact is caused by the fermentation process which converts the starch/sugar into lactic acid. The other sub-steps of the lactic acid and PLA production process account for the following percentages of its impact: 27 % the conversion of lactic acid into lactide, 4 % the conversion of biomass into dextrose/sucrose (milling) and 9 % the polymerization. The biomass cultivation impacts only accounts for 3 % while the transportation only accounts for the 12 %.

- **Acidification.** The life-cycle stage with the highest contribution (45 %) is the production of lactic acid and PLA. This is almost entirely due to the fermentation of the starch/sugar into lactic acid while starch/sugar milling and lactide production have minor impacts. The acidification caused by the lactic acid production is mainly due to the emissions of Sulphur and nitrogen oxides caused by the production of the chemicals used during the conversion process. The 20 % of the impact is associated with the film extrusion process (for the 81 % caused by the electricity used). The acidification caused by the production of the electricity used in film extrusion is due to emissions of sulphur dioxide (77 %) and nitrogen oxides (21 %). The 20 % of the acidification is caused by the transportation of the polymer (mainly Sulphur emissions caused by combustion of diesel burnt) while only 6 % is caused by the transportation of the final product. The 18 % of the acidification is due to the biomass production. During the cultivation stage, the direct emissions due to fertiliser use account for 59 %, while another 10 % are direct emissions from the use of manure. The indirect emissions are 25 % and are mainly due to the production of the electricity used and for diesel burnt in machinery.

- **Freshwater eutrophication.** Freshwater eutrophication is mainly generated during lactic acid and PLA production (58 %). This is followed by film extrusion (19 %) and biomass cultivation (18 %). All the emissions that cause freshwater eutrophication during PLA production (incl. biomass cultivation) are phosphate (60 %) and phosphorous (40 %) released to water and soil. The majority (90 %) of freshwater

eutrophication related to film extrusion is linked to electricity production: phosphate emissions to water accounts for 98 %.

- **Particulate matter.** For particulate matter, the lactic acid and PLA production is the step with the major cradle-to-user impact (55 %). The particulate emissions of this life-cycle stage come almost entirely from the lactic acid production while the lactide production and polymerisation step have a very minor contribution. The particulate matter of lactic acid and PLA unit process is released as particulates <2.5 % um (70 %), sulphur dioxide (26 %) and particulates <10 um (5 %). As regards the sugar-milling of PLA from Thai sugarcane, the bagasse burning can be identified as the main source of particulates. The second major contributor to particulate matter release is the film extrusion stage which causes 16 % of the total. The 62 % of the particulate emissions of film extrusion are a consequence of the production of the electricity used. The particulate matter caused by electricity production is 55 % due to sulphur dioxide emissions, 42 % particulates <2.5 um and nitrogen oxides 3 %. The two transportation steps are also relevant emitters of particulates and they account for a 23 % share. As regards the cultivation of maize and sugarcane, it is responsible for only 7 %.

- **Photochemical ozone formation.** Photochemical ozone formation is generated mainly during three life cycle stages: lactic acid and PLA production (30 %), biomass production (30 %) and the transportation of polymer (22 %). The photochemical ozone formation related to the lactic acid and PLA production is mainly caused by emissions of nitrogen oxides (59 %), NMVOC (29 %) and sulphur dioxide (9 %). The photochemical ozone formation caused by the biomass production is instead mainly due to emissions of nitrogen oxides (81 %). Film extrusion accounts for another 8 % and 88 % is due to the production of the electricity used.

Additionally, several impact categories that are considered relevant for this case study are also assessed. In this case, the ones selected are terrestrial eutrophication where the biomass production is a significant contributor, and the toxicity impact categories due to the peculiar results that are obtained e.g. the negative impact of biomass production in human toxicity without cancer effects.

- **Terrestrial eutrophication.** The feedstock production is the step of the chain that contributes the most (35 %) to this impact category. The majority (83.5 %) of the terrestrial eutrophication caused by maize cultivation is a consequence of direct emissions occurring during this stage. Almost 100 % of these emissions are ammonia released to air due to the use of fertilisers (84.3 %) and manure (15.7 %). The second relevant contribution is caused by the indirect emissions from the production of the electricity used for maize cultivation (10.7 %). As regards the Thai sugarcane cultivation, the direct emissions account instead for 77 %, while the indirect emissions caused by the diesel used accounts for 12 %. The conversion step (biomass feedstock into PLA) is the second origin of terrestrial eutrophication with a 29 % share. Among the different steps involved in the conversion process, the conversion into lactic acid is the one with the highest impact which is estimated to be about 85 % of the total. The other steps contribute as follows: the conversion into lactide (8 %), the polymerisation for (4 %) and milling (3 %). The transportation of granulates has also a relevant share (21 %) while the distribution of the packaging films has only half of it (9 %). Film extrusion represents the 6 % of this impact and this is caused for the most part (74 %) by the electricity used.

- **Human toxicity, non-cancer effects and cancer effects, plus freshwater ecotoxicity**. Human toxicity without cancer effects is almost totally produced during lactic acid and PLA production (about 90 %). This is caused mainly by emissions of zinc to soil (75 %) and mercury (7 %) and zinc (5 %) to air. A negative impact during the

biomass production is observed in the figure. This is almost entirely caused by the zinc absorbed from the soil by sugarcane. Heavy metals are often added to synthetic fertilisers to facilitate the absorption of the nutrients for crops. They are also often found in animal manure (especially zinc).[70] The sugarcane cultivation in Thailand uses both synthetic fertilisers and animal manure (Durlinger et al., 2017). These metals are either leached into the soil or absorbed by the biomass. When there is more zinc absorbed by the biomass than the leached, a negative emission appears in the LCI model (Durlinger et al., 2017). However, the zinc absorbed by the biomass will eventually end up in the environment if the biomass is burned for energy (e.g. bagasse), returned to the soil (residue distillage applying to the soil) or removed from the production lines and ended up in waste (water) treatments. These zinc emissions could potentially explain the high contribution of zinc during the lactic acid and PLA production. [71] Human toxicity with cancer effects is also caused almost entirely by lactic acid and PLA production (94 % of the total cradle-to-user impact). This is mainly caused by emissions of chromium to water (50 %), to air (20 %) and to soil (20 %). Freshwater ecotoxicity comes instead more from biomass production (68 %) compared to lactic acid and PLA production (28 %). The emissions that cause freshwater ecotoxicity are mainly: atrazine (herbicide) to soil (31 %), chlorpyrifos (pesticide) to soil (28 %), chlorpyrifos to water (18 %) and metolachlor (herbicide) to soil (11 %).

3.1.2 Cradle-to-user environmental impacts of bio-based alternative system 1. PLA from European maize

In this section, the impact assessment of the alternative 1 PLA from European maize (PLA, EU maize)) is discussed. Table 183 shows the results of this fictional route of PLA made from EU maize (see description in Section 2.1.2) compared with the bio-based baseline system (results described in Section 3.1.1). It can be seen that by switching the feedstock of PLA from US maize to locally sourced EU maize, out of 16 impact categories, 12 categories have increased impacts. The four impact categories with decreased impacts (i.e. ozone layer depletion, particulate matter, photochemical ozone formation and abiotic depletion (mineral and metal)) reflect the benefits from the reduced length of logistics chains.

The breakdown of the impacts based on life cycle stages reveals that most of these increased impacts are associated with the mixed EU maize cultivation, Figure 160. The reduced impacts of (transoceanic) transportation are apparently not sufficient to compensate for the increased impacts originating from the EU maize cultivation. One of the main reasons is the lower yield of European maize (7.23 ton/ha/yr for maize 15 % moisture estimated from Agri-footprint 3) compared to maize cultivated in Nebraska/Iowa (10.28 ton/ha/yr for maize with 15 % moisture by weight (Vink et al., 2015)). A higher yield corresponds to a more efficient use of land and consequently a lower use of resources per kg of crop e.g. fertilisers. Nevertheless, it should be noticed that the modelled maize represents the European average production. The productivity of maize varies widely among the European countries ranging from 4.06 (Romania) to 11.06 ton/ha/yr (Spain). In consequence, significantly different results would be obtained if e.g. only the Spanish production were considered[72].

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[70] The amount of zinc and copper in pig manure are very high because they are often added in the feed for animal health reasons (Durlinger et al., 2017).

[71] Due to lack of transparency of the industrial data, the exact zinc balance through the entire life cycle cannot be fully reconstructed.

[72] For the remaining EU countries: 9.88 ton/ha/yr in Germany, 9.26 ton/ha/yr in Italy, 9.22 ton/ha/yr in France, 6.67 ton/ha/yr in Poland, 6.04 ton/ha/yr in Hungary and 5.91 in Bulgaria. Based on FAO (2016).

Table 183. Cradle-to-user environmental impacts (16 PEFCR categories + NREU) per functional unit -PLA films made from European maize, compared with the results of the baseline PLA films.

Impact category	Unit	PLA based on EU maize	PLA baseline	Increase (+) or decrease (-) of impacts
Climate change	kg CO_2 eq	7.09	5.11	39 %
Ozone depletion	kg CFC-11eq	4.31E-07	5.61E-07	-23 %
Human toxicity, non-cancer effects	CTUh	4.52E-06	2.27E-06	99 %
Human toxicity, cancer effects	CTUh	8.94E-07	5.93E-07	51 %
Particulate matter	kg PM2.5eq	4.10E-03	4.23E-03	-3 %
Ionizing radiation HH	kBq U235eq	0.99	0.77	28 %
Photochemical ozone formation	kg NMVOCeq	2.81E-02	3.96E-02	-29 %
Acidification	molc H+eq	9.03E-02	6.26E-02	44 %
Terrestrial eutrophication	molc Neq	2.95E-01	1.62E-01	82 %
Freshwater eutrophication	kg Peq	9.70E-04	5.67E-04	71 %
Marine eutrophication	kg Neq	5.46E-02	2.68E-02	104 %
Freshwater ecotoxicity	CTUe	82.22	48.22	71 %
Land use	kg C deficit	79.46	63.61	25 %
Water use	m3	30.38	6.03	404 %
Abiotic depletion	kg Sbeq	2.14E-05	2.50E-05	-14 %
Abiotic depletion (fossil fuels)	MJ	148.66	136.02	9 %
NREU	MJ	183.71	160.50	14 %

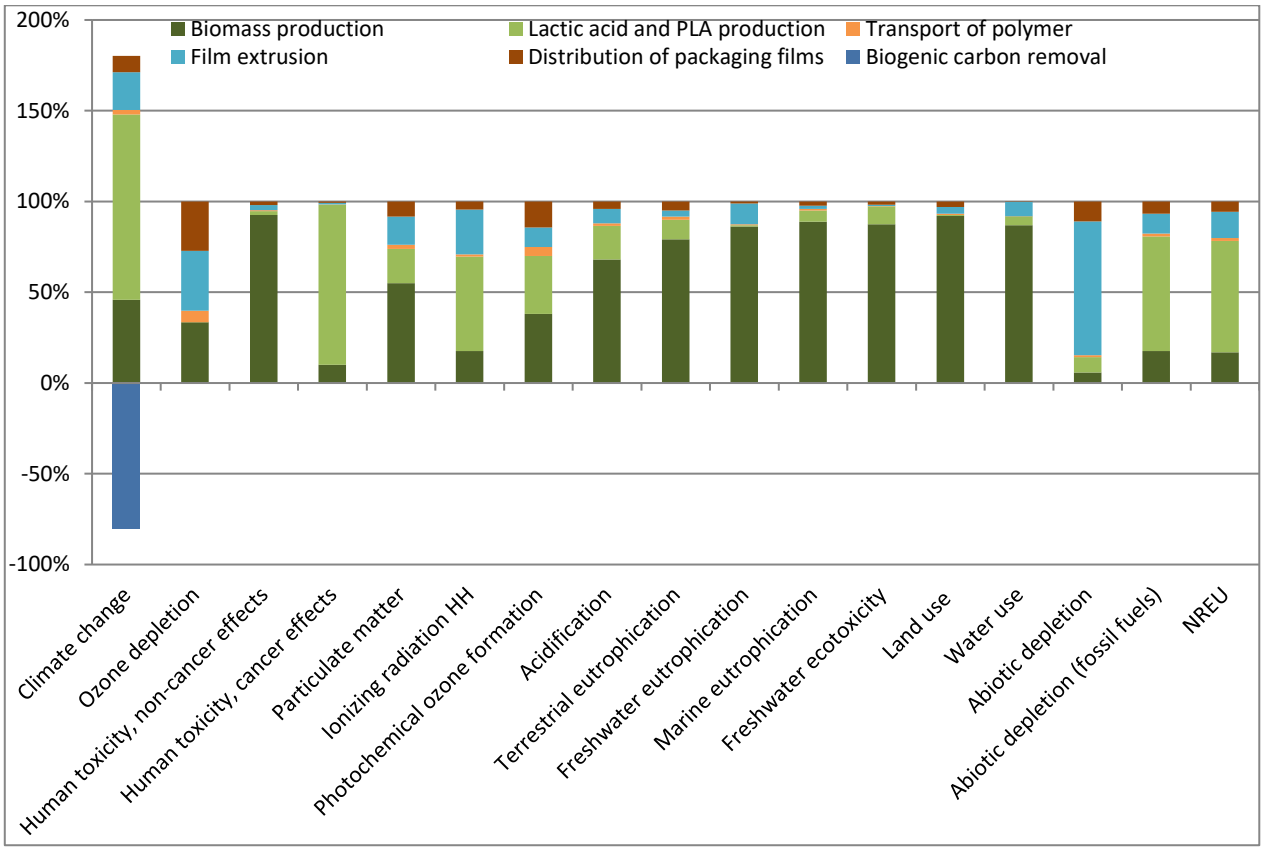


Figure 160. Breakdown of cradle-to-user results for alternative 1 (Bio-based PLA from European maize).

The impact categories showing major differences (e.g. more than ±30 % changes in impacts) between this alternative route and the baseline route in Table 183 are further interpreted below.

- **Climate change.** The climate change impact is assessed to be 7.09 kg CO_2 eq. on a functional unit basis and therefore about 39 % higher compared to the baseline. The climate change impacts of this product system come from the maize cultivation (25 %), from the conversion of maize into PLA (57 %), the 1 % from the transportation of the pellets, from the manufacturing of the packaging films (12 %) and from the final distribution (5 %). In this case the contribution of the material is estimated to be higher because of the more carbon intensive maize production (0.63 kg CO_2 eq. for average European maize) compared to Nebraska and Iowa (0.17 kg CO_2 eq./kg maize (based on Vink et al., 2015)). Among the EU modelled countries, the ones with the most carbon intensive maize production are Romania (0.91 kg CO_2 eq./kg maize) and Poland (0.75 kg CO_2 eq./kg maize). These last two countries cover 17 % and 6 % of the European maize production which is modelled: in terms of climate change impact, they represent respectively 26 % and 8 % of the total. The carbon intensities of maize production in other maize cultivation countries like Germany and France are approximately half of that in Romania, and comparable to the average production of maize in the USA. Moreover, the maize from Iowa/Nebraska which is used by NatureWorks for PLA production has an impact which is 30 % lower on climate change compared to average cultivation in the USA. At emissions level, the maize cultivation in Poland and Romania produces 2-3 times more carbon dioxide of fossil origins (about 60 % of the climate change impact) and 2-4 times more emissions of methane (about 6 % of the climate

change impact) compared to France. The emissions of dinitrogen monoxides (20-40 % of the impact of climate change) are instead released in more similar quantities. The carbon dioxide emissions of fossil origins for the most part (70-80 %) come from diesel burnt in agricultural machinery; 40-60 % of the emissions of methane come from the production of the electricity used and 20-30 % comes from the diesel burnt in machinery. The climate change impact of the lactic acid and PLA production unit process does not change significantly (less than 2 %).

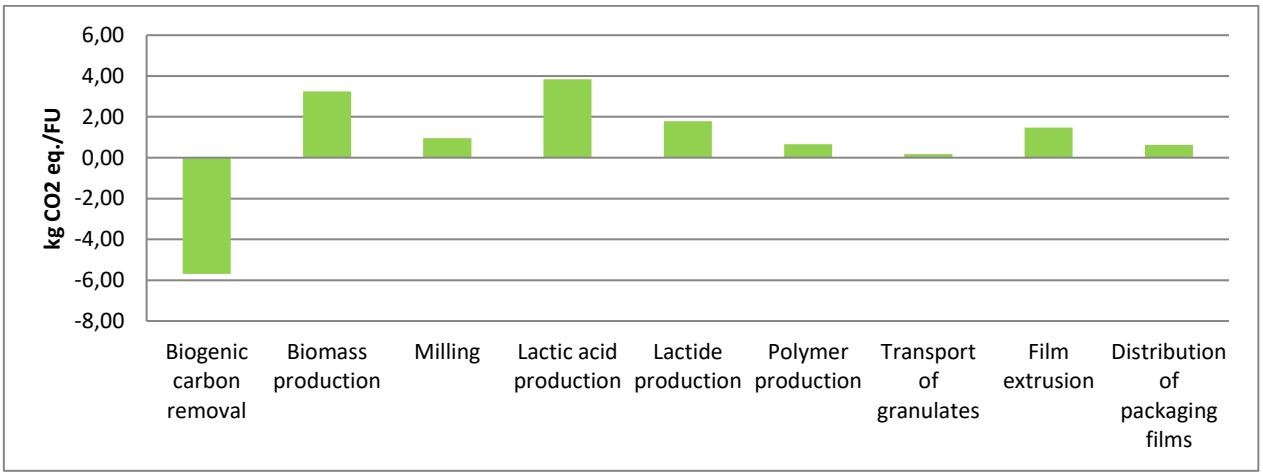


Figure 161. Cradle-to-user breakdown of climate change impacts for PLA (EU maize) packaging films on functional unit basis.

- **Particulate matter.** The European production of maize emits more soot compared to the US. Per kg of PLA, 0.53 g PM2.5 eq are emitted in average US (maize cultivated in Nebraska and Iowa also has better performance compared to the US average) for the production of maize. A corresponding value for European maize would be 0.77 g PM2.5 eq. Nevertheless, the reduction of transportation distance for PLA polymer leads to an overall saving of 3 % of the particulate matter per packaging film. Another reason for this is that the sugarcane cultivation has lower particulate matter emitted per kg of PLA compared to US maize.

- **Photochemical ozone formation.** In the alternative system 1, the emissions per packaging film of NMVOC decrease by 29 % compared to the baseline mainly thanks to the decrease of distance travelled by the granulates. Another reason is that the EU maize cultivation emits 50 % less NMVOC compared to Nebraska/Iowa maize.

- **Acidification.** Acidification per PLA film is in total 44 % higher when PLA is produced from European maize instead that from the mixed Nebraska/Iowa maize and Thai sugarcane. This is mainly because the acidification produced from the European maize cultivation is 79 % higher compared to the US maize used for PLA production (baseline). The acidification caused by European maize is 70 % caused by emissions of ammonia, 18 % by emissions of sulphur dioxide and 8 % by nitrogen oxides. The emissions of ammonia are mainly direct emissions due to the use of fertilisers (65 %) and manure (25 %) while the emissions of sulphur dioxide are mainly indirect emissions deriving from the use of electricity (50 %) and diesel burnt in machinery (30 %).

- **Terrestrial eutrophication.** For the alternative system 1, the feedstock cultivation is responsible for 79 % of the impact while 11 % is caused by the conversion of maize into PLA. Compared to the baseline, terrestrial eutrophication increases by 79 % per film. The main reasons behind this is that the terrestrial eutrophication caused by

European maize is the 70 % higher compared to Nebraska/Iowa cultivation. The second cause is that per kg of PLA, the sugarcane cultivation has a half of the impact compared to US maize used for PLA (baseline).

- **Marine eutrophication.** Marine eutrophication increases by 104 % when PLA is produced from European maize, compared to importing it from US. This is because the feedstock cultivation has a share of 89 % of the total impact and the European maize cultivation release more nitrogen substances compared to US. It is estimated that 1 Kg of maize in Europe causes 6 times higher amounts of kg N eq released compared to maize of Nebraska/Iowa. Marine eutrophication caused by European maize cultivation is due to emissions of nitrate to water (90 %) which is caused by the use of fertilisers (60 %), nitrate emissions from crop residues (30 %) and due to the use of manure (7 %).

- **Water use.** In the alternative system 1, the total water use is four times higher compared to the baseline. The water use from the biomass cultivation (maize from the US and sugarcane from Thailand) in the baseline analysis is 6 m3 per functional unit whereas in this alternative route the EU mixed maize requires 30 m3. There are two reasons for this significant increase of water use. Firstly, the PLA is produced only from maize and no more from the mix maize/sugarcane and the amount of Thai sugarcane needed per kg of PLA required the 60 % less m3 of water compared to the amount of Nebraska/Iowa maize needed for 1 kg of PLA. Secondly, average European maize requires 10 times more water compared to maize cultivated in Nebraska/Iowa. It should be noticed that the water use accounted for with the characterization factors of the AWARE methodology for maize production are largely variable in Europe. For example, the water use impact of maize cultivation is 0.02 m3 in Germany and 31.7 m3 in Spain per kg of maize, although the absolute water withdrawals for irrigation in these two countries are not so substantial because in Spain water is a more constrained resource than in Germany.

- **Land use.** The majority (92 %) of the land use impact is caused by the feedstock production (European maize). For European maize the land use accounted with carbon deficit ranging between 8.9 (Spain) and 24.1 (Romania) kg C deficit per kg of maize: the land use is largely different depending on the country where the maize is produced. For the European average maize, it results 14.0 kg C deficit per kg of maize. The land use for the PLA of the alternative system 1 is 25 % higher compared to the PLA of the baseline product system. The main reason behind is that Nebraska/Iowa maize has a carbon deficit of 10.7 kg C, therefore 23 % lower compared to average European maize.

3.1.3 Cradle-to-user environmental impacts of bio-based alternative system 2: Bio-based PP

A detailed interpretation of the environmental impact of the second alternative product system is reported here. It can be seen from Table 184 that bio-based PP offers substantial impact reduction in nearly all 16 impact categories (the exception is ozone layer depletion, for which the impact decrease is minor).

Table 184. Cradle-to-user environmental impacts (16 PEFCR categories + NREU) of per functional unit bio-based PP films made from used cooking oil, compared with the results of the baseline PLA films.

Impact category	Unit	bio-based PP	PLA baseline	Increase (+) or decrease (-) in impacts
Climate change	kg CO_2 eq.	-5.21	5.11	-202 %

Ozone depletion	kg CFC-11eq	6.02E-07	5.61E-07	7 %
Human toxicity, non-cancer effects	CTUh	3.00E-07	2.27E-06	-87 %
Human toxicity, cancer effects	CTUh	3.34E-08	5.93E-07	-94 %
Particulate matter	kg PM2.5eq	1.61E-03	4.23E-03	-62 %
Ionizing radiation HH	kBq U235eq	4.88E-01	0.77	-36 %
Photochemical ozone formation	kg NMVOCeq	1.36E-02	3.36E-02	-66 %
Acidification	molc H+eq	2.02E-02	6.26E-02	-68 %
Terrestrial eutrophication	molc Neq	4.52E-02	1.62E-01	-72 %
Freshwater eutrophication	kg Peq	1.71E-04	5.67E-04	-70 %
Marine eutrophication	kg Neq	4.11E-03	2.68E-02	-85 %
Freshwater ecotoxicity	CTUe	2.89E+00	48.22	-94 %
Land use	kg C deficit	8.97	63.61	-86 %
Water use	m3	4.74	7.90	-21 %
Abiotic depletion	kg Sbeq	1.90E-05	2.50E-05	-24 %
Abiotic depletion (fossil fuels)	MJ	58.62	136.02	-57 %
NREU	MJ	74.00	160.50	-54 %

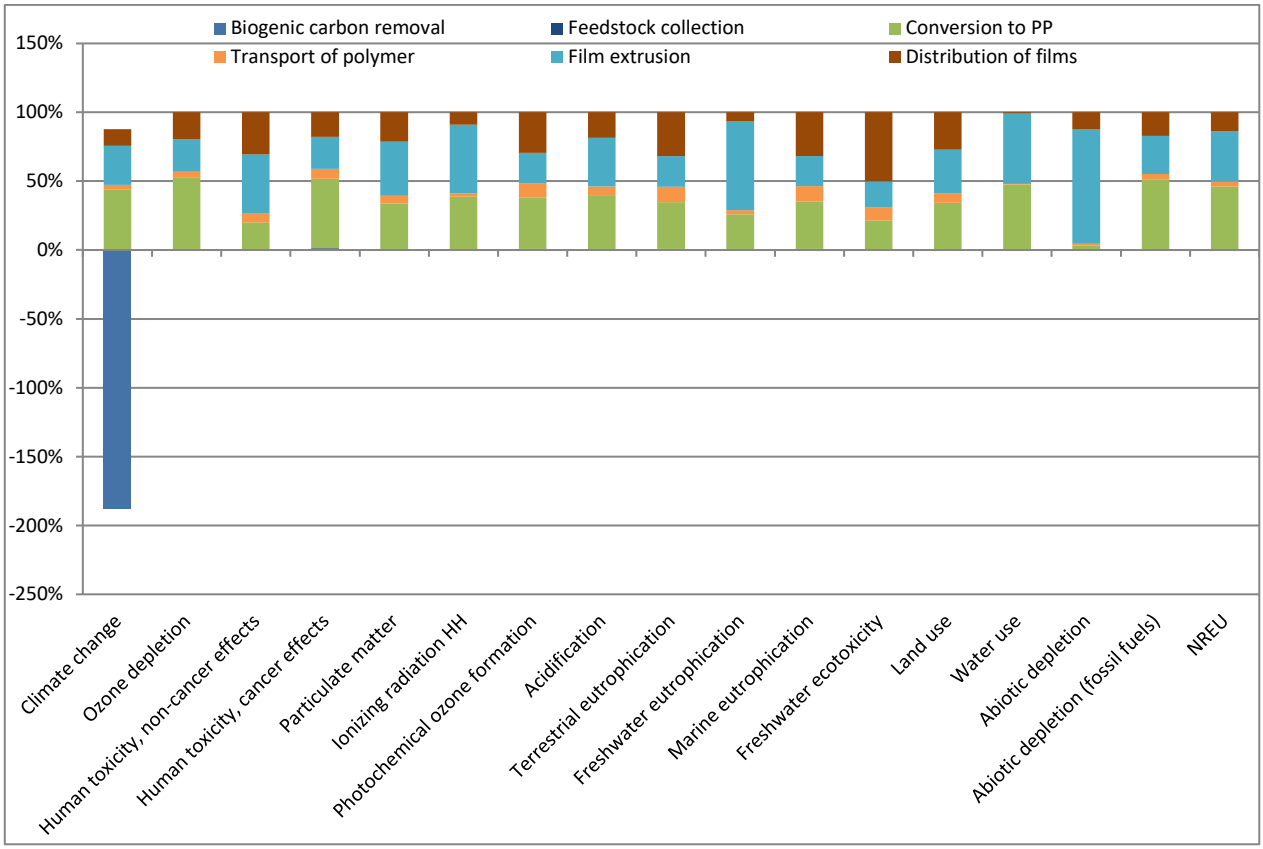


Figure 162. Breakdown of cradle-to-user results for alternative 2 (Bio-based PP from UCO).

From Figure 161 and Figure 162 -showing the breakdown of the environmental impact in the life cycle stages- the following considerations emerge:

- The feedstock collection represents a negligible environmental impact. This is true if the used cooking oil will be locally sourced as assumed.
- The conversion of UCO into polypropylene along with the film extrusion stage are the two major contributors to the environmental burden of this product system and with similar shares. For freshwater eutrophication and abiotic depletion only, the film extrusion process is significantly more impacting.
- The transportation of the polymer which is assumed to happen inside Europe contributes on average 5 % in all the assessed impact categories
- The distribution of the packaging films is negligible only for water use while it is the major environmental hotspot for freshwater ecotoxicity and abiotic depletion.

With a deeper level of detail and for the main impact categories, the main causes of environmental impact are here detailed:

- **Climate change.** The total climate change impact is caused by two main life cycle stages: the conversion of UCO into PP (50 %) and the film extrusion process (32 %). The rest of the climate change impact is caused mainly by the distribution of the packaging films (14 %). Regarding the conversion into PP, the impact is caused by the steam cracking (51 %), by polymerisation (23 %) and by hydrotreatment (21 %) (including transportation of bio-naphtha). The impact of steam cracking is caused primarily (85 %) by emissions of carbon dioxide of fossil origins (these are 30 % direct

emissions, 54 % indirect emissions for the generation of the steam used and 8 % indirect emissions due to LPG production). The climate change impact of hydrotreatment is caused for the most part (86 %) by the production of the hydrogen used. The impact of film extrusion is caused 70 % by the production of the electricity used and 10 % from heat used.

- **Particulate matter.** The emissions of particulate matter are caused by film extrusion (39 %), conversion into PP (33 %) and distribution of films (21 %). The impact of film extrusion comes from electricity used (62 %) and from heat used (11 %). The impact of the conversion stage is distributed as follows: 49 % caused by steam cracking, 28 % by polymerisation and 15 % by hydrotreatment (incl. transportation of naphtha). The impact of hydrotreatment comes 65 % from the production of the hydrogen used and 22 % from the transportation of the naphtha to the stream cracking unit.

- **Photochemical ozone formation.** The impact in terms of photochemical ozone formation is divided as follows: 37 % the conversion into PP, 22 % from the film extrusion and 29 % from the distribution of the packaging films. The impact of the conversion process is 37 % caused by the polymerisation step, 28 % from hydrotreatment and another 28 % from steam cracking. The impact of hydrotreatment is primarily (86 %) due to the production of the hydrogen used. For the steam cracking, 57 % of photochemical ozone formation is a consequence of the production of the steam used, while 42 % comes from the production of the LPG.

- **Acidification.** The contributions to acidification of each life cycle stage are as follows: 39 % from conversion into PP, 35 % from film extrusion, 19 % from distribution of packaging films, 6 % from transport of PP polymer and 1 % from the UCO collection. The majority (81 %) of the acidification caused by film extrusion is a consequence of the production of the electricity used while another 9 % comes from the production of the steam used. The impact of the conversion stage is mainly due to three sub-unit processes: steam cracking (39 %), polymerisation (35 %) and hydrotreatment (20 %). The impact of steam cracking is caused 69 % by the production of the steam used and 31 % by the production of the LPG used. The impact of hydrotreatment is caused mainly by the production of the hydrogen used (81 %).

- **Terrestrial eutrophication.** The life cycle stages that contribute the most are the conversion into PP (33 %), the film extrusion (22 %) and the distribution of the food packaging films (32 %). The transport of the PP polymer represents a share of 11 % while the UCO collection is less than 1 %. Among the processes which belong to the conversion stage, the main contributors are: hydrotreatment (37 %), polymerisation (30 %) and steam cracking (24 %). The majority (87 %) of the impact related to hydrotreatment is due to the production of the hydrogen used. The impact of the steam cracking is caused 63 % by the production of steam and 37 % by production of LPG burnt. Electricity and heat production represent respectively 74 % and 9 % of the environmental burden produced during film extrusion.

- **Marine eutrophication.** The shares of the life cycle impact breakdown for these impact categories are very similar to those related to terrestrial eutrophication. The life cycle stages that contribute the most are the conversion into PP (34 %), the film extrusion (22 %) and the distribution of the packaging films (32 %). The impact of the conversion stage is structured as follows: 37 % hydrotreatment (86 % due to the production of the hydrogen used), 24 % steam cracking (63 % due to steam production and 37 % due to LPG production) and 31 % polymerisation. The majority (97 %) of the impact of film extrusion is due to the production of the electricity used.

- **Land use.** Land use impacts are caused 35 % by the conversion to PP, 31 % by film extrusion, 27 % by the packaging films distribution and 7 % by the transportation of the PP polymer. The land use impact of the conversion step comes almost entirely (82 %) from the steam cracking unit process and this is caused 71 % by LPG production and 29 % by steam production.

- **Water use.** Two life cycle stages cover almost the total water use: the conversion to PP (47 %) and the film extrusion process (51 %). The polymerisation step is entirely responsible for the water use (almost 100 %) of the conversion step. Only 10% of the water use of film extrusion is caused by the electricity production while the other 91 % comes from direct use of water.

- **Abiotic depletion- fossil fuels.** The breakdown for this impact category is structured as follows: 51 % conversion into PP, 27 % film extrusion and 17 % distribution of the packaging films. The steam cracking (58 %) and the polymerisation (27 %) processes are the two main contributors of non-renewable resources used during the conversion stage while the hydrotreatment accounts for another 11 %. The impact of the steam cracking is due to steam production (52 %) and LPG production (47 %). The 73 % of the abiotic depletion-fossil fuels of the film extrusion stage is due to the electricity production while another 13 % is caused by the production of the steam used.

3.2 Cradle-to-user results of petrochemical reference systems

3.2.1 Reference 1 petrochemical PP

With the same level of breakdown showed for PLA packaging films, in this section the results for PP packaging films are collected. In Table 185 it is possible to identify the impact of each stage on the life cycle. Based on Chapter 6 Ranges for environmental impacts from production on fossil-based plastics, seven out of 17 impact categories were found suitable for petrochemical PP interpretation, namely: climate change, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication, abiotic depletion (fossil fuels) and NREU. Further details can be found in Table 35 where the environmental impact of 1 kg of PP and the proposed ranges of variation within each impact category are shown.

The main reasons for the exclusion of most of the assessed impact categories of PlasticsEurope ecoprofile for PP are the large differences identified among the impact assessment results provided by the main LCA databases, non-compliance with several ILCD requirements, and a lack of transparency in the allocation approach. Moreover, impact categories like marine and freshwater ecotoxicity are considered not reliable because there is no distinction within the PlasticsEurope dataset between emissions to fresh water and to seawater.

Table 185. Cradle-to-user environmental impacts (16 PEFCR categories + NREU) per functional unit for petrochemical PP.

Impact category	Unit	Petrochemical PP (ref.1)
Climate change	kg CO_2 eq.	7.76
Particulate matter	kg PM2.5eq	2.17E-03
Photochemical ozone formation	kg NMVOCeq	2.65E-02
Acidification	molc H+eq	2.91E-02
Terrestrial eutrophication	molc Neq	7.05E-02
Abiotic depletion (fossil fuels)	MJ	272.78
NREU	MJ	289.84

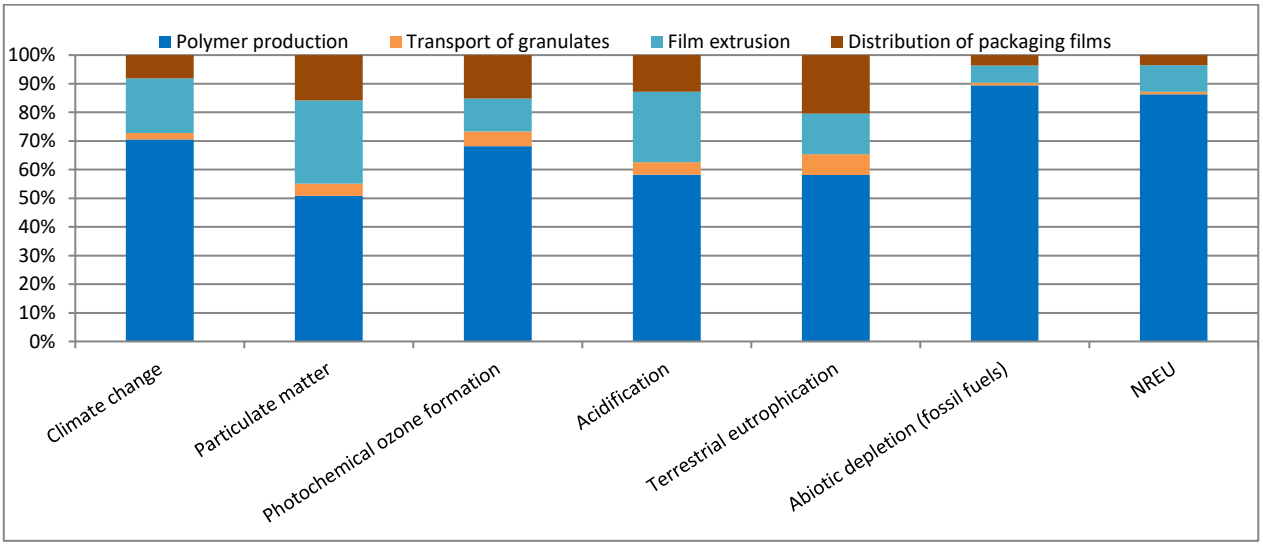


Figure 163. Breakdown of the cradle-to-user results for reference 1 (PP).

From Figure 163 it is noticed that:

- The production of PP granulates generates the highest impact in terms of global warming potential, photochemical ozone formation, terrestrial eutrophication, marine eutrophication, water use, abiotic depletion (fossil fuels) and NREU. Its impact is also very relevant for particulate matter and acidification
- The transport of granulates, assumed to occur inside Europe, does not make any relevant contribution.
- Film extrusion impacts relevantly in almost all of the impact categories therefore it can be perceived as the second source of environmental impact of PP packaging films.

- The distribution of PP packaging films also has a non-negligible impact except for water use, abiotic depletion (fossil fuels) and NREU.

3.3 End of life results and interpretation

The results for the EoL impact assessment modelled in EASETECH are presented in this paragraph. Firstly, the mass and energy flows of all product systems in all technologies are presented, secondly the LCIA results for each product system, thirdly a comparison between product systems, and lastly the EoL mix. In Annex 4 further results are presented, the contribution analysis is in Figure 286 ff. and the weighted results in Table 241 ff.

The LCIA results are shown both with and without food leftovers for the product systems, in order to observe the influence additional food leftovers, have on the results. The results solely on the plastic and its contamination are interesting to investigate, because the packaging would in some cases not include the leftover food, thus the food leftovers could enter a different waste stream.

3.3.1 Mass and energy flows

Table 186 Table 187 and Table 188 present the mass and energy flows, as outputs from EASTECH, for the three product systems presented. Note that these flows are presented with the vegetable waste. The results correspond to the circular footprint formula in PEFCR guidance draft version 6.3 (European Commission, 2017), and in addition the paragraph looks at the carbon flows. The flows correspond to the flows observed in the flow charts for each technology in the approach and methodology chapter of the full report, section 3.4 Average EoL technologies.

It is observed that 70 % of the plastic is recycled and 57 % is substituted in the recycling technology for all three product systems (calculated based on a sorting and technology efficiency as well as a market response, considering the decrease in quality of the plastic - see the methodology for further explanation). Looking at the total mass, the plastic and the vegetable waste, only 5 % of the mass is recycled and 4 % substituted. The vegetable waste that enters the recycling facility and ends up as reject is distributed 53 % to incineration and 47 % to landfilling. The energy recovery from rejects in Table 186 is both from incineration of rejected PLA, and incineration of vegetable waste and landfill gas energy recovery of vegetable waste. For the industrial composting of PLA 11 % of the mass input ends as compost with a content of 45 % of the fertiliser substitutions (N, P and K). In the anaerobic digestion 2 % of the total mass ends as compost, whereas 132 % of the mass input becomes digestate (due to the added water the weight increases), which in total contains 50 % of the nutrients (N, P and K). For the landfilling scenario, 80 % of the mass is stored for the PLA product system and 83 % of the mass for the non-degradable plastic systems.

Regarding energy, 20 % and 23 % of the energy is harvested in the incineration plant for the PLA product system and the PP product systems respectively. When landfilling, 12 % of the energy content is recovered for the PLA system, while 8 % of the energy is obtained from the PP. Incineration of rejects from the composting process for PLA and the recycling process for PP recovers 2 % of the energy. Anaerobically digesting the PLA yields 41 % of the energy in the biogas utilisation, and further 3 % for the incineration of rejects from the AD plant.

Looking at the carbon balance when landfilling films, 22 % of the biogenic carbon and 44 % of the fossil carbon is stored for the PLA product system, while 99.9 % of the carbon is stored in the landfill for the bio-based and petrochemical PP. In the compost from the composting facility 80 % of the biogenic carbon is recovered, and for the AD 43 % of the biogenic carbon is recovered from the compost and digestate together.

Table 186. Material and energy flow for EoL of 1 kg PLA with 100 g contamination and 15 kg of vegetable waste. Box letter refer to flow charts in the approach and methodology chapter (Figure 8 to Figure 13).

PLA including vegatable waste								
Technology	Box	Process	Mass kg	Bio carbon kg	Fossil carbon kg	Gas m^3 CH_4	Energy substitution MJ	Fertiliser substitution kg NPK
Material	I	Material input	1.61E+01	2.15E+00	1.93E-02		8.26E+01	1.19E-01
Recycling	O	Recycled plastic	7.70E-01	3.93E-01	0.00E+00			
	S	Substituted plastic	6.24E-01	3.18E-01	0.00E+00			
	R, E, H	Rejects	1.53E+01	1.76E+00	1.93E-02		1.15E+01	
Incineration w/wo energy recovery *	E, H	Energy production	1.61E+01	2.15E+00	1.93E-02		1.67E+01	
	R1	Fly Ash	4.72E-02	0.00E+00	0.00E+00			
	R2	Bottom ash	1.39E-01	2.15E-03	8.42E-06			
	D	Direct emissions		2.15E+00	1.93E-02			
Landfill	L	Leachate	2.59E+01	0.00E+00	0.00E+00			
	G, E	Landfill gas		1.67E+00		1.66E+00	9.43E+00	
	CS	Storage in landfill	1.29E+01	4.77E-01	8.42E-03			
Compost	O, S	Compost	1.75E+00	1.72E+00	8.00E-03			5.33E-02
	R, E, H	Rejects	1.18E-01	1.46E-01	4.15E-04		1.82E+00	
Anaerobic digestion	B, E1, H1, F	Biogas		1.66E+00		1.93E-01	6.51E+00	
	C, S	Compost	4.03E-01	4.03E-02	9.08E-04			6.96E-03
	D, S	Digestate	2.86E+01	6.54E-01	4.99E-03			4.32E-02
	R, E2, H2	Rejects	5.28E+00	3.06E-01	1.07E-04		5.01E+00	

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

Table 187. Material and energy flow for EoL of 1 kg bio-based PP with 100 g contamination and 15 kg of vegetable waste. Box letter refer to flow charts in the approach and methodology chapter (Figure 8 to Figure 13).

Bio-based PP including vegetable waste							
Technology	Box	Process	Mass kg	Biogenic carbon kg	Fossil carbon kg	Gas m^3 CH_4	Energy substitution MJ
Material	I	Material input	1.61E+01	2.47E+00	1.92E-02		1.03E+02
Recycling	O	Recycled plastic	7.70E-01	5.78E-01	0.00E+00		
	S	Substituted plastic	6.24E-01	4.68E-01	0.00E+00		
	R, E, H	Rejects	1.53E+01	1.89E+00	1.92E-02		3.74E+00
Incineration w/wo energy recovery *	E, H	Energy production	1.61E+01	2.47E+00	1.92E-02		1.25E+01
	R1	Fly Ash	6.33E-02	4.50E-04	0.00E+00		
	R2	Bottom ash	1.84E-01	2.46E-03	8.25E-06		
	D	Direct emissions		2.46E+00	1.91E-02		
Landfill	L	Leachate	2.67E+01	0.00E+00	0.00E+00		
	G, E	Landfill gas		1.46E+00		1.46E+00	0.00E+00
	CS	Storage in landfill	1.33E+01	1.01E+00	8.25E-03		

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

Table 188. Material and energy flow for EoL of 1 kg petrochemical PP with 100 g contamination and 15 kg of vegetable waste. Box letter refer to flow charts in the approach and methodology chapter (Figure 8 to Figure 13).

PP including vegetable waste							
Technology	Box	Process	Mass kg	Bio carbon kg	Fossil carbon kg	Gas m^3 CH_4	Energy substitution MJ
Material	I	Material input	1.61E+01	1.64E+00	8.45E-01		1.03E+02
Recycling	O	Recycled plastic	7.70E-01	0.00E+00	5.78E-01		
	S	Substituted plastic	6.24E-01	0.00E+00	4.68E-01		
	R, E, H	Rejects	1.53E+01	1.64E+00	2.67E-01		1.10E+02

Incineration w/wo energy recovery *	E, H	Energy production	1.61E+01	1.64E+00	8.45E-01		1.99E+02
	R1	Fly Ash	6.33E-02	0.00E+00	4.50E-04		
	R2	Bottom ash	1.84E-01	1.64E-03	8.34E-04		
	D	Direct emissions		1.64E+00	8.44E-01		
Landfill	L	Leachate	2.67E+01	0.00E+00	0.00E+00		
	G, E	Landfill gas		1.46E+00		1.46E+00	0.00E+00
	CS	Storage in landfill	1.33E+01	1.79E-01	8.34E-01		

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

3.3.2 Baseline: Bio-based PLA

The impact assessment results are presented per the EoL-reference flow, i.e. 1 kg plastic, 100 g of contamination and 15 kg of food leftovers for each of the EoL technologies (assuming 100 % disposal via that technology).

Based on the LCIA, only looking into the PLA, the preferred EoL technology for PLA is recycling, especially looking at the results without food leftovers. Table 189 presents the LCIA results for all impact categories for the PLA without food leftovers. The food packaging PLA will cause additional vegetable food waste in the same stream and results for the PLA with the additional vegetable food waste are presented in Table 189. For the results with food leftovers, the best performing EoL option is more unclear. Thus, these characterised results have been weighted, see section 3.2.2.7, and show that industrial composting performs best, secondly anaerobic digestion, thirdly incineration, and the least favoured EoL technology is landfilling.

Further details on the individual EoL options for the bio-based PLA packaging film are presented below, looking into specific impact categories and the contribution of impacts from different processes and elemental emissions.

Table 189. Total results of treating 1 kg PLA (incl. 100 g contamination and 15 kg vegetable food waste) for all impact categories. Highest impact within each category is red and lowest impact green. The intended EoL option is marked with a bold box.

Impact category	Unit	PLA incl. vegetable food waste				
		Recycling	Incineration	Landfilling	Industrial composting	Anaerobic digestion
Climate change - fossil	kg CO_2 eq	-6.95E-01	-3.52E-01	9.52E-02	1.16E+00	-2.78E+00
Climate change - biogenic	kg CO_2 eq	1.55E+01	7.73E+00	2.83E+01	-4.50E-01	4.79E+00
Climate change - total	kg CO_2 eq	1.48E+01	7.38E+00	2.83E+01	7.12E-01	2.01E+00
Ozone depletion	kg CFC-11 eq	6.06E-06	-6.62E-08	1.48E-05	2.36E-08	-2.40E-07
Human toxicity, non-cancer effects	CTUh	8.23E-08	1.03E-07	1.12E-08	3.32E-06	4.16E-06

Human toxicity, cancer effects	CTUh	2.25E-08	4.84E-08	-2.36E-09	-7.97E-08	-3.35E-07
Particulate matter	kg PM2.5 eq	-1.15E-04	1.87E-04	-2.01E-04	2.18E-04	-2.63E-04
Ionizing radiation HH	kBq U235 eq	3.09E-02	1.24E-01	-2.57E-02	6.74E-02	1.67E-01
Photochemical ozone formation	kg NMVOC eq	8.72E-03	1.48E-02	8.75E-03	3.17E-03	-1.11E-03
Acidification	molc H+ eq	6.67E-03	1.68E-02	-3.10E-05	9.31E-03	4.19E-02
Terrestrial eutrophication	molc N eq	3.54E-02	7.61E-02	1.42E-03	3.60E-02	1.88E-01
Freshwater eutrophication	kg P eq	1.69E-03	3.28E-03	-2.55E-05	2.17E-04	3.83E-04
Marine eutrophication	kg N eq	2.95E-03	5.99E-03	6.55E-04	7.68E-03	3.47E-02
Freshwater ecotoxicity	CTUe	8.70E-01	2.38E+00	1.91E-02	1.96E+00	-1.06E+00
Land use	kg C deficit	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	m³	3.02E-01	2.56E-02	-3.97E-01	1.32E+00	-1.32E+00
Abiotic depletion	kg Sb eq	-3.50E-07	-4.38E-07	-4.14E-07	-8.75E-07	-8.83E-06
Abiotic depletion (fossil fuels)	MJ	-3.37E+01	-9.52E+00	-2.11E+00	7.51E+00	-3.32E+01
NREU	MJ	-3.50E+01	-1.25E+01	-1.03E+00	4.15E+00	-3.63E+01

The significance of the colour scale is the same as for all the other product systems (Table 189 and Table 190). . This means that you can compare the colour horizontally across all tables, but only comparing between the same impact category.

3.3.2.1 Intended EoL technology: Industrial composting of PLA

The LCIA results from industrial composting of PLA with vegetable waste show that this EoL technology perfoms best in the impact category of climate change. For the impact categories of NREU and abiotic depletion this EoL performs the worst, as there is no energy recovery in industrial composting as there is in the other EoL technologies. The proportion of vegetable waste entering the composting facility is much higher than the plastic. Emissions from use on land have a big contribution (i.e. direct emissions from when the compost is spread on the field. Primarily from CO_2, but also methane and other substances like N_2O. In total 4.8 kg CO_2 eq. solely from vegetables) and to a smaller degree direct emissions from the composting facility (1.7 kg CO_2 eq.). On the other hand, this is balanced to a great extent by a saving in fertiliser substitution, in total 6 kg CO_2 eq. Thus, leaving a total of 0.5 kg CO_2-eq for industrial composting of the vegetables. The only process that contributed to environmental benefits from industrial composting is the substitution of fertiliser.

Looking solely into the PLA and excluding the vegetable waste, the industrial composting does not perform best in any impact categories compared with the other EoL technologies. The industrial composting performs second best in three impact categories (climate change, photochemical ozone formation and human toxicity (cancer effects). The use of compost

contributes most to climate change, but the impact almost balances out with the fertiliser substitution. The fertiliser substitution also contributes to significant savings in human toxicity (cancer effects). The EoL technology performs worst in two impact categories (marine eutrophication and particulate matter), mainly due to impacts from direct emissions of the composting facility.

The following impact categories perform better when including vegetables matter; climate change - biogenic, human toxicity (cancer effects) and photochemical ozone formation. The reason for the lower impacts is savings from fertiliser substitution from the compost applied on land. The impact categories, apart from climate change, that give a higher impact due to direct emissions from use on land or additional material use, energy use and emission from the composting facility. For human toxicity (non-cancer effects) and freshwater ecotoxicity the impacts from use on land are high. For ozone depletion, particulate matter, ionizing radiation, freshwater and marine eutrophication, water use, abiotic depletion (both) and NREU the additional impacts from the composting plant is the cause.

The contribution analysis for the industrial composting of PLA food packaging film without vegetable waste shows that the contribution to the overall LCIA results is dominated by the substitution of energy and direct emissions from the composting facility. In the three eutrophication impact categories as well as acidification, petrochemical ozone formation and climate change - fossil the direct emissions contribute to the largest share (37-66 %). The heat substitution from incinerating rejects contributes to large savings in the impact categories of NREU, abiotic depletion (fossil fuels) and ozone depletion but to a large impact in the category ionizing radiation. The use on land of the compost and fertiliser substitution give large contributions to impact categories of climate change - biogenic, human toxicity non-cancer and freshwater ecotoxicity, and thereby even each other out in the end result for these categories.

3.3.2.2 Plastic recycling of PLA

Table 189 presents the results of the LCIA of PLA with vegetable waste. Recycling of PLA with the additional vegetables does not show the best performance. The additional vegetable waste will enter the recycling facility and be treated as a reject and incinerated. There are two impact categories, abiotic depletion (fossil fuels) and NREU, where recycling performs the best, by saving in substitution of PP. The extra resources needed in order to transport, treat and sort the vegetable food waste will cause an increase in emissions for the recycling facility and in the incineration facility that will eventually treat all the vegetable food waste.

At the present time the technology for recycling PLA is not on a commercial scale, and therefore this scenario is not considered to be realistic in the near future. The recycled PLA was substituted with PP, which gives high savings in environmental impacts. To date the recycled PLA would most likely substitute PP and therefore the recycling EoL was modelled in this way. In the long term, the recycled PLA would replace PLA and hence the saved emissions would decrease (around one third less when comparing results for climate change impacts for cradle-to-user for PP and PLA, but this is evaluated to be outside of than the temporal scope of this study).

Based on the LCIA for PLA without the vegetable waste, the overall preferred EoL technology for PLA is recycling, with the best performance in 11 out of 16 impact categories. The largest savings in all impact categories in the recycling technology are from the substitution of PP. The emissions from the recycling facility have the highest impact in most categories, except in climate change and freshwater eutrophication, where the largest emission of CO_2-eq. comes from the treatment of rejects (incineration and treatment of bottom ash and fly ash). For human toxicity (non-cancer effects) transportation contributes most to the overall LCIA results for recycling PLA. The recycling facility has the largest contribution to the impact category water use, from washing the plastic before recycling.

3.3.2.3 Landfilling of PLA

Landfilling performs worst in two impact categories, but second best in three and best in six impact categories. Landfilling performs worst in terms of climate change with CO_2 emissions of 28.3 kg CO_2 eq. due to landfill gas combustion and direct emissions of methane from the gas upgrading and direct greenhouse gas emissions to the environment. The direct emissions from landfill gas combustion contribute to a large share of the climate change impacts, ozone depletion and photochemical ozone formation. In the same four categories, the landfill gas that is not collected and oxidised under cover also contributes to a large share.

In all other impact categories, it is the electricity substitution that contributes to the largest share. The results stress the importance of keeping biodegradable plastic and organic waste from entering landfills. The landfill performs best in toxicity categories and freshwater eutrophication. Electricity substitution contribute to 0.2 % of climate change impacts and are especially contributing to savings in acidification (88 % of absolute impact) and NREU (46 % of absolute impact).

3.3.2.4 Anaerobic digestion of PLA

Anaerobic digestion performs best in eight impact categories, second best in nine impact categories and worst in 5. Applying the digestate and compost on agricultural land together with the transportation will give emissions of metals that contribute to human toxicity (non-cancer effects). The AD technology performs best in the categories of abiotic depletion, water use and NREU, indicating that this EoL is better at preserving resources compared to other EoL technologies. For the vegetable waste AD is a relevant technology, however there are several limitations for the AD technology to fully reach its potential in terms of treatment of PLA. First, the PLA only degrades under wet thermophilic conditions and it is estimated that only 23% of the AD plants in Europe are wet thermophilic. Therefore, the largest part of the PLA entering anaerobic digestion plants will have no methane yield to recover energy. Secondly, the methane potential in the PLA is low compared to other organic waste, resulting in a low methane yield and gas production. This is underlined by the fact that energy recovery on average only contributes to 7 % of the impact categories, ranging from 0 % to 21 %, which is dominantly from the food leftovers.

Looking into the contribution analysis, excluding the vegetable waste, the contribution is evenly distributed between several processes. The substitution of heat and electricity from the incineration of rejects contributes to a large share of the overall LCIA results in the categories NREU, abiotic depletion (both), ionising radiation, particulate matter and ozone depletion. The fertiliser substitution gives high savings in human toxicity cancer effects and climate change from biogenic carbon. Treatment of rejects and use on land of compost contribute the most to biogenic climate change impacts, human toxicity non-cancer and freshwater ecotoxicity.

3.3.2.5 Incineration of PLA

Incineration performs best or second best in no impact categories and worst in four impact categories. The impact categories where incineration performs the worst are ionizing radiation, photochemical ozone formation, particulate matter and terrestrial eutrophication. The largest contributor to impacts from incineration of PLA including vegetable waste are the direct emissions from the incineration, especially in biogenic climate change impacts, photochemical ozone formation, acidification, terrestrial and marine eutrophication. The contribution analysis shows that the heat substitution has the largest savings in climate change, freshwater ecotoxicity, NREU and abiotic depletion (fossil fuels), but the largest impact in ionizing radiation – with contributions in absolute percentages from 2 % to 73 %.

3.2.2.6 Weighted end of life results for baseline product system (PLA)

The weighted results of the EoL technologies are presented in Figure 164, and the numerical results can be found in Annex 4. For methodology and choice of normalization and weighing factors, please see the Approach and methodology chapter, section 5.2.3 "Use normalised and weighted results to determine the preference of EoL option" and the values in Table 30. Normalisation factors (NF) and weighting factors used by this study.

The weighted results show that with toxicity, the industrial composting technology is best, secondly anaerobic digestion, thirdly incineration, the second worst is recycling and the least favoured EoL technology is landfilling. If one looks at incineration solely with energy recovery (i.e. excluding plants without energy recovery), the difference between incineration and recycling will minimise.

The largest contribution to the weighted results is the global warming potential impact category for incineration, landfilling and recycling. For the biological treatments other impact categories (e.g. human toxicity categories) are dominant for the weighted results with toxicity. Without toxicity, it is the AD EoL technology that performs the best, with composting in second place. For the impact categories without toxicity it is acidification and marine eutrophication that are more dominant, as well as climate change potential. The savings from the biological recycling are in human toxicity (cancer effects), from savings from fertiliser substitution. The savings in recycling and incineration are from abiotic depletion (fossil fuels), which is due to savings from heat and electricity substitution.

The conclusion from the weighted results are that the biological EoL treatments are the most favourable EoL technologies for PLA packaging film with vegetables, and that landfilling should be a avoided. Industrial composting and anaerobic digestion are favourable mainly due to the low climate change impact, by conserving the carbon in the compost and substituting it with fertiliser. The industrial composting has lower impacts and lower savings compared to the AD.

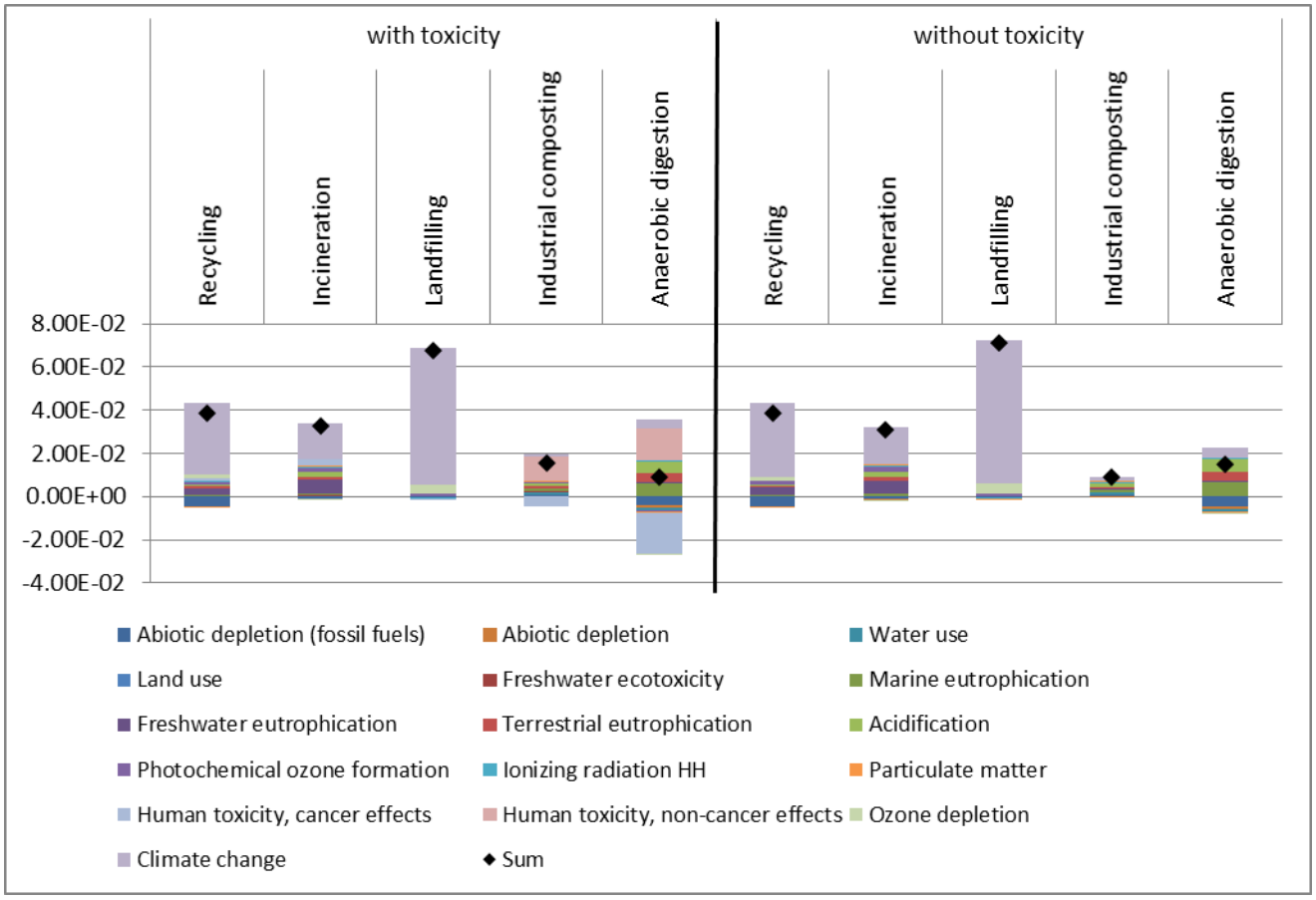


Figure 164. Weighted EoL results for the baseline product system, bio-based PLA food packaging film including vegetable food waste.

3.3.3 Alternative and reference: Bio-Based PP and petrochemical PP

The impact assessment results for the PP food packaging film are presented per the EoL-reference flow, i.e. 1 kg plastic, 100 g of contamination 15 kg food leftovers for each EoL technologies (assuming 100 % disposal via that technology).

Regarding the PP including additional vegetable food waste, see Table 190, a conclusion on the ranking between the EoL technologies is not easily drawn. Each of the EoL technologies performs well in some impact categories, and poorly in others.

Looking into the PP without the additional vegetable food waste, recycling is the preferred EoL technology followed by landfill and at lastly incineration. Incineration of the petrochemical plastic film results in a climate change impact of 2.3 CO_2-eq compared to 0.046 kg CO_2-eq for recycling and 0.016 kg CO_2-eq for landfilling. The high climate change impact from incineration comes mostly from direct emissions of incinerating the PP. The heating and electricity substitution contribute to savings of -0.8 kg CO_2-eq.

Further details on the individual EoL options for the PP packaging film are presented below, looking into specific impact categories and the contribution of impacts from different processes and elemental emissions.

Table 190. Total results of treating 1 kg bio-based and petrochemical PP (incl. 100 g contamination and 15 kg vegetable food waste) for all impact categories. Highest impact within each category is red and lowest impact green. The intended EoL option is marked with a bold box.

Impact category	Unit	Bio-based PP incl. vegetable food waste			Petrochemical PP incl. vegetable food waste		
		Recycling	Incineration	Landfilling	Recycling	Incineration	Landfilling
Climate change - fossil	kg CO_2 eq	-7.92E-01	-6.77E-01	1.07E-01	1.15E-01	2.35E+00	1.07E-01
Climate change - biogenic	kg CO_2 eq	1.58E+01	8.83E+00	2.52E+01	1.49E+01	5.80E+00	2.52E+01
Climate change - total	kg CO_2 eq	1.50E+01	8.15E+00	2.53E+01	1.50E+01	8.15E+00	2.53E+01
Ozone depletion	kg CFC-11 eq	6.05E-06	-1.00E-07	1.30E-05	6.05E-06	-1.00E-07	1.30E-05
Human toxicity, non-cancer effects	CTUh	9.97E-08	1.61E-07	1.61E-08	9.97E-08	1.61E-07	1.61E-08
Human toxicity, cancer effects	CTUh	6.65E-08	1.95E-07	-1.98E-09	6.65E-08	1.95E-07	-1.98E-09
Particulate matter	kg PM2.5 eq	-1.37E-04	1.12E-04	-1.75E-04	-1.37E-04	1.12E-04	-1.75E-04
Ionizing radiation HH	kBq U235 eq	4.43E-02	1.69E-01	-2.18E-02	4.43E-02	1.69E-01	-2.18E-02
Photochemical ozone formation	kg NMVOC eq	8.73E-03	1.49E-02	7.88E-03	8.73E-03	1.49E-02	7.88E-03
Acidification	molc H+ eq	6.85E-03	1.74E-02	7.19E-05	6.85E-03	1.74E-02	7.19E-05
Terrestrial eutrophication	molc N eq	3.57E-02	7.72E-02	1.59E-03	3.57E-02	7.72E-02	1.59E-03
Freshwater eutrophication	kg P eq	1.74E-03	3.44E-03	-2.18E-05	1.74E-03	3.44E-03	-2.18E-05
Marine eutrophication	kg N eq	2.98E-03	6.10E-03	6.83E-04	2.98E-03	6.10E-03	6.83E-04
Freshwater ecotoxicity	CTUe	1.88E+00	5.74E+00	3.23E-02	1.88E+00	5.74E+00	3.23E-02
Land use	kg C deficit	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	m^3	2.54E-01	-1.36E-01	-3.17E-01	2.54E-01	-1.36E-01	-3.17E-01
Abiotic depletion	kg Sb eq	-4.19E-07	-6.68E-07	-3.65E-07	-4.19E-07	-6.68E-07	-3.65E-07
Abiotic depletion (fossil fuels)	MJ	-3.55E+01	-1.56E+01	-1.51E+00	-3.55E+01	-1.56E+01	-1.51E+00

NREU	MJ	-3.71E+01	-1.94E+01	-6.05E-01	-3.71E+01	-1.94E+01	-6.05E-01

The significance of the colour scale is the same as for all the other product systems (Table 189 and Table 190). This means that you can compare the colour horizontally across all tables, but only comparing between the same impact category.

3.3.3.1 Recycling of PP

The LCIA results for all impact categories for recycling of petrochemical PP packaging film are presented Table 190. When the PP packaging film with vegetable waste is recycled, a part of the PP packaging and all the vegetable waste will end up as rejects. The vegetable waste is distributed 47 % to incineration and 43 % to landfilling, and the PP rejects will only be incinerated. The recycling EoL technology only perform the best in two impact categories, the NREU and abiotic depletion (fossil fuels), from energy and PP substitution. The only impact category that the recycling performs worst in is water use, from washing the plastic.

The substitution of PP gives high savings for all impact categories except water use, which has the largest savings from substitution of heat from the incineration of rejects. Treatment of rejects gives the largest impact for climate change, freshwater ecotoxicity, freshwater eutrophication and human toxicity, cancer effects. The contribution analysis for recycling of the bio-based and petrochemical PP food packaging film shows that the plastic substitution is the largest contributor to the overall LCIA results in all impact categories except for biogenic climate change, both human toxicity categories, freshwater ecotoxicity, freshwater eutrophication, water use and abiotic depletion. The plastic recycling plant has the largest contribution to water use from washing the plastic. The transportation has the largest contribution in human toxicity, non-cancer (36 %).

3.3.3.2 Incineration of PP

Incineration performs best in three impact categories compared with the other two relevant EoL technologies. The good performance in these three impact categories (climate change, ozone depletion and abiotic depletion) is due to savings in electricity and heat. However, incineration performs worst in 10 out of 16 impact categories.

Contribution analysis of the incineration of petrochemical and bio-based PP single-use cups shows that heat substitution, electricity substitution and direct emissions contribute the largest amount of impacts (positive or negative) in almost all impact categories. The bottom ash treatment has the largest impact on human toxicity (cancer effects), freshwater eutrophication and freshwater ecotoxicity. The large contribution in cancer effects is due to chromium emissions to water and soil and the freshwater eutrophication comes from phosphate emissions to water and soil. The fly ash treatment has the largest contribution in the human toxicity, non-cancer impact. The heat substitution has the largest savings in the impact categories NREU, abiotic depletion (fossil fuels), ozone depletion and the largest impact in the ionising radiation category. The direct emissions have a large impact on the climate change (biogenic for the bio-based and fossil for the petrochemical PP), photochemical ozone formation terrestrial and marine eutrophication. The electricity substitution has a large contribution in the form of savings for the categories climate change - biogenic, particulate matter, water use and abiotic depletion.

3.3.3.3 Landfilling of PP

Landfilling of PP packaging film with vegetable waste shows the highest impact in climate change. The vegetable waste accounts for the largest share of climate change emissions, as the vegetable waste degrades and generates landfill gas. Landfilling also performs worst in the impact categories of NREU and abiotic depletion (fossil) as there is little energy or material recovery compared with the other EoL technologies. For the three eutrophication impact categories it is landfilling that performs best compared to incineration and recycling.

The limited landfill impacts from the PP packaging film itself come from the construction and operation of the landfill, transportation and leachate treatment. Transportation has a large impact on climate change from fossil carbon, human toxicity non-cancer, photochemical ozone formation, terrestrial eutrophication and NREU. Leachate treatment has a substantial impact of

48 % in freshwater ecotoxicity and 64 % in marine eutrophication but contributes to less than 20 % in other impact categories. The marine eutrophication impact from leachate treatment mostly comes from ammonium that is released to surface water.

3.3.4 Details on climate change impacts

This section looks across both product systems of the food packaging film primarily with vegetable waste in all EoL technologies, i.e. comparing all results in Table 189 to Table 190. For illustration the climate change impact is presented in Figure 165, with a contribution between processes in the technologies. The treatment of vegetable waste is presented in a separate combined process. Note that this section still compares per EoL EoL-reference flow, i.e. 1 kg plastic, 100 g contamination and food leftovers.

Overall, the largest climate change impact emission from EoL technologies of food packaging film, is caused by the treatment of additional vegetable waste. The best performing technology with PLA food packaging is industrial composting, followed by anaerobic digestion. These two technologies are intended for treatment of bio-degradable materials, as they preserve the nutritional value in the organic material and substitute conventional fertilisers. Landfilling the vegetable food waste results in a high generation of methane gas, the PLA will also degrade in the landfill and generate methane gas.

Aside from the treatment of additional vegetable food waste plastic, recycling and fertiliser substitution have significant climate change savings. At the present time the technology for recycling PLA is not available on a commercial scale, and therefore this scenario is not realistic in the near future. Moreover, the recycled PLA was substituted with PP, which gives high savings in environmental impacts. To date the recycled PLA would most likely substitute PP and therefore the recycling EoL was modelled in this way. In the long term, the recycled PLA would replace PLA and hence the saved emissions would decrease. Landfill results for the petrochemical plastic (not including vegetable waste) have an overall better performance than the other technologies, because the plastic is assumed not to degrade within 100 years. Nonetheless, the plastic resource is lost, which is also observed in the fact that landfilling of petrochemical PP performs worst in the impact categories of; abiotic depletion, abiotic depletion (fossil fuels) and NREU.

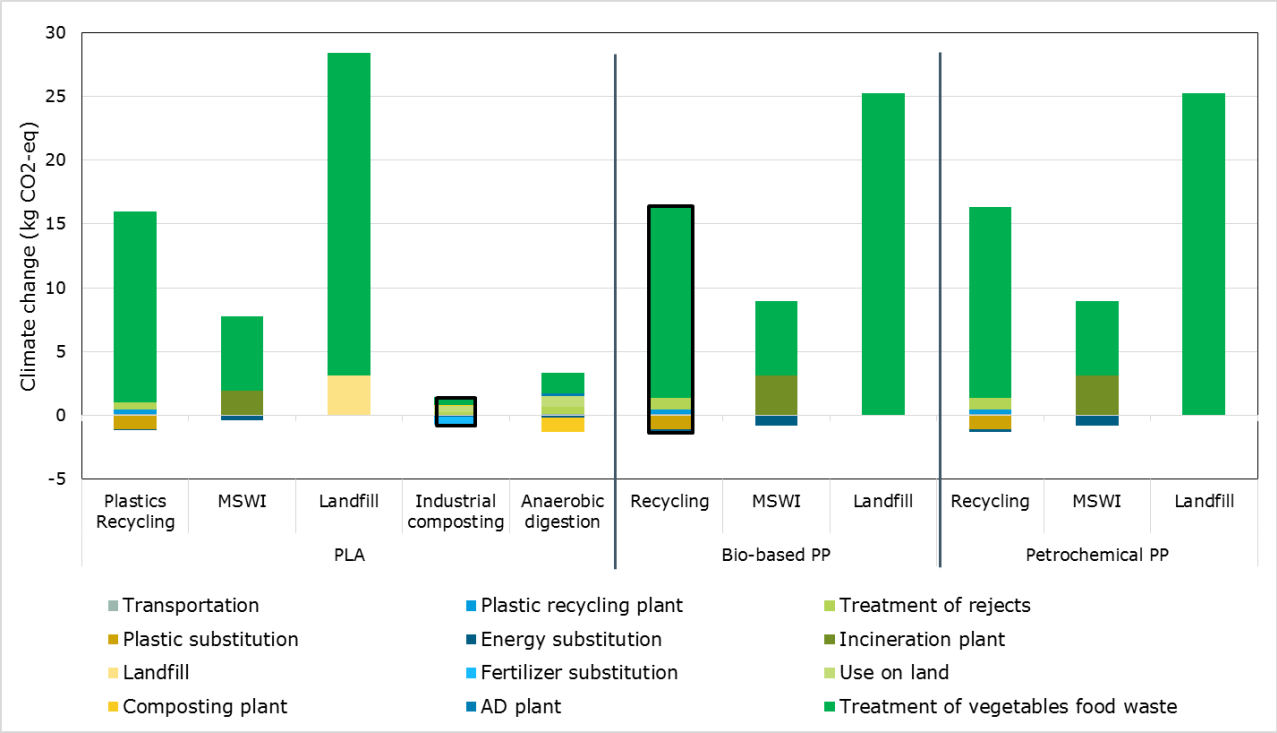


Figure 165. Climate change of EoL technologies for bio-based PLA and petrochemical PP food packaging film for 1 kg plastic and 100 g contamination. The intended EoL option is marked with a black box. See Annex 4 for details of the content of the grouped processes. The results without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amounts of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

3.3.5 Comparing all EoL technologies in all the product systems with weighted results

The LCIA results of all product systems have been weighted with toxicity, Figure 166, per functional unit (i.e. different from the above sections were the results were presented per EoL reference flow with 1 kg plastic, 100 g contamination and food leftovers). The weighted results show that the biological treatments of PLA packaging films are the preferred choice. The incineration of PLA packaging film also shows a lower weighted score compared to all EoL technologies for the PP product systems. Recycling is a slightly better choice of treatment than incineration for the PP product systems and landfilling is worst for all product systems.

Recycling the PLA packaging film contributed primarily to climate change, which is caused by incineration and landfilling of the vegetable food leftovers (impacts from direct emissions from the incineration plant and the landfill gas, which is not collected or flared). Climate change are dominant in the weighted results of incineration of the PLA, which is due to direct emissions from incinerating. Climate change impacts arise from landfilling the PLA due to landfill gas directly emitted or flared. For industrial composting and anaerobic digestion, the emissions in human toxicity (non-cancer effects) and savings in human toxicity (cancer effects) stems from respectively use on land and fertiliser substitution.

For the bio-based and petrochemical PP, recycling contributions mainly to climate change, due to incineration and landfilling of the food leftovers. Incineration of PP contributes to climate change due to direct emissions of the incineration plant. Landfilling PP packaging and the food

leftovers contribute mainly to climate change, where the impacts originate from landfill gas being directly emitted and flared (the gas comes from degradation of the food leftovers).

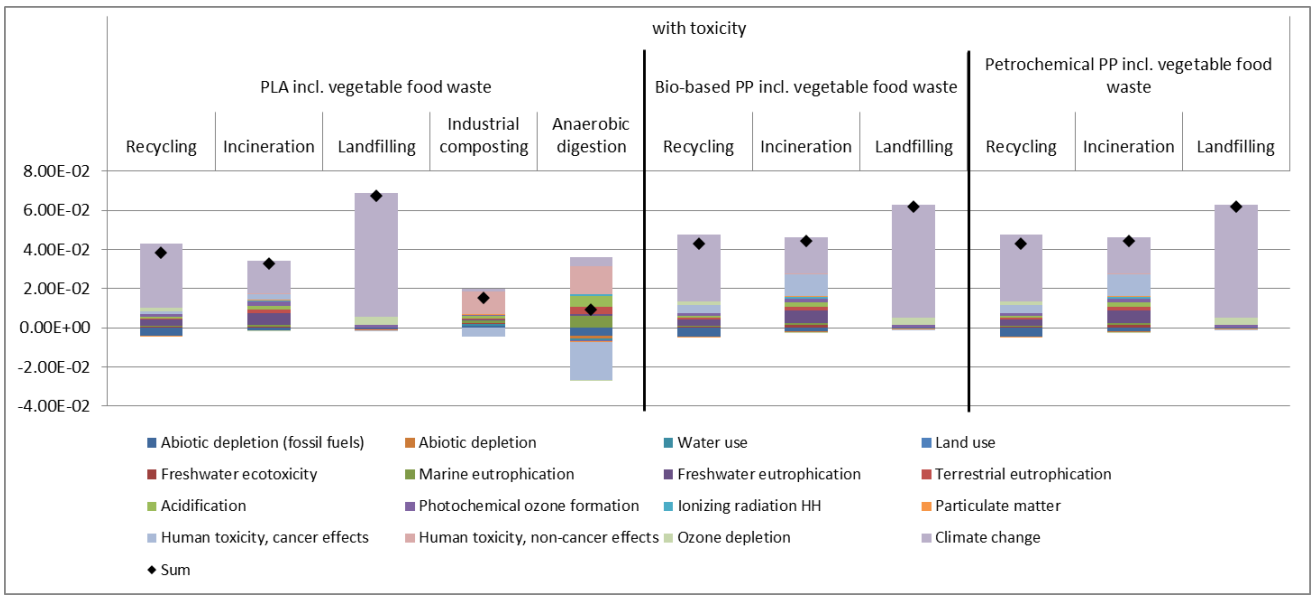


Figure 166. Weighted results with toxicity for the three products systems of food packaging film (including vegetable food leftovers) for their EoL options per 1 kg of plastic (the weight of the functional unit of the products systems are the same). The numerical figures are presented in Annex 4.

3.3.6 End of life mix results

The LCIA results are presented now as a mix of the EoL options including the vegetable food leftovers in Table 191. The EoL mixes of the product systems are presented at the top.

The bio-based PP and the petrochemical PP have the exact same results, except for the biogenic and fossil climate change impacts. Comparing the starch-based food packaging film with the PP based films the PP films perform better in nine impact categories and the starch-based film also in nine out 18 impact categories, see Table 191. It is therefore hard to point out one product system that is better for the mixes chosen. The largest difference (percentile) in human toxicity (cancer effects) and particulate matter favouring the bio-based PLA, because industrial composting is the preferred EoL technology in these categories.

Table 191. LCIA for EoL mix of the food packaging for all product systems per functional unit; 3.12 kg plastic + 320 g organic contamination + 46.8 kg food leftovers.

Impact category	Unit	Bio-based PLA	Bio-based PP	Petrochemical PP
Industiral composting	%	15		
Recycling	%	15	30	30
Incineration	%	39	39	39
Landfill	%	31	31	31
Climate change - fossil	kg CO_2 eq / FU	-1.17E-01	-1.46E+00	3.07E+00
Climate change - biogenic	kg CO_2 eq / FU	4.38E+01	4.99E+01	4.53E+01
Climate change - total	kg CO_2 eq / FU	4.36E+01	4.84E+01	4.84E+01
Ozone depletion	kg CFC-11 eq / FU	1.71E-05	1.81E-05	1.81E-05
Human toxicity, non-cancer effects	CTUh / FU	1.73E-06	3.05E-07	3.05E-07
Human toxicity, cancer effects	CTUh / FU	2.99E-08	2.98E-07	2.98E-07
Particulate matter	kg PM2.5 eq / FU	8.07E-05	-1.61E-04	-1.61E-04
Ionizing radiation HH	kBq U235 eq / FU	1.72E-01	2.26E-01	2.26E-01
Photochemical ozone formation	kg NMVOC eq / FU	3.21E-02	3.39E-02	3.39E-02
Acidification	molc H+ eq / FU	2.79E-02	2.77E-02	2.77E-02
Terrestrial eutrophication	molc N eq / FU	1.27E-01	1.29E-01	1.29E-01
Freshwater eutrophication	kg P eq / FU	4.86E-03	5.80E-03	5.80E-03
Marine eutrophication	kg N eq / FU	1.29E-02	1.09E-02	1.09E-02
Freshwater ecotoxicity	CTUe / FU	4.24E+00	8.77E+00	8.77E+00
Land use	kg C deficit / FU	0.00E+00	0.00E+00	0.00E+00
Water use	m^3 / FU	4.08E-01	-2.35E-01	-2.35E-01
Abiotic depletion	kg Sb eq / FU	-1.51E-06	-1.56E-06	-1.56E-06
Abiotic depletion (fossil fuels)	MJ / FU	-2.59E+01	-5.36E+01	-5.36E+01
NREU	MJ / FU	-3.06E+01	-5.89E+01	-5.89E+01

Looking into the seven comparable categories, as set out in Table 192, it is the PP product systems that performs better in four out of seven impact categories. Depending on the importance of the impact categories, the preferred product system will differ.

Table 192. LCIA results for the seven comparable impact categories for EoL mix of all product systems for food packaging film including vegetable waste per functional unit.

Impact category	Unit	Bio-based PLA	PP product systems
Climate change - total	kg CO_2 eq / FU	4.36E+01	4.84E+01
Particulate matter	kg PM2.5 eq / FU	8.07E-05	-1.61E-04
Photochemical ozone formation	kg NMVOC eq / FU	3.21E-02	3.39E-02
Acidification	molc H+ eq / FU	2.79E-02	2.77E-02
Terrestrial eutrophication	molc N eq / FU	1.27E-01	1.29E-01
Abiotic depletion (fossil fuels)	MJ / FU	-2.59E+01	-5.36E+01
NREU	MJ / FU	-3.06E+01	-5.89E+01

3.4 Cradle-to-grave environmental impacts and interpretation

In this section, the cradle-to-grave results for the investigated product systems are reported. The EoL environmental impact included in the presented cradle-to-grave results is based on

the EoL mix detailed in section *End of Life description, data, assumptions and multifunctionality.*

3.4.1 Bio-based baseline system: PLA

The cradle-to-grave results of PLA (baseline) packaging films are given in Table 193. The breakdown of impacts for the baseline product system between biomass production (including carbon removal), manufacturing (including transportation) and EoL is shown in Figure 167. For PLA, the Eol mix is recycling 15 %, industrial composting 15 %, incineration: 39 % and landfilling 31 %.

Table 193. Cradle-to-grave LCA results of PLA films (baseline) on functional unit basis (excluding DLUC and ILUC effects).

Impact category	Unit	Biomass	Manufacturing	EoL	Total
Climate change	kg CO_2 eq.	-4.72	9.83	43.58	48.69
Ozone depletion	kg CFC-11eq	9.31E-09	5.51E-07	1.71E-05	1.76E-05
Human toxicity, non-cancer effects	CTUh	-8.74E-07	3.14E-06	1.72E-06	3.99E-06
Human toxicity, cancer effects	CTUh	9.59E-09	5.84E-07	2.98E-08	6.23E-07
Particulate matter	kg PM2.5eq	2.91E-04	3.94E-03	8.06E-05	4.31E-03
Ionizing radiation HH	kBq U235eq	6.92E-02	6.99E-01	1.72E-01	9.41E-01
Photochemical ozone formation	kg NMVOCeq	1.17E-02	2.79E-02	3.20E-02	7.16E-02
Acidification	molc H+eq	1.12E-02	5.14E-02	2.79E-02	9.05E-02
Terrestrial eutrophication	molc Neq	5.68E-02	1.05E-01	1.27E-01	2.89E-01
Freshwater eutrophication	kg Peq	1.03E-04	4.64E-04	4.85E-03	5.42E-03
Marine eutrophication	kg Neq	1.86E-02	8.19E-03	1.29E-02	3.97E-02
Freshwater ecotoxicity	CTUe	3.26E+01	1.56E+01	4.23	52.45
Land use	kg C deficit	47.63	15.98	0.00	63.61
Water use	m3	1.98	4.05	0.41	6.43
Abiotic depletion	kg Sbeq	1.02E-06	2.40E-05	-1.51E-06	2.35E-05
Abiotic depletion (fossil fuels)	MJ	8.25	127.78	-25.82	110.20
NREU	MJ	8.88	151.62	-30.59	129.91

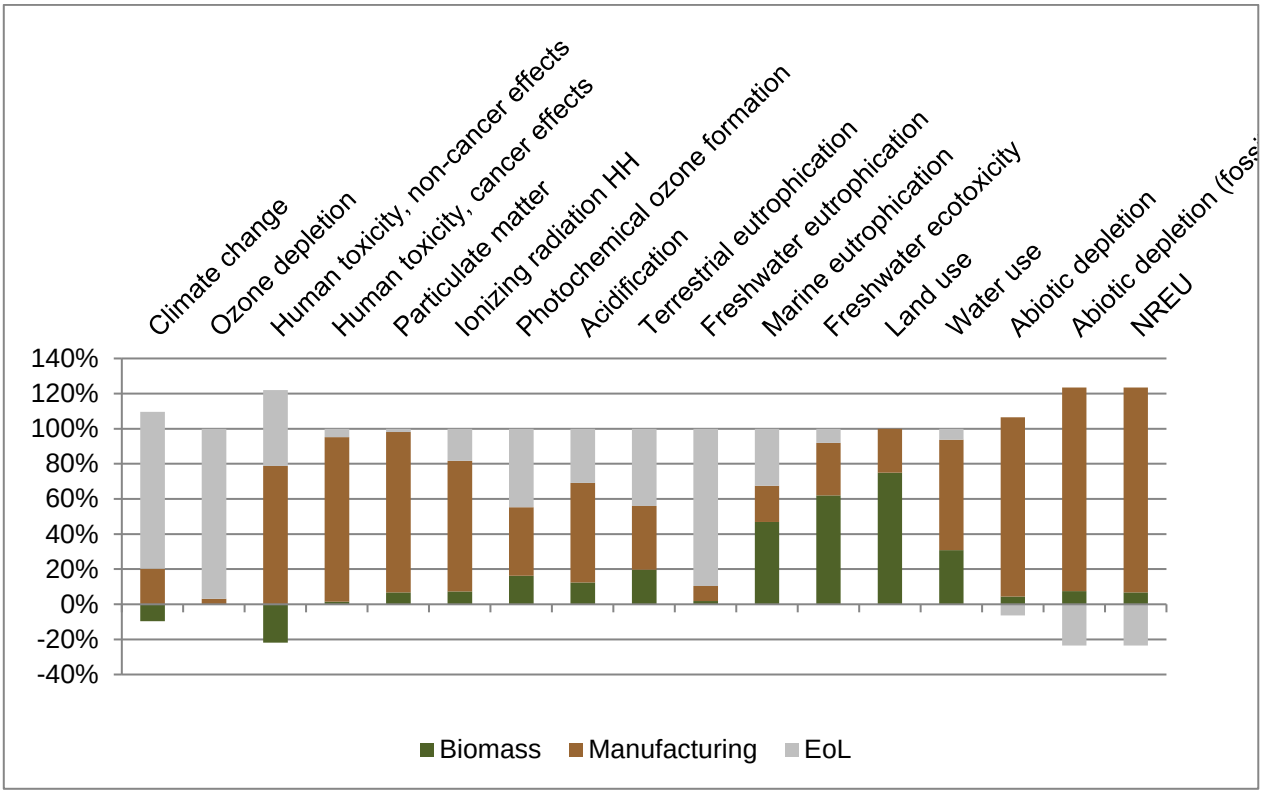
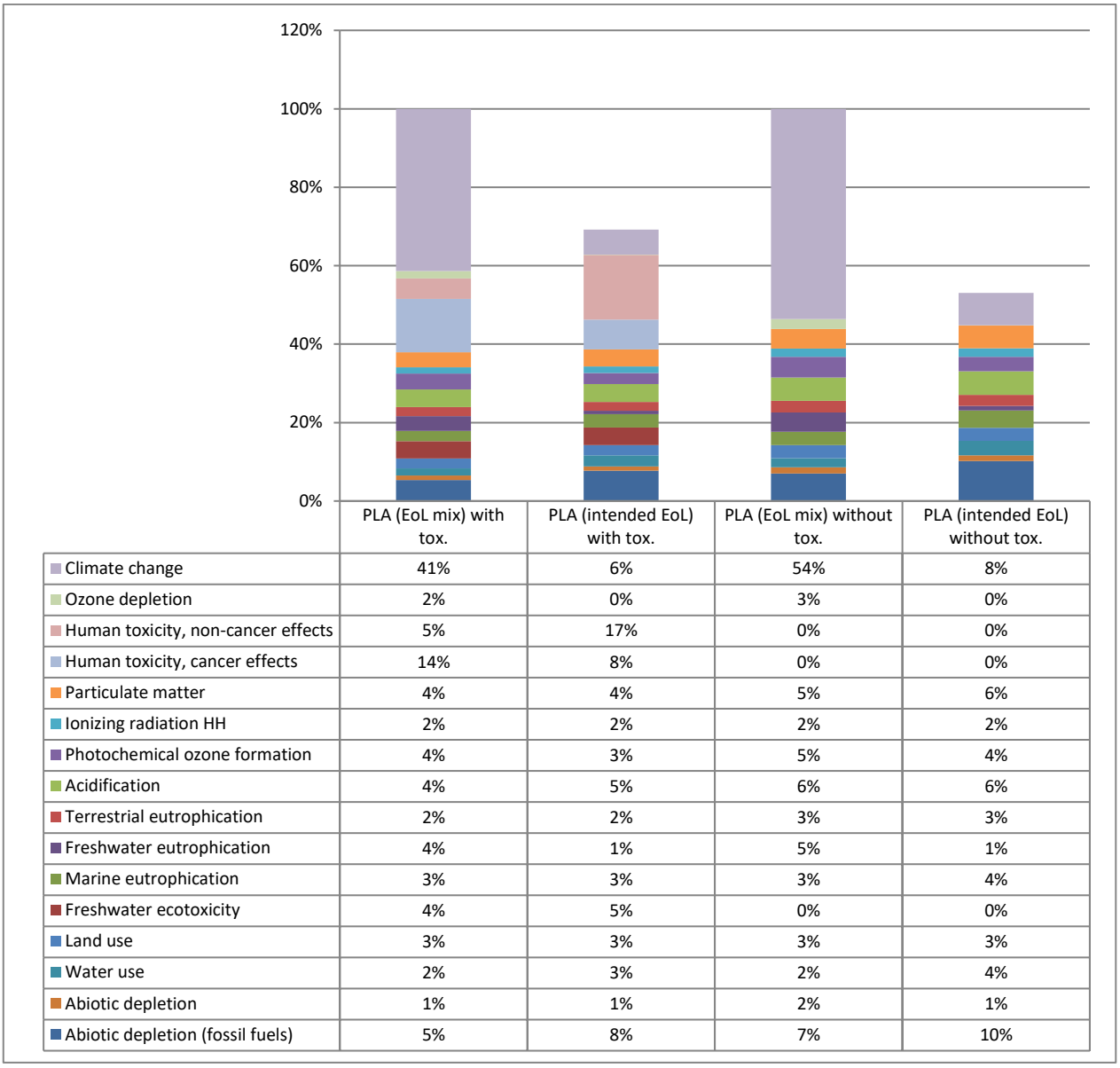


Figure 167. Breakdown of the potential impacts from the PLA (baseline) packaging films across all impact categories, cradle-to-grave results (excluding DLUC and ILUC effects).

Manufacturing (including transportation) along with EoL are the major contributors to the environmental burden in all the impact categories except for marine eutrophication, freshwater ecotoxicity and land use. Nevertheless, manufacturing represents a significantly impacting life cycle stage also in these last impact categories. Biomass production has generally a minor impact (less than 20 %). That threshold is overcome for photochemical ozone formation, terrestrial eutrophication, marine eutrophication, freshwater ecotoxicity, land and water use. At EoL stage significant environmental impacts are seen only in abiotic depletion-fossil fuels (and NREU) owing mainly to the plastic substitution with recycling and the heat and electricity produced during incineration. An increase in environmental impacts due to EoL is instead significant for climate change, ozone depletion, human toxicity without cancer effects, photochemical ozone formation, acidification, terrestrial, freshwater and marine eutrophication.

Normalisation and weighting have been applied to the baseline system (PLA (baseline)) to identify the main hotspots of the aggregated impacts into a single score. Figure 168 shows the weighted results including or not including the toxicity impact categories.



	PLA (EoL mix) with tox.	PLA (intended EoL) with tox.	PLA (EoL mix) without tox.	PLA (intended EoL) without tox.
Climate change	41%	6%	54%	8%
Ozone depletion	2%	0%	3%	0%
Human toxicity, non-cancer effects	5%	17%	0%	0%
Human toxicity, cancer effects	14%	8%	0%	0%
Particulate matter	4%	4%	5%	6%
Ionizing radiation HH	2%	2%	2%	2%
Photochemical ozone formation	4%	3%	5%	4%
Acidification	4%	5%	6%	6%
Terrestrial eutrophication	2%	2%	3%	3%
Freshwater eutrophication	4%	1%	5%	1%
Marine eutrophication	3%	3%	3%	4%
Freshwater ecotoxicity	4%	5%	0%	0%
Land use	3%	3%	3%	3%
Water use	2%	3%	2%	4%
Abiotic depletion	1%	1%	2%	1%
Abiotic depletion (fossil fuels)	5%	8%	7%	10%

Figure 168. Weighted impact assessment result for PLA (baseline) and variation of weighted cradle-to-grave results for PLA packaging films when the intended EoL option achieves the 100 %.

When the impact categories related to toxicity are included in the weighted results, climate change (41 %), human toxicity-cancer effects (14 %), abiotic depletion of fossil fuels (5 %), freshwater ecotoxicity (4 %), acidification (4 %), particulate matter (4 %), human toxicity non cancer effects (5 %) and photochemical ozone formation (4 %) are the most significant categories (about 80 % of total).

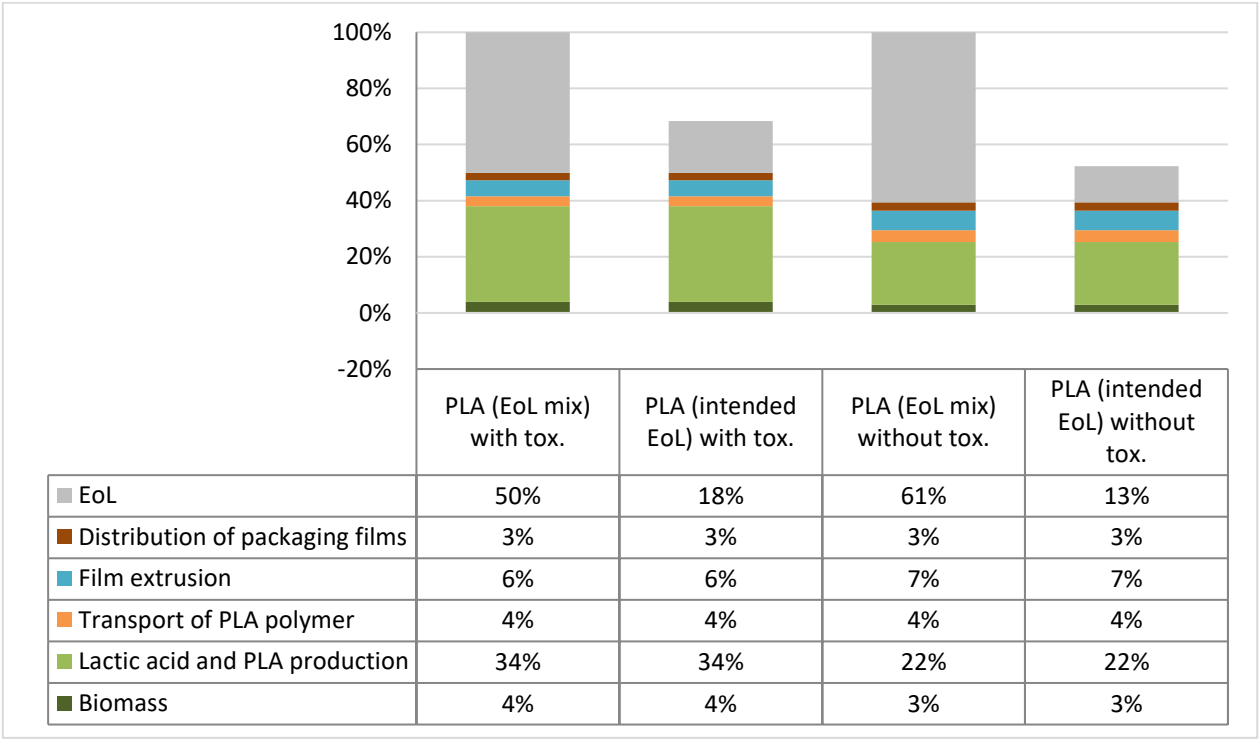
The EoL climate change impact accounts for 37 % of the 41 % climate change contribution, while 12 % out of the 14 % assigned to human-toxicity-cancer effects comes from the lactic acid and PLA production.

From the assessment of the environmental impact excluding toxicity impact categories, what emerges is that the main impact belongs to the impact category climate change (53 %) which is followed by abiotic depletion of fossil fuels (7 %), acidification (6 %), particulate matter (5 %), photochemical ozone formation (5 %) and freshwater eutrophication (5 %). Climate

change is absolutely the most significant and 48 % out of 53 % related to the climate change contribution belongs to EoL.

As noticed, the climate change due to EoL is the major impact (between the 30 and 40 % of the total impact) and 63 % of this impact is due to the landfilling of PLA packaging films with food residues. Another 22 % comes instead from its incineration.

Figure 169 shows the breakdown in life-cycle stages of the weighted single score results. It is clear that manufacturing and EoL are the main stages responsible for the environmental burdens impacting respectively 37-47 % and 49-60 % of the impact depending if toxicity impact categories are accounted for or not. The main difference caused by the accounting or not of the toxicity impact categories is within the manufacturing stage. When toxicity is accounted for the impact of the chemical conversion from biomass to PLA has a higher share. Biomass is responsible only for 3-4 % of the total cradle-to-grave environmental impact. Among the processes of the manufacturing phase, the lactic acid and PLA production is the most impacting, contributing the 23-35 % of the total cradle-to-grave environmental burden. The second unit process with the highest attributed emissions is film extrusion with only an impact of 6-7 % from cradle-to-grave.



	PLA (EoL mix) with tox.	PLA (intended EoL) with tox.	PLA (EoL mix) without tox.	PLA (intended EoL) without tox.
EoL	50%	18%	61%	13%
Distribution of packaging films	3%	3%	3%	3%
Film extrusion	6%	6%	7%	7%
Transport of PLA polymer	4%	4%	4%	4%
Lactic acid and PLA production	34%	34%	22%	22%
Biomass	4%	4%	3%	3%

Figure 169. Breakdown of the weighted impact assessment result for PLA (baseline).

From the weighted results, it emerges that the EoL stage has a very relevant share of the cradle-to-grave environmental impact: between 49 and 60 % depending on whether toxicity is included or not in the analysis. The EoL for PLA has been defined as 15 % recycling, 15 % industrial composting, 39 % incineration and 31 % landfilling. Nevertheless, PLA is a biodegradable plastic, the intended EoL has been identified as industrial composting. It is relevant for this material to estimate how much environmental benefit can be achieved if the share of the intended EoL increased in the EoL mix. Trying to answer this last query, Figure 170 shows that when PLA packaging films are 100 % industrially composted, significant environmental benefits are achieved in terms of climate change (-85 %), ozone depletion (-95 %) and freshwater eutrophication (-75 %). At the same time, worse performances are shown

in terms of human toxicity without cancer effects (+70 %), water use (+30 %) and abiotic depletion of fossil fuels (+30 %).

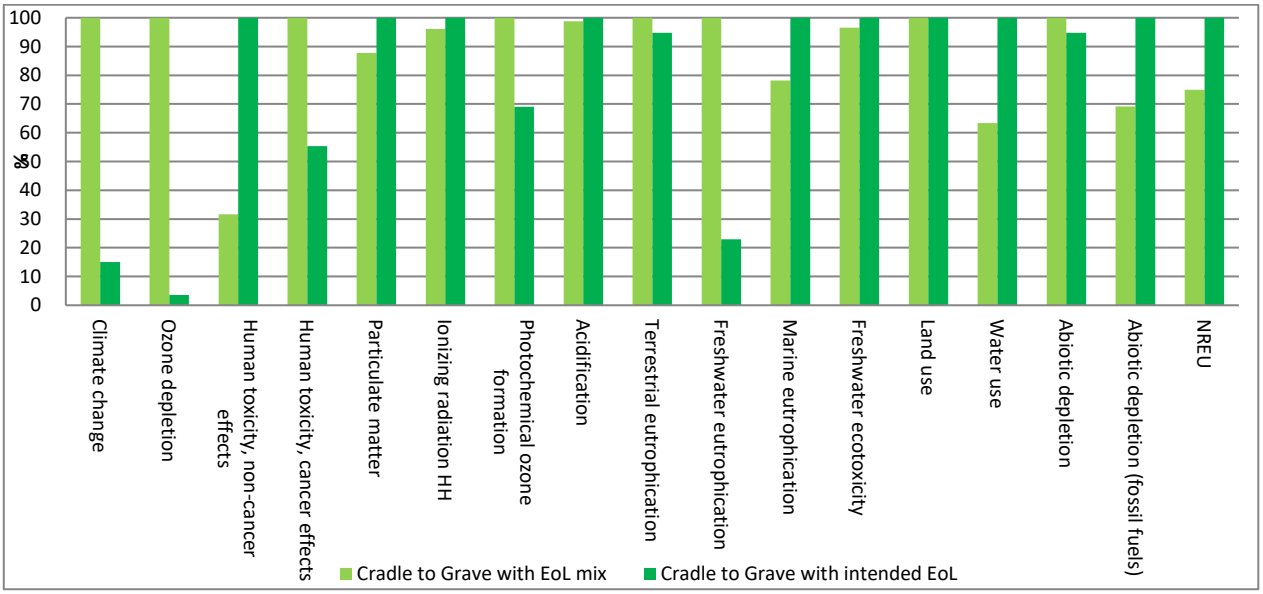


Figure 170. Variation of cradle-to-grave results for PLA packaging films when the intended EoL option achieves the 100 %.

Once weighted, these life cycle impact results become the ones shown in Figure 169. Depending on whether toxicity is accounted or not, used PLA packaging films processed through 100 % industrial composting instead of the 15 % of the current-near future share would lead to environmental benefits estimated between 30 and 45 % which is mainly due to the reduction of 85 % in the cradle-to-grave climate change impacts.

3.4.2 Bio-Based alternative system 1: PLA (EU maize)

The cradle-to-grave results of PLA (EU maize) packaging films are given in Table 194. The breakdown of impacts for the baseline product system between biomass production (including carbon removal), manufacturing (including transportation) and EoL is shown in Figure 171.

Table 194. Cradle-to-grave LCA results of PLA packaging films (EU maize) on functional unit basis (excluding DLUC and ILUC effects).

Impact category	Unit	Biomass	Manufacturing	EoL mix	Total
Climate change	kg CO_2 eq.	-2.43	9.52	43.58	50.67
Ozone depletion	kg CFC-11eq	1.44E-07	2.87E-07	1.71E-05	1.75E-05
Human toxicity, non-cancer effects	CTUh	4.18E-06	3.39E-07	1.72E-06	6.24E-06
Human toxicity, cancer effects	CTUh	8.92E-08	8.05E-07	2.98E-08	9.24E-07
Particulate matter	kg PM2.5eq	2.26E-03	1.84E-03	8.06E-05	4.18E-03
Ionizing radiation HH	kBq U235eq	1.74E-01	8.12E-01	1.72E-01	1.16E+00
Photochemical ozone formation	kg NMVOCeq	1.07E-02	1.74E-02	3.20E-02	6.01E-02
Acidification	molc H+eq	6.15E-02	2.88E-02	2.79E-02	1.18E-01
Terrestrial eutrophication	molc Neq	2.33E-01	6.12E-02	1.27E-01	4.22E-01
Freshwater eutrophication	kg Peq	8.37E-04	1.33E-04	4.85E-03	5.82E-03
Marine eutrophication	kg Neq	4.85E-02	6.06E-03	1.29E-02	6.75E-02
Freshwater ecotoxicity	CTUe	7.19E+01	1.03E+01	4.23E+00	8.65E+01
Land use	kg C deficit	73.07	6.39	0.00	79.46
Water use	m3	26.40	3.98	0.41	30.79
Abiotic depletion	kg Sbeq	1.22E-06	2.02E-05	-1.51E-06	1.99E-05
Abiotic depletion (fossil fuels)	MJ	26.06	122.59	-25.82	122.84
NREU	MJ	31.09	152.62	-30.59	153.13

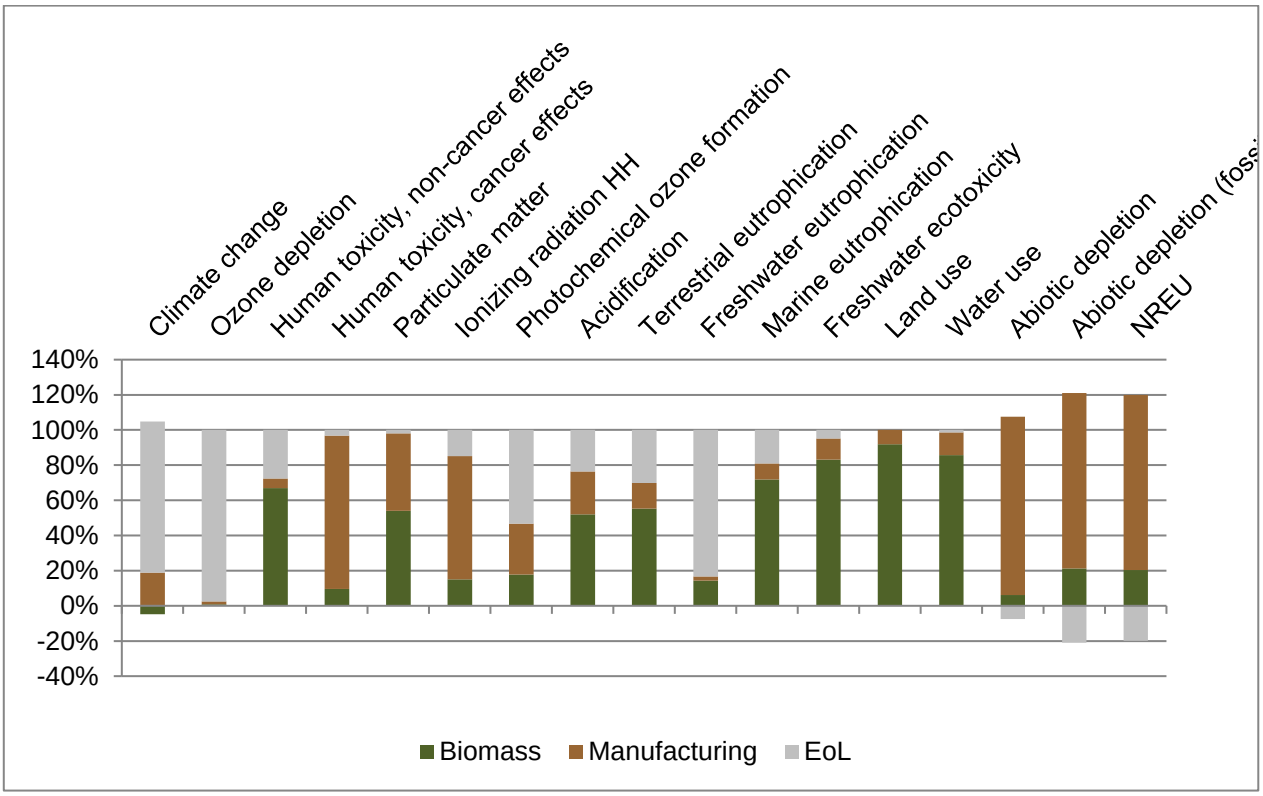


Figure 171. Breakdown of the potential impacts from the PLA (EU maize) packaging films across all impact categories, cradle-to-grave results (excluding DLUC and ILUC effects).

In this alternative product system (PLA packaging films from EU maize). Biomass production no longer has a minor impact and in many of the impact categories contributes equally or even more compared to manufacturing. At the EoL stage sensible savings are achieved as for the baseline system only in terms of abiotic depletion of fossil fuels (and NREU).

3.4.3 Bio-Based alternative system 2: Bio-based PP from UCO

The cradle-to-grave results of bio-based PP packaging films are given in Table 195. The breakdown of impacts for the baseline product system between feedstock (UCO collection and carbon removal), manufacturing (including transportation) and EoL is shown in Figure 172. For bio-based PP, the Eol mix is recycling 30 %, incineration 39 % and landfilling 31 %.

Table 195. Cradle-to-grave LCA results of bio-based PP (UCO) packaging films on functional unit basis (excluding DLUC and ILUC effects).

Impact category	Unit	Feedstock	Manufacturing	EoL mix	Total
Climate change	kg CO_2 eq.	-9.78	4.56	48.33	43.12
Ozone depletion	kg CFC-11eq	1.89E-09	6.00E-07	1.81E-05	1.87E-05
Human toxicity, non-cancer effects	CTUh	1.55E-09	2.98E-07	3.05E-07	6.04E-07
Human toxicity, cancer effects	CTUh	4.24E-10	3.30E-08	2.98E-07	3.31E-07
Particulate matter	kg PM2.5eq	9.89E-06	1.60E-03	-1.60E-04	1.45E-03
Ionizing radiation HH	kBq U235eq	1.30E-03	4.87E-01	2.25E-01	7.13E-01
Photochemical ozone formation	kg NMVOCeq	1.07E-04	1.35E-02	3.38E-02	4.75E-02
Acidification	molc H+eq	1.17E-04	2.01E-02	2.77E-02	4.79E-02
Terrestrial eutrophication	molc Neq	3.97E-04	4.48E-02	1.29E-01	1.74E-01
Freshwater eutrophication	kg Peq	1.06E-06	1.70E-04	5.79E-03	5.96E-03
Marine eutrophication	kg Neq	3.60E-05	4.08E-03	1.09E-02	1.50E-02
Freshwater ecotoxicity	CTUe	0.01	2.88	8.76	11.64
Land use	kg C deficit	0.04	8.93	0.00	8.97
Water use	m3	3.00E-03	4.73E+00	-2.34E-01	4.50E+00
Abiotic depletion	kg Sbeq	2.51E-08	1.90E-05	-1.56E-06	1.74E-05
Abiotic depletion (fossil fuels)	MJ	0.19	58.43	-53.51	5.12
NREU	MJ	0.25	73.75	-58.82	15.19

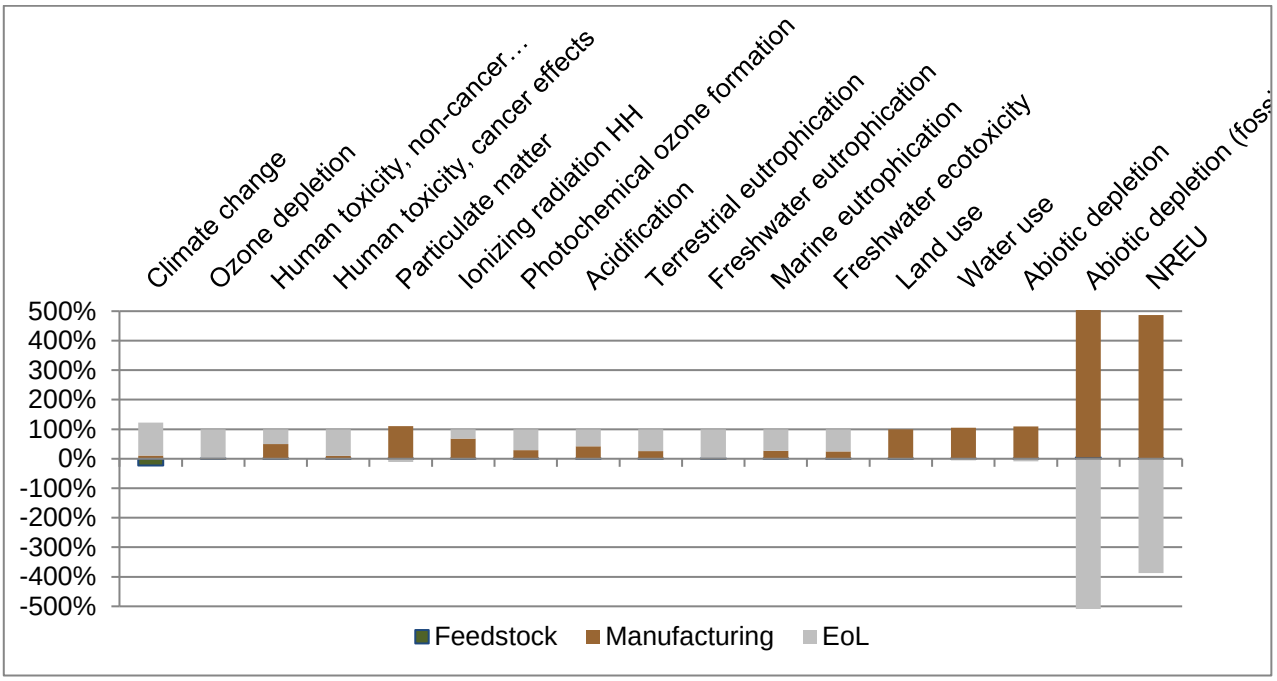


Figure 172. Breakdown of the potential impacts from the bio-based PP (UCO) packaging across all impact categories, cradle-to-grave results (excluding DLUC and ILUC effects).

In this alternative product system (bio-based PP films from UCO). The impact of the feedstock is negligible: it is a waste product and therefore the only impact which is assigned is due to

collection. The total impact in terms of climate change is negative and this is because of the carbon removal which for PP is higher compared to PLA due to a higher carbon content. EoL and manufacturing are the main sources of environmental impact.

During the EoL stage several relevant savings occur in terms of particulate matter (mainly due to plastic substitution of recycling), abiotic depletion (mainly thanks to electricity produced in MSWI), abiotic depletion of fossil fuels-and NREU- (mainly thank to plastic recycling).

3.4.4 Petrochemical reference system

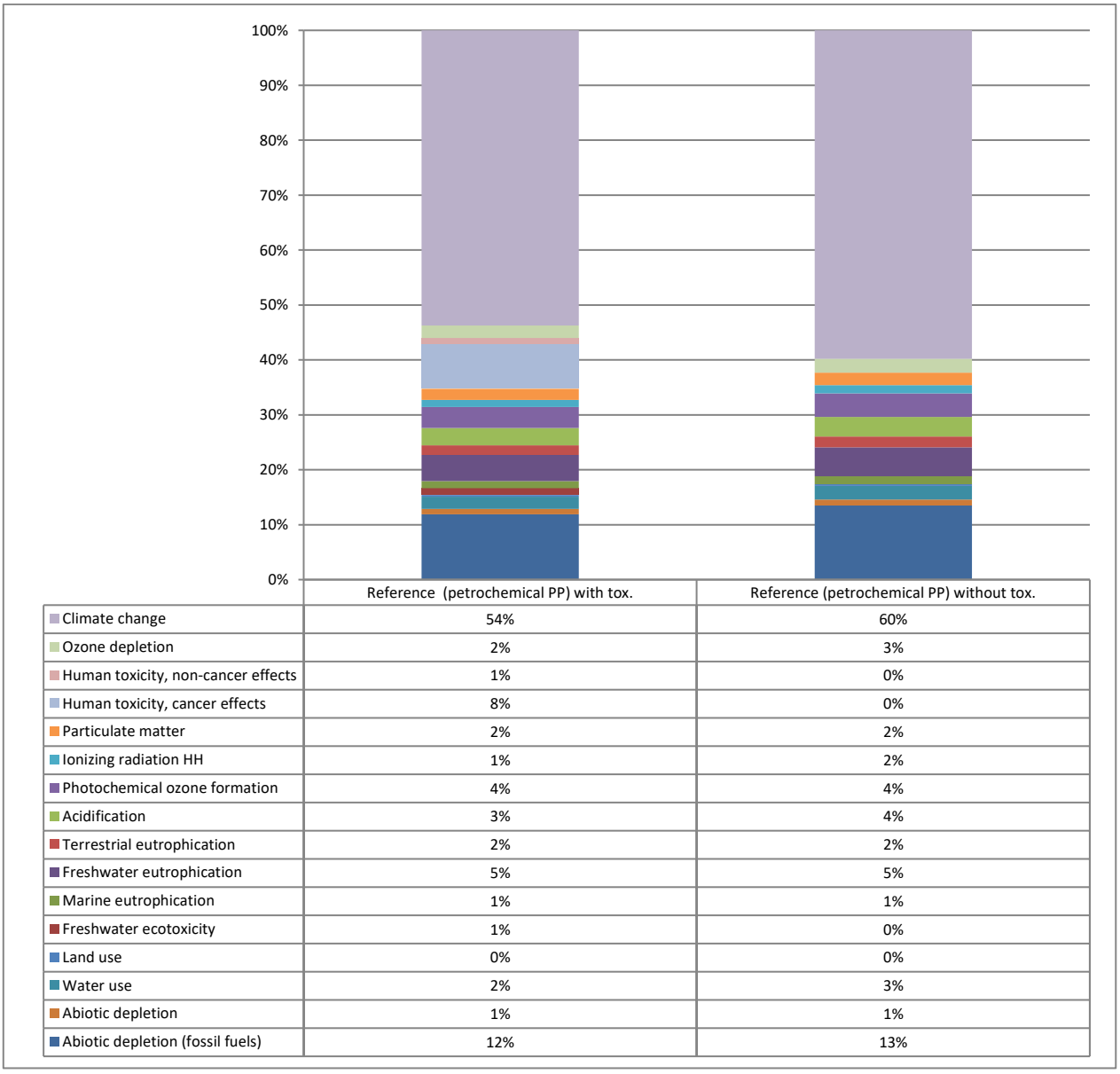
In this section, the cradle-to-grave results of the investigated petrochemical references are reported with the breakdown into manufacturing stage (granulates production, transportation stages and film extrusion) and EoL. For PP, the EoL mix is recycling 30 %, incineration 39 % and landfilling 31 %.

Table 196. Cradle-to-grave LCA results of petrochemical PP packaging films on functional unit basis.

Impact category	Unit	Manufacturing	EoL mix	Total
Climate change	kg CO_2 eq.	7.76	48.34	56.09
Particulate matter	kg PM2.5eq	2.17E-03	-1.60E-04	2.01E-03
Photochemical ozone formation	kg NMVOCeq	2.65E-02	3.38E-02	6.04E-02
Acidification	molc H+eq	2.91E-02	2.77E-02	5.68E-02
Terrestrial eutrophication	molc Neq	7.05E-02	1.29E-01	1.99E-01
Abiotic depletion (fossil fuels)	MJ	272.78	-53.51	219.27
NREU	MJ	289.84	-58.82	231.02

Normalisation and weighting have been applied to the reference system to understand which among the assessed impact categories are more relevant for this application and what the main environmental hotspots are. From Figure 173, it becomes clear that climate change is the most impacting, with a share between 54 and 60 %. This is followed by abiotic depletion of fossil fuels with a share between 12 % and 13 %.

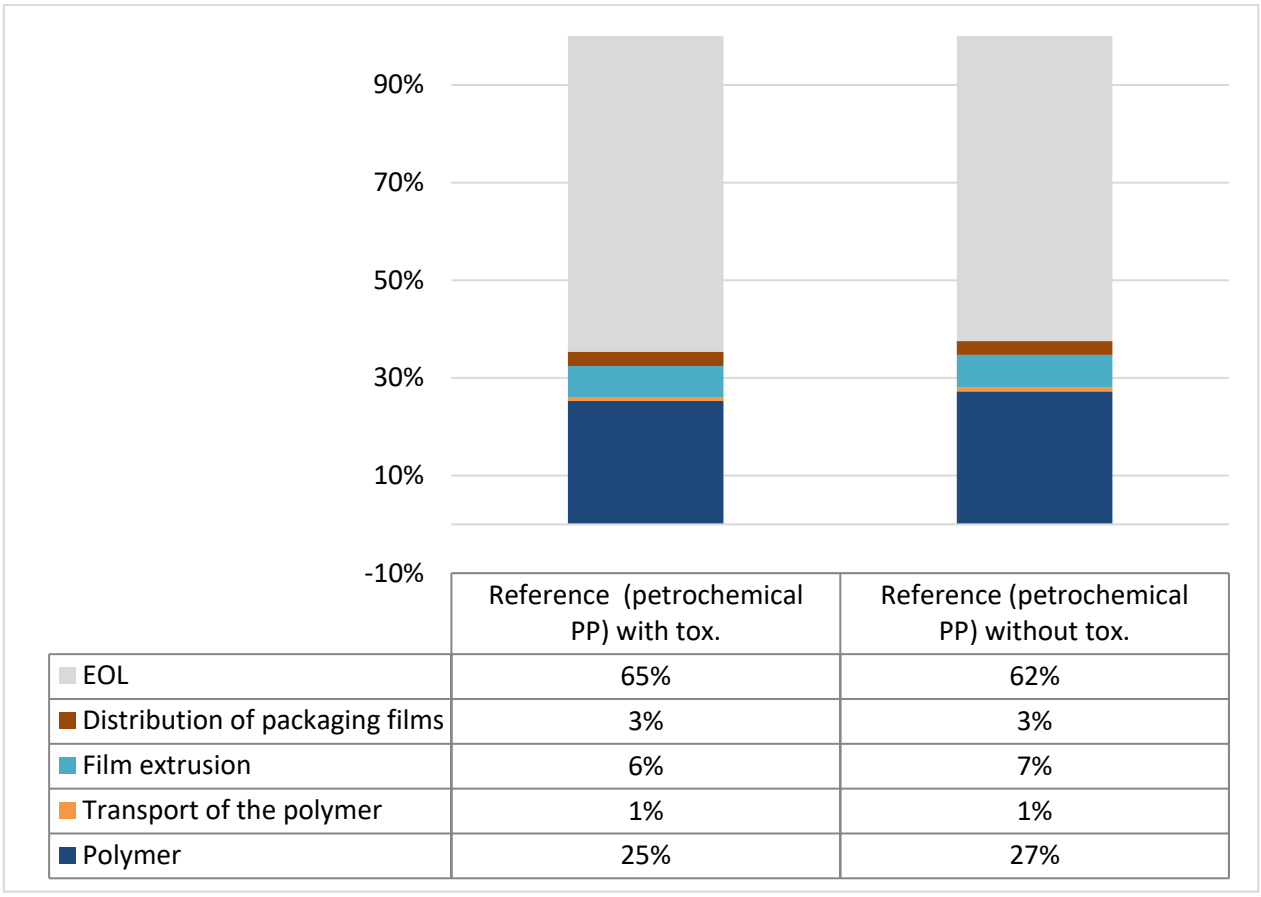
When toxicity is considered, human toxicity with cancer effects becomes a very important impact with 8 % of the impact.



	Reference (petrochemical PP) with tox.	Reference (petrochemical PP) without tox.
Climate change	54%	60%
Ozone depletion	2%	3%
Human toxicity, non-cancer effects	1%	0%
Human toxicity, cancer effects	8%	0%
Particulate matter	2%	2%
Ionizing radiation HH	1%	2%
Photochemical ozone formation	4%	4%
Acidification	3%	4%
Terrestrial eutrophication	2%	2%
Freshwater eutrophication	5%	5%
Marine eutrophication	1%	1%
Freshwater ecotoxicity	1%	0%
Land use	0%	0%
Water use	2%	3%
Abiotic depletion	1%	1%
Abiotic depletion (fossil fuels)	12%	13%

Figure 173. Weighted impact for petrochemical references

Figure 174 shows the weight of each life-cycle stage on the total environmental impact. It is possible to notice that the contribution percentages of the considered life cycle stages do not significantly differ when toxicity is or is not included. For petrochemical packaging films, most of the emissions are related to the EoL: 59-62 %. The polymer production is the second contributor to environmental damage with 27-30 % of the impact. The film extrusion represents a 7 % share.



	Reference (petrochemical PP) with tox.	Reference (petrochemical PP) without tox.
EOL	65%	62%
Distribution of packaging films	3%	3%
Film extrusion	6%	7%
Transport of the polymer	1%	1%
Polymer	25%	27%

Figure 174. Breakdown of the weighted impact assessment results for petrochemical reference PP.

3.4.5 Comparing the Bio-Based baseline system with bio-based alternatives and petrochemical reference

Figure 172 shows the full life-cycle impact assessment comparison for the six PEFCR impact categories found to be appropriate for comparison for petrochemical PP. For the remaining impact categories, the comparison is only carried out for the different bio-based alternatives.

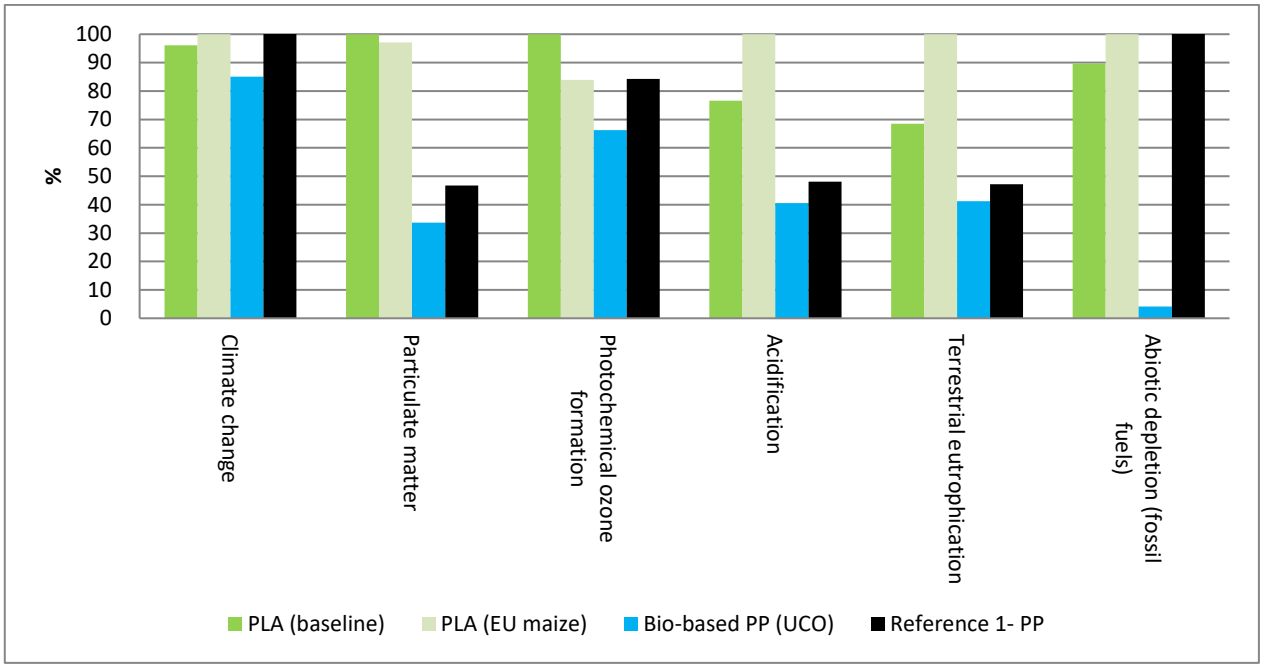


Figure 175. Comparing the cradle-to-grave (EU EoL mix) environmental impact of 1 FU films for the impact categories selected in chapter 6 (Investigation of generic LCA data for fossil-based plastics).

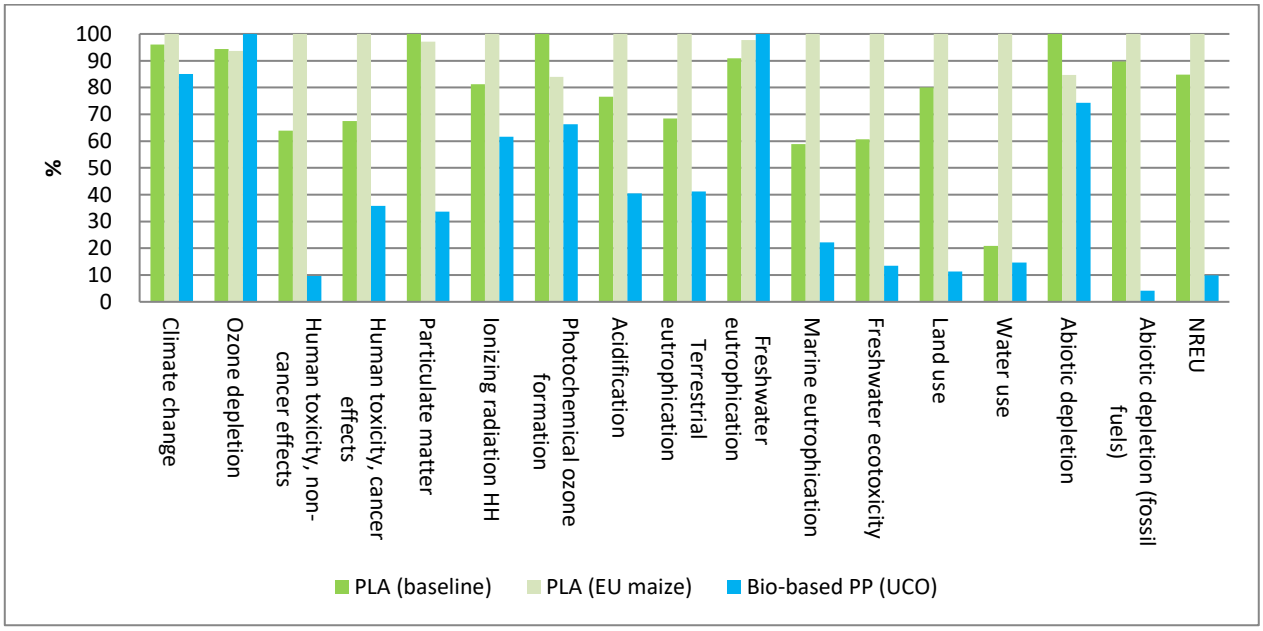


Figure 176. Comparing the cradle-to-grave (EU EoL mix) environmental impact of 1 FU of films.

The following considerations emerge from Figure 175 and Figure 176:

- PLA (baseline) shows better environmental performance compared to the petrochemical reference in abiotic depletion of fossil fuels (-50 %) and climate change (-13 %). Nevertheless, it performs significantly worse in particulate matter (+144 %), photochemical ozone formation (+19 %), acidification (+59 %) and terrestrial eutrophication (+45 %). Compared to PLA from European maize, PLA (baseline)

performs generally better: out of 16 impact categories, it performs significantly worse only for photochemical ozone formation and abiotic depletion.

- PLA (EU maize) -similarly to PLA (baseline) but with lower emissions savings- performs better than the petrochemical reference in climate change and abiotic depletion of fossil fuels.

- Bio-based PP (alternative 2) shows the best performance in all six selected impact categories. This is also reflected in the comparison with bio-based PLA in all the 16 impact categories with a few exceptions where the impact is comparable.

- Petrochemical PP shows significantly better environmental performance compared to PLA (baseline) in four out of six selected impact categories, namely: particulate matter, photochemical ozone formation, acidification and terrestrial eutrophication.

The comparison between the baseline PLA and the petrochemical reference (petrochemical PP) undergoes several changes when the intended EoL is also considered in the comparison. From Figure 177, it is possible to notice that 100 % composted PLA packaging films allows substantial savings (-85 %) in terms of climate change impact compared to petrochemical PP. Moreover, PLA improves its performance significantly in photochemical ozone formation, reaching a lower impact compared to petrochemical PP (-18 %). On the other hand, there is an increase of abiotic depletion of fossil fuels and particulate matter shifting from the EoL mix to the intended EoL due to no replacement of virgin plastic and no production of electricity from its burning. Nevertheless, PLA would still be the best material for abiotic depletion of fossil fuels (-27 %). On the other hand, PLA performs still worse on the other half of impact categories for particulate matter (+140 %), acidification (+61 %) and terrestrial eutrophication (+37 %) compared to petrochemical PP. Nevertheless, on a weighted basis and only focusing on the selected impact categories, it is possible to conclude that the comparison between 100 % composted PLA and petrochemical PP is in favour of PLA.

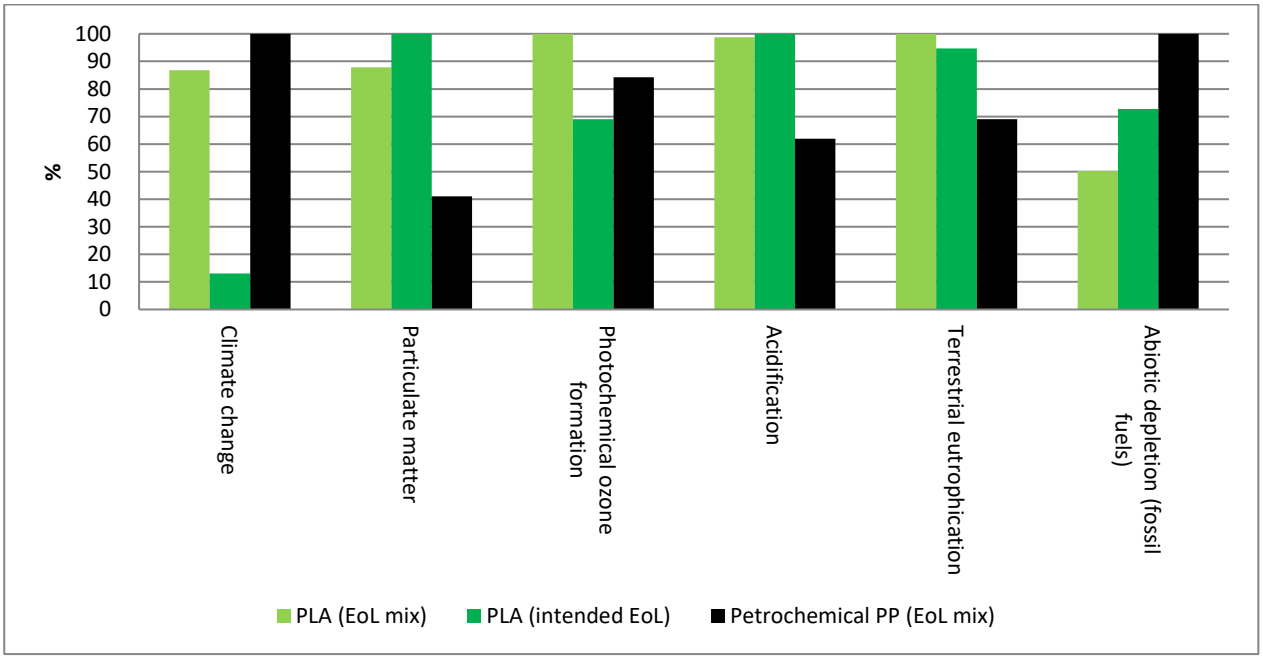


Figure 177. Comparing baseline and reference cradle-to-grave environmental impact the impact categories selected in chapter 6 on FU basis.

4 Discussion

4.1 Sensitivity analysis based on different material substitution factors (MSF)

As mentioned in section 1.2, the thickness of the PLA packaging film was fixed to 25 µm for the application defined in this study. Based on this thickness, the area density assumed for PLA packaging films was 32.1 m2/kg (BI-AX, 2018). Based on communications with companies, the thickness of the PP film was fixed to 36 µm: this corresponds to the thickness of PP films which leads to the same area density of PLA films (32.1 m2/kg).

Nevertheless, the design of bio-based packaging products is influenced by many factors. Usually, they should perform as similarly as possible to the petrochemical benchmark that they aim to replace on the market. On the other hand, the cost of bio-based polymers is generally higher compared to conventional polymers. This last fact leads us to reduce as much as possible the weight of the bio-based products to reach market prices closer to the petrochemical benchmark specifying e.g. the strength of the film. In this context, the material substitution factors (MSFs) can be seen as interesting parameters when different options are assessed.

Based on the different design constraints, the minimal mass of a given material can be calculated. Table 173 of section 1.2 shows the material properties of PLA and PP films. Based on the data reported in that table, the material substitution factors (MSF, see the definition in Section 1.3 of case study 3: Single-use Cups) are calculated.

Table 197 shows the MSFs that are obtained. A MSF below 1 (e.g. MSF = 0.7 in the table) means that the minimal amount of PLA to fulfil a desired design constraint (e.g. oxygen barrier-limited design) is less than the minimal mass required to PP (the reference material). A MSF greater than 1 (e.g. MSF = 1.2 in the table) means that more PLA is required compared to PP to fulfil the design constraint, i.e. the same strength (MSF =1.2). It can be seen that although the tensile strength of PLA is better than PP (see Table 173), because of the high density of PLA, it results in a heavier PLA film in order to be as strong as the a PP film.

Table 197. Material substitution factors for this application and variability of the weight per FU based on strength-limited design and oxygen-barrier property-limited design.

Design constraint	Material indices (Ashby, 1999),(Aggarwal et al., 2018)	MSF (m_{PLA}/m_{PP})	Weight of PLA films (kg/FU)	Weight of PP films (kg/FU)
Strength-limited design	σ_y/ρ	1.2	3.8	3.12
Oxygen barrier-limited design	$(PO_2 * \rho)^{-1}$	0.7	2.1	3.12
Default analysis	MI not used.	1	3.12	3.12

Symbols mentioned in the table: σ_y strength (MPa); PO_2 Oxygen transmission rate (cc/m2/24hr); ρ density(g/cm3) (values from Table 173).

The function-driven MSFs for the substitution of PP with PLA range between 0.7 (same oxygen barriers) and 1.2 (same strenghth) depending on the function property as reported in Table 197. Taking as a reference the weight of 3.12 kg as assumed in the default analysis, the corresponding weight of the PLA films found to be either 2.1 or 3.8 kg on functional unit basis.

Table 197 shows the cradle-to-grave (with EoL mix) environmental impact based on the different MSFs shown in table above. It can be seen that the overall conclusions do not change

regardless of the different MSFs: PLA films offer savings in climate change and abiotic depletion (fossil fuels); it does not offer savings for the other four impact categories. A lower MSF based on same oxygen barrier property allows more savings (for climate change and abiotic depletion fossil fuels), as well as allowing the gaps between PLA and PP for the other four impact categories to be closed. In particular, photochemical ozone formation of the PLA films is reduced so much to become close to the impact of the petrochemical reference (as shown in the Figure 178 the saving is nearly zero).

This sensitivity analysis attempts to understand how the environmental impacts could change based on single constraint. Product design is rarely carried out by only one design constraint. In this case study, both barrier property and strength are important functions, amongst others such as transparency, biodegradability, thermal properties (in case of cooling) and printability, etc. The final design is often a compromise of many trials and errors next to theoretical models.

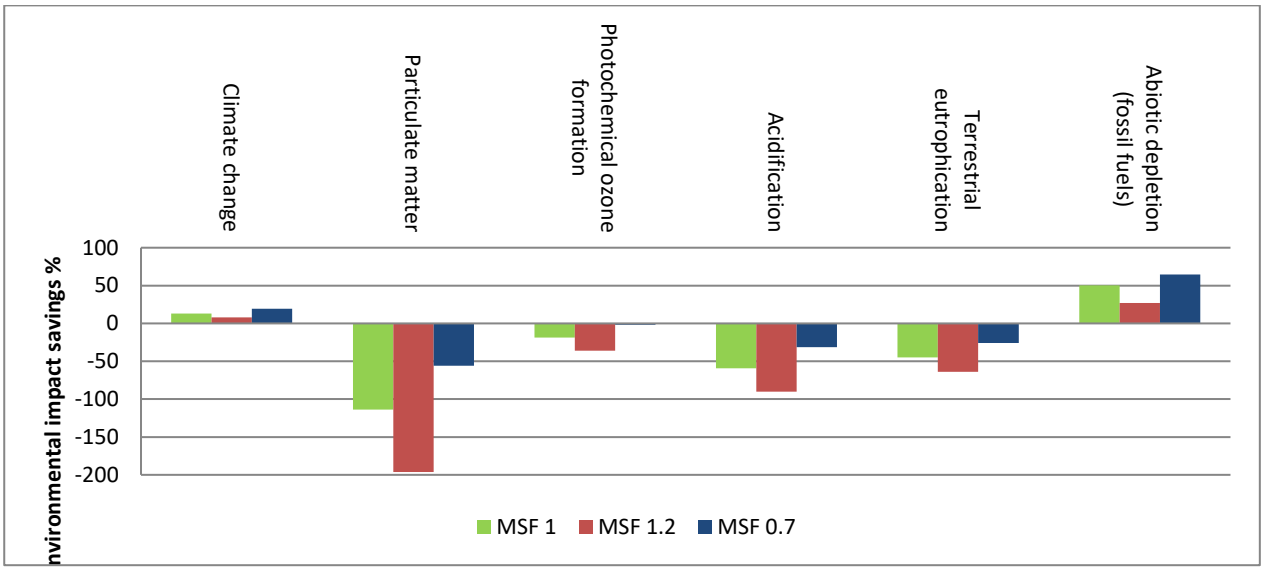


Figure 178. Comparison of environmental impact savings between baseline PLA and petrochemical reference PP for different MSFs (baseline MSF=1). MSF stands for material substitution factor. The bars above zero (positive numbers in the graph) mean savings can be expected from PLA films; the bars below zero (negative numbers) mean that PLA films do not offer environmental savings.

4.2. End of life sensitivity analysis

The sensitivity analysis was performed for the intended EoL option, hence industrial composting and recycling of PLA. The significant parameter chosen, which is likely to change due to an increase in technical improvements in both the plastic products and in the waste management technology, were the degradation percentage of the biogenic carbon in the product. This parameter changes within a given technology was modelled because the average European technology in the EoL scenarios represents a rather broad set of technological variances. This change will provide an insight into the variance within these technologies and depict a target for future waste scenarios. The parameter variations are presented in Table 198.

Furthermore, the amount of vegetable food leftovers which are composted have been analysed, and the range can be observed in Table 198, which is based on the previously described ranges.

Table 198. Variation in input parameters in the sensitivity analysis.

Product system	Technology	Parameter changed	Original EoL (%)	Lower (%)	Higher (%)	References
PLA	Industrial Composting	Biogenic carbon degradation	57.1	55	95	Hermann, 2011
PLA	Recycling	Sorting and technology efficiency	70	39	90.3	EASETECH
PLA	Industrial composting	Amount of vegetable food leftover (kg per kg plastic)	15	4.3	25.6	Manolopoulou, et al., 2012 and FAO, 2011

4.2.1 PLA industrial composting

The sensitivity analysis was performed on the industrial composting technology. The C degradation was changed according to literature limits given in Table 198. Changing the degradation rate of C only affects the climate change impact categories. Figure 179 presents the change in climate change impacts when altering the C degradation compared to the original for industrial composting of bio-based food packaging film including vegetable waste. A higher degradation rate gives a lower total climate change impact.

Using a higher C degradation leads to a decrease in the total climate change of 63 % compared to the original, or 0.1 kg CO_2-eq. There are three main reasons for the changes in the climate change for biogenic carbon:

- Increasing the degradation percentages means that less of the biogenic carbon will be incinerated with the rejects and contribute to direct emissions of biogenic CO_2.
- Less biogenic carbon will be released to the atmosphere in the form of the compost product that is spread on the land later.
- More biogenic carbon is emitted directly from the composting plant when the biodegradation percentage is higher.

Considering the fossil CO_2, a lower C degradation will lead to a small saving of fossil CO_2 as there is less compost to be spread on land with a diesel consuming tractor. A higher C degradation will lead to less savings of fossil CO_2 when the compost is used on land, which is why the change is positive.

The lower and higher carbon degradation would not cause changes in the conclusion that industrial composting performs best in climate change impact for the PLA product system.

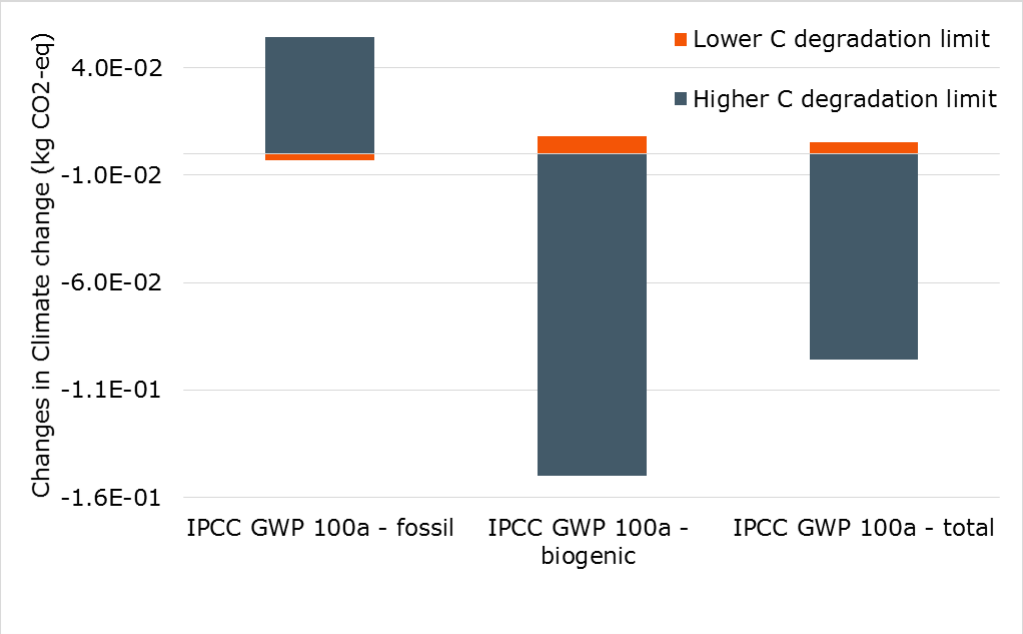


Figure 179. Sensitivity analysis for EoL compost PLA food packaging film.

This sensitivity would decrease the cradle-to-grave climate change results when increasing the C degradation by less than 0.2 % and when decreasing C degradation, the cradle-to-grave results will increase with a total climate change impact of less than 0.01 %.

4.2.2 PLA recycling

A sensitivity analysis was also performed on the recycling efficiency parameter. The results of the analysis are presented in Figure 180, which shows the percentage change in total results compared with the original. When the percentage is negative, the results are lower than for the original and vice versa. Applying the higher limit of the recycling efficiency of 90.3 % means that there are fewer rejects to incinerate.

The biogenic climate change decreases 72 % when applying the higher recycling efficiency due to lower direct emissions from burning the PLA. Freshwater eutrophication has a high percentage change when changing the recycling efficiency. The largest part of the emission contributing to freshwater eutrophication is the substitution of PP, as phosphate and phosphorus emissions to water and soil are avoided, when virgin PP production is avoided. There are big differences in the percentage change for ionizing radiation, as significant savings in this impact category can be achieved by increasing the recycling efficiency and thereby the substituted PP.

Increasing the recycling efficiency will both increase the environmental savings from substituting PP and have less direct emissions from the burning of the PLA. It is clear that increasing the recycling efficiency will benefit all impact categories, except ozone depletion, water use and abiotic depletion. The abiotic depletion is higher when the recycling is increased, due to more resources used in the recycling plant when treating a higher amount, and less electricity generated when there are fewer rejects incinerated for energy generation. The water use will increase when the recycling plant is treating a higher amount of plastic, as water is used for cleaning the plastic and for process steam.

The lower and higher sorting and technology efficiency would not change the conclusion regarding the climate change impact, recycling still performs second worst for the PLA product system (looking at the results with food leftovers). Only in photochemical ozone depletion

category would the conclusion on prioritisation between the EoL technologies change; a higher efficiency would mean that recycling performs a little better than landfilling.

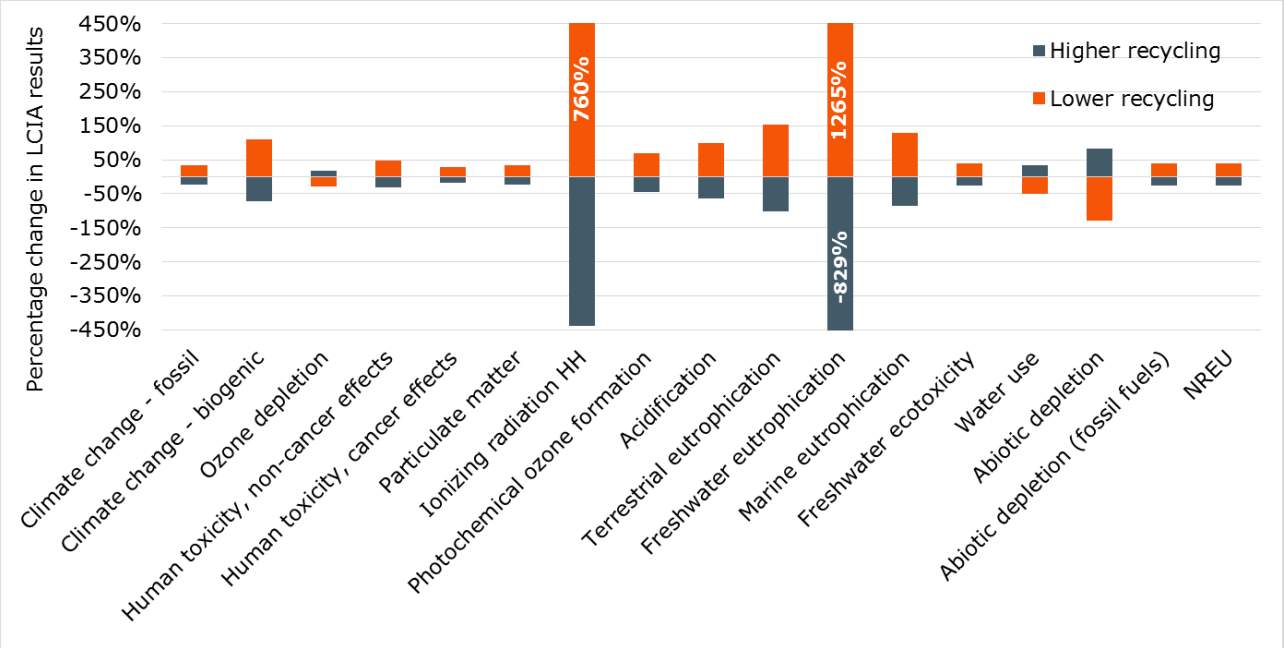


Figure 180. Sensitivity analysis for recycling of PLA packaging film compared to the EoL results of the PLA recycling.

The change in cradle-to-grave results, when changing the sorting and technology efficiency is presented in Table 199, for the seven selected comparable impact categories. The results in percentage change for these seven categories show that changing the recycling and sorting efficiency has a low impact on the overall cradle-to-grave results. Lowering the sorting and technology efficiency will result in an increase of total impacts from 1-5 %, where abiotic depletion (fossil fuels) has the largest percentage change. Increasing the technology efficiency will lead to a decrease of 1-3 % depending on the impact category.

Table 199. The change in percentage of higher and lower sorting and technology efficiency at the PLA recycling facility to the cradle-to-grave results of seven selected impact categories.

Impact category	Lower recycling	Higher recycling
Climate change	4 %	-2 %
Particulate matter	1 %	-1 %
Photochemical ozone formation	3 %	-2 %
Acidification	1 %	-1 %
Terrestrial eutrophication	1 %	-1 %
Abiotic depletion (fossil fuels)	5 %	-3 %
NREU	3 %	-2 %

4.2.3 Varying the amount of vegetable waste in the EoL reference flow

The amount of vegetable waste in the reference flow that enters the waste stream together with the packaging film can be an uncertain parameter. The effect of varying this parameter is analysed. Figure 181 presents the percentage change in the overall results for industrial composting compared to the original. The original vegetable food waste is 15 kg/kg PLA, the

lower limit is 4.3 kg/kg PLA and the higher is 25.6 kg/kg PLA. This implies that the EoL reference flow changes with food leftover, as a range of values were found in the literature.

The sensitivity analysis shows that the additional vegetable food waste has a substantial influence on the LCIA results for industrial composting. To increase the vegetable food waste up to the higher limit will cause the total results from composting to increase/decrease up to ±124 %, depending on the category. The biggest positive percentage change is in the category climate change – biogenic, where more vegetable waste will cause higher savings in terms of fertiliser substitution. The total climate change will however be negatively affected by the increase in vegetable waste. The non-renewable energy use is sensitive to the amount of material incoming to the industrial composting plant, as the energy use is per kg of waste treated.

Lowering the amount of vegetable food waste will lead to a percentage change in the overall results of -80 % to 74 %. Most impact categories will have a lower potential impact when the vegetable waste is decreased, as there is less amount to be treated. For other EoL pathways an increase in vegetable waste will likely have a larger impact.

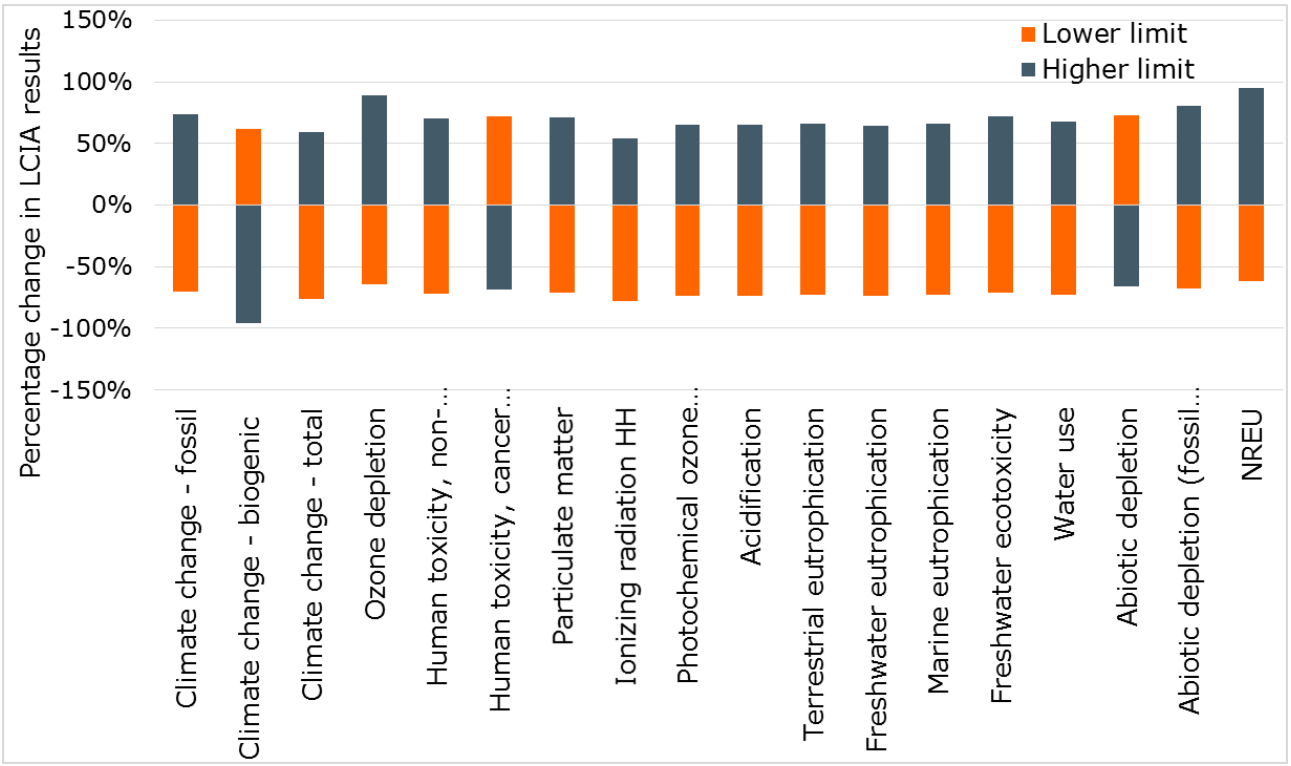


Figure 181. Sensitivity analysis for additional vegetable waste for composting PLA food packaging film, compared to the EoL results on PLA industrial composting.

The sensitivity analysis for the cradle-to-grave results when changing the amount of vegetable waste in the industrial composting technology is presented in Table 200. The sensitivity was calculated by varying the amount of vegetable waste that is treated in an industrial composting facility together with PLA packaging film. The results from increasing/decreasing the amount of vegetable waste in industrial composting are inserted to the EoL mix with other technologies, where the vegetable food waste has been kept to a total of 15 kg. However, it will be assumed that the vegetable waste would increase in all EoL technologies. The results of the sensitivity analysis indicate the importance of the amount of vegetable waste on the cradle-to-grave results. As presented in Table 194, the percentage change when lowering the amount of vegetable food waste will cause a decrease in all seven impact categories, with a percentage between 0.5-2.2 %. Increased use of resources at the composting facility, together with

increased emissions from the use of compost on land and fertiliser substitution will cause a change in the impact categories. The industrial composting however only contributes to 15 % of the EoL mix and the treatment of vegetables has a low impact in industrial composting compared with other EoL technologies.

Table 200. The change in percentage of higher and lower range for vegetable waste to the cradle-to-grave results of seven selected impact categories.

Impact category	Lower amount of vegetable waste	Higher amount of vegetable waste
Climate change	-0.5 %	0.01 %
Particulate matter	-1.1 %	1.1 %
Photochemical ozone formation	-1.4 %	1.3 %
Acidification	-1.9 %	1.7 %
Terrestrial eutrophication	-2.2 %	2.0 %
Abiotic depletion (fossil fuels)	-1.2 %	1.4 %
NREU	-0.5 %	0.7 %

4.2.4 PLA recycling with composting of food leftovers

A waste management system where the organic waste rejects from recycling are treated with industrial composting is a possible scenario, depending on the Member State. A sensitivity analysis is performed where the vegetable waste rejects are treated in an industrial composting facility instead of 53 % with incineration and 47 % with landfilling.

The results of the sensitivity analysis are presented in Figure 182, with percentage difference between the original scenario and with rejects to composting. The sensitivity analysis shows that in 10 impact categories, the impacts increase but in six impact categories the impact decreases. This change has a great effect on the impact categories of human toxicity, the non-cancer impact increases 3906 %, due to higher impacts from the use on land of compost. The human toxicity, cancer effects, decreases as the fertiliser substitution, gives a saving in chromium emissions to soil and water. The change in climate change impact shows a decrease of 97 %. Impact categories related to energy recovery increase, as composting does not provide any energy recovery.

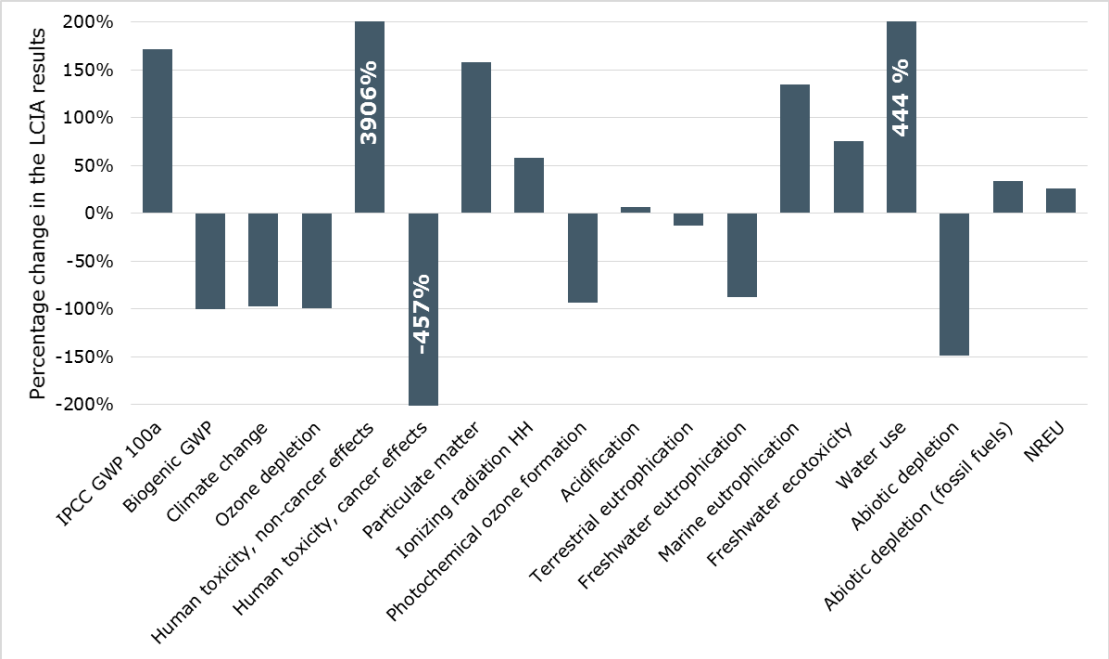


Figure 182. Sensitivity analysis for Recycling of PLA and changing the treatment of vegetable waste recycling rejects from incineration and landfill to industrial composting.

Table 201 shows the change on the overall LCIA cradle-to-grave results when changing the treatment of rejects from recycling. It can be concluded that this change will have a significant effect on the impact category climate change, with an 8.4 % decrease in the cradle-to-grave results. The photochemical ozone formation decreases by a total of 2.3 %, whereas other impact categories show a change of less than 1 %.

Table 201. The change in percentage of changing the treatment of vegetable waste recycling rejects from incineration and landfill to industrial composting to the cradle-to-grave results of seven selected impact categories.

Impact category	Change in cradle-to-grave results
Climate change	-8.4 %
Particulate matter	0.4 %
Photochemical ozone formation	-2.3 %
Acidification	0.0 %
Terrestrial eutrophication	-0.1 %
Abiotic depletion (fossil fuels)	0.8 %
NREU	0.5 %

4.3 Discussion on End of Life uncertainties

The parameters analysed in the sensitivity analysis above and other parameters, which may be uncertain, i.e., estimations, assumptions, huge ranges in industry or EU member states, missing data, etc. are discussed in this section. Methodological shortcomings and additional research requirements are also presented.

Quantity of vegetable waste that follows the packaging film. It is uncertain how much of the vegetables will enter the same waste system as the plastic packaging film. It is realistic that some consumers will throw both packaging and vegetables into the same bin. It is likely that more vegetable waste will be disposed of with bio-based plastic if consumers are aware of the

plastic's biodegradable properties. Other than a difference in density, there can therefore be a difference in the quantity of vegetable waste correlated to plastic type.

In other cases, the packaging will not include food leftovers, thus food leftovers would enter a different waste stream. The results show that including the food leftovers will positively affect the bio-based biodegradable product system due to appropriate EoL treatment for vegetable waste. Inclusion of food leftovers in the reference flow results in less favourable recycling, incineration and landfilling due to an assumption that organic waste is incinerated and landfilled.

Changing the quantity of vegetable waste solely for industrial composting gives a large impact in all impact categories for the EoL reference flow to industrial composting (see Figure 181). Changing this quantity in the cradle-to-grave analysis does not affect the results substantially, only up to 2 % (see Table 200), and industrial composting is still the favoured EoL technology.

For the other technologies, i.e. incineration, landfilling, recycling and anaerobic digestion, vegetable waste is assumed to be more influential on cradle-to-grave results.

Carbon biodegradation in industrial composting - This was analysed with a sensitivity analysis within industrial composting above. Results from the analysis, see Figure 179, show that this parameter will affect the GWP of industrial composting but is not sensitive to the cradle-to-grave results. Also, the level of VS degradation is uncertain in the industrial composting, but this was not investigated with a sensitivity analysis. Adjusting the level of VS degradation is not believed to have a major influence on the cradle-to-grave results.

PLA recycling – there is more than one uncertainty related to PLA recycling. Firstly, the technology is not commercialised yet, which is why there is no literature and little knowledge surrounding sorting and technology efficiency of PLA recycling. For the sake of this analysis it is assumed that it behaves like fossil polymers. As mentioned in the sensitivity analysis this is not seen to significantly impact the cradle-to-grave results, which can be viewed in Table 199 with a change of up to 5 %. The collection rate is also assumed and is, therefore, an uncertainty.

Secondly, recycled PLA is set to substitute PP, but in the future, recycled PLA might substitute virgin PLA. The environmental benefits of secondary PLA replacing virgin PLA are lower because impacts from producing PLA are lower than the production of PP.

In addition, food leftovers, which are included in the EoL reference flow, are assumed to be incinerated and landfilled in the recycling scenario. Citizens may instead sort food leftovers into the organic waste fraction heading to industrial composting, which is why a sensitivity analysis was conducted. This showed that different waste streams for food leftovers both increase and decrease the LCIA results. Landfilling/incineration as compared with industrial composting increases and decreases the LCIA results by up to 8 % (decrease in GWP), see Table 201.

Sorting and technology efficiency of recycling processes. The sensitivity analysis for the PLA sorting and technology efficiency shows that this parameter has a large impact on the LCIA results for the EoL recycling reference flow. Increasing the sorting and technology efficiency by 20% will decrease the cradle-to-grave GWP by 2 % (Table 199).

Vegetable waste to industrial composting instead of MSWI and landfill. In the study it is assumed that organic waste collected in connection with plastic packaging recycling of was sorted between landfill and incineration. It is viable that consumers send organic waste to industrial composting instead of incineration and landfill after collection. A sensitivity analysis was performed for this and the results showed an 8.4 % decrease in GWP. This scenario is probable in many European countries and would be consistent with the Circular Economy Package (European Commission, 2018a), which advocates not sending organic waste to landfill.

Reject quantity in industrial composting and AD is subject to uncertainties because pre-sorting and post-sorting at processing plants have varying technical set-ups. The quantity of rejected biodegradable plastic through the plants is prone to change if more biodegradable plastic enters the waste stream. This is outside of the temporal scope of this study.

Biodegradation in AD is uncertain because of the complexity of the EoL technology and because installed technologies at AD plants can vary. Several factors within AD plants will affect biodegradation, for example, temperature, moisture, time and post treatment practices. Chemical structure and plastic thickness will also affect the degradation process.

Biogenic carbon content in PLA and PP. PLA and PP studied in this case study are composed of 100 % biogenic carbon, although some PLA and PP will include fossil carbon from additives and co-polymers. This will affect the environmental profile, especially in the incineration EoL technology for GWP, where this would have different emission factors between fossil and biogenic carbon emissions.

Apart from the above case study's specific uncertainties, several other general uncertainties that apply to most of the presented case studies are discussed in the Approach and methodology in chapter 3.5., e.g., data quality, critical assumptions and limitations in End of Life modelling. These include uncertainties related to:

- the estimated EoL mix
- the marginal energy technologies utilised
- the development of the EoL technologies (efficiency increases)
- the consumption of materials and energy at the EoL treatment plants
- littering, which is not modelled in this LCA
- the chemical composition of the biodegradable plastics.

Sorting and technology efficiency of both polymers, PLA recycling as well as vegetable leftover quantity (for all EoL technologies excluding industrial composting) may pose significant uncertainties. Further research on organic waste quantities mixed with biodegradable plastic waste is required.

5 Assessing non-PEF impact categories: taking into account ILUC and endpoint of GTP

5.1 Baseline. Bio-Based PLA

5.1.1 Global temperature change potential

As regards climate change assessed with Global Temperature change Potential (GTP)100a as indicator (see Figure 183), it can be noticed that the cradle-to-user impact results are similar to climate change impacts in both absolute values and breakdown shares (Figure 186).

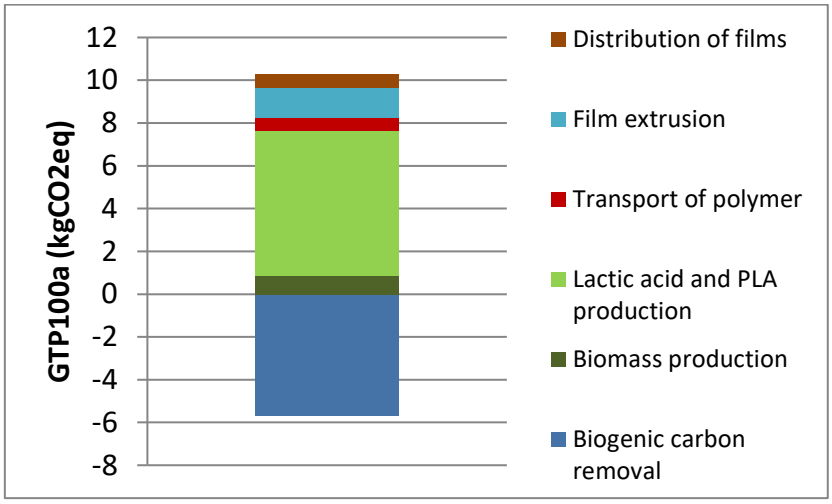


Figure 183. Breakdown of the cradle-to-gate GTP 100a results for baseline PLA packaging films, 1 functional unit, excluding DLUC and ILUC effects.

5.1.2 Land use change emissions

Table 202 presents the cradle-to-grave characterised results broken down into feedstock, manufacturing, EoL, and ILUC. As highlighted in Chapter 4, DLUC and ILUC cannot be summed (it is whether DLUC or ILUC). This case does not involve crops that imply DLUC (namely Brazilian sugarcane, German maize, French wheat and German wheat), on the basis of the methodology from the PEFCR Guidance V6.3 presented in Chapter 4. Figure 184 presents the characterised results for case study 6 (baseline).

Table 202. Characterised cradle-to-grave results broken down per process, including ILUC. Case study 6 Bio-based baseline and EU mix EoL scenario

Impact category	Unit	Biomass produc-tion (incl. biogenic C)	Manufacturing	EoL (EU mix)	ILUC
Climate change	kg CO_2 eq.	-4.72E+00	9.83E+00	4.36E+01	1.73E+00
Ozone depletion	kg CFC-11eq	9.31E-09	5.51E-07	1.71E-05	3.61E-10
Human toxicity, non-cancer effects	CTUh	-8.74E-07	3.14E-06	1.72E-06	2.89E-10
Human toxicity, cancer effects	CTUh	9.59E-09	5.84E-07	2.98E-08	4.60E-11
Particulate matter	kg PM2.5eq	2.91E-04	3.94E-03	8.06E-05	4.56E-05
Ionizing radiation, human health	kBq U235eq	6.92E-02	6.99E-01	1.72E-01	2.49E-05
Photochemical ozone formation	kg NMVOCeq	1.17E-02	2.79E-02	3.20E-02	5.37E-03
Acidification	molc H+eq	1.12E-02	5.14E-02	2.79E-02	1.08E-03
Terrestrial eutrophication	molc Neq	5.68E-02	1.05E-01	1.27E-01	6.04E-03
Freshwater eutrophication	kg Peq	1.03E-04	4.64E-04	4.85E-03	3.98E-06
Marine eutrophication	kg Neq	1.86E-02	8.19E-03	1.29E-02	6.42E-04
Freshwater ecotoxicity	CTUe	3.26E+01	1.56E+01	4.23E+00	3.21E-03
Land use	kg C deficit	4.76E+01	1.60E+01	0.00E+00	4.97E-03
Water use	m3	1.98E+00	4.05E+00	4.07E-01	5.32E-02
Abiotic depletion	kg Sbeq	1.02E-06	2.40E-05	-1.51E-06	1.57E-08
Abiotic depletion (fossil fuels)	MJ	8.25E+00	1.28E+02	-2.58E+01	4.49E-02
NREU	MJ	8.88E+00	1.52E+02	-3.06E+01	4.29E-02

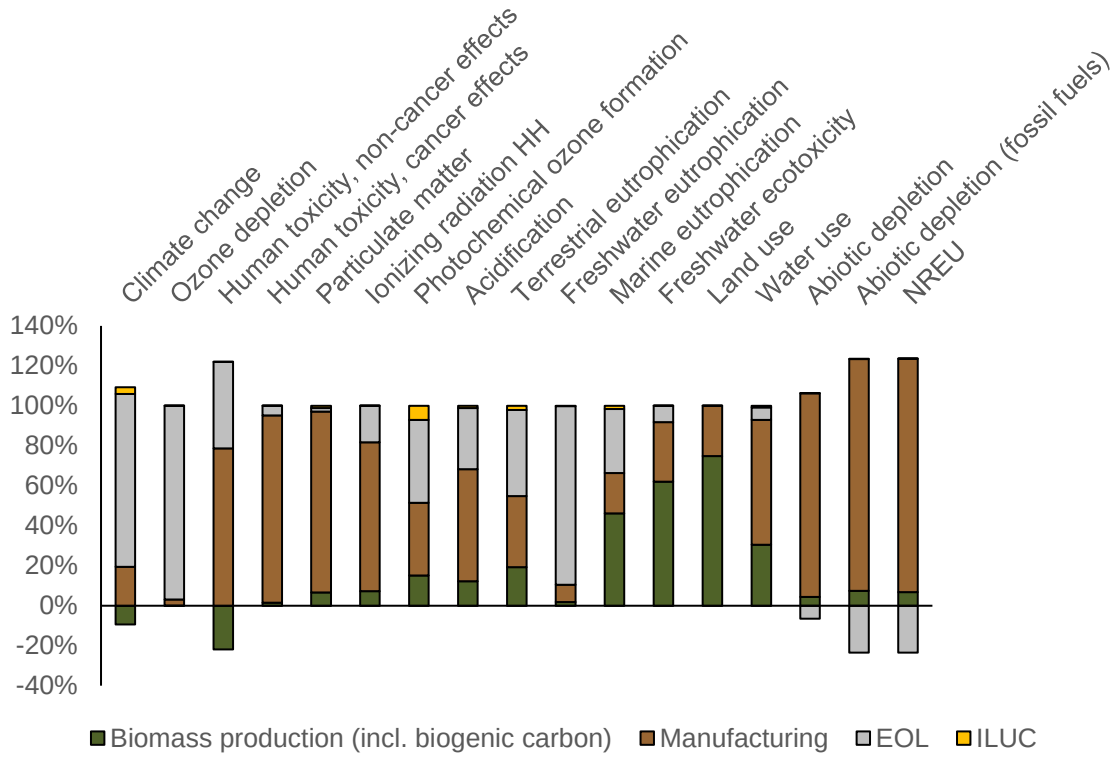


Figure 184. Relative characterised results broken down for all impact categories and including ILUC – case study 6, baseline.

As can be seen from Table 202 and to some extent from Figure 184, ILUC does contribute to all impact categories considered herein, as in previous cases. However, as for previous cases, the contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (3.4 % of the impact), photochemical ozone formation (7 % of the impact), terrestrial eutrophication (2 % of the impact), marine eutrophication (1.6 % of the impact), acidification (1.2 % of the impact) and particulate matter (1 % of the impact). All these impacts are dominated by land expansion (Figure 15). In the case of climate change, as explained earlier, the ILUC impact is essentially due to the CO_2 release resulting from land clearing and in the case of photochemical ozone formation, to carbon monoxide also released during land clearing (burning the cleared biomass; Figure 17). Both acidification, terrestrial- and marine eutrophication are essentially due to nitrogen oxides released during land clearing (more than 80 % of the impact; Figure 17). The particulate matter impact is due to both the carbon monoxide and nitrogen oxides released during land clearing.

In absolute terms, the ILUC impact for climate change here is quite high compared to the one obtained for e.g. case 1 but is still lower than some of the other cases (e.g. case study 2 or case study 5). This reflects the importance of having conversion processes requiring as little biomass as possible, so the climate change impacts from both the crop cultivation and the ILUC can be lower. It is also noteworthy to highlight that the contribution of ILUC (as a process) to photochemical ozone formation is ca. twice as much as for climate change, unlike all other cases. This shows that the other processes (here EoL, in particular) contribute much more to the climate change impact than ILUC.

5.2 Alternative 1. PLA from EU maize

5.2.1 Global temperature change potential

As regards climate change assessed with Global Temperature change Potential (GTP)100a as indicator, it can be noticed that the cradle-to-user impact results are similar to climate change in both absolute values and breakdown shares (Figure 185).

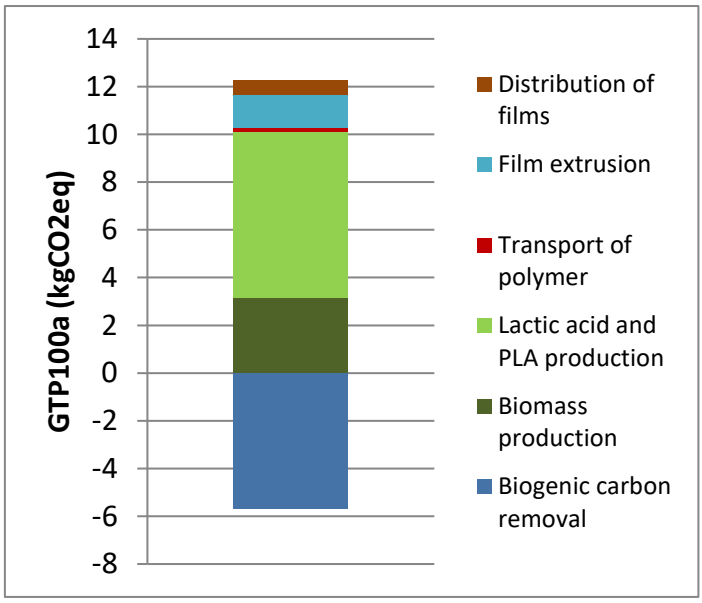


Figure 185. Breakdown of the cradle-to-gate GTP 100a results for EU PLA packaging films, 1 functional unit, excluding DLUC and ILUC effects.

5.2.2 Land use change emissions

Table 203 presents the cradle-to-grave characterised results broken down into feedstock, manufacturing, EoL, DLUC and ILUC. As highlighted in Chapter 4, DLUC and ILUC cannot be summed up (it is whether DLUC or ILUC). DLUC is also presented as it is a "process" that should be considered in the LCA, according to the latest PEFCR Guidance v6.3 (European Commission, 2018). DLUC was calculated as specified in the methodology of the PEFCR Guidance v6.3, as detailed in Chapter 4. Figure 186 presents the characterised results for this variant of case study 6 with the inclusion of ILUC.

Results with the inclusion of DLUC are not shown, as they are rather negligible (Table 203) and barely visible in an illustration such as Figure 186 (contributing to climate change impact only and making 0.0008 % of the impact, when included). As explained in Chapter 4, DLUC is based on the 20-year history of crops in a given country. For the feedstock considered in this study, DLUC is only observed for a few crops, namely Brazilian sugarcane, German maize, French wheat and German wheat. Here, only one of these crops is involved (German maize). The reason why DLUC is insignificant here is reflected in Table 25 of Chapter 4 and is explained by the rather minor amount of German maize per functional unit.

Table 203. Characterised cradle-to-grave results broken down per process, including ILUC. Case study 6, alternative with EU maize.

Impact category	Unit	Biomass production (incl. biogenic C)	Manufacturing	EoL (EU mix)	ILUC	DLUC
Climate change	kg CO_2 eq.	-2.43E+00	9.52E+00	4.36E+01	2.86E+00	3.7E-04
Ozone depletion	kg CFC-11eq	1.44E-07	2.87E-07	1.71E-05	5.97E-10	
Human toxicity, non-cancer effects	CTUh	4.18E-06	3.39E-07	1.72E-06	4.79E-10	
Human toxicity, cancer effects	CTUh	8.92E-08	8.05E-07	2.98E-08	7.61E-11	
Particulate matter	kg PM2.5eq	2.26E-03	1.84E-03	8.06E-05	7.54E-05	
Ionizing radiation, human health	kBq U235eq	1.74E-01	8.12E-01	1.72E-01	4.11E-05	
Photochemical ozone formation	kg NMVOCeq	1.07E-02	1.74E-02	3.20E-02	8.89E-03	
Acidification	molc H+eq	6.15E-02	2.88E-02	2.79E-02	1.78E-03	
Terrestrial eutrophication	molc Neq	2.33E-01	6.12E-02	1.27E-01	9.99E-03	
Freshwater eutrophication	kg Peq	8.37E-04	1.33E-04	4.85E-03	6.59E-06	
Marine eutrophication	kg Neq	4.85E-02	6.06E-03	1.29E-02	1.06E-03	
Freshwater ecotoxicity	CTUe	7.19E+01	1.03E+01	4.23E+00	5.31E-03	
Land use	kg C deficit	7.31E+01	6.39E+00	0.00E+00	8.22E-03	
Water use	m3	2.64E+01	3.98E+00	4.07E-01	8.80E-02	
Abiotic depletion	kg Sbeq	1.22E-06	2.02E-05	-1.51E-06	2.60E-08	
Abiotic depletion (fossil fuels)	MJ	2.61E+01	1.23E+02	-2.58E+01	7.43E-02	
NREU	MJ	3.11E+01	1.53E+02	-3.06E+01	7.10E-02	

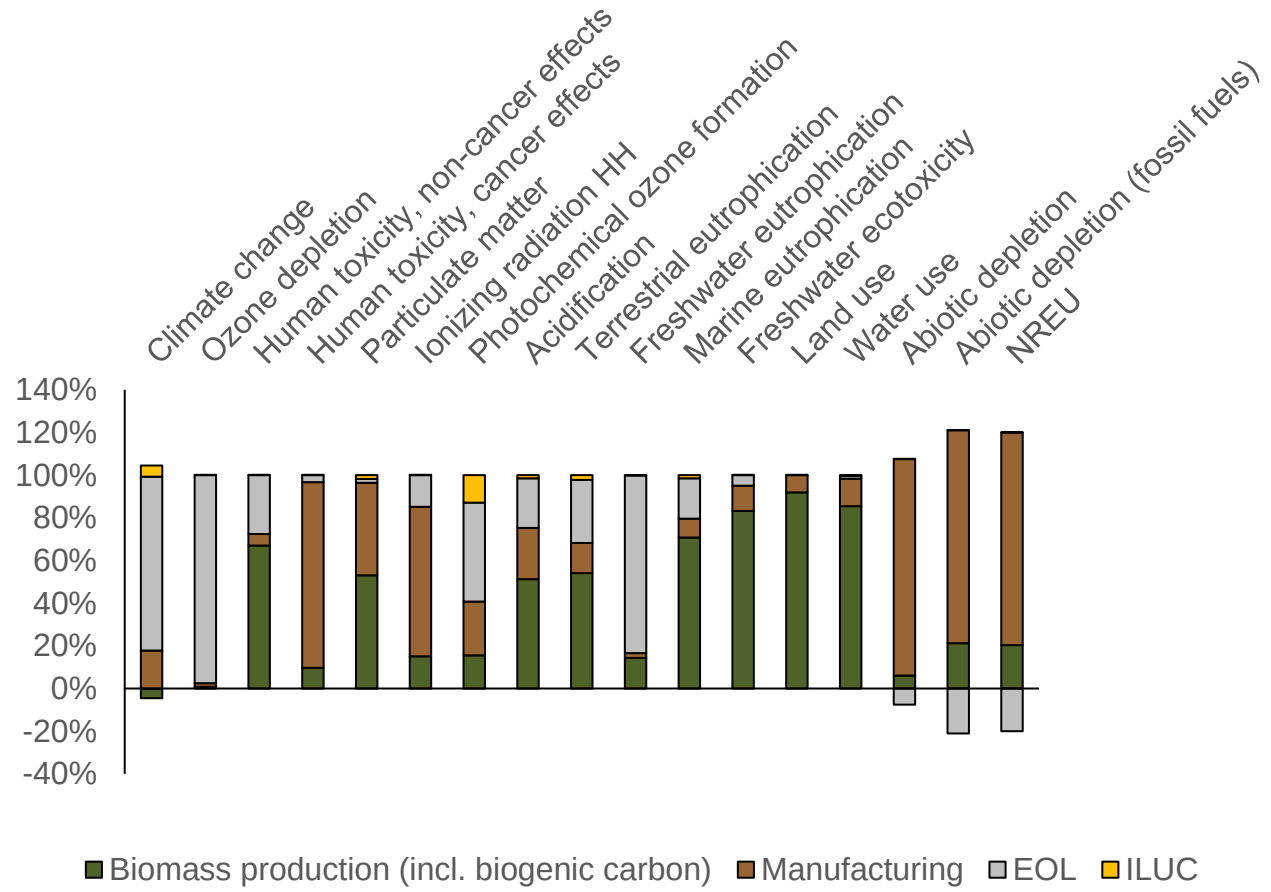


Figure 186. Relative characterised results broken down for all impact categories and including ILUC – case study 6, alternative with EU maize.

As can be seen from Table 203 and to some extent from Figure 186, ILUC does contribute to all impact categories considered herein, as in previous cases. However, as for previous cases, the contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (5 % of the impact), photochemical ozone formation (13 % of the impact), terrestrial eutrophication (2 % of the impact), marine eutrophication (2 % of the impact), acidification (1.5 % of the impact) and particulate matter (1.8 % of the impact). All these impacts are dominated by land expansion (Figure 15). In the case of climate change, as explained earlier, the ILUC impact is essentially due to the CO_2 release resulting from land clearing and in the case of photochemical ozone formation, to carbon monoxide also released during land clearing (burning the cleared biomass; Figure 17). Both acidification, terrestrial- and marine eutrophication are essentially due to nitrogen oxides released during land clearing (more than 80 % of the impact; Figure 17). The particulate matter impact is due to both the carbon monoxide and nitrogen oxides released during land clearing. In absolute terms, the ILUC impact for climate change is not as high as for e.g. case study 2 or Case study 5, but it is approximately 1.5 times higher than the baseline impact. As in the baseline case, the contribution of ILUC (as a process) to photochemical ozone formation is more important than for climate change (here it is 3 times higher for photochemical ozone formation than for climate change), unlike all other cases. This shows that for this case, the other processes (here EoL, in particular) contribute much more to the climate change impact than ILUC.

6 Conclusions, limitations and recommendations

The cradle-to-grave environmental impact of different packaging films has been assessed through the LCA methodology. PLA is one of the most important bio-based materials for film packaging and it has been the focus of the study. The current-near future commercial PLA has been considered as the baseline material under investigation and has been compared to the petrochemical reference for this application (PP) and two alternative bio-based materials (PLA from EU maize and bio-based PP from UCO).

For bio-based packaging films made of PLA, the most important environmental impacts are climate change (41-53 %) and when the impact categories related to toxicity are also considered, human toxicity-cancer effects also become a significant impact (14 %). Independently of whether toxicity impact categories are accounted for or not, EoL is the main source of environmental burden (49-60 %). The current-near future EoL mix for PLA has been defined as 15 % recycling, 15 % industrial composting, 39 % incineration and 31 % landfilling. This is followed by lactic acid and PLA production (23-35 %) while biomass production and transportation are responsible of minor impacts. Landfill is the main source of the high impact of EoL, due to the significant amount of greenhouse gas emissions released. This is also reflected in the life cycle environmental impact of the other analysed polymers for which the share of landfill in the EoL mix has also been fixed to 39 %. The main reason for this high impact of EoL for packaging films is due to the high amount of food waste (15 kg per kg of plastic) that is collected together with the films; landfill is the worst EoL option when such a high amount of food waste is collected with plastic items. Energy consumption and chemicals used in the manufacturing process are the major sources responsible for the impact of lactic acid and PLA production process.

For the baseline PLA, cups made from sugarcane from Thailand and maize from the US, there is no visible DLUC. ILUC does contribute to all impact categories considered herein, as in previous cases. However, as for previous cases, the contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (4 % of the impact), photochemical ozone formation (8 % of the impact), terrestrial eutrophication (2 % of the impact), marine eutrophication (2 % of the impact), acidification (1.2 % of the impact) and particulate matter (1 % of the impact). All these impacts are dominated by land expansion.

Out of the six impact categories identified as 'suitable for comparison' (See Chapter 6), from cradle-to-grave, PLA films undoubtedly offer better environmental performances in terms of abiotic depletion of fossil fuels (-50 %) and climate change (-15 %) compared to petrochemical PP. PLA packaging films have however higher impacts compared to petrochemical PP films in the other four impact categories which are strongly associated with agricultural production and lactic acid production namely; photochemical ozone formation (+10 %), particulate matter (+50 %), acidification (+60 %) and terrestrial eutrophication (+45 %). As a result of this, there is no a best/worst material between PLA and PP packaging films. For both the polymers, the cradle-to-grave environmental impact will significantly decrease in the future thanks to a decreasing use of landfill at EoL. In particular, the impact of PLA films would significantly improve if the share of the intended EoL option (i.e. industrial composting) significantly increased. In a futuristic scenario, if 100 % of PLA films could be composted, PLA would perform better than petrochemical PP films due to the large amount of food waste that contaminate the packaging films and that would be diverted from other waste management practices.

The modelled EU-maize based PLA does not offer a better environmental profile compared to the existing-commercialised PLA. If the average maize currently cultivated in Europe were

used for PLA production, no environmental savings would be obtained from avoiding the global transportation of the polymer.

For EU-maize based PLA, as for the baseline PLA, there is no visible DLUC impact. The contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (5 % of the impact), photochemical ozone formation (13 % of the impact), terrestrial eutrophication (2 % of the impact), marine eutrophication (2 % of the impact), acidification (1.5 % of the impact) and particulate matter (1.8 % of the impact). All these impacts are dominated by land expansion.

About the second alternative bio-based material which is assessed -bio-based PP derived from used cooking oil-, better environmental performances are shown compared to all the analysed polymers suitable for this application. The main reason of the remarkable environmental performances of the analysed bio-based PP is the usage of a vegetable waste oil.

The EoL mix of both product systems have been weighted with and without toxicity. The weighted results show that the PLA packaging film has a lower weighted score, hence it is the preferred choice. The difference in the scores is only 10 % with toxicity and lower without toxicity. A small change in the EoL mix can therefore influence which product system is preferable.

The EoL sensitivity analysis looking at the EoL pathway for the food leftovers, changing the recycling scenario from incineration and landfilling to industrial composting, when recycling the PLA film, is sensitive to the results. The cradle-to-grave results decreased for three of the selected seven impact categories, primarily climate change with 8 %, and increased the cradle-to-grave results slightly (less than 1 %) for the other four impact categories.

The other EoL sensitivity analysis' does not show a major impact onto the cradle-to-grave results (less than 5 % looking at the seven selected impact categories), based on changing the biogenic carbon degradation rate in the industrial composting facility (varied from 55 % to 95 %), changing the sorting and technology efficiency at the PLA recycling facility (between 39 % and 90 %) and changing the amount of vegetable food leftovers of the reference flow in the industrial composting scenario of bio-based PLA (varied from 4.3 kg to 25.6 kg).

DLUC applies only in the variant with EU crops but is rather negligible. ILUC does contribute to all impact categories considered herein, as in previous cases. However, as for previous cases, the contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (3.4 - 5 % of the impact), photochemical ozone formation (7 - 13 % of the impact), terrestrial eutrophication (<2 % of the impact), marine eutrophication (<2 % of the impact), acidification (<2 % of the impact) and particulate matter (<2 % of the impact). All these impacts are dominated by land expansion.

Littering is excluded from this case study due to lack of data and knowledge of the impact assessment models. The exact amount (and relative share) of plastics that are littered is uncertain for a specific application and will differ widely between countries. The impacts of littering and microplastics are currently not properly captured in the toxicity-related impact categories (ecotoxicity and human toxicity). As pointed out by Pant et al. (2018) fate, exposure, and effect modelling for macro- and micro-plastics (including additives) is still in its infancy. Physical impacts (e.g. entanglement; ingestion of larger plastic particles and their effects) and chemical impacts (e.g. due to microplastic formation, microplastics as carrier of other chemicals, unknown impact of additives) and biological impacts (e.g. microplastics as carrier of germs/alien species) are largely unknown and currently not included in LCA. Thus, the impacts of littering (e.g. leaving plastics in the soil or in a marine environment) are most likely underestimated in the current LCA for the non-degradable applications.

CASE STUDY 7: SINGLE-USE PLASTIC CARRIER BAGS

1 Goal and scope definition

1.1 Goal and background

The goal of this case study is to assess the environmental profile of bio-based carrier bags production and compare it with the fossil-based counterpart. Traditionally, plastic carrier bags are made with low-density polyethylene which is not compostable. Bio-based bags can offer the same functionality (strength and toughness) while being in some cases biodegradable and compostable. To achieve compostability, the material should be certified according to the EN 13432 standard. Compostable bags could offer advantages through reuse for organic waste collection. The use of these compostable bags offers improved collection of organic waste (Oever et al., 2017). Currently, much kitchen waste ends up in general waste and is incinerated or landfilled. With the use of compostable bags, the organic waste e.g. food waste can be redirected from incineration to composting, although this is dependent on consumer behaviour and proper sorting of waste. It is important to properly inform consumers to stimulate them to use compostable bags for organic waste management. These compostable bags should not result in additional contamination in the compost produced (ISWA, 2015).

Bio-based carrier bags are produced from starch plastics. Starch plastics represent 10.3 % of the global production capacity of bio-based plastics (Aeschelmann et al., 2017). Europe has a solid position in the production and commercialization of starch blends with several companies being active in this sector, such as Novamont, Rodenburg, Biotec, FKuR and Kingfa (Aeschelmann et al., 2017). Application of starch plastics include flexible packaging, carrier bags, disposable tableware, agriculture and horticulture (mulch films, clips, pots) and loose-fill packaging foams. Carrier bags make up the majority of the consumption (Kaeb et al., 2016).

Starch plastics are blends of starch with one or more polymers (fossil or bio-based polyesters). First, native starch is extracted from starch crops such as maize and potato before it is modified into thermoplastics starch (TPS), which is ready to be blended into starch plastic granules. Depending on the industrial application and required technical properties, TPS is blended via reactive extrusion with complexing agents, which are often fossil and/or bio-based polyesters, e.g. polylactic acid (PLA, bio-based) and/or polybutyrate adipate-co-terephthalate (PBAT, fossil fuel based), to produce starch plastics granules, sometimes also called "polyester-complexed starch biopolymers" (Broeren et al., 2017a). To achieve good adhesion between the polymers, compatibiliser additives are added in the blended extrusion. The starch plastics granules are then converted into final products via various processing technologies, e.g. film extrusion to produce carrier bags.

1.2 Scope

As defined in Chapter 2 "*Approach and methodology*", in this study we aim to analyse carrier bags that are purchased, used and disposed of in Europe. The manufacturing chains including biomass supplies and plastic conversions, can extend globally. For the baseline system, maize and potato, as the sources of the starch, are both cultivated in Europe. PLA, used as copolyester, is produced in the U.S. and shipped to Europe. PLA is made from maize cultivated in the U.S (see Case Study 3 Single-use cups). Other copolyesters and additives are sourced from Europe. Blending the ingredients into the starch plastic granules and extruding the granules into films to produce carrier bags both take place in Europe. For the alternative bio-based LDPE product system, the cultivation of sugarcane and its conversion to bio-based LDPE takes place in Brazil. The bio-based LDPE granules are then transported to Europe and converted into carrier bags.

The temporal scope is current production (2017-2018) with relevant developments foreseen for the near future (5-10 years). The technical scope is a range of commercially available technologies.

1.3 Function and functional unit

The primary function is defined here as carrying groceries.

The functional unit of this case is defined as:

- One single-use all-purpose lightweight plastic carrier bag with the volume of 20 liters and 10 kg weight holding capacity

- The petrochemical counterpart of a bio-based carrier bag is a lightweight plastic carrier bag made of low-density polyethylene (LDPE). The average weight of an LDPE carrier bag to fulfil the defined functional unit is 17.9 g (The Danish Environmental Protection Agency, 2018). Based on personal communication with industrial partners, to fulfil the same functional unit, the bio-based starch biopolymer bag is assumed to be about 25 % heavier and therefore weighs 22.4 g. It should be noted the bio-based carrier bags have the same load capacity (i.e. equivalent strength), they are sometimes less resistant to puncturing and tearing. It could be less suitable for groceries with sharp edges. The bio-based LDPE is identical with petrochemical LDPE and the weight to fulfil the defined functional unit is also the same: 17.9 g.

1.4 Product systems

The baseline for the bio-based product system is polyester-complexed starch biopolymer carrier bags produced from European crops (maize and potato). In the starch plastics industry, very often companies produce starch blends based on their own technologies. The environmental impacts are analysed based on each technology; here, company-specific data are used. Aggregated data are then constructed by making a weighted average based on the market share of the companies. The market share of starch-based plastics from maize is two thirds, whereas the remaining one third of the market is composed of potato starch-based plastics.

An alternative bio-based product system is bio-based LDPE. It is produced from Brazilian sugarcane ethanol. Bioethylene production and polymerization to bio-based LDPE takes place in Brazil. It is then transported to Europe and converted into carrier bags.

The reference petrochemical system is LDPE carrier bags produced in Europe.

1.5 System boundaries

A cradle-to-grave approach is taken including the life cycle stages of feedstock production, manufacturing and EoL. The consumer use phase is excluded from the analysis as it is the same for both product systems and has a negligible impact. Thus, the life cycle can be divided into cradle-to-gate and EoL.

2 Life cycle inventory analysis

2.1 Process description, data, assumptions and multifunctionality

2.1.1 Baseline: Bio-Based carrier bags from maize/potato

For this process, data is obtained from the largest industrial starch-based bioplastic producers, one producer based on maize starch and another based on potato starch. The biodegradable and bio-based polymer grades used for carrier bag applications have the following properties. They are biodegradable thermoplastic materials made with maize/potato starch and biodegradable copolyesters based on proprietary technology. The copolyester can be bio-based, fossil based or a mix of both. For starch plastics made from potato, two copolyesters are blended: namely, PLA (bio-based) and PBAT (petrochemical) (Broeren et al. 2017a). For starch plastics made from maize, the copolyester is fossil fuel-based. Proprietary technology is investigated, and the ingredients are confidential. Maize is cultivated in Italy. Potato is cultivated Germany. The harvested starch crops are converted to native starch and shipped to the starch plastics granulate production. Copolyester and other raw materials are produced in Europe from bio and fossil resources and supplied to the plastic producer. The starch-based bioplastic production consists of a combination (by means of extrusion) of the native maize/potato starch with the copolyesters and other raw materials (i.e. additives such as compatibilisers and plasticisers). Possible plastic scraps from the manufacturing process are directly recycled in the extrusion process. The thermoplastic granules (plastic pellets), ready to be processed into the final application, are then sent to other economic operators to be converted into carrier bags through blown film extrusion technology.

The process flow diagram of the bio-based carrier bag product system is shown in Figure 187 for maize-based carrier bags and in Figure 188 for potato-based carrier bags.

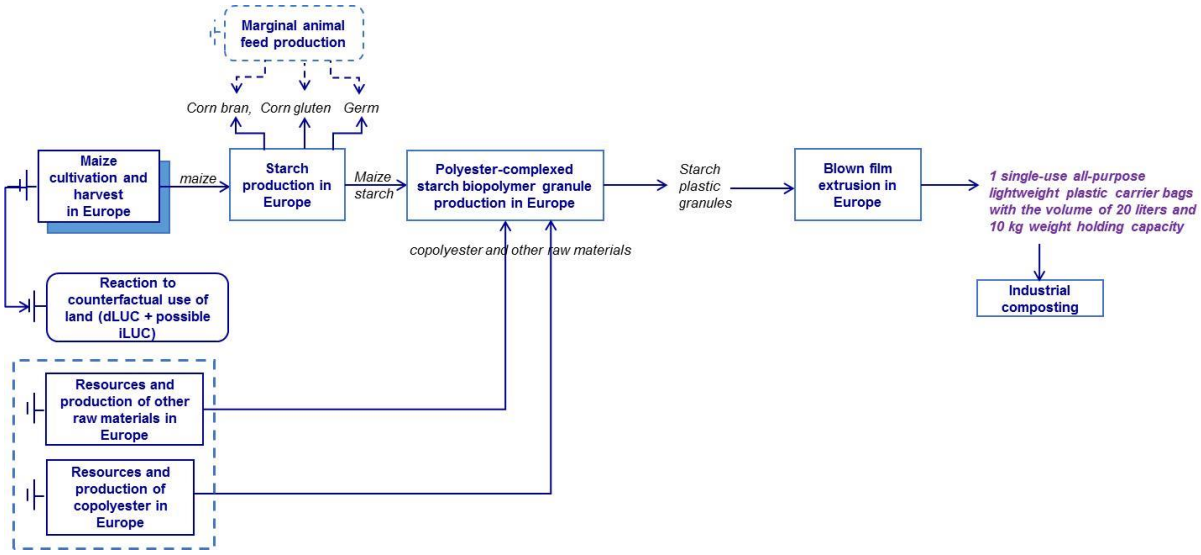


Figure 187. Process flow diagram of bio-based carrier bag made from maize starch.

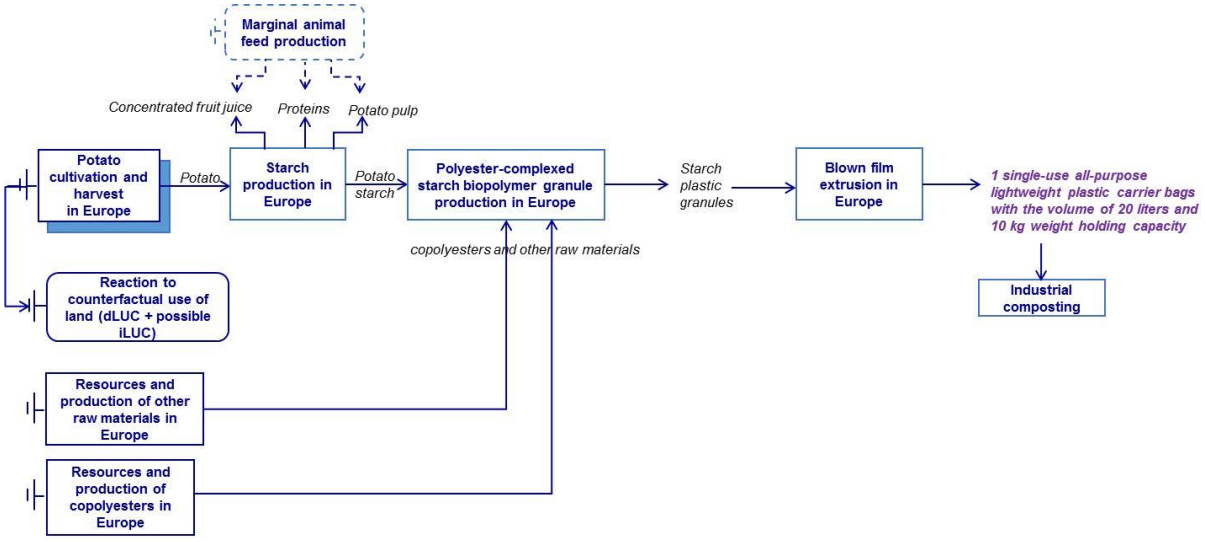


Figure 188. Process flow diagram of bio-based carrier bag from potato starch.

2.1.1.1 Bio-Based carrier bag from maize

Ecoinvent 3.3 data *Polyester-complexed starch biopolymer {RER}| production | Alloc Rec, U* is used to calculate the environmental profile of film grade biodegradable plastic.

The processes involved can be divided into the following key unit processes:

2.1.1.1.1 Maize cultivation and harvest

For maize cultivation and harvest, the same inventory data as the mulch film case is assumed (see case study 5, 2.1.1.1). The data reflects maize produced in Italy. From the elemental composition of the polyester-complexed starch biopolymer, the biogenic carbon content is used to calculate the biogenic carbon embedded in the bio-based carrier bag (0.41 kg CO_2/kg starch-based polymer).

2.1.1.1.2 Maize starch production

For maize starch production, the same inventory data as the mulch film case is assumed (see case study 5, 2.1.1.1). The data reflects maize starch produced in Italy in a wet milling process. The starch is extracted and then refined. Maize bran, maize gluten and germ are co-products of the starch extraction process. These are used as animal feed.

2.1.1.1.3 Maize starch transportation

The maize starch is transported to the granulate production site by lorry. For the specific transportation distance, it is taken to be same as the mulch film case (see case study 5, 2.1.1.1).

2.1.1.1.4 Copolyester and other natural and renewable raw materials (i.e. additives such as compatibilisers and plasticisers) production

The environmental burdens regarding the fossil fuel-based copolyester have been approximated by the inputs of naphtha, natural gas and coal suggested by the unit process in the Ecoinvent 3.3 *Polyester-complexed starch biopolymer {RER}| production | Alloc Rec, U* dataset. The specific information regarding the type of the copolyester is not provided by the

database. The breakdown between copolyester and additives is not available. However, the majority of the impacts of this unit process is contributed by the copolyester production, with additives and other raw materials contributing only marginally.

2.1.1.1.5 Copolyester and other raw materials transportation

The copolyester and the other raw materials produced are transported to the granulate production site by lorry. The specific transportation distances are taken to be same as the mulch film case (see case study 5, 2.1.1.1).

2.1.1.1.6 Polyester-complexed starch biopolymer granulate production

Native starch is blended and extruded with copolyester and other natural and renewable raw materials to form a bio-based biodegradable thermoplastic in the form of granulate based on proprietary technology. The impacts of granulate production are mainly associated with electricity consumption and waste treatments within the Ecoinvent 3.3 *Polyester-complexed starch biopolymer {RER}| production | Alloc Rec, U* dataset. The "gate to gate" electricity consumption in this process was corrected from 0.555 kWh/kg (in Ecoinvent) to 0.402 kWh/kg based on a publicly available Environmental Product Declaration (Novamont, 2004). The water consumption and wastewater treatment were not provided in this dataset and are additionally included.

2.1.1.1.7 Granulate transportation

The starch-based biodegradable bioplastic granulate produced is transported to the converter site by lorry. Average approximate transportation distance between the production site and the film producers is considered.

2.1.1.1.8 Blown film extrusion

Carrier bags are produced through blown film extrusion of the starch-based granulates. With 1 kg of starch-based granulate, 0.995 kg of carrier bag is obtained. The key activity level data is electricity consumption which is 1.07 kWh/kg (Environmental Agency, 2011).

The foreground data sources used for the bio-based carrier bags from maize are summarised in Table 204. For background data, e.g. grid electricity and heat, other utilities and production of chemicals and materials, Ecoinvent 3.3 is used.

Table 204. Data used in the bio-based carrier bags from maize.

Processes	Data sources	Comments
Maize cultivation and harvest	See case study 5, section 2.1.1.1	Represents production of maize in Italy.
Maize starch production	See case study 5, section 2.1.1.1	Represents production of maize starch in Italy.
Copolyester (fossil-based) and other natural and renewable raw materials production	Ecoinvent 3.3	Naphtha, natural gas and coal inputs provided within Ecoinvent 3.3 *Polyester-complexed starch biopolymer {RER}| production | Alloc Rec, U* dataset
Transportation (of maize starch, copolyester and other raw materials)	See case study 5, section 2.1.1.1	Represents distances from the suppliers to the granulate production site. All transportation is within Europe by lorry.
Maize-starch based granulate production	Ecoinvent 3.3 & Novamont, 2004	Electricity consumption and waste treatments within *Ecoinvent 3.3 Polyester-complexed starch biopolymer {RER}| production | Alloc Rec, U* dataset. Electricity consumption corrected from 0.555 kWh/kg to 0.402 kWh/kg granulate based on Novamont, 2004.
Granulate trasportation	See case study 5, section 2.1.1.1	Represents average distance from granulate production site to extrusion site. Transportation is within Europe by lorry.
Blown film extrusion	Environmental Agency, 2011	Key activity level data is electricity consumption: 1.07 kWh/kg extruded plastic.

2.1.1.1.9 Multifunctionality

Maize starch production process involves co-production of maize bran, maize gluten and germ. To avoid allocation, the system is expanded to include their animal feed replacement. The co-products are displaced with a mix of marginal feed ingredients with the same standardised feed unit. The marginal protein feed is taken as soybean meal, the marginal carbohydrate is taken as maize, and the marginal oil is taken as palm oil, based on Tonini et al. (2016). These choices are based on detailed elaboration of recent demand trends and future projections (FAPRI, 2012; FAOSTAT, 2014).

2.1.1.2 Bio-based carrier bag from potato

For the potato-based starch plastic the processes involved can be divided into the following key unit processes:

2.1.1.2.1 Potato cultivation and harvest

Potatoes are sourced mainly from Germany. The Agrifootprint database is used for the inventory data of potato production in Germany. From the composition of the polyester-complexed starch biopolymer provided by the plastic producer, the biogenic carbon content is used to calculate the biogenic carbon embedded in the bio-based carrier bag (1.09 kg CO_2 eq./kg potato starch-based polymer).

2.1.1.2.2 Potato starch production

After harvesting, potatoes are cleaned and grinded into pulp. The starch slurry and juice are then separated. The juice is further processed to produce proteins and concentrated fruit juice. Starch slurry is refined and dried to produce native starch to be sent to granulate production.

Concentrated fruit juice, proteins and pulp are obtained as co-products of the starch production process. The Agrifootprint database is used for the inventory data reflecting a typical potato wet milling process in Europe. Drying of the starch to the required moisture level is included in the inventory.

2.1.1.2.3 Potato starch transportation

The potato starch is transported to the granulate production site by lorry.

2.1.1.2.4 Copolyesters production

Bio-based PLA from maize and petrochemical polybutylene adipate-co-terephthalate (PBAT) are used as copolyesters. For PLA, data is obtained from the PLA producer located in the US. Please refer to case study 3 for further information. For PBAT, data was not available in the databases. It was modelled using stoichiometric amounts of 1,4-butanediol, adipic acid and purified terephthalic acid (PTA). For polymerization of the monomers to PBAT it was taken to be equal to PET polymerization. The same data was used for PTA and polymerization as in case study 1 (PET beverage bottles). The data for 1,4-butanediol and adipic acid was obtained from the Ecoinvent 3.3 database.

2.1.1.2.5 Copolyesters transportation

The copolyesters are transported to the granulate production site by combination of lorry, bulk carrier and freight train.

2.1.1.2.6 Other raw materials (i.e. additives such as compatibilisers and plasticisers) production

Other renewable and non-renewable raw materials (additives) are produced in Europe. Data from the Ecoinvent 3.3 database is used for them.

2.1.1.2.7 Other raw materials transportation

Other raw materials used in granulate production are transported by lorry to the granulate production site.

2.1.1.2.8 Polyester-complexed starch biopolymer granulate production

Starch is combined with copolyesters and other raw materials to get a thermoplastic bio-based compostable material (i.e. granulate) based on proprietary technology. The key activity level data are the electricity consumption and waste treatments. Foreground data is based on Broeren et al. (2017a).

2.1.1.2.9 Granulate transportation

The starch-based biodegradable bioplastic granulate produced is transported to the converter site by lorry.

2.1.1.2.10 Blown film extrusion

Carrier bags are produced through blown film extrusion of the starch-based granulates. The same data as for the maize-based production is used (section 2.1.1.1).

The foreground data sources used for the bio-based carrier bags from potato are summarised in Table 205. For granulate transportation and blown film extrusion processes refer to Table 204. For background data, e.g. grid electricity and heat, other utilities and production of chemicals and materials, Ecoinvent 3.3 is used.

Table 205. Data used in the bio-based carrier bags from potato.

Processes	Data sources,	Comments
Potato cultivation and harvest	Agri-footprint	Represents potato cultivation in Germany. Crop yields derived mainly from 2010-2014 FAO statistics (FAOSTAT, 2014)
Potato starch production	Agri-footprint	Represents a typical potato wet milling process in Europe, reference year of 2014
Copolyesters production	For PLA: Vink et al., 2015 For PBAT: Broeren et al., 2017a	PLA based on data from industrial producer, see case study 3 for details with updated background data using Gabi database version 2018 (Personal communication with E. Vink). PBAT based on modelling done by Broeren et al., 2017a
Other raw materials (i.e. compatibilsers and plasticisers) production	Broeren et al., 2017a & Ecoinvent 3.3	Represents average European processes, reference year 2016
Transportation (of potato starch, copolyesters and other raw materials)	Broeren et al., 2017a	Represents average distances from the suppliers to the granulate production site. Within Europe transportation by lorry, from U.S. transportation by combination of lorry, bulk carrier and freight train
Potato-starch based granulate production	Broeren et al., 2017a	Represents average compounding extruder, key activity level data is electricity consumption: 1.9 MJ/kg granulate

2.1.1.2.11 Multifunctionality

The potato starch production process involves co-production of potato pulp, proteins and concentrated fruit juice. To avoid allocation, the system is expanded to include their animal feed replacement. The co-products are displaced with a mix of marginal feed ingredients with the same standardised feed unit. The marginal protein feed is taken as soybean meal, the marginal carbohydrate is taken as maize, and the marginal oil is taken as palm oil, based on Tonini et al. (2016). These choices are based on detailed elaboration of recent demand trends and future projections (FAPRI, 2012; FAOSTAT, 2014).

2.1.2 Bio-based alternative: bio-based LDPE carrier bag

Bio-based LDPE (100 % bio-based) was chosen as the bio-based alternative for starch plastic carrier bag (30-70 % bio-based). The process flow diagram of the bio-based LDPE carrier bag product system is shown in Figure 189. The processes involved, and the foreground data sources used are summarised in Table 206. Bio-based LDPE is produced 100 % from Brazilian sugarcane ethanol. The ethanol production is modelled in case study 1 beverage bottles, please refer to it for detailed process descriptions. Bioethylene production from ethanol takes place in Brazil. Proprietary data is used in modelling this process (Tsiropoulos et al., 2015). With high pressure polymerisation ethylene is converted into LDPE. No specific data for polymerisation of bio-based ethylene was available. However, as this process is 100 % identical to fossil-based polymerisation, the impacts of the polymerisation process are approximated by subtracting impacts of fossil-based ethylene (Ecoinvent 3.3 data) from the impacts of petrochemical LDPE (Industry data 2.0). The impacts for climate change and NREU of the polymerisation process calculated this way are checked to confirm agreement with the results provided in the LDPE ecoprofile (PlasticsEurope, 2016). LDPE is then transported to Europe based on transportation distances in Tsiropoulos et al. (2015). Finally, they are converted into carrier bags by blown film extrusion. The electricity requirement for this process

was modelled based on data from a bag producer in Europe for petrochemical LDPE bags (Environmental Agency, 2011).

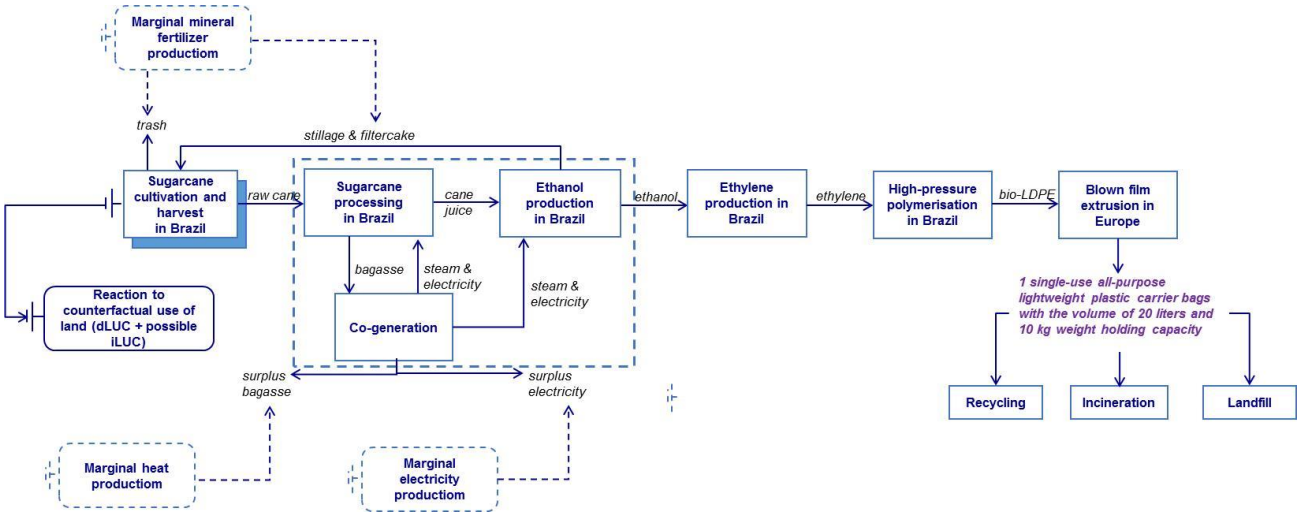


Figure 189. Process flow diagram of alternative, bio-based LDPE carrier bag system.

Table 206. Overview of data used in the bio-based LDPE carrier bag.

Processes	Data sources, foreground	Comments
Sugarcane cultivation	Tsiropoulos et al., 2014 & Seabra et al., 2011	Represents south-central Brazil 2008, based on sugarcane technology centre CTC database (CTC, 2009), with own modification
Bioethanol production	Tsiropoulos et al., 2014 & Seabra et al., 2011	Represents south-central Brazil 2008, based on sugarcane technology centre CTC database (CTC, 2009)
Bio-ethylene production	Tsiropoulos et al., 2015	Represents production in Brazil in 2011 based on data from technology licensor
Polymerisation to bio-based LDPE	PlasticsEurope 2016 & Ecoinvent 3.3	Represents European average, reference year of 2011
Transportation to Europe	Tsiropoulos et al., 2015	Lorry between polymerisation plant in Brazil to port, then bulk carrier to Rotterdam
Blown film extrusion	Environmental Agency 2011	Represents European average, reference year 2006 (0.97 kWh/kg, 99.5 % efficiency)

2.1.3 The reference system: petrochemical LDPE

The reference petrochemical system is petrochemical LDPE carrier bag. PlasticsEurope released in 2016 the latest eco-profile for petrochemical LDPE. This data is taken as the reference since it provides the most up to date eco-profile based on European plastic producers. However, as explained Chapter 6 (Ranges for environmental impacts from production of fossil-based plastics), PlasticsEurope data for petrochemical plastics are not transparent and do not comply with ILCD requirements. Differences between the available data from different data sources are found to be large and some impact categories were found not suitable (Chapter 6 Ranges for environmental impacts from production of fossil-based plastics). The data and ranges are provided in Table 37 for the suitable impact categories for 1 kg LDPE granulate production. LDPE carrier bags are then produced through blown film extrusion of the LDPE granulates. With 1 kg of LDPE granulate, 0.995 kg of carrier bag is attained. The key activity level data is electricity consumption which is 0.97 kWh/kg (Environmental Agency, 2011).

2.2 End of Life description, data, assumptions and multifunctionality

The carrier bags have different possible EoL options depending on the product system, as observed in Table 207.

The petrochemical and bio-based LDPE single-use carrier bags have three possible EoL options;

- **Mechanical plastic recycling**; the process includes the energy and material requirements for the transportation to the facility, sorting, cleaning and recycling processes. The recycled LDPE is assumed to substitute the virgin LDPE production. The rejects from the recycling process are sent to incineration (both with and without energy recovery) and this includes transportation. The flowchart for plastic recycling is presented in Figure 8.

- **MSW Incineration with and without energy recovery**; a generic MSW incineration plant is assumed, which represents the average EU waste incineration with and without energy recovery with an overall heat efficiency of 22 % and electrical efficiency of 9 % (CEWEP, 2013; Eurostat, 2017a). The process includes the direct emissions from waste incineration as well as indirect emissions from the production of the input materials, the combustion of fossil fuels as well as treatment of bottom ash and fly ash. The flowchart for MSW incineration is presented in Figure 9.

- **MSW landfilling**: the process includes the construction and operation of an average landfill site, direct emissions of landfill gas, landfill gas collection, flaring and recovery system, treatment of collected leachate and direct emissions from uncollected leachate. 29 % of the collected landfill gas is used to generate electricity (OpenLCA Nexus, 2015). The electricity conversion efficiency is 37 % (Christensen, 2011). The flowchart for MSW landfilling is presented in Figure 10.

The starch-based plastic, has the following EoL options;

- **MSW Incineration with and without energy recovery**: a generic MSW incineration plant is assumed, which represents the average EU waste incineration with and without energy recovery, with an overall heat efficiency of 22 % and electrical efficiency of 9 % (CEWEP, 2013; Eurostat, 2017a). The process includes the direct emissions from waste incineration as well as indirect emissions from the production of the input materials, the combustion of fossil fuels as well as treatment of bottom ash and fly ash. The flowchart for MSW incineration is presented in Figure 9.

- **MSW landfilling**: the process includes the construction and operation of an average landfill site, direct emissions of landfill gas, landfill gas collection, flaring and recovery system, treatment of collected leachate and direct emissions from uncollected leachate. Twenty-nine percent of the collected landfill gas is used to generate electricity (OpenLCA Nexus, 2015). The electricity conversion efficiency is 37 % (Christensen, 2011). The flowchart for MSW landfilling is presented in Figure 10.

- **Industrial composting**: this includes indirect emissions from material and energy consumption, direct emissions from the plant and use on land, fertiliser substitution from compost and rejects sent to incineration (30 % of plastic and 5 % of organic waste). The flowchart for industrial composting is presented in Figure 12.

- **Anaerobic digestion**: a mix of technologies is modelled (combined with and without post-maturation (aerobic)), which includes indirect emissions from material and energy consumption, pre-treatment (rejects sent to incineration - 30 % of plastic and 5 % of organic waste), electricity, heat and fuel generation from biogas, direct emissions from the plant and use on land, and fertiliser substitution from compost and digestate. The flowchart for anaerobic digestion is presented in Figure 13.

An uncertainty in this case study is the fact that the bags are prone to littering, however, the impacts from littering are not modelled in this LCA.

The intended EoL technology for the bio-based LDPE is recycling and for the bio-based starch plastic industrial composting.

Apart from modelling full use (100 %) for each technology, an EoL mix for Europe is modelled and is presented in Table 207. These figures are for the petrochemical EoL mix based on the available waste statistics in Europe on plastic waste treatment. The current collection for treatment of plastic waste is 30 % recycling, 39 % incineration and 31 % landfilling in Europe (European Commission, 2018). This is assumed to be applicable for the petrochemical and bio-based LDPE. The current recycling rate of lightweight plastic carrier bags is however lower (European Parliament, 2015), than the assumed 30 %, but as the time frame of the study is 5-10 years, this collection rate has been assumed.

For the EoL mix for the starch carrier bag, it is estimated that the Europeans can sort the same amount of starch carrier bags which they today are able to sort for recycling of the commercial plastics, hence 30 %. This amount would then be sorted to the intended technology of industrial composting.

Table 207. The EoL options, the intended EoL option and the estimated EoL mix.

Product system	EoL options modelled individually (100 % scenarios)	Intended EoL option	Estimated EoL collection mix of EU (based on statistics of waste treatments of waste plastics)
Bio-based starch plastic	Incineration Landfilling Industrial composting Anaerobic digestion *Reuse as carrier bag* *Reuse as waste bag* *Reuse as organic waste bag*	Industrial composting	Industrial composting: 30 % Incineration: 39 % Landfilling: 31 %
Bio-based LDPE	Recycling Incineration Landfilling *Reuse as carrier bag* *Reuse as waste bag*	Recycling	Recycling: 30 % Incineration: 39 % Landfilling: 31 %
Petrochemical LDPE	Recycling Incineration Landfilling *Reuse as carrier bag* *Reuse as waste bag*		Recycling: 30 % Incineration: 39 % Landfilling: 31 %

The average European EoL technologies are described in the approach and methodology chapter in the section regarding EoL. In that section, factors that do not depend on the specific product system are described, e.g. the energy efficiency of the incineration plant and the consumption of energy and materials in each technology. Furthermore, principles for substitution are described. This involves the energy produced in the EoL technologies being substituted with marginal electricity and heat, as well as the recycling substitution methodology.

In addition, there are several product system dependent factors, which are presented in Table 208, for each possible EoL technology. The recycled LDPE plastic is substituted with petrochemical PE. The recycling rate is calculated by multiplying the collection rate and the sorting and technology efficiency.

Table 208. Values for product system dependent factors for carrier bags.

EoL Technology	Product system dependent factor	Unit	Bio-based starch	Reference	Petrochemical and bio-based LDPE	Reference
Plastics recycling	Sorting and technology efficiency (amount collected for recycling which is recycled)	% of input			70 %	Estimated based on Plastic Recyclers Europe, 2017, Replast, n.d., SWEREC, 2007, Franklin Associates, 2011, Giugliano et al., 2011 and Dall et al., 2003
MSWI with and without energy recovery	Energy content	MJ/kg	23.27	Ecoinvent 3.4	41.45	Götze et al., 2016 (Modified C for bio-based)
Landfilling	1st order decay rate for methane generation	1/s	0.032	Estimated	0	Estimated
Industrial composting	VS Degradation (%)	% of VS	80	Hermann et al. (2011)		
	C Degradation (%)	% of C bio	80	Hermann et al. (2011)		
Anaerobic digestion	Anaerobically biodegradable biogenic carbon	% of bio C	72	Ecoinvent 3.4		

The single-use plastic carrier bags have the following material composition, see Table 209, which is utilised as the input to the EoL LCA model. The full chemical composition is presented in Annex 1. The intermediate EoL-reference flow for analysing the impact of the EOL of the carrier bags is:

- 1 kg of plastic and 100 g contamination

The contamination consists of organic waste – see further details in Approach and methodology chapter, section 3.3.2 Contamination composition.

Table 209. Material composition of single-use plastic carrier bags (1 kg plastic + 100 g contamination).

Chemical component	Unit	Bio-based starch	Bio-based LDPE	Petrochemical LDPE
Water	% of total	3.2	9.5	9.5

TS (VS+ash)	% of total	90.7	90.5	90.5
VS	% of TS	91.6	90.2	98.0
C fossil	% of TS	35.5	0.2	80.9
C biogenic	% of TS	16.2	81.0	4.5
Reference		Ecoinvent 3.4	Götze et al., 2016	Götze et al., 2016

3 Life cycle impact assessment results and interpretation

3.1 Cradle-to-gate environmental impacts and interpretation, bio-based system(s)

3.1.1 Cradle-to-gate environmental impacts of Bio-Based baseline system: Bio-Based carrier bags from maize/potato starch

The aggregated cradle-to-gate LCA results for one functional unit of bio-based carrier bag (22.4 g) is presented in Table 210. This represents the aggregated results based on the market share of 2/3 for starch plastics from maize and 1/3 for starch plastics from potato as described in section 1.4.

Table 210. Cradle-to-factory gate LCA results of 1 functional unit bio-based carrier bag (baseline) (excluding DLUC and ILUC effects).

Impact category	Unit	Bio-based carrier bag (baseline)
Climate change	kg CO_2 eq.	4.47E-02
Ozone depletion	kg CFC-11 eq	1.45E-08
Human toxicity, non-cancer effects	CTUh	1.23E-08
Human toxicity, cancer effects	CTUh	3.04E-09
Particulate matter	kg PM2.5 eq	2.80E-05
Ionizing radiation, human health	kBq U235 eq	1.19E-02
Photochemical ozone formation	kg NMVOC eq	1.81E-04
Acidification	molc H+ eq	3.49E-04
Terrestrial eutrophication	molc N eq	6.31E-04
Freshwater eutrophication	kg P eq	2.14E-05
Marine eutrophication	kg N eq	1.15E-04
Freshwater ecotoxicity	CTUe	3.23E-01
Land use	kg C deficit	2.08E-01
Water use	m3	4.02E-02
Abiotic depletion	kg Sb eq	5.96E-08
Abiotic depletion (fossil fuels)	MJ	1.22E+00
NREU	MJ	1.41E+00

In Figure 190, the impacts of starch plastic carrier bags are broken down into the main unit processes described in section 2.1 inventory analysis. For the maize based polymer, it is not possible to show the impacts of copolyesters and other raw materials separately. Therefore, they are combined into one category for the aggregated results with the potato-based polymer.

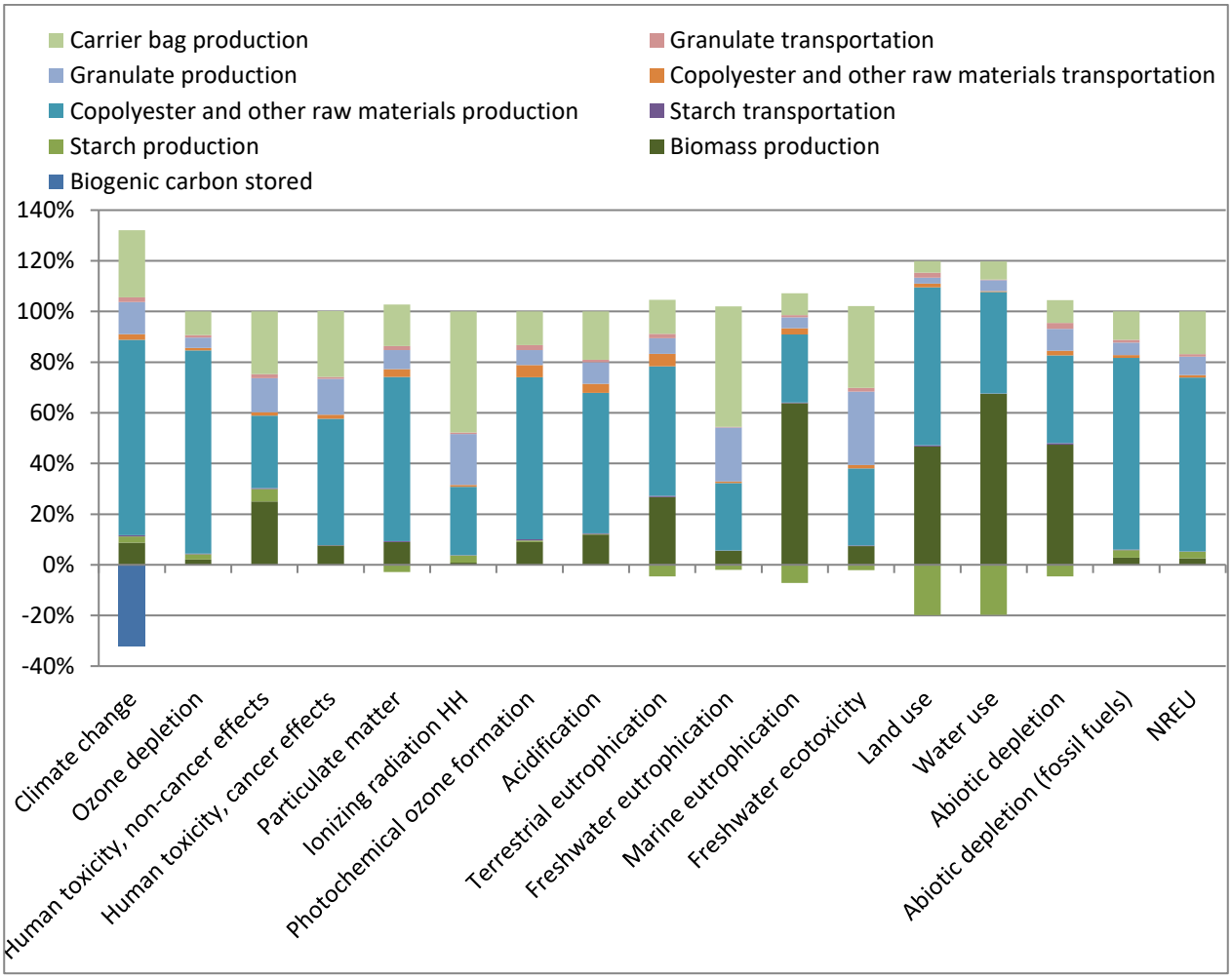


Figure 190. Breakdown of the cradle-to-gate results for bio-based carrier bags (baseline), excluding DLUC and ILUC effects.

Biomass (maize/potato) production contributes significantly to terrestrial eutrophication (27 %), marine eutrophication (64 %), land use (47 %), human toxicity (non-cancer effects) (25 %), water use (68 %) and abiotic depletion (47 %). The high contribution to water use is mainly due to irrigation. The contribution to marine and terrestrial eutrophication comes from direct nitrous emissions caused by the application of fertilisers. The pesticides production process causes the contribution of maize production to abiotic depletion. For other impact categories, contribution from maize production is up to 12 %.

Starch production does not have a significant contribution to any category except for land use and water use for which it provides a 20 % saving. Due to the avoided animal feed production from the starch by-products, smaller savings are also seen in the eutrophication and abiotic depletion categories.

The environmental impact of bio-based carrier bags is dominated by copolyester and other raw material production. Their contribution ranges from 27 % to 80 % in all impact categories. This is caused predominantly by the petrochemical copolyesters and raw materials; their impacts arise from the naphtha, natural gas and coal used in the production. For the bio-based copolyester of PLA, the detailed breakdown into unit processes can be seen in case study 3. This bio-based polyester shows a major contribution in the land use category (44 % of the total impacts from copolyester and other raw material production, see further breakdown of the impacts in section 3.1.9).

The contribution of transportation requirements (starch and copolyester transportation to granulate production, granulate transportation to carrier bag production) to the environmental impact of bio-based carrier bags are minor (maximum 7 % contribution to impact categories).

Granulate production and carrier bag production processes have a parallel contribution to the impact categories because the key activity level data in both processes is electricity consumption. A significant contribution of these processes is seen for ionizing radiation, freshwater eutrophication and freshwater ecotoxicity arising from the production of electricity. Combined they contribute to about 70 % of the environmental impact of carrier bags.

The results for the bio-based carrier bag baseline are further elaborated below. Following the PEFCR guidance (i.e. contributing to 80% of the total impact) the most relevant impact categories are identified as: abiotic depletion (fossil fuels), climate change, particulate matter, ionizing radiation, acidification, freshwater eutrophication and photochemical ozone formation. They cumulatively contribute to 84 % of the total environmental impact (excluding toxicity related impact categories) based on the normalised and weighted results (see section 3.4.1). Detailed analysis is made to trace the origin of the impact in the major contributing processes for these impact categories and also for impact categories where biomass production has significant impact.

The seven most relevant impact categories are interpreted in detail in the following bullets point.

- **Abiotic depletion (fossil fuels):** The majority (76 %) of abiotic depletion (fossil fuels) comes from copolyester and additives. Sixty-one percent of the aggregated impacts are caused by the petrochemical copolyester in the maize-starch plastics; and the impacts arise from naphtha (46 %) and natural gas (39 %) used. Thirty-nine percent of the aggregated impacts are caused by the potato-starch plastics where 37 % of the impact comes from PBAT production (41 % caused by adipic acid, 32 % caused by 1,4 butanediol production), 30 % from PLA production and 33 % from other raw materials. Electricity consumption for the granulate and carrier bag production contributes to 16 % of the impact.

- **Climate change:** The biogenic carbon embedded in the product is calculated as 0.64 kg CO_2 eq./kg bio-based starch plastic. When the biogenic carbon removal is considered, the cradle to factory gate climate change impact of the bio-based carrier bag is 2 kg CO_2 eq./kg or 4.47×10^{-2}kg CO_2 eq./FU. Again, it is emphasized that this GHG accounting does not consider the effects of DLUC and ILUC. The copolyester production accounts for 60 % of the gross GHG emissions[73] (Figure 190). About 30 % of the gross emissions come from the petrochemical copolyester used in the maize starch-based polymer (where heat produced from coal contributes 62 % of this impact). The bio-based PLA and petrochemical PBAT each contribute 18 % to the gross impact. For PBAT, the GHG emissions are dominated by adipic acid (56 %) and butanediol (31 %). The remaining contributors are carrier bag production (20 % of the gross emissions), granulate production (10 %) and biomass production (7 %). The GHG emissions of granulate and carrier bag production processes are mainly caused by the electricity consumption. For biomass production, 38 % of the impact is based on the diesel use in machinery, 26 % from direct field emissions from fertilisers and urea decomposition and 10 % from ammonium nitrate chemical fertiliser use.

[73] "Gross GHG emissions" defined as the total GHG emissions excluding the biogenic carbon removal (the negative value on the graph).

- **Particulate matter:** Copolyester production contributes to 65 % of particulate matter. Of this 65 %, about 60 % of the impact originate from copolyester used in the maize-starch plastic and about 40 % of the impacts are caused by the potato-based plastic, of which PBAT plays a dominant role (70 %)[74], followed by PLA (15 %) and other raw materials such as additives (15 %). Electricity consumption in the granulate and carrier bag production contribute to in total 24 % of impact which comes from the direct particulates and sulphur dioxide emissions in electricity production from coal and lignite in the electricity mix.

- **Ionizing radiation:** For ionizing radiation, about 68 % of the impact comes from granulate production (i.e. converting native starch into plastic granulates) and carrier bag production (i.e. converting plastic granulates into bags) where the impact is dominated by electricity. Copolyester and other raw material such as additives account for 27 % of the impact.

- **Acidification:** Copolyesters and other raw materials (such as additives) production account for 55 % of the acidification impact. Of this 55 %, about 56 % is caused by the maize starch-based plastic and 44 % is caused by the potato starch-based plastic of which 48 % of the impact comes from PBAT, 35 % from PLA production and 17 % from other raw materials. Biomass production contributes 12 % to the acidification impact due to direct ammonia emissions to air from the use of fertilisers and nitrogen oxides emissions from the diesel burned in machinery. A total of 28 % of the contribution to acidification impact comes from granulate production (9 %) and carrier bag production (19 %), caused by the sulphur dioxide and nitrogen oxides emission from the electricity use.

- **Freshwater eutrophication:** For freshwater eutrophication, about 69 % of it comes from carrier bag production (48 %) and granulate production (21 %) where electricity consumption is the most important activity. Looking at the elementary flows, the high impact of electricity production is caused by phosphate emissions to water from treatment of spoil from lignite and hard coal mining. Copolyester and other raw material production account for 27 % of freshwater eutrophication, where maize starch-based plastic and potato starch based plastic show almost equal contribution. For maize starch-based plastic, which uses petrochemical copolyester, the impacts arise from coal used to produce the copolyester (67 %). For potato starch-based plastic, 83 % of the impacts are caused by petrochemical PBAT to produce the chemicals used (adipic acid (42 %) and 1,4 butanediol (43 %)).

- **Photochemical ozone formation:** Copolyester and other raw material such as additives account for 64 % of photochemical ozone formation where maize starch-based plastic and potato starch based plastic show almost equal contributiona. The most important elementary flows are nitrogen oxides and non-methane volatile organic compounds, which are emitted during the production of various chemicals which is used by the process. The remaining contribution mainly comes from from carrier bag production (13 %) and granulate production (6 %) where the most important activity is electricity consumption.

The interpretation of several other impact categories that are relevant for biomass production are also here reported:

[74] Breakdown of PBAT production: 40 % caused by adipic acid, 44 % caused by 1,4 butanediol production.

- **Marine eutrophication:** For marine eutrophication, 64 % of the contribution comes from biomass production. The majority (88 %) of this impact originates from direct nitrate emissions to water from application of fertilisers and from crop residues.

- **Land use:** About 40 % of gross impact of land use[75] (in kg carbon deficit) originates from land occupation for biomass (maize/potato). Fifty-two percent of the gross land use impact comes from copolyester production, including the land use for PLA production and the land use for onshore drilling for naphtha for the petrochemical copolyesters. Due to the avoided animal feed production by the starch by-products, about 17 % of the gross land use impact is reduced.

- **Water use:** Biomass production contributes 56 % of the gross water use[76], of which the majority (97 %) comes from irrigation requirements for maize cultivation.[77] About 34 % of the gross water use comes from copolyesters: 53 % arises from PBAT,[78] 24 % from PLA production and 15 % from additives. Due to the avoided animal feed production from the starch by-products, about 17 % of the gross water use can be avoided.

- **Abiotic depletion.** Biomass production contributes 47 % of abiotic depletion. The impacts are mainly (>70 %) caused by the production of pesticides. Copolyester and other raw materials production contributes 35 %, of which PBAT contributes 51 %, PLA contributes 16 % from PLA production and the remaining additives 19 %.

3.1.2 Cradle-to-gate environmental impacts of bio-based alternative system: Bio-based LDPE carrier bags

The cradle-to-gate LCA results for one functional unit of bio-based alternative system of bio-based LDPE carrier bag (17.9 g) is presented in Table 211.

Table 211. Cradle-to-factory gate LCA results of 1 functional unit bio-based carrier bag (alternative: bio-based LDPE) (excluding DLUC and ILUC effects).

Impact category	Unit	Bio-based carrier bag (alternative, bio-based LDPE)
Climate change	kg CO_2 eq.	-4.80E-03
Ozone depletion	kg CFC-11 eq	8.02E-09
Human toxicity, non-cancer effects	CTUh	5.48E-08
Human toxicity, cancer effects	CTUh	1.94E-09
Particulate matter	kg PM2.5 eq	7.61E-04
Ionizing radiation human health	kBq U235 eq	6.94E-03
Photochemical ozone formation	kg NMVOC eq	4.80E-04
Acidification	molc H^+ eq	5.75E-04

[75] "Gross impact of land use" refer to the land use impact without taking into account the credits from avoided land use of by-products (from maize and potato milling).

[76] "Gross water use" refers to the water use impact without taking into account the credits received from avoided water use of by-products (from maize and potato milling).

[77] Note that little or no irrigation is required for European potatoes.

[78] For the water use impact of PBAT production: 16 % caused by adipic acid, 63 % caused by 1,4 butanediol production.

Terrestrial eutrophication	molc N eq	2.21E-03
Freshwater eutrophication	kg P eq	1.59E-05
Marine eutrophication	kg N eq	1.45E-04
Freshwater ecotoxicity	CTUe	3.03E-01
Land use	kg C deficit	7.66E-01
Water use	m^3	2.31E-01
Abiotic depletion	kg Sb eq	7.35E-08
Abiotic depletion (fossil fuels)	MJ	5.46E-01
NREU	MJ	6.56E-01

In Figure 191, the impacts of bio-based LDPE carrier bags are broken down into the main unit processes described in the section 2.1.2 inventory analysis. In Figure 192, the breakdown of climate change LCIA results are shown separately.

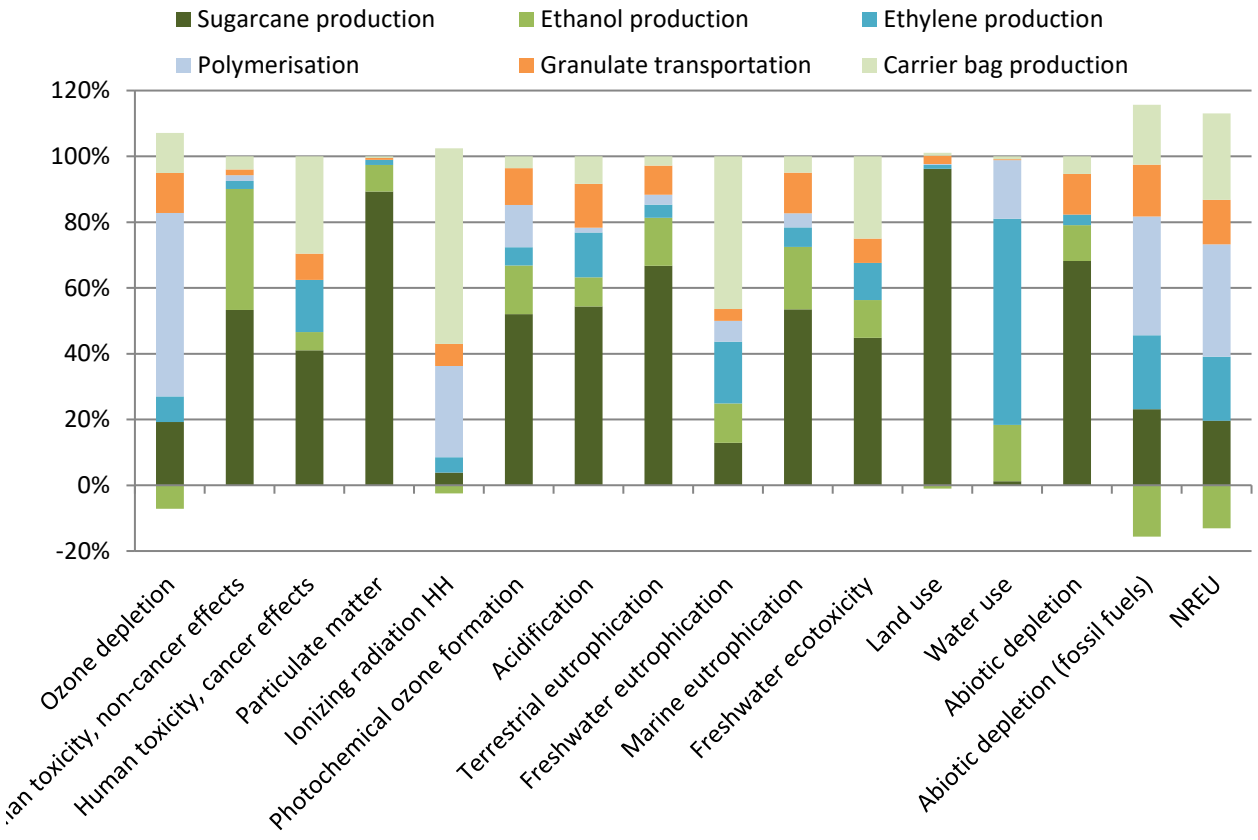


Figure 191. Breakdown of the cradle-to-gate results for bio-based carrier bags (alternative: bio-based LDPE), excluding DLUC and ILUC effects.

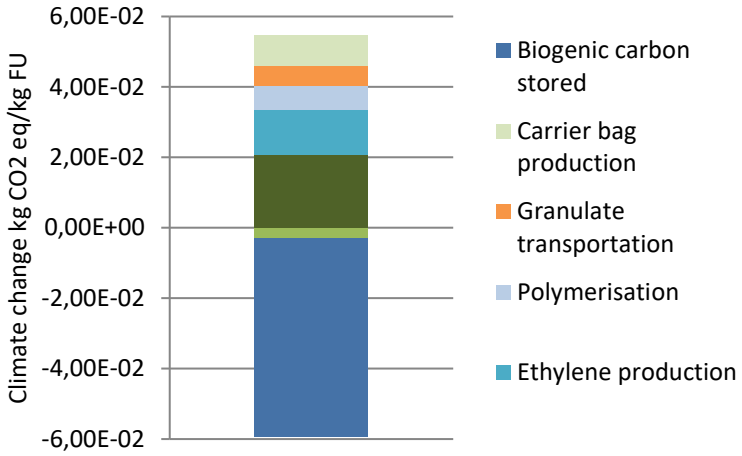


Figure 192. Breakdown of the cradle-to-gate climate change impact results for bio-based carrier bags (alternative: bio-based LDPE), excluding DLUC and ILUC effects.

This is a 100 % bio-based product and the biogenic carbon embedded in the product is calculated as 3.14 kg CO_2 eq./kg bio-based starch plastic. When the biogenic carbon removal is taken into account, the cradle-to-factory-gate climate change impact of the bio-based LDPE carrier bag results in net CO_2 storage of 0.27 kg CO_2 eq./kg or 4.80×10^{-3}kg CO_2 eq./FU, as shown in Figure 192. Again, it is emphasized that this GHG accounting does not take into account the effects of DLUC and ILUC.

Sugarcane production contributes significantly to many impact categories including particulate matter (89 %), land use (96 %), abiotic depletion (68 %), terrestrial eutrophication (67 %), marine eutrophication (53 %), photochemical ozone formation (52 %) and human toxicity (non-cancer) (53 %). Particulate matter is caused by the direct emissions due to the cane waste burning during harvest. The pesticides used have an effect on the human toxicity (non-cancer) and abiotic depletion categories. Nitrogen oxide emissions due to application of fertilisers result in photochemical ozone formation, terrestrial and marine eutrophication.

The bioethanol production contributes to human toxicity (non-cancer) (37 %) caused by the emissions to air from bagasse burning. It shows a 19 % contribution to marine eutrophication and 15 % contribution to terrestrial eutrophication due to nitrogen oxides emissions from bagasse burning. It also shows a 17 % contribution to water use due to water used in the process. Savings are seen in climate change, abiotic depletion (fossil fuels) and NREU due to the use of the by-product bagasse for internal energy supply and the credits gained by excess electricity production.

The water use for ethylene production results in a high contribution of this process to the overall water use impact (63 %). The heat requirement of the process results in about a 20 % contribution to abiotic depletion (fossil fuels) and NREU impact categories.

Transportation of the bio-based LDPE from Brazil to Europe shows a contribution to abiotic depletion (fossil fuels) (16 %) and NREU (13 %) impact categories due to use of crude oil for transportation fuels. Transportation also has a 12 % contribution to marine eutrophication, 9 % to terrestrial eutrophication and 11 % to photochemical ozone formation due to the nitrogen oxides emissions to air.

The carrier bag production by blown film extrusion shows a significant contribution to the ionizing radiation (59 %) and freshwater eutrophication (46 %) categories coming from the electricity production processes.

3.2 Cradle-to-gate environmental impacts of the petrochemical reference system

The cradle-to-factory gate LCA results for one functional unit of petrochemical LDPE carrier bags (17.9 g/FU) is presented in Table 212. This is calculated by multiplying the impact of plastic granulate production (given in Table 37 for 1 kg LDPE granulate) with 0.0179 kg to convert per FU. Impacts of granulate transportation and and blown film extrusion to carrier bags are added to this.

Based on Chapter 6 *Ranges for environmental impacts from production of fossil-based plastics*, eight out of 17 impact categories were found suitable for LDPE, namely: climate change, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication, marine eutrophication, abiotic depletion (fossil fuels) and NREU (see chapter 6). This is because comparison of different datasets revealed large differences for the other impact categories. Further details can be found in Table 37 where the environmental impact of 1 kg of LDPE and the proposed ranges of variation within each impact category are shown. The data used is based on PlasticsEurope, since it provides the most up to date eco-profile for European fossil-based plastics. However, it does not conform with ILCD requirements in its presentation of inventory details or impact assessment results. It does not distinguish between emissions to fresh water and to seawater, and it is therefore not possible to calculate the impacts on human health and ecosystems as well as freshwater eutrophication, because the receiving compartment is an important element in determining the fate and effect of an emission.

Table 212. Cradle-to-gate LCA results of petrochemical LDPE carrier bag for 1 FU (=17.9 g).

Impact category	Unit	Petrochemical LDPE carrier bags
Climate change	kg CO_2 eq.	4.35E-02
Particulate matter	kg PM2.5 eq	9.52E-06
Photochemical ozone formation	kg NMVOC eq	1.67E-04
Acidification	molc H^+ eq	1.43E-04
Terrestrial eutrophication	molc N eq	3.11E-04
Marine eutrophication	kg N eq	3.03E-05
Abiotic depletion (fossil fuels)	MJ	1.48E+00
Non-renewable energy use (NREU)	MJ	1.61E+00

In Figure 193, the impacts of petrochemical LDPE carrier bags are broken down into the main unit processes of granulate production, transportation and carrier bag production. It is seen that production of LDPE granulate shows the highest contribution in all (selected as suitable for comparison) impact categories. Its contribution ranges between 61-93 % for all categories. The ecoprofile for the LDPE granulate comprises extraction and refining of crude oil and natural gas, steam cracking of hydrocarbons into lower olefins, and polymerisation of the monomers into polyethylene. However, it is not possible to show the breakdown of the impact of granulate production into these unit processes due to lack of transparancy of the PlasticsEurope data. The transportation is within Europe and shows a negligible contribution to impacts. Blown film extrusion process shows significant (35 %) contribution in the particulate matter and acidification impact categories. The main activity level data of this process is electricity consumption.

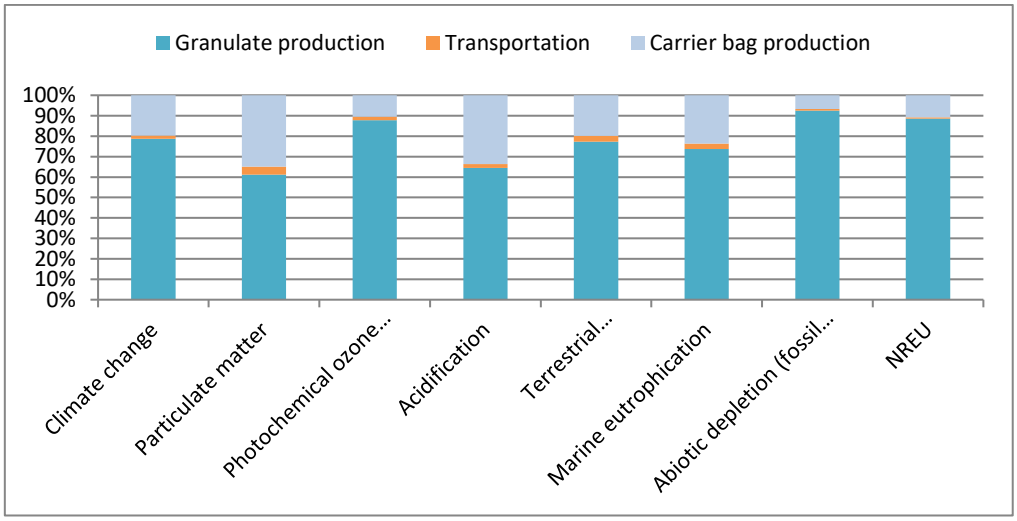


Figure 193. Breakdown of the cradle-to-gate LCA results for petrochemical LDPE carrier bags

3.3 End of life results and interpretation

The results for the EoL impact assessment modelled in EASETECH are presented in this paragraph. Firstly, the mass and energy flows of all product systems in all technologies are presented, secondly the LCIA results for each product system, thirdly a comparison between product systems, and lastly the EoL mix. In Annex 4 further results are presented, the contribution analysis in Figure 308 ff. and weighted results are in Table 247 ff..

3.3.1 Mass and energy flows

The mass and energy flows of EoL waste management, as outputs from EASTECH, are presented in Table 213 to Table 215 for all product systems. The results, correspond to the circular footprint formula in PEFCR guidance draft version 6.3 (European Commission, 2017), and in addition this paragraph looks at the carbon flows. The flows correspond to those observed in the flow charts for each technology in the approach and methodology chapter of the full report, section 3.4 Average End of Life technologies.

The mass flows of the product systems and EoL scenarios are as follows. Landfilling the carrier bags results in 73 % of the total mass being stored for the starch, but 98 % of the mass is stored for the non-degradable product systems bio-based LDPE and petrochemical LDPE. For the biological treatments of starch carrier bags, 21 % of the mass ends up as compost from the composting facility, and from the AD facility, 12 % as compost and almost 550 % of the input weight as digestate (due to added water in the process). From the composting facility 31 % of the nutrients end up back on fields as compost, whereas the same recovery for the anaerobic digestion is 38 %. In the recycling scenario 70 % of the LDPE plastics is recycled and 57 % substituted (calculated based on a sorting and technology efficiency as well as a market response, considering the decrease in quality of the plastic - see the methodology for further explanation).

The energy recovery is highest for the incineration scenarios, where all product systems recover 31 % (the energy efficiency of the incineration plant). There is additionally recovery of energy from incineration of rejects; 9 % in the recycling of the LDPEs, 9 % in the composting of PLA and 9 % in the anaerobic digestion of PLA. Furthermore 15 % of the energy content is recovered in the biogas from the AD facility, recovering in total 24 % of the energy content of

the waste input. The utilisation of landfill gas when landfilling PLA recovers 6 % the energy content.

Looking at the carbon flows, it is observed that 18 % of the biogenic carbon is stored in the landfill for starch, but all the carbon for the non-degradable LDPEs. In the biological technologies respectively 15 % and 69 % of the biogenic carbon is recovered in industrial composting and AD as compost, digestate or biogas.

Table 213. Material and energy flow for EoL of 1 kg starch with 100 g contamination. Box letter refer to flow charts in the approach and methodology chapter (Figure 8 to Figure 13).

Bio-based starch								
Technology	**Box**	**Process**	**Mass (kg)**	**Bio carbon (kg)**	**Fossil carbon (kg)**	**Gas (m^3 CH4)**	**Energy substitution (MJ)**	**Fertiliser substitution (kg NPK)**
Material	I	Material input	1.10E+00	1.90E-01	3.90E-01		2.39E+01	1.65E-03
Incineration w/wo energy recovery *	E, H	Energy production	1.10E+00	1.90E-01	3.90E-01		7.35E+00	
	R1	Fly Ash	1.39E-03	0.00E+00	0.00E+00			
	R2	Bottom ash	4.96E-03	1.90E-04	3.90E-04			
	D	Direct emissions		1.90E-01	3.90E-01			
Landfill	L	Leachate	1.61E+00	0.00E+00	0.00E+00			
	G, E	Landfill gas		1.56E-01		6.30E+01	1.38E+00	
	CS	Storage in landfill	8.05E-01	3.38E-02	3.90E-01			
Compost	O, S	Compost	2.29E-01	2.83E-02	2.73E-01			5.07E-04
	R, E, H	Rejects	6.32E-02	1.05E-02	4.42E-02		2.16E+00	
Anaerobic digestion	B, E1, H1, F	Biogas		8.61E-02		1.01E-01	3.48E+00	
	C, S	Compost	1.35E-01	6.77E-03	5.16E-03			7.37E-05
	D, S	Digestate	5.97E+00	3.82E-02	2.31E-01			5.56E-04
	R, E2, H2	Rejects	3.30E-01	5.70E-02	1.17E-01		2.21E+00	

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

Table 214. Material and energy flow for EoL of 1 kg bio-based LDPE with 100 g contamination. Box letter refer to flow charts in the approach and methodology chapter (Figure 8 to Figure 13).

Bio-based LDPE							
Technology	**Box**	**Process**	**Mass (kg)**	**Biogenic carbon (kg)**	**Fossil carbon (kg)**	**Gas (m^3 CH_4)**	**Energy substitution (MJ)**
Material	I	Material input	1.10E+00	8.19E-01	1.79E-04		4.04E+01
Recycling	O	Recycled plastic	7.70E-01	5.73E-01	1.25E-04		
	S	Substituted plastic	6.24E-01	4.64E-01	1.01E-04		
	R, E, H	Rejects	3.30E-01	2.46E-01	5.36E-05		3.73E+00
Incineration w/wo energy recovery *	E, H	Energy production	1.10E+00	8.19E-01	1.79E-04		1.24E+01
	R1	Fly Ash	4.59E-03	0.00E+00	0.00E+00		
	R2	Bottom ash	1.40E-02	8.19E-04	1.79E-07		
	D	Direct emissions		8.18E-01	1.79E-04		
Landfill	L	Leachate	2.15E+00	0.00E+00	0.00E+00		
	G, E	Landfill gas				2.01E-03	7.63E-02
	CS	Storage in landfill	1.08E+00	8.06E-01	1.79E-04		

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

Table 215. Material and energy flow for EoL of 1 kg petrochemical LDPE with 100 g contamination. Box letter refer to flow charts in the approach and methodology chapter (Figure 8 to Figure 13).

Petrochemical LDPE						
Technology	**Box**	**Process**	**Mass kg**	**Fossil carbon kg**	**Gas m^3 CH4**	**Energy substitution MJ**
Material	I	Material input	1.10E+00	8.03E-01		4.04E+01
Recycling	O	Recycled plastic	7.70E-01	5.23E-01		
	S	Substituted plastic	6.24E-01	4.24E-01		
	R, E, H	Rejects	3.30E-01	2.80E-01		3.73E+00
Incineration w/wo energy recovery *	E, H	Energy production	1.10E+00	8.03E-01		1.24E+01
	R1	Fly Ash	4.59E-03	0.00E+00		
	R2	Bottom ash	1.40E-02	8.03E-04		
	D	Direct emissions		8.02E-01		
Landfill	L	Leachate	2.15E+00	0.00E+00		
	G, E	Landfill gas			2.01E-03	7.63E-02
	CS	Storage in landfill	1.08E+00	8.03E-01		

* Results for incineration without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

3.3.2 Baseline: Bio-Based starch

The impact assessment results are presented per the EoL-reference flow, i.e. 1 kg plastic and 100 g of contamination for each EoL technologies (assuming 100 % disposal via that technology). The overall LCIA results for the bio-based starch carrier bag is presented in Table 216 for all EoL technologies.

The LCIA results show that for the bio-based starch carrier bag there is no clear best EoL technology. Choosing the best performing EoL technology for bio-based starch carrier bags depends highly on which impact categories are found to be most important.

In the weighted results, section 3.2.2.5, it is observed that industrial composting performs environmentally best out of the EoL options relevant for the bio-based starch plastic carrier bags.

Further details on the individual EoL options for the starch carrier bag are presented below, looking into specific impact categories and contribution of impacts from different processes and elemental emissions.

Table 216. Total results of treating 1 kg bio-based starch carrier bag for all impact categories. Highest impact within each category is red and lowest impact is green. The intended EoL option is marked with a bold box.

Impact category (unit)	Bio-based starch			
	MSWI	Landfilling	Industiral composting	AD
Climate change – fossil (kg CO_2 eq.)	1.05E+00	-5.18E-02	3.83E-01	2.64E-01
Climate change – biogenic (kg CO_2 eq.)	6.32E-01	3.17E+00	9.89E-03	3.52E-01
Climate change – total (kg CO_2 eq.)	1.69E+00	3.12E+00	3.93E-01	6.16E-01
Ozone depletion (kg CFC-11 eq.)	-3.64E-08	1.30E-06	-1.11E-08	-2.80E-08
Human toxicity, non-cancer effects (CTUh)	-1.09E-08	-2.40E-09	3.52E-08	3.33E-08
Human toxicity, cancer effects (CTUh)	-1.64E-09	-5.55E-10	-2.88E-09	-3.22E-09
Particulate matter (kg PM2.5 eq.)	-6.11E-05	-2.74E-05	-1.74E-05	-7.45E-05
Ionizing radiation HH (kBq U235 eq.)	5.05E-02	-1.12E-02	1.47E-02	2.91E-02
Photochemical ozone formation (kg NMVOC eq.)	8.45E-04	8.55E-04	1.43E-04	4.19E-04
Acidification (molc H+ eq.)	1.54E-03	-3.51E-04	3.87E-04	1.03E-03
Terrestrial eutrophication (molc N eq.)	5.40E-03	-3.56E-04	1.16E-03	3.85E-03
Freshwater eutrophication (kg P eq.)	7.50E-05	-7.85E-06	3.67E-06	2.23E-05
Marine eutrophication (kg N eq.)	4.35E-04	6.73E-06	1.14E-04	4.36E-04
Freshwater ecotoxicity (CTUe)	-1.35E-01	-1.17E-02	-4.36E-02	-8.67E-02
Land use (kg C deficit)	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use (m^3)	-1.84E-01	-1.07E-02	-2.20E-02	-1.18E-01
Abiotic depletion (kg Sb eq.)	-2.48E-07	-3.73E-08	-8.93E-08	-2.62E-07
Abiotic depletion (fossil fuels) (MJ)	-6.93E+00	-1.33E+00	-1.26E+00	-4.03E+00
NREU (MJ)	-7.95E+00	-6.97E-01	-1.75E+00	-4.57E+00

The significance of the colour scale is the same as for all the other product systems (Table 216 and Table 217
Table 217). This means that you can compare the colour horizontally across all tables, but only comparing between the same impact category.

3.3.2.1 Intended EoL technology: Industrial composting of starch carrier bag

Industrial composting has the lowest impact in two impact categories, the second lowest in six and the highest impact in three categories. Industrial composting has the lowest climate change impact of 0.39 kg CO_2-eq., due to low impacts from the composting facility compared with other technologies and from substitution of fertilisers, electricity and heat. The industrial

composting technology however has the highest impact compared to the other EoL technologies on human toxicity (non-cancer effect) and particulate matter.

Looking into the contribution analysis, the use of compost on agricultural land contributes the most to the climate change impact, but the impact is balanced out by emissions from savings in fertiliser substitution. The substitution of heat and electricity from incineration of rejects has the largest contribution in most categories, nine out of 18. The substitution of heat has a 45-60 % contribution on the overall EoL results in NREU, abiotic depletion (fossil) and ionizing radiation. The substitution of electricity contributes 63 % to the overall EoL impact result in particulate matter, 55 % in water use and 67 % in abiotic depletion. In impact categories where nitrogen emissions have an effect (e.g. freshwater ecotoxicity), the use of compost and fertiliser substitution has a large contribution. Moreover, the use of compost and fertiliser substitution have a large contribution where heavy metal emissions to soil and water cause impacts (e.g. in human toxicity impact categories and freshwater ecotoxicity). The direct emissions from incineration of rejects has a substantial impact on climate change – total (53 %).

3.3.2.2 Anaerobic digestion of starch carrier bag

Anaerobic digestion has the lowest impact on human toxicity (cancer effects), particulate matter and abiotic depletion. The AD has the second lowest impact in seven impact categories compared to the other EoL options. AD of starch carrier bags has the highest impact for the impact category marine eutrophication, caused by nitrate emissions to water from use of digestate on land.

The substitution of heat and electricity has a large contribution in most impact categories (e.g. NREU, abiotic depletion (both), water use, particulate matter) ranging from 74 % to 81% of the impacts. Energy recovery contributes to 25 % of the climate change impact. The fertiliser substitution has the largest contribution to the overall LCIA results in human toxicity (cancer), mainly from savings in chromium emissions from substitution of fertiliser. The direct emissions from incineration and the emissions from treatment of fly ash have a high contribution in impact categories of climate change - fossil (52 %), photochemical ozone formation (26 %) and freshwater ecotoxicity (35 %).

3.3.2.3 Landfilling of starch carrier bag

Landfilling shows the highest impact in eight impact categories. For climate change impact, landfilling performs worst with CO_2 emissions of 3.12 kg CO_2 eq. due to landfill gas combustion and direct emissions of methane from the gas upgrading as well as direct greenhouse gas emissions to the environment. The electricity substitution is only 3 % of the climate change impacts. Landfilling performs best compared to the other EoL technologies in five impact categories; acidification, eutrophication categories and ionizing radiation. There is very little energy generation from the landfilling compared with the other EoL technologies and therefore landfilling performs worst in NREU, abiotic depletion and water use (as substitution of energy will give high savings in these impact categories for the other EoL technologies).

The contribution analysis shows that the small electricity substitution does contribute to the largest share of the EoL LCIA results in 14 out 18 impact categories. As mentioned previously, the landfill gas causes a large contribution to climate change, but also ozone depletion and photochemical ozone formation. The ozone depletion occurs from emissions of CFC gasses to air from non-collected methane that is oxidised in top covers. The high impact from photochemical ozone formation is mainly due to methane, non-fossil, emissions together with nitrogen oxides and toluene emissions in air.

3.3.2.4 Incineration of starch carrier bags

Incineration performs the best in six impact categories, second best in two and worst in four impact categories. Savings in electricity substitution lead to large savings in abiotic depletion (fossil fuels), NREU and water use which is why incineration performs best in these categories (ranging from 94 to 96 % of the impacts). Incineration has the highest impact in ionizing radiation, acidification and terrestrial and freshwater eutrophication. Substitution of heat gives a high impact in ionizing radiation. The direct emissions from incineration contribute to a large impact on terrestrial eutrophication and the treatment of bottom ash contributes to the largest share of freshwater eutrophication impacts.

The contribution analysis reveals that the substitution of heat and electricity have the largest contribution on the overall LCIA EoL results for 9 out of 18 impact categories, while the other are distributed between direct emissions (climate change, photochemical ozone formation, terrestrial and marine eutrophication) and bottom ash treatment (human toxicity, cancer effects, freshwater eutrophication and freshwater ecotoxicity).

3.3.2.5 Weighted end of life results for the baseline – starch carrier bag

The results have been normalised and weighted to reflect the importance of each impact category. For methodology and choice of normalisation and weighing factors, please see the Approach and methodology chapter, section 5.2.3 "Use of normalised and weighted results to determine the preference of EoL option" and the values in Table 30. Normalisation factors (NF) and weighting factors used by this study.

The results of weighting the EoL technologies for the baseline product system are shown in Figure 194, and the numerical results can be found in Annex 4. The weighted results show that both with and without toxicity, the industrial composting technology is the best, secondly, anaerobic digestion, thirdly incineration and the least favoured EoL technology is landfilling.

The largest contribution to the weighted results is the global warming potential impact category in all EoL technologies. Both industrial composting and anaerobic digestion have substantially lower impacts on climate change than landfilling and incineration. The incineration has the largest savings, mostly from savings in abiotic depletion (fossil fuels) from substitution of heat and electricity. Ozone depletion has a visible impact in the weighted results for landfilling, due to landfill gas that is collected and combusted or not collected but oxidised in top covers.

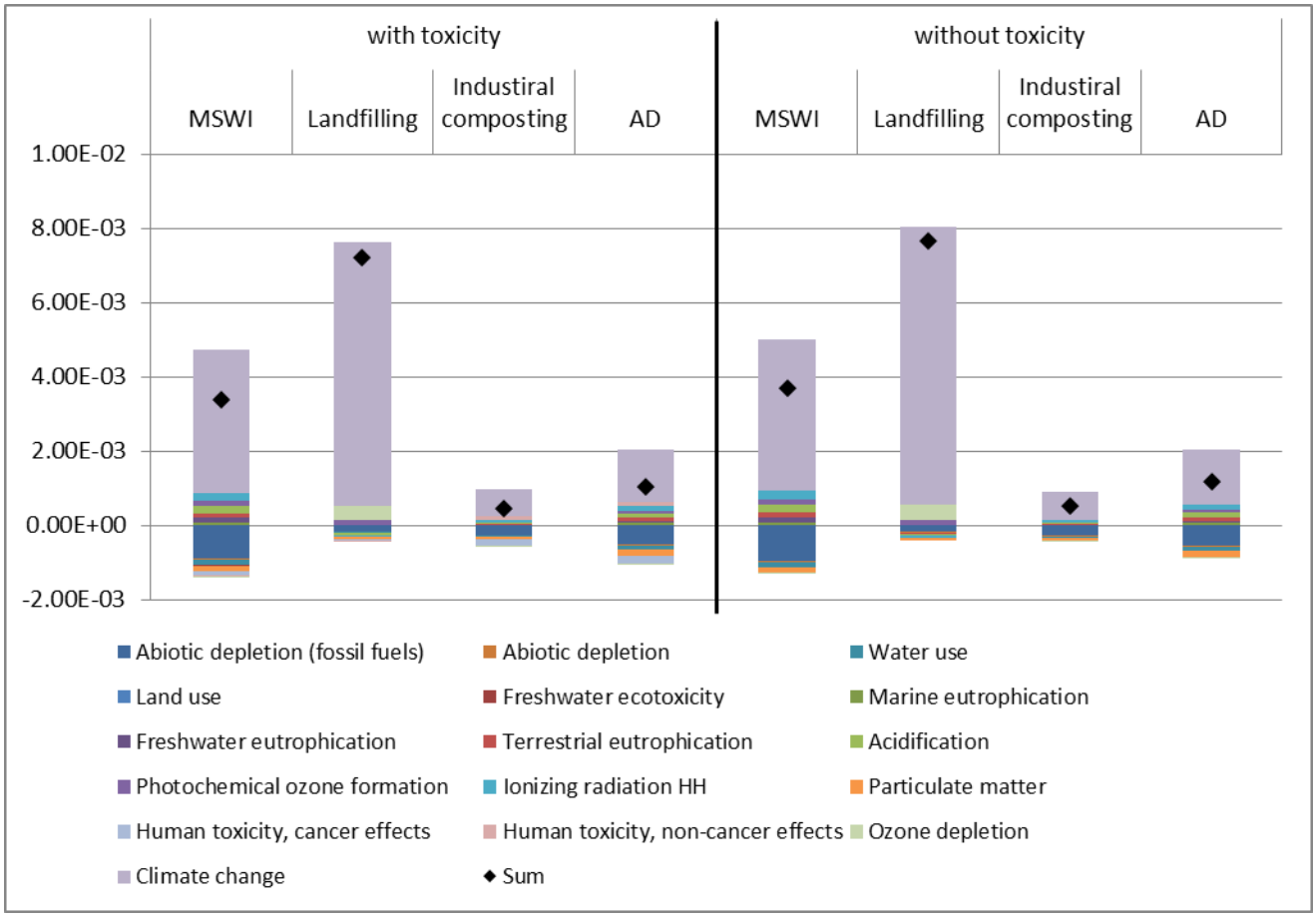


Figure 194. Weighted EoL results for the baseline product system, bio-based starch carrier bag. Numerical values are found in Annex 4.

3.3.3 Alternative and reference: Bio-Based LDPE and petrochemical LDPE

All results are given per the EoL-reference flow, i.e. 1 kg plastic and 100 g of contamination for the full EoL technologies (100 %). The overall LCIA results for the bio-based LDPE and petrochemical LDPE carrier bag are presented in Table 217 for all EoL technologies.

By looking at the LCIA results in Table 216, recycling seems to be the preferred EoL technology, performing best in most of the impact categories. Secondly, the EoL option of landfilling, which performs second-best in most impact categories and lastly incineration of LDPE which performs worst in most impact categories. For some impact categories however, the picture is reversed, with incineration performing best.

Further details on the individual EoL options for the LDPE carrier bag are presented below, looking into specific impact categories and the contribution of impacts from different processes and elemental emissions. To determine which EoL technology is the most preferable, the results have been weighted, in section 3.3.4.

Table 217. Total results of treating 1 kg petrochemical LDPE and bio-based LDPE (incl. 100 g contamination) for all impact categories. Highest impact within each category is red and lowest impact green. The intended EoL option is marked with a bold box.

Impact category (Unit)	Bio-based LDPE			Petrochemical LDPE		
	Recycling	**Incineration**	**Landfilling**	**Recycling**	**Incineration**	**Landfilling**
Climate change – fossil (kg CO_2 eq.)	-9.17E-01	-6.46E-01	1.15E-02	-3.48E-02	2.30E+00	1.15E-02
Climate change – biogenic (kg CO_2 eq.)	8.40E-01	2.89E+00	2.29E-01	-4.22E-02	-5.02E-02	2.29E-01
Climate change – total (kg CO_2 eq.)	-7.70E-02	2.25E+00	2.40E-01	-7.70E-02	2.25E+00	2.40E-01
Ozone depletion (kg CFC-11 eq.)	-3.23E-08	-6.25E-08	1.09E-07	-3.23E-08	-6.25E-08	1.09E-07
Human toxicity, non-cancer effects (CTUh)	2.18E-08	1.93E-08	3.53E-09	2.18E-08	1.93E-08	3.53E-09
Human toxicity, cancer effects (CTUh)	-2.63E-09	1.81E-08	3.76E-11	-2.63E-09	1.81E-08	3.76E-11
Particulate matter (kg PM2.5 eq.)	-1.80E-04	-1.20E-04	1.20E-06	-1.80E-04	-1.20E-04	1.20E-06
Ionizing radiation HH (kBq U235 eq.)	8.66E-03	8.47E-02	-2.54E-05	8.66E-03	8.47E-02	-2.54E-05
Photochemical ozone formation (kg NMVOC eq.)	-2.53E-03	7.60E-04	1.09E-04	-2.53E-03	7.60E-04	1.09E-04
Acidification (molc H+ eq.)	-1.44E-03	1.91E-03	3.82E-05	-1.44E-03	1.91E-03	3.82E-05
Terrestrial eutrophication (molc N eq.)	-2.96E-03	5.80E-03	1.65E-04	-2.96E-03	5.80E-03	1.65E-04
Freshwater eutrophication (kg P eq.)	-1.46E-05	8.94E-05	4.00E-08	-1.46E-05	8.94E-05	4.00E-08
Marine eutrophication (kg N eq.)	-2.98E-04	4.77E-04	5.72E-05	-2.98E-04	4.77E-04	5.72E-05
Freshwater ecotoxicity (CTUe)	-4.52E-01	8.37E-01	8.13E-03	-4.52E-01	8.37E-01	8.13E-03
Land use (kg C deficit)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use (m^3)	1.03E-01	-3.26E-01	7.52E-03	1.03E-01	-3.26E-01	7.52E-03
Abiotic depletion (kg Sb eq.)	-7.10E-08	-4.25E-07	2.32E-10	-7.10E-08	-4.25E-07	2.32E-10
Abiotic depletion (fossil fuels) (MJ)	-3.20E+01	-1.19E+01	1.43E-01	-3.20E+01	-1.19E+01	1.43E-01
NREU (MJ)	-3.29E+01	-1.36E+01	1.44E-01	-3.29E+01	-1.36E+01	1.44E-01

The significance of the colour scale is the same as for all the other product systems (

Table 216). This means that you can compare the colour horizontally across all tables, but only comparing between the same impact category.

3.3.3.1 Intended EoL technology: Recycling of LDPE

For both LDPE carrier bags, recycling performs the best in 11 impact categories and the worst in 2 impact categories. The substitution of LDPE contributes to a large amount of savings in impact categories such as abiotic depletion (fossil fuels), NREU and particulate matter. In the impact category water use, the recycling EoL performs worst, due to water consumption in the recycling facility. Transportation has the largest contribution in the other impact category were recycling performs the worst, human toxicity (non-cancer). The contribution analysis of the recycling shows that the substitution of plastic has the largest contribution in 14 out of 18 impact categories. Altering the technical substitution rate and the sorting efficiency will therefore have a large impact on the overall LCIA results. In the bio-based LDPE product system, treatment of rejects from incineration contributes to 91 % of the total biogenic climate change impact, but for the total climate change impact, treatment of rejects only contributes to 33 %. However, for the petrochemical LDPE carrier bag the treatment of rejects only contributes to 17 % of total climate change impact.

3.3.3.2 Landfilling of LDPE

For both LDPE carrier bag product systems, landfilling performs the best in 2 impact categories, second in 4 and worst in 5. The LDPE carrier bags do not degrade and therefore generates no landfill gas, avoiding direct emissions to the environment. Other impacts from landfill originate from construction and operation of the landfill, and leachate treatment. The contamination is assumed to consist of organic waste, that does degrade in the landfill, and thereby generates a small amount of methane gas. The construction and operation of the landfill together with the transport have the largest contribution in 9 out of 18 impact categories. In the impact categories of freshwater ecotoxicity and marine eutrophication the leachate treatment has the highest contribution. The small amount of landfill gas from the contamination causes the highest impacts in terms of biogenic and total climate change, ozone depletion and photochemical ozone formation.

3.3.3.3 Incineration of LDPE

Incineration

of both types of LDPE carrier bags performs best in 3 impact categories, the second best in 4 and the worst in 9 impact categories. Incineration has the lowest impact in water use and abiotic depletion due to savings from electricity substitution. Direct emissions from the incineration plant contribute to the largest part of the impact categories. The incineration performs the worst in 9 categories, mainly due to impacts from the direct emissions in categories such as total climate change, photochemical ozone formation, terrestrial and marine eutrophication.

The contribution analysis of incineration of both LDPE product systems shows that the substitution of heat and electricity has the largest contribution to the overall LCIA EoL results in most impact categories. In the impact categories of freshwater ecotoxicity and human toxicity (cancer effects), the treatment of bottom ash and fly ash have the highest contribution, mostly due to emissions of metals to water and soil.

3.3.4 Details on climate change

This section analysis across all three carrier bag product systems and all EoL technologies, i.e. comparing all results in Table 216 and Table 217. For illustration, the climate change is presented in, Figure 195 with a contribution broken down between processes in the technologies. Note that this section still compares per EoL-reference flow, i.e. 1 kg plastic and

100 g contamination (does not consider the difference in weight for the FU – the bio-based starch bag is 25 % heavier).

The recycling of the LDPE product systems has the lowest total impact of -0.077 kg CO_2-eq. The intended EoL technology for the alternative system of bio-based LDPE is recycling, which also performs best out of the other relevant EoL technologies. This is mostly due to high savings from plastic substitution.

The climate change impact of landfilling both bio-based and petrochemical LDPE carrier bags also has a low impact of 0.24 kg CO_2-eq. Industrial composting of bio-based starch carrier bags performs the third best in terms of climate change out of the relevant EoL technologies (emission of 0.39 kg CO_2 eq. / 1 kg starch) and anaerobic digestion of bio-based starch (0.6 kg CO_2 eq. / kg starch) is the fourth best. Incineration of the starch performs a little better than the LDPE bags. Regarding climate change, landfilling of starch is clearly the worst performing pathway for the bio-based starch carrier bag, with an emission of 3.12 kg of CO_2 eq. per kg starch.

If these climate change impacts were compared per FU, the ranking would be similar, though the incineration of the three product systems would be more similar.

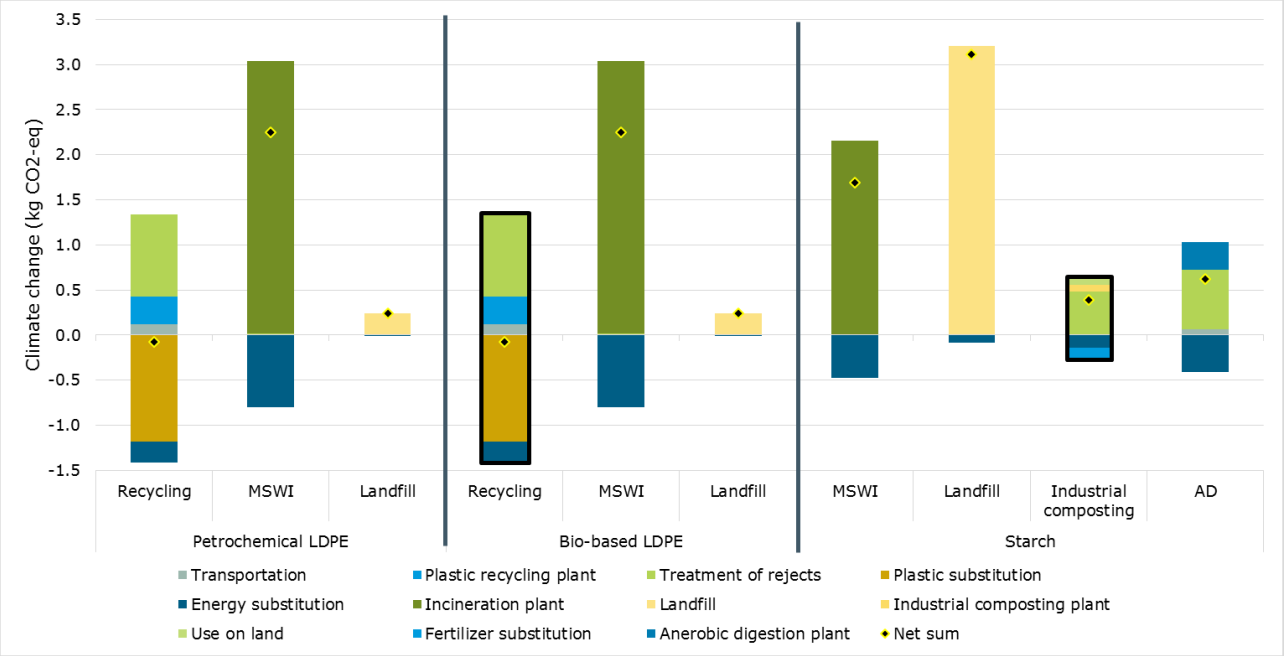


Figure 195. Climate change of EoL technologies for petrochemical LDPE, bio-based LDPE and starch carrier bags for 1 kg plastic and 100 g contamination. The intended EoL option is marked with a black box. See Annex 4 for details of the content of the grouped processes. The results without energy recovery can be interpreted by excluding the energy substitution, and for incineration with energy substitution the energy substitution shall be multiplied by 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

3.3.5 Comparing the EoL technologies in all product systems with weighted results

In the sections above, results have been shown for the EoL-reference flow of 1 kg plastic and 100 g contamination. To make a comparison, the weighted results for all product systems is presented per functional unit.

The impacts are similar overall for many of the relevant EoL technologies and it is therefore dependent on the weighting of each impact category, which EoL performs the best. The weighted results with toxicity for all product systems and EoL technologies are presented in Figure 196.

From the figure, it can be concluded that recycling of both LDPE product systems is the preferred choice, with the lowest weighted score. The weighted results show that the landfilling of both LDPE product systems and the industrial composting of bio-based starch are similar. However, the industrial composting of bio-based starch carrier bag has a lower total weighted score with toxicity. There are savings from recycling the organic materials in several impact categories, but no impact savings are achieved for landfilling LDPE. The anaerobic digestion follows as the second best EoL technology for the bio-based starch carrier bag, third is incineration and the least preferred EoL option is landfilling.

The contributions from all EoL options, except recycling of the LDPE, are pre-dominantly to climate change, while for recycling the savings are from abiotic depletion of fossil fuels. The climate change contribution from incineration stem from direct emissions for all product systems. Regarding landfilling, the contribution for the carrier bags is from un-collected LFG and flared LFG. Industrial composting and anaerobic digestion of the starch have emissions from use on land as well as savings from fertiliser substitution. The contribution from abiotic depletion of fossil fuels when recycling the LDPE are from plastic substitution.

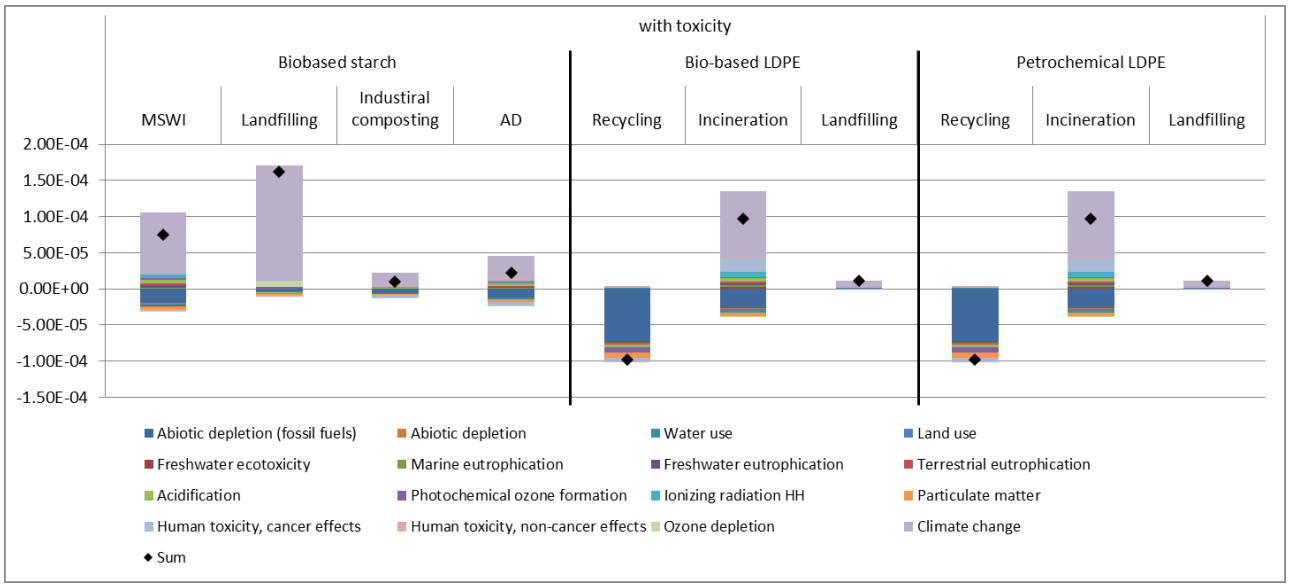


Figure 196. Weighted results with toxicity for the single-use carrier bags and their individual EoL options per functional unit. The numerical values can be found in Annex 4.

3.3.6 End of life mix results

The EoL mix results are presented per functional unit in Table 218. The EoL mix for the bio-based starch carrier bag is composed of industrial composting, incineration and landfilling. The bio-based LDPE and petrochemical LDPE EoL mix is composed of recycling, incineration and landfilling.

The results for both LDPE carrier bags are the same, except for the biogenic carbon, as the material composition is the same, but the carbon is biogenic instead of fossil for the bio-based LDPE. Overall, the LDPE carrier bags seems to perform better than the starch-based carrier bag with the current EoL mix. There is large share of landfilling in the EoL mix of the starch carrier bag, as landfill has the poorest performance of all EoL technologies for most impact categories. The landfilling of starch performs much worse than landfilling LDPE, because the

starch is degradable and emits e.g. greenhouse gasses. This is also due to the fact that industrial composting of the starch plastic bags performs worse than the recycling of the LDPE. In the impact category of ozone depletion, there is a large difference between the LDPE bags and the starch bag. Other categories where the LDPE bags performs much better are climate change, acidification, terrestrial and marine eutrophication.

Table 218. LCIA for EoL mix of the carrier bag for all product systems per functional unit (1 carrier bag).

Impact category	Unit	Bio-based starch	Bio-based LDPE	Petrochemical LDE
Industrial composting	%	30		
Recycling	%		30	30
Incineration	%	39	39	39
Landfill	%	31	31	31
Climate change - fossil	kg CO_2 eq / FU	1.14E-02	-9.37E-03	1.59E-02
Climate change - biogenic	kg CO_2 eq / FU	2.76E-02	2.60E-02	6.93E-04
Climate change - total	kg CO_2 eq / FU	3.90E-02	1.66E-02	1.66E-02
Ozone depletion	kg CFC-11 eq / FU	8.60E-09	-2.39E-12	-2.39E-12
Human toxicity, non-cancer effects	CTUh / FU	1.25E-10	2.72E-10	2.72E-10
Human toxicity, cancer effects	CTUh / FU	-3.75E-11	1.12E-10	1.12E-10
Particulate matter	kg PM2.5 eq / FU	-8.41E-07	-1.80E-06	-1.80E-06
Ionizing radiation HH	kBq U235 eq / FU	4.62E-04	6.38E-04	6.38E-04
Photochemical ozone formation	kg NMVOC eq / FU	1.43E-05	-7.66E-06	-7.66E-06
Acidification	molc H+ eq / FU	1.36E-05	5.80E-06	5.80E-06
Terrestrial eutrophication	molc N eq / FU	5.25E-05	2.55E-05	2.55E-05
Freshwater eutrophication	kg P eq / FU	6.26E-07	5.46E-07	5.46E-07
Marine eutrophication	kg N eq / FU	4.62E-06	2.04E-06	2.04E-06
Freshwater ecotoxicity	CTUe / FU	-1.55E-03	3.46E-03	3.46E-03
Land use	kg C deficit / FU	0.00E+00	0.00E+00	0.00E+00
Water use	m^3 / FU	-1.83E-03	-1.68E-03	-1.68E-03
Abiotic depletion	kg Sb eq / FU	-3.02E-09	-3.35E-09	-3.35E-09
Abiotic depletion (fossil fuels)	MJ / FU	-7.82E-02	-2.54E-01	-2.54E-01
NREU	MJ / FU	-8.61E-02	-2.71E-01	-2.71E-01

3.4 Cradle-to-grave results and interpretation

3.4.1 Bio-Based baseline system

The aggregated cradle-to-grave LCA results from the assessment of 1 functional unit of bio-based carrier bags (22.4 g) are given in Table 219. The breakdown of impacts between biomass production, manufacturing and EoL is shown in Figure 197. For bio-based carrier bags, 100 % industrial composting is the desired/intended EoL scenario. The EU mix of real EoL scenario is modelled as 30 % industrial composting, 39 % incineration and 31 % landfilling (see section 2.2). In Table 219, EoL values per functional unit for both scenarios are given. In Figure 197, the desired EoL is taken as the base case (assigned 100 %). The manufacturing phase (conversion of biomass to carrier bags) dominates the impacts in most categories except for marine eutrophication, land use, water use and abiotic depletion. In these categories biomass production has a significant impact. Biomass production has a minor contribution to the other impact categories. EoL has negligible impacts in the overall cradle-to-grave results except for climate change. The climate change impact of the real EoL was found to be significantly higher than the desired EoL of industrial composting. This is due to the

direct emissions from the incineration plant and landfill gas produced in landfilling (both from oxidation under cover and from direct emissions from flaring) of the starch plastic.

Table 219. Cradle-to-grave LCA results breakdown of 1 functional unit bio-based carrier bags, baseline (excluding DLUC and ILUC effects).

Impact category	Unit	Biomass	Manufacturing	EoL (Intended)	EoL (EU mix)
Climate change	kg CO_2 eq.	-1.04E-02	5.51E-02	7.16E-03	3.85E-02
Ozone depletion	kg CFC-11eq	3.03E-10	1.42E-08	-2.48E-10	8.59E-09
Human toxicity, non-cancer effects	CTUh	3.07E-09	9.24E-09	7.59E-10	1.16E-10
Human toxicity, cancer effects	CTUh	2.30E-10	2.81E-09	-6.47E-11	-3.75E-11
Particulate matter	kg PM2.5eq	2.50E-06	2.55E-05	-5.37E-07	-8.84E-07
Ionizing radiation HH	kBq U235eq	1.02E-04	1.18E-02	3.27E-04	4.61E-04
Photochemical ozone formation	kg NMVOCeq	1.63E-05	1.65E-04	9.53E-07	1.36E-05
Acidification	molc H+eq	4.18E-05	3.07E-04	5.59E-06	1.27E-05
Terrestrial eutrophication	molc Neq	1.69E-04	4.62E-04	1.40E-05	4.89E-05
Freshwater eutrophication	kg Peq	1.20E-06	2.02E-05	8.21E-08	6.25E-07
Marine eutrophication	kg Neq	7.35E-05	4.16E-05	1.73E-06	4.36E-06
Freshwater ecotoxicity	CTUe	2.36E-02	2.99E-01	-9.89E-04	-1.56E-03
Land use	kg C deficit	9.71E-02	1.11E-01	0.00E+00	0.00E+00
Water use	m^3	2.72E-02	1.30E-02	-1.22E-03	-2.04E-03
Abiotic depletion	kg Sbeq	2.83E-08	3.13E-08	-2.00E-09	-3.02E-09
Abiotic depletion (fossil fuels)	MJ	3.47E-02	1.19E+00	-4.48E-02	-8.31E-02
NREU	MJ	3.64E-02	1.37E+00	-5.12E-02	-8.96E-02

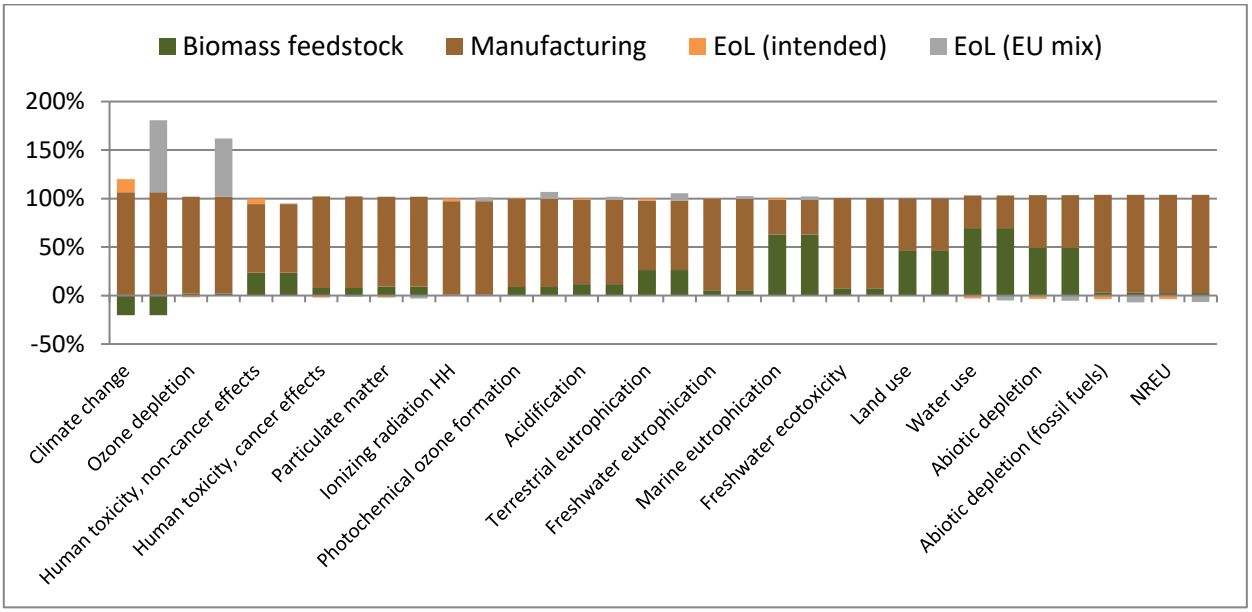
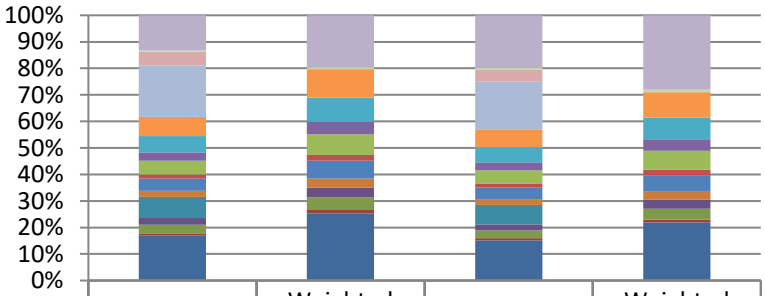


Figure 197. Breakdown of the potential impacts from the starch-based carrier bags across all impact categories, cradle-to-grave results (excluding DLUC and ILUC effects). The graph is normalised to the 100 % desired EoL technology.

According to PEFCR guidance v 6.3, the most relevant impact categories are identified as all impact categories that cumulatively contribute to at least 80 % of the total environmental impact (excluding toxicity related impact categories). Based on the normalised and weighted results shown in Figure 198 for both EoL options, excluding toxicity related impacts, the most important impact categories are: abiotic depletion (fossil fuels), climate change, particulate matter, ionizing radiation, acidification, freshwater eutrophication and photochemical ozone formation. These seven impact categories form ca. 84 % of the weighted impacts. These impact categories are identified as the hot spots for this case study and detailed analysis is made for each of them in section 3.1.1.

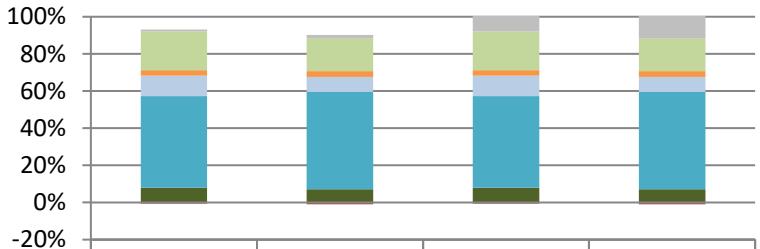


	Based on intended EoL		Based on EoL EU mix	
	Weighted score	Weighted score (excluding toxicity categories)	Weighted score	Weighted score (excluding toxicity categories)
Climate change	13%	20%	20%	28%
Ozone depletion	0%	1%	1%	1%
Human toxicity, non-cancer effects	5%	0%	4%	0%
Human toxicity, cancer effects	19%	0%	18%	0%
Particulate matter	7%	11%	7%	10%
Ionizing radiation human health	6%	9%	6%	8%
Photochemical ozone formation	3%	5%	3%	4%
Acidification	5%	8%	5%	7%
Terrestrial eutrophication	2%	2%	1%	2%
Freshwater eutrophication	5%	7%	4%	6%
Marine eutrophication	2%	3%	2%	3%
Freshwater ecotoxicity	8%	0%	7%	0%
Land use	2%	4%	2%	3%
Water use	3%	5%	3%	4%
Abiotic depletion	1%	1%	1%	1%
Abiotic depletion (fossil fuels)	17%	25%	15%	22%

Figure 198. Contribution of the 16 PEF categories based on normalised and weighted results of bio-based starch plastic carrier bags, with and without toxicity categories

Contributions of the life cycle stages and the different processes within are shown in Figure 199. Please refer to section 3.1.1 for a more detailed elaboration per impact category of the

contributing processes. As shown in Figure 199, the manufacturing phase (conversion of biomass to carrier bags) is the most relevant life cycle stage cumulatively contributing to 80/83 % of the impacts. The copolyester and other raw material production is the most significant process with about a 50 % contribution followed by carrier bag (blown film extrusion) (18-21 %) and granulate production (8-11 %) processes. For biomass production, the biogenic carbon sequestered results in low overall impact for this stage (7-8 %). The EoL is found to have very low contribution (1-2 %) to the overall impacts with, the desired EoL scenario of industrial composting showing low impacts to the environment. Using the real scenario for EoL, the contribution of this life stage significantly increases to 9-12 %, showing higher impacts associated with incineration and landfilling EoL options.



	Based on intended EoL		Based on EoL EU mix	
	Weighted score	Weighted score (excluding toxicity categories)	Weighted score	Weighted score (excluding toxicity categories)
EoL	1%	2%	9%	12%
Carrier bag production	21%	18%	21%	18%
Transportation	3%	3%	3%	3%
Granulate production	11%	8%	11%	8%
Copolyester and other raw materials production	49%	52%	49%	52%
Starch production	-1%	-1%	-1%	-1%
Biomass production	8%	7%	8%	7%

Figure 199. Contribution of the life cycle stages and processes based on normalised and weighted results of bio-based starch plastic carrier bags, with and without toxicity categories

3.4.2 Bio-Based alternative system (Bio-Based LDPE carrier bags)

The cradle-to-grave LCA results from the assessment of 1 functional unit of bio-based LDPE carrier bags (17.9 g) are given in Table 220. The breakdown of impacts between biomass production, manufacturing and EoL is shown in Figure 200. For EoL of bio-based LDPE carrier bags, the EoL mix is modelled as 30 % recycling, 39 % incineration and 31 % landfilling (see section 2.2).

Because this is a 100 % bio-based material, the biogenic carbon stored in the bag provides significant savings. The biomass production dominates the impacts in many impact categories including particulate matter, land use, abiotic depletion, terrestrial eutrophication, marine eutrophication, and photochemical ozone formation.

The manufacturing phase (conversion of biomass to carrier bags) shows most significant contribution in water use (due to ethylene production), in ozone depletion (due to

polymerization), in ionizing radiation, and in freshwater eutrophication (due to blown film extrusion).

The EoL has a negligible impact in the overall cradle-to-grave results except for climate change impacts, abiotic depletion (fossil fuels) and NREU. The climate change impact is due to the direct emissions from the incineration plant and landfill gas produced in landfilling (both from oxidation in covers and from direct emissions from flaring) of the starch plastic. The savings in abiotic depletion (fossil fuels) and NREU are due to the recycling from plastic substitution.

Table 220. Cradle-to-grave LCA results breakdown of 1 functional unit bio-based LDPE carrier bags, alternative (excluding DLUC and ILUC effects).

Impact category	Unit	Biomass	Manufacturing	EoL (mix)
Climate change	kg CO_2 eq.	-3.55E-02	3.07E-02	1.66E-02
Ozone depletion	kg CFC-11eq	1.54E-09	6.48E-09	-2.39E-12
Human toxicity, non-cancer effects	CTUh	2.92E-08	2.56E-08	2.72E-10
Human toxicity, cancer effects	CTUh	7.94E-10	1.14E-09	1.12E-10
Particulate matter	kg PM2.5eq	6.80E-04	8.14E-05	-1.80E-06
Ionizing radiation, human toxicity	kBq U235eq	2.67E-04	6.67E-03	6.38E-04
Photochemical ozone formation	kg NMVOCeq	2.50E-04	2.30E-04	-7.66E-06
Acidification	molc H+eq	3.13E-04	2.62E-04	5.80E-06
Terrestrial eutrophication	molc Neq	1.48E-03	7.36E-04	2.55E-05
Freshwater eutrophication	kg Peq	2.05E-06	1.39E-05	5.46E-07
Marine eutrophication	kg Neq	7.74E-05	6.74E-05	2.04E-06
Freshwater ecotoxicity	CTUe	1.36E-01	1.67E-01	3.46E-03
Land use	kg C deficit	7.37E-01	2.92E-02	0.00E+00
Water use	m^3	2.87E-03	2.28E-01	-1.68E-03
Abiotic depletion	kg Sbeq	5.01E-08	2.34E-08	-3.35E-09
Abiotic depletion (fossil fuels)	MJ	1.26E-01	4.20E-01	-2.54E-01
NREU	MJ	1.29E-01	5.28E-01	-2.71E-01

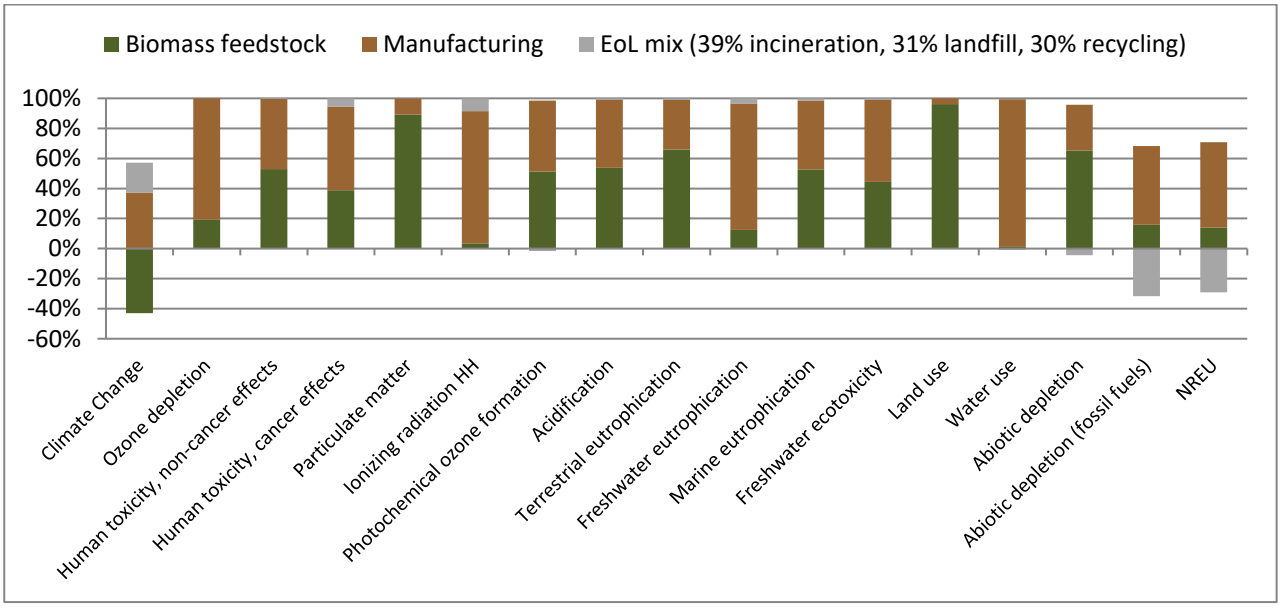


Figure 200. Breakdown of the potential impacts from the bio-based LDPE carrier bags, alternative across all impact categories, cradle-to-grave results (excluding DLUC and ILUC effects).

3.4.3 Petrochemical reference system

The cradle-to-grave LCA results for 1 functional unit (17.9 g) of petrochemical LDPE carrier bags are presented in Table 221. The breakdown of impacts between manufacturing and EoL is shown in Figure 201. The estimated EoL mix for petrochemical LDPE carrier bag is 30 % recycling, 39 % incineration and 31 % landfilling (see section 2.2). The EoL has a 28 % contribution to climate change caused by the direct emissions from the incineration plant. The EoL provides savings in particulate matter, abiotic depletion (fossil fuels) and NREU due to the plastic substitution with recycling and energy/heat substitution with incineration.

Table 221. Cradle-to-grave LCA results of 1 functional unit petrochemical LDPE carrier bags.

Impact category	Unit	Manufacturing	EoL	Total
Climate change	kg CO_2 eq.	4.35E-02	1.66E-02	6.01E-02
Particulate matter	kg PM2.5 eq	9.52E-06	-1.80E-06	7.72E-06
Photochemical ozone formation	kg NMVOC eq	1.67E-04	-7.66E-06	1.59E-04
Acidification	molc H+ eq	1.43E-04	5.80E-06	1.49E-04
Terrestrial eutrophication	molc N eq	3.11E-04	2.55E-05	3.36E-04
Marine eutrophication	kg N eq	3.03E-05	2.04E-06	3.24E-05
Abiotic depletion (fossil fuels)	MJ	1.48E+00	-2.54E-01	1.23E+00
Non-renewable energy use (NREU)	MJ	1.61E+00	-2.71E-01	1.34E+00

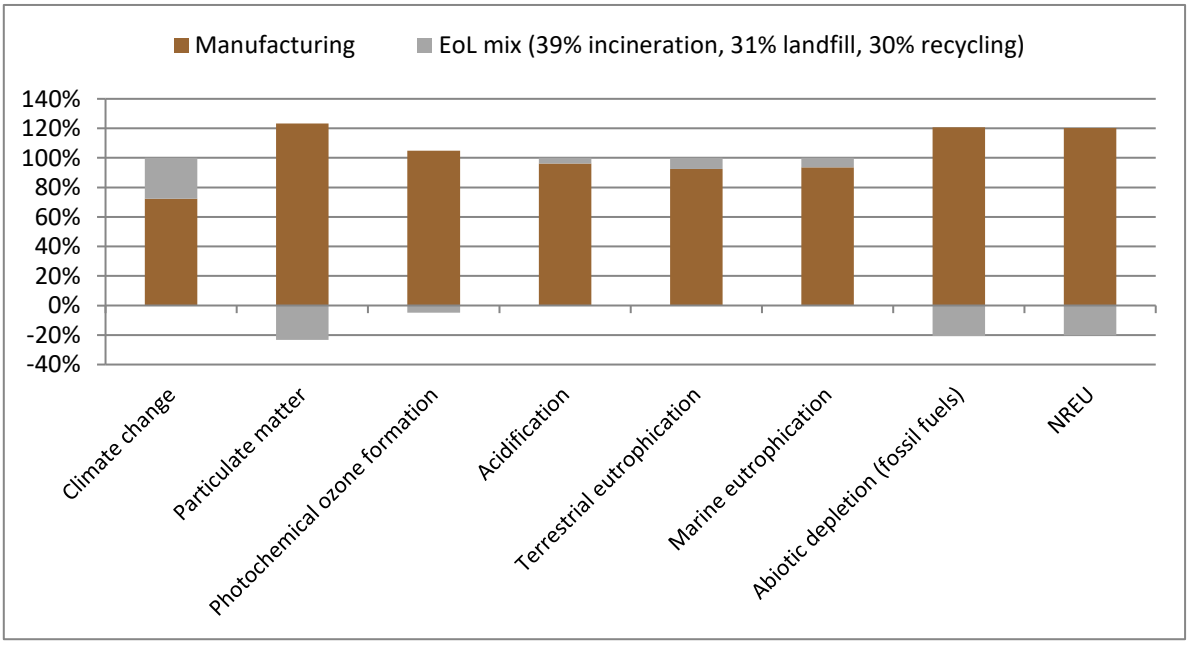


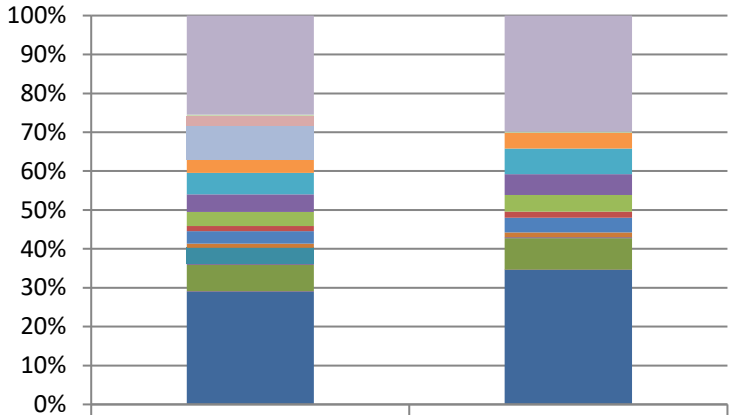
Figure 201. Breakdown of the cradle-to-gate LCA results for petrochemical LDPE carrier bag

Normalisation and weighting have been applied to the reference system (petrochemical LDPE carrier bags) to understand which among the assessed impact categories are more relevant for this application and what the main environmental hotspots are. To do this, all the impact categories needed to be taken into the calculation. Hence, the weighted results are subject to considerable uncertainties, as highlighted in *Chapter 6 Ranges for environmental impacts from production of fossil-based plastics*, therefore their numerical values must be considered only as indicative.

Based on the normalised and weighted results shown in Figure 203, excluding toxicity related impacts, the most relevant impact categories are: abiotic depletion (fossil fuels) (35 %), climate change (30 %), water use (8 %) and ionizing radiation (7 %), which cumulatively

contribute 80 % of the total impacts. The rest of the impact categories are less relevant with each contributing up to 5 %.

Contributions of the life cycle stages and the different processes within are given in Figure 202. Manufacturing of granulate and bags are the most relevant life cycle stage, contributing to a cumulative 98 % of the impacts with 69 % (79 % excluding toxicity categories) of the contribution coming from granulate production (extraction and refining of crude oil and natural gas, steam cracking of hydrocarbons into lower olefins, and polymerisation of the monomers into polyethylene). The electricity used for carrier bags production also has a significant contribution (27 % with /19 % without toxicity categories). The EoL life cycle stage is seen to have a negligible contribution due to the positive and negative impacts from the incineration, landfilling and recycling processes balancing each other out in the overall analysis (Figure 203).



	Weighted score	Weighted score (excluding toxicity categories)
Climate change	25%	30%
Ozone depletion	0%	0%
Human toxicity, non-cancer effects	2%	0%
Human toxicity, cancer effects	9%	0%
Particulate matter	3%	4%
Ionizing radiation HH	6%	7%
Photochemical ozone formation	4%	5%
Acidification	4%	4%
Terrestrial eutrophication	1%	2%
Freshwater eutrophication	3%	4%
Marine eutrophication	1%	1%
Freshwater ecotoxicity	4%	0%
Land use	0%	0%
Water use	7%	8%
Abiotic depletion	0%	0%
Abiotic depletion (fossil fuels)	29%	35%

Figure 202. Contribution of the 16 PEF categories based on normalised and weighted results of petrochemical LDPE carrier bags, with and without toxicity categories.

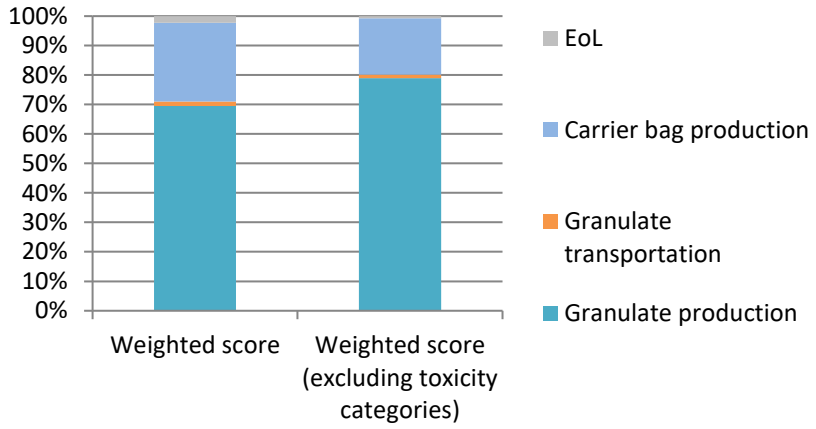


Figure 203. Contribution of the life cycle stages and processes based on normalised and weighted results of petrochemical LDPE carrier bags, cradle-to-grave with and without toxicity categories.

3.4.4 Comparing the bio-based baseline system with the bio-based alternative and petrochemical reference

Comparing the bio-based baseline system with the bio-based alternative

The comparison of cradle-to-grave results for the bio-based carrier bags baseline system per functional unit (22.4 g) with the alternative system of bio-based LDPE (17.9 g) is given in Figure 204. For the baseline (starch plastic carrier bags), both EoL scenarios are provided. The 100 % bio-based alternative system of bio-based LDPE performs significantly better (>75 %) in three categories: climate change, abiotic depletion (fossil fuels) and NREU. Because of the particulate emissions from sugarcane waste burning, it performs much worse (by a factor of 30) in this category. Compared to the baseline system, which is about 30-60 % bio-based, the effects coming from biomass production are more pronounced. This results in significantly higher impacts in the photochemical ozone formation, acidification, terrestrial eutrophication, land use and water use categories.

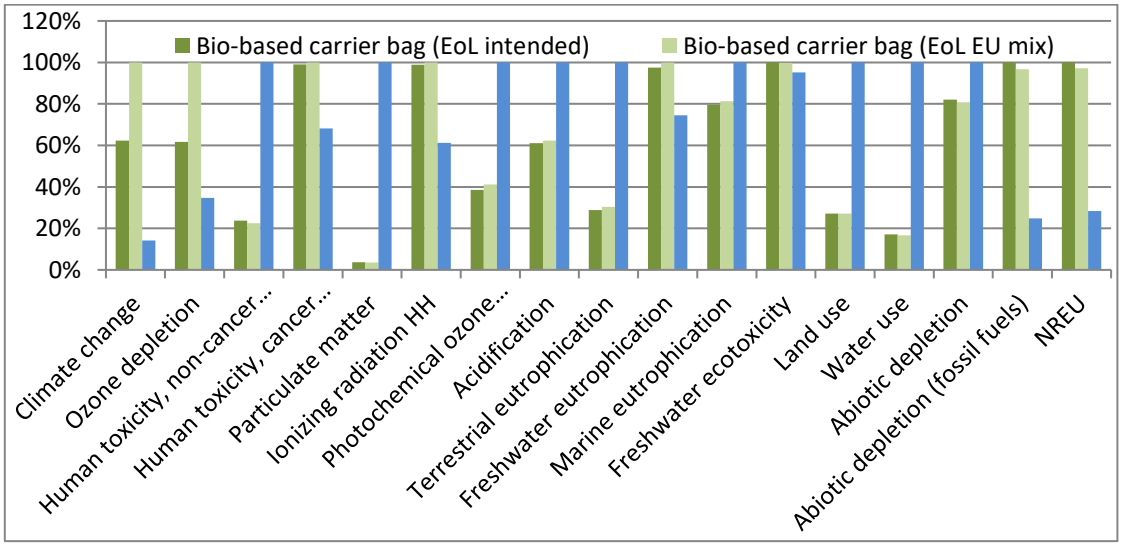


Figure 204. Comparing bio-based carrier bags baseline (starch plastic) system with bio-based alternative (bio-based LDPE) system, cradle-to-grave results (excluding DLUC and ILUC effects).

Comparing the bio-based systems with petrochemical reference

The comparison of cradle-to-grave results of bio-based starch plastic bags (22.4 g/FU), bio-based LDPE bags (17.9 g/FU) and petrochemical carrier bag (LDPE, 17.9 g/FU) per functional unit is provided in Figure 205. The bio-based carrier bags perform significantly worse (up to 75 %) in four impact categories (particulate matter, acidification, terrestrial and marine eutrophication). For photochemical ozone formation, the baseline system is comparable with the petrochemical system, but bio-based LDPE performs significantly worse. For the abiotic depletion (fossil fuels) and NREU categories, starch plastic and petrochemical carrier bags show similar performance. However, bio-based LDPE performs >70 % better. In the climate change category, considering the desired EoL scenario, starch plastic carrier bags perform 10 % better than the petrochemical bags. However, with the current real EoL scenario, 30 % lower performance than petrochemical is seen. Bio-based LDPE gives 80 % better climate change performance than the petrochemical. Overall, starch plastic carrier bags don't provide clear environmental benefits in any of the impact categories selected as suitable for comparison. The bio-based LDPE carrier bags offer better performance in climate change, abiotic depletion (fossil fuels) and NREU.

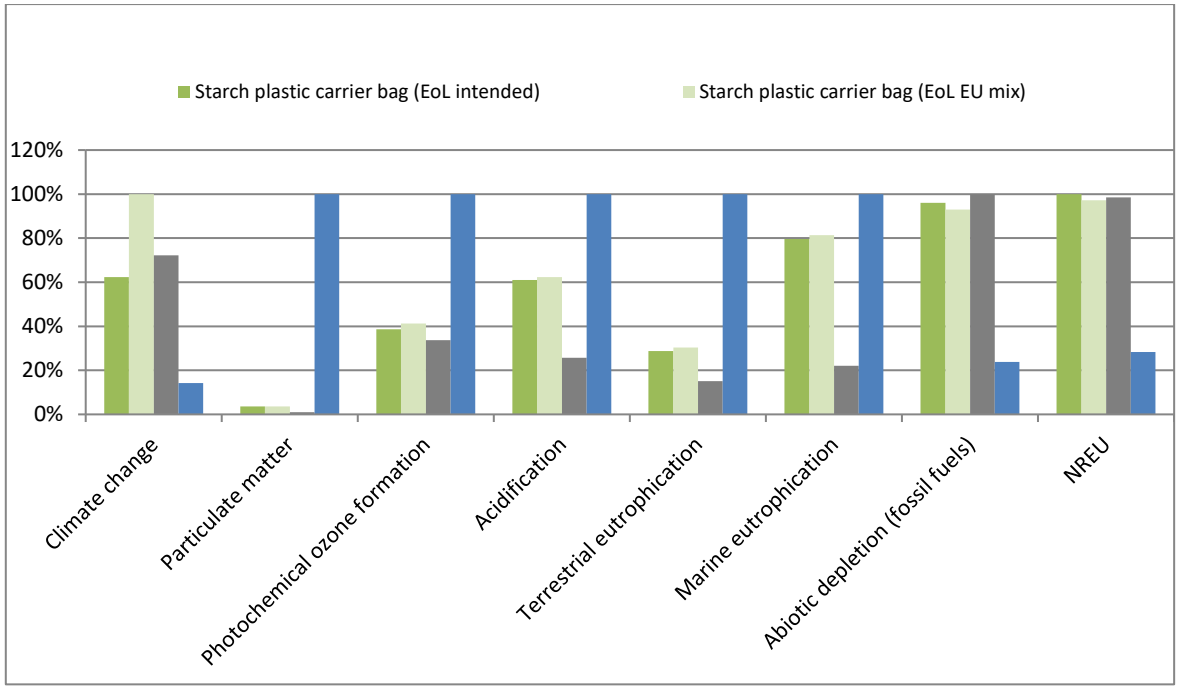


Figure 205. Comparing bio-based carrier bags with petrochemical carrier bag, cradle-to-grave results (excluding DLUC and ILUC effects).

4 Sensitivity analysis and discussion

4.1 Sensitivity analysis

4.1.1 Sensitivity analysis for carrier bag weight

A sensitivity analysis was performed to assess the variation of the impact assessment results due to the variation of the carrier bags weight. The average weight of an LDPE bag was selected as 17.9 g in section 1.3 to fulfil the functional unit. The Literature review showed that the weight of the LDPE carrier bags can be 17 % lower (14.8 g) (Mori et al., 2013). It can also be 17 % higher (20.9 g) (Muthu and Li, 2014). This variation affects proportionally all the assessed environmental impacts, i.e. all the impacts are 17 % higher or lower. Therefore, the weight of the carrier bag is one of most sensitive parameter to be chosen once the functional unit has been fixed. The weight of the bio-based starch biopolymer bag is 25 % higher, so it would vary between 18.5-26.1 g. Since the change happens proportionally, the comparison between bio-based and fossil-based carrier bags doesn't change, but the absolute values of the impact assessment results change.

4.1.2 Starch industrial composting

The sensitivity analysis was performed for the intended EoL option, hence industrial composting of starch. The significant parameter chosen, which is likely to change due to an increase in technical improvements in both the plastic products and in the waste management technology, was the degradation percentage of the biogenic carbon in the product. Changes in this parameter within a given technology were modelled because the average European technology in the EoL scenarios represent a rather broad set of technological variances. This change provides insight into the variance within these technologies and depicst a target for future waste scenarios. The parameter variations are presented in Table 222.

Table 222. Variation in input parameters in the sensitivity analysis.

Product system	Technology	Parameter changed	Current	Lower	Higher	References
Starch	Industrial Composting	Biogenic carbon degradation (%)	80	63	97	Hermann, 2011

The sensitivity analysis was performed on the industrial composting technology. The C degradation was changed according literature limits given in Table 222. Changing the degradation rate of C only affects the climate change impact categories. Figure 206 presents the change in climate change when altering the C degradation compared to the original for industrial composting of bio-based starch carrier bags. A higher degradation rate gives a lower total climate change impact.

Using a higher C degradation leads to a decrease in the total climate change impact of 4 % compared to the original. The climate change impact change is higher for the biogenic carbon and there are three main reasons for these changes: 1) Increasing the degradation percentages means that less of the biogenic carbon will be incinerated with the rejects and contribute to direct emissions of biogenic CO_2; 2) Less biogenic carbon will be released to the atmosphere in the form of the compost product that is spread on the land later o;. 3) More biogenic carbon is emitted directly from the composting plant when the biodegradation percentage is higher.

Considering the fossil CO_2, a lower C degradation will lead to a small saving of fossil CO_2 as there is less compost to be spread on land with a diesel consuming tractor. A higher C

degradation will lead to less savings of fossil CO_2 when the compost is used on land, which is why the change is positive.

The lower and higher carbon degradation would not cause changes in the conclusion that industrial composting preforms best in terms of climate change impacts for PLA.

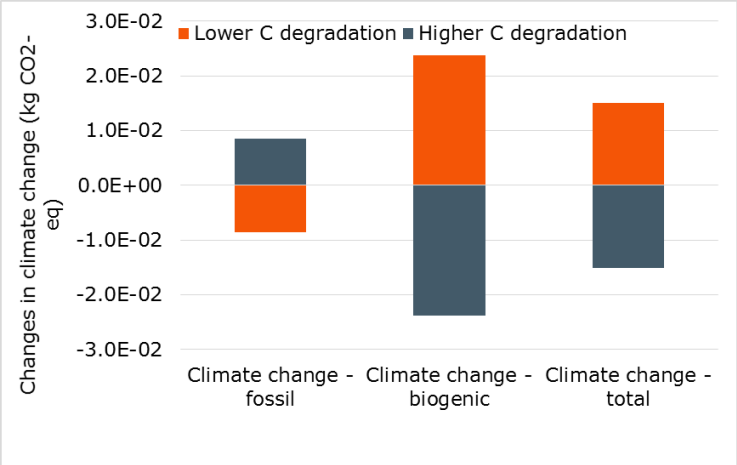


Figure 206. Sensitivity analysis for bio-based starch carrier bags EoL.

This sensitivity would change the cradle-to-grave climate change results for the EoL mix of the starch-based carrier bag by +0.2 % and −0.2 % for respectively a lower and a higher carbon degradation.

4.2 Reuse scenarios

The carrier bag is of a heavier quality and is therefore likely to be reused by the consumer. Taking this into account the carrier bags have in addition to the full EoL technology and the EoL mix, been modelled with both primary and secondary reuse, before disposal. In the use phase it is assumed there is not impacts.

- Primary reuse is defined as reuse of the carrier bag as a carrier bag again. Reusing the carrier bag X times allows for an avoidance of production, use and disposal X times of a carrier bag (either petrochemical LDPE or bio-based starch).

- Secondary reuse is defined as reuse of the carrier bag as a waste disposal bag. The effect the reuse as a waste bag allows for the avoidance of production and disposal of a thin waste bag liner (corresponding to the product system bag material; petrochemical LDPE, bio-based LDPE or bio-based starch).

Furthermore, the primary and secondary reuse scenarios can be combined, as illustrated in Figure 207. The primary reuse of carrier bags before disposal with the possibility of secondary reuse as a waste bag afterwards was analysed for all the carrier bag product systems.

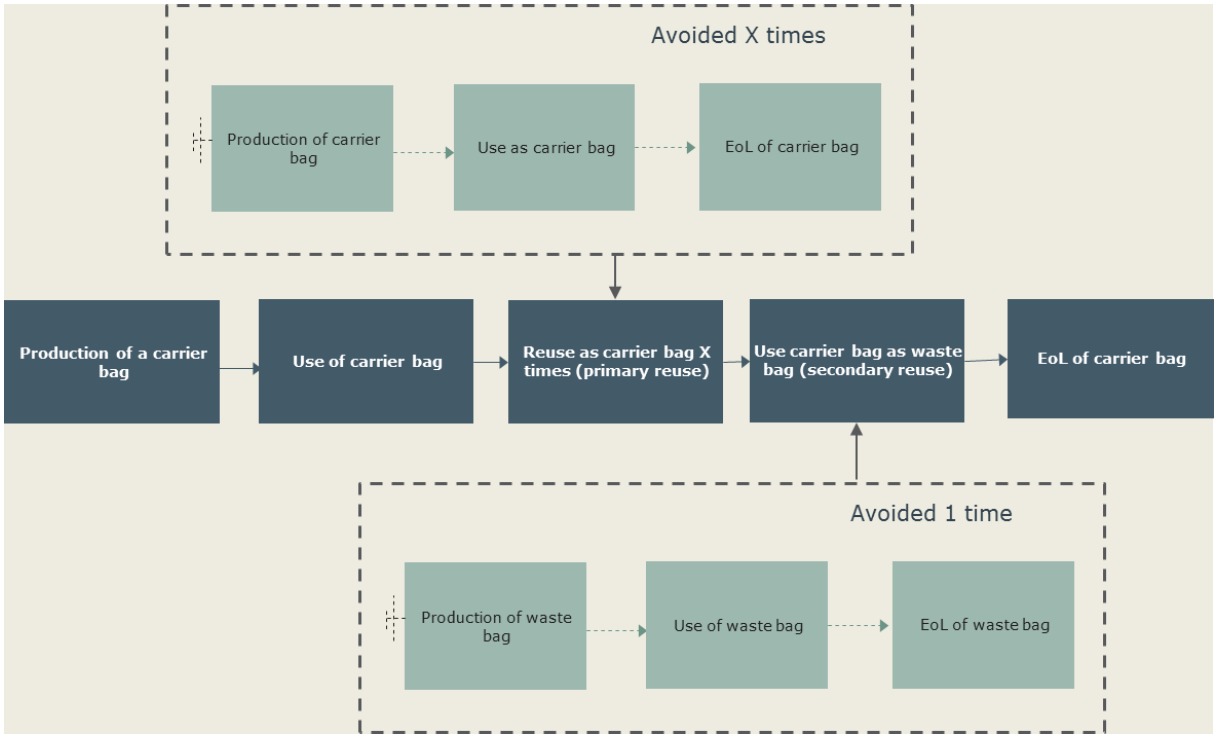


Figure 207. Flow chart primary and secondary reuse as carrier bag.

The results of the reuse scenario are presented in Annex 4 (Case 7: Carrier bag reuse scenarios). The analysis for reuse of all the EoL technologies was performed by analysing the effect on climate change impact results when the carrier bag was reused up to 20 times and then disposed of via different EoL technologies (primary reuse). Furthermore, the EoL mix for all products systems is presented with results for all impact categories, showing the performance of 1) no reuse, 2) 1 x primary reuse, 3) 1 x primary and secondary reuse, 4) 10 x primary reuse and 5) 10 x primary and secondary reuse.

The overall conclusion of the reuse scenarios is that, the more often the carrier bag is used the more environmental savings are achieved. Reusing the carrier bag gives greater savings than using it as a waste bag. The best scenario is to reuse it as many times as possible and then use it as a waste bag. Similar environmental assessments on carrier bags have been performed with the same results, showing the more reuse, the more environmental savings (Miljøstyrelsen, 2018; Edwards et al., 2011).

4.3 Discussion on End of Life uncertainties

The parameters analysed in the sensitivity analysis above and other parameters, which may be uncertain, i.e., estimations, assumptions, huge ranges in industry or EU member states, missing data, etc. are discussed in this section. Methodological shortcomings and additional research requirements are also presented.

Frequency of re-use. This factor is highly dependent on the quality of a plastic carrier bag; the better the quality, the higher is the likelihood that the bag is reused. If the biodegradable plastic bag has a lifespan of one year before it starts to degrade it is less likely to be reused multiple times compared will an LDPE bag. The carrier bag will likely serve as a waste bag before it enters the waste stream. However, it may be that the biodegradable bag is used for organic waste and thereby enters a different waste stream than the fossil-based bag. The when and whereabouts of disposed plastic bags depends on the consumer, which is subject to a level of uncertainty.

Biodegradation of biogenic C in starch carrier bags in industrial composting facilities. A sensitivity analysis for this parameter was performed. Increasing/decreasing the C biodegradation affects the cradle-to-grave climate change results for EoL mix by only ±0.2 % (see Figure 206), which shows that this is not a sensitive parameter for the overall cradle-to-gate results.

Biodegradation levels in AD are uncertain because this EoL technology is complex and the AD plants vary depending on installed technologies. Several factors within the AD plant will affect the biodegradation, for example temperature, moisture, time and post treatment practices. Chemical structure and plastic thickness will also affect the degradation process. Plastic bags can cause challenges in wet AD systems if not shredded in advance of processing, as they have a tendency to get entangled on rotating pumps or other devices (De Wilde, 2016).

Littering of plastic bags has been a topic of discussion in recent years. Light weight plastic is more prone to being blown from waste bins and thereby become a littering problem. In this study, littering has not been modelled in the LCA, as no standard method exists to perform such modelling, and the justification for this is explained in section 3.1.4 (Approach and Methodology chapter).

Collection and sorting and technology efficiencies for recycling LDPE and bio-based LDPE. The collection effect is set to 30%, just as the other plastic types. It is possible that collection and recycling of carrier bags is lower than the other packaging products mentioned in this report, because a quantity are used as waste bin liners. The sorting and technology efficiency is based on literature values for LDPE plastic. This parameter is assumed to react sensitively in the cradle-to-grave results. This assumption is drawn from sensitivity analyses in other case studies.

Substitution of LDPE should also be considered uncertain, as post-consumer soft plastics can be troublesome to sort at processing plants. LDPE is more likely to end up in a mixed polymer fraction as a lower quality secondary raw material.

The level of biodegradation in landfills is uncertain and it will depend on landfill conditions such as temperature, landfill waste composition and bioplastic type.

Apart from the above case study's specific uncertainties, several other general uncertainties that apply to most of the presented case studies are discussed in the Approach and methodology in chapter 3.5., e.g., data quality, critical assumptions and limitations in End of Life modelling. These include uncertainties related to

- the estimated EoL mix
- the marginal energy technologies utilised
- the development of the EoL technologies (efficiency increases)
- the consumption of materials and energy at the EoL treatment plants
- the amount of rejects in biological treatment
- the chemical composition of the biodegradable plastics.

Assumptions related to the exclusion of the reuse of carrier bags, sorting and technology efficiency of LDPE, substitution rate of LDPE and the biodegradation of starch in landfills present significant uncertainties. Further research on littering, as well as biodegradation rates of biodegradable plastics in non-intended EoL technologies is required.

5 Assessing non-PEF impact categories: taking into account ILUC and endpoint of GTP

5.1 Baseline with starch plastics

5.1.1 Global Temperature Change Potential

The GTP impacts of bio-based carrier bags broken down into the main unit processes described in the section 2.1 inventory analysis is shown in Figure 208. When the biogenic carbon removal is taken into account, the cradle-to-gate GTP 100a is 0.04 kg CO_2 eq./FU. Observations for GTP are very similar to those for the climate change impacts described in section 3.1.1.

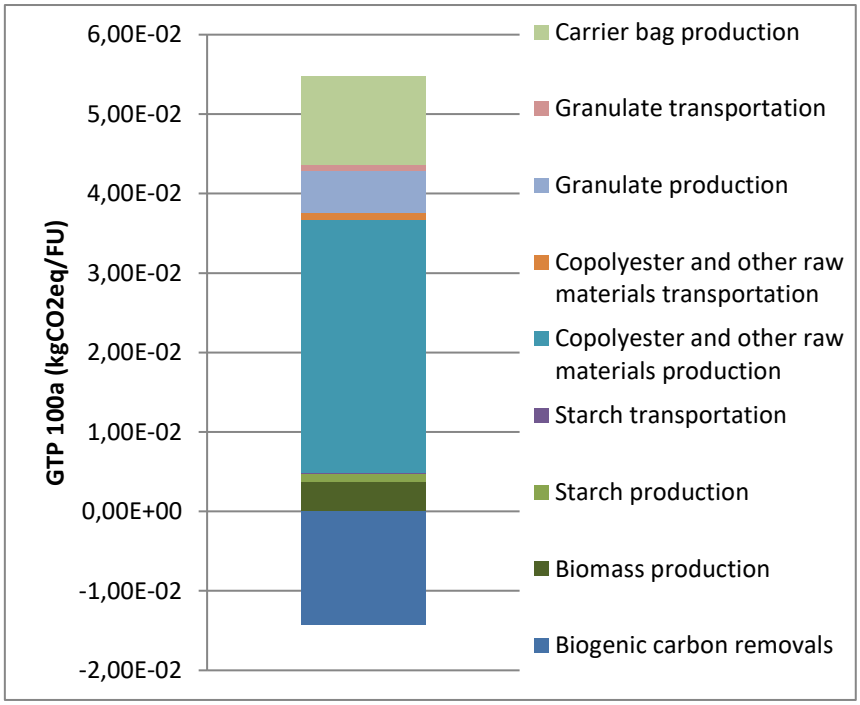


Figure 208. Breakdown of the cradle-to-gate GTP 100a results for bio-based carrier bags (baseline), 1 functional unit, excluding DLUC and ILUC effects.

5.1.2 Land Use Change emissions

Table 223 presents the cradle-to-grave characterised results broken down into feedstock (for potato used for starch, maize used for starch and maize used for PLA; with biogenic carbon stored separately), manufacturing, EoL and ILUC (for potato used for starch; maize used for starch and maize used for PLA). As highlighted in Chapter 4, DLUC and ILUC cannot be summed up (it is whether DLUC or ILUC). This case does not involve crops that imply DLUC (namely Brazilian sugarcane, German maize, French wheat and German wheat), based on the methodology from the PEFCR Guidance V6.3 presented in Chapter 4. Figure 209 presents the characterised results for case study 7 with the inclusion of ILUC.

Table 223. Characterised cradle-to-grave results broken down per process, including ILUC. Case study 7, baseline.

Impact category	Biogenic carbon removals	iLUC (potato for starch)	iLUC (maize IT for starch)	iLUC (maize for PLA)	Potato production (for starch)	Maize IT production (for starch)	Maize production (for PLA, US)	Manufacturing	EoL
Climate change (kg CO_2 e)	-1.43E-02	4.70E-04	3.41E-03	2.06E-03	3.37E-04	3.55E-03	8.55E-04	5.42E-02	7.16E-03
Ozone depletion (kg CFC-11 e)		9.83E-14	7.10E-13	4.28E-13	4.21E-12	2.99E-10	3.07E-14	1.42E-08	-2.48E-10
Human toxicity, non-cancer effects (CTUh)		2.14E-13	1.55E-12	3.44E-13	2.46E-09	6.09E-10	2.22E-11	9.22E-09	7.59E-10
Human toxicity, cancer effects (CTUh)		2.03E-14	1.47E-13	5.50E-14	2.00E-11	2.10E-10	9.92E-12	2.80E-09	-6.47E-11
Particulate matter (kg PM2.5 e)		1.24E-08	8.96E-08	5.43E-08	2.27E-07	2.27E-06	3.11E-07	2.52E-05	-5.37E-07
Ionizing radiation HH (kBq U235 e)		-8.71E-09	-6.21E-08	2.96E-08	5.11E-06	9.68E-05	8.80E-05	1.17E-02	3.27E-04
Photochemical ozone formation (kg NMVOC e)		1.46E-06	1.06E-05	6.40E-06	2.68E-07	1.60E-05	1.69E-05	1.48E-04	9.53E-07
Acidification (molc H+ e)		2.93E-07	2.12E-06	1.29E-06	9.92E-06	3.19E-05	1.28E-05	2.95E-04	5.59E-06
Terrestrial eutrophication (molc N e)		1.64E-06	1.19E-05	7.20E-06	4.36E-05	1.25E-04	6.94E-05	3.93E-04	1.40E-05
Freshwater eutrophication (kg P e)		1.08E-09	7.82E-09	4.76E-09	2.42E-07	9.55E-07	4.32E-08	2.02E-05	8.21E-08
Marine eutrophication (kg N e)		1.75E-07	1.26E-06	7.66E-07	9.17E-06	6.43E-05	7.29E-06	3.43E-05	1.73E-06
Freshwater ecotoxicity (CTUe)		3.62E-06	2.62E-05	3.82E-06	5.50E-03	1.81E-02	1.63E-02	2.83E-01	-9.89E-04
Land use (kg C deficit)		1.35E-06	9.78E-06	5.92E-06	1.22E-02	8.50E-02	5.66E-02	5.43E-02	0.00E+00
Water use (m^3)		1.45E-05	1.05E-04	6.34E-05	7.12E-05	2.71E-02	2.48E-03	1.06E-02	-1.22E-03

Abiotic depletion (kg Sb e)		4.28E-12	3.10E-11	1.88E-11	3.02E-11	2.82E-08	1.45E-09	2.99E-08	-2.00E-09
Abiotic depletion (fossil fuels) (MJ)		1.22E-05	8.84E-05	5.35E-05	1.65E-03	3.30E-02	1.00E-02	1.18E+00	-4.48E-02
NREU (MJ)		1.17E-05	8.46E-05	5.12E-05	1.80E-03	3.46E-02	1.05E-02	1.36E+00	-5.12E-02

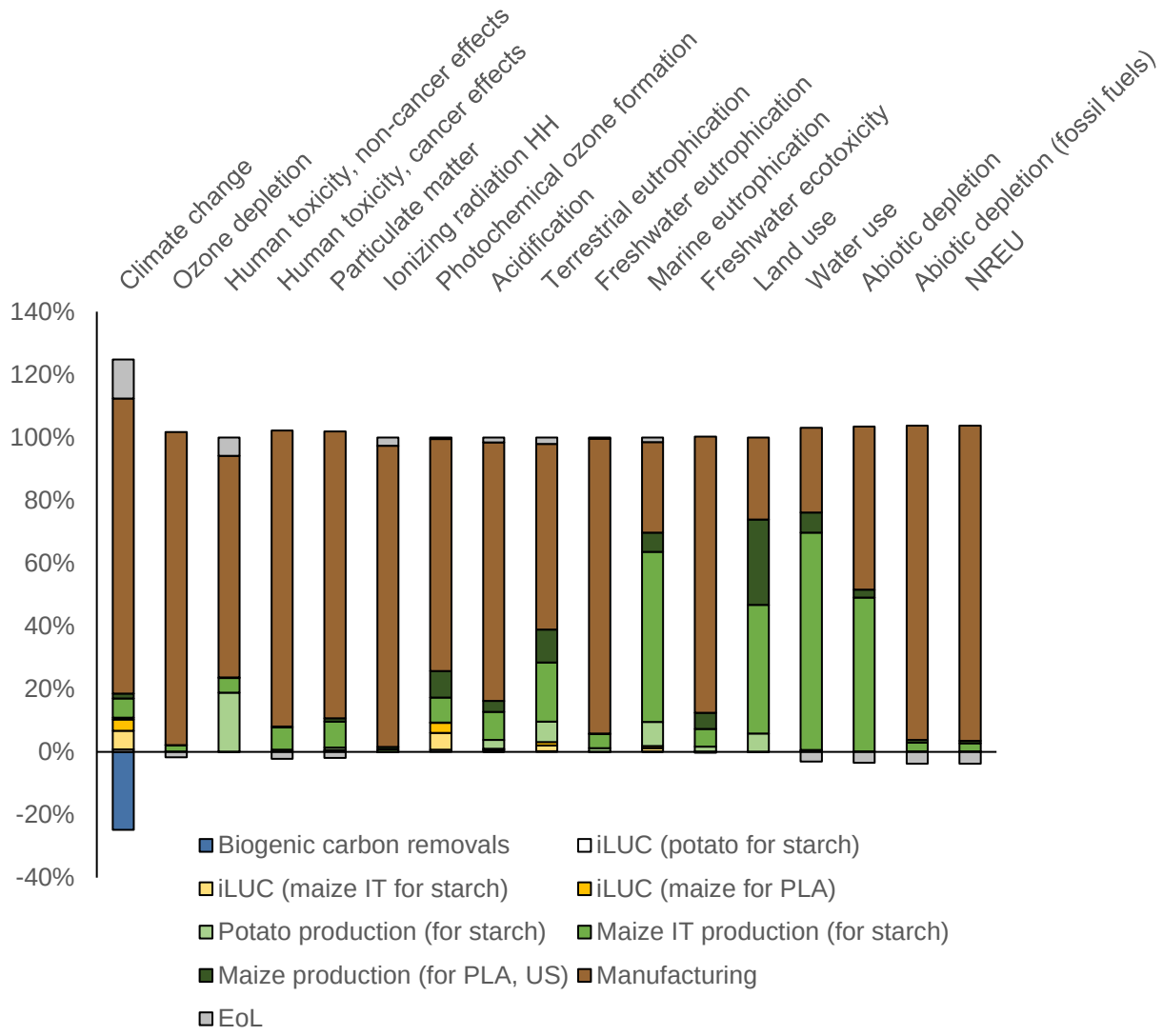


Figure 209. Relative characterised results broken down for all impact categories and including ILUC – case study 7, baseline.

As can be seen from Table 223 and to some extent from Figure 209, ILUC does contribute to all impact categories considered herein, as in previous cases. However, as for previous cases, the contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (10 % of the impact summing all 3 ILUC processes), photochemical ozone formation (9 % of the impact), terrestrial euthrophication (2.8 % of the impact) and marine eutrophication (1.8 % of the impact). All of these impacts are dominated by land expansion (Figure 15). In the case of climate change, as explained earlier, the ILUC impact is essentially due to the CO_2 releases resulting from land clearing and in the case of photochemical ozone formation, to carbon monoxide also released during land clearing (burning the cleared biomass; Figure 17). Acidification, terrestrial- and marine euthrophication are essentially due to nitrogen oxides released during land clearing (more than 80 % of the impact; Figure 17).

The contribution of ILUC to climate change has a magnitude slightly higher than the impact from producing the crops themselves (Figure 209; crop cultivation represents 8 % of the

climate change impact when ILUC is included). As in case study 2 and 5, ILUC is here shown in white to yellowish tones for all crops contributing to it (for the climate change impact, ILUC potato starch represents 0.8 %, ILUC maize starch 5.9 % and ILUC maize PLA 3.6 %). Similarly all crop cultivation processes are shown separately (greenish tones).

Although potato has a lower yield than maize, its ILUC impact is lower than for maize as less is used per functional unit. In absolute terms, the ILUC impact for climate change (summing all 3 ILUC) here is rather low compared to all other cases. This reflects the importance of having conversion processes requiring as little biomass as possible, so the climate change impacts from both the crop cultivation and the ILUC can be lower.

5.2 Alternative with bio-based LDPE

5.2.1 Global Temperature Change Potential

The GTP impacts of bio-based LDPE carrier bag, broken down into the main unit processes described in the section 2.1 inventory analysis, are shown in Figure 210. When the biogenic carbon removal is taken into account, the cradle-to-gate GTP 100a is -0.007 kg CO_2 eq./FU. Observations for GTP are very similar to those of climate change impacts described in section 3.1.2.

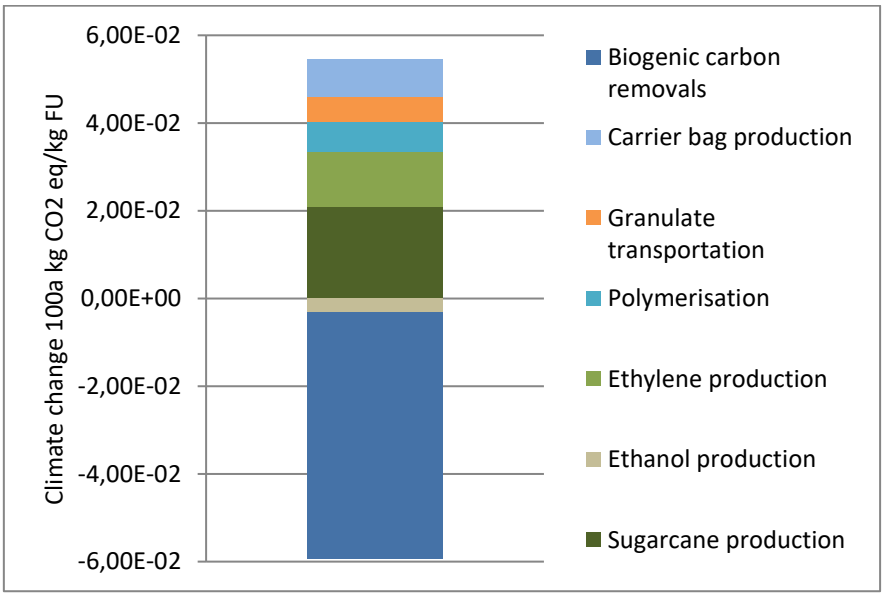


Figure 210. Breakdown of the cradle-to-gate GTP 100a results for bio-based carrier bags (baseline), 1 functional unit, excluding DLUC and ILUC effects.

5.2.2 Land Use Change Emissions

Table 224 presents the cradle-to-grave characterised results broken down into feedstock (with biogenic carbon stored separately), manufacturing, EoL, DLUC and ILUC. As highlighted in Chapter 4, DLUC and ILUC cannot be summed up. Figure 211 presents the characterised results for this variant of case study 7 with the inclusion of ILUC.

DLUC is also presented as it is a "process" that should be considered in the LCA, according to the latest PEFCR Guidance v6.3 (European Commission, 2018). Results with the inclusion of DLUC are shown here (Figure 212), as visible (contributing 27 % of the climate change impact). As explained in Chapter 4, DLUC is based on the 20-year history of crops in a given country. For the feedstock considered in this study, DLUC is only observed for a few crops,

namely Brazilian sugarcane (used here for the bio-based LDPE), German maize, French wheat and German wheat (the latter three are not used in this case). The reason why DLUC is so significant here is reflected in Table 25 of Chapter 4 (a high nature conversion area is considered for Brazilian sugarcane, that conversion is from forestland and it causes high GHG emission factors). As opposed to e.g. case 1, it reflects the fact that an important amount of (sugarcane) feedstock is required per functional unit.

Table 224. Characterised cradle-to-grave results broken down per process, including ILUC. Case study 7, alternative with bio-based LDPE.

Impact category	Unit	Biogenic carbon removals	dLUC	iLUC	Sugarcane production	Manufacturing	EoL
Climate change	kg CO_2 eq.	-3.14E+00	2.44E-01	1.44E+00	1.16E+00	1.71E+00	9.27E-01
Ozone depletion	kg CFC-11 eq			3.01E-10	8.61E-08	1.01E-06	-1.33E-10
Human toxicity, non-cancer effects	CTUh			6.55E-10	1.63E-06	9.35E-07	1.52E-08
Human toxicity, cancer effects	CTUh			6.22E-11	4.44E-08	2.54E-07	6.27E-09
Particulate matter	kg PM2.5 eq			3.80E-05	3.80E-02	1.65E-03	-1.01E-04
Ionizing radiation, human health	kBq U235 eq			-2.62E-05	1.49E-02	8.59E-01	3.56E-02
Photochemical ozone formation	kg NMVOC eq			4.47E-03	1.39E-02	1.23E-02	-4.28E-04
Acidification	molc H+ eq			8.97E-04	1.75E-02	2.23E-02	3.24E-04
Terrestrial eutrophication	molc N eq			5.04E-03	8.25E-02	4.61E-02	1.42E-03
Freshwater eutrophication	kg P eq			3.32E-06	1.15E-04	1.54E-03	3.05E-05
Marine eutrophication	kg N eq			5.35E-04	4.32E-03	5.24E-03	1.14E-04
Freshwater ecotoxicity	CTUe			1.11E-02	7.58E+00	2.19E+01	1.93E-01
Land use	kg C deficit			4.14E-03	4.12E+01	1.11E+01	0.00E+00
Water use	m3			4.43E-02	1.61E-01	2.14E+00	-9.40E-02
Abiotic depletion	kg Sb eq			1.31E-08	2.80E-06	3.15E-06	-1.87E-07
Abiotic depletion (fossil fuels)	MJ			3.74E-02	7.05E+00	6.70E+01	-1.42E+01
NREU	MJ			-1.63E-03	7.20E+00	8.28E+01	-1.51E+01

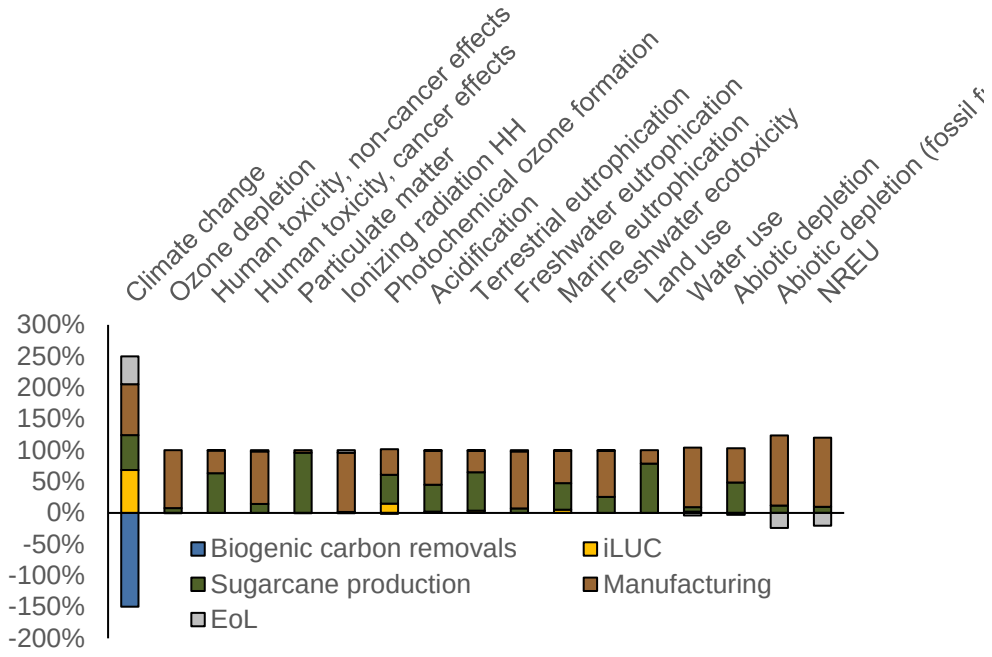


Figure 211. Relative characterised results broken down for all impact categories and including ILUC – case study 7, alternative with bio-based LDPE.

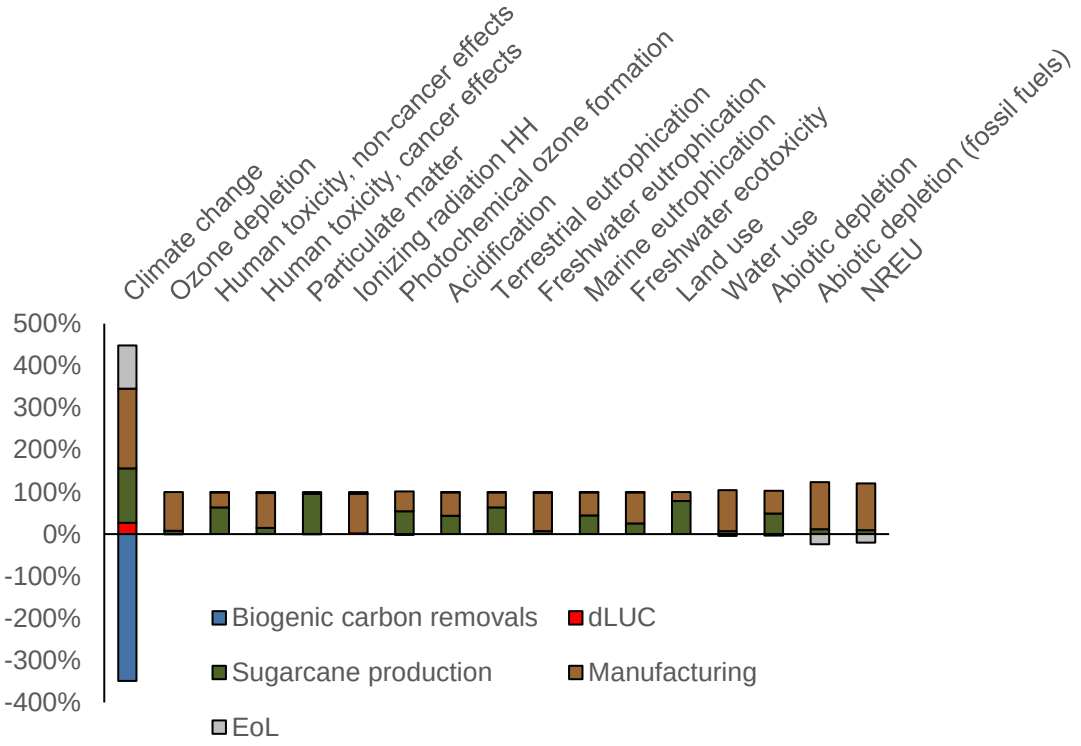


Figure 212. Relative characterised results broken down for all impact categories and including DLUC – case study 7, alternative with bio-based LDPE.

As can be seen from Table 224 and to some extent from Figure 211, ILUC does contribute to all impact categories considered herein, as in previous cases. However, as for previous cases, the contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (69 % of the impact), photochemical ozone formation (15 % of the impact), terrestrial eutrophication (3.7 % of the impact), marine eutrophication (5.2 % of the impact), acidification (2.2 % of the impact) and water use (2 % of the impact). All these impacts are dominated by land expansion, except for water use which is fully dominated by intensification (Figure 15). In the case of climate change, as earlier explained, the ILUC impact is essentially due to the CO_2 releases resulting from land clearing and in the case of photochemical ozone formation, to carbon monoxide also released during land clearing (burning the cleared biomass; Figure 17). Acidification, terrestrial- and marine eutrophication are essentially due to nitrogen oxides released during land clearing (more than 80 % of the impact; Figure 17). The water use impact, on the other hand, essentially reflects the hydropower electricity demanded for the (additional) fertiliser production (intensification process), in particular phosphorus (diammonium phosphate).

The contribution of ILUC to climate change has a magnitude slightly higher than the impact from producing the sugarcane (Figure 211; crop cultivation represents 55 % of the climate change impact when ILUC is included). In absolute terms, the ILUC impact for climate change here is quite high compared to the one obtained for e.g. case 1 but is still lower than the one of other cases (e.g. case study 2 or case study 5). This reflects the importance of having conversion processes requiring as little biomass as possible, so the climate change impacts from both the crop cultivation and the ILUC can be lower. Nevertheless, it should be highlighted that it is many orders of magnitude higher than the ILUC of the baseline case (1.44 kg CO_2 eq. per functional unit; it is 0.0059 for the baseline).

6 Conclusions, limitations and recommendations

The goal of this case study was to assess the environmental profile of bio-based carrier bag production and compare it with the fossil-based counterpart. Although the LDPE carrier bags considered in this case study are mostly produced in Europe, other types of petrochemical carrier bags (HDPE and PP) that find wide application, are produced in China and transported to Europe. Due to the difference in the production systems in China and the requirement of transportation to Europe, the impacts of petrochemical carrier bags can be higher.

For starch-based bio-based carrier bags, two EoL scenarios were compared. Based on the cradle-to-grave comparison with the petrochemical reference of LDPE bags, starch plastic bags did not provide clear environmental benefits in any of the selected impact categories. The bio-based LDPE carrier bags offer better performance in climate change, abiotic depletion (fossil fuels) and NREU categories. However, bio-based LDPE performs significantly worse in the particulate matter, acidification, terrestrial and marine eutrophication categories. Thus, it is concluded that based on these eight impact categories, there is no clear better alternative.

It should be noted, however, that the environmental impacts of sugarcane production could be reduced – e.g. burning of sugarcane is already phased out in Brazil, and thus the particulate matter impacts are likely to be reduced in the near future. On the other hand, the climate change and abiotic depletion effects of fossil carrier bags are arguably likely to remain the same. Also, from the ILUC analysis, it appears that the climate change impact could increase by about 8 %, and so the main benefit in this category arises when taking possible indirect impacts into account.

ILUC does contribute to all impact categories considered herein, as in previous cases. However, as for previous cases, the contribution of ILUC is rather negligible (contributing to <1 % of the total impact) for most impact categories except for climate change (10 - 69 % of the impact), photochemical ozone formation (9 - 15 % of the impact), terrestrial eutrophication (<4 % of the impact), marine eutrophication (2 - 5 % of the impact), acidification (<2 % of the impact) and water use (<2 % of the impact). All these impacts are dominated by land expansion, except for water use which is fully dominated by intensification. As opposed to all other cases, DLUC has a significant contribution in the case of the LDPE variant with EU crops where it contributes to 27% of the climate change impact.

With regard to EoL, choosing the best performing EoL technology for bio-based starch carrier bag depends highly on which impact categories are found to be most important, as often specific EoL categories perform best in some impact categories, and worst in others. This raises questions on for example the general prioritization of recycling over incineration with energy recovery or landfilling for all plastics. For the weighted results, industrial composting is the best-performing technology. For the starch-based bags, changing from the current mix of EoL options to industrial composting would reduce the EoL contribution to the total weighted impacts from about 9-12 % to a mere 1-2 %. This depends of course to some extent on specific assumptions, e.g. the source of electricity used/replaced, or whether methane emissions from landfilled biodegradable plastics are captured ad utilised or not, and these conditions may change over time.

Based on the weighted LCIA results of the EoL mix for each product system, the conclusion is that bio-based or petrochemical LDPE carrier bags perfom better looking at their EoL mix. The preferred EoL technology for the bio-based starch carrier bags based on weighted results was industrial composting, followed by anaerobic digestion. Landfilling of the bio-based starch carrier bag showed the highest impacts, stressing the importance of keeping biodegradable waste from landfills.

This sensitivity analysis would change the cradle-to-grave climate change results for EoL mix of the starch-based carrier bag by +0.2 % and –0.2 % (looking at the seven selected impact

categories) for respectively a lower and a higher carbon degradation at the industrial composting facility (varying from 63 % to 97 %).

The added value of biodegradable/compostable plastics in the EoL scenario has not been taken on board in the LCA methodology. Using biodegradable bags as waste bags for organic material has been shown to increase the purity of organic recycling. This is however not included in the scope of this case study. Moreover, littering and specifically marine littering are not recognised environmental indicators, which are particulary relevant to note with regards to landfilling. Another critical aspect concerning carrier bags is the interpretation of the primary re-use of carrier bags, which was not included in the EoL mix scenarios: increasing the number of re-uses would reduce all impacts and improve the performance of the bags.

Littering is excluded from this case study due to lack of data and knowledge of the impact assessment models. The exact amount (and relative share) of plastics that are littered is uncertain for a specific application and will differ widely between countries. The impacts of littering and microplastics are currently not properly captured in the toxicity-related impact categories (ecotoxicity and human toxicity). As pointed out by Pant et al. (2018) fate, exposure, and effect modelling for macro- and micro-plastics (incl. additives) is still in its infancy. Physical impacts (e.g. entanglement; ingestion of larger plastic particles and their effects) and chemical impacts (e.g. due to microplastic formation, microplastics as carrier of other chemicals, unknown impact of additives) and biological impacts (e.g. microplastics as carrier of germs/alien species) are largely unknown and currently not included in LCA. Thus, the impacts of littering (e.g. leaving plastics in the soil or in a marine environment) are most likely underestimated in the current LCA for the non-degradable applications.

CONCLUSIONS, LIMITATIONS AND RECOMMENDATIONS

1 Key findings of the seven LCA case studies

1.1 Findings based on cradle-to-grave LCA results for the Bio-Based products

In this study, seven LCA case studies covered three major commercialised bio-based polymers, namely bio-based PET (polyethylene terephthalate, case study "Beverage Bottles"), PLA (polylactic acid, case studies "Single-use cups for cold beverages", "Single-use Cutlery" and "Food Packaging films") and starch plastics (case studies "Agricultural Clips", "Agricultural Mulch films" and "Single-use Carrier bags").

Based on the normalised and weighted results of the seven cradle-to-grave LCA case studies for 16 impact categories, without taking the effects of direct and indirect land use changes,[79] we found that for both the bio-based products and the petrochemical references, the most important environmental impact categories identified are climate change, abiotic depletion (fossil fuels)[80] and human toxicity (cancer effects). Together, these three impact categories account for approximately 30-60 % of the total cradle-to-grave environmental impacts of the bio-based products These three impact categories are highly associated with (predominantly cumulative fossil) energy use in the cradle-to-user phase, and the direct emissions from the EoL phase.

Figure 213 shows the overview of the breakdown of the weighted LCA results for the cradle-to-grave environmental impacts. The weighting was carried out based on the weighting factors recommended by the PEFCR guidance version 6.3 including all 16 impact categories (see Chapter 5 Interpretation). The total impacts are broken down into major life cycle stages consisting of:

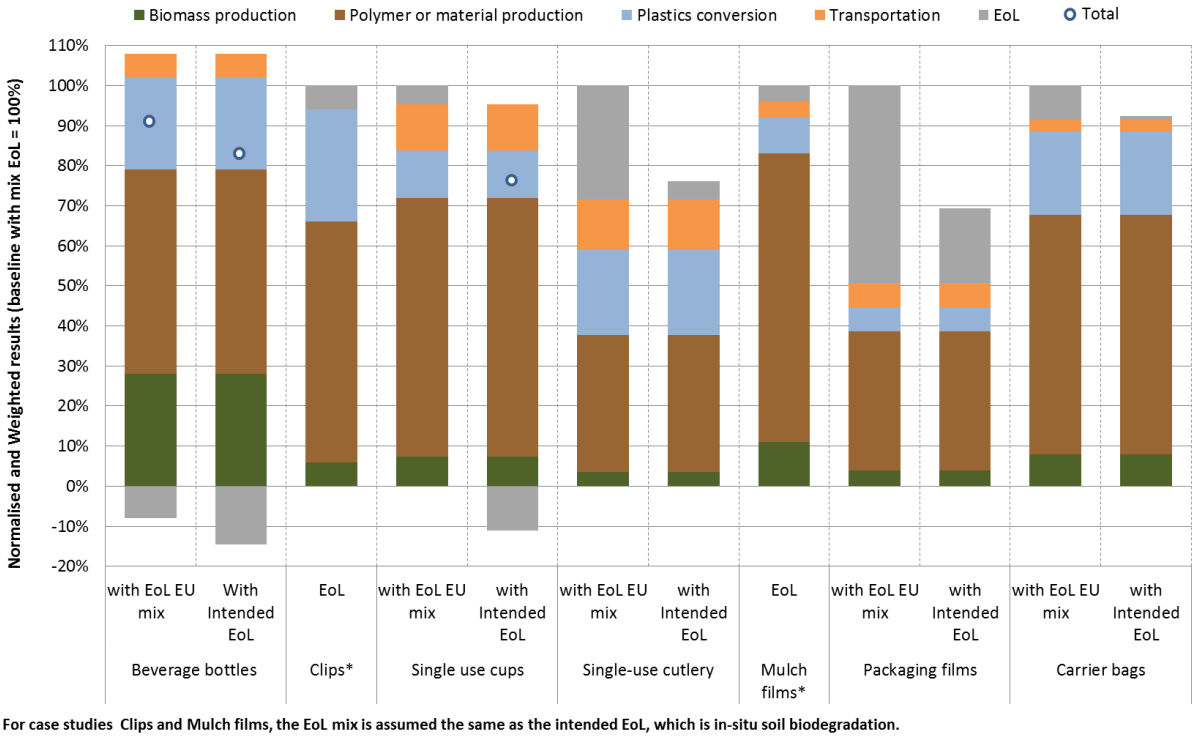
- biomass *production*, including direct and indirect emissions from cultivation and harvesting as well as the net biogenic carbon removals,
- *polymer or material production*, including the conversion from biomass to bio-based polymers and often copolymerised and/or chemical/physically blended with fossil fuel-based polymers,
- *plastics conversion*, including the conversion of bio-based polymer or materials into the final *products*, e.g. by using injection moulding to produce cutlery, or by using thermoforming to produce single-use cups,
- *transportation*, taking into account transportation services required for the main material/carbon flows[81] including shipping all the intermediate products as well as distributing the final products to the end users, and

[79] The additional land use changes share to the overall cradle-to-grave impacts is relatively marginal: on average 14% for GWP_{100}, 10% for photochemical ozone formation, and negligible shares (ca. 0.01-2.4%) for all other impact categories. See conclusions drawn in Section 1.2 of this chapter,

[80] In the latest PEFCR guidance (v6.3) this impact category is re-named to "Resources (fossil fuels)".

[81] By "main material/carbon flows", we refer to the transportation associated to the end products as well as all the intermediate products. The transportation of chemical, energy and all other utilities used in the production process are included in the background data. Those are not separately listed here. The transportation required in the waste management is included in the EoL models.

- *EoL* waste management phase, where the results shown in Figure 213**Error! Reference source not found.** also provide the comparisons between the case where the EU mixed EoL technologies are analysed to reflect the status-quo, and the intended EoL technology representing an ideal scenario (see Section 3.1 of the *Approach and Methodology* chapter). The intended EoL scenario is also in line with the European legislation on waste management and the lower impacts shows compliance with the waste hierarchy.



* For case studies Clips and Mulch films, the EoL mix is assumed the same as the intended EoL, which is in-situ soil biodegradation.

Figure 213. Cradle-to-grave environmental impacts of seven bio-based products, weighted results based on 16 PEFCR impact categories (default weighting factors from PEFCR guidance version 6.3, including toxicity categories), broken down into the major life cycle stages. The results based on EoL EU mix are then compared with the results based on intended EoL scenarios. LCA results here are without the effect of direct or indirect land use changes.

Biomass production. In principle, biomass feedstock production for bio-based products typically requires (more) land than the extraction and use of fossil fuels, and thus land use and associated impacts are often specifically investigated. In this study, we found that the impacts from land use and water use are rather limited compared to the impacts from the manufacturing phase. The biomass production phase in most cases accounts for less than 10 % of the overall cradle-to-grave environmental impacts. The exception is in the case study beverage bottles made from bio-based PET, where the high contribution from biomass production (ca. 28 % of total cradle-to-grave impacts) is caused by particulate matter emissions in the sugarcane harvesting in Brazil. It should be noted while this study was conducted in 2017-2018, Brazil is in the meantime phasing out the burning of cane trash by legislation. By 2031, the burning of cane trash should be fully phased out in the country (São Paulo State Law n. 11241/02) (Carvalho, 2016). Thus, the impact of particulate matter is expected to be substantially reduced in the near future for sugarcane ethanol from Brazil.

The manufacturing phase has the highest contribution of all life cycle stage: it accounts for approximately 50 % of the cradle-to-grave impacts of nearly all seven bio-based products studied. The impacts of the manufacturing phase consist of the impacts from three sub-stages:

the production of bio-based polymers (i.e. converting biomass into polymers), the production of (often) fossil fuel-based copolymers, and the plastic conversion step when the final products are formed. The first two sub-stages together are shown as "polymer or material production" Figure 213. The contributions from these three sub-stages vary depending on the type of polymers or materials studied:

- For **bio-based PET** (30 % bio-based content), the manufacturing phase accounts for nearly 75 % of the total impacts, with the highest contributor of PET polymer production, including bio-MEG, PTA and polymerisation. These three processes contribute to over 50 % of the total cradle-to-grave impacts, followed by the plastics conversion step (bottle blown moulding), contributing to 23 % of the total impacts.

- For **PLA** (100 % bio-based), the conversion from sugarcane or maize into polymer PLA has the dominant impact in the life cycles of the products studied, accounting for 40-60 % of the total cradle-to-grave impacts based the results of three PLA case studies (i.e. Single-use cups, Single-use cutlery and Packaging films). Process energy (electricity and heat) and chemical used in the lactic acid and PLA production are responsible for the major parts of the total impacts. The energy consumption is highly associated with the process water and its removal that needs to be dealt with in a bio-chemical conversion (e.g. fermentation of lactic acid). In the studied cases, a lion's share of the process energy (especially heat supply) is fossil fuels based, which lead to high impacts of climate change and abiotic depletion (fossil fuels). The second most important contributor in the manufacturing phase of PLA is the plastics conversion steps (e.g. thermal forming of cups, injection moulding of cutlery and film extrusion). Electricity is the key input of these conversion processes. Depending on the fuel mix of the electricity where the conversion step is located, the step could contribute to about 10 % of the total cradle-to-grave impacts (e.g. single-use cups thermoformed in Europe) or up to 25 % of the total impacts (e.g. cutlery moulded in China where the electricity is more carbon-intensive than the EU-average).

- **Starch plastics** are thermoplastics starch blended with biodegradable polymer(s). The former ingredient is bio-based, the latter is often fossil-fuel based. The bio-based content varies considerably depending on the formulation of the starch plastics for specific application. For the three starch plastics applications studied, the bio-based contents range from 33 % to 68 %. The environmental impacts of studied starch plastics are dominated by the manufacturing phase (ca. 80 % of the total cradle-to-grave impact), of which the impacts from the fossil fuel-based co-copolymers are dominant (ca. 50-60 % of the total cradle-to-grave impacts) and the impacts from plastics conversions (e.g. injection moulding, film extrusion) account for 20-30 % of the total impacts.

The bio-based industry is still in its infancy stage. There are on-going efforts to optimise manufacturing process. The efforts would likely lead to a lower environmental impact for the future bio-based products than today. The potential (decrease in) environmental impacts of bio-based plastics due to technological progress and the exogenously changing energy system should be assessed in the future. The impacts from *transportation* are in general insignificant. The exceptions are when the global supply chain is very long. For instance, based on the current supply chain of biomass, polymer and end-products, the bio-based carbon in the PLA cutlery travels nearly three quarters of the globe to come to the demand centre – Europe. In this case, the impact of transportation is no longer negligible: about 13 % of total impacts of the PLA cutlery are contributed by the transportation and distribution of the product. However, there are also trade-offs with benefits of imported feedstocks (see below).

From cradle to user, the energy requirements along the chain, from process heat and electricity to transportation fuels, play an important role. Energy consumption directly leads to

the impact of abiotic depletion (fossil fuels) and climate change due to CO_2 emission from the combustion of fossil fuels (for heat and electricity). Combusting fossil fuel leads to the impact of photochemical ozone formation (caused by nitrogen oxides emitted during fuel combustion) and particulate matter (dust emitted from coal combustion). .

The impacts of the *EoL* of the bio-based products studied are highly dependent on the applications. Except for PET bottles, the other six bio-based baseline cases are all for biodegradable applications for which the materials are the newcomers for the existing waste management system. It is not possible for this study to establish a so-called "status-quo" waste management for these newcomers, because the quantity of the bio-based products in the waste stream is still statistically insignificant. Therefore, an EoL mix is assumed for each case study to estimate how these bio-based materials would possibly end up in our waste management system (see also the methodology chapter of EoL in the beginning of this report, chapter Approach and methodology). The individual EoL technologies is modelled as average European e.g. meaning that incineration includes both with and without energy recovery and anaerobic digestion includes all combinations of dry/wet, thermophilic/mesophilic and with/without post maturation. For the six case studies where biodegradable materials are involved, the uncertainty of EoL assessment is considered high.

- For PET bottles, since it is a currently highly recycled material (more than 60 % collected for recycling in 2016 in Europe), the EoL waste treatment of PET bottles receives a net credit from an environmental point of view (see Figure 212). Based on the weighted LCA results for all 16 impact categories, the most favourable EoL technology of PET is recycling. By moving from 60 % collection rate for recycling of today to 100 % collection rate for recycling, the overall impacts of the bio-based PET bottle could be reduced by 7 %. Based on the comparisons of weighted results of different EoL options for all 16 impact categories, recycling is identified as the most preferred EoL option for PET bottles, followed by landfilling and incineration.

- For single-use cutleries and food packaging films, where the reference flow includes food leftovers, the mixed EoL have a significant or even dominant contribution: this phase accounts for 30 % and 50 % of the cradle-to-grave environmental impacts for the two products respectively. The results are highly sensitive to two factors: 1) the assumed mix of waste management technologies and 2) the assumed food residues, which go together with the bio-based products.

    - From Figure 212 it can be seen that if the "intended" EoL (industrial composting) is implemented, the overall impacts of PLA cutlery and food packaging films could decrease by 25 % and 30 % respectively, especially due to the avoidance of methane emissions from landfilling.
    - Based on the weighted LCA results for all 16 impact categories, the most favourable EoL technology of both PLA cutlery and food packaging films is industrial composting, followed by anaerobic digestion, incineration and landfilling.

- For PLA cups, since less organic residue/contamination is assumed (compared to the PLA cutlery and packaging films), the EoL EU mix has a limited impact contribution (ca. 5 %) of the cradle-to-grave results. When switching to EoL "intended" (50 % industrial composting and 50 % recycling), the cradle-to-grave impacts of PLA cups would reduce by 25 %. The most favourable EoL for PLA cups is recycling, followed by industrial composting, anaerobic digestion, incineration and landfilling.

- For carrier bags made from starch plastics, the EoL mix contribute with 9 % of the weighted cradle-to-grave results. Introducing industrial composting as the "intended" EoL technology reduces to contribute to 1 % of the weighted cradle-to-grave results.

Industrial composting is the most preferred EoL option, followed by anaerobic digestion, incineration and landfilling.

- For starch plastics used for in-situ soil biodegradation in the case studies clips and mulch films, the impacts of EoL is not substantial (less than 10 % of the total weighted cradle-to-grave impacts). However, the results are highly sensitive to the assumed amount of soil collected together with the plastics.

The weighted results (with toxicity) of the individual EoL option (100 % scenarios) for each product system does not always comply with the waste hierarchy (the waste framework directive). The fossil and bio-based non-biodegradable plastics have low impacts from landfilling. When expanding the reference flow to include food leftovers landfilling show the highest impacts and the hierarchy is the valid here (excluding mechanical recycling). Regarding the bio-based and biodegradable plastics the results neither show complete compliance with the waste hierarchy. For the agricultural applications landfilling is preferred over incineration. The other biodegradable plastics comply with the waste hierarchy, except that for some case studies industrial composting is preferred over anaerobic digestion (however, uncertainties apply here, and the results are within a small range and hence seen as comparable). An exception is that recycling of PLA with food leftovers which does not perform well, which is due to the assumption that the food leftover is sent for incineration and landfilling. A sensitivity analysis sending the food leftovers for industrial composting, shows that recycling would be similarly attractive as industrial composting and anaerobic digestion.

In this study, for some cases *alternative feedstock* of biomass were investigated. In the case studies of PET bottles and PLA (for cases cups, cutleries and packaging films), using average EU maize do not necessarily offer environmental benefits. This is observed in comparing three fictional cases with the baseline (status-quo): PET bottle using ethanol made from average EU maize, wheat and sugar beet, and PLA cups, cutleries and packaging films made from average EU-maize. All cases show that by using EU average biomass the impacts from the logistics services are reduced. However, the saved impacts do not offer sufficient environmental benefits to overcome the more carbon-intensive average EU crop production compared to the status-quo market choice of imported biogenic carbon. The main reason is the lower yield of *average* European crops used in our fictional cases, e.g. maize (average yield of 7.2 ton ha^{-1} yr^{-1}) [82], compared to crops sourced from a high yield areas in the US, e.g. maize cultivated in Nebraska/Iowa has an yield of 10.3 ton ha^{-1} yr^{-1}), or a high yield crop, e.g. sugarcane cultivated from Thailand has a typical yield of 60-70 ton ha^{-1} yr^{-1}. Low yields lead to more extensive use of land and consequently more resources and utilities such as water, fertiliser, pesticides as well as direct energy requirements and direct emissions. All these factors eventually lead to a higher environmental impact of average EU maize. For example, in the case of single-use cups made from PLA. The weighted impact of average EU maize is about three times higher than that of the baseline biomass (i.e. the mix of the US-Nebraska maize and Thai sugarcane, comparison on per functional unit basis). The most affected impact categories are ozone layer depletion, water use and human toxicity. It should be noted there is a huge variation of the maize yield within the EU. Of the top eight maize production countries, the yield could be as high as 9-11 ton ha^{-1} yr^{-1} (e.g. in France, Germany, Italy and Spain), or as low as 5-6 ton ha^{-1} yr^{-1} (e.g. in Hungary and Bulgaria). For PLA, choosing a high yield biomass feedstock is critical factor to compete with the current market PLA products where the biogenic carbon is sourced outside of EU.

[82] Average yield of eight top maize production countries in Europe, accounting for 85% of total EU maize production, see sections 2.1.2 and 3.1.2 of the case study "Single-use cups".

A crucial point however, is that the high yield of crops in Brazil and the US are a result of large-scale monoculture depending on GMO breeds with other adverse impacts which are not included in the LCA for methodological and data availability reasons. However, the possible impact on biodiversity of different (local) agricultural practices in the EU and other countries was not specifically investigated in this study, part from an ecoregion level test for one case and one crop (maize). The test, using Chaudhary (2015) shows higher risk of biodiversity loss from US Maize as compared to Italian Maize, but the finding cannot be extrapolated to other crops or cases. Therefore, despite the studies investigations relying on best available methods, it is not complete, and results must be interpreted with care: It is thus not possible to draw conclusions about the possible effects of extensive growing of monocultures as drawback of high yields.

In two case studies, the effects of using alternative feedstocks such as *a waste stream instead of a cultivated crop* (as in the baseline) were investigated. These are starch plastics made from potato peels used to make horticultural clips and bio-based polypropylene (PP) cups made from used cooking oils (UCO). It is observed that by using waste as a resource is not necessary a guarantee for a lower environmental impact of the studied bio-based products; it strongly depends on the type of waste and efficiency of the conversion technologies.

- In the case of potato peel-based starch clips, it is unveiled that although the impact of biomass production is substantially reduced, the overall impacts of the clips are overwhelmingly dominated by the impacts of fossil fuel-based copolymers. The benefits of reduced impacts from feedstock production are marginal. The future improvement of starch plastics should be focused on achieving a full bio-based composition and decrease the dependency on the fossil fuel-based copolymers. This case highlights how more research and development into processing technology and materials is needed before significant improvements in environmental impacts can be achieved. The case further shows how a rather immature technology still has some distance to develop before competitive scale production can be achieved, while possible technically.

- In the case of bio-based PP cups made from UCO[83], the environmental benefit offered by this technology is substantial. Bio-based PP made from UCO offers a negative GHG emissions based on the cradle-to-grave comparison with the baseline PLA cups (See Case study chapter Single-use cups). In other words, the product stores more (biogenic) carbon than emitted. The benefits are derived from not only low impacts of raw feedstock (only one process of waste oil collections) but also the overall low impacts of the conversion processes benefit from two reasons: 1) an efficient thermochemical conversion which does not need to deal with a large amount of water in the production system (unlike the production of e.g. lactic acid); 2) by using a side stream (bio-naphtha) from bio-energy (i.e. renewable diesel) production as feedstock.

1.2 Findings on impacts of direct or indirect land use changes

An attempt was made to account for land use changes in this study. Direct land use changes were accounted for based on the PEFCR Guidance v6.3 recommendations. For the impacts of indirect land use changes a deterministic approach based on historical deforestation data (2000-2010) was developed within this study, building on existing frameworks. It resulted in an average indirect land use change (ILUC) factor of 4.0 t CO_2 eq. ha^{-1} y^{-1}, with a range of [1.22 – 5.20] t CO_2 eq. ha^{-1} y^{-1}. The main impacts affected by the inclusion of ILUC were

[83] Note that bio-based PP made from UCO is not a commercialised product. It has not yet reached the technology-readiness-level of 9.

climate change and photochemical ozone formation. When the effects of land use changes are accounted for, the impacts of biomass production do increase. However, while the share of biomass production in the overall impacts was not dominant in the first place, the additional land use changes share to the overall cradle-to-grave impacts is relatively marginal: on average 14% for GWP_{100}, 10% for photochemical ozone formation, and between 0.01-2.4% for all other impact categories.

1.3 Data uncertainties in the inventory models

The LCA case studies of bio-based products are built in three modules: cradle-to-user, EoL and land use changes. The uncertainties associated with the data used by these modules are different.

- For the cradle-to-user phases of the bio-based baseline systems for all seven cases, primary data were gathered, and in most cases, real supply chains were analysed. For this part of the module, we consider that the data quality is high, and the data uncertainty is low. In some cases where no company-specific data were available, or data provided by different companies were extremely diverse (and no plausible explanations can be found), generic data from literature are used, often verified by industrial experts before they were put into the model. This is especially the case for the plastics conversion technologies in the cradle-to-user phase.

- For the EoL phase, there are in many areas a lack of data. This corresponds to the variance in EoL technologies throughout EU and between collection schemes. Due to the many data lacks several assumptions were made, which is addressed further in section 4 under methodological challenges. This includes data quality issues is e.g. on:

    - the chemical composition of the biodegradable plastics (composition of the product itself is utilised, and not the plastic waste – i.e. additional items like ink or labels is not included),

    - assuming average European technologies, where several factors like energy consumption vary between e.g. Member States and specific plants, and

    - waste streams of specific products, i.e. the share of the waste products, which are sent for different waste treatment technologies or are littered.

- For the LUC models, the method relies on well acknowledged data (IPCC Guidelines for National GHG Inventories and the Forest Resource Assessment report of FAO). However, the former is being updated, and a new guideline titled "2019 refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories" is being written. Yet, the method also relies on choices (e.g. share of intensification and expansion; type of intensification) that can greatly affect the results. In an endeavour to address the impact of this, an interval for ILUC has been derived by changing these choices. It resulted in an average ILUC factor of 4.0 t CO_2 eq. ha^{-1} y^{-1}, with a range of [1.22 – 5.20] t CO_2 eq. ha^{-1} y^{-1}.

1.4 Comparing the LCA results with the petrochemical counterparts

One of the main aims of this study is to compare bio-based plastics and their petrochemical counterparts to provide evidence for policy debates for the future plastics policy. For petrochemical references, the key data of polymer production were taken from the latest PlasticsEurope's Eco-profiles (hereafter: the Eco-profiles).

The Eco-profiles have been established and maintained over decades by the industrial association and have been recognized as *the* LCI database representing the average EU

refineries and polymer production. The Eco-profiles have been widely used as a secondary data in LCAs or as benchmarks to compare. In this study, we also used the Eco-profiles (for the production of the polymers) to build the LCAs for the petrochemical counterparts for the comparison.

However, to conduct a full comparative LCA, it is important that insights of all options compared are obtained. In this study, a deeper insight was gained for the bio-based, whereas for the petrochemical polymers, the Eco-profiles appear as a black box. It is therefore hard to interpret the uncertainties of the LCA results based on the Eco-profiles.

In Chapter 6 of this report, we reviewed the Eco-profiles' data and compared the LCA results of all 16 PEFCR categories with other similar databases for the four polymers used in the case studies: PET (cases Beverage Bottles and Single-use Cups), LDPE (cases packaging films, mulch films and carrier bags), PP (cases Single-use cups and Clips) and PS (case Single-use Cutlery). The review results show that out of 16 impact categories, five categories are considered to have low discrepancy (i.e. less than ±50 %) in LCA results. These five categories are used for the overall comparisons in this study:

- climate change,
- abiotic depletion (fossil fuels),
- particulate matters,
- photochemical ozone formation, and
- terrestrial eutrophication.

Figure 214 shows the results of the comparison. We conclude that for all seven case studies, from cradle-to-grave the bio-based products offer environmental benefits in two impact categories: *climate change* and *abiotic depletion fossil fuels)*.

All seven bio-based products have a higher impact in particulate matter compared to the petrochemical counterparts. However, the results deviate from the median value (87%) significantly. Thus, thus median value of 87% higher PM impact for the bio-based in Figure 214 should be taken as an indicative value and should be treated with caution. The higher impact of particulate matter of bio-based products differs substantially from case to case. Bio-based PET has over six times higher impacts on particulate matter than the petrochemical PET due to the combustion of sugarcane trash in the biomass cultivation and harvesting phase. It should be noted that the burning cane trash will be soon phased out in Brazil. The impact of particulate matter impact of bio-based PET caused sugarcane production is expected to reduce substantially in the near future. In the case of PLA packaging films and cups, the particulate matter impacts are over two times as much as the impact of their PP counterparts. Compared to PP, PLA has a slightly higher energy requirement in the manufacturing phase and has much less avoided impact from the EoL phase. For all other cases, particulate matter is only marginally higher (3-8%) for the bio-based products than for their petrochemical counterparts.

For the remaining two impact categories i.e., photochemical ozone formation and terrestrial eutrophication some bio-based products (i.e. bio-based PET bottles, bio-based carrier bags and PLA cups, films and cutleries) do not offer impact reductions, whereas for some other cases, especially where in-situ soil biodegradation is assumed as EoL waste management (i.e. bio-based mulch films and clips), substantial impact reductions are observed. Therefore, no generic trend was observed for the comparisons for these two categories. The results are greatly case-dependent.

		Savings*, with EoL EU mix	Savings*, with intended EoL
Lower	Climate Change (GWP 100a)	-14% (-81% to 38%)	-66% (-87% to -14%)
	Abiotic depletion (fossil fuels)	-45% (-72% to -7%)	-46% (-72% to -3%)
Higher	Particulate matter	87% (2% to 603%)	94% (-2% to 597%)
Inconclusive	Photochemical ozone formation	23% (±79%)	12% (-79% to 58%)
	Terrestrial eutrophication	66% (-77% to 143%)	61% (-77% to 129%)

*Median savings based on the eight comparisons of the seven case studies (two comparisons were made for single-use cups).

Figure 214. Comparing cradle-to-grave environmental impacts of the seven bio-based products with their petrochemical counterparts, results indicated as environmental impact median savings of seven bio-based products (negative values stand for savings). The comparisons between the bio-based products with *intended EoL* are listed in the second column. Savings are without considering the effect of direct and indirect land use changes.

The choice of EoL has a strong influence on the overall savings of the bio-based products. The influence is especially significant for climate change impact: with intended EoL, the bio-based products on average could offer more than 65 % of the GHG emissions savings, instead of 14 % in the Baseline, compared to their petrochemical counterparts. This indicates the great potential of low carbon bio-based products if the EoL waste management are implemented appropriately.

We recognise that showing such a comparison can easily start a discussion of "comparing apples with oranges", like in many comparative LCAs. We nevertheless demonstrate these results, not only to fulfil the goal of this study, but also to use this study as a stepping stone for future discussion. For any future policy decisions on bio-based products, a robust and transparent benchmark of petrochemical counterparts should be available. This requires more accessible and transparent data provided by the plastics industry as well as LCI data providers.

2 Findings of biodiversity assessments

The novel Chaudhary 2015 methodology have been demonstrated for assessment of land use impacts on terrestrial biodiversity associated with the cultivation of feedstocks used as baseline for the case studies carried out in the project.

Two assessments have been carried out at different levels of specificity in this study:

- at country level for cultivation of all case study baseline feedstocks;
- at ecoregion level for cultivation of Thai sugarcane and US maize used as base materials for production of PLA (case study single-use cups, single-use cutlery and packaging films).

In the **first assessment** with country aggregated characterisation factors aspects of biodiversity loss is put into the picture where it otherwise would not have been considered due to lack of specific information. The results showing differences between countries can be used to get an idea of possible hot spots and may guide further investigation.

In the **second assessment** at ecoregion level locality is narrowed down to a specific radius of mills processing feedstocks, and relevant ecoregions have been identified and ecoregion characterisation factors applied. This provides a more accurate picture of the ecosystem pressure related to cultivation compared to the previous assessment made using country average characterisation factors.

With country aggregated characterisation factors, the global impact assessment predicted the highest potential biodiversity loss for maize cultivation in Italy compared to cultivation of other feedstocks in other countries. Secondly, the highest potential biodiversity loss was associated with maize cultivation in USA and sugarcane cultivation in Thailand. With ecoregion characterisation factors for cultivation of sugarcane (Thailand) and maize (USA), the global impact assessment predicted a slightly higher potential biodiversity loss associated with sugarcane cultivation in Thailand.

However, before entering decision-making about whether to use one or the other feedstock for production of bio-based products more in-depth investigations must be performed. Firstly, specific locations of cultivation must be considered for all feedstocks because the impact may vary greatly with location. Secondly, biodiversity assessment at ecoregion level must be carried out (as was done for US maize and Thai sugarcane). Thirdly, local conditions influencing biodiversity not considered by the performed biodiversity assessment and local agricultural practices must be considered. Precautions to protect biodiversity may already have been put in place and agricultural management may be different than assumed. Furthermore, uncertainty of applied characterisation factors, land use requirements and identification of ecoregions should be further scrutinized, and the known limitations of the Chaudhary 2015 methodology in terms of predicting biodiversity loss should be considered as well.

In agreement with the recommendations of UNEP-SETAC (2016) the results obtained with the Chaudhary 2015 methodology in this project cannot be used to support comparative assertions. It is thus not possible to conclude about preferred feedstocks in bio-based products in terms of biodiversity loss based on the biodiversity assessments carried out in this project. Therefore, it is recommended that the results be supplemented with further assessment of the local biodiversity risks and management practices.

3 Socio-economic effects

Bio-based products are by many expected to provide socio-economic benefits. Examples of such benefits are less dependency on limited and increasingly expensive fossil resources, support to rural development, and increased industrial competitiveness through innovative eco-efficient bio-based products. Furthermore, job standards and job creation properties may differ between a bio-based production processes and that for the alternative fossil-based product.

The aim of the socio-economic assessment in this project was to assess whether any of the seven case studies' bio-based products have positive or negative socio-economic characteristics that should be noted and possibly be responded to when promoting them. This assessment was carried out based on a literature review, and not linked to quantitative modelling. Overall, the tentative conclusion is that there seems to be a tendency for bio-based products having better socio-economic properties compared to their fossil-based alternatives. In particular, jobs in rural areas are created by the bio-based feedstock production, although sometimes under adverse working conditions. Furthermore, the training/educational content is relatively high in the innovative bio-based production processes.

4 Methodological challenges

4.1 Life Cycle Assessment

There are many methodological challenges for the LCA of bio-based and fossil plastics. In this section, we do not aim to be comprehensive, but point out several the methodological issues encountered as part of this study. The coverage of challenges follows the life cycle itself, starting with challenges associated with feedstock production. The order does not represent any other ranking on the side of the authors as concerns importance or pertinence. The below outlined challenges relates to LCA of bio-based plastics, however the LCAs for fossil plastic face a range of challenges and are not complete either. The challenges faced by LCAs for fossil plastic includes direct land use, which are not considered. These challenges, among other issue related to fossil plastic LCAs, are described in section 4.2.

The first important life cycle stage of many bio-based plastics is **feedstock production on agricultural land**, e.g. sugarcane, maize and potatoes. The methodologies to account for most direct impacts of cultivating and supplying these production systems are generally well agreed upon. However, modelling of soil carbon C interactions, N_2O formation, as well as all soil nutrient immobilisation and loss still needs further work, among others. Also, as explained elsewhere in the report the impacts on biodiversity (and ecosystems and their services) of intense agricultural production remains sparsely covered and for this there is currently no method or data mature enough to support policy making.

The issue of how to take direct and indirect land use changes impacts into account remains a challenge. In this study, the GHG emissions caused by direct land use changes were considered through the methodology recommended by the latest PEFCR Guidance (v.6.3), i.e. a methodology based on the PAS2050:2011 and PAS2050-1:2012 (BSI, 2012, 2011). Field emissions occurring in the agricultural field in relation to the feedstock production were accounted for based on state-of-the-art life cycle inventory datasets (AgriFootprint v3.0; Ecoinvent v3.4). To comply with the latest PEFCR Guidance (v.6.3; European Commission, 2018), the emissions from indirect land use changes should be excluded from the LCA. In this study, these were nevertheless included separately, as a "non-PEF impact". To this end, a deterministic approach based on historical deforestation data (2000-2010) was developed within this study, building on the approaches recently presented by Tonini et al. (2016) and Edwards et al. (2014). As it builds upon historic data, the method does not estimate the impacts of the future cropland demand for EU bio-based products but provides an estimate of the overall deforestation and intensification emissions associated with demanding one hectare of cropland in the past. It was here the hypothesis that this estimate provides useful insights to anticipate the magnitude of these emissions in the future. Nevertheless, our approach allowed to present ready-to-use figures to assess the ILUC impacts per area of cropland demanded (rather than focusing on the use as e.g. fuel or for polymer production). The lack of such figures were a key issue prior to undertaking this study.

Further, the modelling of indirect effects is so far limited to GHG emissions (and often only CO_2) and climate impacts but should also be extended to non GHG-related impacts (Pant et al. 2018). By accounting, on top of well-mixed GHGs, for nitrous oxides and carbon monoxide from land expansion, as well as induced fertilisers production and ammonia, nitrates & phosphorus losses from land intensification, our ILUC approach do allow to reflect the impacts caused in all environmental impact categories recommended by the PEF, and not only climate change. Specifically, within the frame of this study, we point out that DLUC/ILUC impacts were assessed associated with the production of only the main feedstock. As also part of the additives / copolymers can be based on agricultural feedstocks (and thus can cause land use change), it should be highlighted that these impacts could be higher than modelled in this study. On the other hand, land use change impacts only occur once, i.e. after they have been written off (typically over 20 or 30 years), there is no more impact.

Moreover, any activities interacting, directly or indirectly, with cropland does involve land use changes emissions. Here, a rough (and among firsts, to authors knowledge) attempt was made to illustrate this for the additional demand of urban land (i.e. including the land demanded for petrochemical activities).

The **manufacturing phase** has the highest contribution for the life cycle impacts of the bio-based products. A lions's share of the impacts is associated with the fossil fuel energy consumption for process heat and electricity. If in the future the decarbonisation of the energy sector succeeds, the impacts from manufacturing phase would decrease for the bio-based as well as for the petrochemical. However, given that biogenic carbon stored in the product, especially if recycling of bio-based products can be realized, the decarbonisation of the energy sector would benefit more for the bio-based products than the petrochemical counterparts.

Another key life cycle phase is the **End of Life** with multiple uncertainties. Typically, the current situation is a mix of different EoL technologies, and so estimating what fraction of the waste product ends up in which EoL technology is already a challenge, for which in many cases scarce and insufficient data exists for the specific products. Only in the study for the beverage bottles the product-specific data were obtained, as PET bottles are currently the most recycled plastic and the recycling statistics have been constantly reported in literature and statistics.

There are numerous challenges for each EoL option as well, which are briefly highlighted below.

- Case studies 2, 4, 5 and 6 were assessed with an expansion of the reference flow to include food leftovers or soil, the amount is an uncertain parameter. For amount of food leftovers were analysed in a sensitivity analysis, where this was not highly sensitive to the cradle to grave results, except for the impact category terrestrial eutrophication.

- The biological technologies (*industrial composting, anaerobic digestion and in-situ based biodegradation)*, there are several uncertainties;

    - a significant uncertainty in relation to insufficient data about the actual biodegradability performance of the plastic e.g. biodegradable plastics in anaerobic digestion (De Wilde, 2017) deemed it impossible to make a standard certification for products to enter AD plants, because the typology is too complex with the large span of technical options). This was tested in the sensitivity analysis on the intended technologies, i.e. industrial composting, and did not show major influence of the cradle-to-grave results.

    - Furthermore, the amount of rejects from industrial composting and anaerobic digestions of the products are supposed sensitive. The data is uncertain because the pre- and post-sorting of the plants have different technical set-ups. The amount of reject of biodegradable plastic in the plants is also prone to change if more biodegradable plastic enters the waste stream. This was stemmed out of the temporal scope of this study.

- Typically, the preferred EoL option is *recycling*, when this is a possible EoL option (unless in-situ soil biodegradation is specifically aimed for, for agricultural applications). But also, here, there are significant differences between ideal scenarios with perfect mechanical separation and recycling and the actual, current practice. Challenges on *recycling* include

    - the amount of collected and recycled plastic is a sensitive parameter, as observed in cases. The collection rate and the sorting and technology efficiency varies tremendously depending on e.g. the EU member state, the collection scheme, the recycling facility and which new product the recycled material is

utilised for. This is especially true for PS recycling, where it is uncertain how many of the recycling facilities actually recycle PS (depends e.g. on current market size and development of the technology).

- Regarding PLA recycling there are several uncertain areas. First, the technology is not commercially available yet. No literature and little knowledge regarding the sorting and technology efficiency of PLA recycling. In this study, it was assumed that the sorting and technology efficiency of PLA recycling to be equal to those of the other fossil polymers. Second, the substitution of recycled PLA is set to substitute PP or PET, but in the future, PLA might substitute PLA which would decrease the environmental benefit of PLA recycling, as impacts from producing PLA are lower than the production of PP/PET.

- the problems with recycling specific bio-based polymers (PLA) and the potential recycling incompatibilities between bio-based/biodegradable and fossil-based plastics in mechanical recycling (Pant et al. 2018) (which was not included in the study), especially also with many additives unknown. It is also a question whether mechanical separation and recycling is ultimately the ideal solution for all types of bio-plastics. As pointed out in Task 3, mixed urban organic waste streams could also provide an interesting and abundantly available resource for the production of bio-based products. These different recycling options introduce additional uncertainty regarding our scenarios, but assessing these different options was outside of the scope of this study.

- In case studies 4 and 6, the plastic is disposed of together with food leftovers. In the baseline analysis, if the plastic is collected for recycling, the food leftovers are treated 53% with incineration and 47% with landfilling. A sensitivity analysis was carried out for case study 6, where the food leftovers are sent for industrial composting instead of incineration and landfill. By 2023 this will likely be more realistic, as organic waste is expected to be collected separately in all Member States (European commission, 2018a). The sensitivity analysis shows that changing this waste management of the food leftovers will affect the cradle-to-grave results with -8.4% for GWP up to 0.8% for abiotic depletion – fossil fuels.

- *Landfilling* poses similar challenges, as there is significant uncertainty with regard to the (anaerobic) decomposition of various bio-based plastics over time. Also, it is then questionable if and how much of the volatile organic compounds released are captured and combusted with energy recovery or end up in the atmosphere as fugitive emissions. For landfilling the degradation of the biodegradable plastic lack data and is deemed to have a major influence because direct emissions have a large contribution to the EoL phase for landfilling. Last but not least, landfilling is typically regarded as the least preferable solution of all EoL options. Yet, from *a climate change perspective*, non-biodegradable (bio-)plastics that lock away carbon for long time periods might actually be the most beneficial solution. However, climate change is not the only challange here. The ecosystem damages caused by potentially unwanted littering due to e.g., poor management of landfill sites, are still to be understood especially for non-degradable and persistant compounds.

- Finally, yet importantly, *littering* is nowadays one of the key environmental concerns both in the public and on EU policy level, yet the exact amount (and relative share) of plastics that are littered is very uncertain and will largely differ between countries and end-use applications. Also, the impacts of littering and microplastics are currently not properly captured in the toxicity-related impact categories (ecotoxicity and human toxicity). As pointed out by Pant et al. (2018) fate, exposure, and effect modelling for macro- and micro-plastics (incl. additives) is still in its infancy. Physical impacts (e.g.

entanglement; ingestion of larger plastic particles and their effects) and chemical impacts (e.g. due to microplastic formation, microplastics as carrier of other chemicals, unknown impact of additives) and biological impacts (e.g. microplastics as carrier of germs/alien species) are largely unknown and currently not included in LCA. Thus, the impacts of littering (e.g. leaving plastics in the soil or in a marine environment) are most likely underestimated in the current LCA results. However, it will likely take significant time and research before any data on these impacts will become available, so for the time being, this is going to remain a major uncertainty for this EoL modelling. It thus will also remain a difficult trade-off to compare e.g. the impacts of the feedstock production of biodegradable plastics (such as the bio-based starch film and clips) to the EoL impacts of their fossil counterparts. Having said that, non-biodegradable bio-based plastics and additives will cause similar problems as their fossil counterparts.

One general issue that is not taken into account in current LCA methodology is **the timing of emissions**, i.e. whether a GHG gas is emitted during or very soon after the use phase of whether this occurs only a long time after the production and use of the product. In the seven case studies investigated, average lifetimes are all very short (typically less than one year), but especially for more long-lived biomaterials and plastics (e.g. used in construction), this could lead to significant additional benefits of bio-based products. The issue of timing of emissions has been extensively discussed in literature for bioenergy-related applications (especially woody biomass for electricity and heat production, see e.g. Lamers and Junginger, 2013) and should also be investigated in more detail for bio-based plastics.

In this study, the PEFCR guidance (version 6.3) has been used a guiding document. The PEFCR guidance has been established for existing products, whereas in this study many **niche applications** are assessed. This also created methodological challenges. A summary of lessons learned is listed below.

- When defining the functional unit, the weights of the new products are often not determined by minimal engineering requirements but by many practical reasons. For example, the dimensions of a bio-based product are often observed in the market the same as the petrochemical plastics, even when the properties of the bio-based would allow a thinner option. The reason behind is that the process technologies are not yet optimized for new materials. The new materials are often processed in the same machines as the conventional plastics. In the future, if the demand of bio-based plastics would increase, there would sufficient incentives for the plastics conversion industry to optimize their moulds and processes, which could further lower the impacts of the bio-based products.

- As explained earlier, assumptions were made to establish the average EoL for niche applications. We compared the results between the EoL mix and the EoL intended to provide the perspective for the range of impacts of EoL. Given lack of data of EoL of bio-based plastics and many EoL technologies are not yet in place in practice (e.g. recycling of PLA), we consider this approach as currently the optimal one to provide the answer to the research question.

- Based on our results, recycling or industrial composting are the preferred options for some of the bio-based and biodegradable plastics. However, to achieve 100 % recycling or 100 % industrial composting of the bio-based and biodegradable plastics still requires large efforts in the recycling technology developments, the adaptation of waste management infrastructure to accommodate the newcomers and the communication/education for the public.

It is also important to remember that the weighting factors applied in this study are the default weighting factors recommended by the PEFCR guidance (v6.3). The stakeholders involved in

determining these weighting factors do not necessarily reflect the opinion of the stakeholders of the bio-based economy, who may give heavier weight to natural resources and ecosystem services than the default weightings.

Last, the decision context of the LCAs in this study remained in the micro-level decision context. In many cases we looked at the activities retrospectively to be able to establish the "average impacts" of a product. However, when we looked into a novel product or a perspective technology, it is more important to understand the change-oriented effects for the future decisions. In that case, a more consequential thinking is required and a macro/meso-level decision context could be chosen to support policy debate.

4.2 Comparing the environmental benefits of bioplastics to bioenergetic use

As shown above, bioplastics do in general show a somewhat better environmental performance than their fossil-based counter parts, but advantages appear often also to be limited. Especially when comparing overall GHG performance for the seven case studies investigated, while there is a wide variation, on average, the savings are (far) less than the 65 % reduction required for biofuels from 2021 onwards and the 70 % / 80 % reduction for production of heat and electricity required after 2021/20-26 respectively, as was recently announced as thresholds as outcome of the trialogue for the RED-II. This ultimately raises the question what the best use of biomass is from an environmental and climate perspective. However, a direct comparison is difficult for many reasons:

- In the RED-II methodology, the energy allocation method is used for by-products, whereas in this study, system expansion was applied. Therefore, the GHG reductions cannot be directly compared, especially for feedstock conversion routes that produce significant quantities of non-energy products (e.g. routes involving starch or vegetable oils)

- The relative reduction compared to the fossil reference is only one possible metric; others could be the absolute reduction per kg of final of product, or (for routes based on feedstocks from agricultural land) the emission reduction per hectare of land. These may lead to different results and insights and should thus be evaluated next to each other.

- Comparing the performance of energy vs. material use of biomass based on a single impact category (GWP) is misleading – ideally, as the performance may vary substantially for different impact categories. All impact categories should be compared.

- Bio-based industries are a nascent sector. Their production processes not fully optimised for efficiency. In the future, through process optimisation and decarbonisation of the energy sector, the contribution of the manufacturing phase to the total environmental impact of bio-based products could be expected to decrease.

Through cascaded use, biomass can first be used for materials and then for energy, which ultimately is likely to provide the highest overall GHG benefits. However, what typical tipping points would be to go for recycling vs. energetic use depends again on many factors and are clearly beyond the scope of this study.

5 Recommendations

Based on the findings of the 7 case studies, the following recommendations are made on how to reduce problems with data availability, address methodological challenges, and what additional research is needed to provide additional evidence for policy making.

During developing the seven case studies, frequently, missing data hampered the comparison of bio-based and petrochemical plastics. The following recommendations are made:

- Specially to produce petrochemical polymers, most data are in principle available, but typically treated as confidential and not publicly accessible. While also much of the data on the production of bio-based plastics is confidential, many producers decided to provide data under NDA's for this report. On the other hand, PlasticsEurope and the LCI data owners did decide not to provide equivalent support. This ultimately led to the situation that the number of impact categories that could be compared in all case studies was reduced from 16 to 5. It is recommended that PlasticsEurope and the LCI data owners provide more transparent data that allows the data users to have a full interpretation. Only by this way it is possible to make the comparison with full insights.

- EoL scenarios often depend on the actual composition of post-consumer waste streams – but these are in practice largely unknown. Better monitoring of the composition of e.g. urban wastes would allow a more accurate determination of the possible EoL scenarios.

- Behaviour of new bio-based materials in e.g. industrial composting facilities, landfills and recycling facilities is still not well-understood, and can have a significant impact on the EoL performance. Better monitoring and modelling of these behaviours would allow reducing uncertainties regarding the assessment of EoL impacts.

Also, many methodological challenges were encountered. The most important ones are listed below:

- This study started out assessing the environmental impacts of bio-based materials, and – given the additional land-use of feedstock production – also focussed to the indirect impacts of land use. However, also fossil fuel production requires land (especially when exploiting tar sands) or may have incidental impacts on land (e.g. due to oil spills) which are poorly or not taken into account in the current impact assessment methodology but would be required to properly assess and compare impacts on land-use.

- In general, environmental impacts of the primary production of the feedstocks for the bioeconomy is not sufficiently addressed within LCA. The consideration of different origins of the feedstock for bio-based products (e.g. Europe and US) in this study indicates that certain production systems are more favourable than others. However, a crucial point is that several impacts are not included such as soil health, pesticide use, and impacts on biodiversity and ecosystems. This is partly because limited detailed data are available and partly because there are no mature methods suitable for policy support. It should be noted though, that as the environmental impacts of a particular biomass production are end-use agnostic, the lack of data and methods for assessment of impacts of primary production is relevant for all other end uses such as food, feed or e.g. bioenergy and are not related exclusively to the bio-based products covered in this study.

- As pointed out above, littering of (non-biodegradable) plastics may have significant long-term impacts in marine, freshwater and terrestrial environments, which are so far

poorly understood, and which also lack proper assessment in the current LCA methodology. Both additional inventory data collection and advances in impact assessment methods development are needed to better understand the impacts of littering and allow a more adequate comparison with biodegradable alternatives. It is possible to list littering as a separate environmental indicator to facilitate the reporting of the problem instead of often being neglected due to the data and method limitation of the cause-effect models in LCIA. The definition of such a separate environmental indicator of littering should be carefully drafted by inviting science as well as engaging stakeholder dialogue.

- Both biodiversity and socio-economic impact have only been briefly assessed in this report. Biodiversity impacts are currently not included in LCA methodology. The impacts of feedstock production on biodiversity are end-use agnostic and thus need to be assessed also for the production of food, feed, bioenergy and other end-uses, which goes beyond the scope of this study. Socio-economic impacts of bio-based and fossil plastic production and use also include the use and EoL phase and are therefore product dependent. But also, here, as social LCA is still in the early stages of development, a better methodological framework needs to be developed before social impacts can adequately and quantitatively be assessed.

To support the further policy development on bio-based products, the following recommendations are made:

- In this study, only seven bio-based products, notably all single-use items, were compared, based on a limited number of feedstocks. To make a better and more comprehensive analysis, we recommend carrying out a larger number of case studies, including different feedstocks, but also different end-uses. In the current case study, only short-lived and largely single use applications were evaluated, which also limits the possibility to draw wider conclusions.

- The current analysis only shows relatively limited benefits of bio-based plastics compared to petrochemical plastics. However, even commercially available bioplastics are fairly young compared to their fossil counterparts, and in many cases, additional research and development could improve their properties, reducing e.g. the weight per functional units or process energy required, further improving their environmental performance. Also, endogenous developments, such as improvements in agriculture and increasing use of renewable process energy would probably favour the bio-based alternatives. We recommend carrying out such an assessment, including assessment of technological developments and (fully) consequential life-cycle analysis.

- The assessment of the End of Life phase of the seven case studies is a good example of the benefits of integrating bioeconomy and circular economy research and policy. The conclusions of the EoL phase for the seven case studies show that environmental impacts of bio-plastics typically, but not always, follow the waste hierarchy. Recycling is, when possible, often preferred, except for products which tend to include food leftovers. Here, industrial composting and anaerobic digestion are preferred. Furthermore, this study highlights the importance of the EU target of maximum 10 % landfilling by 2030 as the bio-based products showed high impacts when landfilled. The biodegradable clips and mulch films performed well compared to their petrochemical references, however, the impact of littering was not modelled. The many sensitivity analyses performed showed that the sorting and technology efficiency of the modelled recycling facilities were sensitive to the overall cradle-to-grave results for the bio-based non-biodegradable products. It is recommended that new targets are set with focus on the actual recycled material and not on collection rates for recycling. The work on EoL

criteria for biobased and conventional plastic waste is important to guide producers and consumers towards the preferred EoL management.

Ultimately, this study provides a good starting point of discussion for comparing bio-based and fossil fuel-based plastics but cannot be used on its own for formulation for general policies for bio-based plastics.

LIST OF ABBREVIATIONS

AP	Acidification
ASTM	American Society for Testing and Materials
B2B	Business to Business
B2C	Business to Consumer
CCFB	Climate-carbon feedback
CF	Characterisation Factor
COD	Chemical oxygen demand
COM	Commission
DEG	DiEthyelene Glycol
DLUC	Direct Land Use Change
EASETECH	Environmental Assessment System for Environmental TECHnologies
ELCD	European Life Cycle Database
EoL	End of Life
FDCA	FuranDiCarboxylic Acid
FU	Functional Unit
GHG	Greenhouse gas
GTP	Global Temperature change Potential
ILUC	Indirect Land Use Change
IPCC	Intergovernmental Panel for Climate Change
ISO	International Organization for Standardization
IUCN	International Union for Conservation of Nature
JRC	Joint Research Centre
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LDPE	Low Density Polyethylene
LHV	Lower heating value
MEG	MonoEthylene Glycol
MSW	Municipal Solid Waste
MSWI	Municipal Solid Waste Incineration
NDA	Non-Disclosure Agreements
NMVOCs	Non-Methane Volatile Organic Compound(s)
NREU	Non-renewable energy use
ODP	Ozone Depletion Potential

PBS	Polybutylene Succinate
PDF	Disappeared Faction
PDLA	Poly D-lactide
PDO	Propanediol
PE	Polyethylene
PEF	Product Environmental Footprint
PEF	Polyethylene furanoate
PEFCR	Product Environmental Footprint Category Rules
PET	Polyethylene Terephthalate
PHB	Polyhydroxybutyrate
PLA	Polylactide
PLLA	Poly L-lactide
PP	Polypropylene
PS	Polystyrene
PSL	Potential Species Loss
PTA	Purified Teraphthalic Acid
SAR	Species-Area Relationship
TEG	Triethylene Glycol
TPS	Thermoplastics
TS	Total solids
UCO	Used Cooking Oil
VS	Vulnerability Score
VS	Volatile solids
WRI	World Resources Institute

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ANNEX 1 LIFE CYCLE INVENTORIES FOR END OF LIFE

In this annex, the modelling of EoL in EASETECH is explained in greater detail.

Full Material compositions

Here, the full material physio-chemical composition of each plastic material is presented. This is the input to the EASETECH model. The case(s) where the composition is used is indicated. The unit indicated with % means % of total solids (TS).

Case #	Material name	With/withour contamination	Water (%)	TS (%)	VS (%TS)	Ash (%TS)	Energy (MJ/kgTS)	C bio (%TS)	C bio and (%TS)	C fossil (%TS)	Ca (%TS)	Cl (%TS)	F (%TS)
1	Biogenic 30% PET (packaging)	with	3.25E+00	9.68E+01	9.90E+01	1.03E+00	2.29E+01	1.92E+01	1.92E+01	4.48E+01	1.57E-01	2.61E-01	1.00E-03
3, 6	Biogenic PLA (packaging)	without	1.00E-01	9.99E+01	1.00E+02	0.00E+00	1.87E+01	5.00E+01	2.34E+01	0.00E+00	0.00E+00	0.00E+00	0.00E+00
3	Biogenic PP (packaging)	with	3.25E+00	9.68E+01	9.40E+01	6.01E+00	3.78E+01	7.76E+01	0.00E+00	0.00E+00	7.03E-01	1.32E-01	2.02E-03
7	Biogenic starch (carrier bags)	without	3.63E-01	9.86E+01	9.99E+01	3.32E-01	2.45E+01	1.74E+01	1.47E+01	3.90E+01	0.00E+00	0.00E+00	0.00E+00
2	Biogenic starch (clips)	without	3.30E-02	9.90E+01	6.60E+01	3.30E-01	2.16E+01	3.23E+01	2.34E+01	2.04E+01	0.00E+00	0.00E+00	0.00E+00
5	Biogenic starch (mulching film)	without	3.63E-01	9.90E+01	9.83E+01	3.63E-01	2.33E+01	1.74E+01	1.47E-01	3.90E+01	0.00E+00	0.00E+00	0.00E+00
5	Fossil LDPE (non-packaging)	without	7.13E+00	9.29E+01	9.89E+01	1.14E+00	4.12E+01	0.00E+00	0.00E+00	8.22E+01	1.80E+00	1.22E-01	2.40E-03
7	Fossil LDPE (packaging)	with	3.25E+00	9.68E+01	9.84E+01	1.63E+00	4.15E+01	0.00E+00	0.00E+00	8.38E+01	1.69E-01	3.58E-02	1.00E-03
7	Fossil LDPE (packaging)	with	3.25E+00	9.68E+01	9.84E+01	1.63E+00	4.15E+01	8.38E+01	0.00E+00	0.00E+00	1.69E-01	3.58E-02	1.00E-03
1, 3	Fossil PET (packaging)	with	3.25E+00	9.68E+01	9.90E+01	1.03E+00	2.29E+01	0.00E+00	0.00E+00	6.40E+01	1.57E-01	2.61E-01	1.00E-03
6	Fossil PP (packaging)	with	3.25E+00	9.68E+01	9.40E+01	6.01E+00	3.78E+01	0.00E+00	0.00E+00	7.76E+01	7.03E-01	1.32E-01	2.02E-03
4	Fossil PS (packaging)	with	3.25E+00	9.68E+01	8.97E+01	1.03E+01	3.60E+01	0.00E+00	0.00E+00	8.32E+01	7.47E-01	7.47E-02	1.36E-03
4	PLA/starch mix	without	6.34E-02	6.73E+01	6.33E+01	3.96E+00	1.19E+01	3.16E+01	1.48E+01	0.00E+00	0.00E+00	0.00E+00	0.00E+00
3,4,6,7	Organic waste contamination		7.20E+01	2.80E+01	9.39E+01	6.08E+00	1.99E+01	4.95E+01	4.07E+01	4.62E-01	1.44E+00	8.28E-01	1.00E-02
2,5	Agricultural soil contamination		3.20E+01	6.80E+01	9.00E+00	9.10E+01	1.02E+00	0.00E+00	0.00E+00	7.35E-01	4.30E-01	3.00E-02	5.00E-03

Continued

Case #	Material name	H (%TS)	K (%TS)	N (%TS)	Na (%TS)	O (%TS)	P (%TS)	S (%TS)	Ag (%TS)	Al (%TS)	As (%TS)
1	Biogenic 30% PET (packaging)	5.20E+00	2.04E-02	1.58E-01	6.21E-02	2.76E+01	7.50E-03	5.03E-02	4.74E-04	2.97E-02	2.15E-04
3, 6	Biogenic PLA (packaging)	5.56E+00	0.00E+00	0.00E+00	0.00E+00	4.44E+01	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
3	Biogenic PP (packaging)	1.34E+01	4.41E-02	3.53E-01	1.56E-01	0.00E+00	5.16E-02	4.50E-02	1.79E-03	1.39E-01	2.92E-04
7	Biogenic starch (carrier bags)	6.61E-02	0.00E+00	0.00E+00	0.00E+00	3.57E-01	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
2	Biogenic starch (clips)	6.07E-02	0.00E+00	2.64E-03	0.00E+00	3.82E-01	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
5	Biogenic starch (mulching film)	6.61E-02	0.00E+00	0.00E+00	0.00E+00	3.41E-01	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
5	Fossil LDPE (non-packaging)	1.39E+01	2.15E-02	6.63E-01	6.88E-01	0.00E+00	4.36E-03	7.73E-02	4.98E-04	2.26E-01	3.51E-04
7	Fossil LDPE (packaging)	1.47E+01	2.00E-02	1.61E-01	9.13E-02	0.00E+00	1.87E-03	5.03E-02	4.64E-04	8.19E-02	2.11E-04
7	Fossil LDPE (packaging)	1.47E+01	2.00E-02	1.61E-01	9.13E-02	0.00E+00	1.87E-03	5.03E-02	4.64E-04	8.19E-02	2.11E-04
1, 3	Fossil PET (packaging)	5.20E+00	2.04E-02	1.58E-01	6.21E-02	2.76E+01	7.50E-03	5.03E-02	4.74E-04	2.97E-02	2.15E-04
6	Fossil PP (packaging)	1.34E+01	4.41E-02	3.53E-01	1.56E-01	0.00E+00	5.16E-02	4.50E-02	1.79E-03	1.39E-01	2.92E-04
4	Fossil PS (packaging)	8.60E+00	2.00E-02	5.33E-01	5.63E-02	0.00E+00	3.28E-02	3.99E-02	8.82E-04	6.16E-02	2.64E-04
4	PLA/starch mix	3.54E-02	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
3,4,6,7	Organic waste contamination	6.93E+00	1.09E+00	3.18E+00	5.04E-01	3.42E+01	4.22E-01	2.33E-01	0.00E+00	8.45E-02	3.63E-05
2,5	Agricultural soil contamination	7.00E-01	9.80E-01	3.00E-01	3.60E-01	3.00E+00	8.00E-02	4.00E-02	0.00E+00	1.84E+00	2.28E-04

Continued

Case #	Material name	Au (%TS)	Ba (%TS)	Be (%TS)	Br (%TS)	Cd (%TS)	Cr (%TS)	Cu (%TS)	Fe (%TS)	Ga (%TS)	Hg (%TS)
1	Biogenic 30% PET (packaging)	5.05E-06	3.64E-03	2.20E-05	1.00E-03	4.57E-05	1.05E-03	2.01E-03	3.43E-02	6.67E-05	2.40E-07
3, 6	Biogenic PLA (packaging)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
3	Biogenic PP (packaging)	5.06E-06	3.50E-03	3.69E-04	1.00E-03	5.13E-04	4.89E-03	2.88E-03	1.70E-01	1.18E-04	4.93E-07
7	Biogenic starch (carrier bags)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
2	Biogenic starch (clips)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
5	Biogenic starch (mulching film)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
5	Fossil LDPE (non-packaging)	5.31E-06	4.67E-02	2.31E-05	1.00E-03	4.80E-05	4.70E-03	4.60E-03	2.04E-01	4.80E-04	6.17E-07
7	Fossil LDPE (packaging)	2.74E-05	8.15E-03	2.15E-05	1.00E-03	4.48E-05	7.58E-04	6.10E-04	3.98E-02	8.98E-05	2.40E-07
7	Fossil LDPE (packaging)	2.74E-05	8.15E-03	2.15E-05	1.00E-03	4.48E-05	7.58E-04	6.10E-04	3.98E-02	8.98E-05	2.40E-07
1, 3	Fossil PET (packaging)	5.05E-06	3.64E-03	2.20E-05	1.00E-03	4.57E-05	1.05E-03	2.01E-03	3.43E-02	6.67E-05	2.40E-07
6	Fossil PP (packaging)	5.06E-06	3.50E-03	3.69E-04	1.00E-03	5.13E-04	4.89E-03	2.88E-03	1.70E-01	1.18E-04	4.93E-07
4	Fossil PS (packaging)	4.95E-06	7.06E-03	2.13E-04	6.80E-03	3.94E-04	5.77E-04	6.52E-04	4.74E-02	1.33E-04	9.60E-07
4	PLA/starch mix	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
3,4,6,7	Organic waste contamination	0.00E+00	0.00E+00	0.00E+00	0.00E+00	9.92E-06	4.23E-04	1.09E-03	2.45E-02	0.00E+00	2.00E-06
2,5	Agricultural soil contamination	0.00E+00	0.00E+00	0.00E+00	0.00E+00	2.00E-05	3.34E-03	1.29E-03	0.00E+00	0.00E+00	7.00E-06

Continued

Case #	Material name	In (%TS)	Li (%TS)	Mg (%TS)	Mn (%TS)	Mo (%TS)	Ni (%TS)	Pb (%TS)	Pd (%TS)	Pt (%TS)	Sb (%TS)
1	Biogenic 30% PET (packaging)	1.40E-06	8.22E-05	9.07E-03	5.61E-04	9.13E-05	8.27E-04	7.47E-05	1.05E-05	1.51E-06	2.71E-02
3, 6	Biogenic PLA (packaging)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
3	Biogenic PP (packaging)	1.89E-06	1.19E-04	7.28E-01	1.96E-03	1.20E-04	2.98E-03	4.31E-03	1.76E-05	1.51E-06	8.52E-04
7	Biogenic starch (carrier bags)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
2	Biogenic starch (clips)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
5	Biogenic starch (mulching film)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
5	Fossil LDPE (non-packaging)	1.54E-06	1.48E-04	9.55E-02	2.90E-03	3.58E-04	8.68E-04	8.16E-03	1.10E-05	1.58E-06	1.01E-03
7	Fossil LDPE (packaging)	1.55E-06	2.89E-05	5.92E-03	3.89E-04	1.40E-04	8.10E-04	7.95E-05	1.03E-05	1.48E-06	3.91E-04
7	Fossil LDPE (packaging)	1.55E-06	2.89E-05	5.92E-03	3.89E-04	1.40E-04	8.10E-04	7.95E-05	1.03E-05	1.48E-06	3.91E-04
1, 3	Fossil PET (packaging)	1.40E-06	8.22E-05	9.07E-03	5.61E-04	9.13E-05	8.27E-04	7.47E-05	1.05E-05	1.51E-06	2.71E-02
6	Fossil PP (packaging)	1.89E-06	1.19E-04	7.28E-01	1.96E-03	1.20E-04	2.98E-03	4.31E-03	1.76E-05	1.51E-06	8.52E-04
4	Fossil PS (packaging)	1.37E-06	5.12E-04	2.63E-02	1.54E-03	7.50E-05	8.60E-04	6.12E-04	1.28E-05	5.02E-06	2.80E-03
4	PLA/starch mix	0.00E+00	0.00E+00	1.17E-02	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
3,4,6,7	Organic waste contamination	0.00E+00	0.00E+00	1.17E-01	6.63E-03	7.31E-05	2.10E-04	7.98E-05	0.00E+00	0.00E+00	0.00E+00
2,5	Agricultural soil contamination	0.00E+00	0.00E+00	1.15E-01	2.40E-02	4.00E-04	2.55E-04	1.19E-03	0.00E+00	0.00E+00	0.00E+00

Continued

Case #	Material name	Se (%TS)	Si (%TS)	Sn (%TS)	Sr (%TS)	Ta (%TS)	Ti (%TS)	V (%TS)	Zn (%TS)	Zr (%TS)
1	Biogenic 30% PET (packaging)	0.00E+00	2.88E-01	3.69E-04	3.50E-04	2.14E-05	3.88E-02	3.64E-05	6.58E-04	1.42E-04
3, 6	Biogenic PLA (packaging)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
3	Biogenic PP (packaging)	0.00E+00	1.94E+00	1.52E-04	1.05E-03	1.33E-05	2.63E-01	1.55E-04	5.50E-03	5.05E-04
7	Biogenic starch (carrier bags)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
2	Biogenic starch (clips)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
5	Biogenic starch (mulching film)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
5	Fossil LDPE (non-packaging)	0.00E+00	3.44E+00	8.79E-04	2.79E-03	1.03E-05	2.53E-01	1.65E-04	1.49E-02	9.38E-04
7	Fossil LDPE (packaging)	0.00E+00	2.41E-01	1.43E-03	6.72E-04	1.23E-05	3.10E-01	3.86E-05	3.57E-03	2.46E-04
7	Fossil LDPE (packaging)	0.00E+00	2.41E-01	1.43E-03	6.72E-04	1.23E-05	3.10E-01	3.86E-05	3.57E-03	2.46E-04
1, 3	Fossil PET (packaging)	0.00E+00	2.88E-01	3.69E-04	3.50E-04	2.14E-05	3.88E-02	3.64E-05	6.58E-04	1.42E-04
6	Fossil PP (packaging)	0.00E+00	1.94E+00	1.52E-04	1.05E-03	1.33E-05	2.63E-01	1.55E-04	5.50E-03	5.05E-04
4	Fossil PS (packaging)	9.05E-05	7.58E-01	3.64E-04	1.30E-03	1.81E-04	6.19E-01	7.55E-05	1.37E-02	5.65E-04
4	PLA/starch mix	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
3,4,6,7	Organic waste contamination	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	3.11E-03	0.00E+00
2,5	Agricultural soil contamination	0.00E+00	0.00E+00	1.50E-03	0.00E+00	0.00E+00	0.00E+00	0.00E+00	2.86E-03	0.00E+00

Average European technologies

In this section, the consumption of components and energy for each technology is presented and well as a simple explanation of the EASETECH model.

Plastic Recycling

Pastic recycling	Amount	Unit	Amount source	Process name	Process	Comment	Data quality
Avoided products							(TeR,GR,TiR,P)
Substitution of plastic material	0.81	kg / kg recycled (ww)	Case dependent	Case dependent	Plastics Europe's Ecoprofiles	*Substitution, the amount recycled multiplied with the market ratio for each specific fossil plastic.*	Case dependent
Inputs							
Process water	2.96E+00	kg / kg recycled (ww)	EASETECH, Based on Perugini et al, 2005	Process water from surface water	ELCD		
Process steam	2.56E+00	MJ / kg recycled (ww)	EASETECH, Based on Rigamonti et al., 2014	Process steam from natural gas 90%; consumption mix, EU-27, 2008, ELCD	ELCD	*Methane consumption were calculated as an average of different sources (the min. methane consumption was 2.29 MJ per kg recycl. PET and the max. was 2.90 MJ per kg recycl. PET). All the methane in Rigamonti et al. has been modelled as natural gas in EASETECH.*	
Sodium hydroxide	3.00E-03	kg / kg recycled (ww)	EASETECH, Based on Perugini et al., 2005	Sodium hydroxide, 1996, RER, ELCD	ELCD	*Soap component for cleaning the plastic*	
Electricity	3.20E-01	kg / kg recycled (ww)	EASETECH, Based on Perugini et al., 2005	market group for electricity, marginal, RER	Ecoinvent 3.4	*Electricity consumption*	1,1,1,1
Emissions from treatment of waste water	2.93E-02	kg / kg waste (ww)	EASETECH	*direct emissionsto surface water*		*Composition of COD, TOC, BOD, nitrogen, phosphorous and cadmium*	
Incineration of residues	Case dependent	kg / kg waste (ww)	See MSWI inventory	See MSWI inventory	See MSWI inventory	*All the residues are assumed to be sent to an average European incineration plant.*	See MSWI Inventory

Incineration

Process emissions are used as they are provided in EASETECH, i.e., the consumption of chemicals and energy. The following parameters are calculated to model an average European incineration plant:

Heat and electricity efficiency have been calculated as an average for all European plants (both with and without energy recovery). The average efficiencies for MSWI with energy recovery from the Confederation of European Waste-to-Energy Plants (CEWEP, 2013) for heat and electricity are 34.6 % and 14.9 % respectively. This is based on 314 waste-to-energy plants in Europe between 2007 and 2010. The amount of waste treated with energy recovery is reported to be 59,000,000 tonnes per year. Eurostat (2017a) report MSWI without energy recovery at 35,675,000 tonnes treated material per year

(avg. of 2006, 2008, and 2010). A heat efficiency of 22 % and an electricity efficiency of 9 % were calculated using a weighted average.

The amount of solid residue from incineration is calculated by EASETECH and dependant on the input. Hereafter a division of treatment representing an average European plant is provided:

- The percentage of fly ash considered to be utilised in salt mines is 43 % and 57 % of fly ash is considered to be landfilled (Manders, 2008).
- The percentage of bottom ash considered to be recycled is 60 % and 40 % is landfilled based on Manders (2008).

The landfilled fly ash from incineration was a duplicate of the bottom ash landfill process from incineration due to lack of data. The fly ash landfilling accounts for under 1 % of the combined impact and was therefore not further addressed. Wastewater treatment is not included in the model.

	Amount	Unit	Amount source	Process name	Process source	Comment	Data quality
Avoided products							(TeR,GR,TiR,P)
Heat	Case dependent	MJ/MJ waste (ww)	EASETECH, CEWEP and EUROSTAT	Marginal heat, district or industiral, natural gas; Europe without Switzerland	Ecoinvent 3.4	*Substitution calculated by EASETECH depending on input. Heat efficiency for MSWI in Europe both with and without energy recovery is 22%.* *With energy recovery: Average of 314 WtE plants in Europe operated by CEWEP. It includes 17 European countries (15 EU countries +CH+NO). The amount of waste treated in this plants corresponds to 85% of the total waste in the EU25 from the 314 plants 71 come from Northern Europe, 188 from Central Europe and 55 from South-Western Europe. It does not include plants without energy recovery. Total amount: 59 mio. tons/year.* *Without energy recovery: Calculated from EUROSTAT: Treatment of waste by waste category, hazardousness and waste operations. Total: 35.7 mio. tons/year*	1,1,1,1
Electricity	Case dependent	kWh/MJ waste (ww)	EASETECH, CEWEP and EUROSTAT	Market group for electricity, medium voltage; RER	Ecoinvent 3.4	*Substitution calculated by EASETECH depending on input. Electricity efficiency for MSWI in Europe both with and without energy recovery is 9%.* *With energy recovery: Average of 314 WtE plants in Europe operated by CEWEP. It includes 17 European countries (15 EU countries +CH+NO). The amount of waste treated in this plants corresponds to 85% of the total wate in the EU25 from the 314 plants 71 come from Northern Europe, 188 from Central Europe and 55 from South-Western Europe. It does not include plans without energy recovery. Total amount: 59 mio. tons/year.* *Without energy recovery: Calculated from EUROSTAT: Treatment of waste by waste category, hazardousness and waste operations. Total: 35.7 mio. tons/year*	1,1,1,1

	Amount	Unit	Amount source	Process name	Process source	Comment	Data quality
Inputs							(TeR,GR,TiR,P)
Sodium hydroxide	2.40E-05	kg/kg waste (ww)	EASETECH	Sodium hypochlorite production, product in 15% solution state, RER	Ecoinvent 3.4	*Used in the incineration facility*	
Hydrogen chloride gas	5.60E-06	kg/kg waste (ww)	EASETECH	Hydrogen chloride gas (HCL), 1996, RER, ELCD	ELCD	*Used in the incineration facility*	
Limestone	5.67E-03	kg/kg waste (ww)	EASETECH	Limestone production, crushed, for mill; CH	Ecoinvent 3.4	*Used in the incineration facility*	
Process water	3.97E-01	kg/kg waste (ww)	EASETECH	Process water from surface water, RER, 2005, ELCD	ELCD	*Used in the incineration facility*	
Polyethylene, high denstity granulate	6.00E-07	kg/kg waste (ww)	EASETECH	Polyethylene production, high denstity granulate, RER	Ecoinvent 3.4	*Used in the incineration facility*	
Lime	3.40E-04	kg/kg waste (ww)	EASETECH	Lime production, Hydrated, loose weight; CH	Ecoinvent 3.4	*Used in the incineration facility*	
Activated Carbon	1.04E-03	kg/kg waste	EASETECH			*Used in the incineration facility*	
Ammonia	1.53E-03	kg/kg waste (ww)	EASETECH	Market for Ammonia, liquid, RER	Ecoinvent 3.4	*Used in the incineration facility*	
Bottomash aggregate in Road soil subgrade	5.50E-03	kg/ kg waste (ww)	EASETECH and Manders, 2008	Combined processes		*The amount of solid residues from the incineration is calulated by EASETECH depending on the input. The share of bottom ash considered to be recycled is 60% based on Manders, 2008 (Life Cycle Assessment of the treatment of MSW in "the average" European Waste-to-Energy plant)*	2,3,2,2
Treatment of bottom ash for landfilling	3.67E-03	kg/ kg waste (ww)	EASETECH and Manders, 2008	Combined processes		*The amount of solid residues from the incineration is calulated by EASETECH depending on the input. The share of bottom ash considered to be landfilled is 40% based on Manders, 2008 (Life Cycle Assessment of the treatment of MSW in "the average" European Waste-to-Energy plant)*	4,4,2,4
Fly ash for utilization in salt mines	1.23E-03	kg/ kg waste (ww)	EASETECH and Manders, 2008	Combined processes		*The amount of solid residues from the incineration is calulated by EASETECH depending on the input. The share of fly ash considered to be utilized for salt mines is 43% based on Manders, 2008 (Life Cycle Assessment of the treatment of MSW in "the average" European Waste-to-Energy plant)*	3,2,2,3
Treatment of fly ash for landfill	1.63E-03	kg/ kg waste (ww)	EASETECH and Manders, 2008	Combined processes		*The amount of solid residues from the incineration is calulated by EASETECH depending on the input. The share of fly ash considered to be landfilled is 57% based on Manders, 2008 (Life Cycle Assessment of the treatment of MSW in "the average" European Waste-to-Energy plant)*	4,4,2,4

Landfill

Process emissions are used as they are in EASETECH based on an average MSW landfill. Processes with substantial impact on the results were further assessed and the following default values representing a MSW landfill were changed to fit plastic landfilling.

The leachate profile was changed to match a leachate profile for plastic in a landfill (modelled in four time periods in EASETECH for a total of 100 years). The leachate composition was based on Manfredi et al. (2010). The COD and BOD concentrations were based on the TOC concentration (Manfredi et al., 2010) and were estimated to be different for non-degradable (petrochemical and bio-based) and biodegradable plastic:

mg/L for 100 years	Non-degradable plastic	Biodegradable plastic
BOD	100	200
COD	600	150

Estimated based on ResearchGate (N.D).

The amount of leachate is based on different parameters including precipitation, which was set to 200 mm per year (Manfredi et al., 2010). A part of the leachate is collected and treated in a wastewater treatment plant and another part is not collected and ends up as an emission to surface water (also allocated into four time periods).

The amount of landfill gas produced is calculated by the k rate (1st order decay rate for methane generation), which is a product specific factor. The composition of the landfill gas is an average gas for MSW landfills.

The collected gas is time-dependent; gas is collected at varying rates between 35-75 % during the first 55 years (three periods). For the last 45 years it is assumed that none of the produced gas is collected. For the gas collected it is assumed that 22 % of the gas is flared, 29 % is used to generate energy and the last 49 % is leaked (OpenLCA Nexus, 2015). This data was validated between 2015 and 2018. The electricity efficiency of the landfill gas utilisation is 37 % (Christensen, 2011).

The landfill gas that is not collected is oxidised in top covers, which is dependent on the period it is produced within (daily cover for the first five years, intermediate cover for years five-15 and 15-100 years for the final cover). Landfill gas substances have uncertainties as they are not based on specific input values.

	Amount	Unit	Data source	Process	Process	Comment	Data quality
Avoided							(TeR,GR,TiR,P)
Heat	Case dependent	MJ/MJ waste (ww)	EASETECH, CEWEP and EUROSTAT	Marginal heat, district or industiral, natural gas; Europe without Switzerland	Ecoinvent 3.4	*Substitution calculated by EASETECH depending on input. Heat efficiency for MSWI in Europe both with and without energy recovery is 22%.* *With energy recovery: Average of 314 WtE plants in Europe operated by CEWEP. It includes 17 European countries (15 EU countries +CH+NO). The amount of waste treated in this plants corresponds to 85% of the total waste in the EU25 from the 314 plants 71 come from Northern Europe, 188 from Central Europe and 55 from South-Western Europe. It does not include plants without energy recovery. Total amount: 59 mio. tons/year.* *Without energy recovery: Calculated from EUROSTAT: Treatment of waste by waste category, hazardousness and waste operations. Total: 35.7 mio. tons/year*	1,1,1,1
Electricity	Case dependent	kWh/MJ waste (ww)	EASETECH, CEWEP and EUROSTAT	Market group for electricity, medium voltage; RER	Ecoinvent 3.4	Substitution calculated by EASETECH depending on input. Electricity efficiency for MSWI in Europe both with and without energy recovery is 9%. With energy recovery: Average of 314 WtE plants in Europe operated by CEWEP. It includes 17 European countries (15 EU countries +CH+NO). The amount of waste treated in this plants corresponds to 85% of the total wate in the EU25 from the 314 plants 71 come from Northern Europe, 188 from Central Europe and 55 from South-Western Europe. It does not include plans without energy recovery. Total amount: 59 mio. tons/year. Without energy recovery: Calculated from EUROSTAT: Treatment of waste by waste category, hazardousness and waste operations. Total: 35.7 mio. tons/year	1,1,1,1

Continued

	Amount	Unit	Amount source	Process	Process	Comment	Data quality
Inputs							(TeR,GR,TiR,P)
Electricity	8.00E-03	kWh/kg waste (ww)	EASETECH	Marginal heat, district or industiral, natural gas; Europe without Switzerland	Ecoinvent 3.4	*Used in construction and operation of landfill*	1,1,1,1
Diesel Oil in Truck	2.02E-04	kg/kg waste (ww)	EDIP, IPU-NF-	Production	EASETECH	*Used in construction and operation of landfill*	
Polyvinylchloride	1.00E-05	kg/kg waste (ww)	EASETECH	Polyvinylchlori	ELCD	*Used in construction and operation of landfill*	
Polyethylene high	2.30E-04	kg/kg waste (ww)	EASETECH	Polyethylene h	ELCD	*Used in construction and operation of landfill*	
Propylene produc	4.00E-08	kg/kg waste (ww)	EASETECH	Propylene proc	Ecoinvent 3.4	*Used in construction and operation of landfill*	
Aluminum	5.80E-08	kg/kg waste (ww)	EASETECH based on: Brogaard et al., 2013	Aluminum, Al (	EASETECH	*Used in construction and operation of landfill*	
Steel Sheets	1.40E-04	kg/kg waste (ww)	EASETECH based on: Brogaard et al., 2013	Steel Sheets (97.75% primary), Sweden, 2008	EASETECH	*Used in construction and operation of landfill (Steel + reinforcement steel)*	
Treatment of coll	Case dependent	kg/kg waste (ww)	EASETECH	Combined process		*depends e.g. on precipitaion, age of landfill, bulk density and*	2,3,2,2
Leachate uncolle	Case dependent	kg/kg waste (ww)	EASETECH	Combined process		*depends e.g. on precipitaion, age of landfill, bulk density and*	2,3,2,2
Landfill gas	Case dependent	kg/kg waste (ww)	EASETECH	Combined process		*depends e.g. on chemical compostition, rate of methane*	2,2,3,3

In-situ

	Amount	Unit	Amount source	Process	Process source	Comment	Data quality
Avoided products							(TeR,GR,TiR,P)
Collection Vehicle	-0.003	l/kg waste (ww)	EASETECH based on: Drivsholm et al., 2002.	Collection Vehicle, 10t Euro6, urban traffic, 1 liter diesel, 2012	EASETECH	*Avoided collection*	1,1,2,2
Inputs							
Farm tractor	0.01	l/kg waste (ww)	EASETECH, based on ILCD and LIPASTO	Farm tractor, combustion 1L of diesel, 2008/2011	EASETECH		2,2,3,1
Use on land	1.1	kg/kg waste	EASETECH based on: 1) Hansen et al., 2006 2) Bernstad and Jansen., 2011. 3) Bernstad and Jansen., 2011. 4) Hansenet al., 2012. 5) Børgesen et al., 2001. 6) Yoshida et al., n.d. 7) Klinglmair et al., n.d.	OFMSW Composted, Use on land, Coarse sandy soil – Ave. DK Crop rotation (High), DK, 2015	EASETECH	*The process describes the emissions to air, surface water, groundwater and soil accumulation from land application of composted organic fraction.*	2,2,3,3

Industrial composting

Pre-treatment and post-treatment are the most important processes to be included in an analysis. The purpose of pre-treatment is to remove larger objects before the materials enter into the treatment process, for example, by particle size reduction. Post-treatment is mainly sieving of the material which can remove large objects e.g., large types of bio-based plastic products, branches, etc. (De Wilde, 2018).

Default process emissions in EASETECH were used for compost processing plants. This includes usage of diesel for wheeled loaders, electricity and wood pellets (structure), which are dependent on input amount. The reject rate is set to 30 % of the plastic input and 5 % of the contamination input (EASETECH).

Land application impact depends on soil conditions, which are diverse around Europe. The soil conditions are here modelled as sandy loam soil (Tonini, 2018). Carbon in the biodegradable products that are disposed of on land and degraded on fields is distributed between CO_2 air, CH4 air and carbon in soil storage. The distribution of the carbon on a sandy loam soil is 88.68 % to CO_2, 0.01 % CH4 and 11.31 % to carbon in soil based on one dimensional simulation by an agro-ecosystem model. The values are simulated based on Danish weather data, which is not representative for average European weather and therefore contributes an uncertainty. Nitrogen degradation is not relevant for bioplastics because N content in bio-based biodegradable plastic products is negligible (De Wilde, 2018). Furthermore, metals Cr, Zn, Ni, Hg, Ca, Co and Pb are modelled as emissions to soil based on N, P and K substitutions.

	Amount	Unit	Amount source	Process	Processs source	Comment	Data quality
Avoided products							(TeR,GR,TiR,P)
Farm tractor	-0.01	l/kg sum of N, K and P	EASETECH, based on ILCD and LIPASTO	Farm tractor, combustion 1L of diesel, 2008/2011	EASETECH	*Application of N, K and P fertilizer avoided*	2,2,3,1
Potassium fertiliser	-1	kg/kg K	EASETECH	market for potassium fertiliser K2O	Ecoinvent 3.4	*K fertilizer substitution*	1,4,2,1
Nitrogen fertiliser	-0.2	kg/kg N	EASETECH	market for nitrogen fertiliser, as N; GLO	Ecoinvent 3.4	*N fertilizer substitution*	1,4,2,1
Phosphate fertiliser	-1	kg/kg P	EASETECH	market for phostphate fertiliser, as P2O5	Ecoinvent 3.4	*P fertilizer substitution*	2,4,2,2
Inputs							
Composting tunnel	0.75	kg/kg waste (ww)	EASETECH based on: 1) Marco & Marco., 2007. 2) Boldrin et al,. 2009	Composting, tunnel, green waste, Treviso, IT	EASETECH	*The process simulates an enclosed channel composting facility, processing 35000 tonnes/year. The process includes shredding, storing, sorting, electricity consumption, fuel consumption and land use changes.*	2,2,3,1
Use on land	0.75	kg/kg waste (ww)	EASETECH based on: 1) Hansen et al., 2006 2) Bernstad and Jansen., 2011. 3) Bernstad and Jansen., 2011. 4) Hansenet al., 2012. 5) Børgesen et al., 2001. 6) Yoshida et al., n.d. 7) Klinglmair et al., n.d.	OFMSW composted, Use on land, Sandy loam soil – Ave. DK Crop rotation (High), DK, 2015	EASETECH	*The process describes emissions to air, surface water, groundwater and soil accumulation from land application of composted organic fraction. The data includes an operation of diesel powered spreader*	1,2,2,2
Incineration of residues	0.3	kg/kg plastic waste (ww)	See MSWI Inventory	Combined process		See MSWI Inventory	See MSWI Inventory
Incineration of residues	0.05	kg/kg contaminataion	See MSWI Inventory	Combined process		See MSWI Inventory	See MSWI Inventory

Anaerobic digestion

Default process emissions from EASETECH for a biogas plant have been used. 23 % of the processing plants are thermophilic wet and 77 % are mesophilic or thermophilic dry technologies (Bruno & Baera, 2015).

The plants are modelled with two screenings (pre-treatment and post-treatment). Pre-treatment and post-treatment are the most important processes to be included in an analysis. The purpose of pre-treatment is to remove larger objects before the materials enter into the treatment process, for example, by particle size reduction. Post-treatment is mainly sieving of the material which can remove large objects e.g., large types of bio-based plastic products, branches, etc. (De Wilde, 2018). The reject rate is set to 30 % of the plastic input and 5 % of the organic waste contamination (EASETECH; De Wilde & Siebert, 2018).

The average biogas plant has a gas yield of 70 % based on measured data from a number of European biogas plants (EASETECH). The biogas composition of methane is 63 %. Of the biogas that is generated, 90 % is used in a combined heat and power unit, and the last 10 % is generated as vehicle fuel (European Biogas Association, 2011). The distribution of biogas utilisation is largely dependent on country. Sweden and Switzerland produce mostly fuels while other countries produce predominately electricity or heat. Efficiency of the combined heat and power unit is as follows: electricity 40 %, heat 50 %; the remaining 10 % is lost (Euopean Comission, 2016). 2 % of the produced methane is leaked to the atmosphere.

Bio plastic is not degraded fully under other conditions than wet thermophilic but the contamination of organic waste will degrade and generate biogas under dry and mesophilic conditions. For dry processing plants there is a split of digestate into either direct land use or to post-maturation, which becomes compost. The impact of land application of bioplastics depends on soil conditions, which are diverse around Europe. The soil conditions are here modelled as sandy loam soil (Tonini, 2018). Carbon in the biodegradable products that are disposed of on land and degraded on fields, is distributed between CO_2 air, CH4 air and carbon in soil storage. The distribution of the carbon on a sandy loam soil is 88.68 % to CO_2, 0.01 % CH_4 and 11.31 % to carbon in soil based on one dimensional simulation by an agro-ecosystem model. The distribution of carbon from digestate applied on land differs from the compost due to the water content and becomes 86.75 % CO_2 in air, 0.05 % CH_4 in air and 13.2 % C to soil. The values are simulated based on Danish weather data, which is not representative for average European weather and therefore contributes an uncertainty. Nitrogen degradation is not relevant for bioplastics because N content in bio-based biodegradable plastic products is negligible (De Wilde, 2018). Furthermore, metals Cr, Zn, Ni, Hg, Ca, Co and Pb are modelled as emissions to soil based on N, P and K substitutions.

	Amount	Unit	Amount source	Process	Process source	Comment	Data quality
Avoided products							**(TeR,GR,TiR,P)**
Heat	Case dependent	MJ/MJ waste (ww)	EASETECH, CEWEP and EUROSTAT	Marginal heat, district or industiral, natural gas; Europe without Switzerland	Ecoinvent 3.4	*Substitution calculated by EASETECH depending on input. Heat efficiency for MSWI in Europe both with and without energy recovery is 22%. With energy recovery: Average of 314 WtE plants in Europe operated by CEWEP. It includes 17 European countries (15 EU countries +CH+NO). The amount of waste treated in this plants corresponds to 85% of the total waste in the EU25 from the 314 plants 71 come from Northern Europe, 188 from Central Europe and 55 from South-Western Europe. It does not include plants without energy recovery. Total amount: 59 mio. tons/year. Without energy recovery: Calculated from EUROSTAT: Treatment of waste by waste category, hazardousness and waste operations. Total: 35.7 mio. tons/year*	1,1,1,1
Electricity	Case dependent	kWh/MJ waste (ww)	EASETECH, CEWEP and EUROSTAT	Market group for electricity, medium voltage; RER	Ecoinvent 3.4	*Substitution calculated by EASETECH depending on input. Electricity efficiency for MSWI in Europe both with and without energy recovery is 9%. With energy recovery: Average of 314 WtE plants in Europe operated by CEWEP. It includes 17 European countries (15 EU countries +CH+NO). The amount of waste treated in this plants corresponds to 85% of the total wate in the EU25 from the 314 plants 71 come from Northern Europe, 188 from Central Europe and 55 from South-Western Europe. It does not include plans without energy recovery. Total amount: 59 mio. tons/year. Without energy recovery: Calculated from EUROSTAT: Treatment of waste by waste category, hazardousness and waste operations. Total: 35.7 mio. tons/year*	1,1,1,1
Farm tractor	-0.01	l/kg sum of N, K and P	EASETECH, based on ILCD and	Farm tractor, combustion 1L of diesel, 2008/2011	EASETECH, based on ILCD	*Application of N, K and P fertilizer avoided*	2,2,3,1
Nitrogen fertiliser	-1	kg/kg K	EASETECH	market for nitrogen fertiliser,	Ecoinvent 3.4	*K fertilizer substitution*	1,4,2,1
Potassium fertiliser	-0.2	kg/kg N	EASETECH	market for potassium fertiliser	Ecoinvent 3.4	*N fertilizer substitution from compost*	1,4,2,1
Nitrogen fertiliser	-1	kg/kg P	EASETECH	market for nitrogen fertiliser,	Ecoinvent 3.4	*P fertilizer substitution*	2,4,2,2
Phosphate fertiliser	-0.4	kg/kg N	EASETECH	market for phostphate	Ecoinvent 3.4	*N fertilizer substitution from digestate*	1,4,2,1
Fuel	Case dependent	MJ/m^3 CH4	EASTETECH	Substitution of conventional liquid transport fuel (Production and combustion	EASTETECH	*Substitution of conventional liquid transport fue (from biogas upgrading)*	3,2,3,3

Continued

	Amount	Unit	Amount source	Process	Process source	Comment	Data quality
Inputs							**(TeR,GR,TiR,P)**
Incineration of residues	0.3	kg/kg plastic waste (ww)	See MSWI Inventory	Combined process		See MSWI Inventory	See MSWI Inventory
Incineration of residues	0.05	kg/kg contaminata	See MSWI Inventory	Combined process		See MSWI Inventory	See MSWI Inventory
Anaerobic digestion [Thermophilic wet]	0.17	kg/kg waste (ww)	EASTETECH	Biogas production, Thermophilic, Generic	EASTETECH	*The process simulates a A hypothetical one stage "wet" thermophilic anaerobic digestion plant treating municipal organic solid waste. The digestion temperature is typically around 53 - 55 °C for thermophilic digestion. Included in data-set are energy use for heating the reactor to thermophilic temperature, diesel to operate machinery at the plant and electricity for pumps, mixers etc.. The nutrients in form of digestate will be used on land and substitute fertilisers*	2,1,3,2
Anaerobic digestion [other technologies]	0.58	kg/kg waste (ww)	EASTETECH	Biogas production	EASTETECH	*Included in data-set are energy use for heating the reactor, diesel to operate machinery at the plant and electricity for pumps, mixers etc.. The nutrients in form of digestate are divided into either entering post composting or used moredirectly as digestate where both will be used on land to substitute fertilisers.*	4,4,4,4
Post compost	Case depender	kg/kg waste (ww)	EASETECH based on: 1) Andersen et al., 2010a 2) Andersen et al., 2010b	Composting windrows AD other technologies	EASETECH	*The dataset describes an open-air windrow composting facility. The data includes emissions to air, soil and water as well as consumption of materials (e.g. grinder, wheeled excavator lubricating grease) and energy use (e.g. electricity consumption, motor oil)*	3,2,3,3
Biogas upgrading	Case dependent	MJ/m^3 CH4	EASTETECH	Upgrading of biogas	EASETECH	*The upgrading of biogas using a membrane technlogy includes energy use, gas prompression, emission to environment and gas distribution losses*	3,2,3,3

Plastic Substitution

Case #	Recycled plastic	Substituted plastic	Process name	Process source	Comment	Data quality (TeR,GR,TiR,P)
1,3	Petrochemical PET	Petrochemical PET	PET, bottle grade, at plant, [RER]	Plastics Europe	*The process describes the production of bottle grade polyethylene terephthalate (PET) from cradle to gate (from crude oil extraction to PET granulate at plant).*	2,1,1,2
1	Biobased PET (30%)	Petrochemical PET	PET, bottle grade, at plant, [RER]	Plastics Europe	*The process describes the production of bottle grade polyethylene terephthalate (PET) from cradle to gate (from crude oil extraction to PET granulate at plant).*	2,1,1,2
3	PLA	Petrochemical PET	PET, bottle grade, at plant, [RER]	Plastics Europe	*The process describes the production of bottle grade polyethylene terephthalate (PET) from cradle to gate (from crude oil extraction to PET granulate at plant).*	2,1,1,2
6	PLA	Petrochemical PP	polypropylene, PP, granulate, at plant, RER	Plastics Europe	*The process describes the production of polypropylene (PP) from cradle to gate (i.e. from raw material extraction to polymer resin at plant)*	2,1,2,2
3	Biobased PP	Petrochemical PP	polypropylene, PP, granulate, at plant, RER	Plastics Europe	*The process describes the production of polypropylene (PP) from cradle to gate (i.e. from raw material extraction to polymer resin at plant)*	2,1,2,2
5,7	Petrochemical LDPE	Petrochemical LLDPE	polyethylene, LLDPE resin, granulate, at plant, RER	Plastics Europe	*This process describes the production of LLDPE from cradle to gate (i.e. from raw mate-rial extraction to polymer resin at plant).*	2,1,1,2
3,6	Petrochemical PP	Petrochemical PP	polypropylene, PP, granulate, at plant, RER	Plastics Europe	*The process describes the production of polypropylene (PP) from cradle to gate (i.e. from raw material extraction to polymer resin at plant)*	2,1,2,2
4	Petrochemical PS	Petrochemical PS	General purpose polysturene (GPPS), PlasticEurope, EU-27	Plastics Europe	This process describes the production of the General Purpose Polystyrene from cradle to gate (from crude oil extraction to granules or resin at plant, i.e. polystyrene production site output).	1,1,1,1
7	Biobased LDPE	Petrochemical LLDPE	polyethylene, LLDPE resin, granulate, at plant, RER	Plastics Europe	*This process describes the production of LLDPE from cradle to gate (i.e. from raw mate-rial extraction to polymer resin at plant).*	2,1,1,2

ANNEX 2 BIODEGRADABILITY

Overview of biodegradability of bio-based plastics in biological EoL technologies.

EoL Technology	Plastic type	Conditions	Time	Followed standard	Biodegradation indications (disintegration, biodegradation or not defined)	Reference
Industrial composting	Various Mater-Bi classes and grades	Aerobic conditions	45 days	ASTM D 5338-92	90 % of carbon	Bastioli (1998)
	Starch plastic	Aerobic conditions	72 days		26.7 % by weight	Mohee et al. (2007)
	Starch plastic	Aerobic conditions	1 week	ASTM D 5338-92 and EN 14046 standard	100 % disintegration	Unmar and Mohee (2008)
	PLA	Aerobic conditions			76 %	Hermann et al., 2011
	Maize starch	Aerobic conditions			73 %	Hermann et al., 2011
	Starch	Aerobic conditions			93 %	Hermann et al., 2011
	Mater-Bi	Aerobic conditions			63 %	Hermann et al., 2011
Anaerobic digestion	Different biodegradable plastics	Anaerobic digestion	81 days		10 % (pellets) to 21 % (film)	Scandola (1998)
	Starch plastic (60 % starch)	Anaerobic conditions, sludge inoculum	32 days	ASTM D 5526 – 94	34.7 % of VS	Mohee et al. (2007)
	Starch-PCL mix	Anaerobic	139 days		88 %	Cho et al. (2011)
	Maize and co-polymer	Mesophilic conditions	50 days		20.2 % carbon loss	Gómez (2013)
Combined AD and industrial	Mater-Bi (3.5 % and 3.8 % of the	Anaerobic digestion plants followed by	5-10		Degradation started during the AD phase, however 100	Novamont

composting	input material)	composting of the digestate	weeks		% degradation only at the composting phase	(2017)
In-situ	Starch plastic	Buried in aqueous biotic soil temperature is 20 ºC, the humidity was 40 %	9 months		50 % by weight	Di Franco et al. (2004)
	Maize and co-polymer	Soil burial test, 20 ºC, 60 % water content	660 days		50 % carbon loss	Gómez (2013)

Overview of biodegradability standards

Standard	Description
EN 14045: 2003	This standard tests if the product is disintegrated in the beginning of the biodegradation phase, and that the leftover products is not visible in the compost under defined aerob conditions.
ASTM D 5338-92 standard / US standard for biodegradability during composting	This standard determines the degree of aerobic biodegradability of plastic material in controlled compost environment under laboratory conditions. The chosen plastic and the inoculum are placed in a composting vessel for 45 days. The inoculum must be a 2 to 4 months old, well-aerated compost, deriving from municipal organic waste. The temperature is kept stable at 58 ºC (± 2 ºC) and the vessel is placed in the dark. Aeration, temperature and humidity are monitored regularly. The experiment measures the total carbon dioxide produced over the study period, which is then translated into the biodegradability rate as indicated in fomula (4)
ASTM D-6400-99 standard /US standard for composability of plastic	This standard is a guide for assessing the composability of the plastics based on the results of ASTM D 5338-92 standard above, which assesses the biodegradation rate of plastics under composting conditions. The standard states that a plastic material can be considered compostable in the following cases: › minimum 60 % of the organic carbon is converted into carbon dioxide when the material contains only one type of polymer.

	› minimum 90 % of the organic carbon is converted into carbon dioxide when the material contains more than one co-polymer ((block copolymers, segmented copolymers, blends, or addition of low molecular weight additives).
EN 13432 Norm / EU standard for compostability	The European EN 13432 standard defines the requirements under which a plastic can be considered compostable, and can acquire the sidling logo (European bioplastics trademark) for compostable material. According to this standard, the following characteristics should be tested: › Biodegradability › Disintegration during biological treatment › Absence of negative effects on the composting process. › Amount of heavy metals For the biodegradability test, the plastic should be composted under industrial composting conditions for 26 weeks (6 months) at temperature of 58 ^{0}C and 50 % moisture. After that period, the biodegradability ratio defined in formula (4) should be higher than 90 %. The standard for this test is EN 14046 (also published as ISO 14855). The second test measures disintegration. Samples are placed in compost soil in sacks or bags. The conditions of the compost soil are moisture of at least 50 % and 58 ^{0}C temperature and the duration of the test is 12 weeks (84 days). After the end of the test, at least 90 % of the dry weight of the plastic should be able to pass through a 2mm sieve. EN 14040 is the used standard for this test. The third test specifies the ability of the compost to support plant growth by phytotoxicity and very low regulated heavy metal content. Phytotoxicity is tested by placing 2 seed in the compost soil. Presence of plants after 10 days indicates positive plant growth conditions. Heavy metals have to be below regulated maximum values at the end of the biodegradability test.
ASTM D 5526 – 94/ US Standard for biodegradability under anaerobic conditions	This standard is used to define the biodegradability of plastic material under anaerobic conditions (or accelerated landfill conditions). The plastic material should be placed in an anaerobic reactor along with anaerobic

	inoculum derived from a laboratory or full scale anaerobic digester operating at 35 ^{0}C. The experiment should proceed in mesophilic conditions with temperature of 35 ^{0}C and more than 30 % total solids in static nonmixed conditions. The experiment should include a positive reference i.e. a thin layer of cellulose that ensures that degradation conditions are fulfilled and that defines the end of the test. The test is considered over when more than 70 % of the positive control has degraded. A typical hydraulic retention time under mesophilic conditions is 25-30 days. Therefore, in order for a biodegradability outcome to be realistic, satisfactory biodegradation rates should occur within this period of time. For the interpretation of the test the Danish Teknologisk Institute mentions a value of at least a 50 % within 60 days.
ASTM D 5338-03 standard / US standard for biodegradability in contact with soil at aerobic conditions.	This standard is equivalent to ISO 17556:2003 and measures the degree and rate of aerobic biodegradation of plastic materials in contact with soil, or a mixture of soil and mature compost, under laboratory conditions. The latter is to determine the degradation of plastic residues found in compost that is afterwards spread on soil. The plastic under test is placed in contact with soil or a soil-mature compost mix at a ratio that corresponds to a typical agriculture application i.e. top soil: compost ratio. The sample are placed in the dark at 20±2 ^{0}C. The sample is considered biodegradable if >90 % of it is degraded in 2 years.
ISO 15985:2017	Determination of complete anaerobic biological degradation under dry conditions (test method for EN13432:2000)
ISO14853:2017	Determination of complete anaerobic biological degradation under wet conditions (test method for EN13432:2000)

ANNEX 3 MULTI-FUNCTIONALITY BIO-BASED PP CUPS/PACKAGING FILMS PRODUCT SYSTEM

In the modelling of the second alternative product system, the multifunctionality of hydrotreatment and steam cracking unit processes has been solved through energy-exergy allocation. Hence, the environmeantal burdens are allocated among co-products according to their energy-exergy output shares. The considered product system is stricky linked to biofuels production: indeed, only a very minor fraction of the collected UCO will be converted in bio-based propylene while the major fraction will be the precursor of the hydrotreated oil sold to the market as renewable diesel. This link with biofuels production is the main reason for which energy allocation has been chosen. Allocation by energy content, specifically the lower heating value of the products, is recommended in the Renewable Energy Directive (RED) for calculating GHG emissions savings from the use of biofuels and in co-production of refineries. However, the European Commission's impact assessment of the 2009 Renewable Energy Directive also states that for the purpose of policy analysis the substitution method is more suitable in biofuels emissions calculations and a marginal calculation for oil refineries (COM, 2008). Energy allocation is considered by the RED "the most appropriate method because it is easy to apply, is predictable over time, minimizes counter-productive incentives and produces results that are generally comparable with those produced by the substitution method". With respect to this conversion route, this is true when renewable diesel is the investigated product obtained from UCO: Thamsiriroj et al. applied energy allocation, a no-allocation approach and the substitution approach to bio-diesel from UCO and concluded that "UCO biodiesel is relatively unaffected by allocation methodology" (Thamsiriroj & Murphy, 2011). The same pathway was also investigated within the Wheel to Wheel model of the Joint Research Centre (JEC (2014)) and a no-allocation approach was followed ("the very small naphtha production is neglected"). Unfortunately, the RED is focused only on GHG emissions and not on all the environmental impacts assessed according to PEFCR guidelines. The aforementioned statement is not referred to all the 16 impact categories and it can not be extended to all them without caution: e.g. it is not assured that assigning abiotic depletion or water use through energy allocation to the different co-products would lead to results comparable with those produced by substitution. In our case, the investigated product is bio-based polypropylene. Bio-based polypropylene is derived from the minor production of naphtha and the use of the substitution method would lead to non comparable results with those produced by the energy allocation method especially if bio-based ethylene is substituted with commercial or petrochemical ethylene. The main reason for this is that the focus is a physically non-dominant product (bio-based PP is produced with a lower yield compared to bio-based ethylene): this implies that the choice of the allocation method deserves much more attention because not enough emphasised in the current guidelines such as ISO 14044 and ILCD handbook (Sandin et al., 2015). Moreover, there are also other critical substitution choices that affect significanltly the results.

In this section, the variation of the impact assessment results -when system expansion is applied consistently with the hierarchy reported in section *Procedure of multi-functionality processes*- is shown. Three different allocation approaches allowed by ISO 14044 are compared:

- **Energy/Exergy allocation.** This is the approach applied that has been applied to the alternative 2 product system (bio-based PP cups). Energy allocation has been applied in hydrotreatment and steam cracking unit processes.

- **Cut-off.** Due to the very minor production of bio-based naphtha compared to renewable diesel, the impact of UCO collection and hydrotreatment are assigned only to

renewable diesel. Bio-based naphta becomes an 'emissions free' by-product obtained from renewable diesel production because of the applied cut-off.

- **Substitution of co-products.** System expansion is applied to co-products produced during bio-based PP production. More in detail the following unit processes and co-products are expanded:

    - Hydrotreatment. The process used for this substitution of petrochemical propane is Propane {GLO}| market for | Conseq Def, U. The electricity production from this propane is outside the system boundaries.

    - Steam cracking. This case of multi-functionality is modelled through system expansion with substitution as follow:

        - The amount of heat produced has been estimated to be 5.12 MJ per kg of propylene produced (based on Karimzade et al. (Karimzade et al. 2009)). This is assumed to substitute industrial heat (Heat, in chemical industry {RER}| market for | Conseq Def, U of Ecoinvent 3.3).

        - Hydrogen by-product (assumed to be the 1.68 % in weight of the cracked gases (Karimzadeh, 2009)) is substituted by hydrogen obtained through steam reforming of natural gas. Hydrogen (reformer) E belonging to Industry 2.0 dataset of PlasticsEurope has been used for this substitution.

        - Methane and other light gases have been substituted by Natural gas, from medium pressure network (0.1-1 bar), at service station {GLO}| market for | Conseq Def, U.

        - Benzene has been substituted with Benzene, at plant/RER based on PlasticsEurope Industry 2.0 database.

        - Bio-based ethylene is assumed to substitute conventional ethylene (Ethylene, at plant/RER of Industry 2.0 dataset of PlasticsEurope).

Figure 215 clearly shows that energy allocation, cut-off approach and the substitution of the co-products for the alternative 2 product system lead to similar results only in several impact categories. When system expansion is applied also to ethylene, large differences are shown compared to the other allocation approaches.

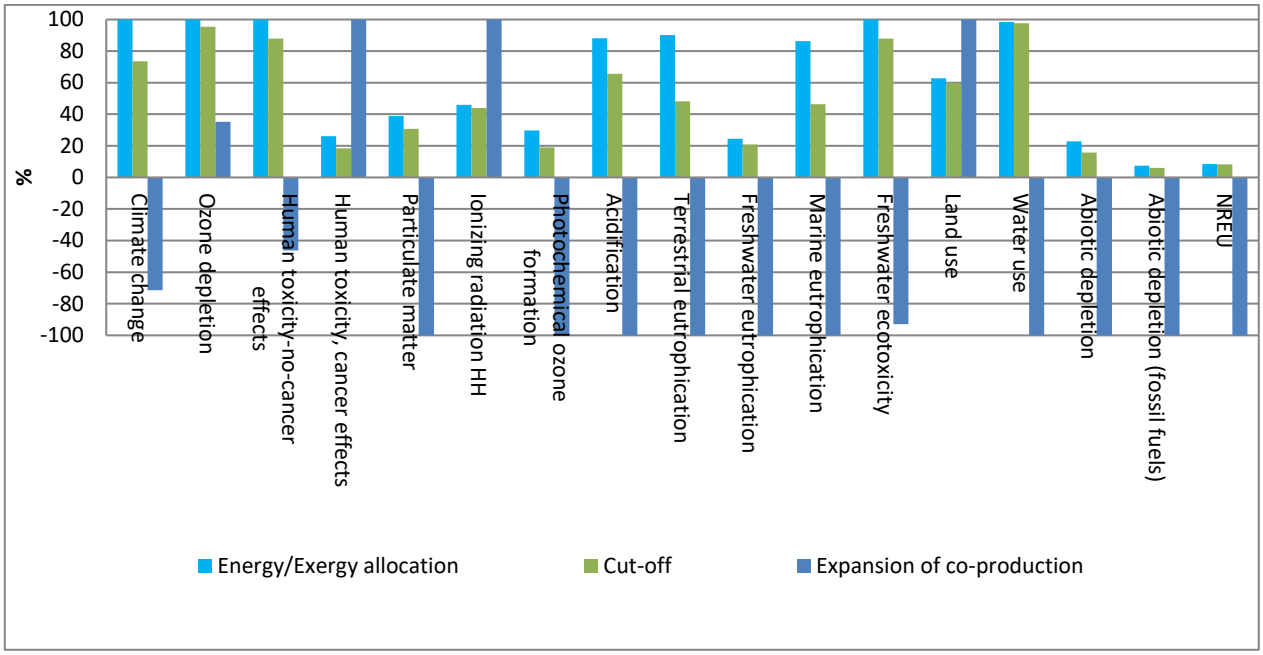

Figure 215. Multifunctionality of bio-based PP production (results variations per kg of bio-based PP)

ANNEX 4 ADDITIONAL RESULTS FOR END OF LIFE

In this annex the following abbreviations are used in the figures.

Impact Category	Unit	Abbreviation
Global Warming Potential (GWP) 100a	kg CO_2 eq.	GWP 100a fossil
Ozone depletion	kg CFC-11eq	Ozone depl.
Human toxicity, non-cancer effects	CTUh	Human tox. non-cancer
Human toxicity, cancer effects	CTUh	Human tox. cancer effc.
Particulate matter	kg PM2.5eq	Particulate matter
Ionizing radiation HH	kBq U235eq	Ionizing rad. HH
Photochemical ozone formation	kg NMVOCeq	Photochemical ozone
Acidification	molc H+eq	Acidification
Terrestrial eutrophication	molc Neq	Terrestrial eutroph.
Freshwater eutrophication	kg Peq	Freshwater eutroph.
Marine eutrophication	kg Neq	Marine eutroph.
Freshwater ecotoxicity	CTUe	Freshwater ecotox.
Land transformation*	kg C deficit	Land use
Water use	m3	Water use
Abiotic depletion	kg Sbeq	Abiotic depl.
Abiotic depletion (fossil fuels)	MJ	Abiotic depl. fossil fuels
Non-renewable energy use. (NREU)	MJ	NREU

Grouped processes

In several figures with EoL results grouped processes have been utilised to present the results. The grouped processes represent the following;

EoL Treatment	Results summed to	Includes these processes
Plastic recycling	Transportation	Transport to facility and rejects to incineraton
	Plastic recycling plant	incl. treatmen of waste water
	Treatment of rejects	Direct and indirect emissions
	Plastic substitution	
	Heat substitution	from incineration of rejects
	Electricity substitution	from incineration of rejects
MSWI	Transportation	Transport to facility and ash to further treatment
	Incineration plant, direct emissions	
	Fly ash treatment and utilization	
	Bottom ash treatment and utilization	
	Heat substitution	
	Electricity substitution	
Landfill	Transportation	Transport to landfill
	Landfill construction and operation	
	Uncollected leachate	Direct emisions to surface water
	Leachate treatment and utilization	Treatment at waste water treatment plant with biogas production
	LFG not collected, oxidation in covers	Direct emissions of LFG to air, considering oxidation in different covers
	LFG combustion, direct emissions	Flared and direct emissions from upgrading gas

	Electricity substitution	from gas
Composting	Transportation	Transport to facility and rejects to incineraton
	Industrial composting plant	Direct and indirect emissions
	Treatment of rejects	Incineration - direct and indirect emisisons
	Use on land	Directs emisisons to air and soil
	Fertiliser substitution	N, P and K substitution
	Heat substitution	from incineration of rejects
	Electricity substitution	from incineration of rejects
AD	Transportation	Transport to facility and rejects to incineraton
	Anerobic digestion plant	All types of AD plants + post-maturation
	Treatment of rejects	Incineration - direct and indirect emisisons
	Biogas combustion and upgrading	the direct emissions
	Use on land	Direct emisisons to air and soil from both compost and digestate
	Fertiliser substitution	N, P and K substitutione from both compost and digestate
	Fuel substitution	from biogas, includes direct emisisons from the upgrading process
	Heat substitution	from incineration of rejects and biogas
	Electricity substitution	from incineration of rejects and biogas
In-situ	Avoided collection	
	Use on land	Direct emisions to air and soil

Contribution analysis

The figures below show the contribution analysis for each EoL treatment for all plastic materials modelled in each Case study. The contribution analysis presents how much a process contributes to the overall results. The numbers are in absolute values and it should be observed that the real values are both positive and negative.

The results are for incineration presented with and without energy recovery. The results of incineration without energy recovery can be interpreted by excluding the energy substitution (heat and electricity), and for incineration with energy substitution the energy substitution shall be multiplied with 1.65 (factor between amount of waste treated in incineration plants with and without energy recovery, see Approach and methodology chapter section 3.4.5 Waste incineration).

Case study 1: Beverage bottles

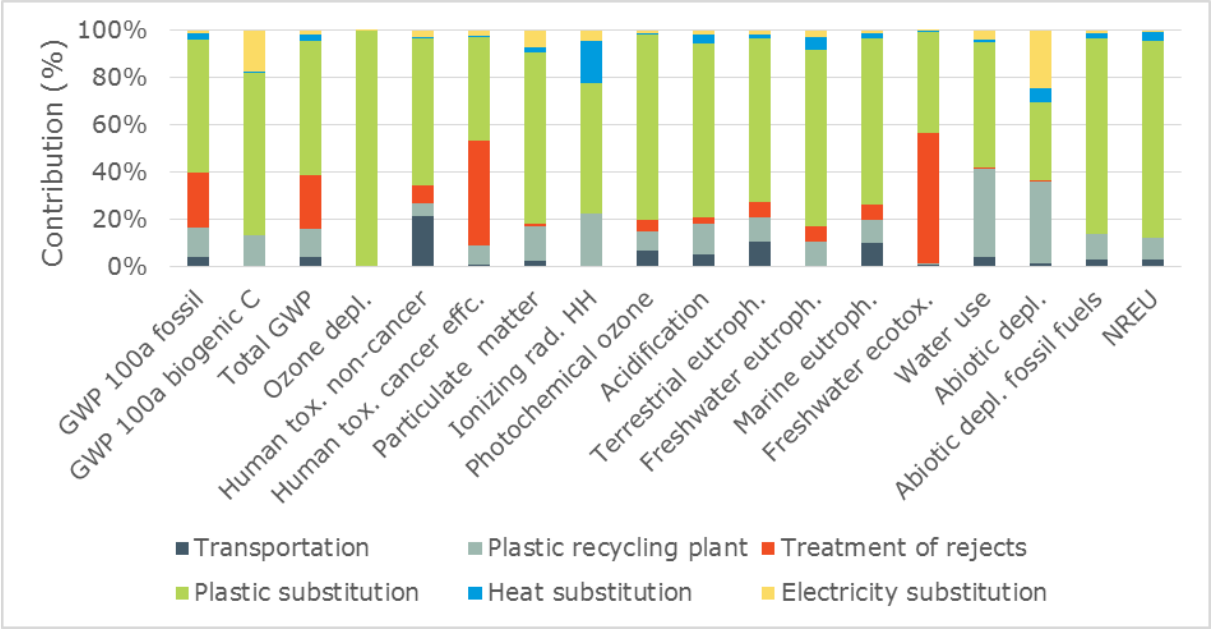

Figure 216. Contribution analysis, Recycling, Petrochemical PET in absolute percentage.

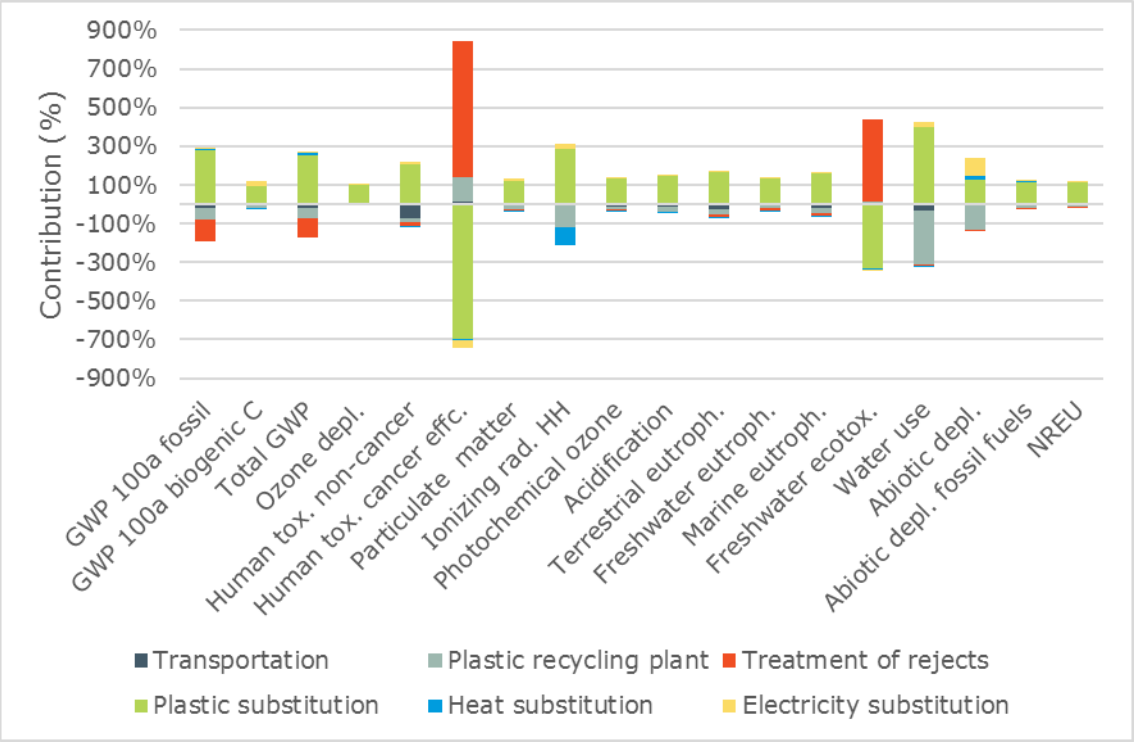

Figure 217 Contribution analysis, Recycling, Petrochemical PET.

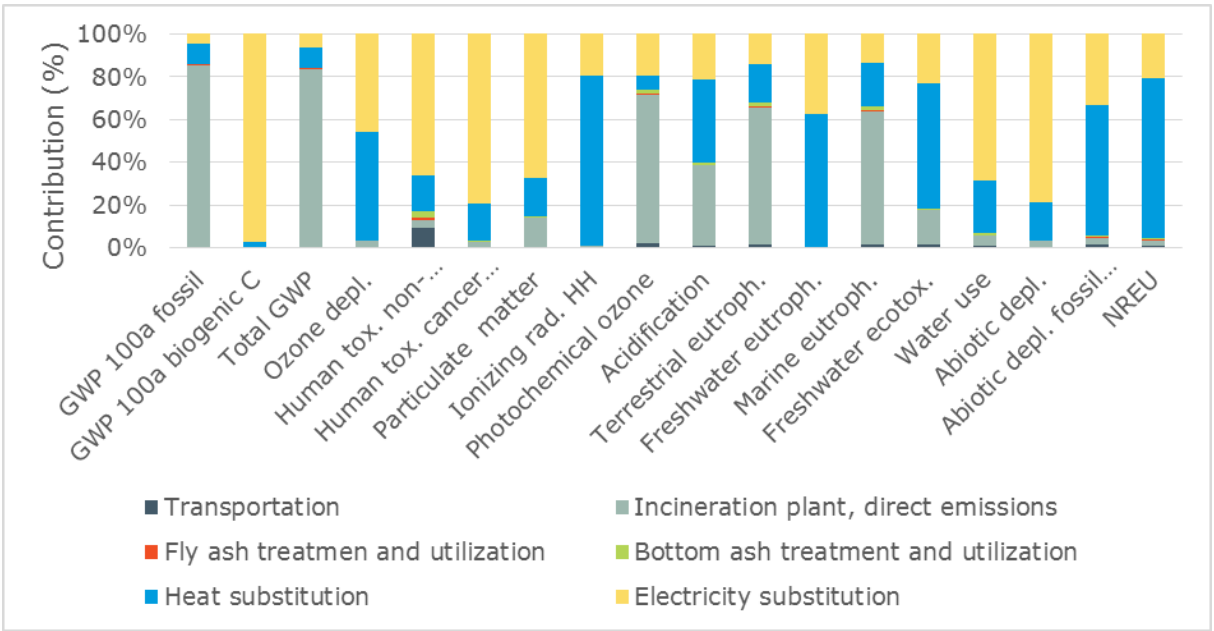

Figure 218. Contribution analysis, Incineration, Petrochemical PET in absolute percentages.

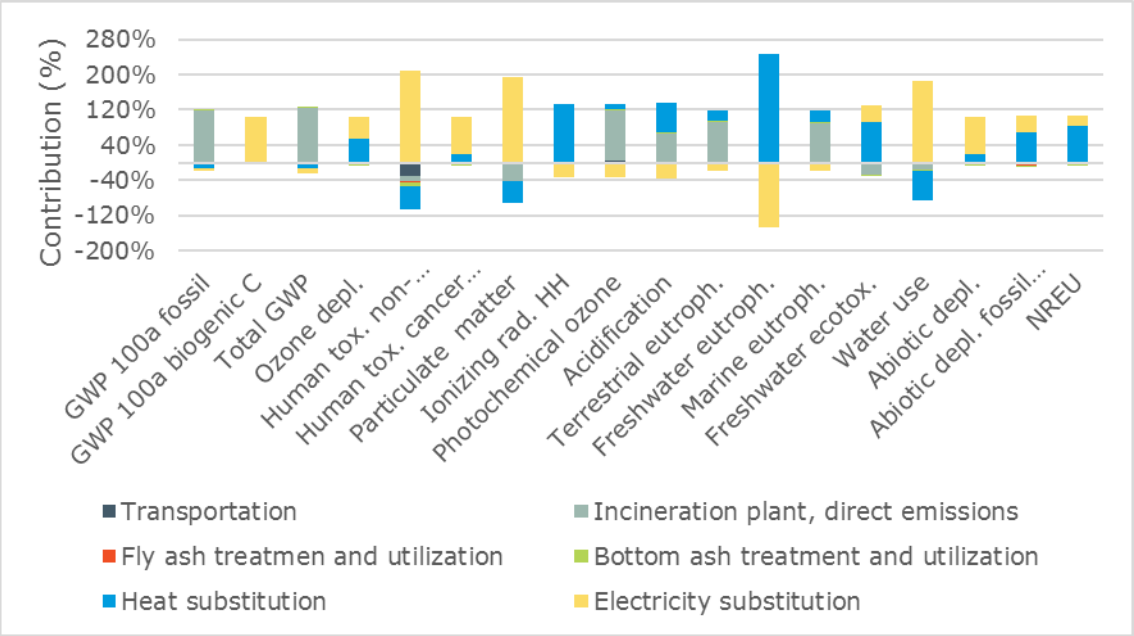

Figure 219 Contribution analysis, Incineration, Petrochemical PET.

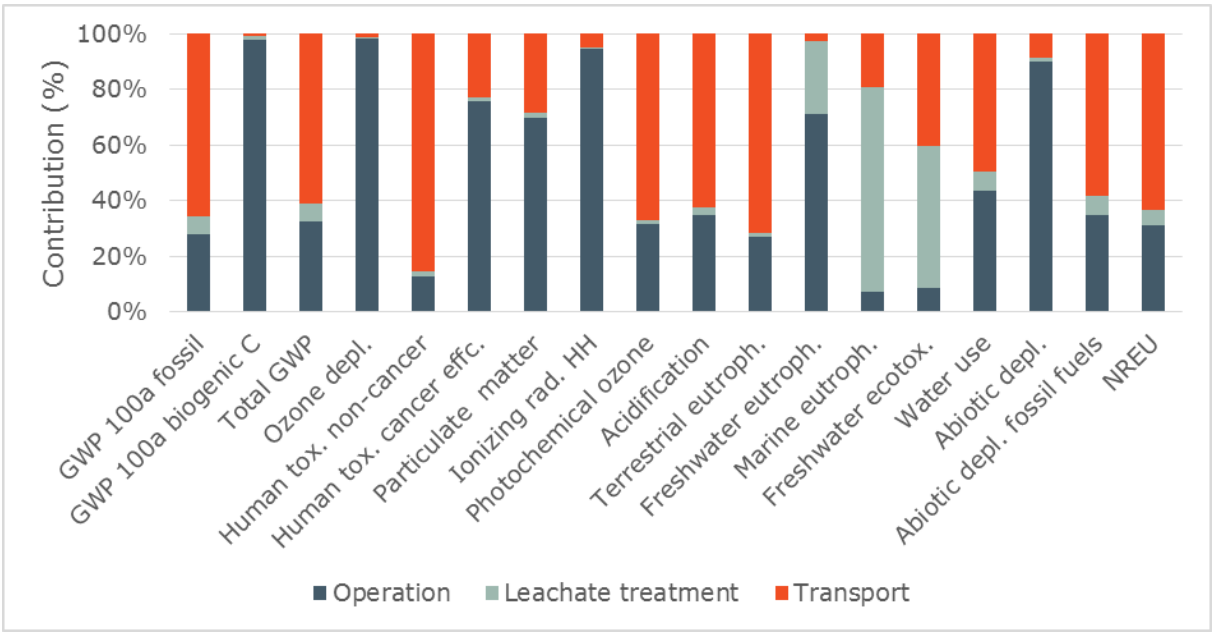

Figure 220. Contribution analysis, Landfill, Petrochemical PET.

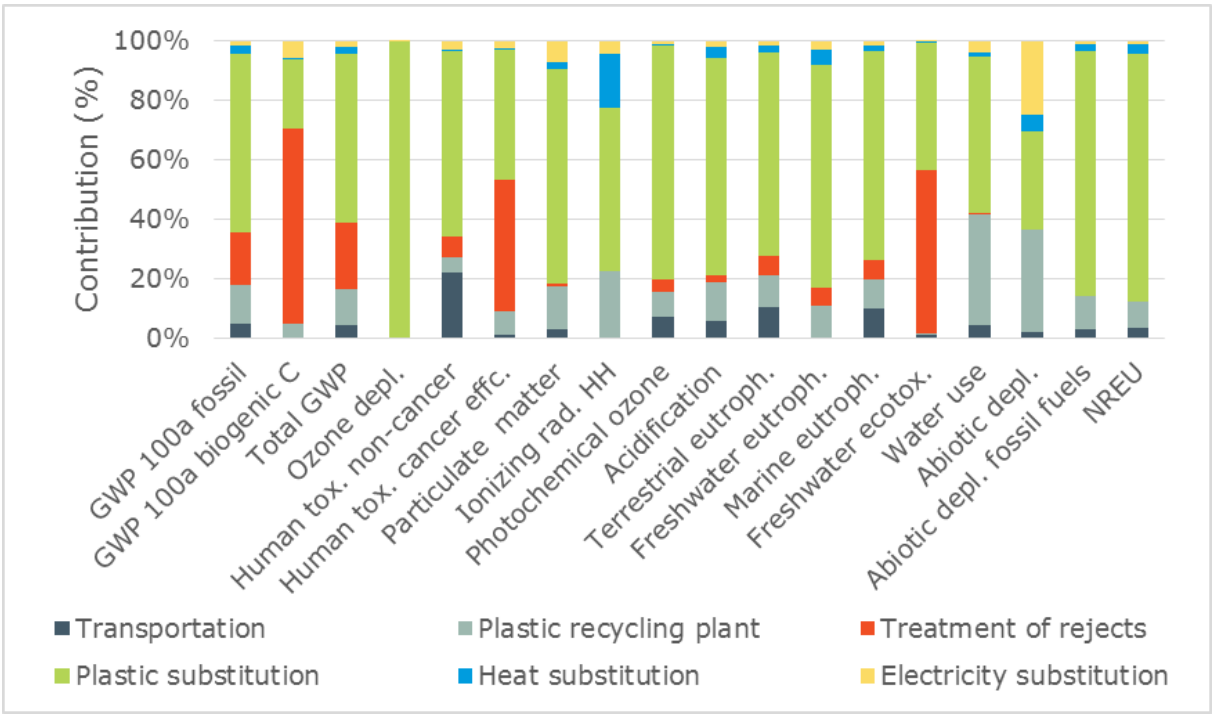

Figure 221. Contribution analysis, Recycling, Bio-based PET in absolute percentages.

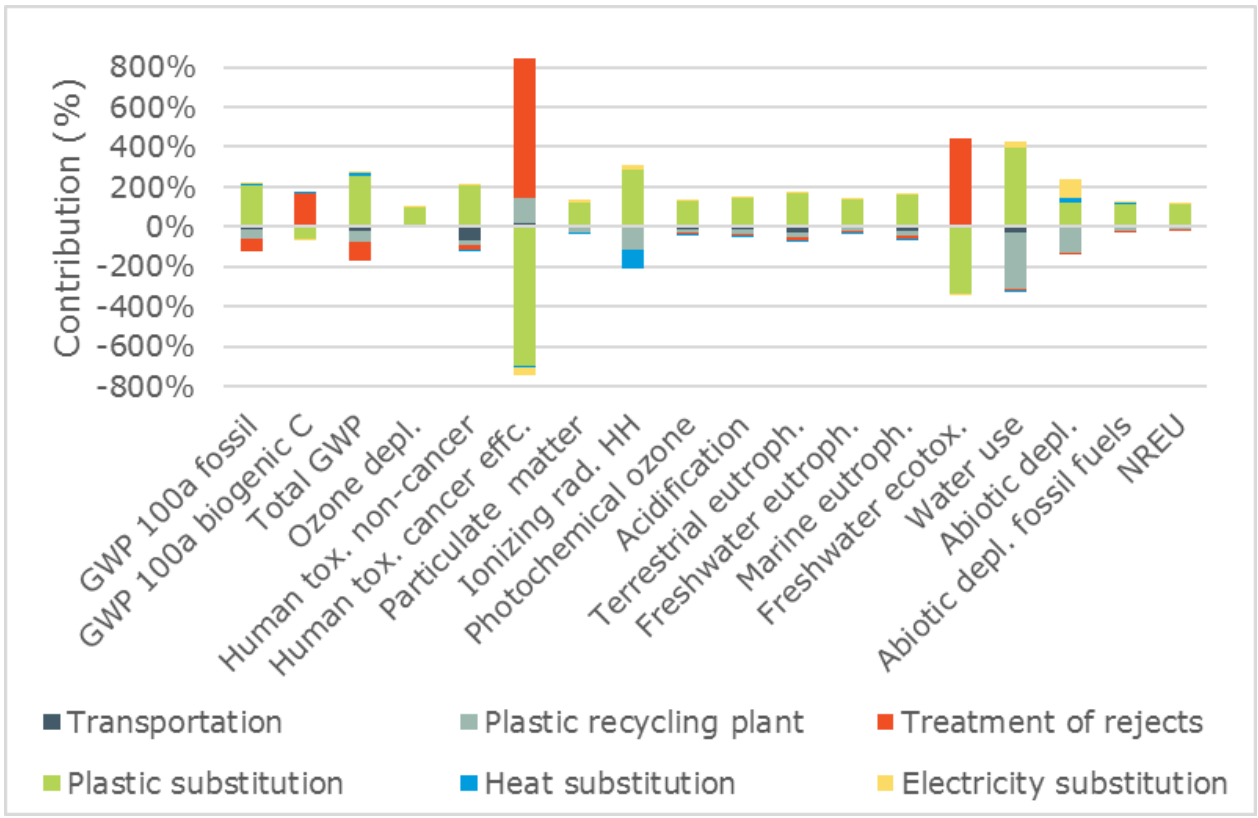

Figure 222. Contribution analysis, Recycling, Bio-based PET.

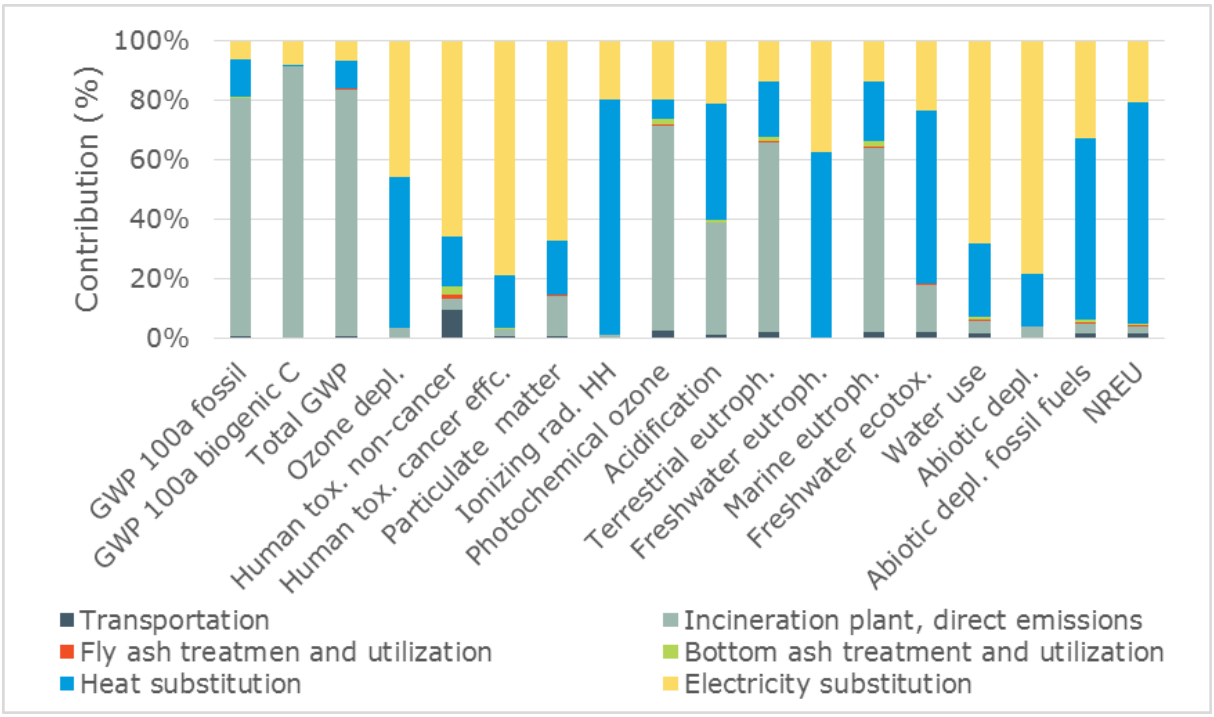

Figure 223. Contribution analysis, Incineration, Bio-based PET in absolute percentages.

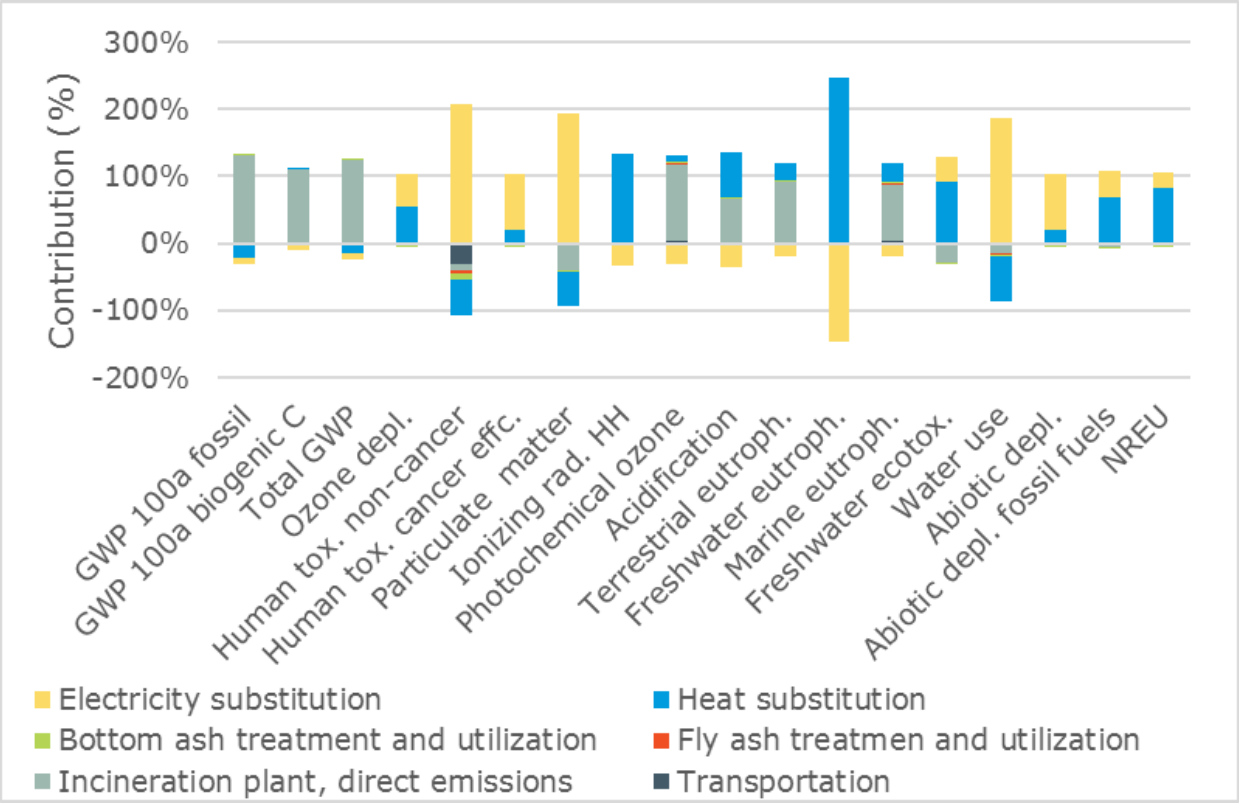

Figure 224. Contribution analysis for incineration of biobased PET.

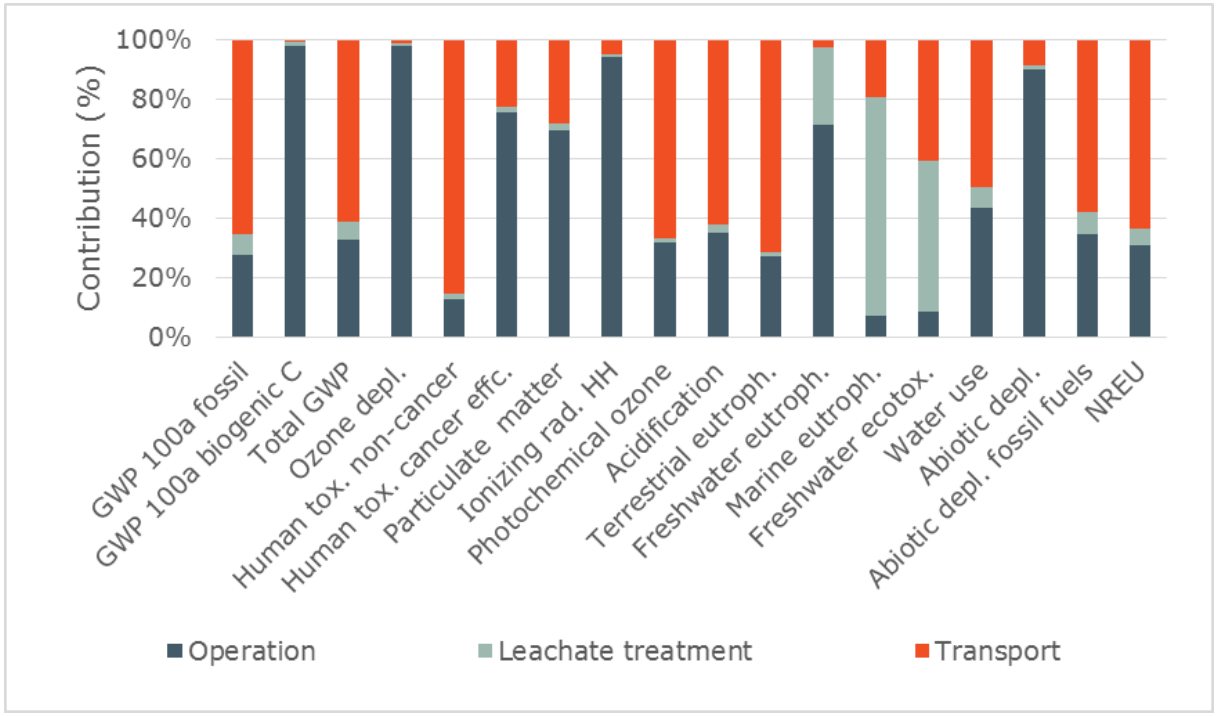

Figure 225. Contribution analysis, Landfill, Bio-based PET.

Case study 2: Single-use clips

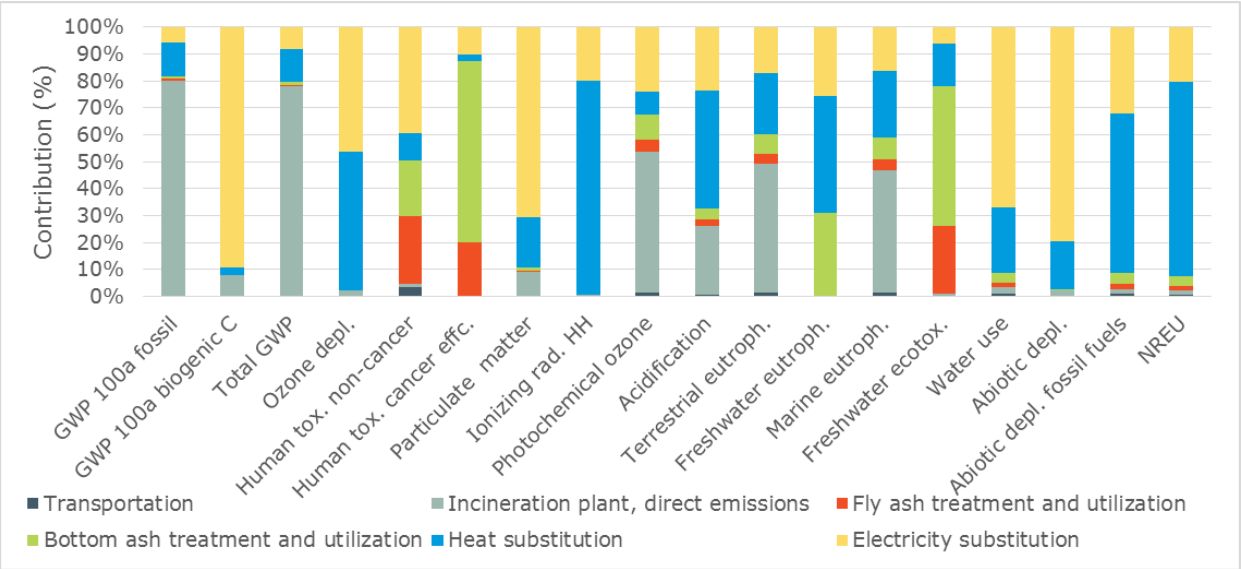

Figure 226. Contribution analysis, Incineration, PP, excl. soil (in absolute percentages).

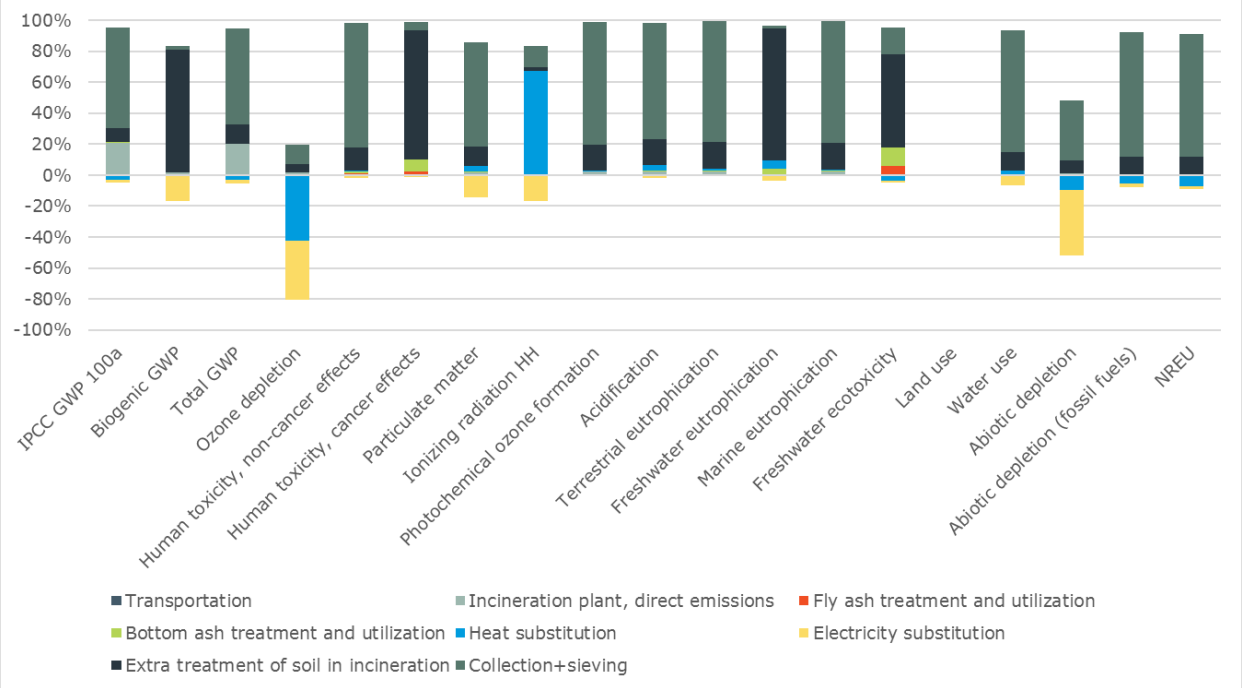

Figure 227. Contribution analysis, Incineration, PP, including soil.

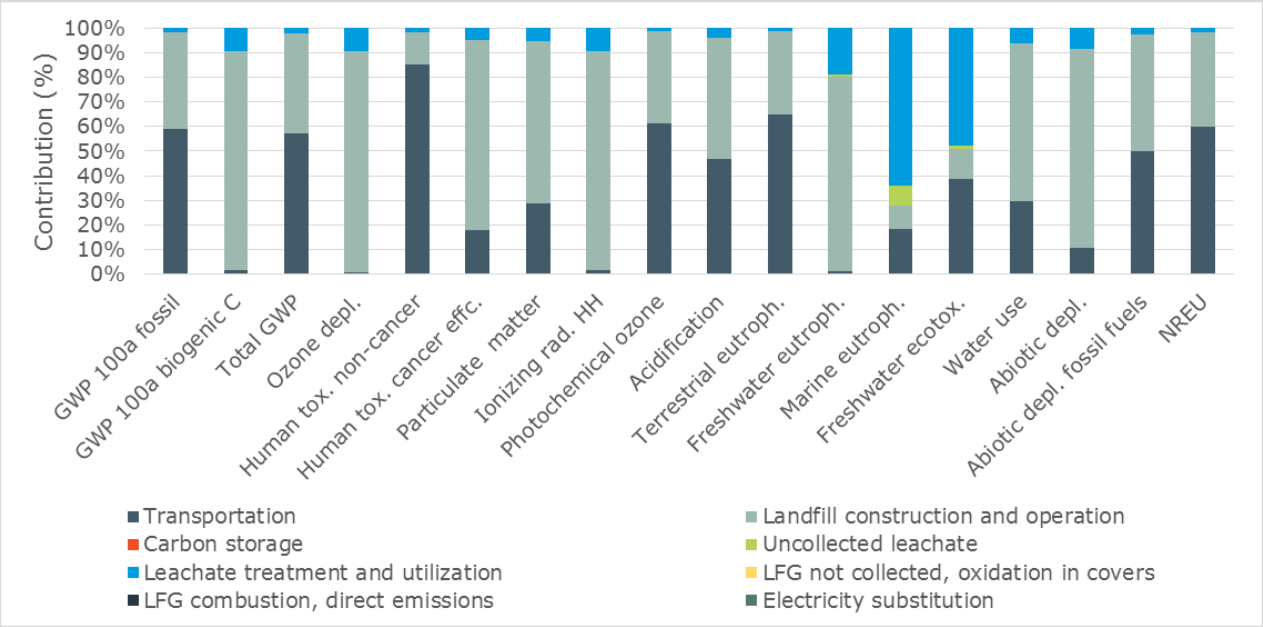

Figure 228. Contribution analysis, Landfill, PP, excl. soil (absolute percentages).

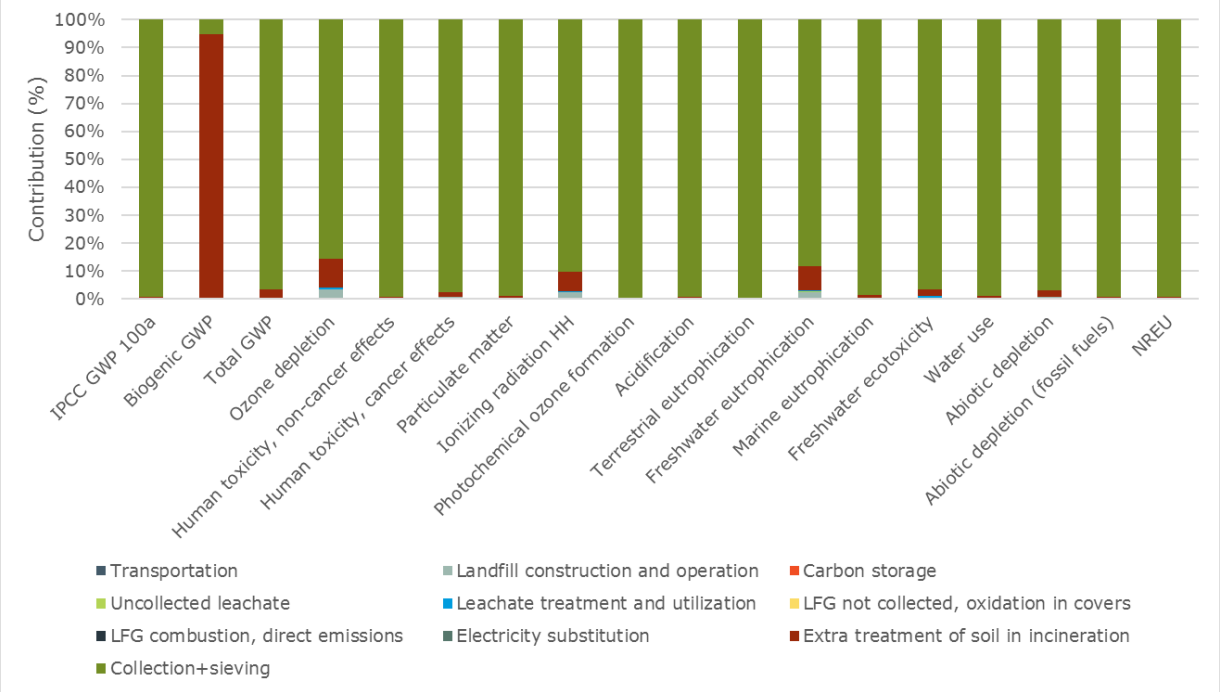

Figure 229. Contribution analysis, Landfill, PP, including soil.

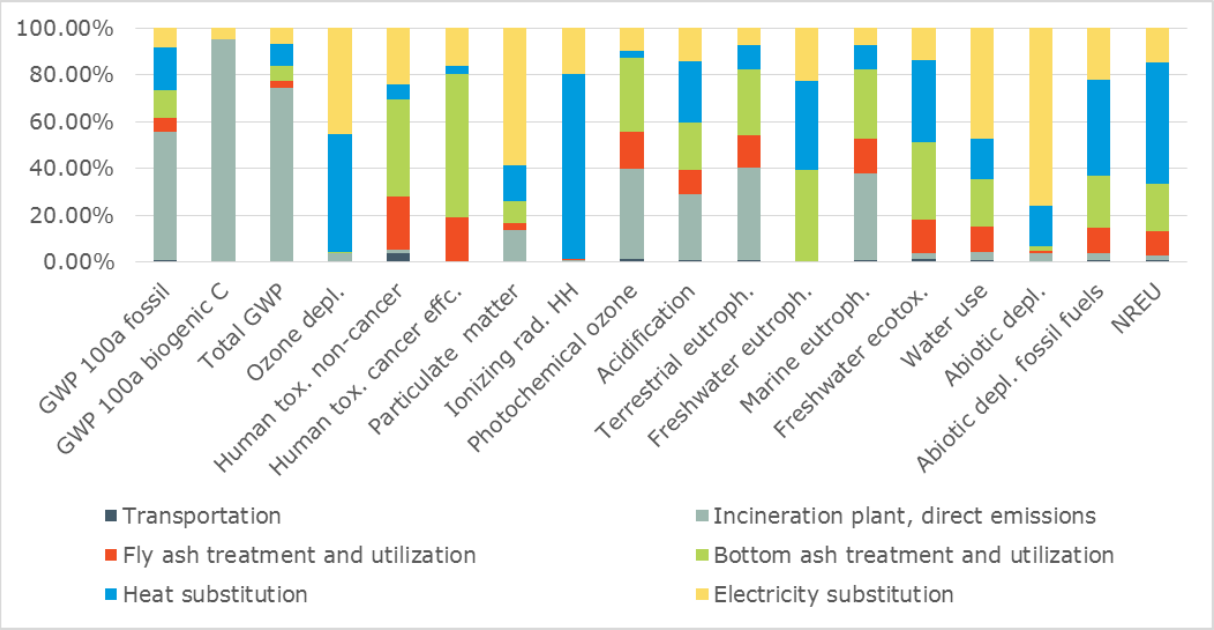

Figure 230. Contribution analysis, Incineration, Starch, excl. soil (absolute percentages).

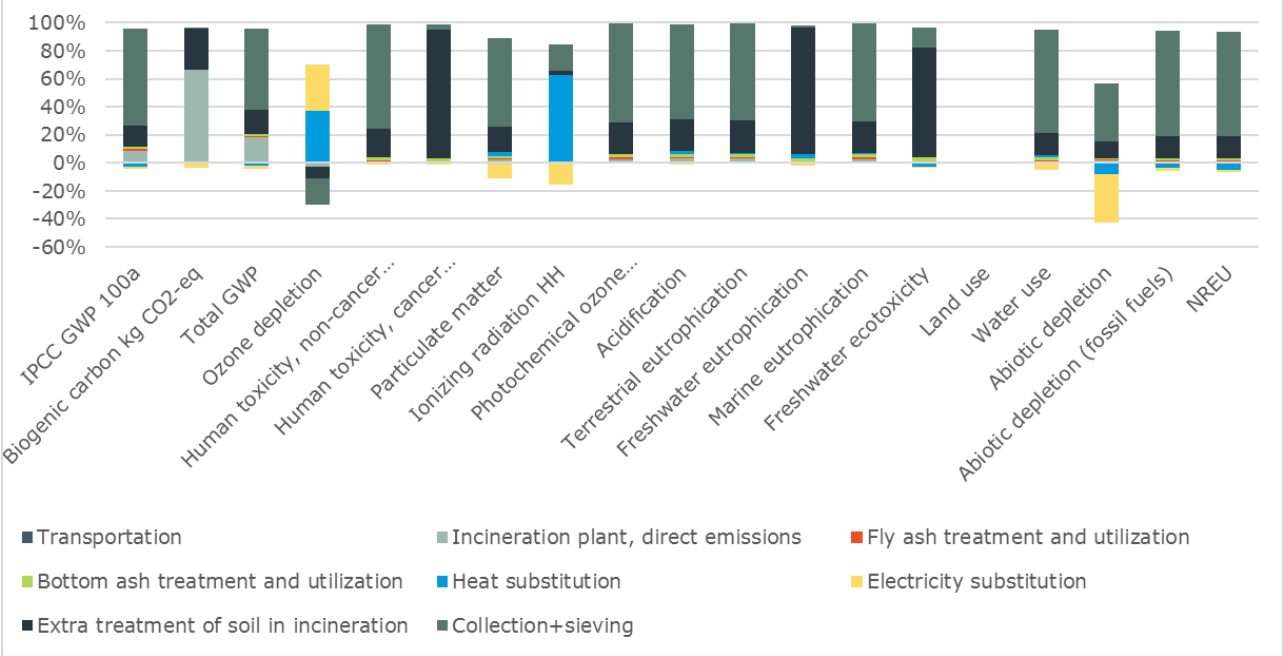

Figure 231. Contribution analysis, Incineration, Starch, including. soil.

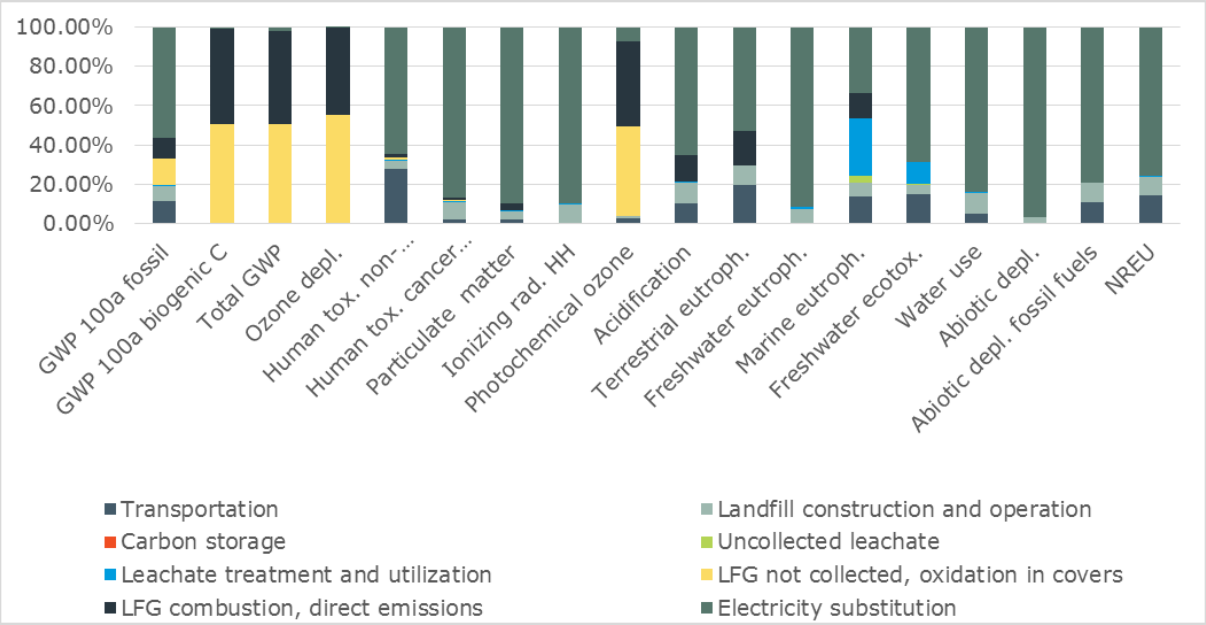

Figure 232. Contribution analysis, Landfill, Starch, excl. soil. (absolute percentages).

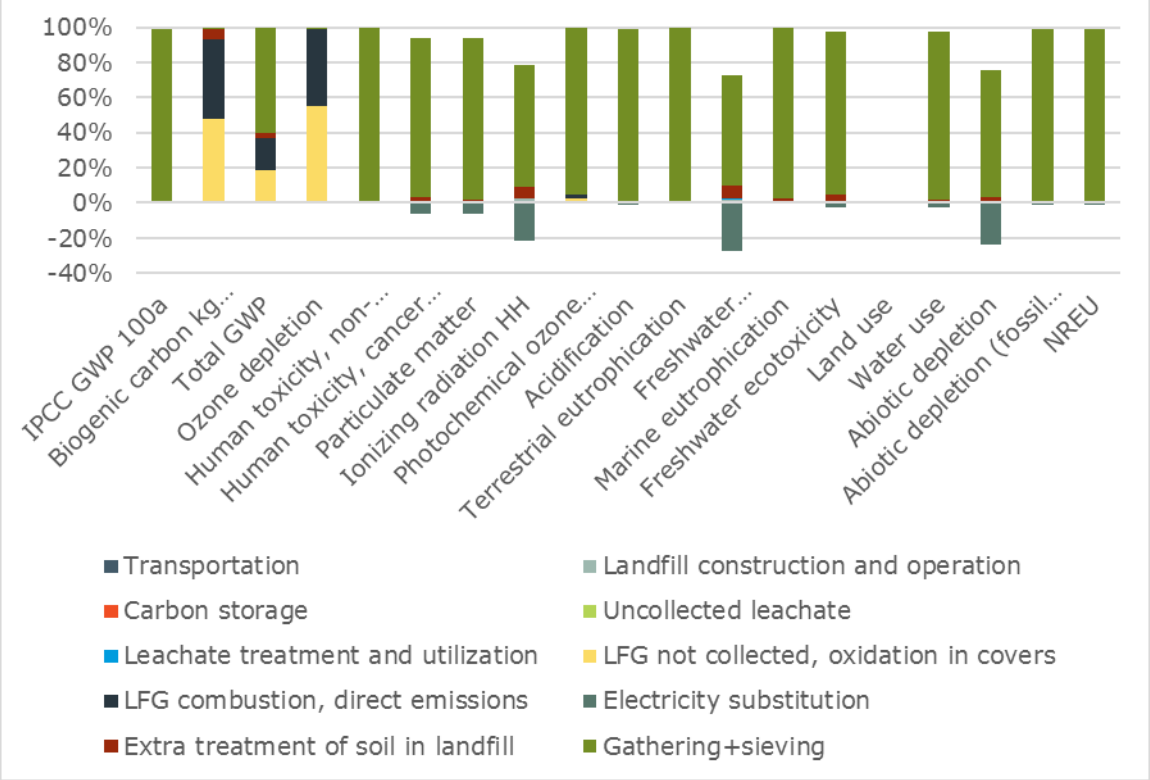

Figure 233. Contribution analysis, Landfill, Starch, including soil.

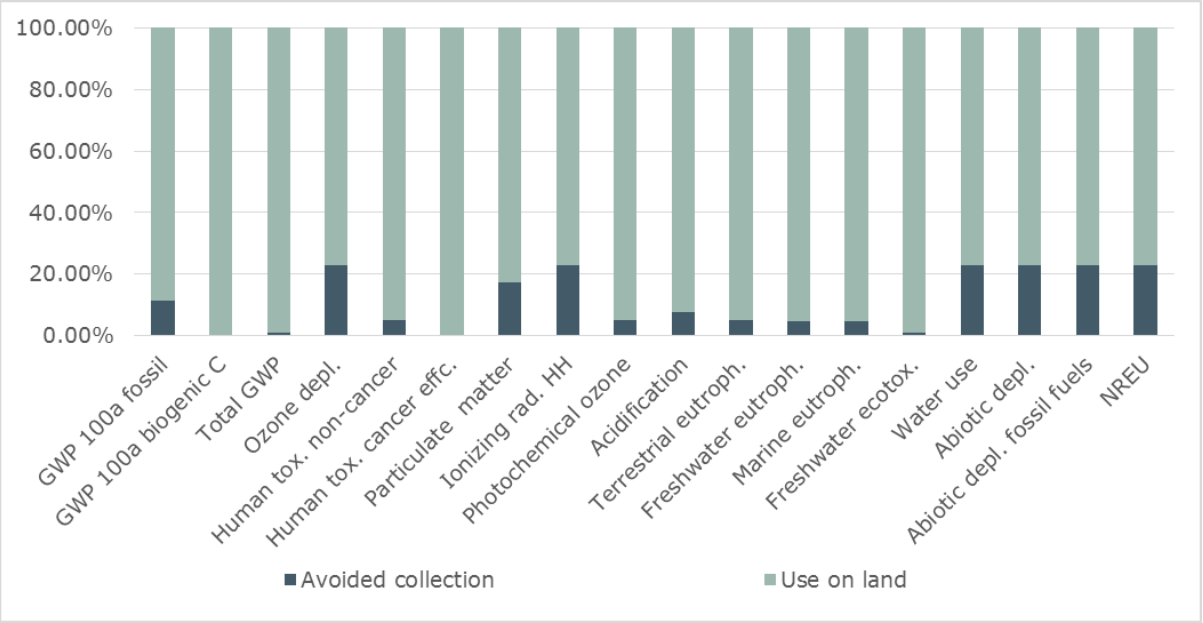

Figure 234. Contribution analysis, In-situ, Starch.

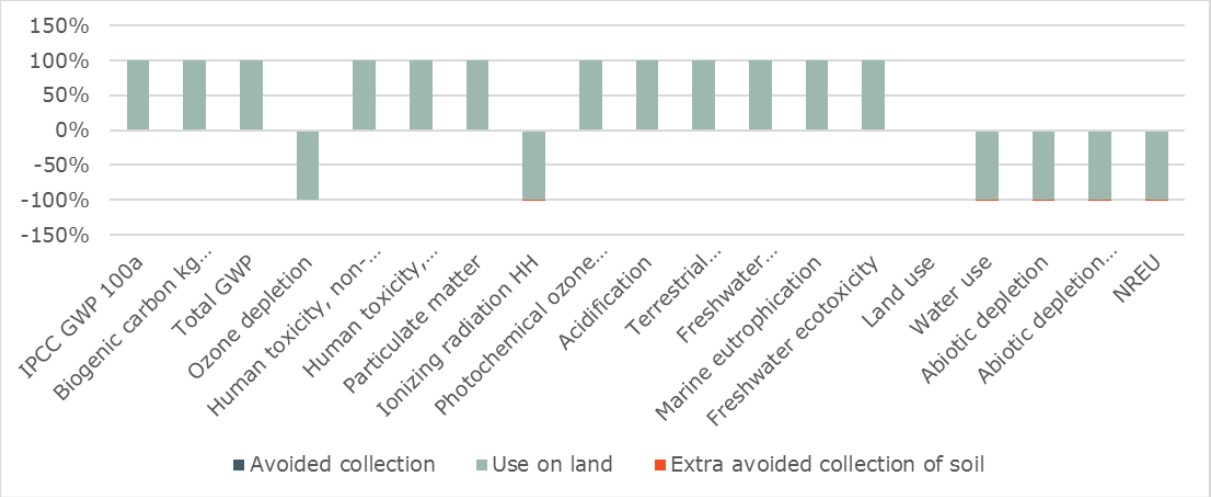

Figure 235. Contribution analysis, In-situ, Starch (including soil).

Case study 3: Single-use cups

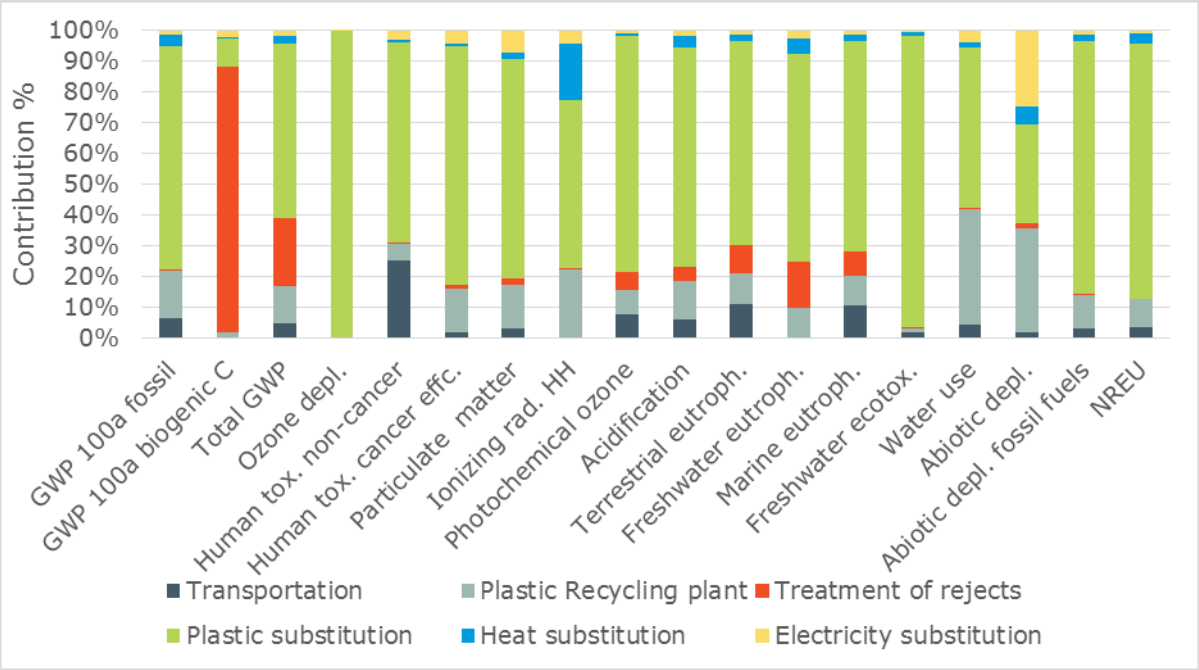

Figure 236. Contribution analysis, Plastic recycling, PLA (absolute percentages).

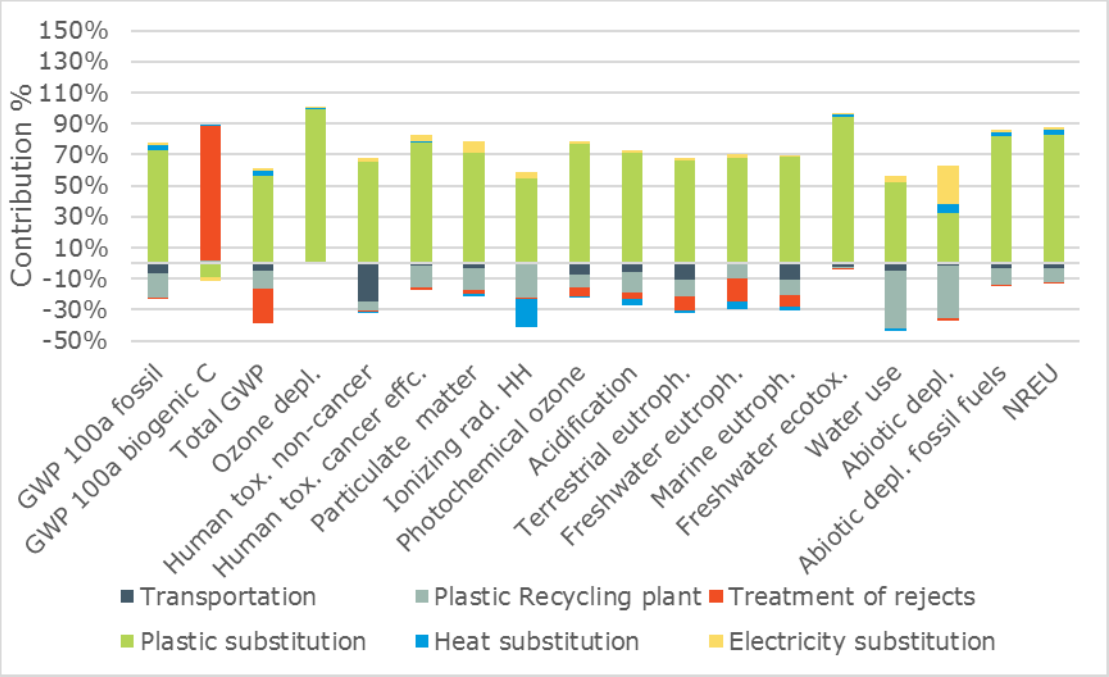

Figure 237. Contribution analysis, Plastic recycling, PLA.

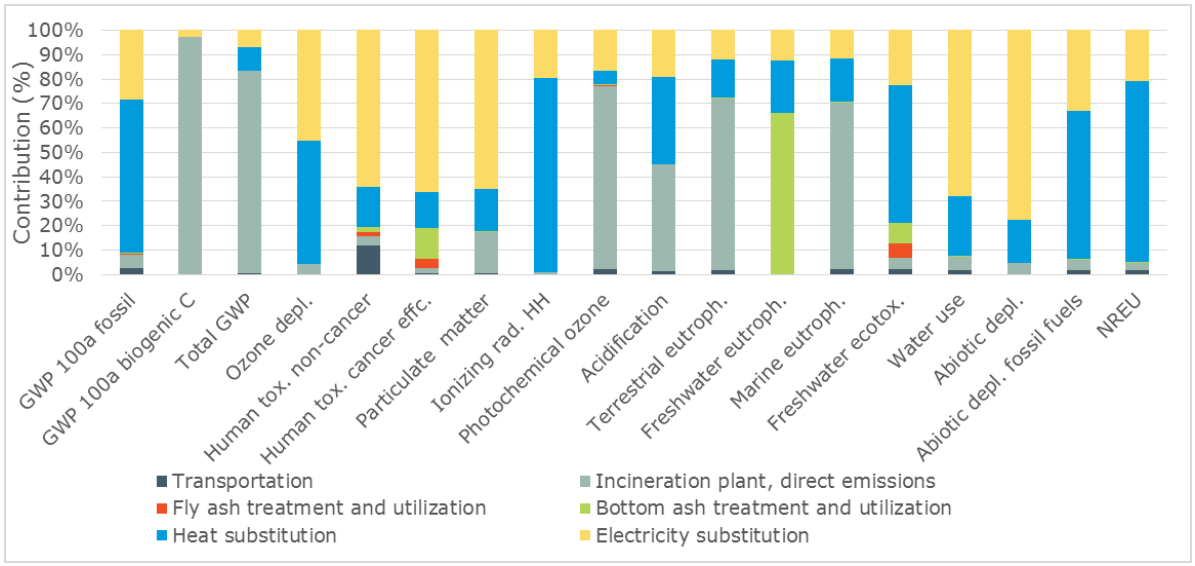

Figure 238. Contribution analysis, Incineration, PLA (absolute percentages).

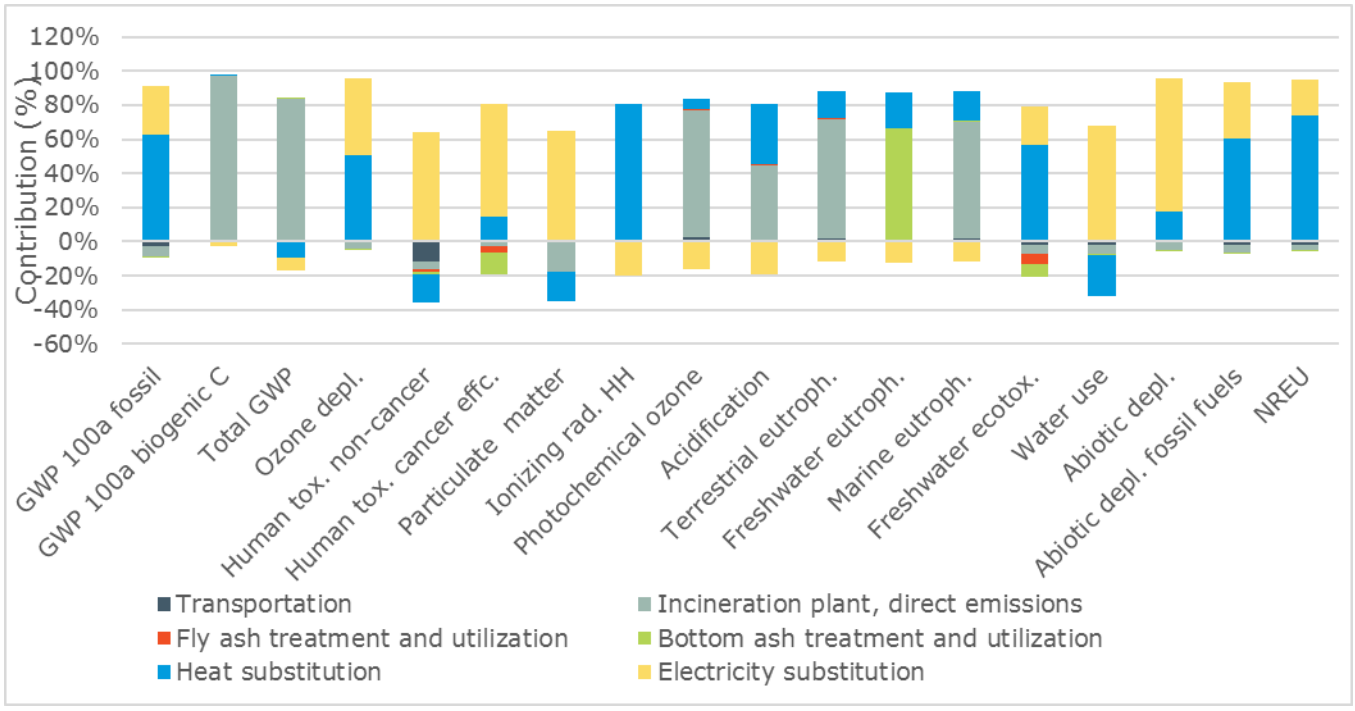

Figure 239. Contribution analysis, Incineration, PLA.

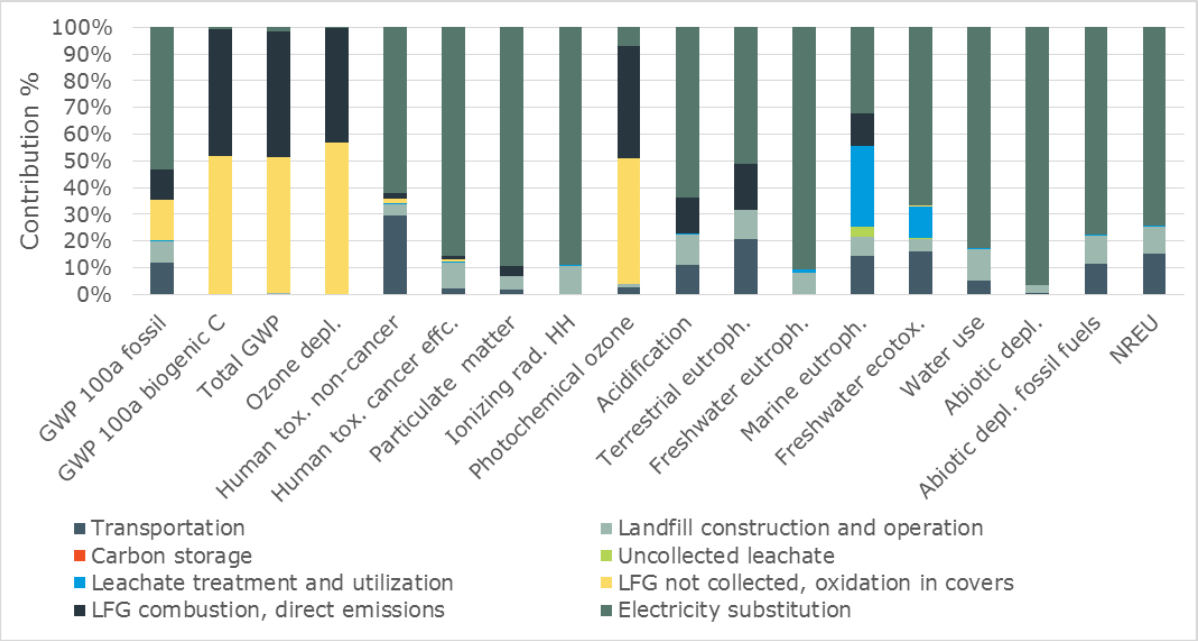

Figure 240. Contribution analysis, Landfill, PLA (absolute percentages).

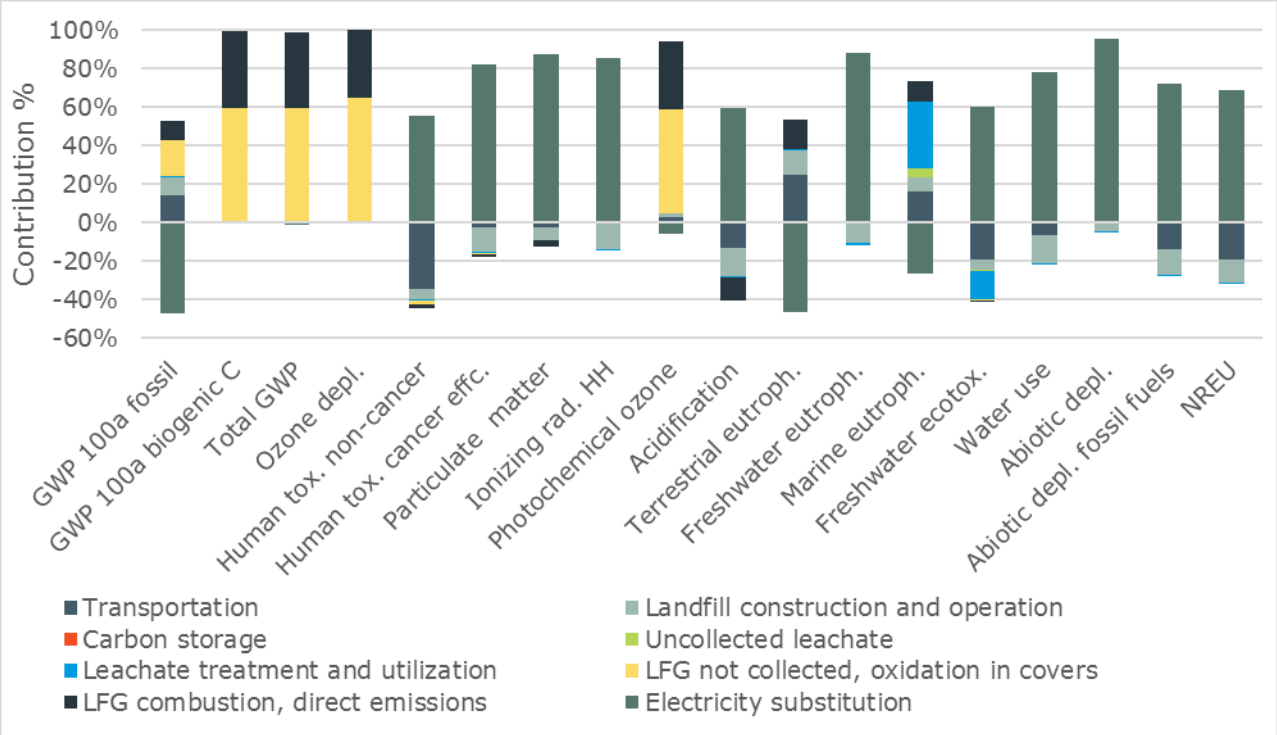

Figure 241. Contribution analysis, Landfill, PLA

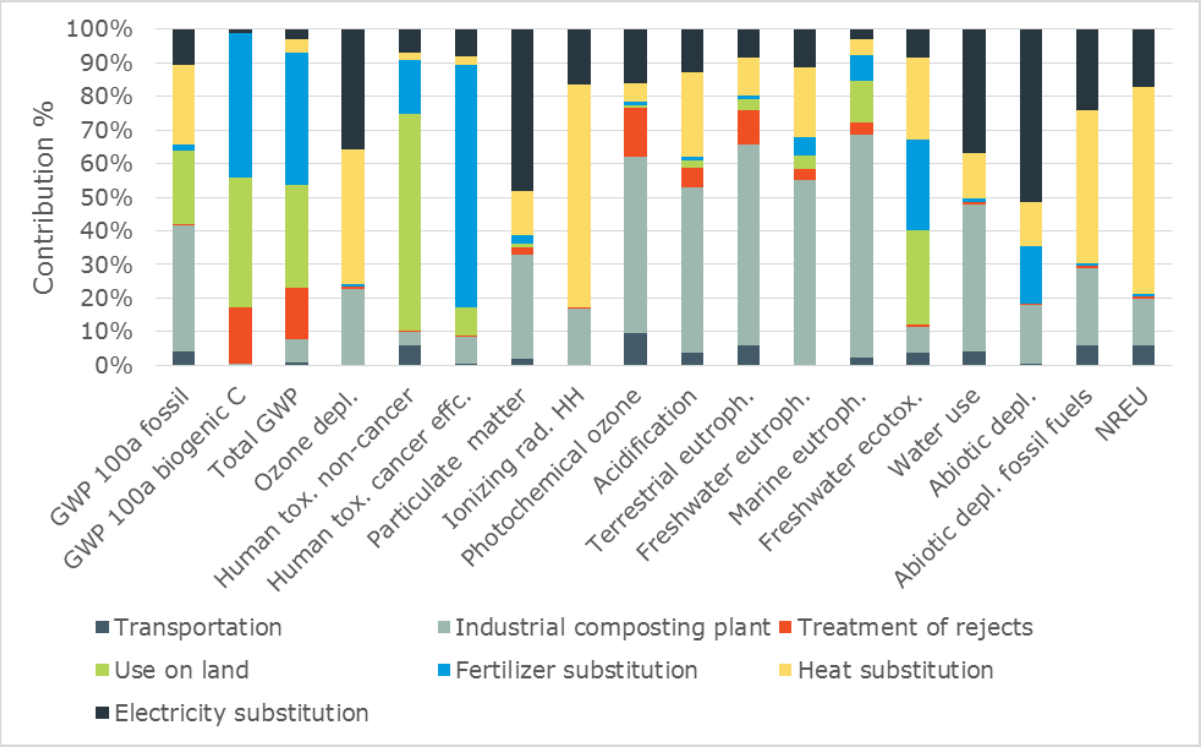

Figure 242. Contribution analysis, Industrial composting, PLA (absolute percentages).

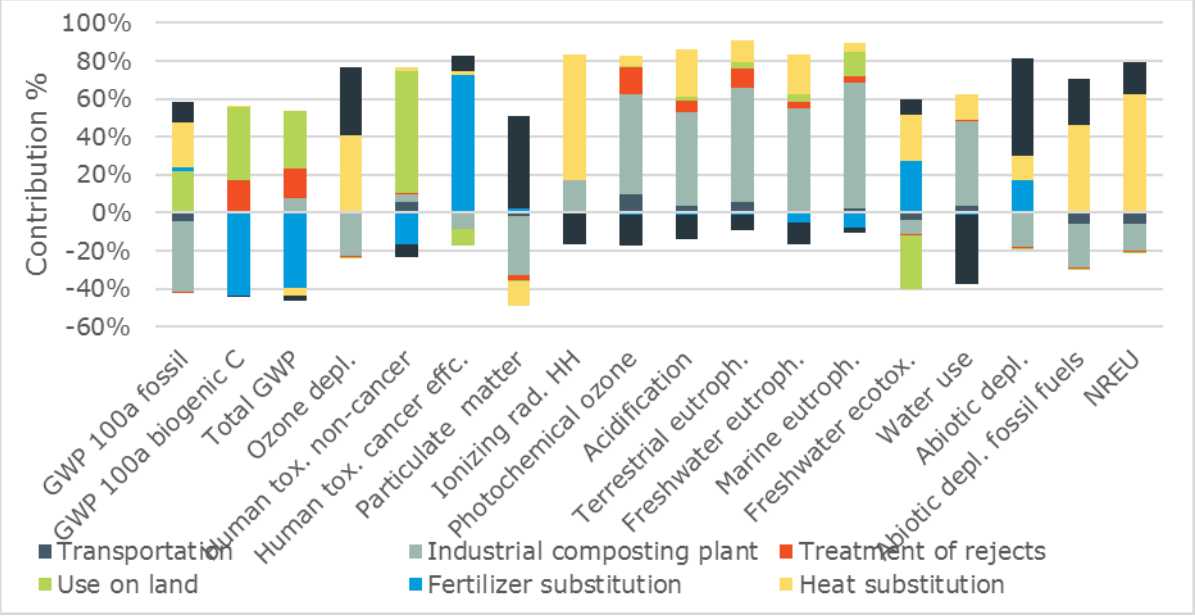

Figure 243. Contribution analysis, Industrial composting, PLA.

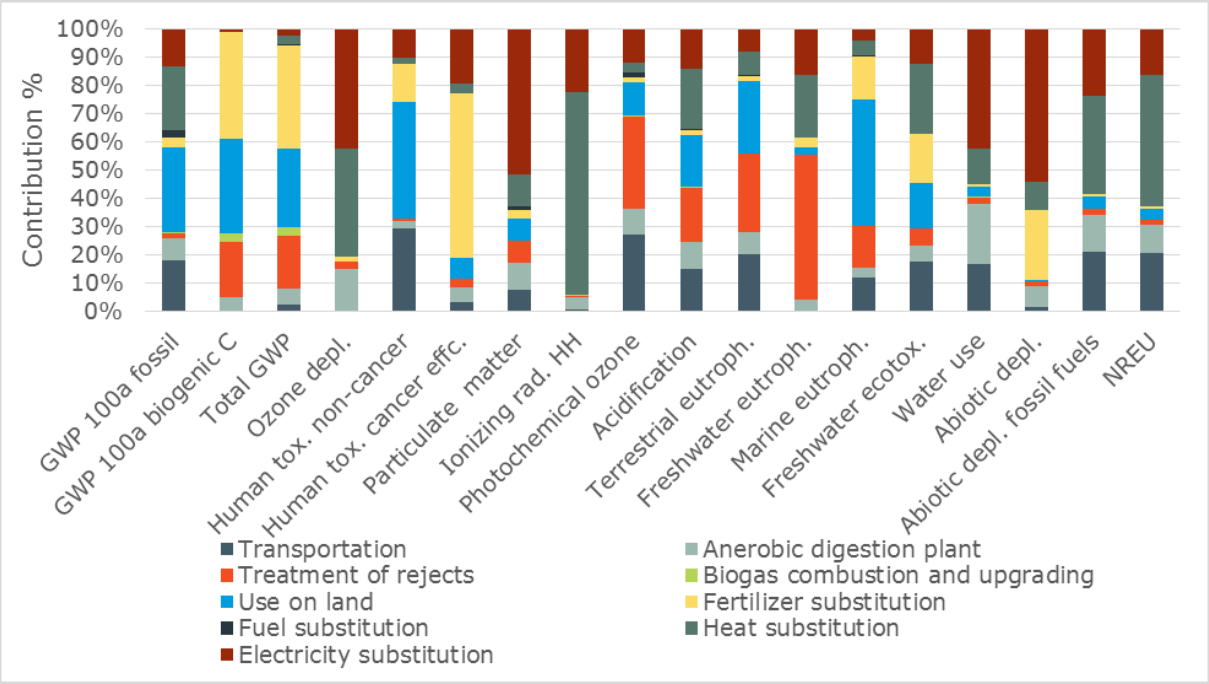

Figure 244. Contribution analysis, Anaerobic digestion, PLA (absolute percentages).

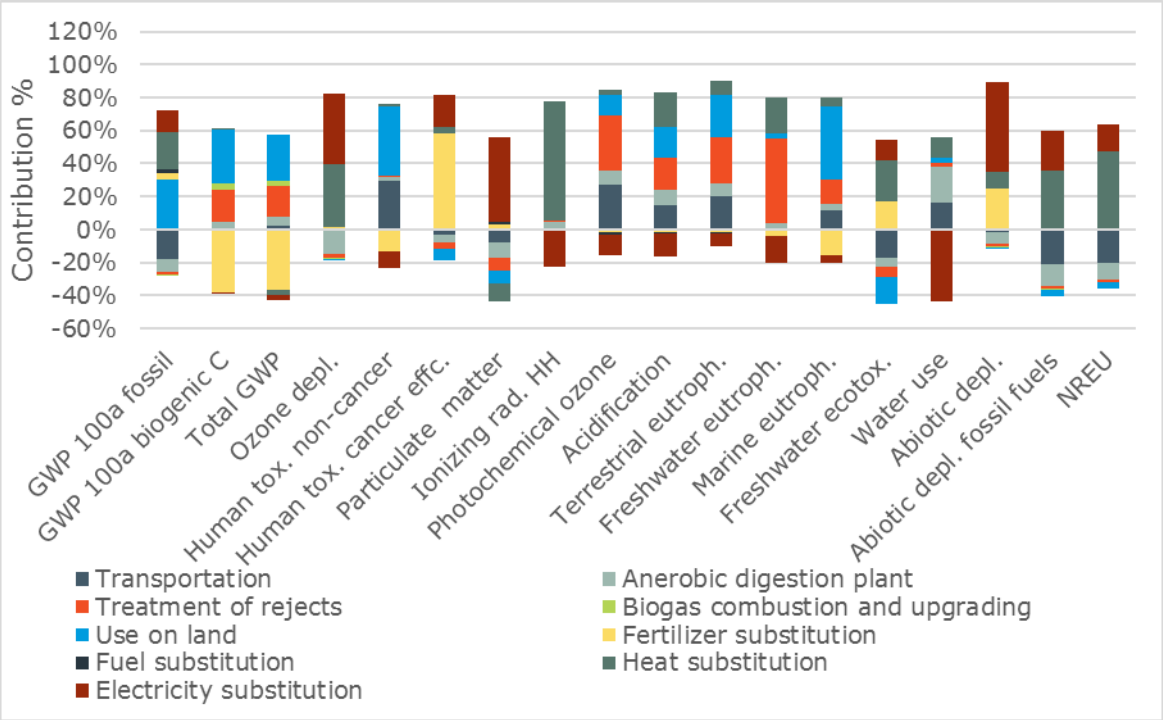

Figure 245. Contribution analysis, Anaerobic digestion, PLA

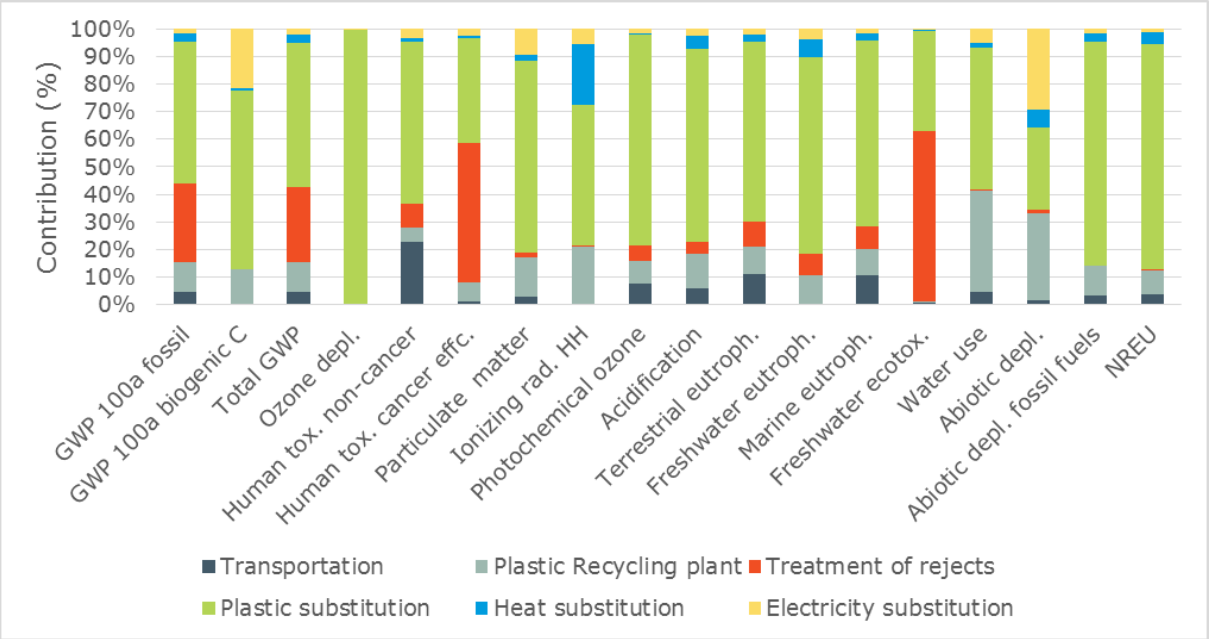

Figure 246. Contribution analysis, Plastic recycling, PET (absolute percentages).

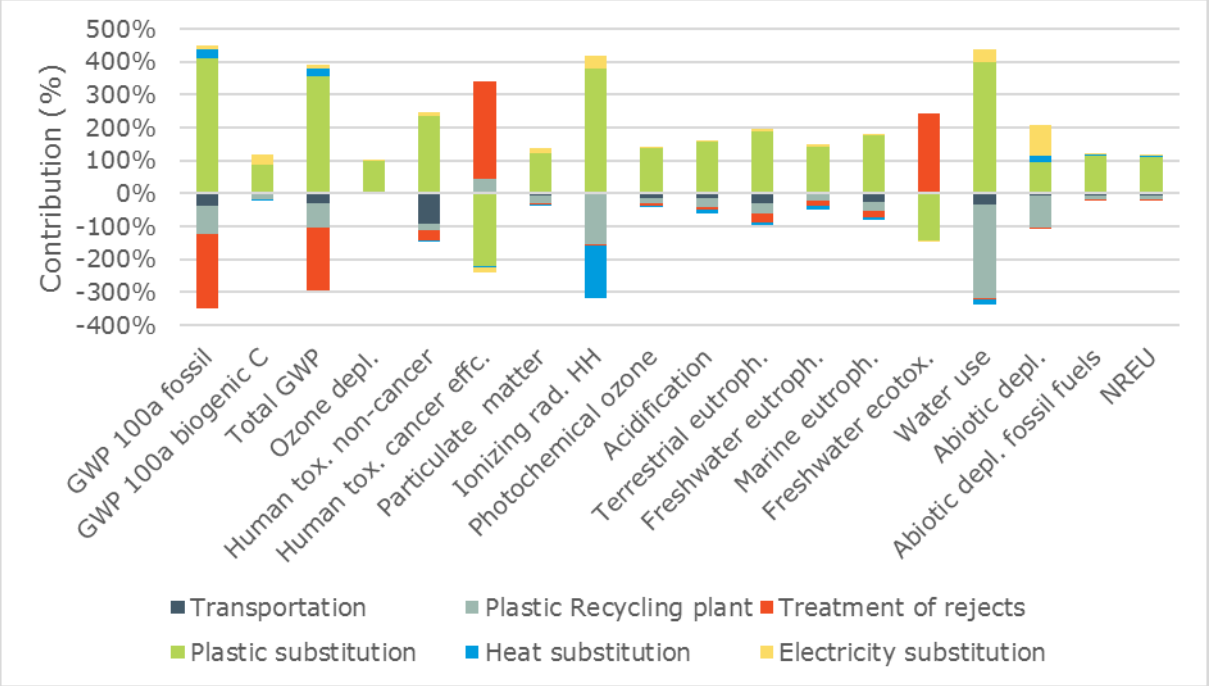

Figure 247. Contribution analysis, Plastic recycling, PET

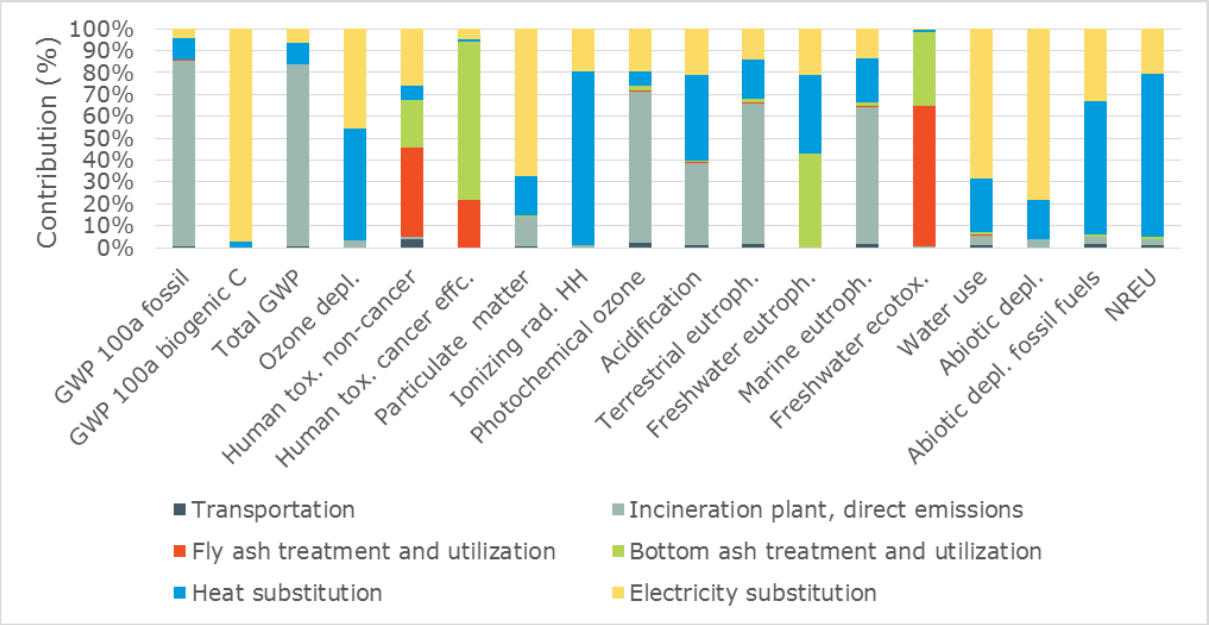

Figure 248. Contribution analysis, incineration, PET (absolute percentages).

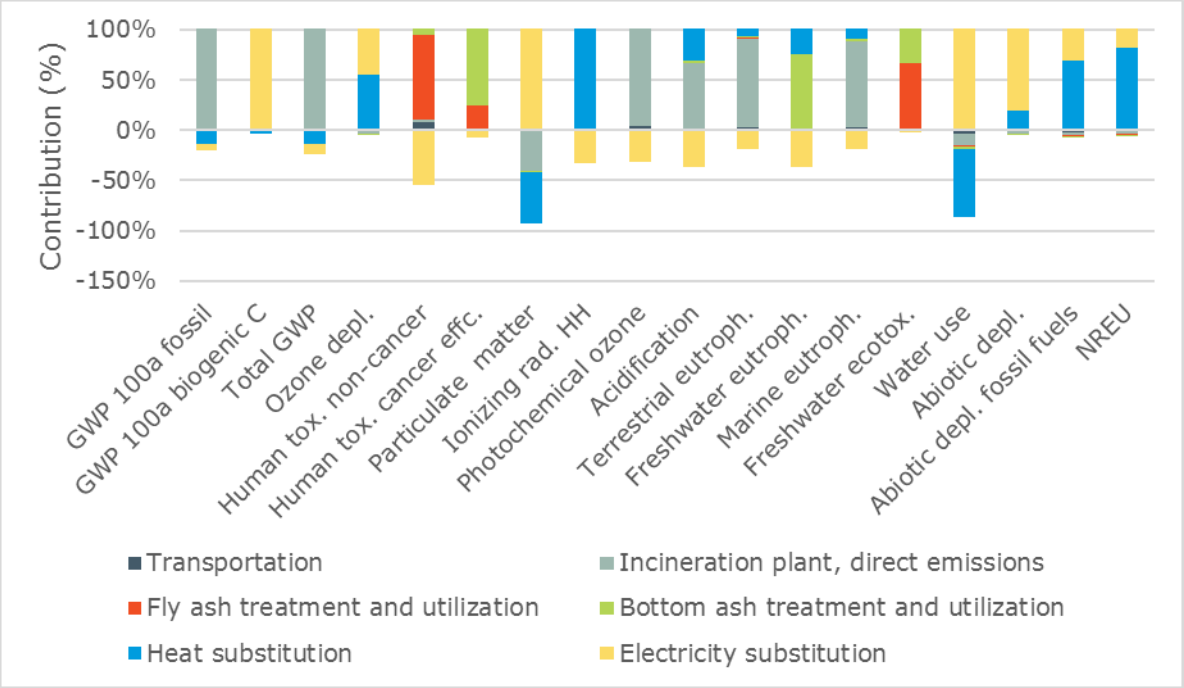

Figure 249. Contribution analysis, incineration, PET

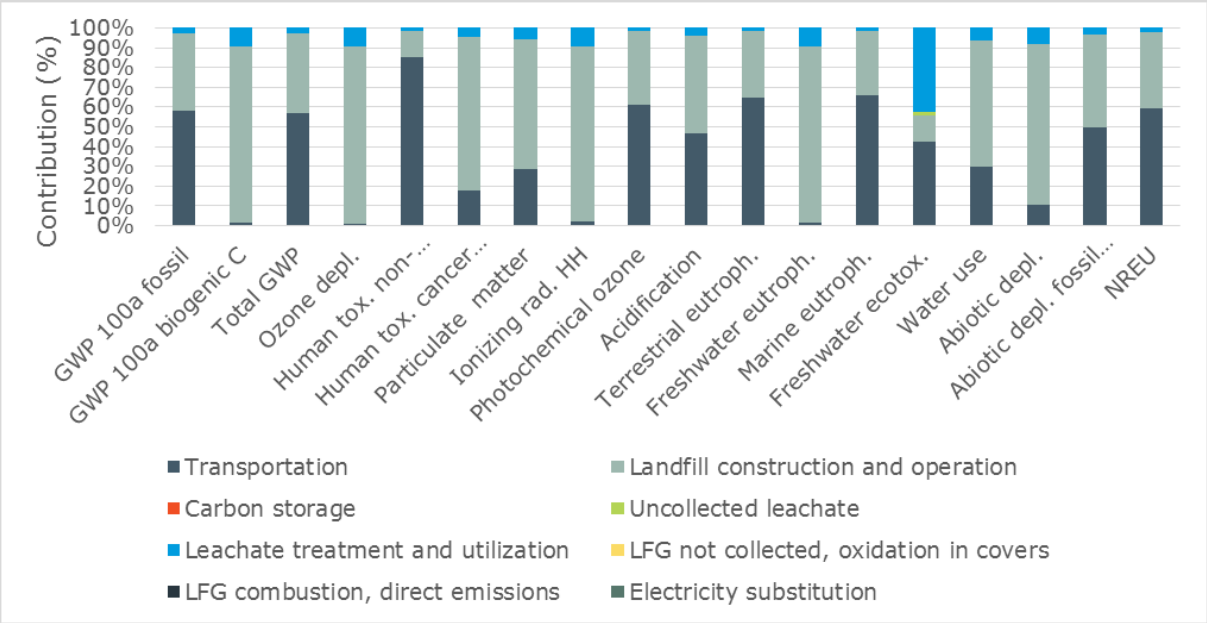

Figure 250. Contribution analysis, landfill, PET.

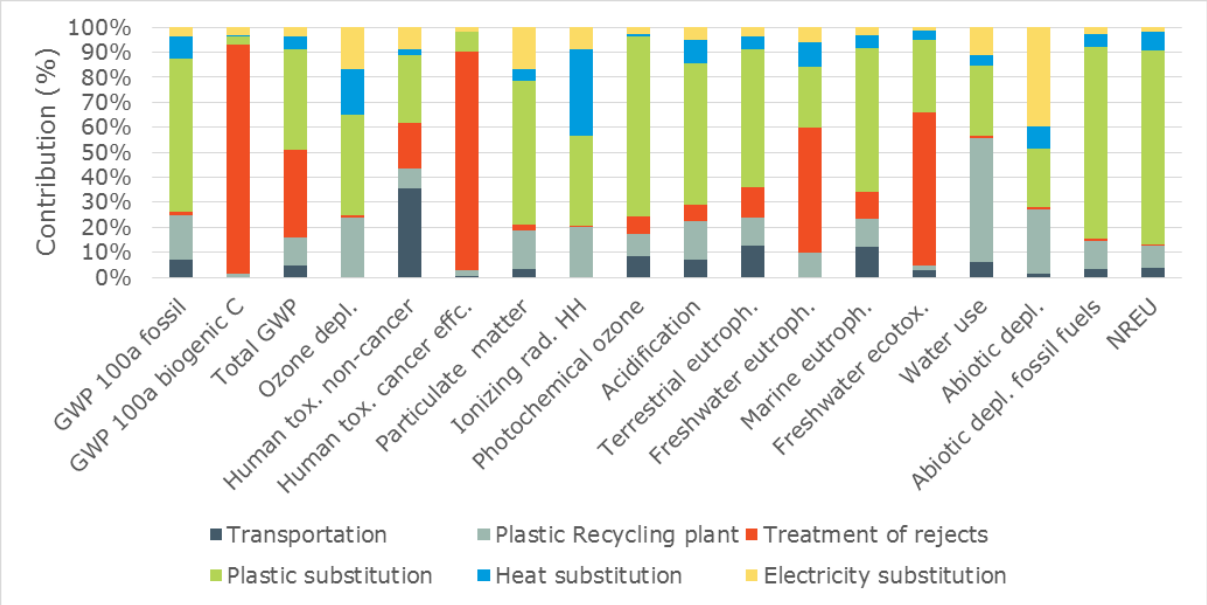

Figure 251. Contribution analysis, Plastic recycling, Bio-based PP (absolute percentages).

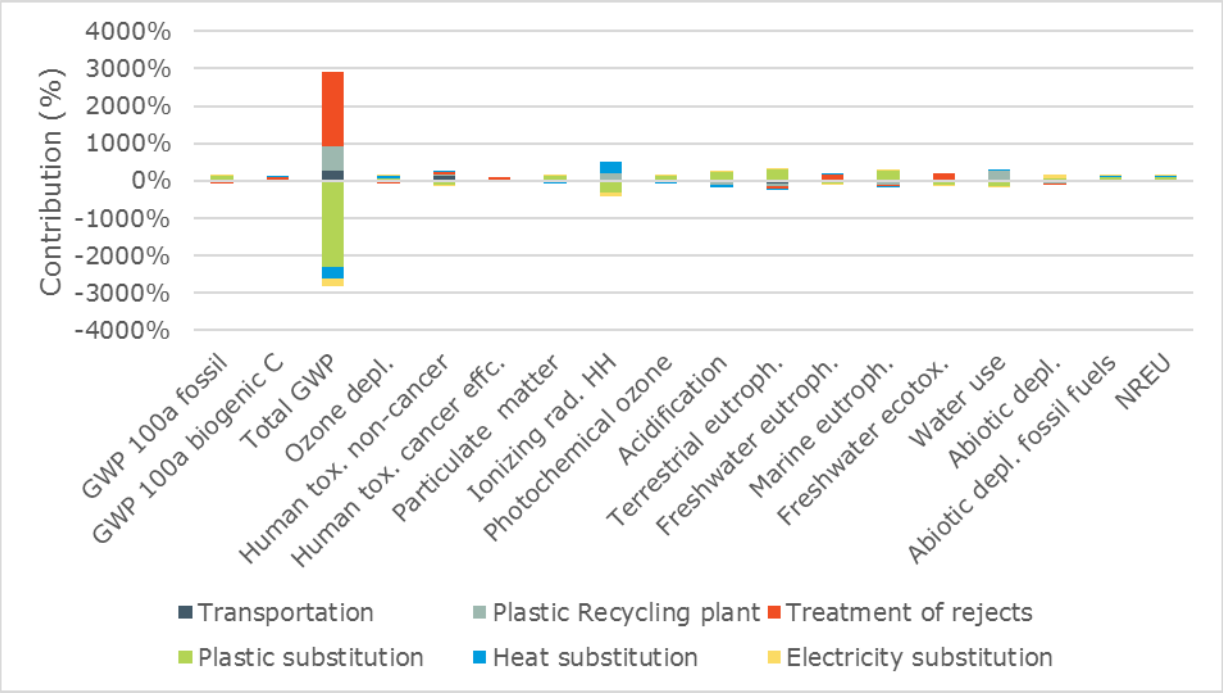

Figure 252. Contribution analysis, Plastic recycling, Bio-based PP.

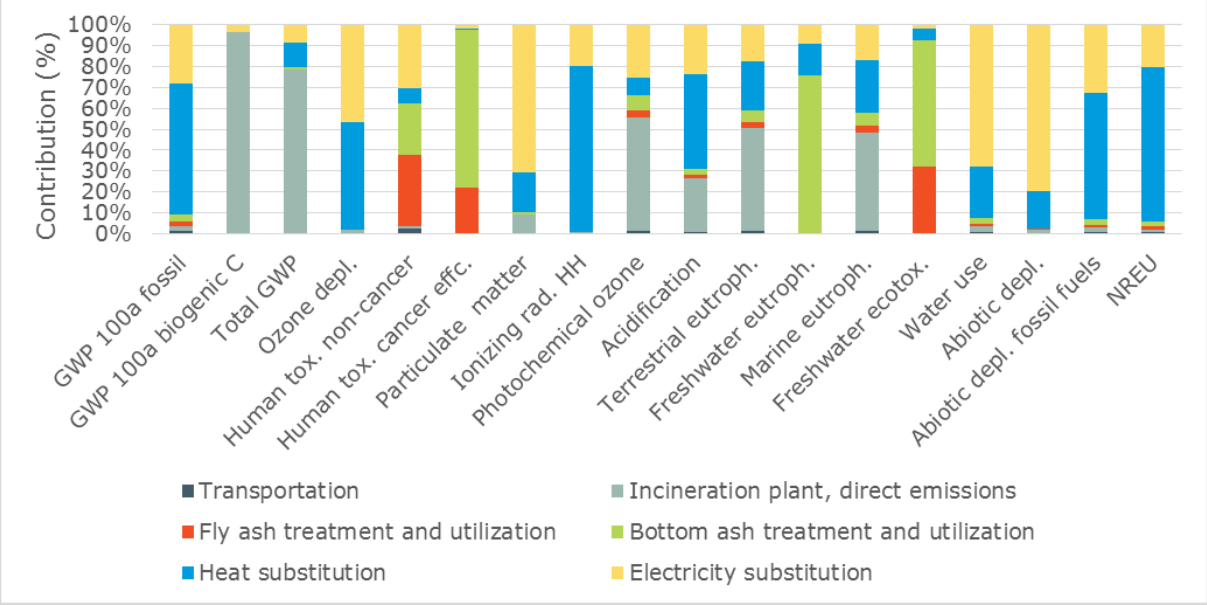

Figure 253. Contribution analysis, Incineration, Bio-based PP (absolute percentages).

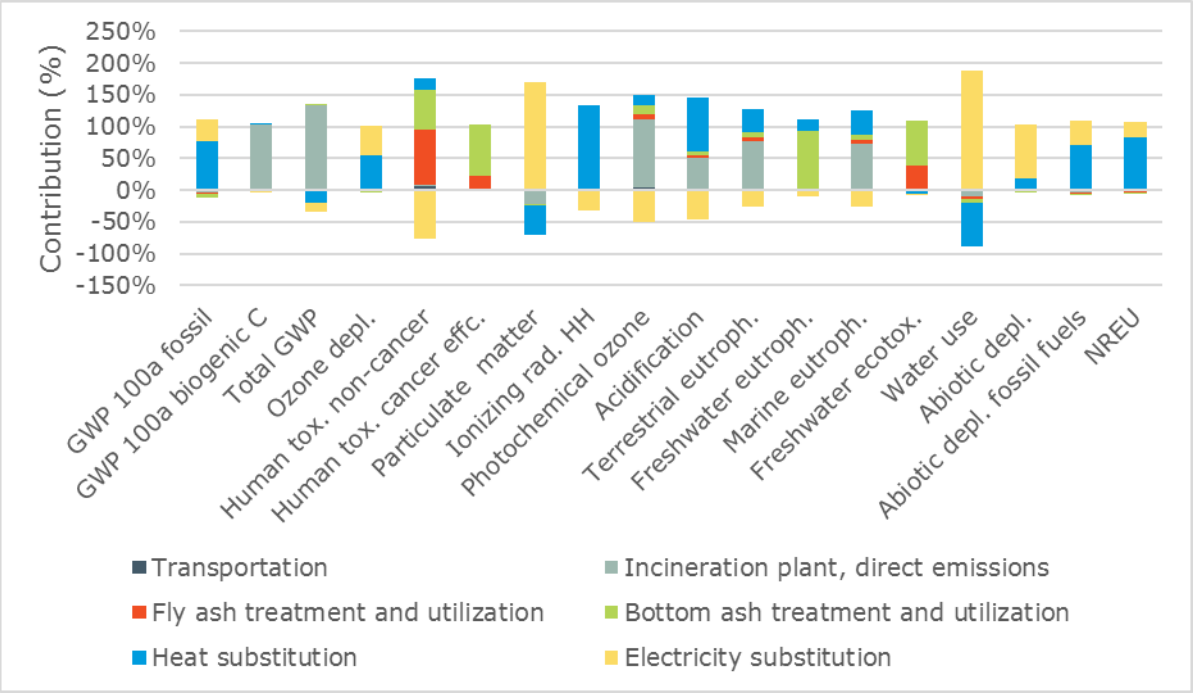

Figure 254. Contribution analysis, Incineration, Bio-based PP.

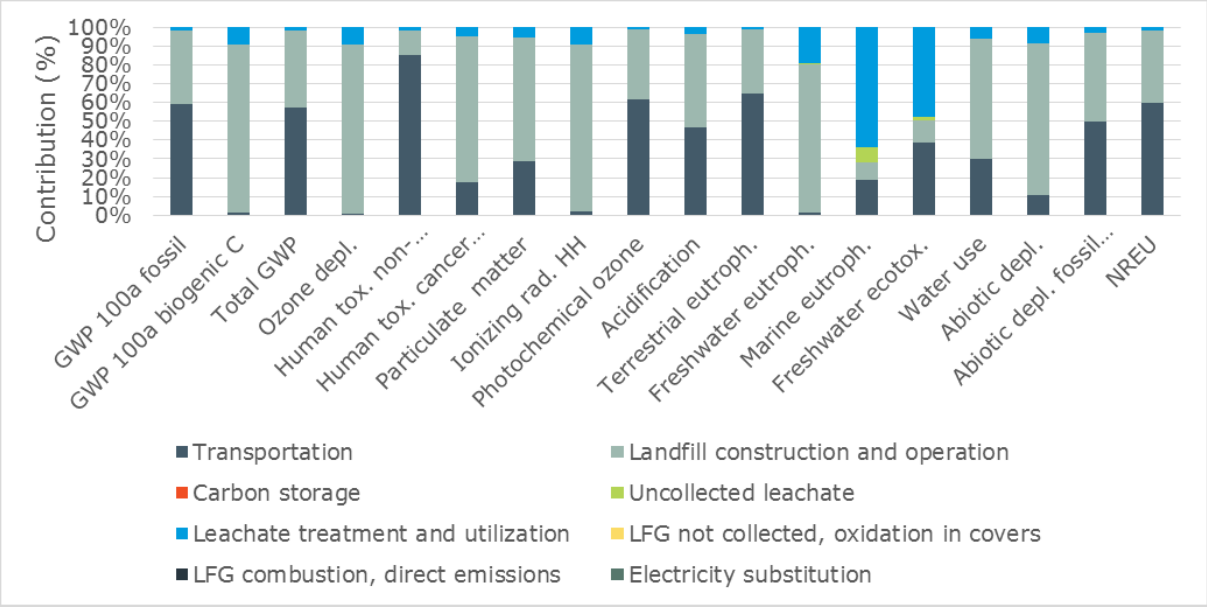

Figure 255. Contribution analysis, Landfill, Bio-based PP.

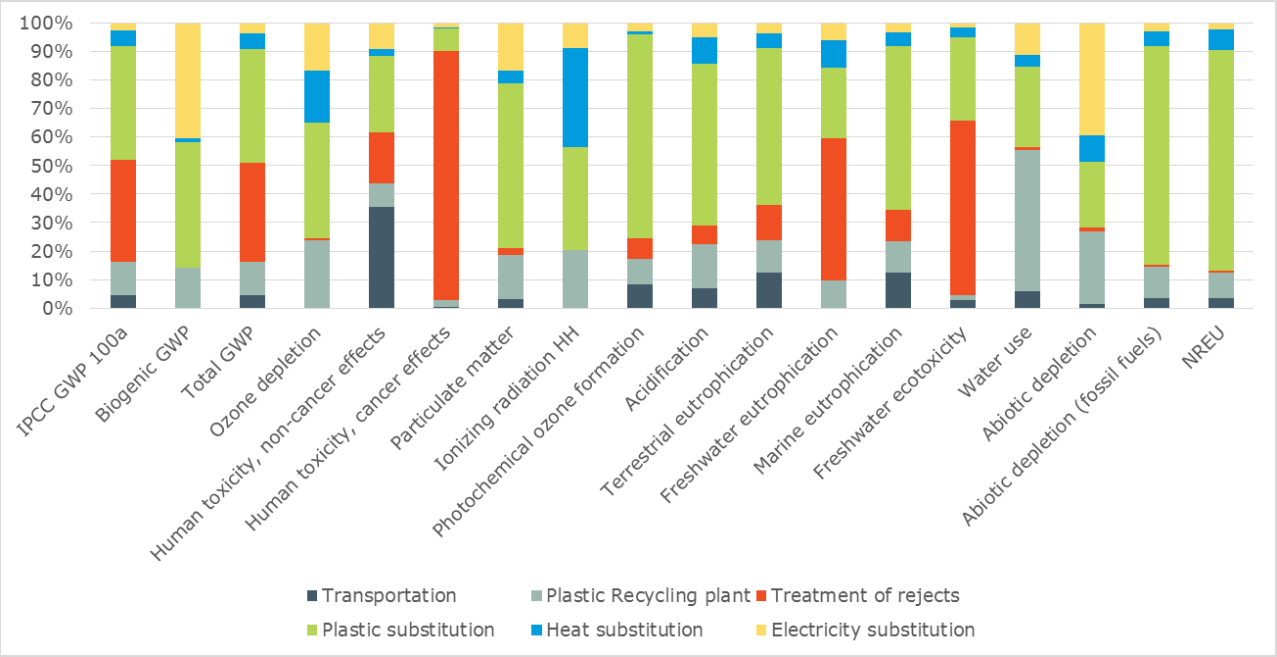

Figure 256. Contribution analysis, Plastic recycling, petrochemical PP (absolute percentages).

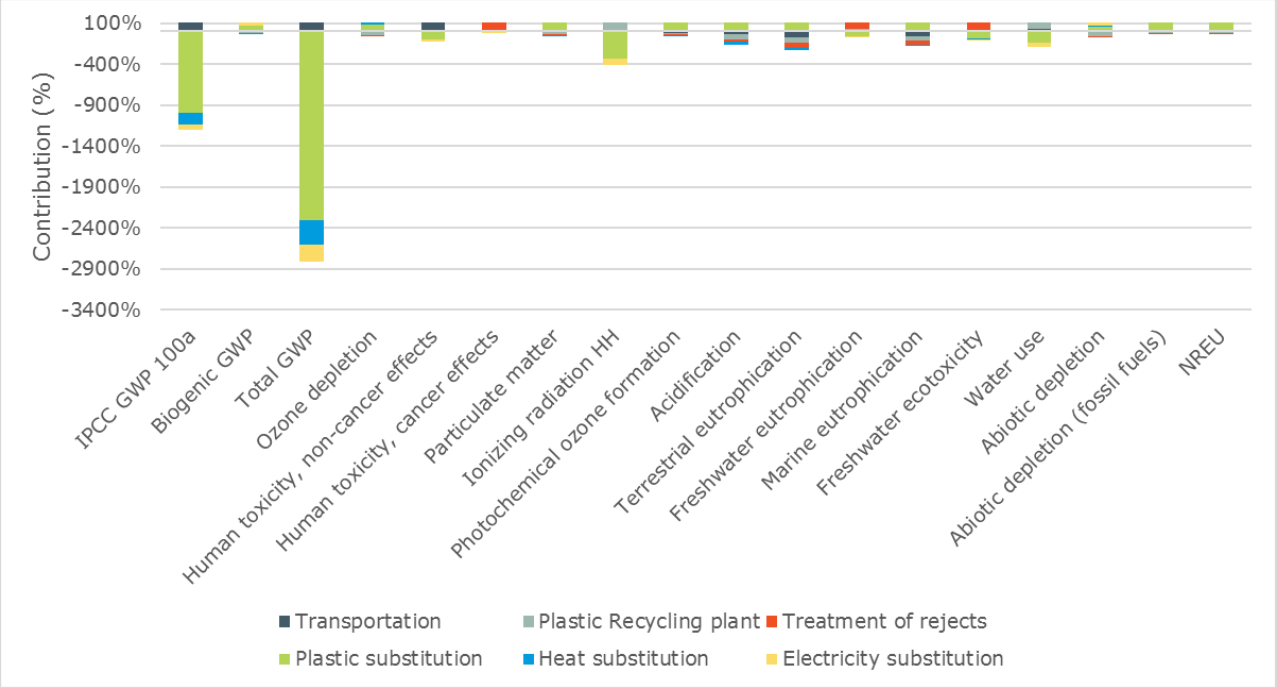

Figure 257. Contribution analysis, Plastic recycling, petrochemical PP.

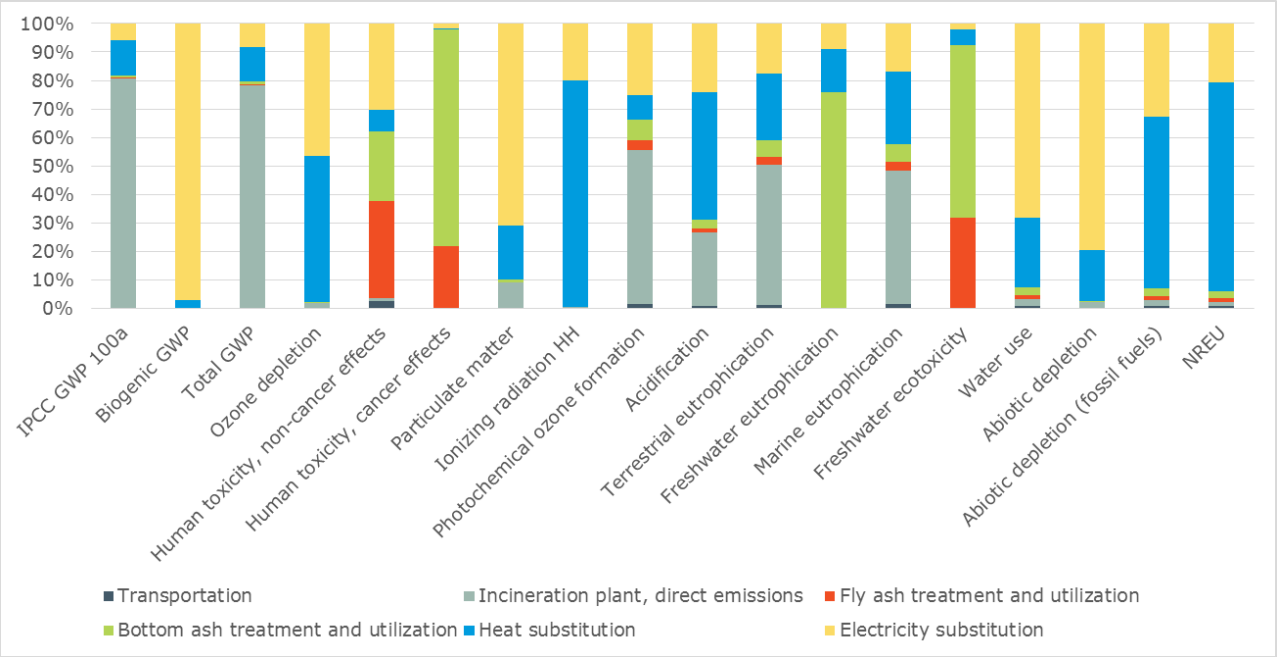

Figure 258. Contribution analysis, Incineration, petrochemical PP (absolute percentages).

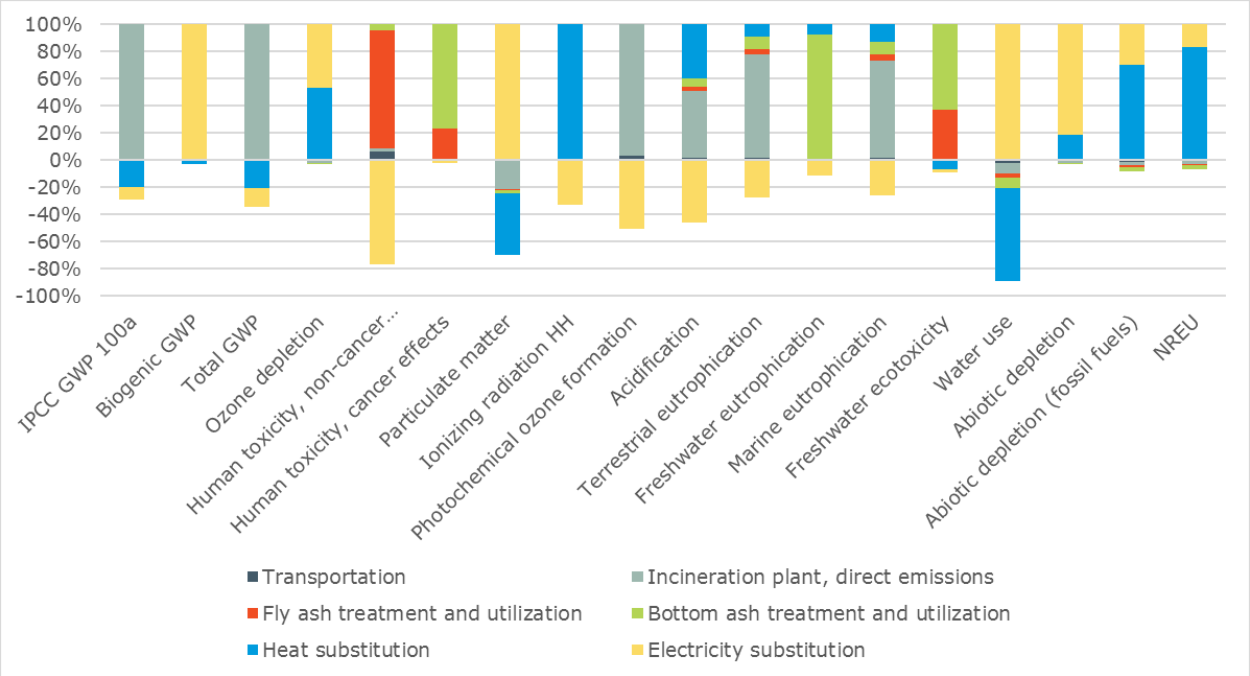

Figure 259. Contribution analysis, Incineration, petrochemical PP.

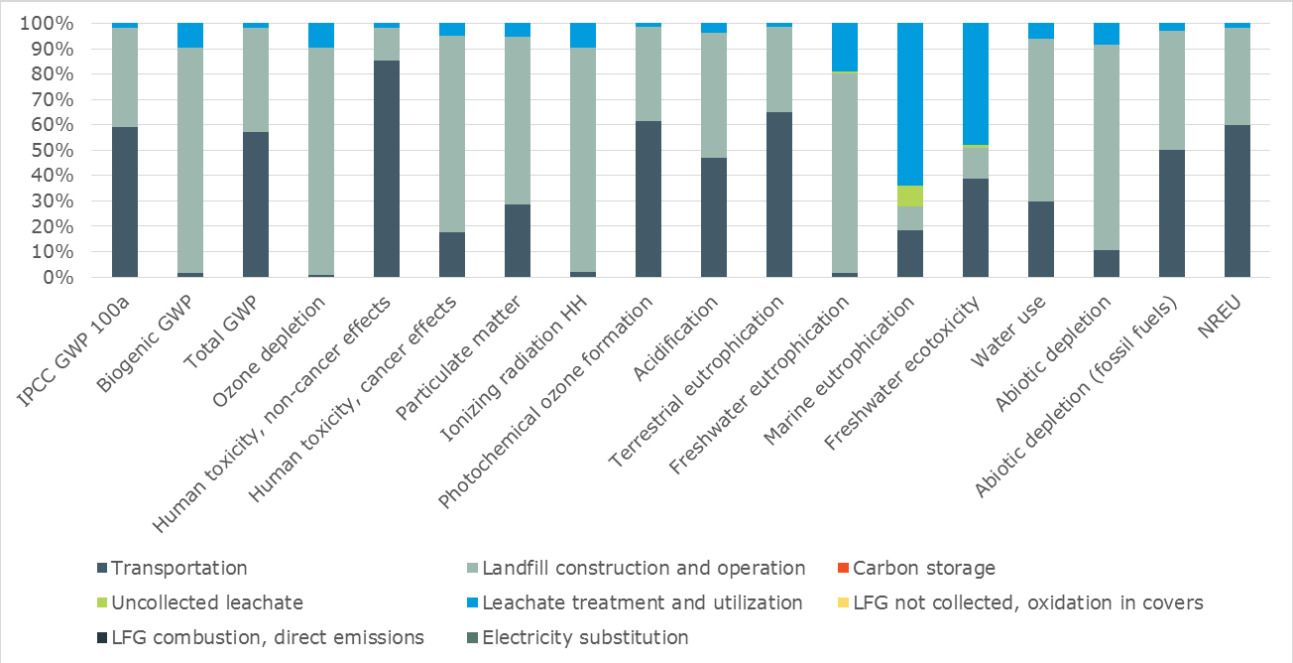

Figure 260. Contribution analysis, landfill, petrochemical PP.

Case study 4: Single-use cutlery

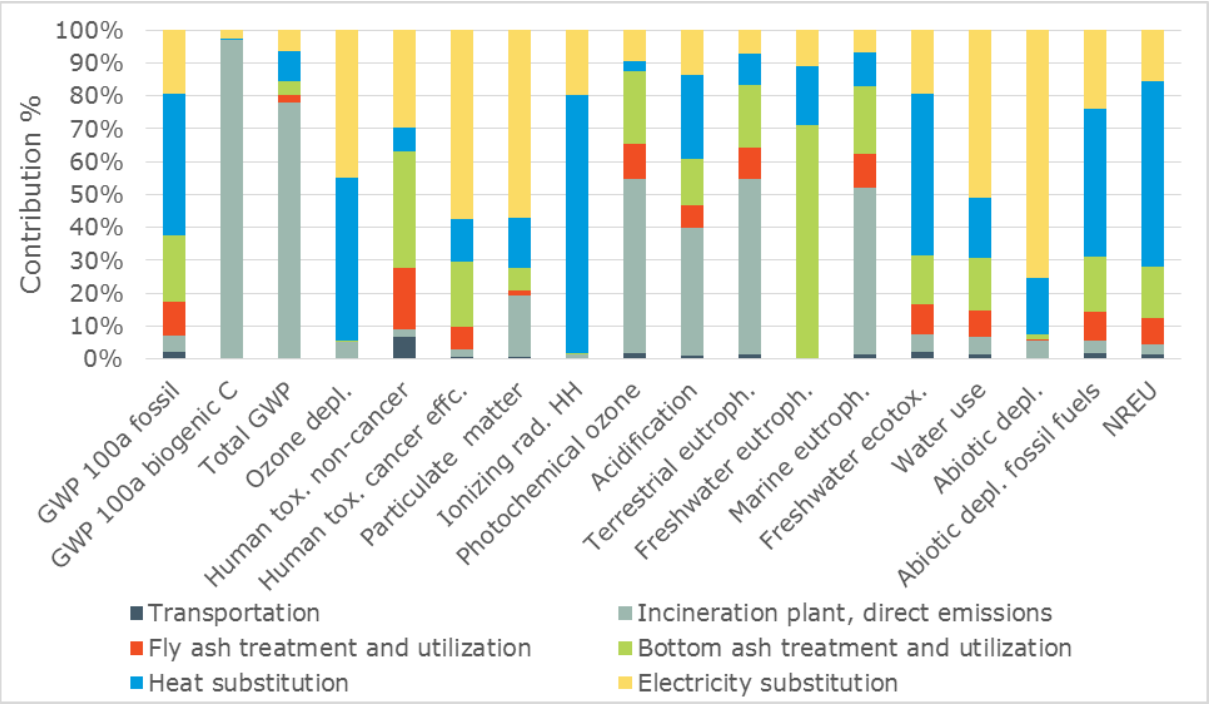

Figure 261. Contribution analysis, Incineration, cPLA excl. food leftovers.

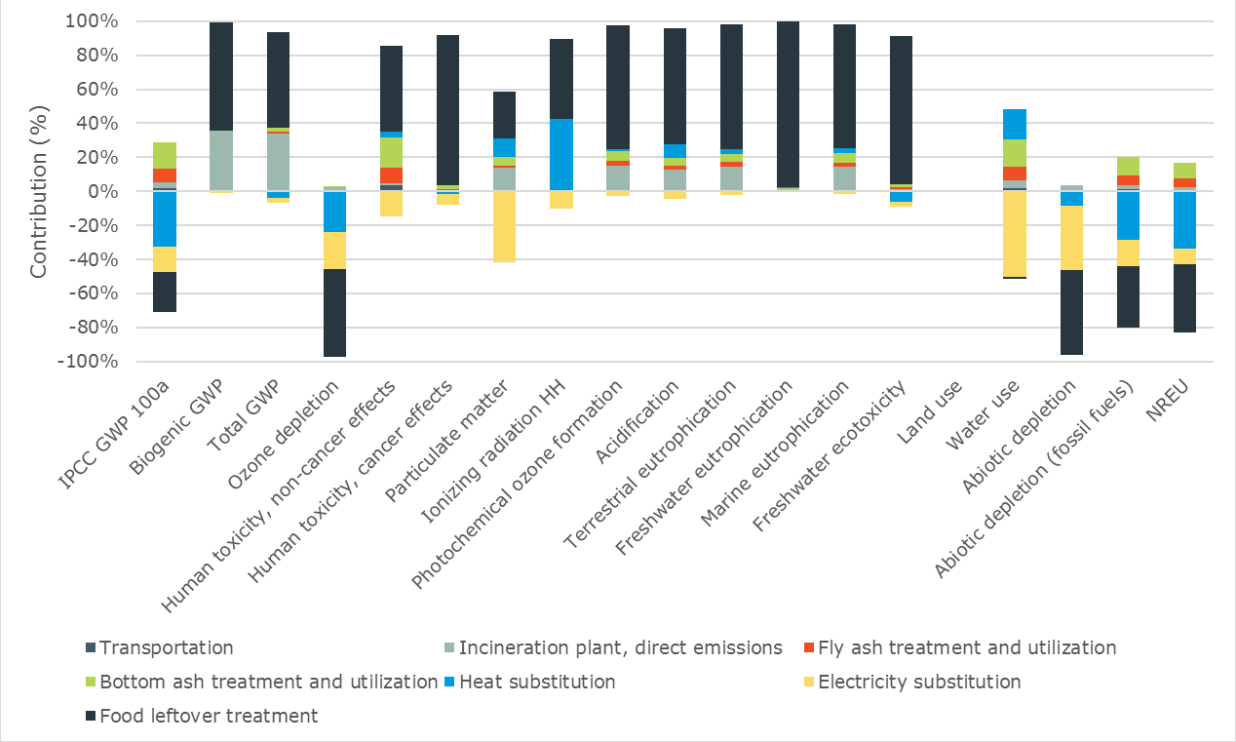

Figure 262. Contribution analysis, Incineration, cPLA incl. food leftovers (absolute percentages).

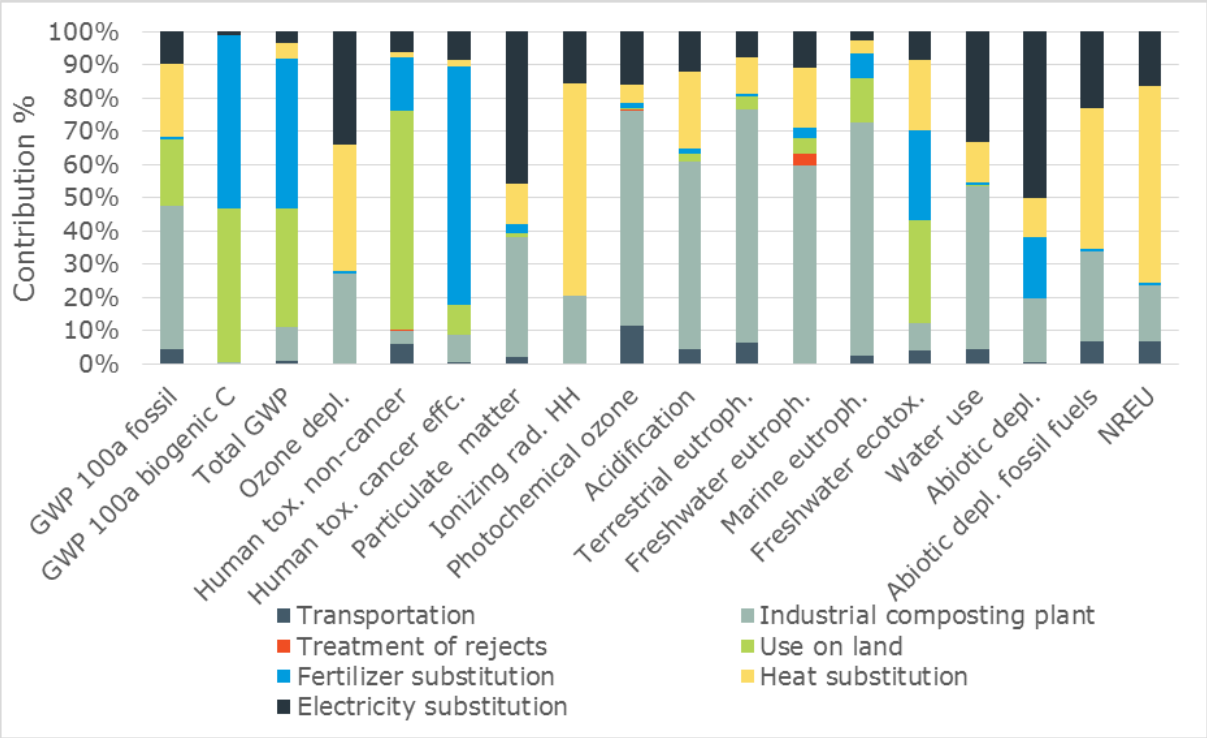

Figure 263. Contribution analysis for case 4, Composting cPLA excl. food leftovers (absolute percentages)

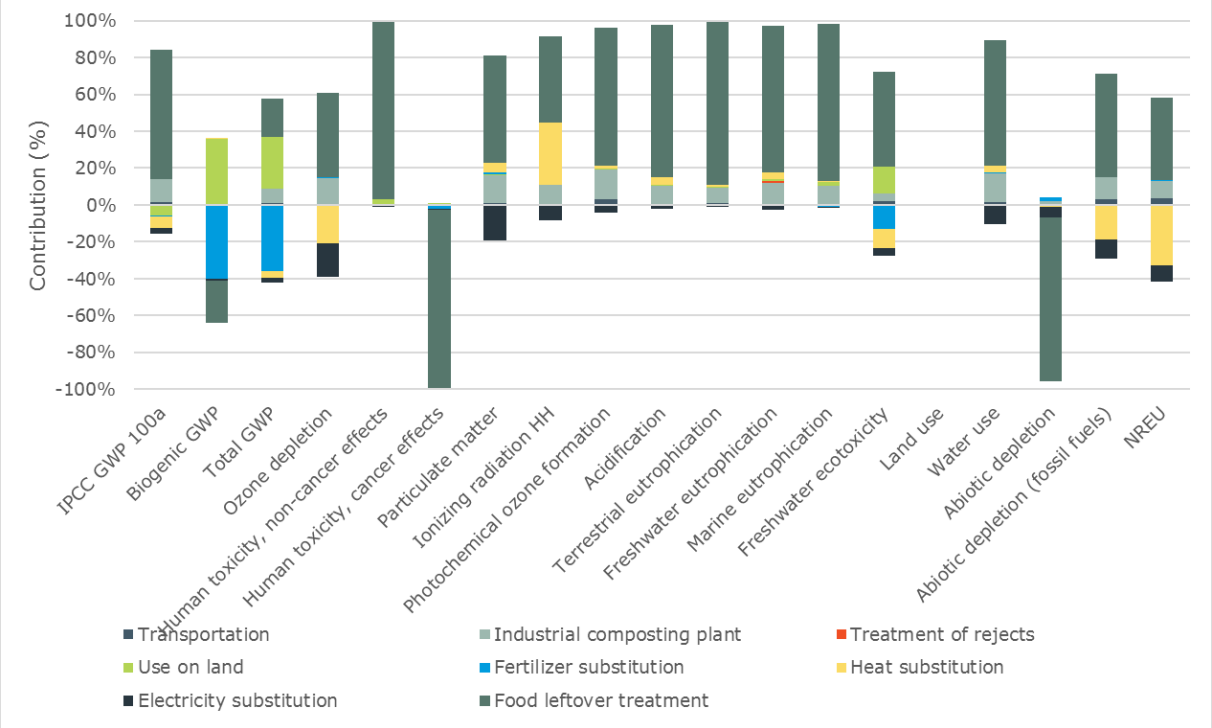

Figure 264. Contribution analysis for case 4, Composting cPLA including food leftovers.

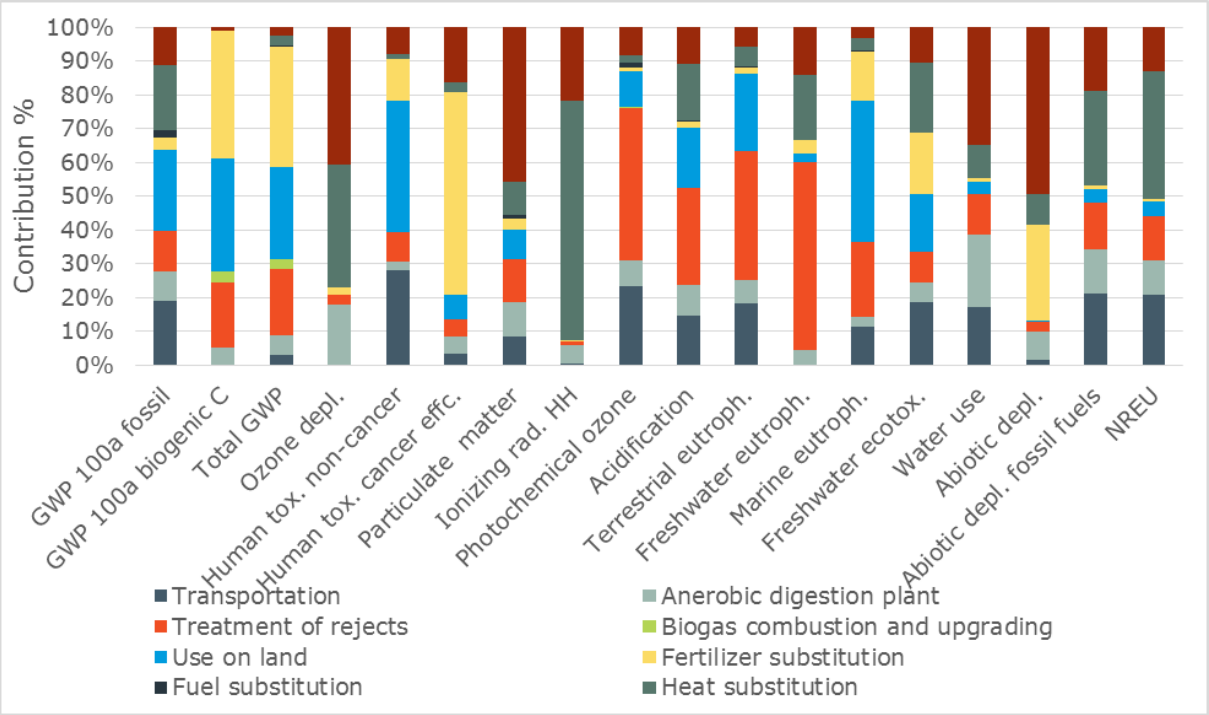

Figure 265. Contribution case 4, AD cPLA excl. food leftovers (absolute percentages).

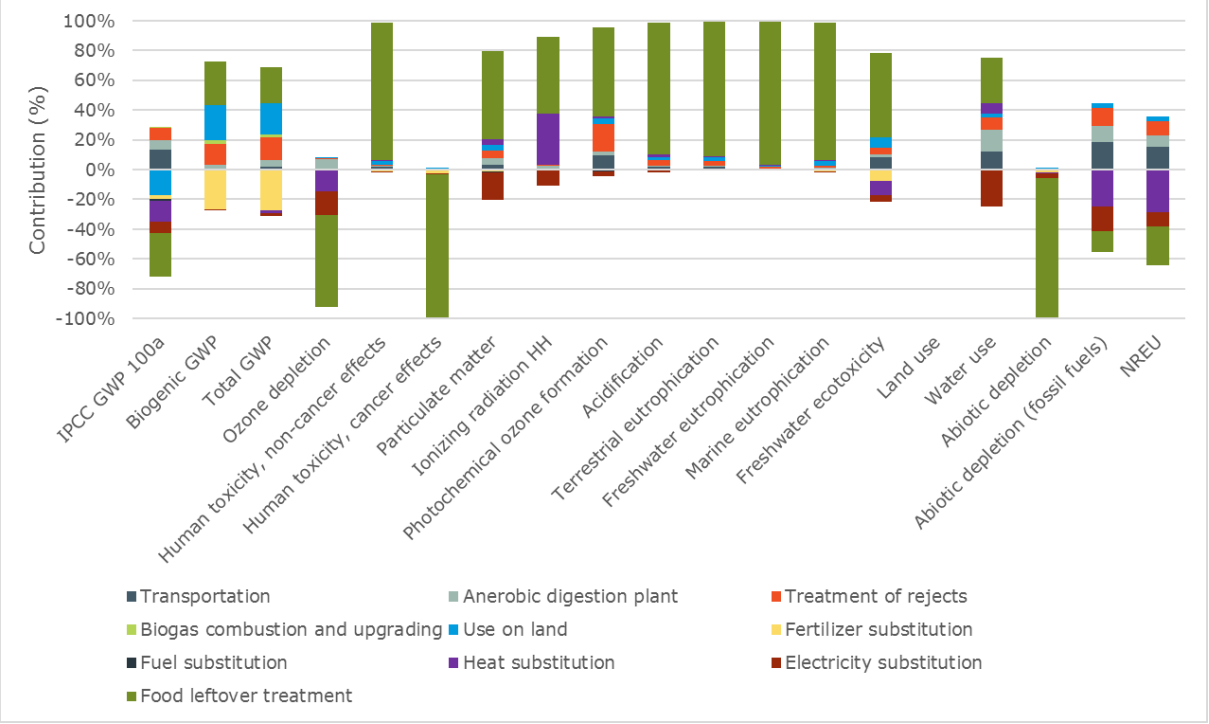

Figure 266. Contribution case 4, AD cPLA incl. food leftovers.

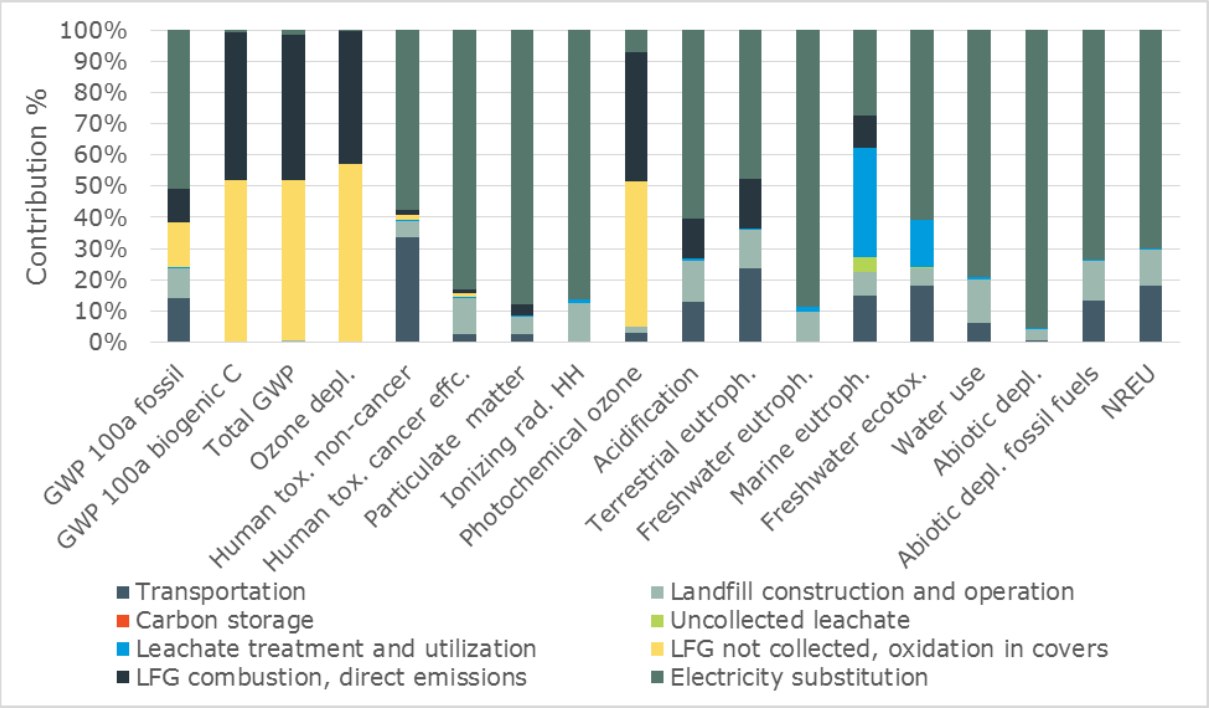

Figure 267. Contribution analysis for case 4, Landfilling cPLA excl. food leftovers (absolute percentages).

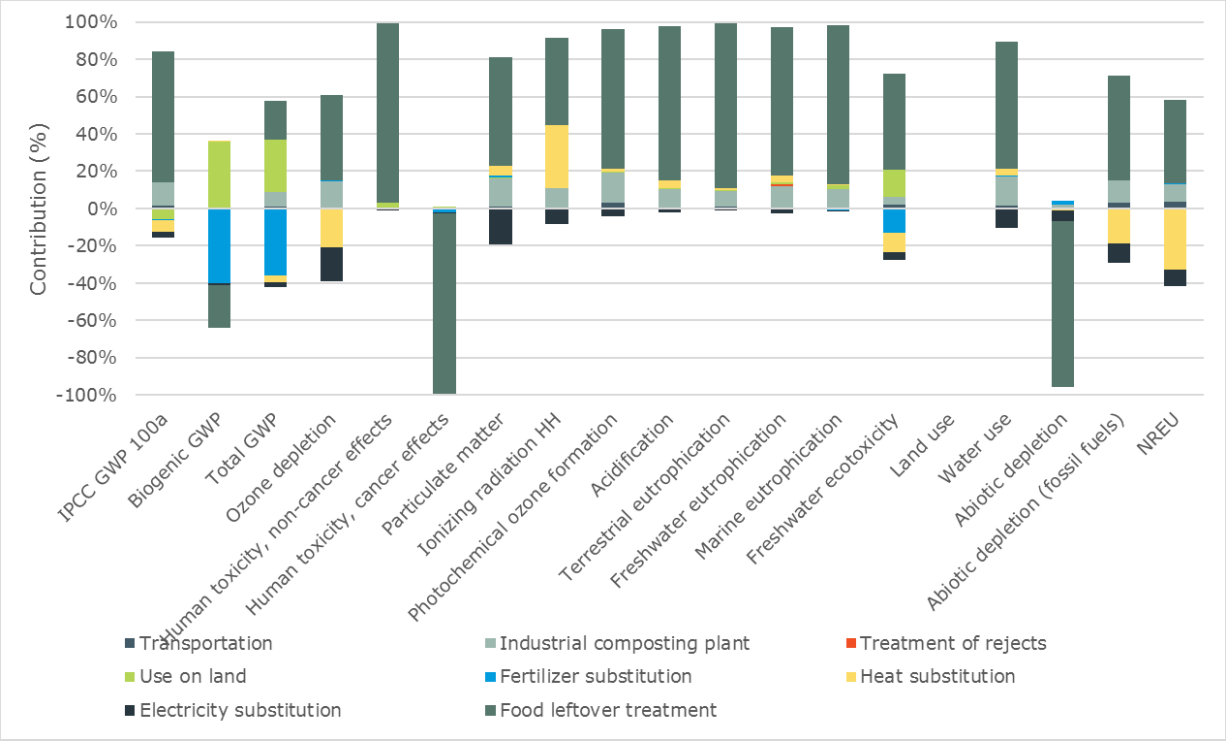

Figure 268. Contribution analysis for case 4, Landfilling cPLA incl. food leftovers

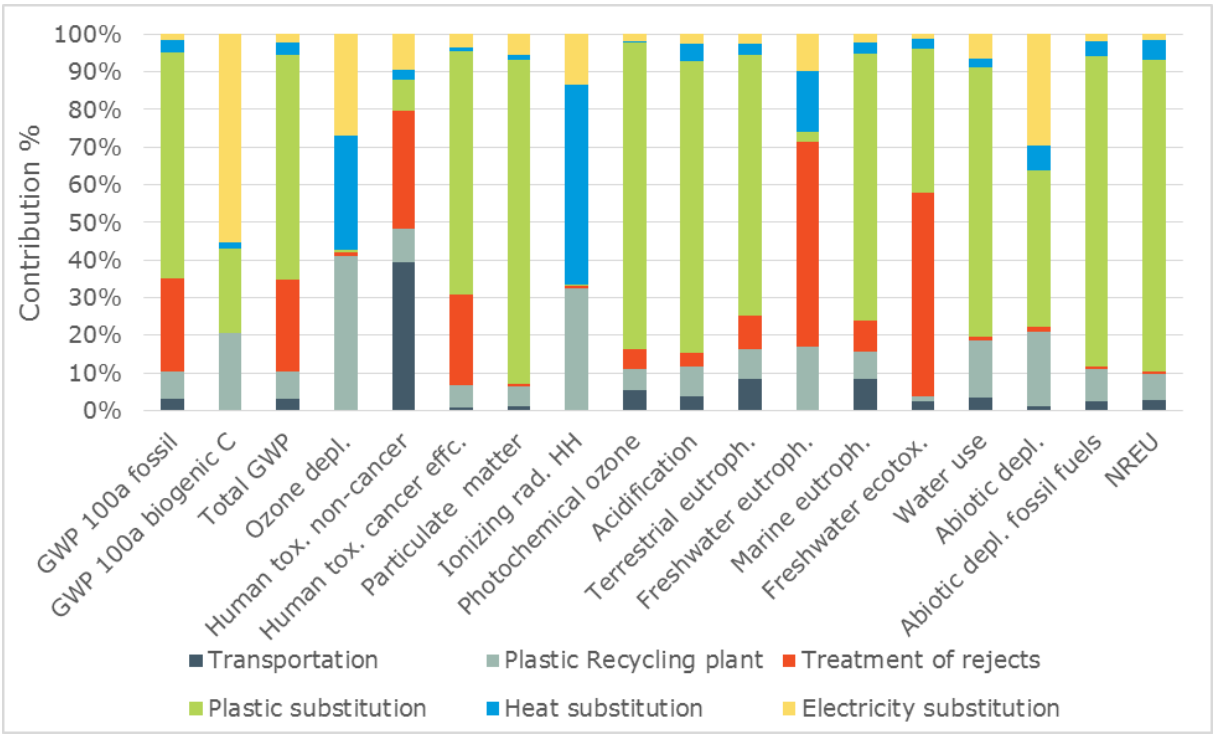

Figure 269. Contribution analysis for case 4, Plastics recycling, PS excl. food leftovers (absolute percentages).

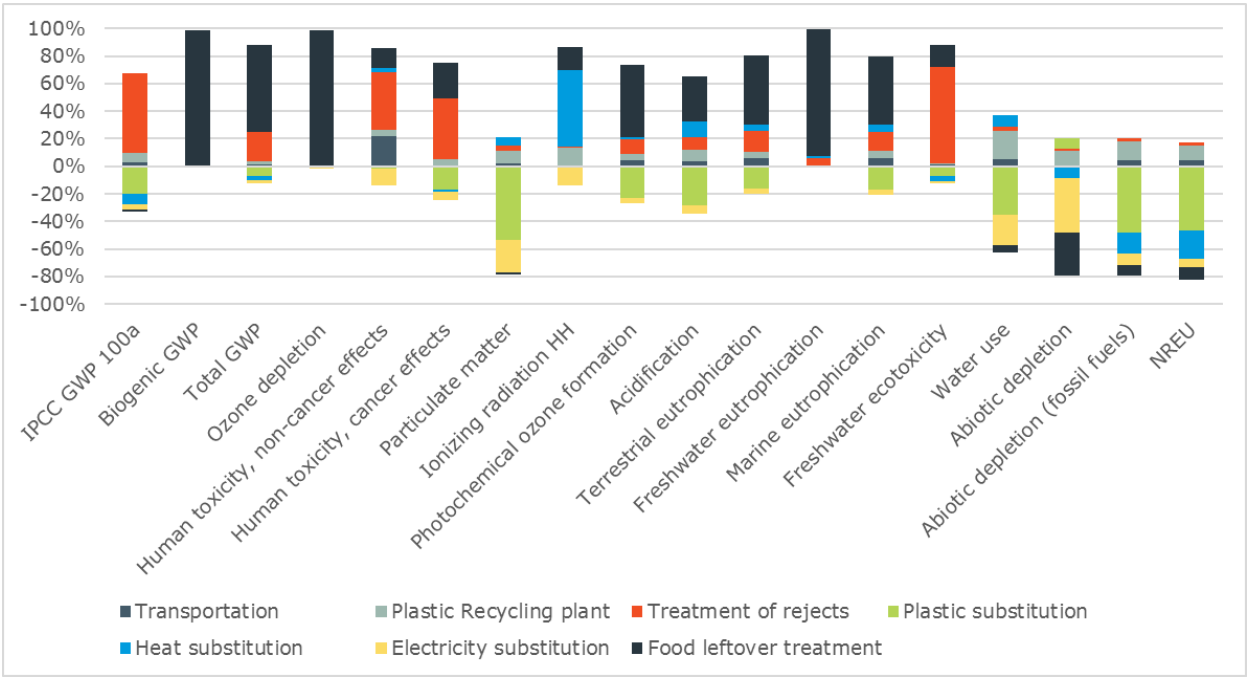

Figure 270. Contribution analysis for case 4, Plastics recycling, PS including food leftovers.

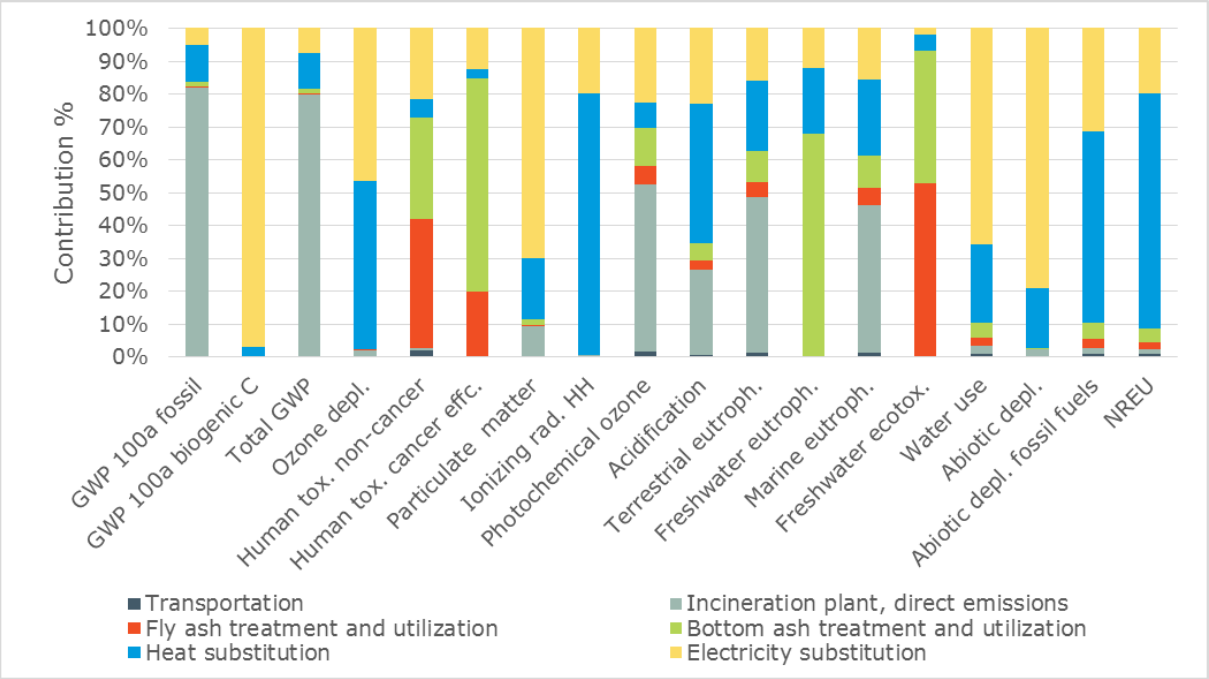

Figure 271. Contribution of case 4, MSWI, PS excl. food leftovers (absolute percentages).

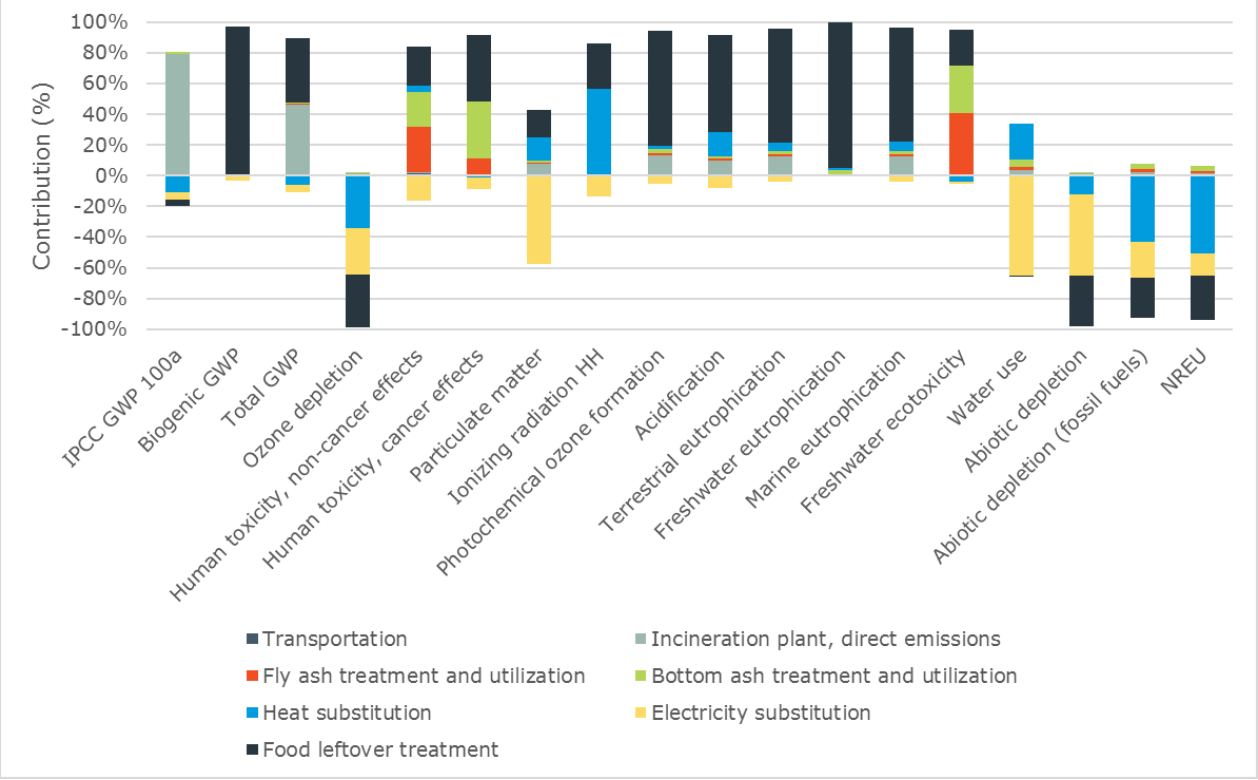

Figure 272. Contribution of case 4, MSWI, PS including. food leftovers.

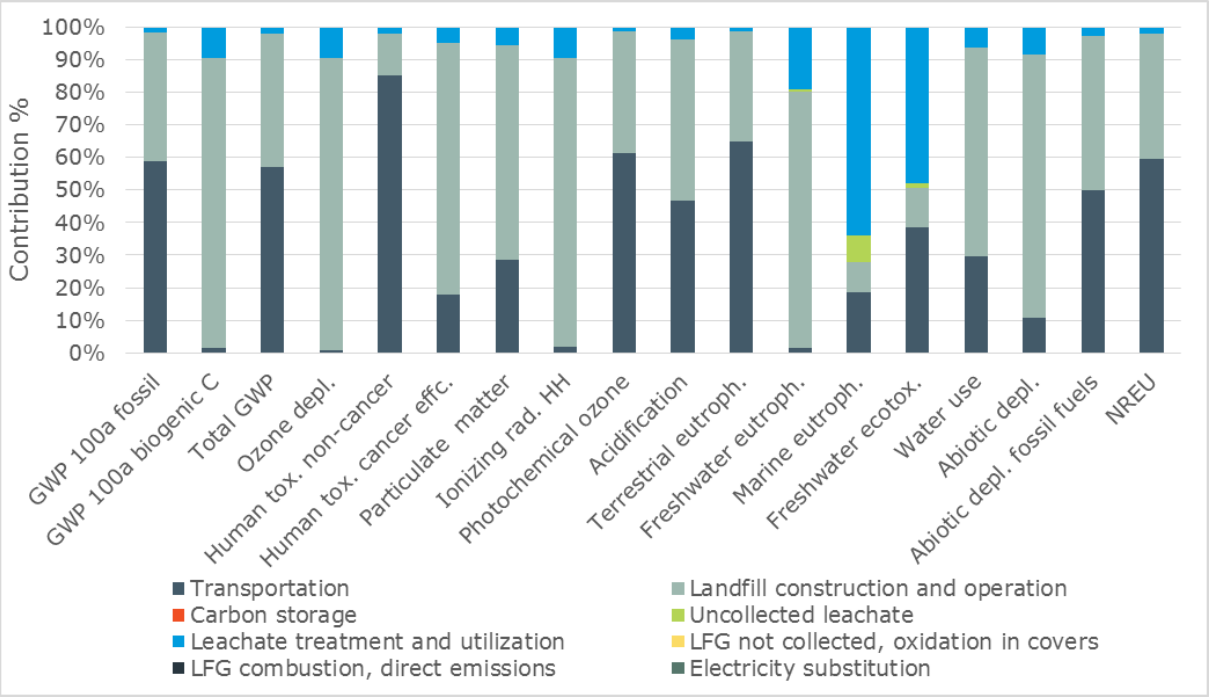

Figure 273. Contribution of case 4, Land filling PS, excl. food leftovers.

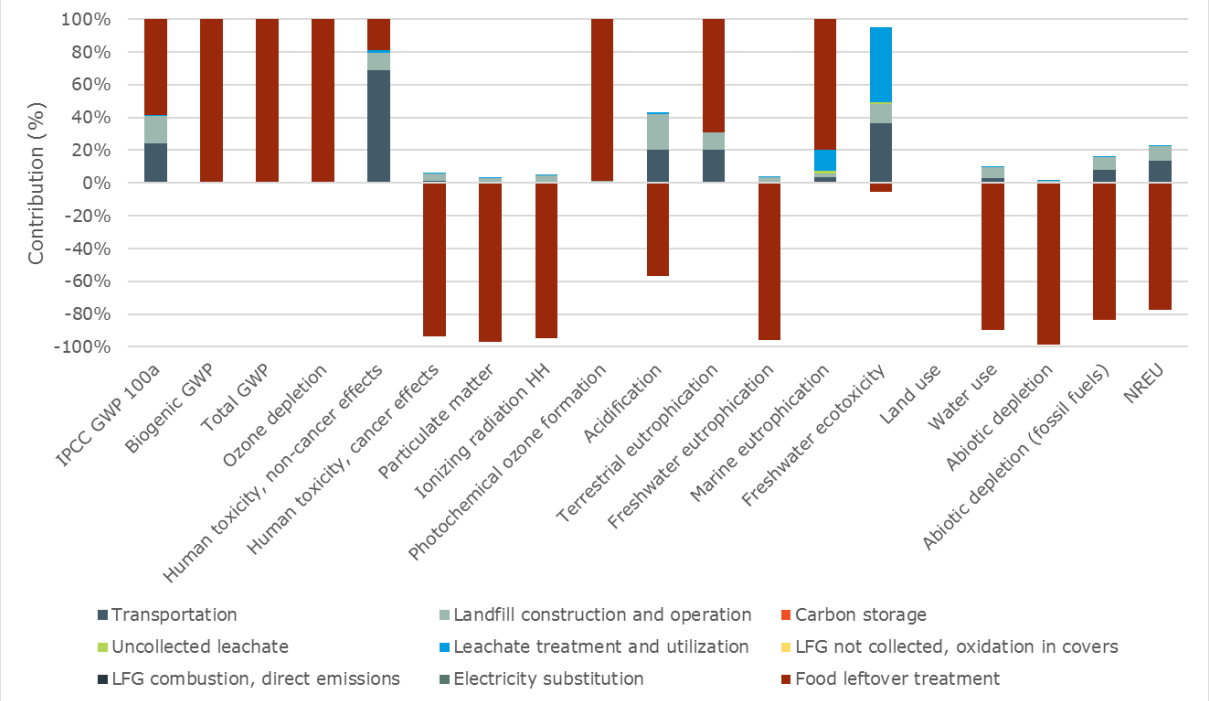

Figure 274. Contribution of case 4, Landfilling PS including. food leftovers

Case study 5: Agricultural mulching films

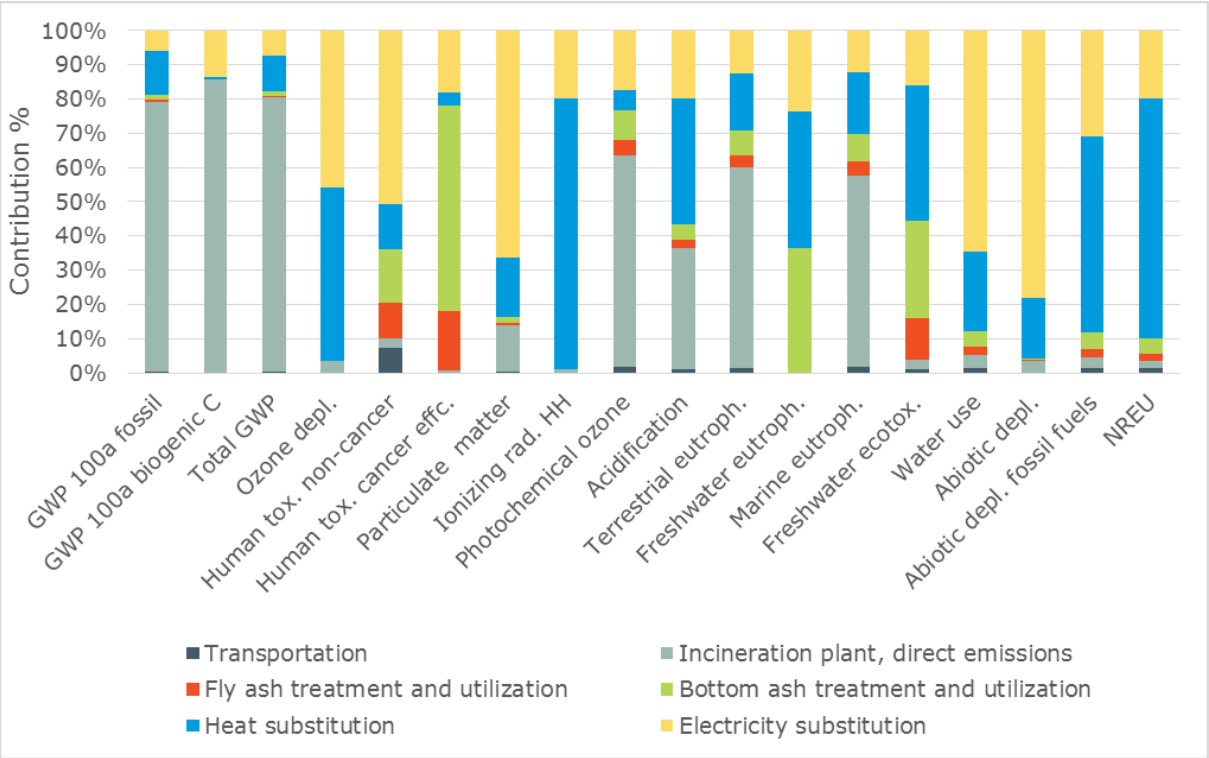

Figure 275. Contribution of case 5 MSWI Bio-based starch mulcing film excl. soil (absolute percentages).

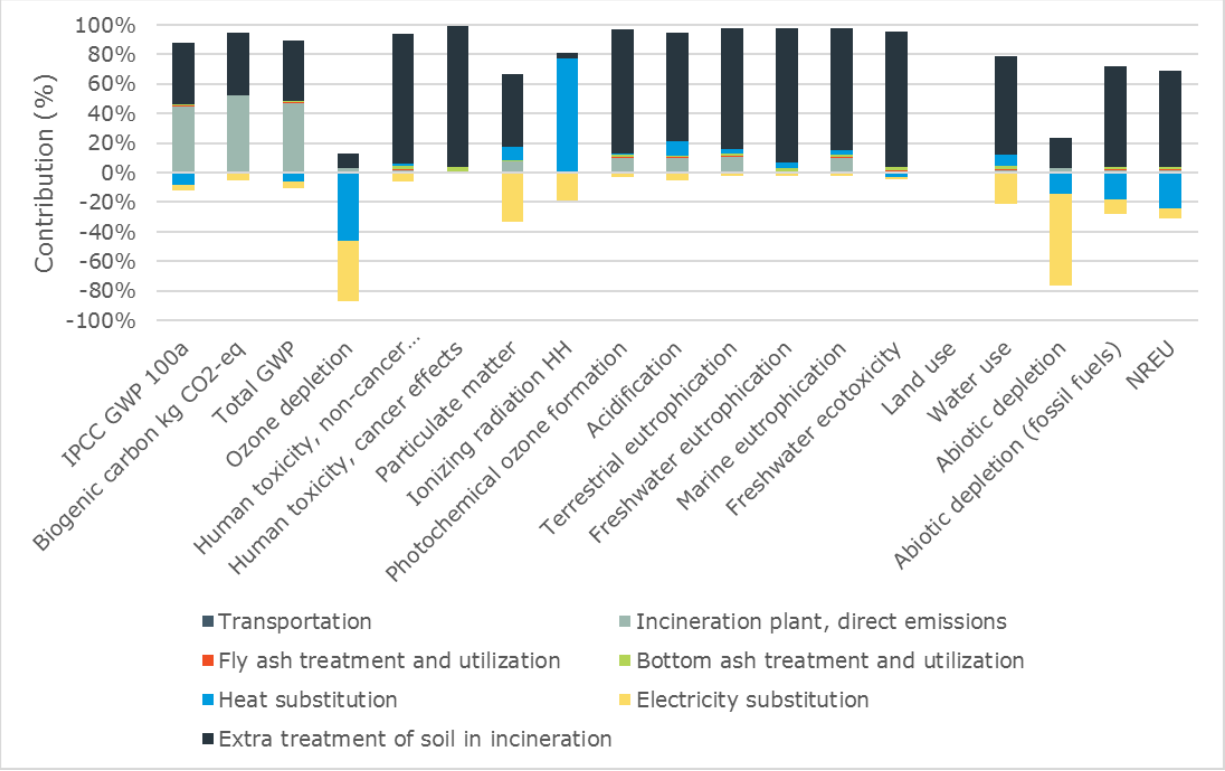

Figure 276. Contribution of case 5 MSWI Bio-based starch mulching film including soil.

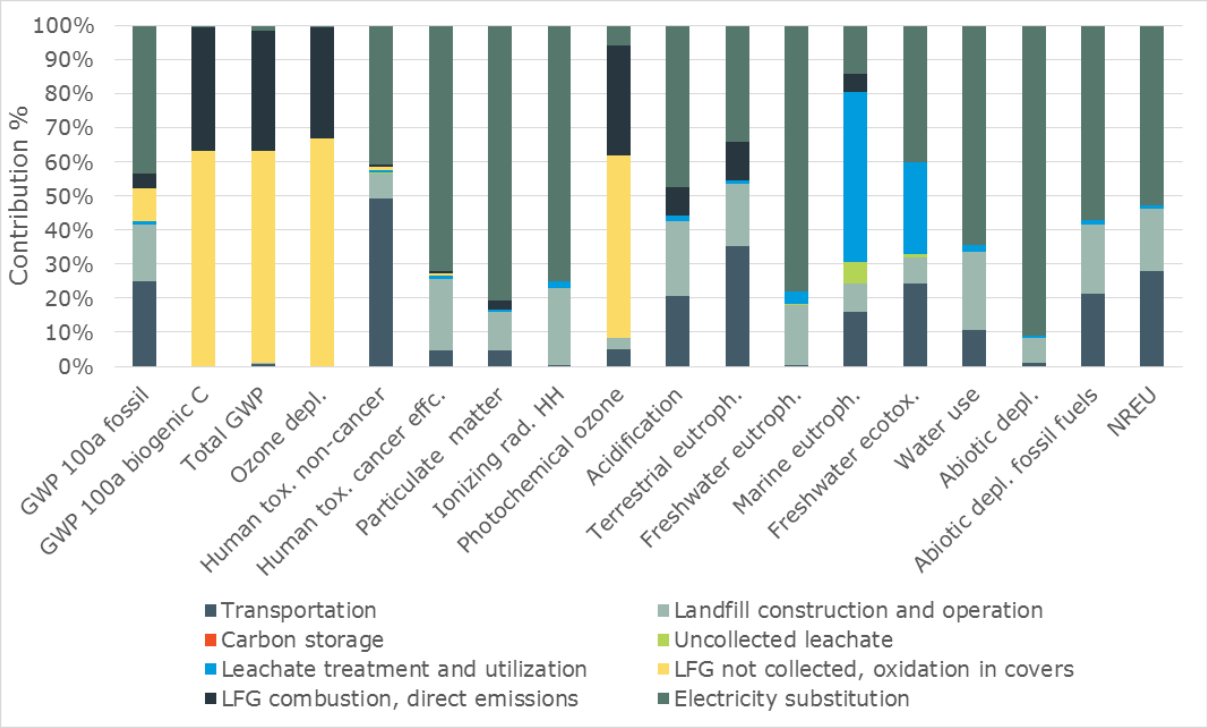

Figure 277. Contribution of case 5, Landfilling Bio-based starch mulcing film excl. soil.

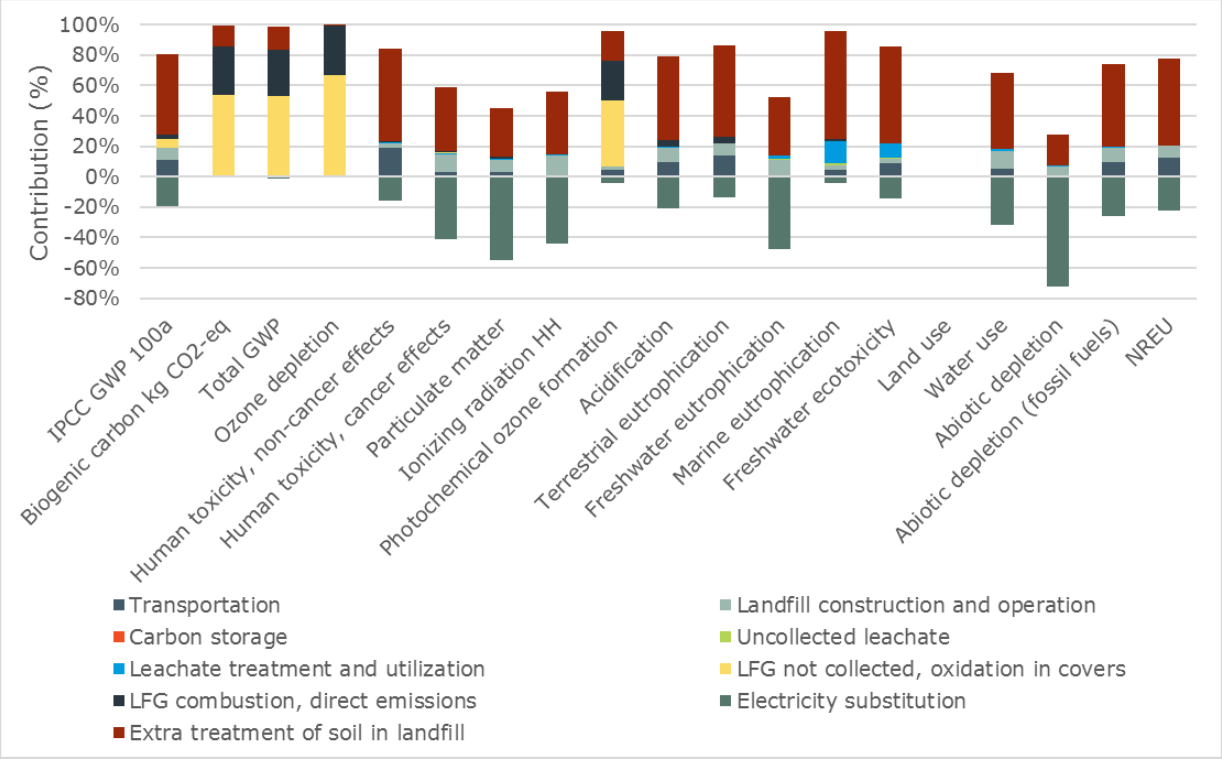

Figure 278. Contribution of case 5, Landfilling Bio-based starch mulching film including soil.

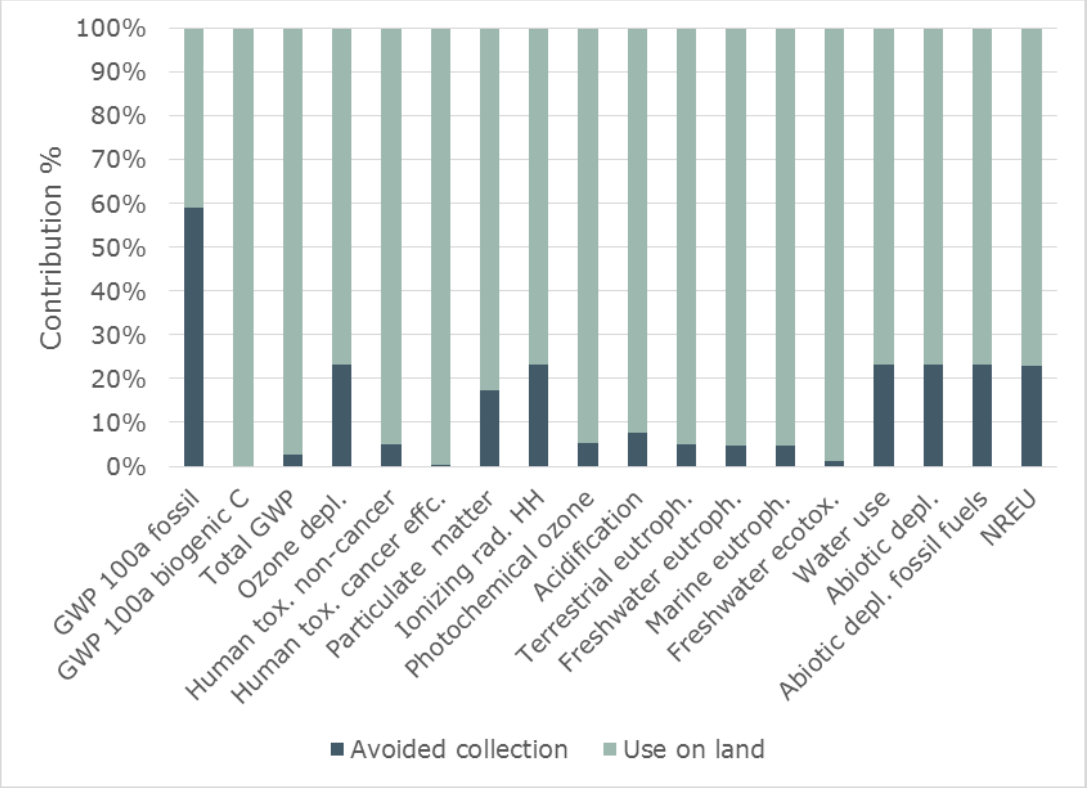

Figure 279. Contribution of case 5, In-situ Bio-based starch mulching film excl. soil (absolute values).

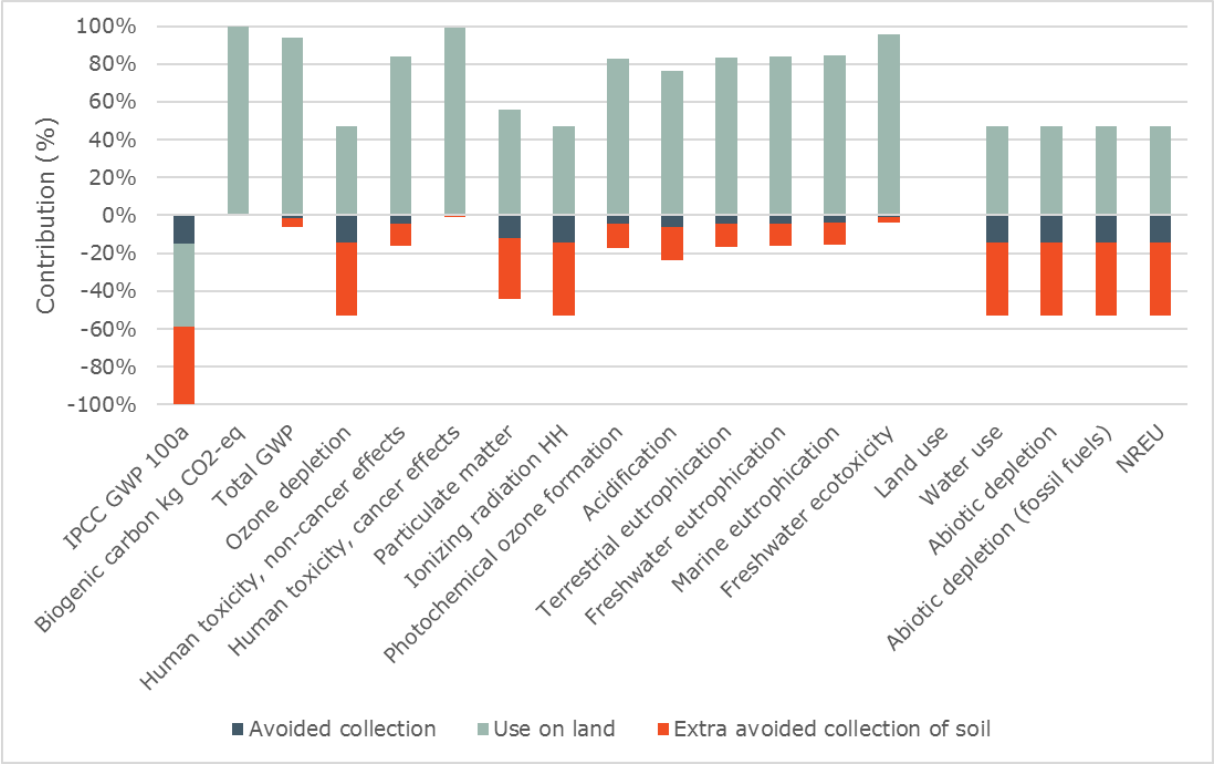

Figure 280. Contribution of case 5, In-situ Bio-based starch mulching film incl. soil.

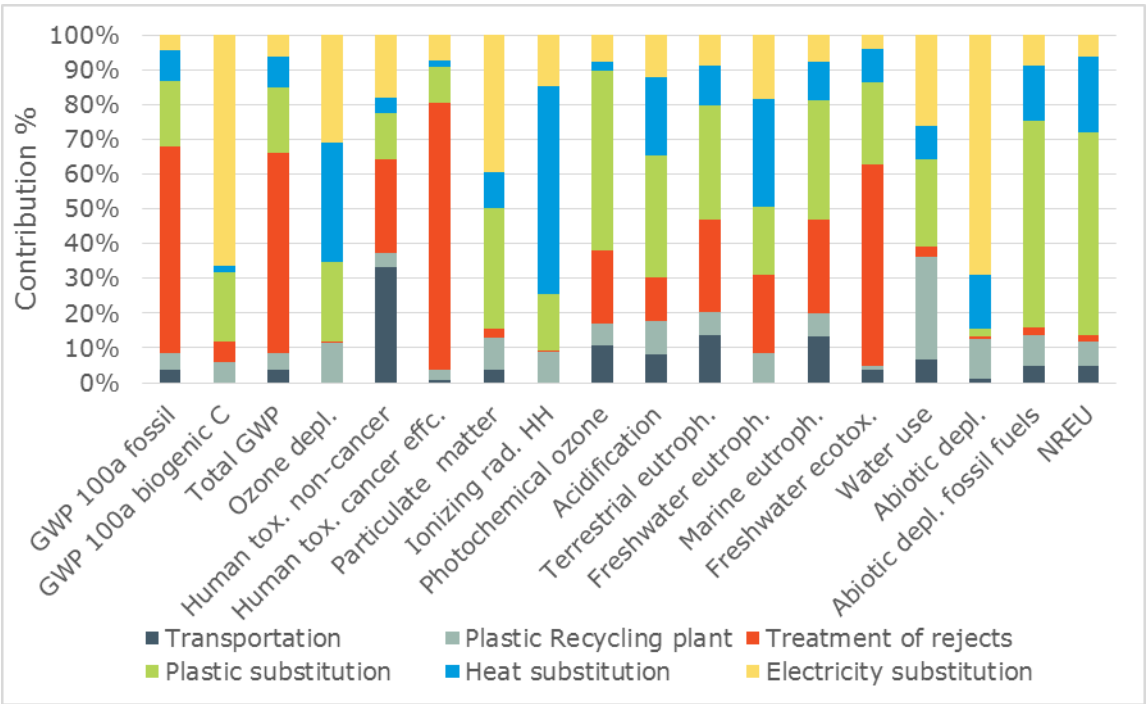

Figure 281. Contribution analysis for case 5, Plastics Recycling, Fossil excl. soil (absolute percentages).

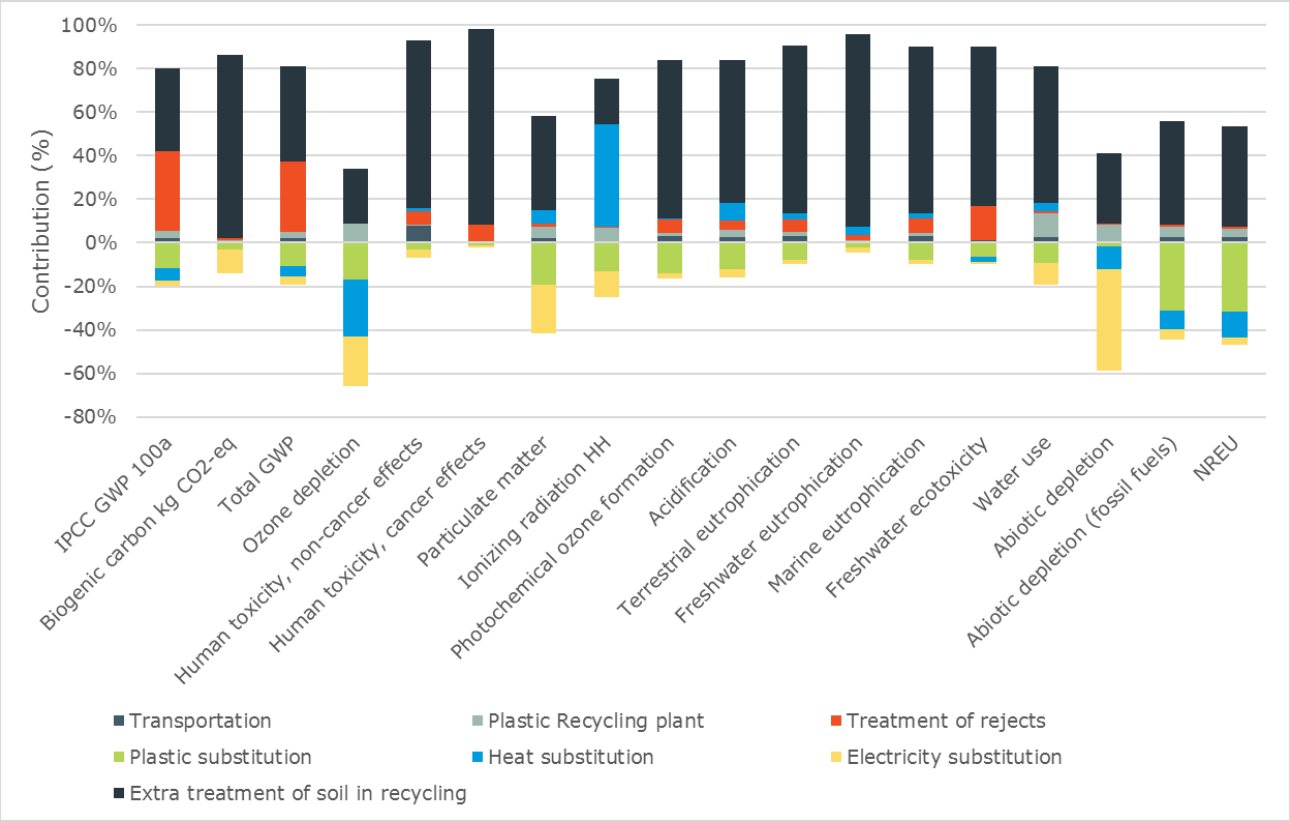

Figure 282. Contribution analysis for case 5, Plastics Recycling, Fossil including soil.

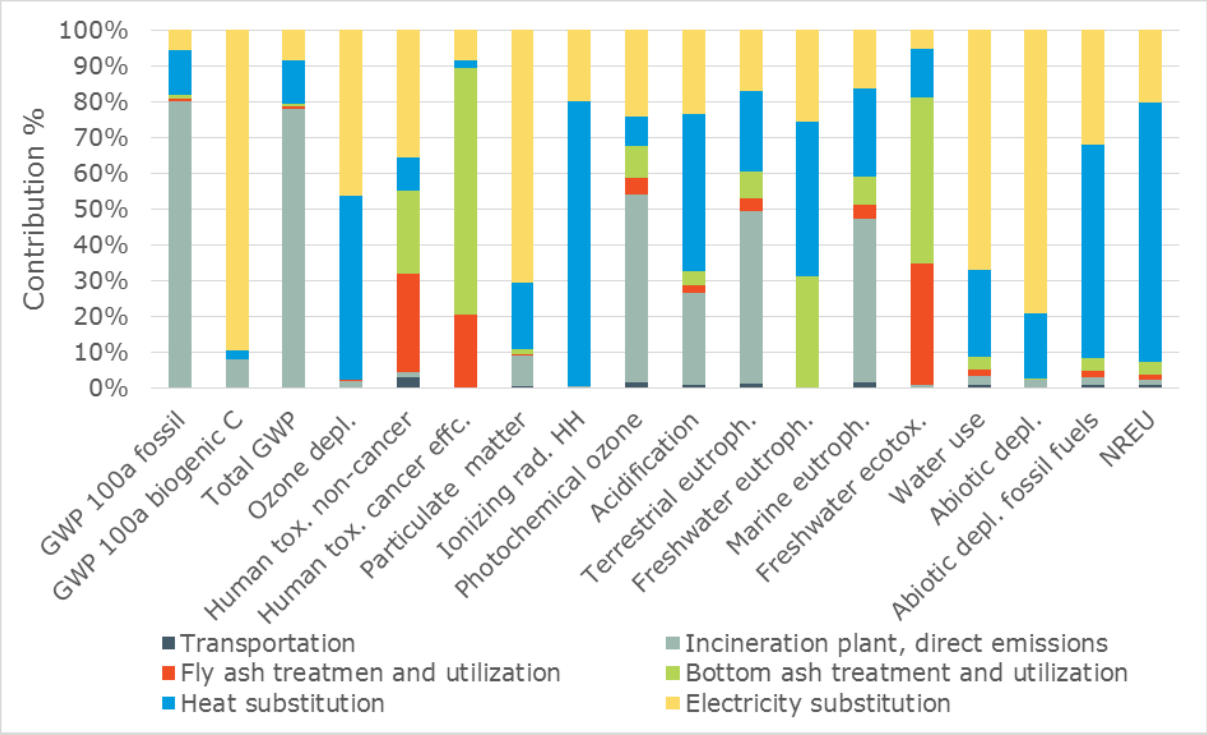

Figure 283. Contribution analysis for case 5, MSWI, Fossil excl. soil (absolute percentages).

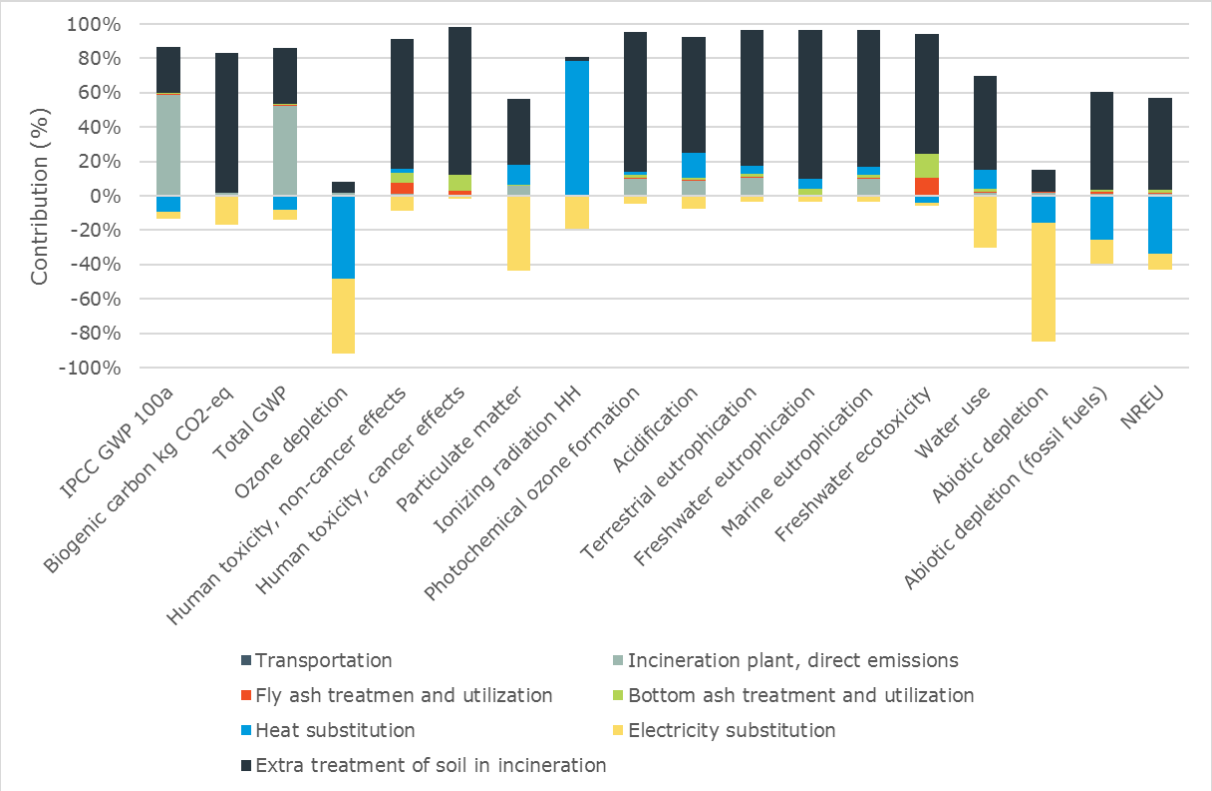

Figure 284. Contribution analysis for case 5, MSWI, Fossil including soil.

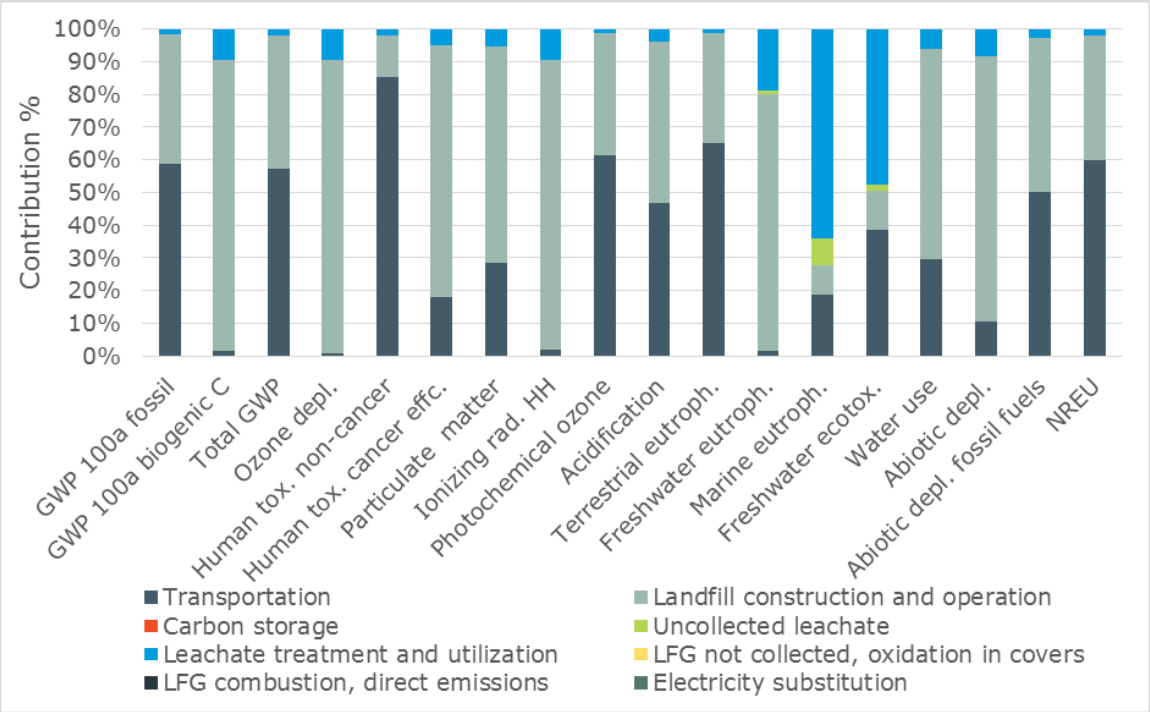

Figure 285. Contribution of case 5, Land filling, Fossil, excl. soil.

Case study 6: Food packing films

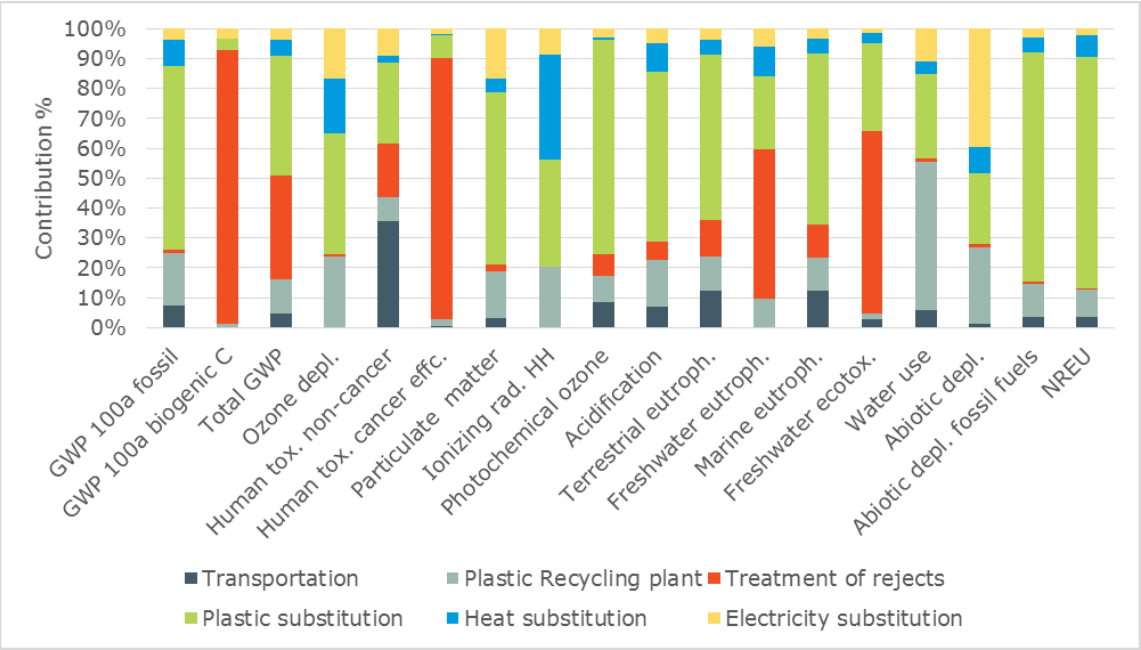

Figure 286. Contribution analysis of case 6 with plastics recycling BioPP excl. vegetable waste (absolute percentages).

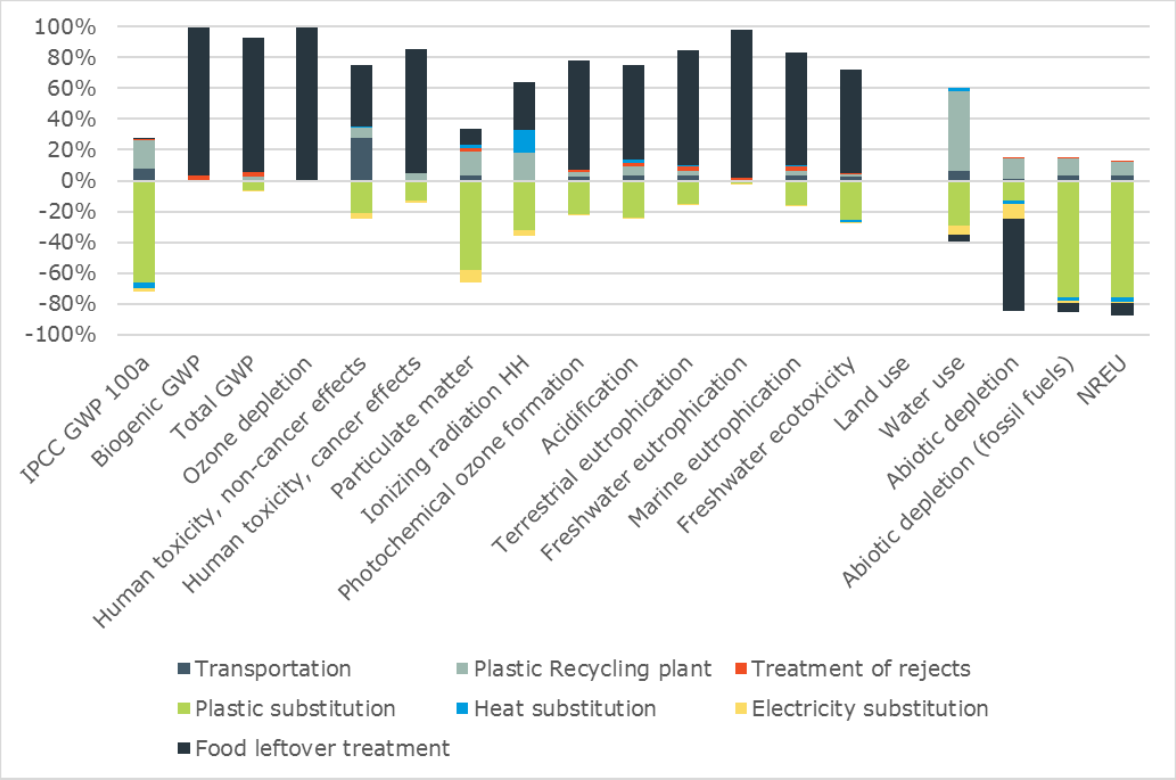

Figure 287. Contribution analysis of case 6 with plastics recycling BioPP including vegetable waste.

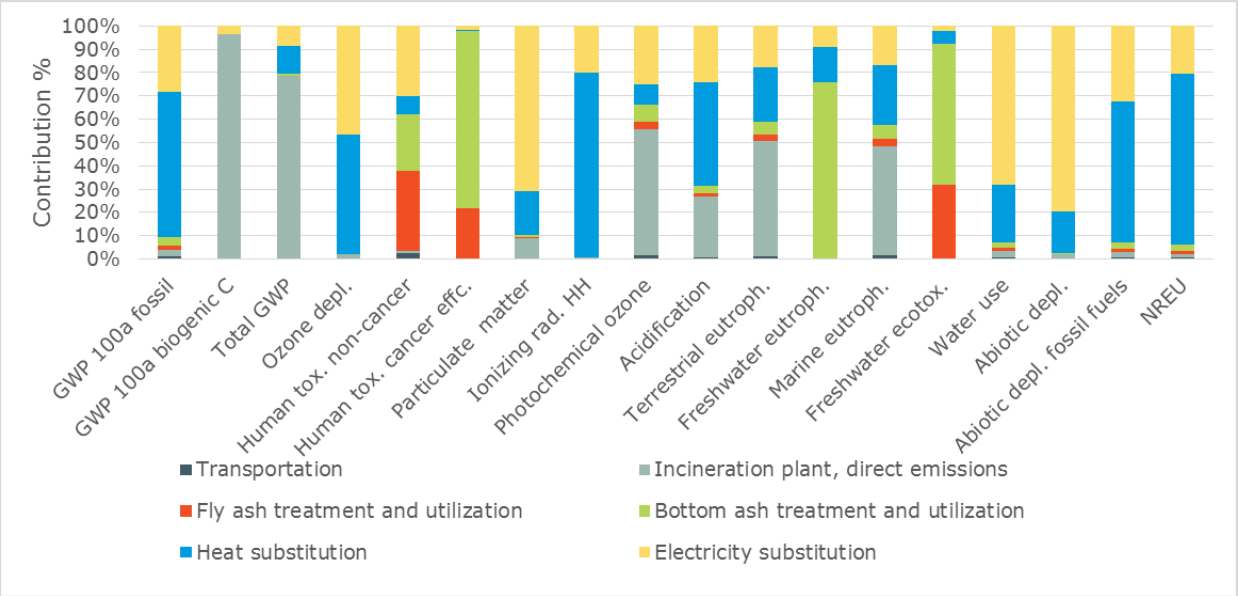

Figure 288. Contribution analysis of case 6 with MSW-Incineration BioPP excl. vegetable waste (absolute percentages).

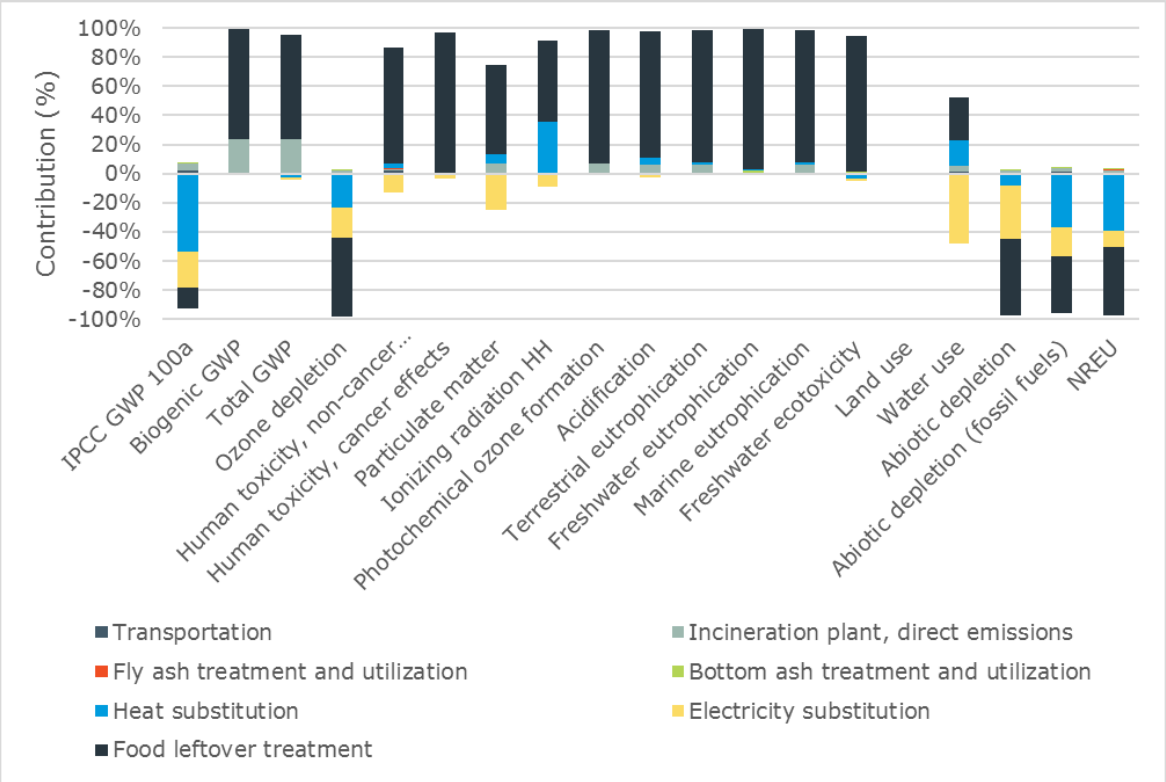

Figure 289. Contribution analysis of case 6 with MSWI BioPP including vegetable waste.

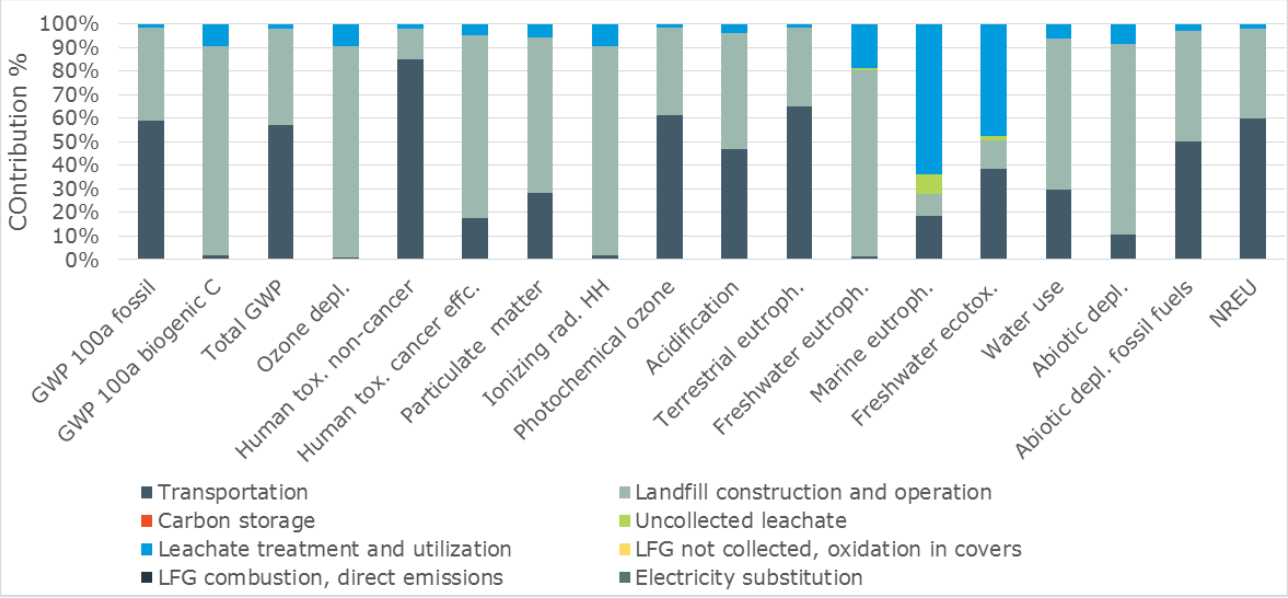

Figure 290. Contribution analysis for case 6, Landfilling BioPP excl. vegetable waste (absolute percentages)

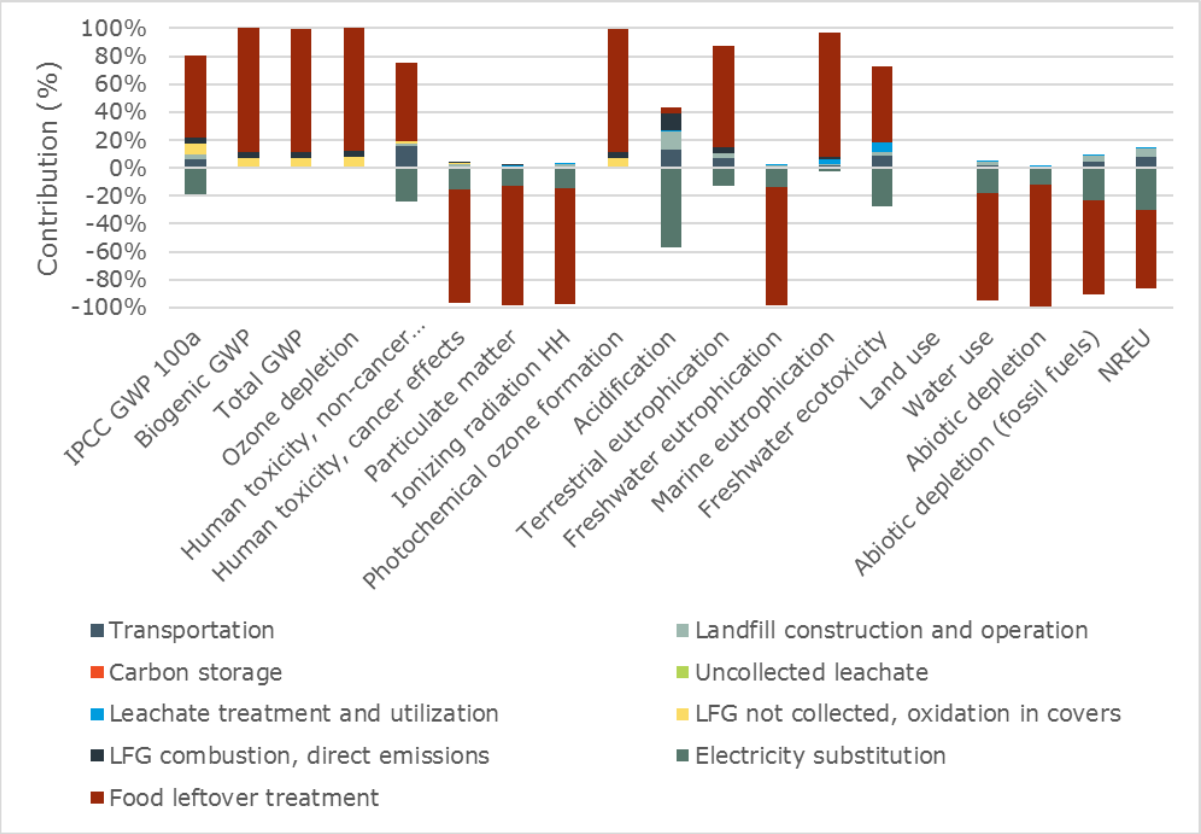

Figure 291. Contribution analysis for case 6, Landfilling BioPP including vegetable waste.

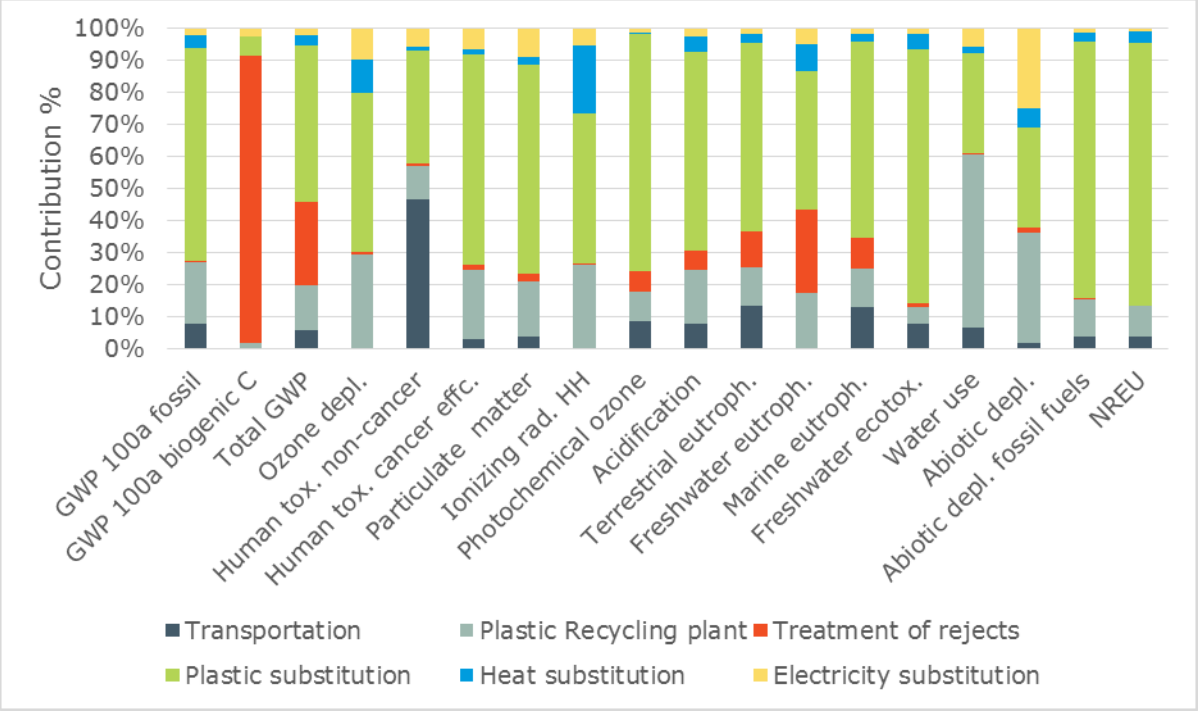

Figure 292. Contribution of case 6 Plastics recycling PLA excl. vegetable waste.

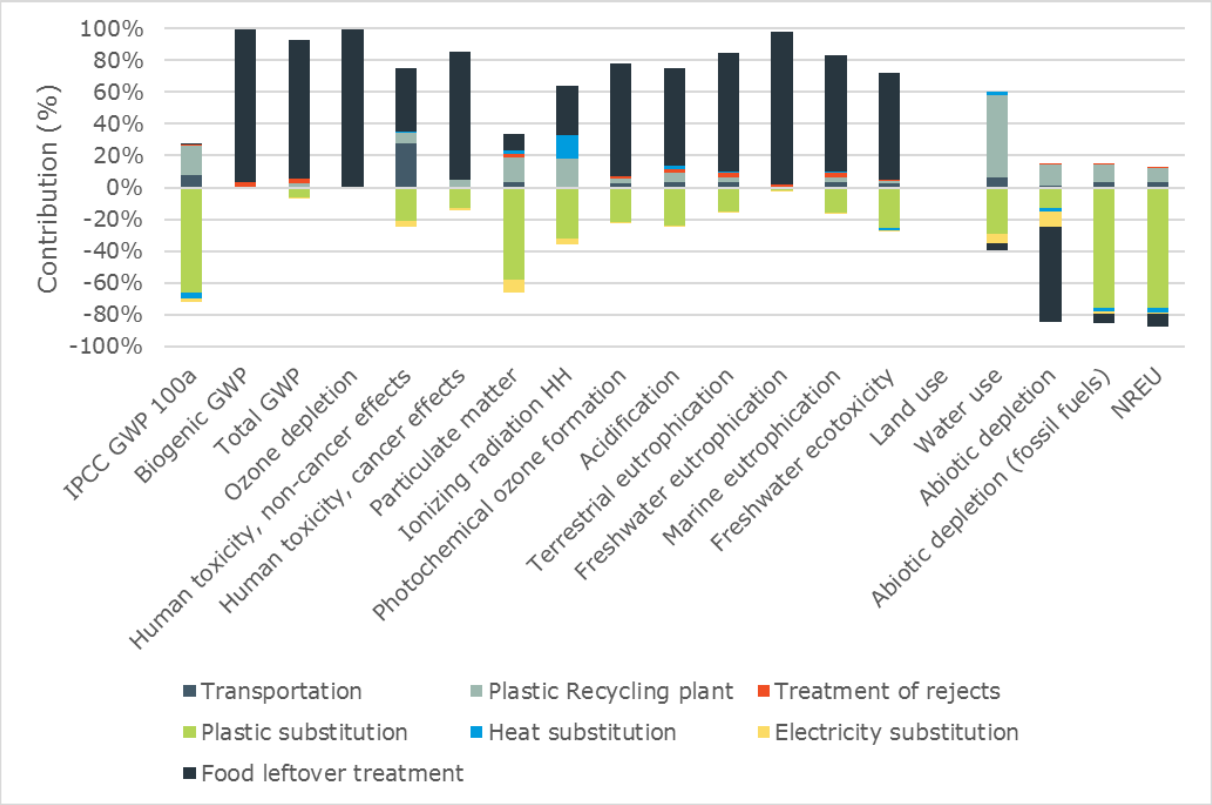

Figure 293. Contribution analysis for case 6 PLA food packaging film incl. vegetable waste

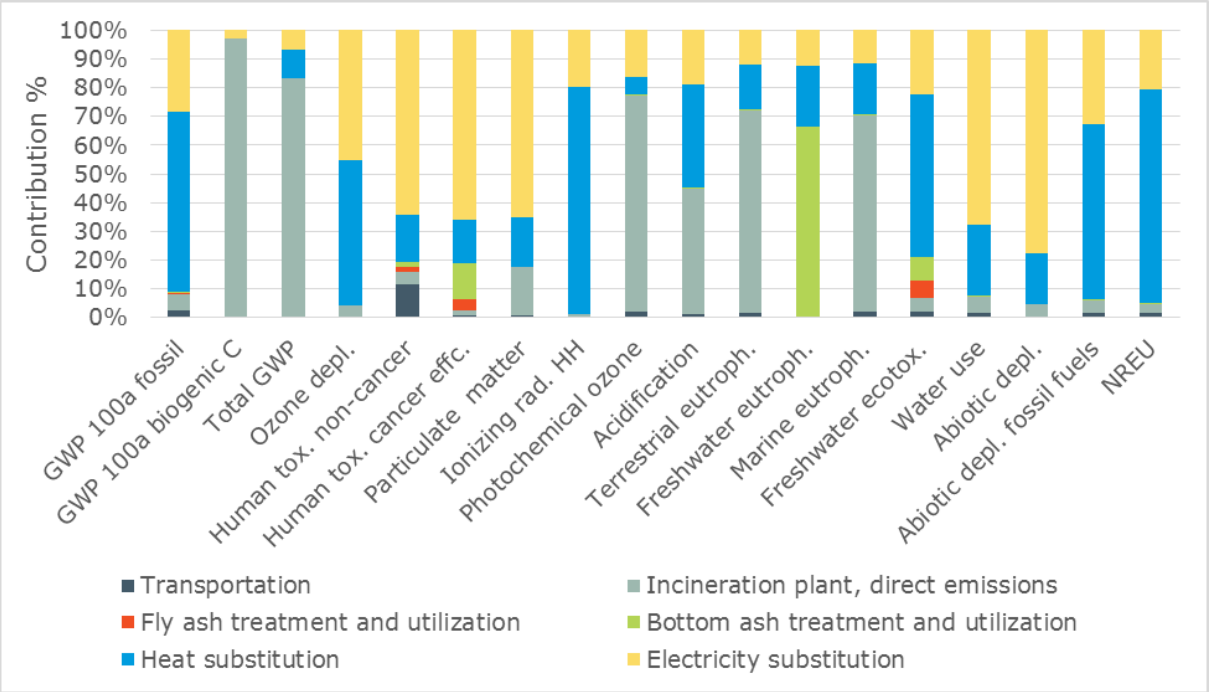

Figure 294. Contribution analysis for case 6, Incineration PLA excl. vegetable waste (absolute percentages).

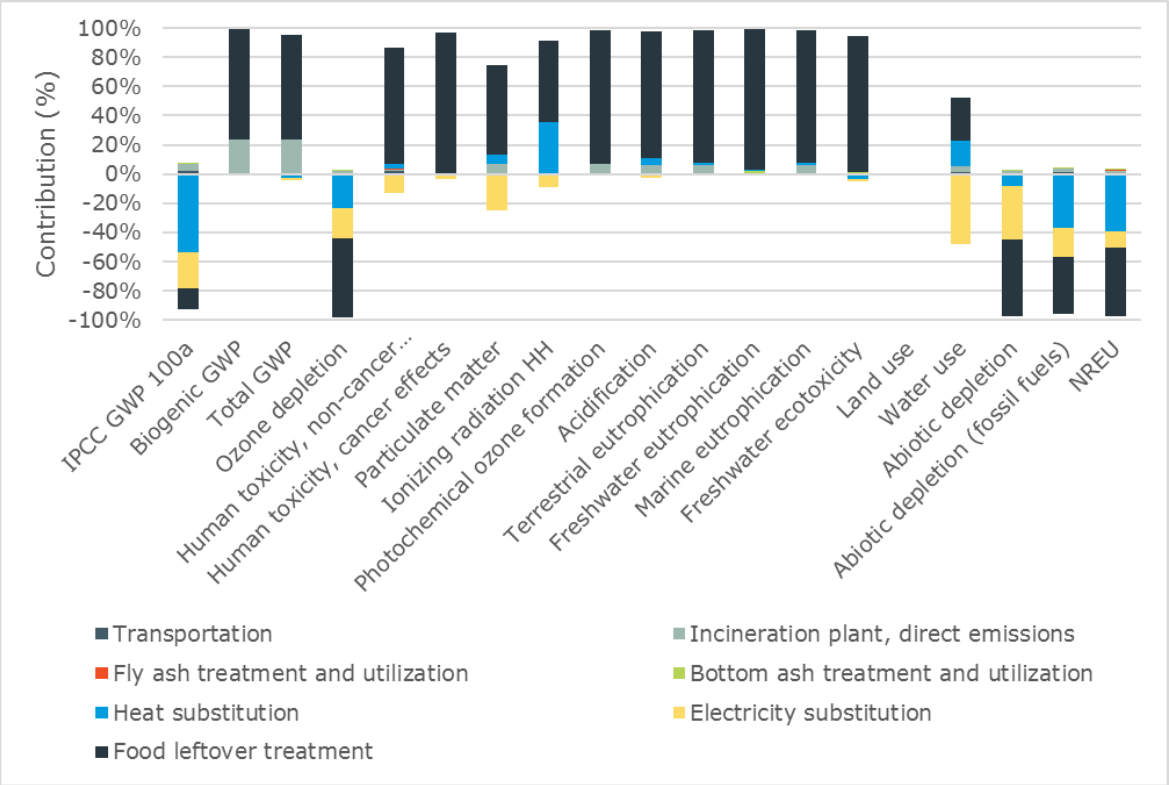

Figure 295. Contribution analysis for case 6, Incineration PLA incl. vegetable waste

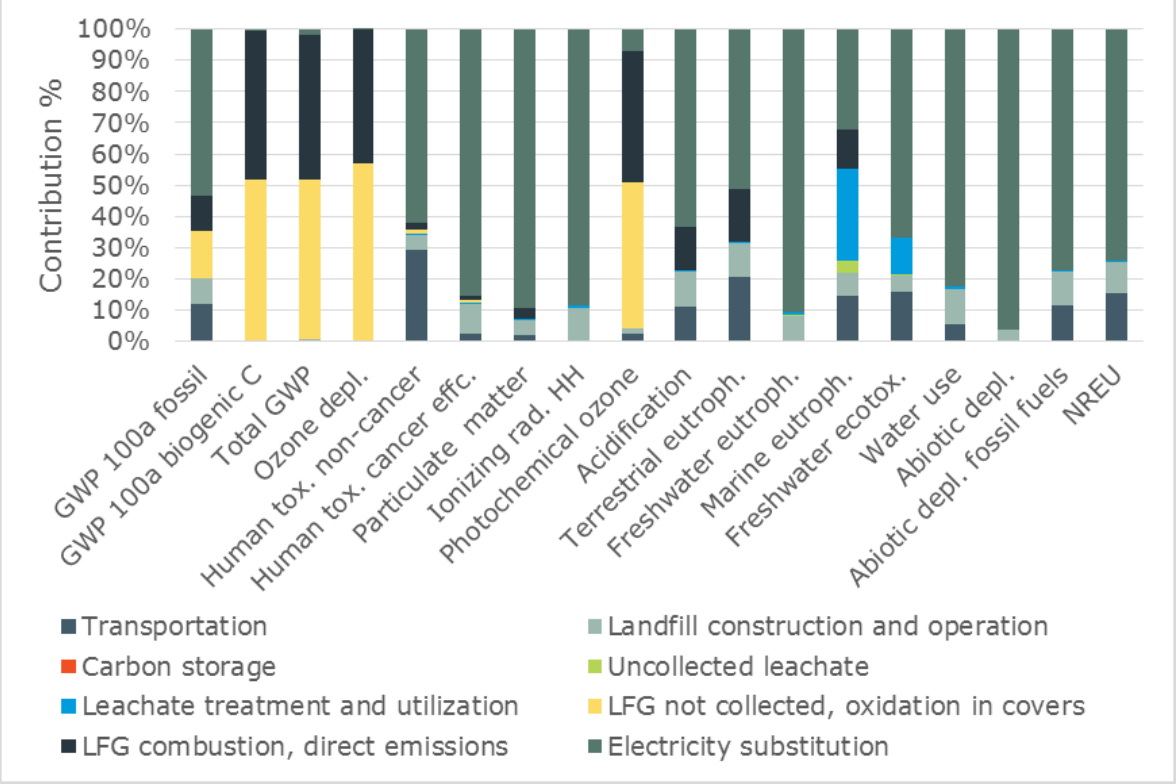

Figure 296. Contribution analysis for case 6, Landfilling, PLA excl. vegetable waste (absolute percentages).

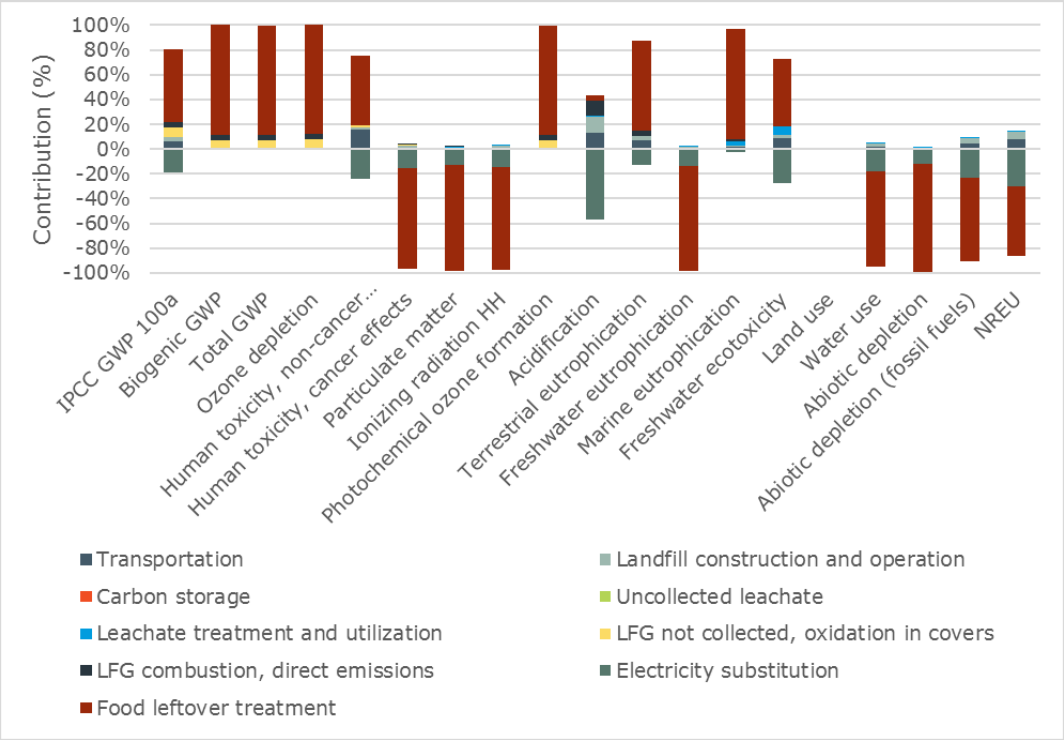

Figure 297. Contribution analysis for case 6, Landfilling, PLA incl. vegetable waste.

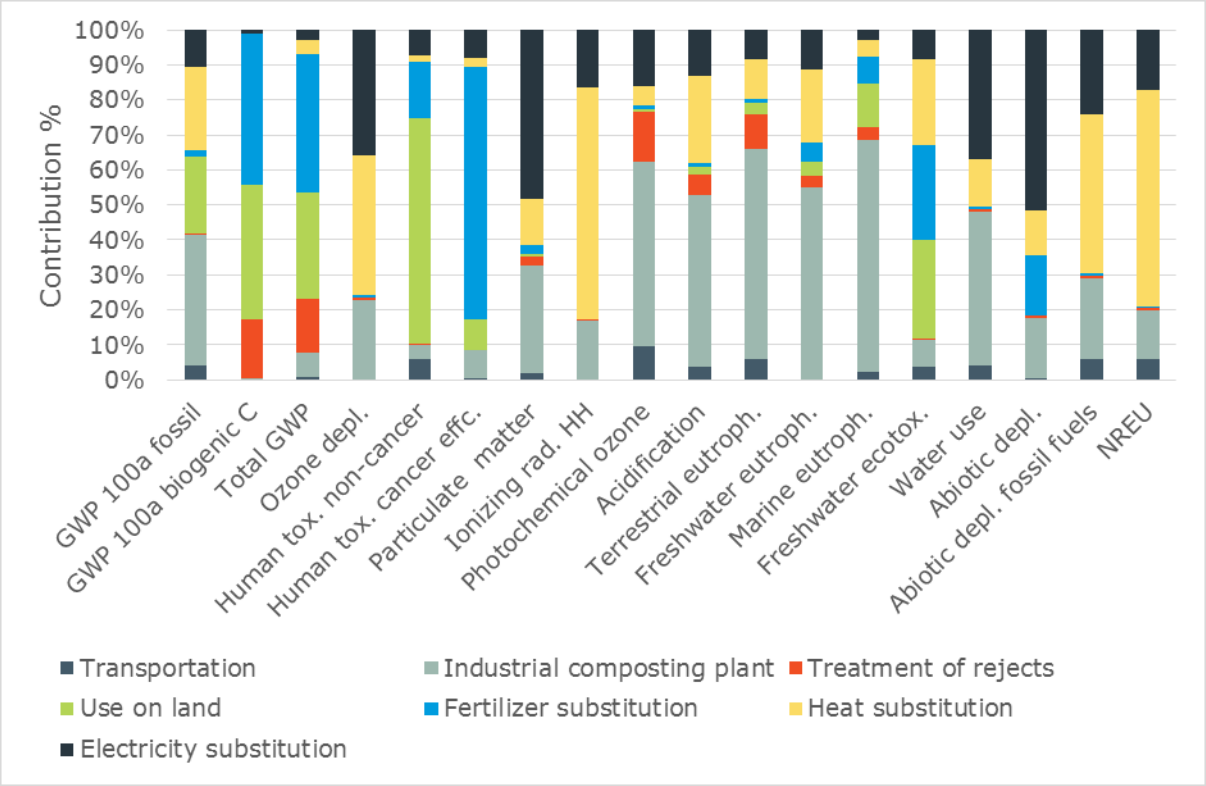

Figure 298. Contribution analysis for case 6, Composting PLA excl. vegetable waste (absolute percentages).

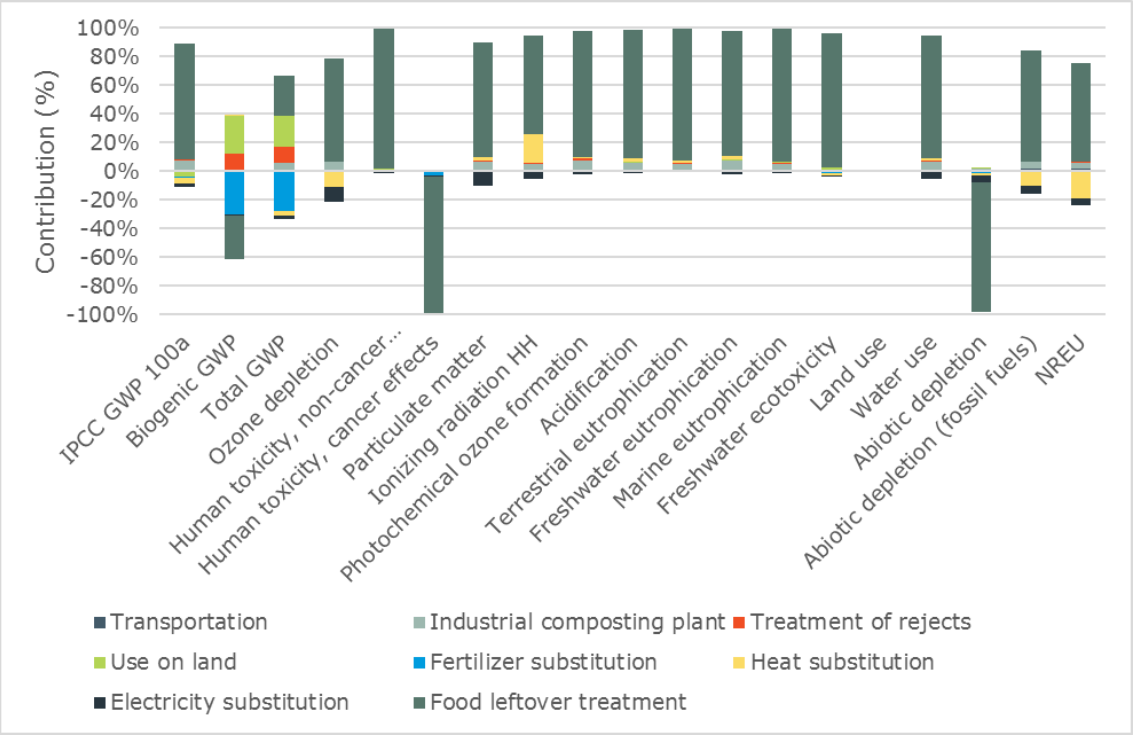

Figure 299. Contribution analysis for case 6, Composting PLA incl. vegetable waste.

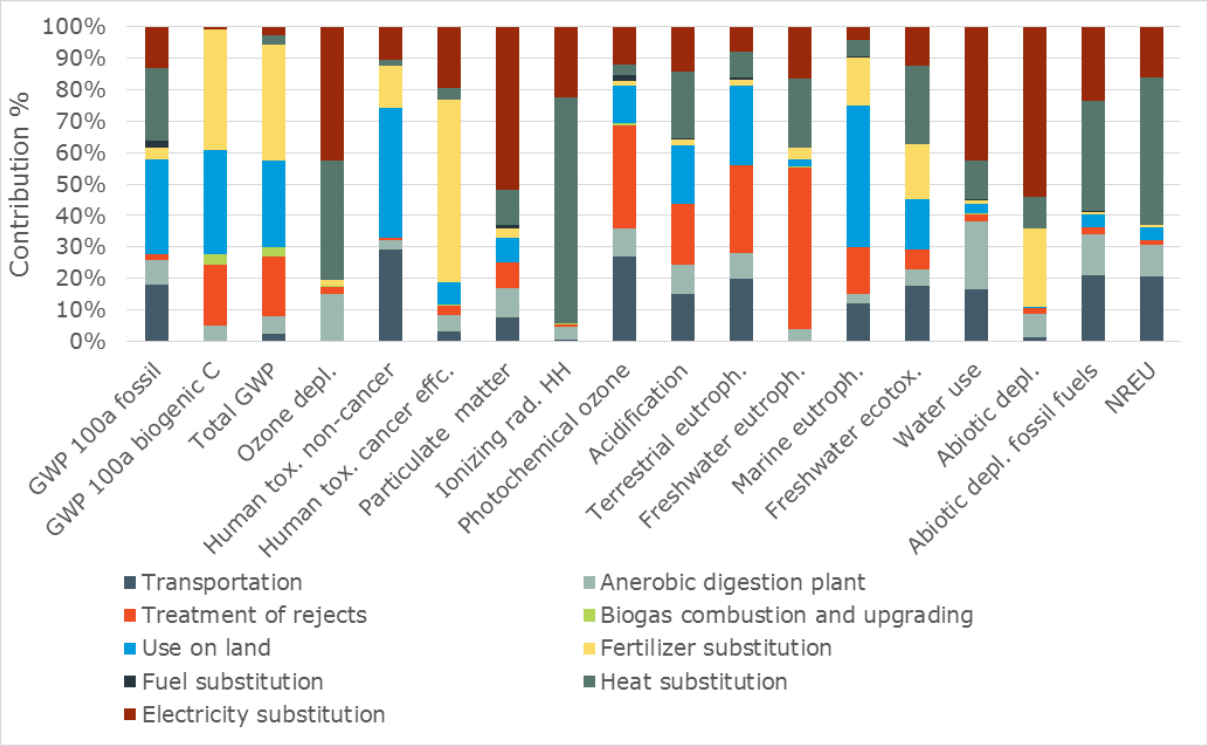

Figure 300. Contribution analysis of case 6, AD, PLA excl. vegetable waste (absolute percentages).

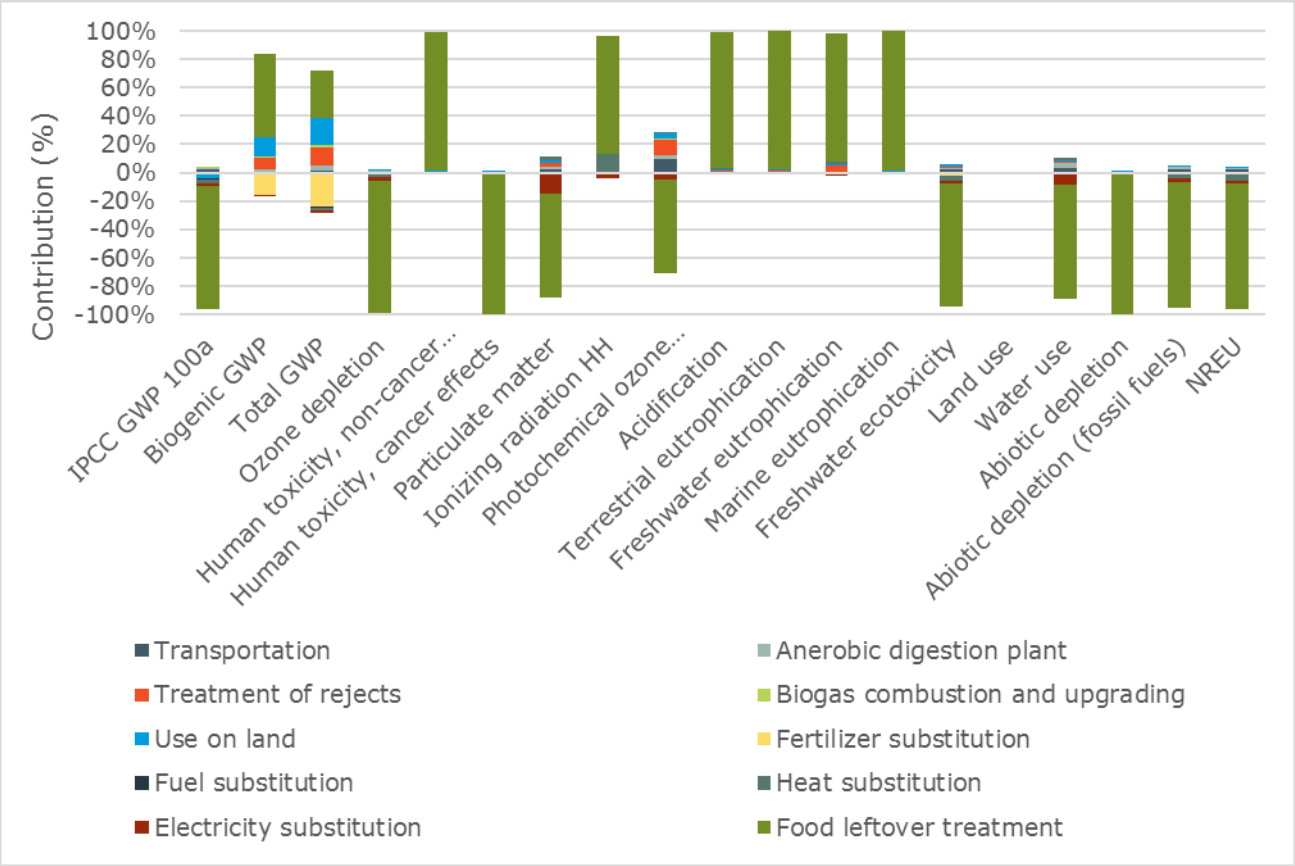

Figure 301. Contribution analysis of case 6, AD, PLA incl. vegetable waste.

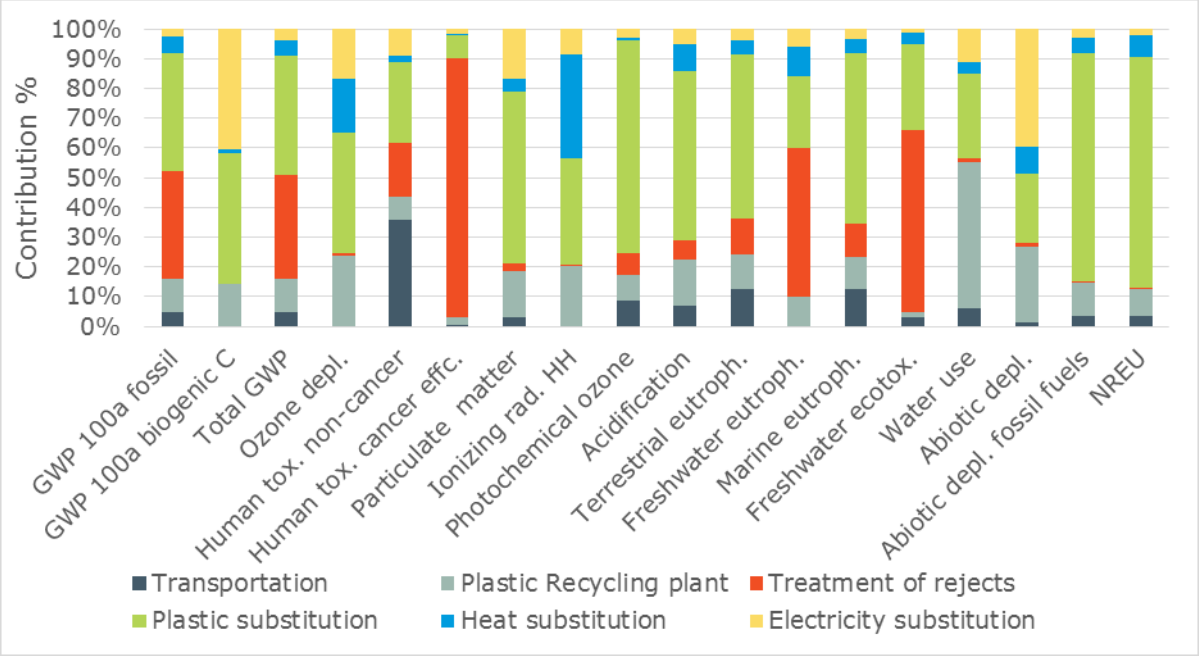

Figure 302. Contribution of case 6, Plastics recycling PP, excl. vegetable waste (absolute percentages).

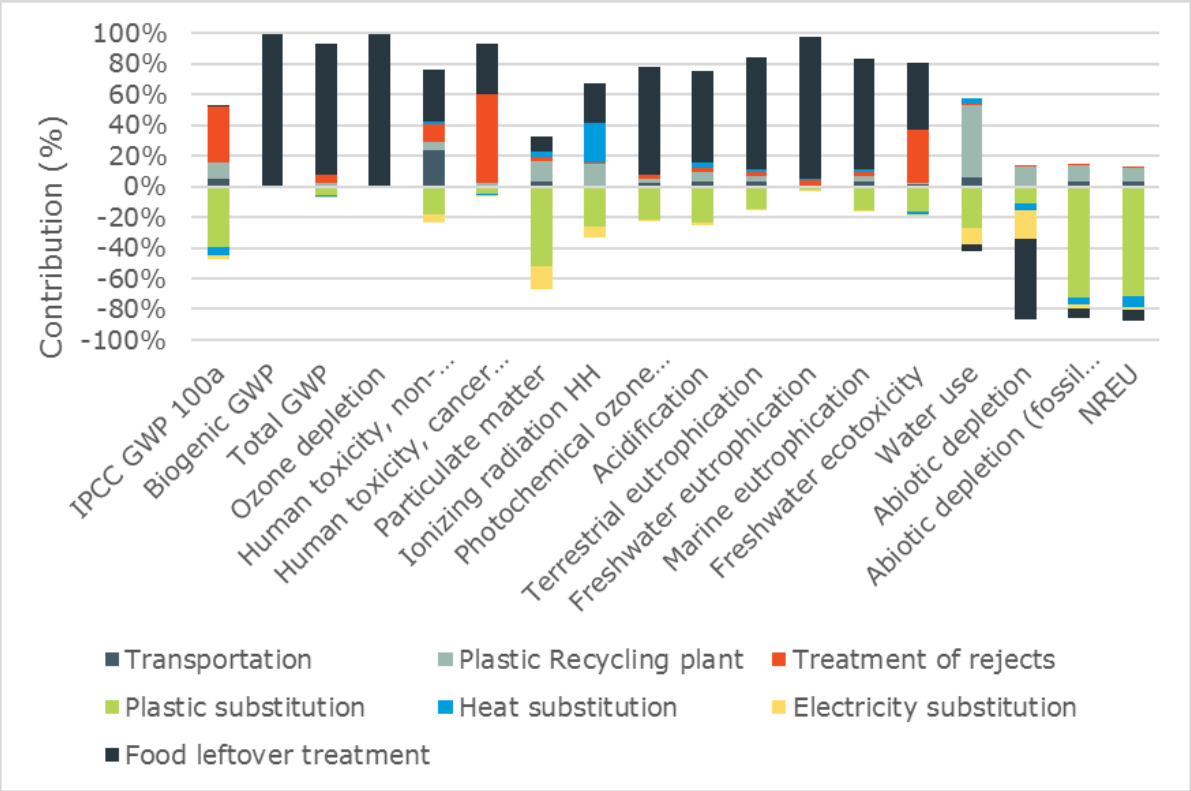

Figure 303. Contribution of case 6, Plastics recycling PP, incl. vegetable waste.

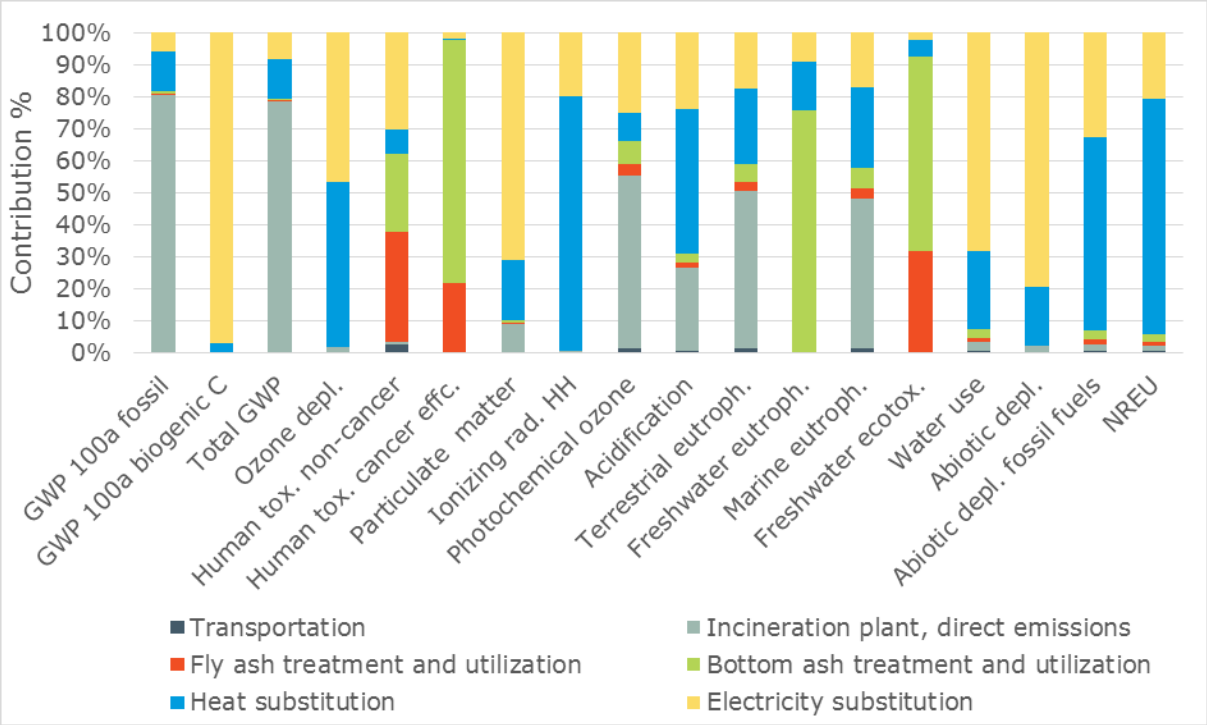

Figure 304. Contribution of case 6, MSWI, PP excl. vegetable waste (absolute percentages).

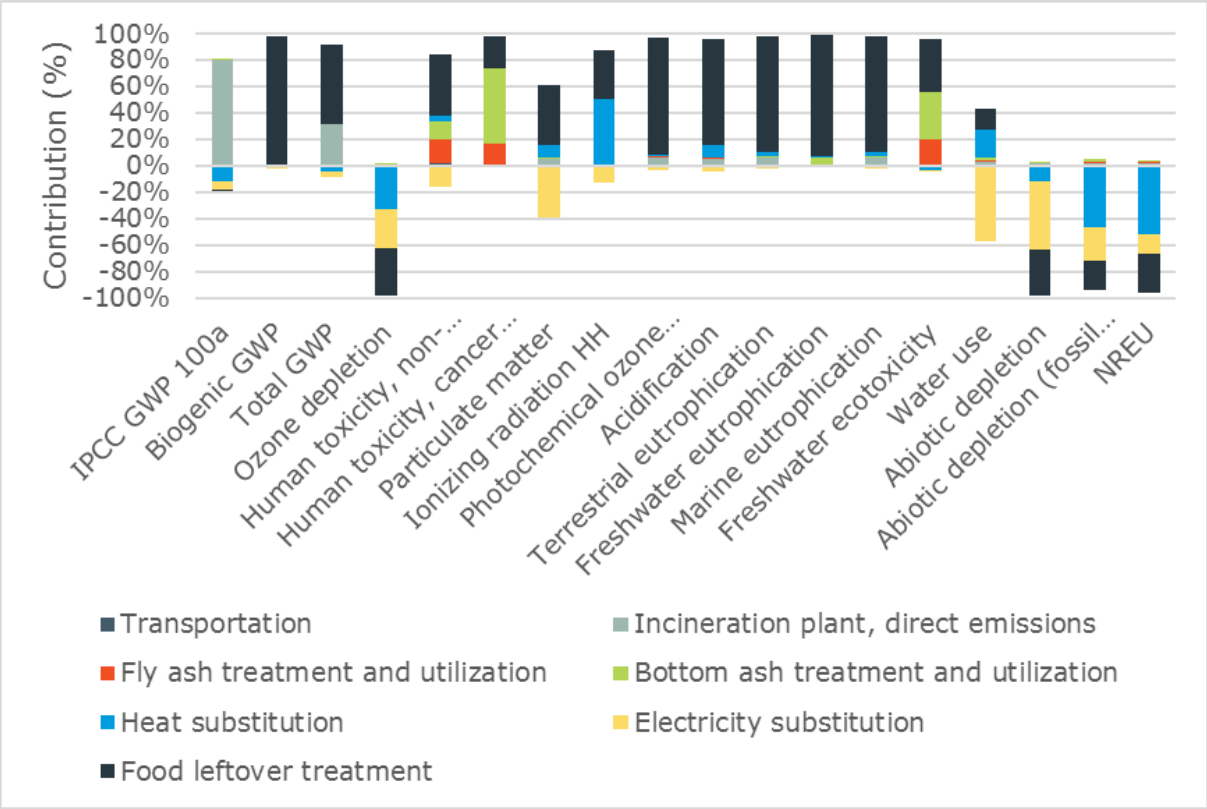

Figure 305. Contribution of case 6, MSWI, PP incl. vegetable waste.

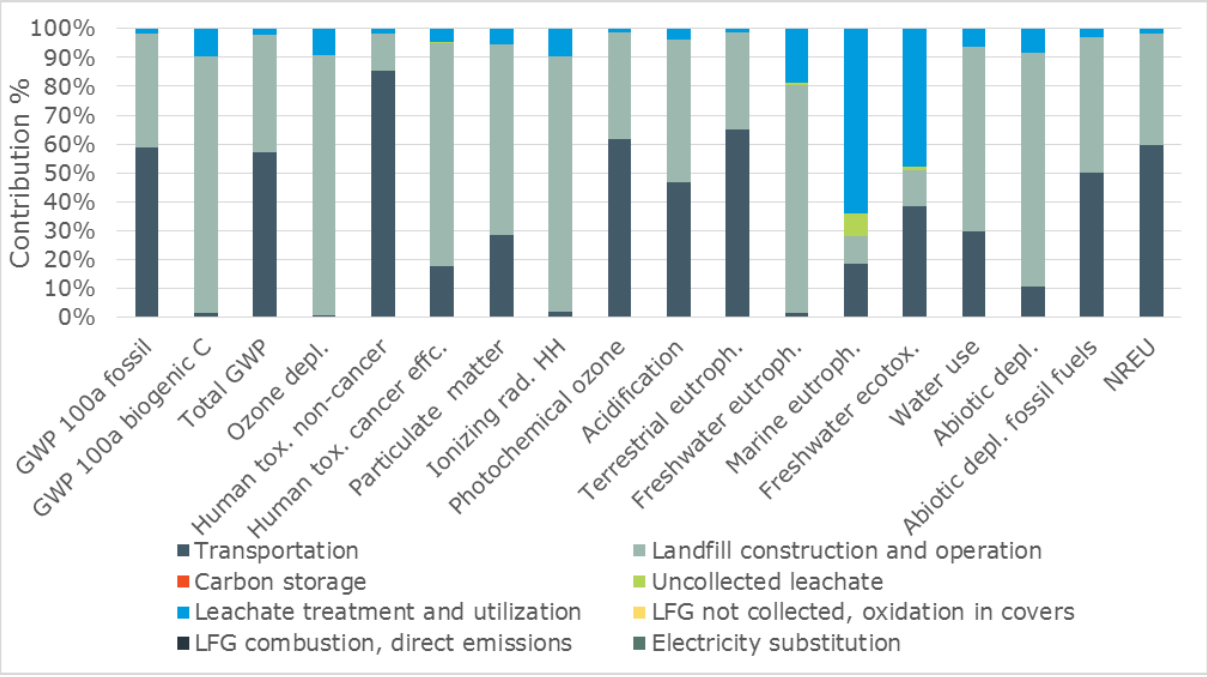

Figure 306. Contribution analysis for case 6, Landfilling, PP excl. vegetable waste (absolute percentages).

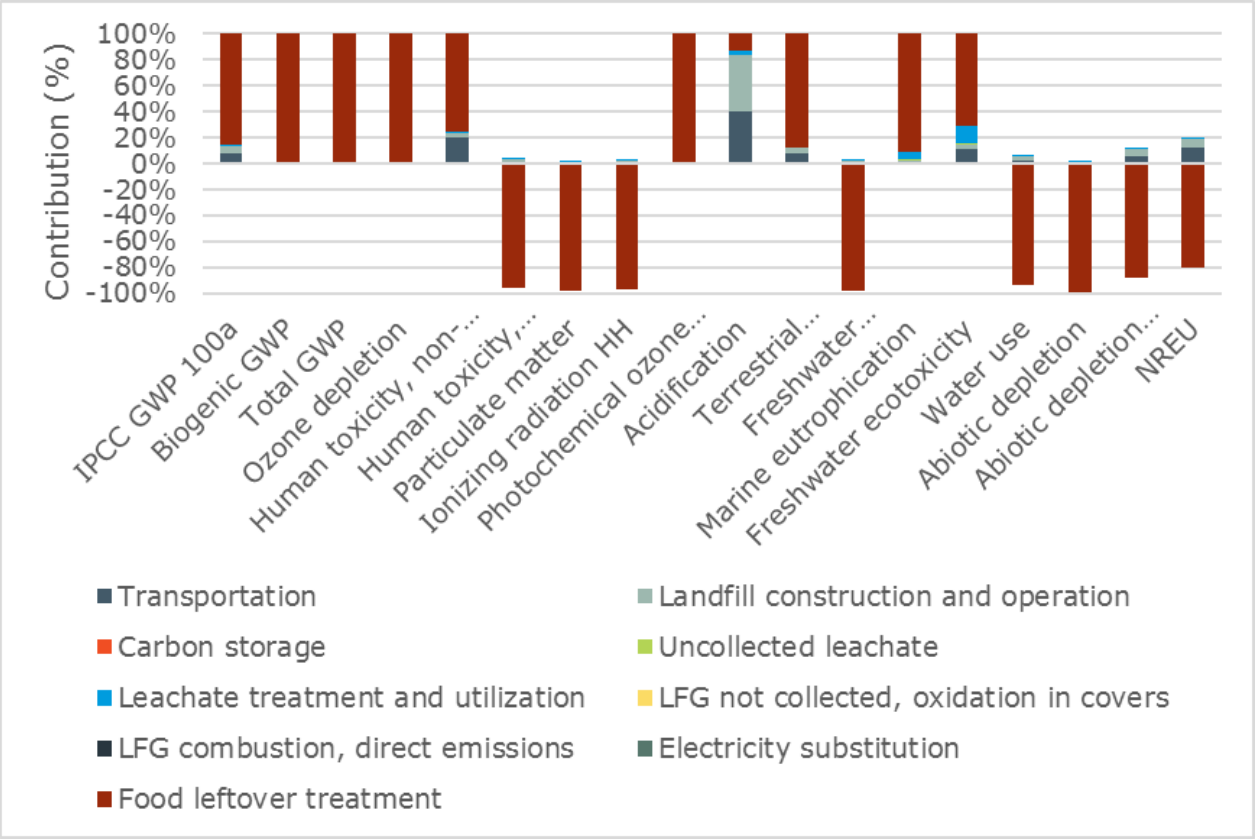

Figure 307. Contribution analysis for case 6, Landfilling, PP incl. vegetable waste.

Case study 7: Single-use plastic carrier bags

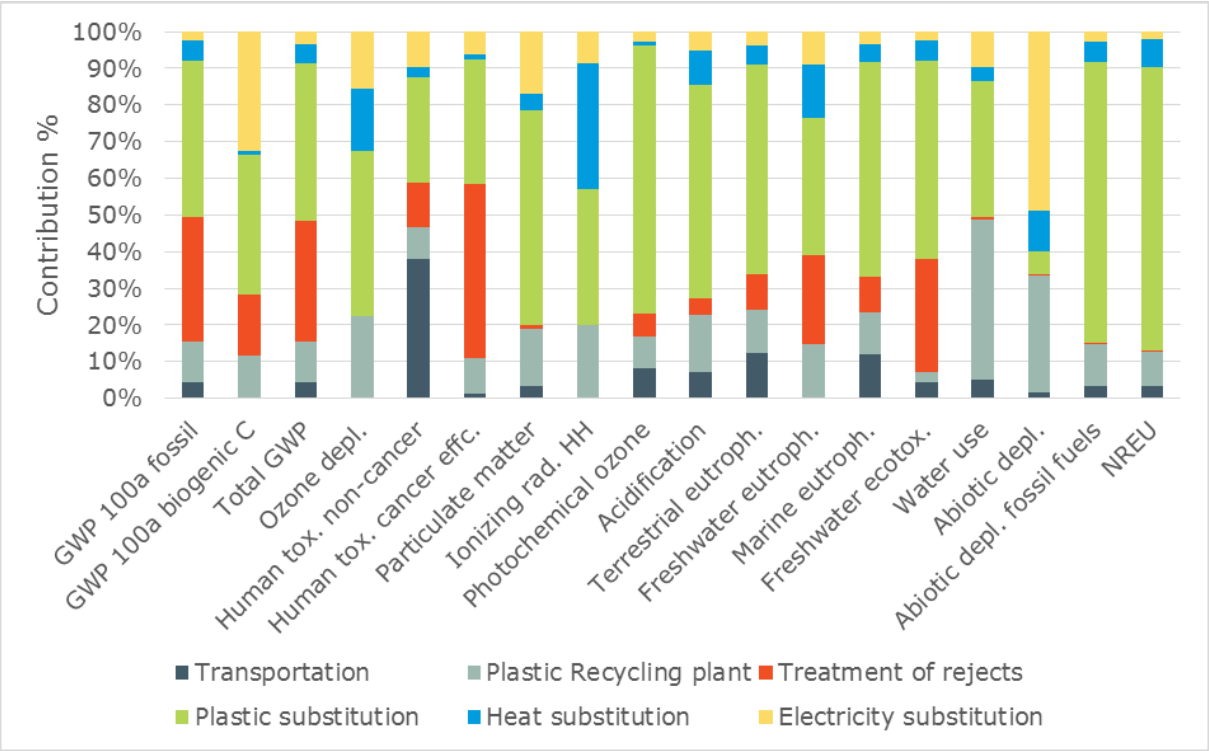

Figure 308. Contribution analysis of case 7 Plastics recycling of petrocehmcial LDPE (absolute percentages).

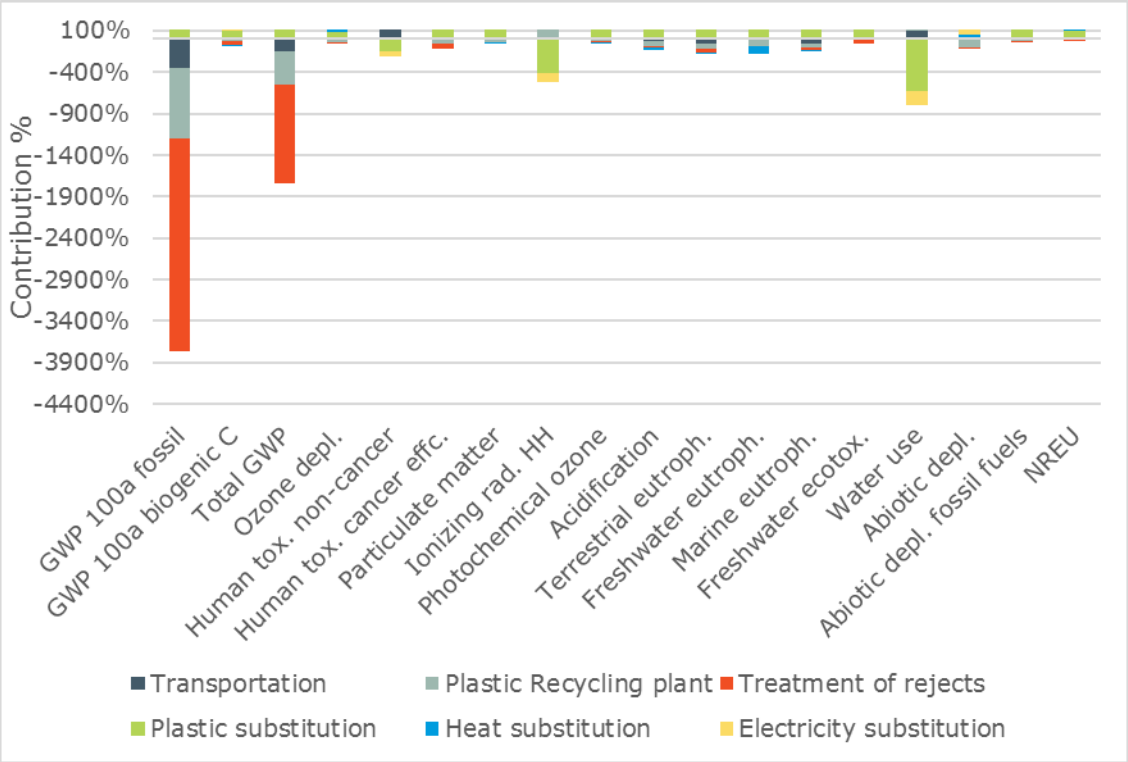

Figure 309. Contribution analysis of case 7 Plastics recycling of petrocehmcial LDPE.

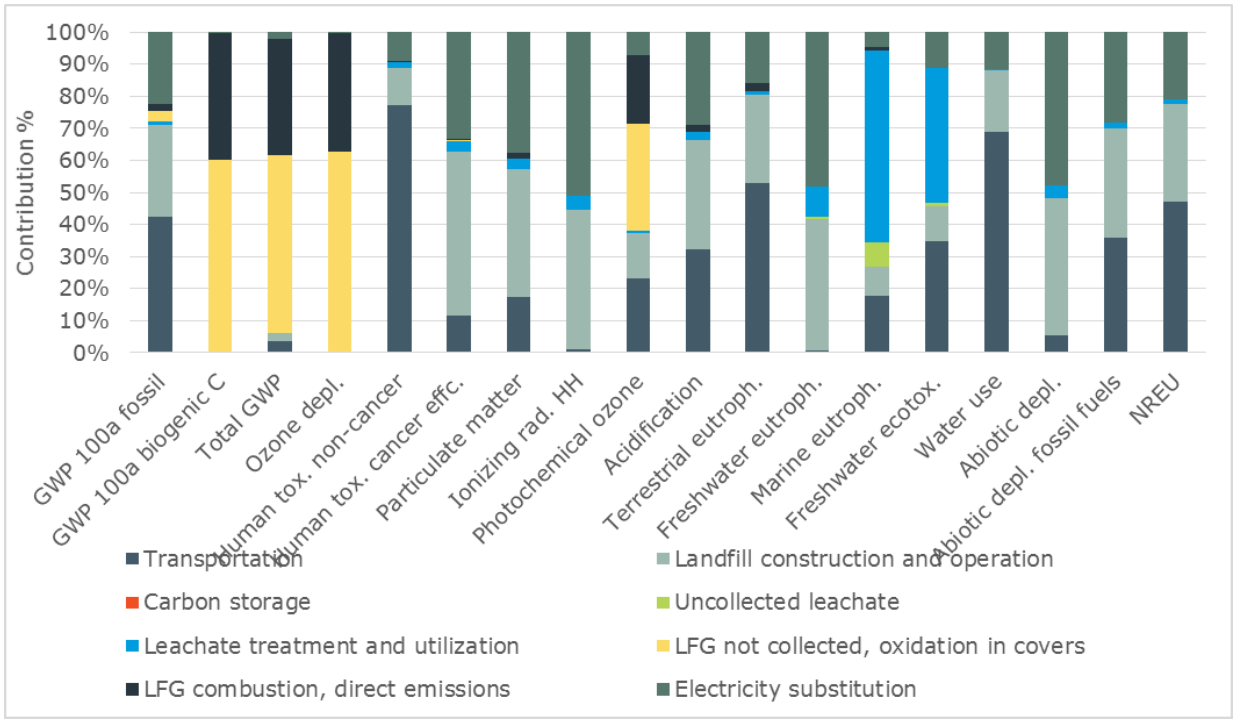

Figure 310. Contribution analysis of case 7 with landfilling petrochemical LDPE (absolute percentages).

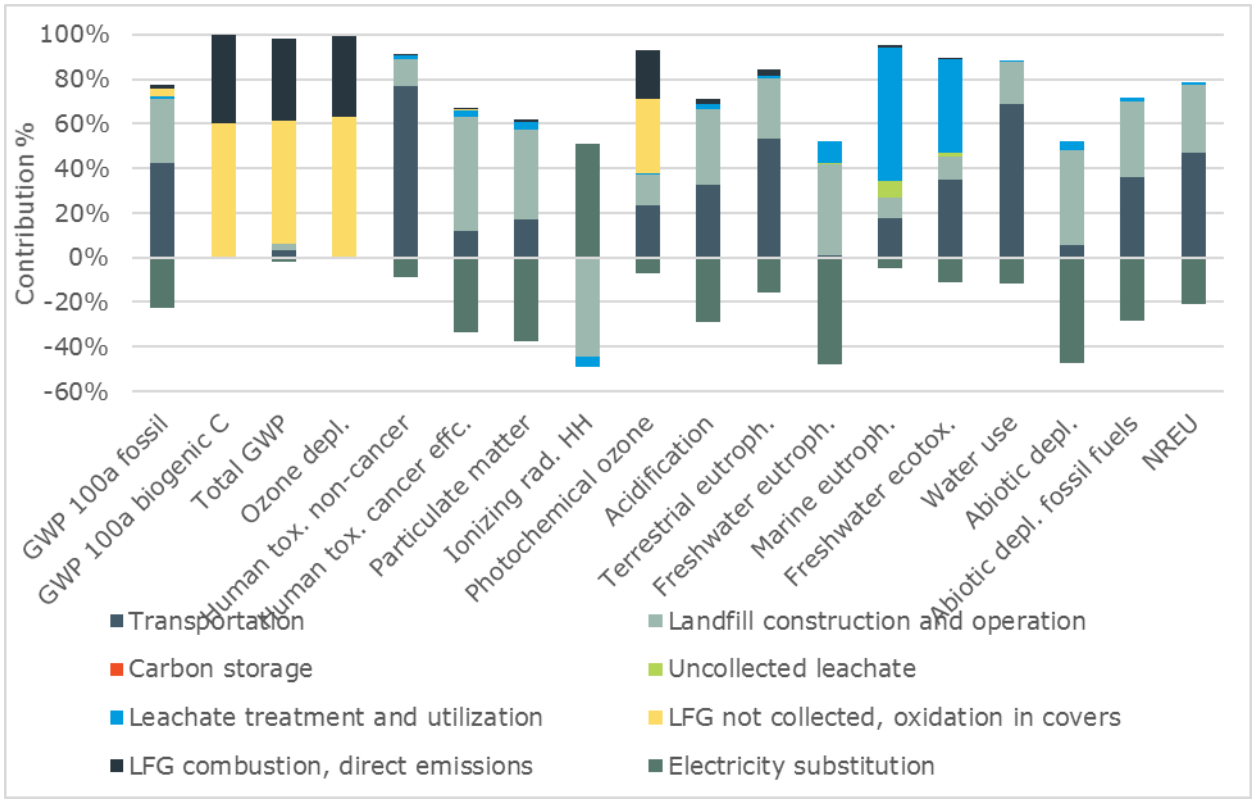

Figure 311. Contribution analysis of case 7 with landfilling petrochemical LDPE.

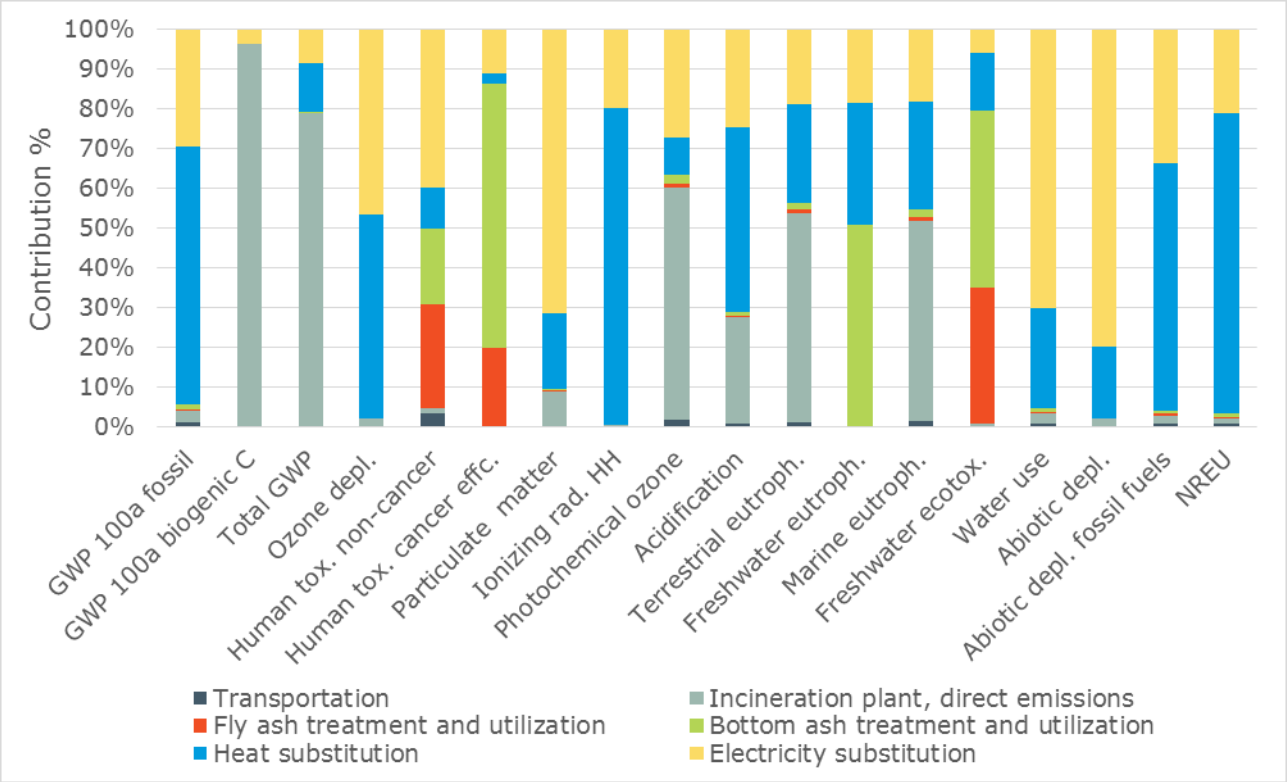

Figure 312. Contribution analysis of case 7 with incineration petrochemical LDPE (absolute percentages).

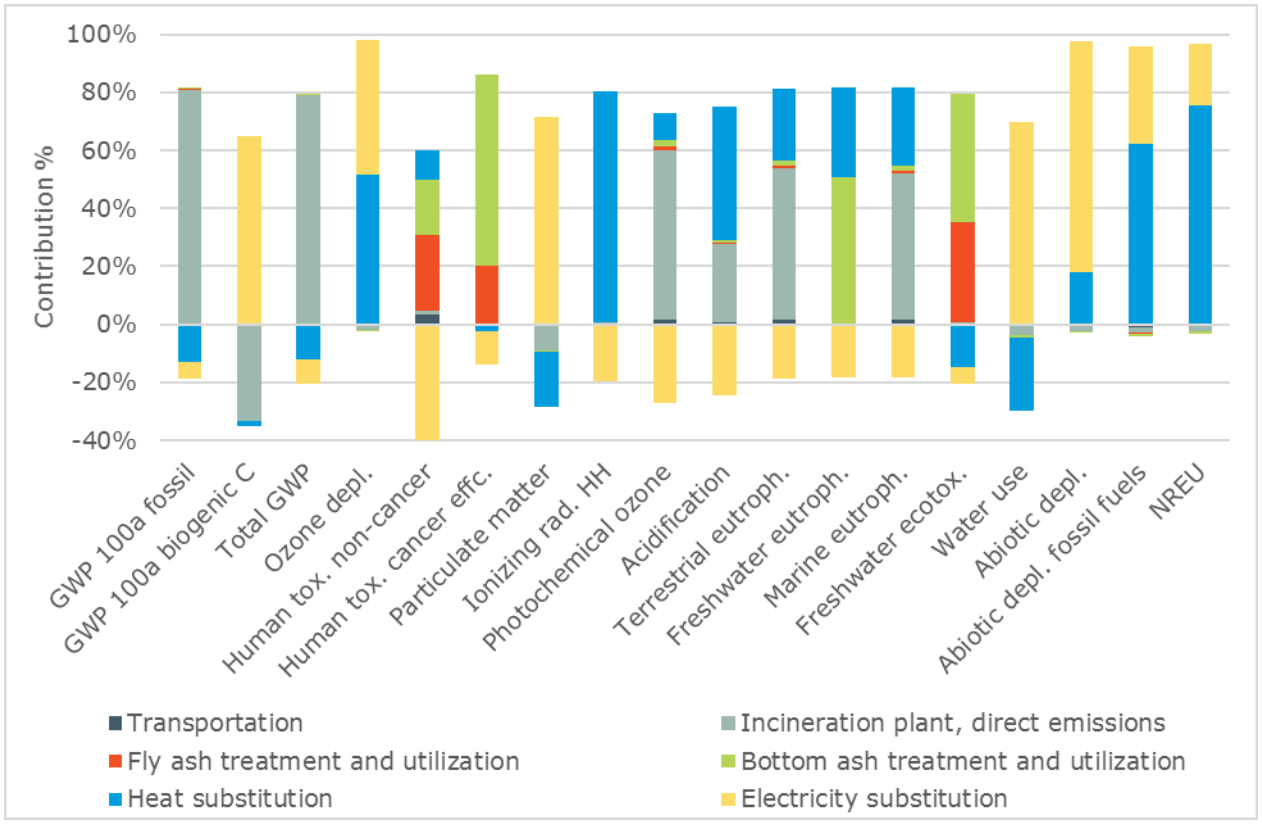

Figure 313. Contribution analysis of case 7 with incineration petrochemical LDPE.

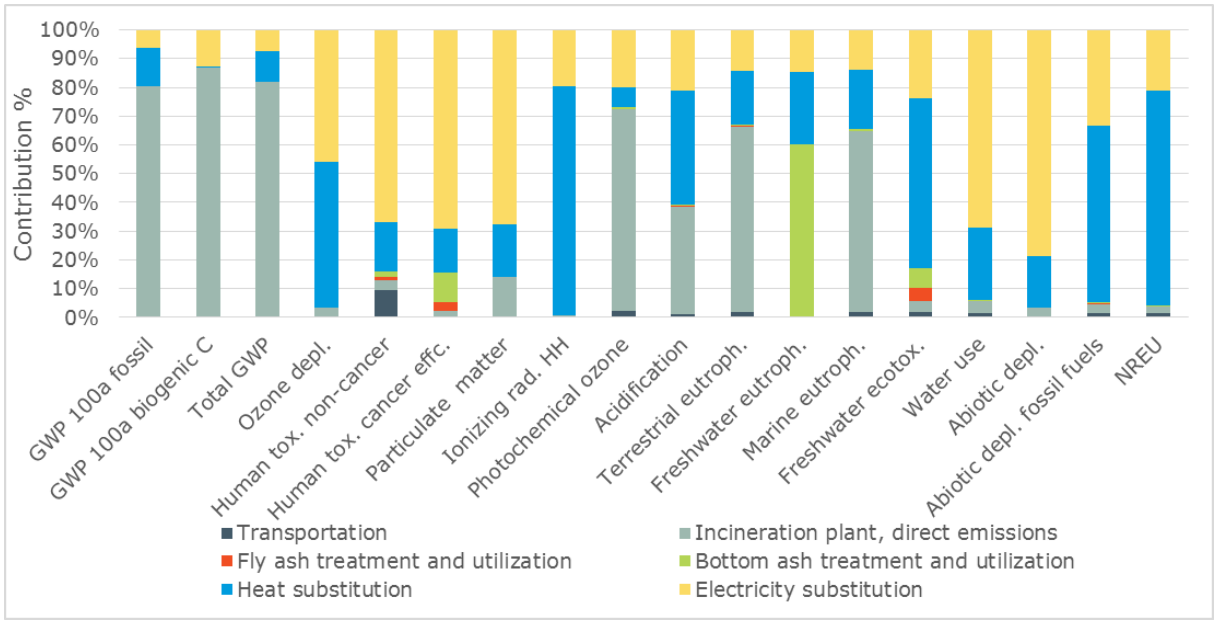

Figure 314. Contribution analysis for case 7, Incineration Starch based carrier bag (absolute percentages).

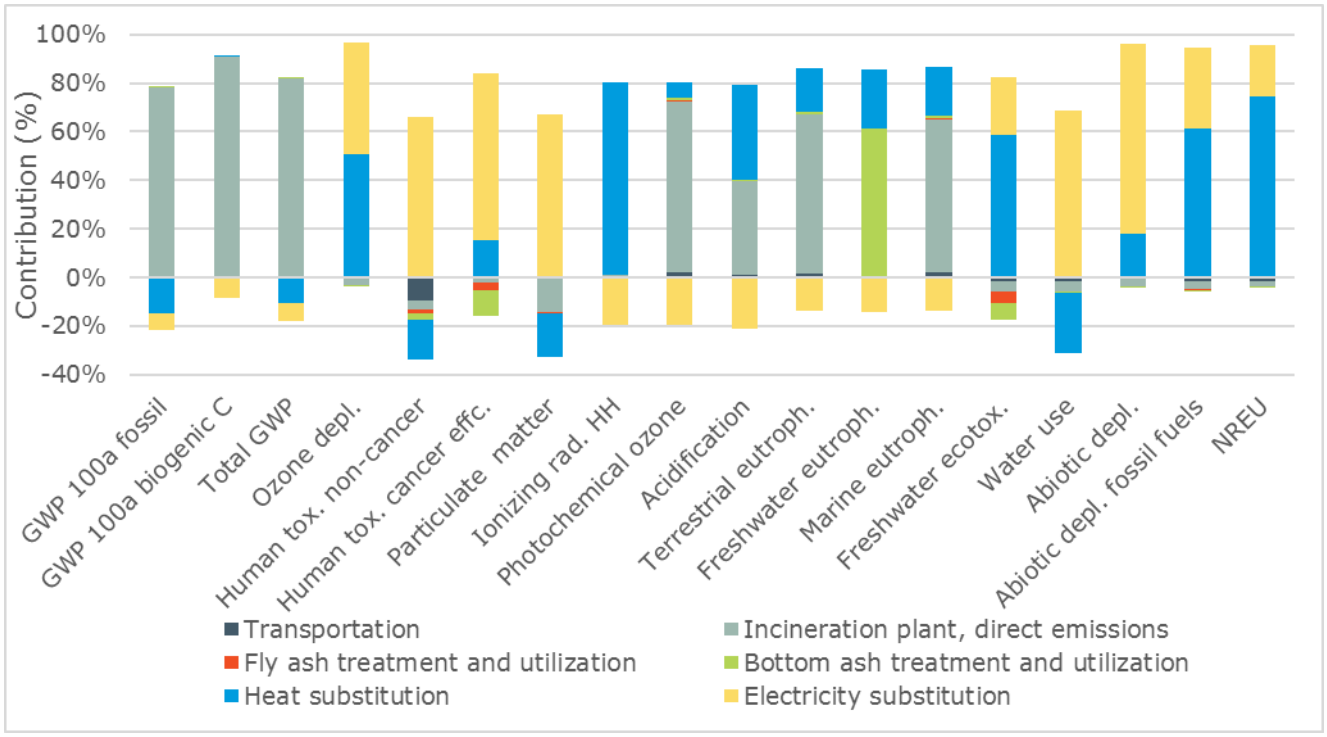

Figure 315. Contribution analysis for case 7, Incineration Starch based carrier bag.

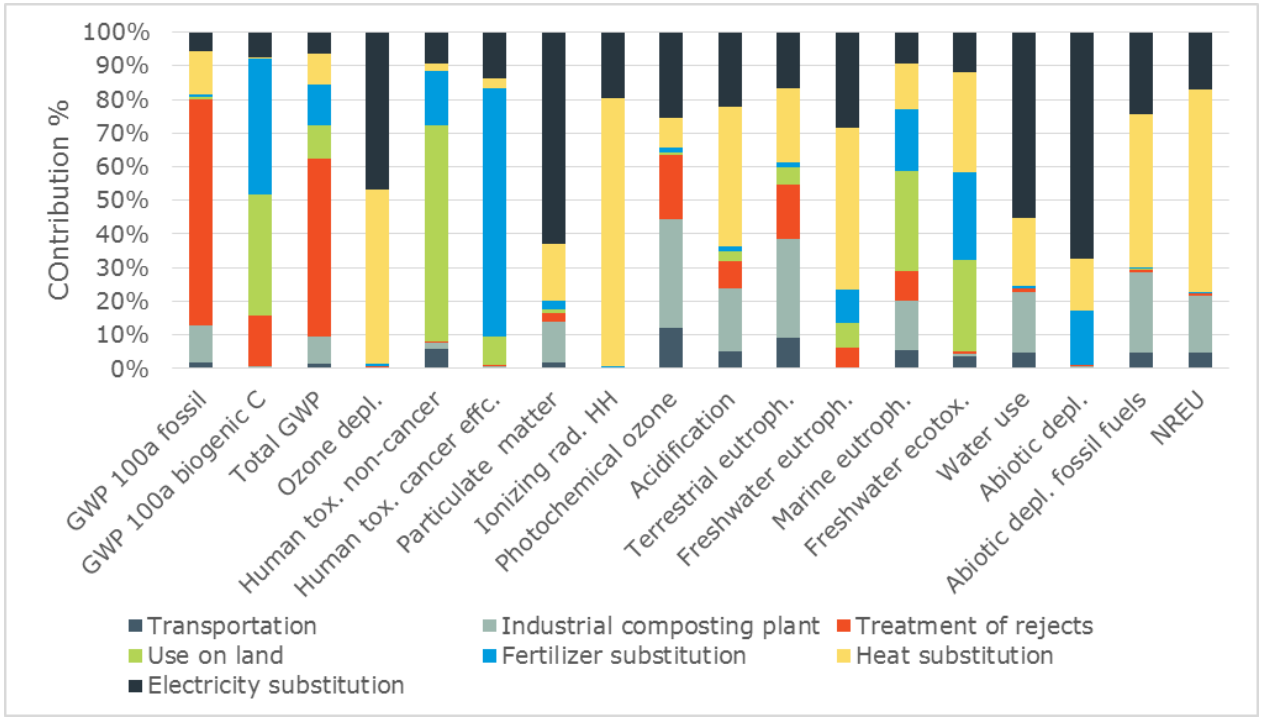

Figure 316. Contribution analysis for case 7, Industrial composting, starch (absolute percentages).

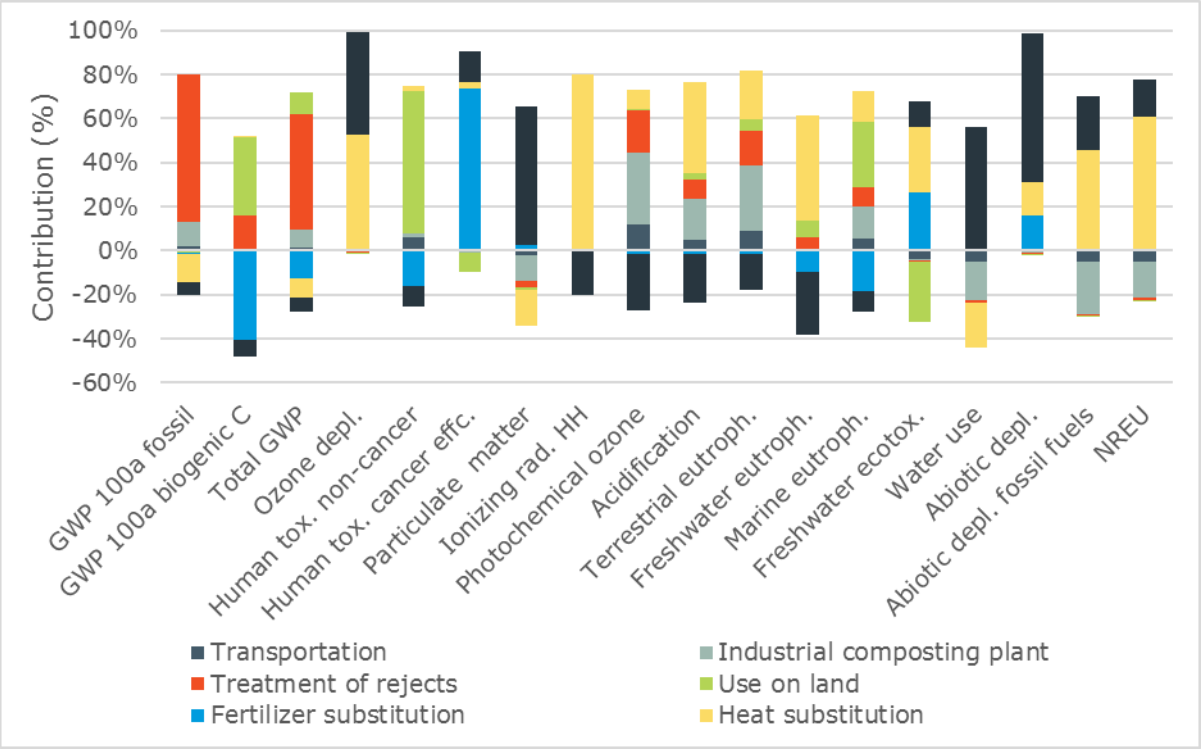

Figure 317. Contribution analysis for case 7, Industrial composting, starch.

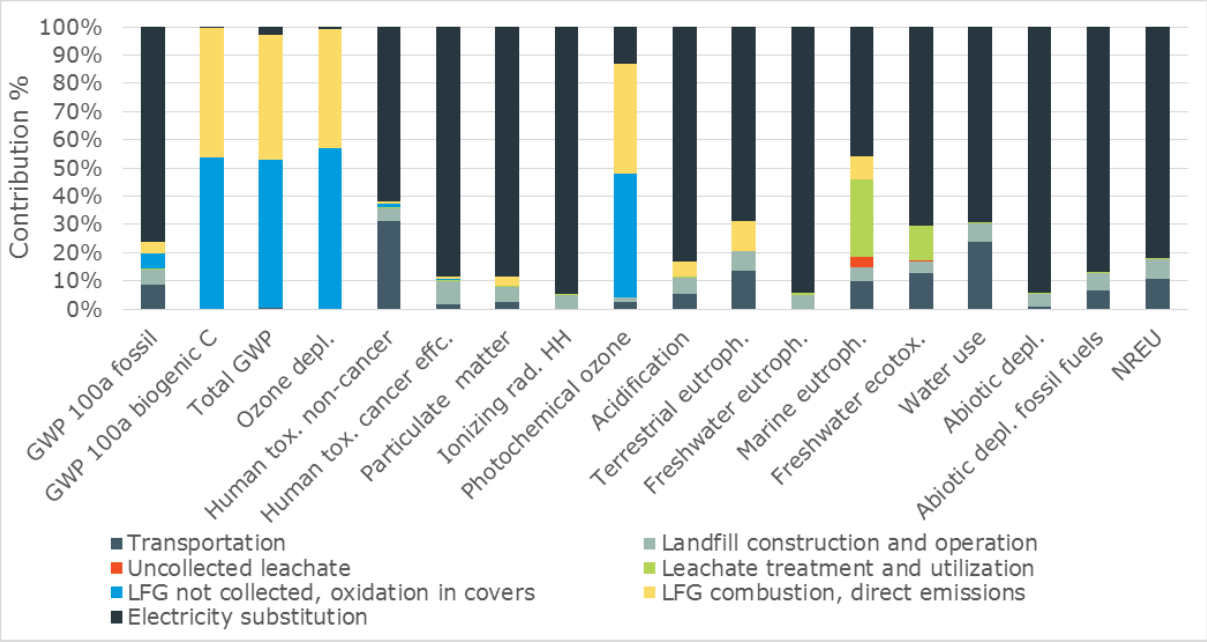

Figure 318. Contribution analysis for case 7, Landfilling, starch (absolute percentages).

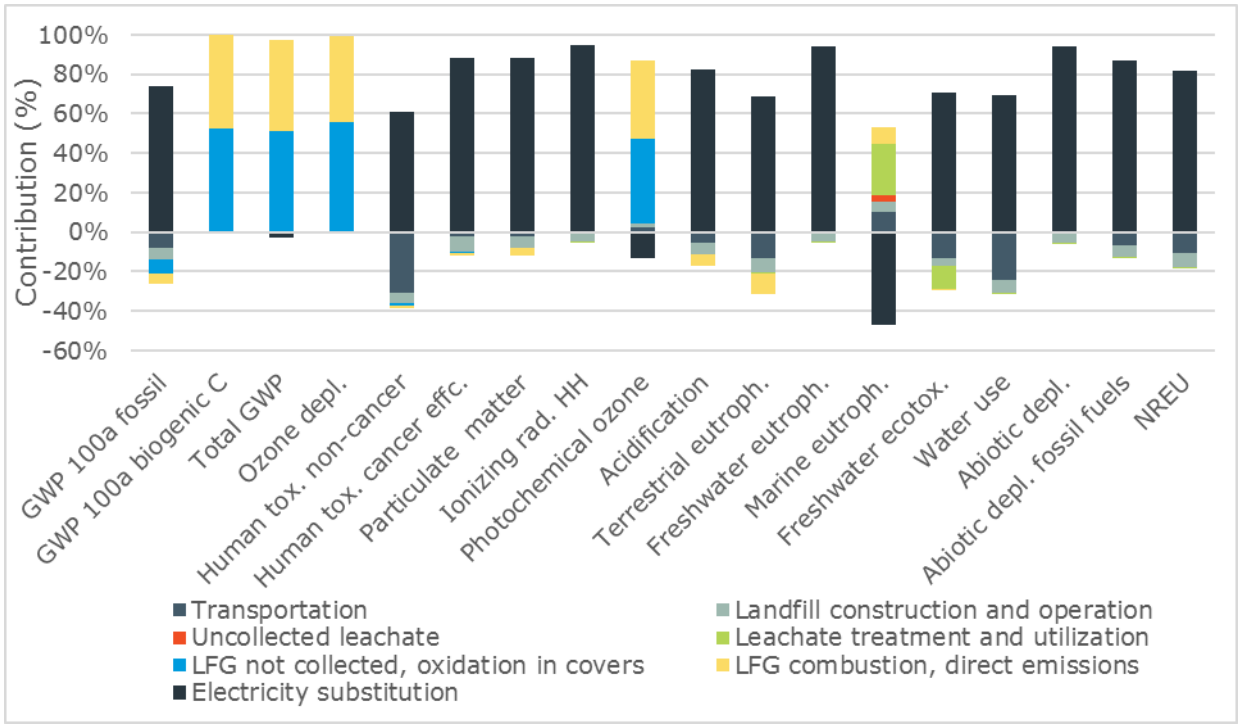

Figure 319. Contribution analysis for case 7, Landfilling, starch.

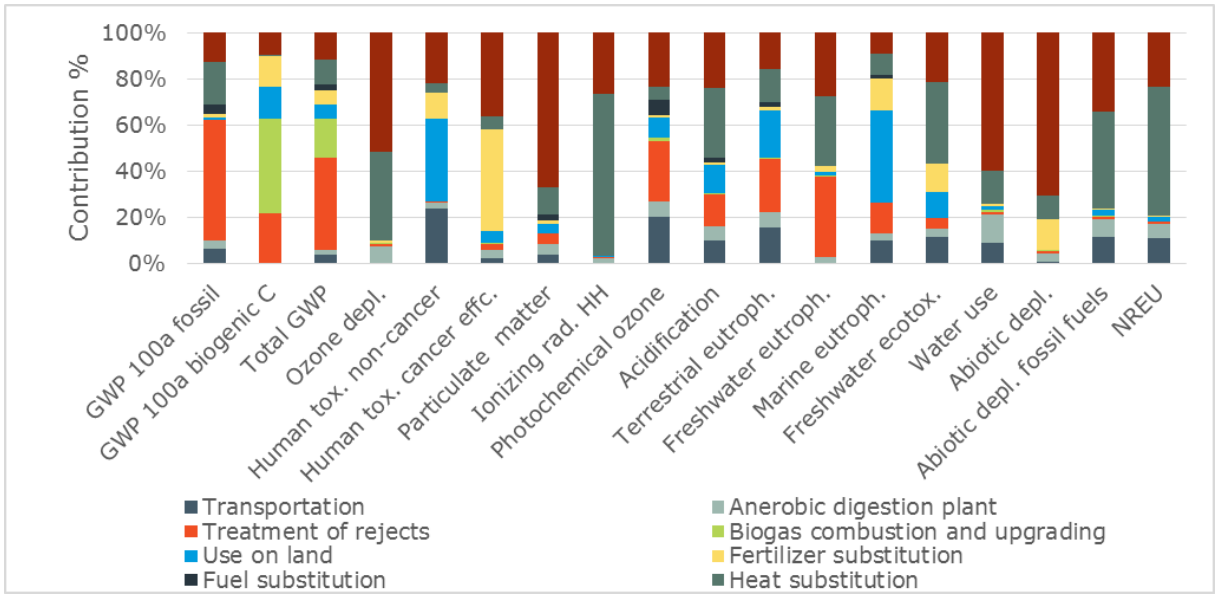

Figure 320. Contribution analysis for case 7, AD starch (absolute percentages).

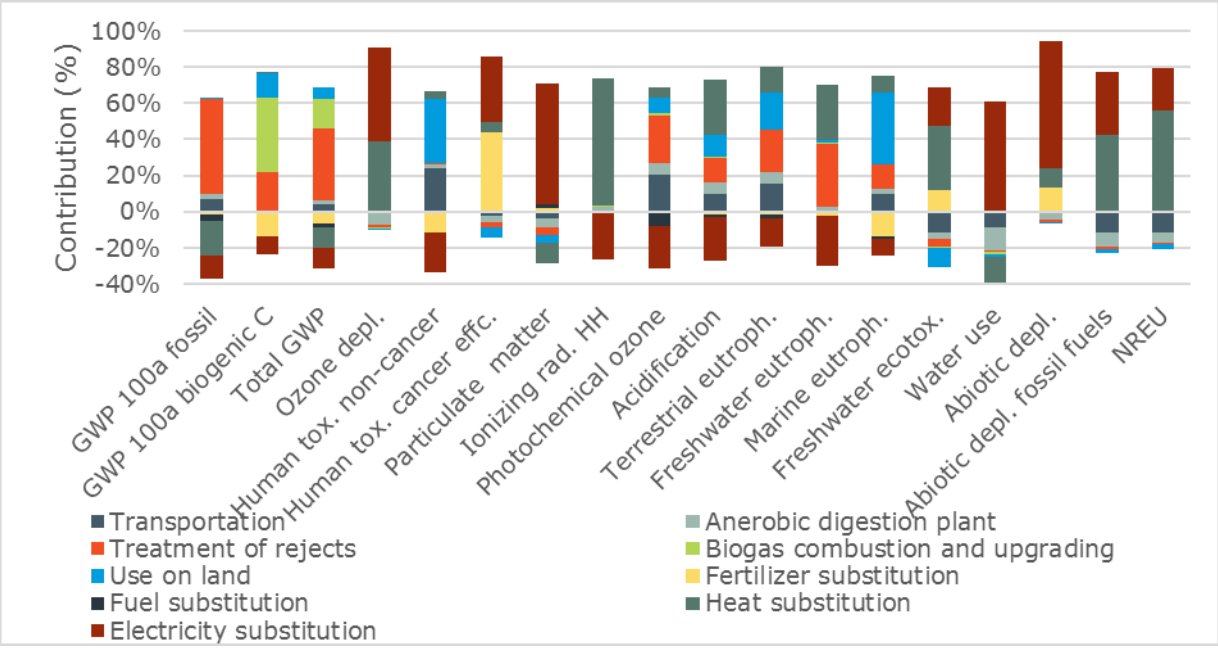

Figure 321. Contribution analysis for case 7, AD starch.

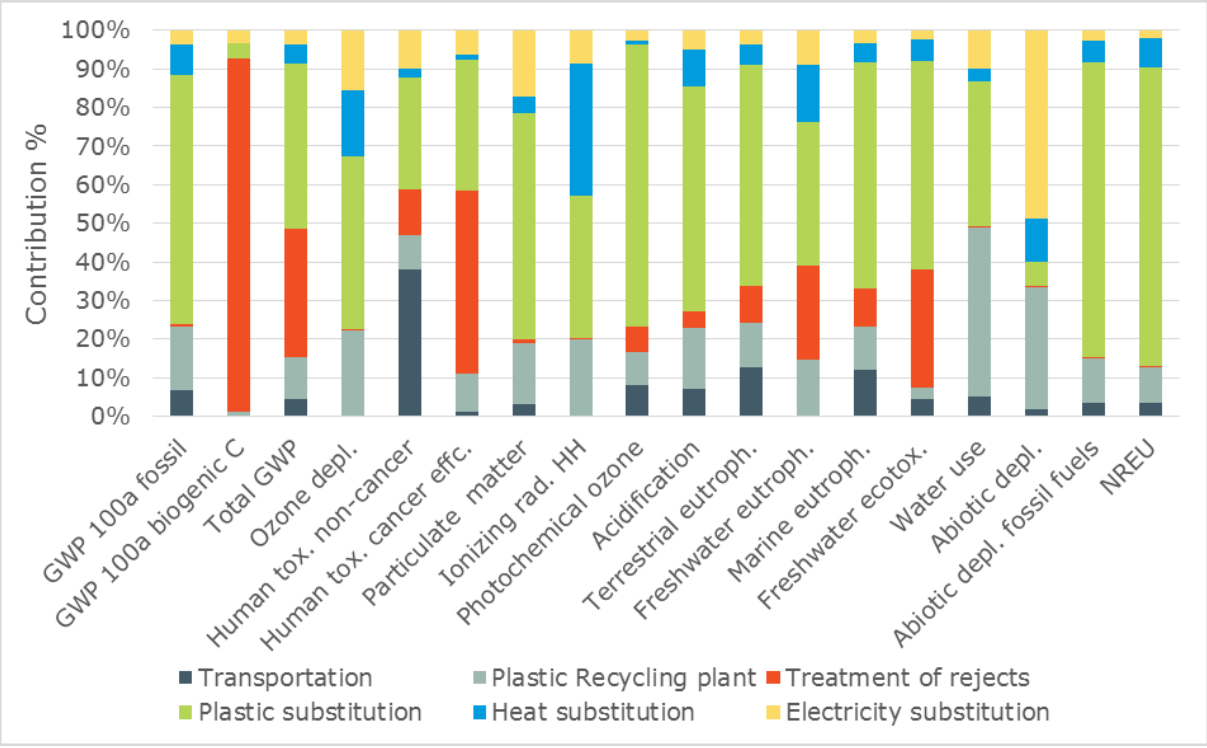

Figure 322. Contribution analysis for case 7, Plastics recycling BioLDPE (absolute percentages).

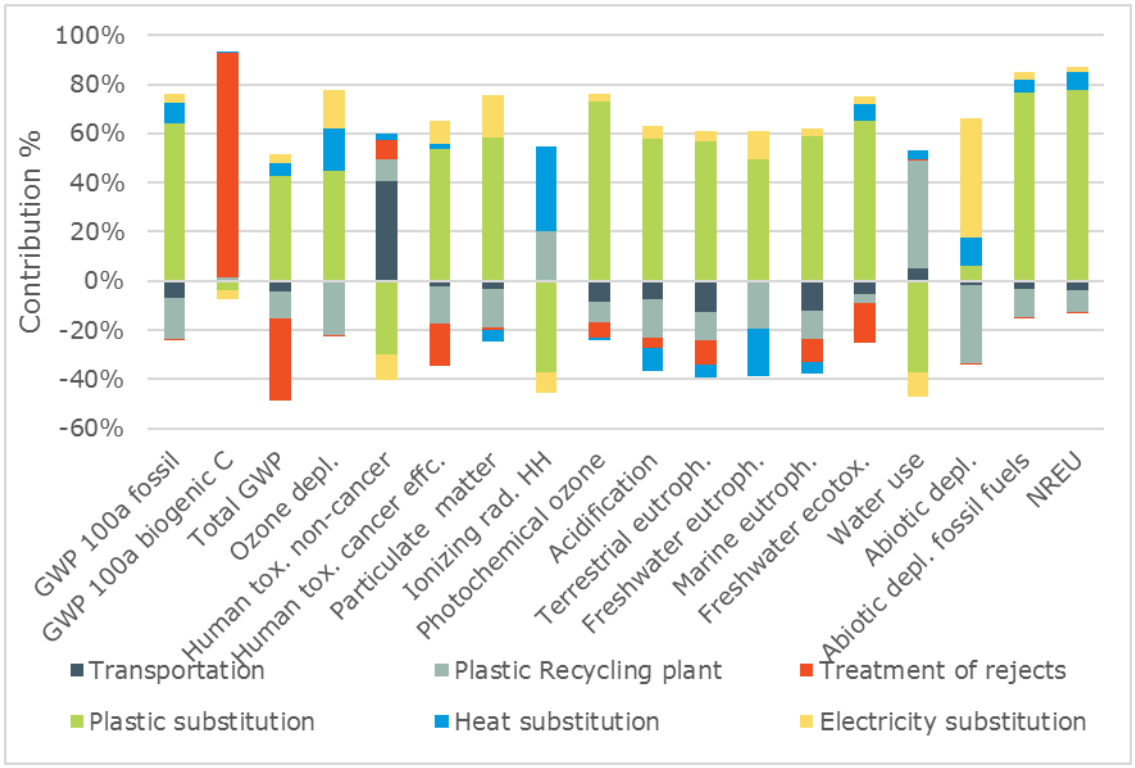

Figure 323. Contribution analysis for case 7, Plastics recycling BioLDPE.

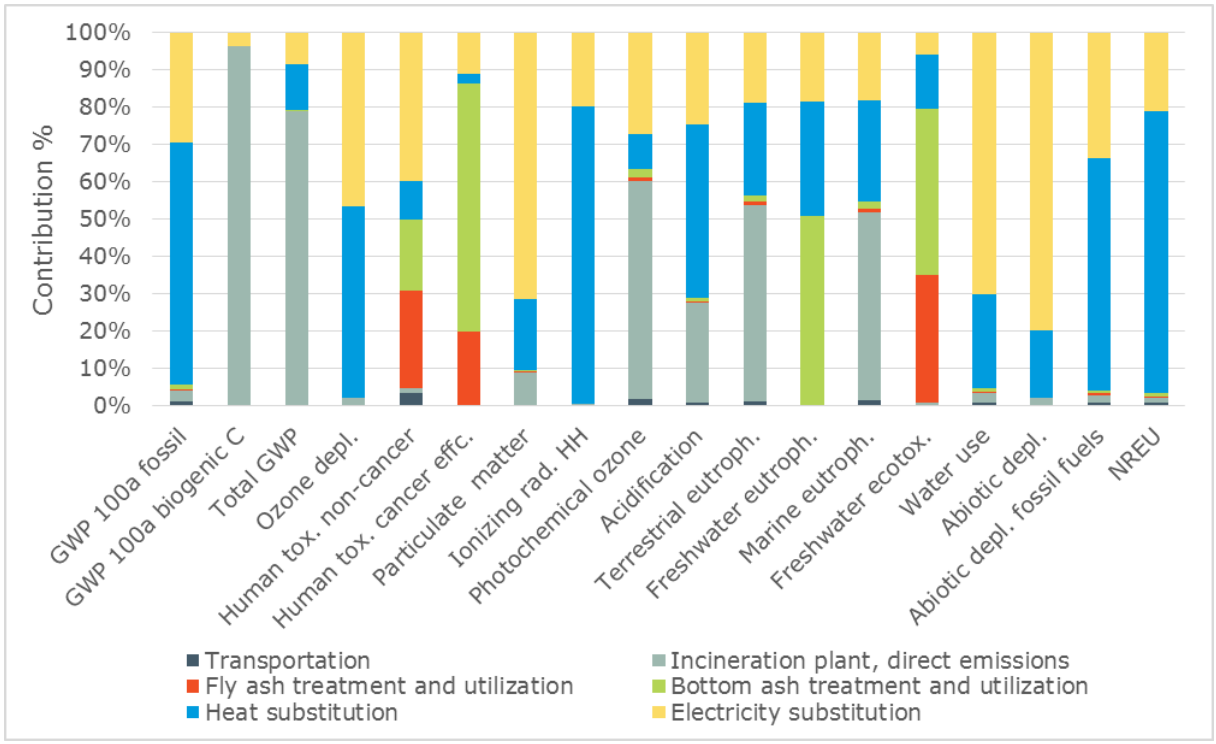

Figure 324. Contribution analysis for case 7, Incineration BioLDPE (absolute percentages).

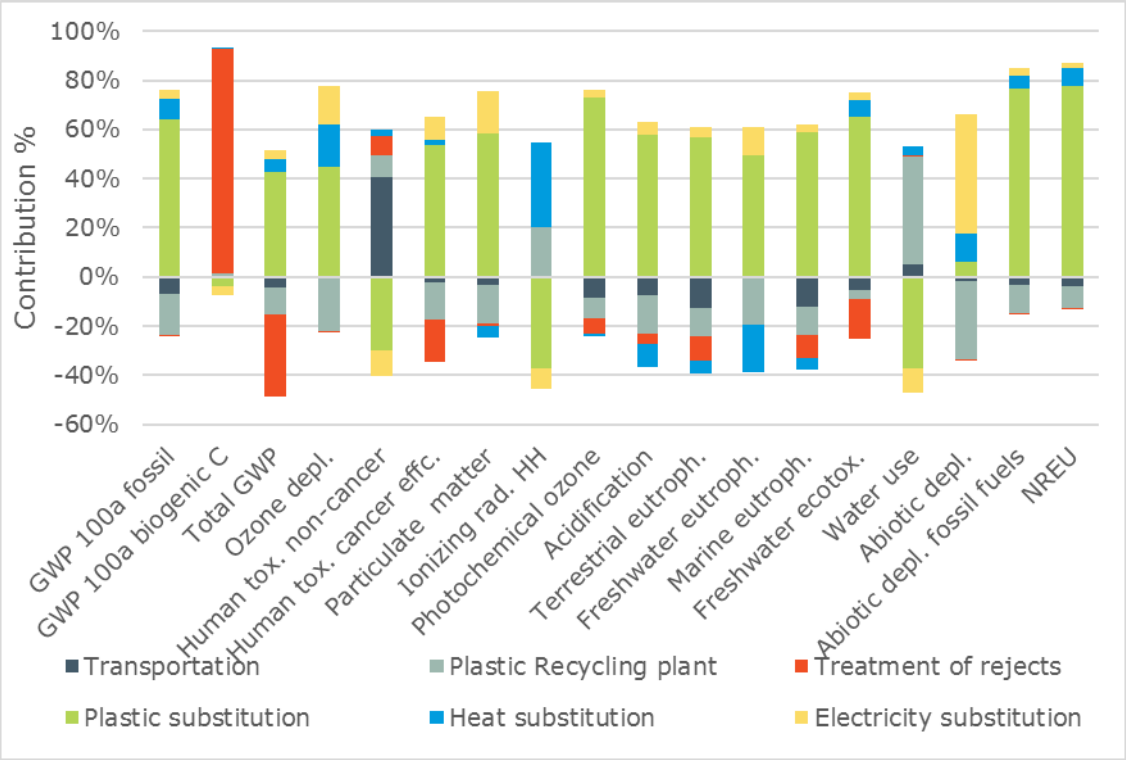

Figure 325. Contribution analysis for case 7, Incineration BioLDPE.

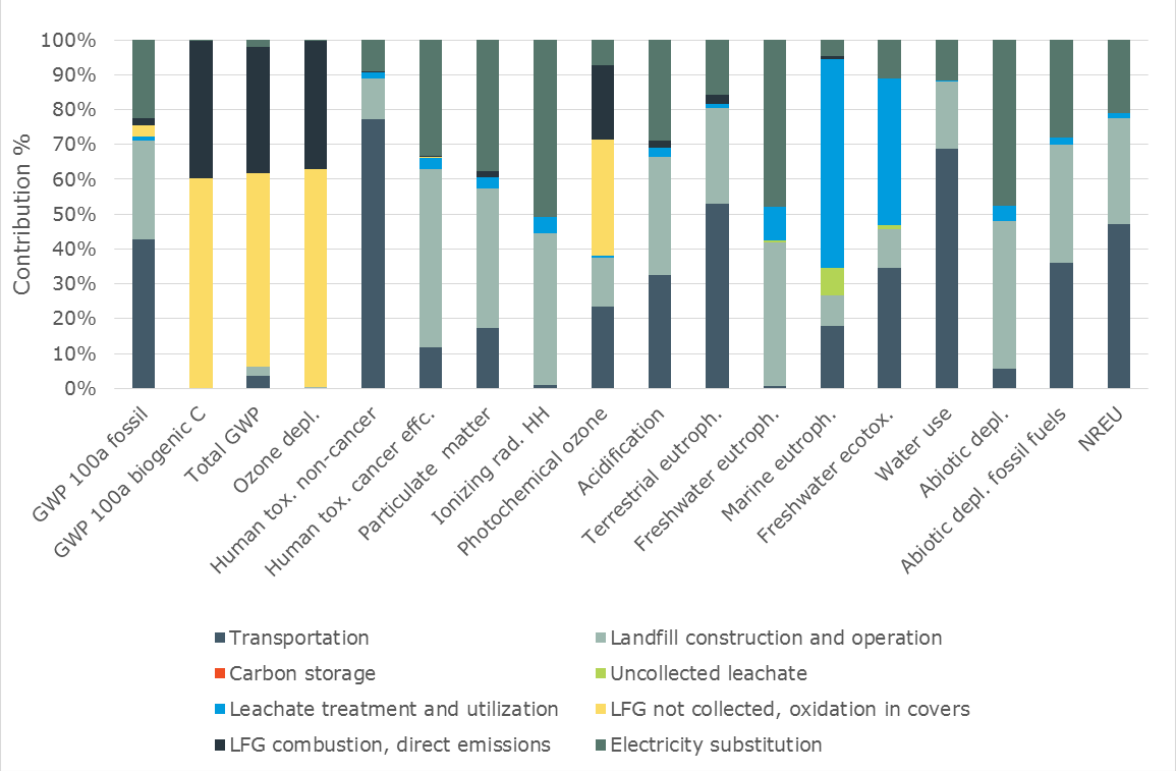

Figure 326. Contribution analysis for case 7, Landfilling, BioLDPE.

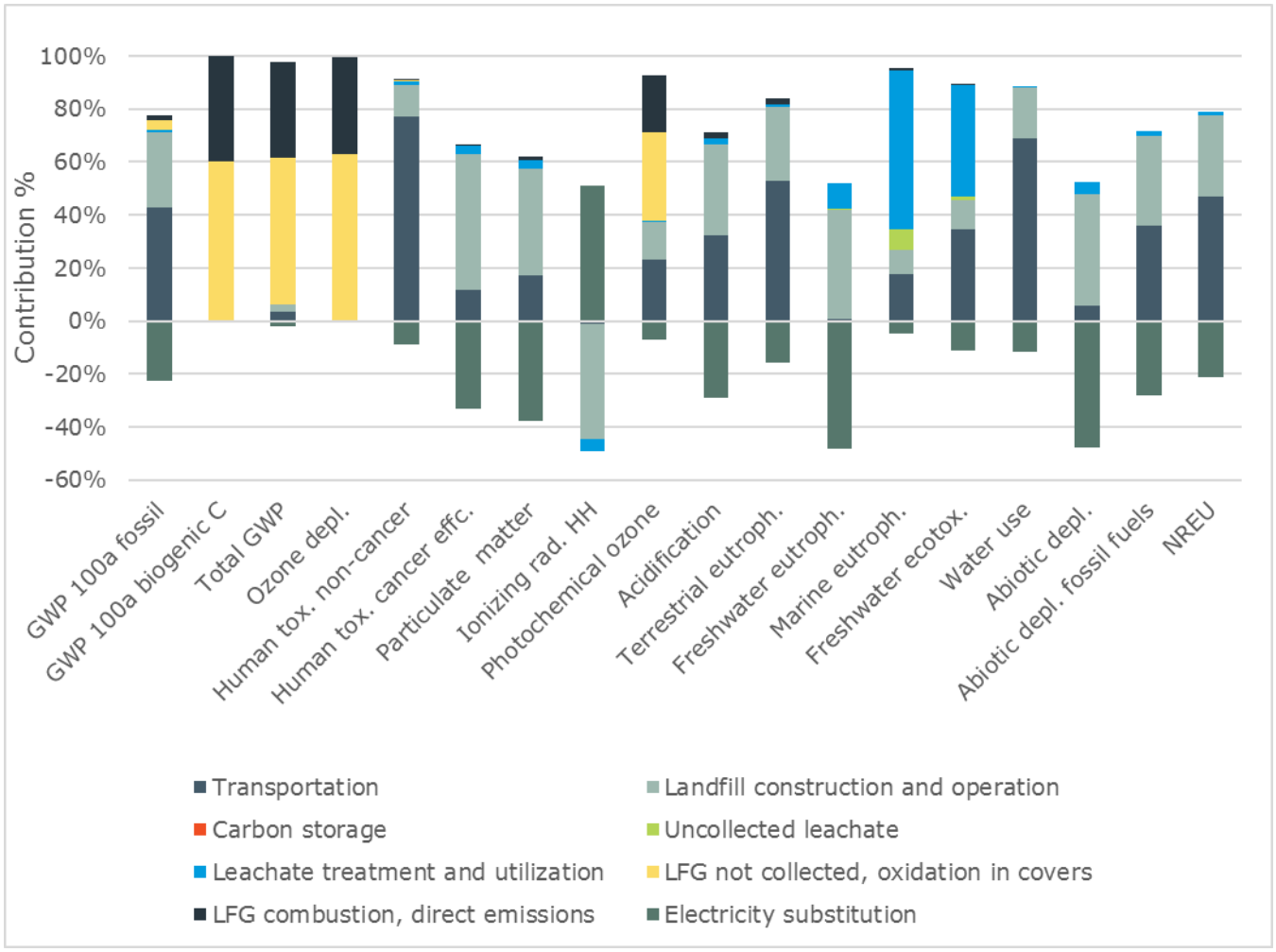

Figure 327. Contribution analysis for case 7, Landfilling, BioLDPE.

Weighted results for the product systems

The results have been normalised and weighted in order to reflect the importance of each impact category. For methodology and choice of normalisation and weighing factors, please see the Approach and methodology chapter, section 5.2.3 "Use of normalized and weighted results to determine the preference of EoL option" and the values in Table 30. Normalisation factors (NF) and weighting factors used by this study.

The weighted results have been calculated based on the ILCD 2015 normalisation factors and on EU27 2010 per capita in accordance to the CFs used in this study. The exception is abiotic depletions, where no NFs are available for both categories, in this case the global NFs per captia was used.

Case study 1: Beverage bottles – baseline

Table 225. Weighted results for Case study 1: Beverage bottles – baseline

Weighted results	with toxicity			without toxicity		
per 1 kg for each technology	Recycling	Incineration	Landfilling	Recycling	Incineration	Landfilling

Climate change	-1.39E-03	4.68E-03	3.33E-05	-1.47E-03	4.93E-03	3.51E-05
Ozone depletion	-2.00E-03	-1.09E-05	8.19E-08	-2.14E-03	-1.17E-05	8.76E-08
Human toxicity, non-cancer effects	-2.11E-04	-3.86E-05	1.34E-05			
Human toxicity, cancer effects	6.52E-05	-1.15E-04	3.37E-06			
Particulate matter	-5.75E-04	-1.48E-04	7.11E-06	-6.12E-04	-1.58E-04	7.57E-06
Ionizing radiation HH	-7.90E-05	2.29E-04	1.14E-06	-8.47E-05	2.46E-04	1.23E-06
Photochemical ozone formation	-4.96E-04	1.29E-04	6.69E-06	-5.30E-04	1.38E-04	7.14E-06
Acidification	-4.69E-04	2.05E-04	6.17E-06	-5.02E-04	2.20E-04	6.61E-06
Terrestrial eutrophication	-1.42E-04	1.16E-04	3.79E-06	-1.50E-04	1.22E-04	3.99E-06
Freshwater eutrophication	-1.35E-04	2.03E-05	5.55E-07	-1.42E-04	2.14E-05	5.84E-07
Marine eutrophication	-1.14E-04	7.76E-05	1.04E-05	-1.21E-04	8.18E-05	1.09E-05
Freshwater ecotoxicity	1.66E-04	-2.99E-05	1.96E-06			
Land use	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	-2.14E-04	-1.38E-04	1.01E-05	-2.27E-04	-1.46E-04	1.07E-05
Abiotic depletion	-7.51E-06	-3.32E-05	4.17E-07	-8.04E-06	-3.55E-05	4.46E-07
Abiotic depletion (fossil fuels)	-4.77E-03	-9.00E-04	2.58E-05	-5.11E-03	-9.65E-04	2.77E-05
Sum	**-1.04E-02**	**4.04E-03**	**1.24E-04**	**-1.11E-02**	**4.44E-03**	**1.12E-04**

Case study 1: beverage bottles – reference

Table 226 Weighted results for Case study 1: beverage bottles – reference

Weighted results	with toxicity			without toxicity		
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**	**Recycling**	**Incineration**	**Landfilling**
Climate change	-1.39E-03	4.68E-03	3.33E-05	-1.47E-03	4.93E-03	3.51E-05
Ozone depletion	-2.00E-03	-1.09E-05	8.19E-08	-2.14E-03	-1.17E-05	8.76E-08
Human toxicity, non-cancer effects	-2.11E-04	-3.86E-05	1.34E-05			
Human toxicity, cancer effects	6.52E-05	-1.15E-04	3.37E-06			
Particulate matter	-5.75E-04	-1.48E-04	7.11E-06	-6.12E-04	-1.58E-04	7.57E-06
Ionizing radiation HH	-7.90E-05	2.29E-04	1.14E-06	-8.47E-05	2.46E-04	1.23E-06
Photochemical ozone formation	-4.96E-04	1.29E-04	6.69E-06	-5.30E-04	1.38E-04	7.14E-06
Acidification	-4.69E-04	2.05E-04	6.17E-06	-5.02E-04	2.20E-04	6.61E-06
Terrestrial eutrophication	-1.42E-04	1.16E-04	3.79E-06	-1.50E-04	1.22E-04	3.99E-06
Freshwater eutrophication	-1.35E-04	2.03E-05	5.55E-07	-1.42E-04	2.14E-05	5.84E-07
Marine eutrophication	-1.14E-04	7.76E-05	1.04E-05	-1.21E-04	8.18E-05	1.09E-05
Freshwater ecotoxicity	1.66E-04	-2.99E-05	1.96E-06			
Land use	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	-2.14E-04	-1.38E-04	1.01E-05	-2.27E-04	-1.46E-04	1.07E-05
Abiotic depletion	-7.51E-06	-3.32E-05	4.17E-07	-8.04E-06	-3.55E-05	4.46E-07

Abiotic depletion (fossil fuels)	-4.77E-03	-9.00E-04	2.58E-05	-5.11E-03	-9.65E-04	2.77E-05
Sum	**-1.04E-02**	**4.04E-03**	**1.24E-04**	**-1.11E-02**	**4.44E-03**	**1.12E-04**

Case study 2: single-use clips – baseline

Table 227 Weighted results for Case study 2: single-use clips – baseline

Weighted results	**with toxicity**			**without toxicity**		
per 1 kg for each technology	**Incineration**	**Landfilling**	**In-situ**	**Incineration**	**Landfilling**	**In-situ**
Climate change	2.26E-02	2.36E-02	2.18E-03	2.39E-02	2.49E-02	2.30E-03
Ozone depletion	-8.45E-06	5.52E-04	-4.20E-10	-9.04E-06	5.91E-04	-4.50E-10
Human toxicity, non-cancer effects	1.29E-03	2.36E-05	1.07E-04			
Human toxicity, cancer effects	1.17E-02	-1.72E-05	3.98E-04			
Particulate matter	3.34E-04	-6.98E-05	3.24E-06	3.56E-04	-7.43E-05	3.45E-06
Ionizing radiation HH	2.20E-04	-1.50E-05	-2.52E-08	2.35E-04	-1.61E-05	-2.70E-08
Photochemical ozone formation	1.56E-03	1.87E-04	3.17E-05	1.66E-03	1.99E-04	3.38E-05
Acidification	1.38E-03	9.89E-06	1.99E-05	1.48E-03	1.06E-05	2.13E-05
Terrestrial eutrophication	1.02E-03	1.04E-05	1.89E-05	1.07E-03	1.10E-05	1.99E-05
Freshwater eutrophication	1.30E-03	-6.67E-06	2.62E-07	1.37E-03	-7.03E-06	2.76E-07
Marine eutrophication	7.13E-04	3.35E-05	1.44E-05	7.52E-04	3.53E-05	1.52E-05
Freshwater ecotoxicity	7.73E-04	3.46E-06	2.07E-05			
Land use	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	7.88E-04	-2.69E-05	-2.28E-06	8.36E-04	-2.85E-05	-2.42E-06
Abiotic depletion	-1.97E-05	-8.54E-06	-1.63E-08	-2.10E-05	-9.13E-06	-1.75E-08
Abiotic depletion (fossil fuels)	1.88E-03	-5.54E-06	-6.84E-06	2.02E-03	-5.94E-06	-7.34E-06
Sum	**4.56E-02**	**2.43E-02**	**2.78E-03**	**3.36E-02**	**2.56E-02**	**2.38E-03**

Case study 2: single-use clips – reference

Table 228 Weighted results for Case study 2: single-use clips – reference

Weighted results	**with toxicity**		**without toxicity**	
per 1 kg for each technology	**Incineration**	**Landfilling**	**Incineration**	**Landfilling**
Climate change	3.08E-02	2.21E-02	3.24E-02	2.33E-02

Ozone depletion	-1.41E-05	3.47E-06	-1.50E-05	3.71E-06
Human toxicity, non-cancer effects	7.38E-03	6.21E-03		
Human toxicity, cancer effects	1.32E-02	7.21E-04		
Particulate matter	2.30E-03	2.20E-03	2.45E-03	2.34E-03
Ionizing radiation HH	4.93E-04	1.14E-04	5.28E-04	1.23E-04
Photochemical ozone formation	7.26E-03	5.89E-03	7.75E-03	6.29E-03
Acidification	6.04E-03	4.75E-03	6.47E-03	5.09E-03
Terrestrial eutrophication	4.41E-03	3.49E-03	4.65E-03	3.68E-03
Freshwater eutrophication	1.36E-03	3.22E-05	1.43E-03	3.40E-05
Marine eutrophication	3.21E-03	2.59E-03	3.38E-03	2.73E-03
Freshwater ecotoxicity	1.15E-03	2.23E-04		
Land use	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	5.81E-03	5.33E-03	6.16E-03	5.65E-03
Abiotic depletion	-4.49E-06	4.34E-05	-4.81E-06	4.64E-05
Abiotic depletion (fossil fuels)	1.68E-02	1.61E-02	1.80E-02	1.72E-02
Sum	**1.00E-01**	**6.98E-02**	**8.32E-02**	**6.65E-02**

Case study 3: single-use cups – baseline

Table 229. Weighted results for Case study 3: single-use cups – baseline

Weighted results	**with toxicity**				
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**	**Industrial composting**	**Anaerobic digestion**
Climate change	-1.28E-03	3.48E-03	7.07E-03	2.58E-04	1.05E-03
Ozone depletion	-1.85E-03	-8.43E-06	5.31E-04	-1.80E-06	-3.20E-06
Human toxicity, non-cancer effects	-2.26E-04	-2.75E-05	-3.68E-06	1.31E-04	1.60E-04
Human toxicity, cancer effects	-3.56E-04	-7.12E-05	-1.74E-05	-1.38E-04	-1.23E-04
Particulate matter	-5.19E-04	-1.03E-04	-5.61E-05	-2.44E-06	-2.71E-05
Ionizing radiation HH	-7.05E-05	1.80E-04	-1.42E-05	6.91E-05	7.25E-05
Photochemical ozone formation	-4.45E-04	1.31E-04	1.39E-04	3.82E-05	9.36E-05
Acidification	-4.12E-04	1.88E-04	-5.30E-06	9.12E-05	1.38E-04
Terrestrial eutrophication	-1.18E-04	1.12E-04	7.70E-07	4.78E-05	8.81E-05
Freshwater eutrophication	-1.01E-04	1.38E-04	-5.88E-06	3.64E-05	4.27E-05
Marine eutrophication	-9.90E-05	7.42E-05	5.78E-06	8.33E-05	8.05E-05
Freshwater ecotoxicity	-5.04E-04	-2.21E-05	-8.28E-07	-5.19E-06	-3.26E-06
Land use	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	-1.89E-04	-1.05E-04	-4.24E-05	3.85E-05	2.97E-05
Abiotic depletion	-6.78E-06	-2.56E-05	-6.11E-06	-7.07E-06	-1.59E-05

Abiotic depletion (fossil fuels)	-4.40E-03	-6.99E-04	-4.76E-05	-1.27E-04	-1.08E-04
Sum	**-1.06E-02**	**3.24E-03**	**7.55E-03**	**5.13E-04**	**1.47E-03**

Table 230. Weighted results for Case study 3: single-use cups – baseline

Weighted results	**without toxicity**				
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**	**Industrial composting**	**Anaerobic digestion**
Climate change	-1.35E-03	3.67E-03	7.45E-03	2.72E-04	1.10E-03
Ozone depletion	-1.98E-03	-9.01E-06	5.68E-04	-1.92E-06	-3.42E-06
Human toxicity, non-cancer effects					
Human toxicity, cancer effects					
Particulate matter	-5.52E-04	-1.10E-04	-5.97E-05	-2.60E-06	-2.88E-05
Ionizing radiation HH	-7.55E-05	1.92E-04	-1.53E-05	7.40E-05	7.77E-05
Photochemical ozone formation	-4.75E-04	1.40E-04	1.48E-04	4.08E-05	9.98E-05
Acidification	-4.41E-04	2.01E-04	-5.68E-06	9.77E-05	1.48E-04
Terrestrial eutrophication	-1.24E-04	1.18E-04	8.12E-07	5.04E-05	9.28E-05
Freshwater eutrophication	-1.07E-04	1.45E-04	-6.19E-06	3.84E-05	4.50E-05
Marine eutrophication	-1.04E-04	7.82E-05	6.10E-06	8.78E-05	8.49E-05
Freshwater ecotoxicity					
Land use	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	-2.01E-04	-1.12E-04	-4.50E-05	4.09E-05	3.15E-05
Abiotic depletion	-7.26E-06	-2.74E-05	-6.54E-06	-7.56E-06	-1.71E-05
Abiotic depletion (fossil fuels)	-4.72E-03	-7.49E-04	-5.10E-05	-1.36E-04	-1.16E-04
Sum	**-1.01E-02**	**3.53E-03**	**7.99E-03**	**5.54E-04**	**1.52E-03**

Case study 3: single-use cups – alternative 2

Table 231. Weighted results for Case study 3: single-use cups – alternative 2

Weighted results	**with toxicity**		
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**
Climate change	1.06E-04	5.25E-03	3.55E-05
Ozone depletion	-8.12E-06	-1.83E-05	1.34E-07
Human toxicity, non-cancer effects	1.21E-04	1.73E-04	1.34E-05
Human toxicity, cancer effects	2.38E-03	8.41E-03	4.30E-06
Particulate matter	-4.14E-04	-2.79E-04	7.02E-06
Ionizing radiation HH	4.65E-05	3.77E-04	2.99E-06
Photochemical ozone formation	-3.54E-04	1.35E-04	7.28E-06
Acidification	-1.74E-04	2.67E-04	8.19E-06
Terrestrial eutrophication	-5.24E-05	1.35E-04	4.17E-06
Freshwater eutrophication	9.82E-05	4.49E-04	1.01E-06
Marine eutrophication	-4.69E-05	9.28E-05	1.06E-05

Freshwater ecotoxicity	1.21E-04	7.15E-04	2.06E-06
Land use	0.00E+00	0.00E+00	0.00E+00
Water use	2.41E-04	-2.25E-04	1.69E-05
Abiotic depletion	-1.54E-05	-5.56E-05	3.37E-07
Abiotic depletion (fossil fuels)	-4.14E-03	-1.47E-03	3.00E-05
Sum	**-2.10E-03**	**1.40E-02**	**1.44E-04**

Table 232. Weighted results for Case study 3: single-use cups – alternative 2

Weighted results	without toxicity		
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**
Climate change	1.11E-04	5.53E-03	3.74E-05
Ozone depletion	-8.69E-06	-1.96E-05	1.43E-07
Human toxicity, non-cancer effects			
Human toxicity, cancer effects			
Particulate matter	-4.41E-04	-2.97E-04	7.48E-06
Ionizing radiation HH	4.98E-05	4.04E-04	3.20E-06
Photochemical ozone formation	-3.78E-04	1.45E-04	7.77E-06
Acidification	-1.86E-04	2.86E-04	8.77E-06
Terrestrial eutrophication	-5.53E-05	1.42E-04	4.40E-06
Freshwater eutrophication	1.03E-04	4.73E-04	1.06E-06
Marine eutrophication	-4.94E-05	9.78E-05	1.12E-05
Freshwater ecotoxicity			
Land use	0.00E+00	0.00E+00	0.00E+00
Water use	2.56E-04	-2.39E-04	1.79E-05
Abiotic depletion	-1.65E-05	-5.95E-05	3.60E-07
Abiotic depletion (fossil fuels)	-4.44E-03	-1.58E-03	3.21E-05
Sum	**-5.06E-03**	**4.89E-03**	**1.32E-04**

Case study 3: single-use cups – reference 1

Table 233. Weighted results for Case study 3: single-use cups – reference 1

Weighted results	with toxicity		
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**
Climate change	1.06E-04	5.25E-03	3.55E-05
Ozone depletion	-8.12E-06	-1.83E-05	1.34E-07
Human toxicity, non-cancer effects	1.21E-04	1.73E-04	1.34E-05
Human toxicity, cancer effects	2.38E-03	8.41E-03	4.30E-06

Particulate matter	-4.14E-04	-2.79E-04	7.02E-06
Ionizing radiation HH	4.65E-05	3.77E-04	2.99E-06
Photochemical ozone formation	-3.54E-04	1.35E-04	7.28E-06
Acidification	-1.74E-04	2.67E-04	8.19E-06
Terrestrial eutrophication	-5.24E-05	1.35E-04	4.17E-06
Freshwater eutrophication	9.82E-05	4.49E-04	1.01E-06
Marine eutrophication	-4.69E-05	9.28E-05	1.06E-05
Freshwater ecotoxicity	1.21E-04	7.15E-04	2.06E-06
Land use	0.00E+00	0.00E+00	0.00E+00
Water use	2.41E-04	-2.25E-04	1.69E-05
Abiotic depletion	-1.54E-05	-5.56E-05	3.37E-07
Abiotic depletion (fossil fuels)	-4.14E-03	-1.47E-03	3.00E-05
Sum	**-2.10E-03**	**1.40E-02**	**1.44E-04**

Table 234. Weighted results for Case study 3: single-use cups – reference 1

Weighted results	**without toxicity**		
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**
Climate change	1.11E-04	5.53E-03	3.74E-05
Ozone depletion	-8.69E-06	-1.96E-05	1.43E-07
Human toxicity, non-cancer effects			
Human toxicity, cancer effects			
Particulate matter	-4.41E-04	-2.97E-04	7.48E-06
Ionizing radiation HH	4.98E-05	4.04E-04	3.20E-06
Photochemical ozone formation	-3.78E-04	1.45E-04	7.77E-06
Acidification	-1.86E-04	2.86E-04	8.77E-06
Terrestrial eutrophication	-5.53E-05	1.42E-04	4.40E-06
Freshwater eutrophication	1.03E-04	4.73E-04	1.06E-06
Marine eutrophication	-4.94E-05	9.78E-05	1.12E-05
Freshwater ecotoxicity			
Land use	0.00E+00	0.00E+00	0.00E+00
Water use	2.56E-04	-2.39E-04	1.79E-05
Abiotic depletion	-1.65E-05	-5.95E-05	3.60E-07
Abiotic depletion (fossil fuels)	-4.44E-03	-1.58E-03	3.21E-05
Sum	**-5.06E-03**	**4.89E-03**	**1.32E-04**

Case study 3: single-use cups – reference 2

Table 235. Weighted results for Case study 3: single-use cups – reference 2

Weighted results	with toxicity		
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**
Climate change	-9.24E-04	4.68E-03	3.59E-05
Ozone depletion	-1.85E-03	-1.09E-05	1.34E-07
Human toxicity, non-cancer effects	-1.73E-04	1.48E-04	1.34E-05
Human toxicity, cancer effects	1.89E-04	1.75E-03	4.29E-06
Particulate matter	-5.32E-04	-1.48E-04	7.04E-06
Ionizing radiation HH	-5.56E-05	2.29E-04	2.99E-06
Photochemical ozone formation	-4.45E-04	1.29E-04	7.31E-06
Acidification	-4.07E-04	2.05E-04	8.20E-06
Terrestrial eutrophication	-1.17E-04	1.16E-04	4.19E-06
Freshwater eutrophication	-1.19E-04	8.07E-05	8.96E-07
Marine eutrophication	-9.79E-05	7.76E-05	3.00E-06
Freshwater ecotoxicity	3.65E-04	2.88E-03	1.87E-06
Land use	0.00E+00	0.00E+00	0.00E+00
Water use	-1.99E-04	-1.38E-04	1.69E-05
Abiotic depletion	-9.05E-06	-3.32E-05	3.37E-07
Abiotic depletion (fossil fuels)	-4.47E-03	-9.00E-04	3.02E-05
Sum	**-8.84E-03**	**9.06E-03**	**1.37E-04**

Table 236. Weighted results for Case study 3: single-use cups – reference 2

Weighted results	without toxicity		
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**
Climate change	-9.74E-04	4.93E-03	3.78E-05
Ozone depletion	-1.98E-03	-1.17E-05	1.43E-07
Human toxicity, non-cancer effects			
Human toxicity, cancer effects			
Particulate matter	-5.67E-04	-1.58E-04	7.50E-06
Ionizing radiation HH	-5.96E-05	2.46E-04	3.20E-06
Photochemical ozone formation	-4.75E-04	1.38E-04	7.80E-06
Acidification	-4.35E-04	2.20E-04	8.78E-06
Terrestrial eutrophication	-1.23E-04	1.22E-04	4.41E-06
Freshwater eutrophication	-1.25E-04	8.50E-05	9.44E-07
Marine eutrophication	-1.03E-04	8.18E-05	3.17E-06
Freshwater ecotoxicity			

Land use	0.00E+00	0.00E+00	0.00E+00
Water use	-2.11E-04	-1.46E-04	1.79E-05
Abiotic depletion	-9.68E-06	-3.55E-05	3.60E-07
Abiotic depletion (fossil fuels)	-4.79E-03	-9.65E-04	3.24E-05
Sum	**-9.85E-03**	**4.51E-03**	**1.24E-04**

Case study 4: single-use cutlery – baseline

Table 237. Weighted results for Case study 4: single-use cutlery – baseline

Weighted results	**with toxicity**				**without toxicity**			
per 1 kg for each technology	**Incineration**	**Landfilling**	**Industrial composting**	**Anaerobic digestion**	**Incineration**	**Landfilling**	**Industrial composting**	**Anaerobic digestion**
Climate change	8.87E-03	3.06E-02	4.83E-04	2.82E-03	9.35E-03	3.22E-02	5.09E-04	2.97E-03
Ozone depletion	-1.47E-05	1.95E-03	1.14E-06	-9.17E-06	-1.57E-05	2.08E-03	1.22E-06	-9.81E-06
Human toxicity, non-cancer effects	2.40E-04	2.17E-06	5.89E-03	4.06E-03				
Human toxicity, cancer effects	7.67E-04	-6.98E-05	-6.41E-03	-4.67E-03				
Particulate matter	7.14E-05	-2.34E-04	1.69E-04	2.85E-04	7.60E-05	-2.49E-04	1.80E-04	3.03E-04
Ionizing radiation HH	3.58E-04	-5.75E-05	1.36E-04	1.74E-04	3.84E-04	-6.17E-05	1.45E-04	1.86E-04
Photochemical ozone formation	9.64E-04	6.15E-04	1.70E-04	3.62E-04	1.03E-03	6.56E-04	1.81E-04	3.86E-04
Acidification	9.91E-04	-1.19E-05	6.24E-04	1.77E-03	1.06E-03	-1.27E-05	6.68E-04	1.90E-03
Terrestrial eutrophication	6.97E-04	9.29E-06	4.10E-04	1.28E-03	7.35E-04	9.79E-06	4.32E-04	1.35E-03
Freshwater eutrophication	5.96E-03	-2.40E-05	2.33E-04	1.76E-03	6.28E-03	-2.53E-05	2.46E-04	1.85E-03
Marine eutrophication	4.63E-04	4.23E-05	6.64E-04	1.81E-03	4.88E-04	4.46E-05	7.00E-04	1.90E-03
Freshwater ecotoxicity	2.21E-04	-3.87E-07	2.17E-05	4.43E-05				
Land use	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	-9.77E-06	-1.60E-04	3.46E-04	1.69E-04	-1.04E-05	-1.69E-04	3.67E-04	1.79E-04
Abiotic depletion	-4.34E-05	-2.60E-05	-8.38E-05	-2.70E-04	-4.65E-05	-2.79E-05	-8.97E-05	-2.89E-04
Abiotic depletion	-8.23E-	-	2.56E-	-	-8.83E-04	-	2.74E-	-

(fossil fuels)	04	1.63E-04	04	6.69E-05		1.75E-04	04	7.17E-05
Sum	**1.87E-02**	**3.24E-02**	**2.92E-03**	**9.51E-03**	**1.84E-02**	**3.43E-02**	**3.61E-03**	**1.07E-02**

Case study 4: single-use cutlery – reference

Table 238. Weighted results for Case study 4: single-use cutlery – reference

Weighted results	**with toxicity**			**without toxicity**		
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**	**Recycling**	**Incineration**	**Landfilling**
Climate change	1.45E-02	1.27E-02	2.91E-02	1.52E-02	1.33E-02	3.07E-02
Ozone depletion	8.30E-04	-2.67E-05	1.78E-03	8.88E-04	-2.86E-05	1.90E-03
Human toxicity, non-cancer effects	3.62E-04	5.33E-04	1.66E-05			
Human toxicity, cancer effects	-2.31E-05	1.81E-03	-6.19E-05			
Particulate matter	-2.09E-03	-1.19E-04	-2.15E-04	-2.22E-03	-1.27E-04	-2.29E-04
Ionizing radiation HH	3.02E-04	6.09E-04	-5.16E-05	3.23E-04	6.52E-04	-5.53E-05
Photochemical ozone formation	8.99E-07	1.03E-03	5.95E-04	9.59E-07	1.10E-03	6.35E-04
Acidification	-4.12E-04	1.15E-03	-2.48E-06	-4.41E-04	1.23E-03	-2.66E-06
Terrestrial eutrophication	1.49E-04	7.72E-04	1.34E-05	1.57E-04	8.14E-04	1.42E-05
Freshwater eutrophication	3.72E-03	7.14E-03	-2.18E-05	3.92E-03	7.52E-03	-2.30E-05
Marine eutrophication	8.84E-05	5.12E-04	5.25E-05	9.32E-05	5.40E-04	5.53E-05
Freshwater ecotoxicity	2.19E-04	1.06E-03	1.95E-06			
Land use	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	-1.14E-03	-2.00E-04	-1.34E-04	-1.21E-03	-2.12E-04	-1.42E-04
Abiotic depletion	-1.38E-05	-8.04E-05	-2.45E-05	-1.48E-05	-8.60E-05	-2.62E-05
Abiotic depletion (fossil fuels)	-6.44E-03	-1.92E-03	-1.23E-04	-6.90E-03	-2.06E-03	-1.32E-04
Sum	**1.00E-02**	**2.49E-02**	**3.10E-02**	**9.82E-03**	**2.27E-02**	**3.27E-02**

Case study 5: agricultural mulching film – baseline

Table 239. Weighted results for Case study 5: agricultural mulching film – baseline

Weighted results	**with toxicity**			**without toxicity**		
per 1 kg for each technology	**Incineration**	**Landfilling**	**In-situ**	**Incineration**	**Landfilling**	**In-situ**
Climate change	8.11E-03	4.40E-03	1.18E-03	8.55E-03	4.64E-03	1.25E-03
Ozone depletion	-9.24E-06	1.96E-04	-4.20E-10	-9.88E-06	2.10E-04	-4.50E-10

Human toxicity, non-cancer effects	1.14E-03	4.13E-05	1.07E-04			
Human toxicity, cancer effects	1.17E-02	5.03E-06	3.98E-04			
Particulate matter	2.77E-04	-5.64E-06	3.24E-06	2.95E-04	-6.00E-06	3.45E-06
Ionizing radiation HH	2.34E-04	2.48E-06	-2.52E-08	2.51E-04	2.65E-06	-2.70E-08
Photochemical ozone formation	1.41E-03	9.60E-05	3.17E-05	1.50E-03	1.02E-04	3.38E-05
Acidification	1.27E-03	2.36E-05	1.99E-05	1.36E-03	2.53E-05	2.13E-05
Terrestrial eutrophication	9.27E-04	1.39E-05	1.89E-05	9.77E-04	1.46E-05	1.99E-05
Freshwater eutrophication	1.30E-03	3.07E-07	2.62E-07	1.37E-03	3.23E-07	2.76E-07
Marine eutrophication	6.48E-04	3.76E-05	1.44E-05	6.83E-04	3.96E-05	1.52E-05
Freshwater ecotoxicity	7.65E-04	6.29E-06	2.07E-05			
Land use	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	6.53E-04	3.33E-05	-2.28E-06	6.93E-04	3.53E-05	-2.42E-06
Abiotic depletion	-2.29E-05	-1.99E-06	-1.63E-08	-2.45E-05	-2.13E-06	-1.75E-08
Abiotic depletion (fossil fuels)	1.45E-03	7.34E-05	-6.84E-06	1.55E-03	7.86E-05	-7.34E-06
Sum	**2.98E-02**	**4.92E-03**	**1.79E-03**	**1.72E-02**	**5.14E-03**	**1.33E-03**

Case study 5: agricultural mulching film – reference

Table 240. Weighted results for Case study 5: agricultural mulching film – reference

Weighted results	**with toxicity**			**without toxicity**		
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**	**Recycling**	**Incineration**	**Landfilling**
Climate change	8.25E-03	9.39E-03	7.31E-04	8.70E-03	9.89E-03	7.70E-04
Ozone depletion	-7.59E-06	-1.68E-05	4.99E-07	-8.12E-06	-1.80E-05	5.34E-07
Human toxicity, non-cancer effects	1.73E-03	1.25E-03	5.01E-05			
Human toxicity, cancer effects	1.23E-02	1.28E-02	1.60E-05			
Particulate matter	2.17E-04	1.37E-04	2.62E-05	2.31E-04	1.46E-04	2.79E-05
Ionizing radiation HH	3.30E-04	3.86E-04	1.11E-05	3.54E-04	4.14E-04	1.19E-05
Photochemical ozone formation	1.35E-03	1.40E-03	2.72E-05	1.44E-03	1.49E-03	2.90E-05
Acidification	1.25E-03	1.32E-03	3.06E-05	1.34E-03	1.41E-03	3.28E-05
Terrestrial eutrophication	9.13E-04	9.35E-04	1.56E-05	9.63E-04	9.86E-04	1.64E-05

Freshwater eutrophication	1.31E-03	1.33E-03	3.76E-06	1.38E-03	1.40E-03	3.97E-06
Marine eutrophication	6.75E-04	6.55E-04	3.96E-05	7.11E-04	6.91E-04	4.17E-05
Freshwater ecotoxicity	8.80E-04	9.79E-04	7.69E-06			
Land use	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	1.68E-03	5.47E-04	6.30E-05	1.78E-03	5.81E-04	6.69E-05
Abiotic depletion	-1.09E-05	-4.61E-05	1.26E-06	-1.16E-05	-4.93E-05	1.34E-06
Abiotic depletion (fossil fuels)	8.80E-04	8.14E-04	1.12E-04	9.43E-04	8.72E-04	1.20E-04
Sum	**3.18E-02**	**3.19E-02**	**1.14E-03**	**1.78E-02**	**1.78E-02**	**1.12E-03**

Case study 6: food packaging film – baseline

Table 241. Weighted results for Case study 6: food packaging film – baseline

Weighted results	**with toxicity**				
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**	**Industrial composting**	**Anaerobic digestion**
Climate change	3.29E-02	1.65E-02	6.32E-02	1.59E-03	4.59E-03
Ozone depletion	1.72E-03	-1.90E-05	4.22E-03	6.67E-06	-7.01E-05
Human toxicity, non-cancer effects	2.78E-04	3.46E-04	3.74E-05	1.11E-02	1.44E-02
Human toxicity, cancer effects	1.26E-03	2.72E-03	-1.33E-04	-4.48E-03	-1.93E-02
Particulate matter	-2.72E-04	4.25E-04	-4.64E-04	5.00E-04	-6.20E-04
Ionizing radiation HH	1.33E-04	5.41E-04	-1.11E-04	2.93E-04	7.40E-04
Photochemical ozone formation	1.27E-03	2.18E-03	1.29E-03	4.67E-04	-1.68E-04
Acidification	8.45E-04	2.15E-03	-4.10E-06	1.19E-03	5.49E-03
Terrestrial eutrophication	7.24E-04	1.56E-03	2.92E-05	7.39E-04	3.97E-03
Freshwater eutrophication	3.11E-03	6.04E-03	-4.71E-05	4.00E-04	7.25E-04
Marine eutrophication	5.01E-04	1.02E-03	1.12E-04	1.31E-03	6.07E-03
Freshwater ecotoxicity	1.83E-04	5.09E-04	4.06E-06	4.18E-04	-2.34E-04
Land use	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	2.26E-04	1.57E-05	-2.88E-04	9.56E-04	-9.82E-04
Abiotic depletion	-4.46E-05	-5.63E-05	-5.28E-05	-1.11E-04	-1.15E-03
Abiotic depletion (fossil fuels)	-4.28E-03	-1.20E-03	-2.64E-04	9.29E-04	-4.23E-03
Sum	**3.85E-02**	**3.27E-02**	**6.75E-02**	**1.54E-02**	**9.14E-03**

Table 242. Weighted results for Case study 6: food packaging film – baseline

Weighted results	**without toxicity**				
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**	**Industrial composting**	**Anaerobic digestion**
Climate change	3.46E-02	1.74E-02	6.66E-02	1.68E-03	4.84E-03

Ozone depletion	1.84E-03	-2.04E-05	4.52E-03	7.13E-06	-7.50E-05
Human toxicity, non-cancer effects					
Human toxicity, cancer effects					
Particulate matter	-2.90E-04	4.53E-04	-4.94E-04	5.33E-04	-6.60E-04
Ionizing radiation HH	1.43E-04	5.80E-04	-1.19E-04	3.14E-04	7.93E-04
Photochemical ozone formation	1.36E-03	2.33E-03	1.37E-03	4.98E-04	-1.79E-04
Acidification	9.05E-04	2.31E-03	-4.39E-06	1.27E-03	5.88E-03
Terrestrial eutrophication	7.63E-04	1.65E-03	3.08E-05	7.79E-04	4.18E-03
Freshwater eutrophication	3.28E-03	6.36E-03	-4.96E-05	4.21E-04	7.63E-04
Marine eutrophication	5.28E-04	1.08E-03	1.18E-04	1.38E-03	6.40E-03
Freshwater ecotoxicity					
Land use	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	2.39E-04	1.66E-05	-3.05E-04	1.01E-03	-1.04E-03
Abiotic depletion	-4.78E-05	-6.03E-05	-5.65E-05	-1.19E-04	-1.23E-03
Abiotic depletion (fossil fuels)	-4.59E-03	-1.29E-03	-2.83E-04	9.96E-04	-4.53E-03
Sum	**3.87E-02**	**3.08E-02**	**7.13E-02**	**8.78E-03**	**1.51E-02**

Case study 6: food packaging film – alternative

Table 243. Weighted results for Case study 6: food packaging film – alternative

Weighted results	with toxicity		
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**
Climate change	3.43E-02	1.86E-02	5.77E-02
Ozone depletion	1.77E-03	-2.92E-05	3.79E-03
Human toxicity, non-cancer effects	3.44E-04	5.57E-04	5.57E-05
Human toxicity, cancer effects	3.84E-03	1.13E-02	-1.14E-04
Particulate matter	-3.23E-04	2.64E-04	-4.12E-04
Ionizing radiation HH	1.96E-04	7.48E-04	-9.67E-05
Photochemical ozone formation	1.32E-03	2.24E-03	1.19E-03
Acidification	8.97E-04	2.29E-03	9.43E-06
Terrestrial eutrophication	7.52E-04	1.63E-03	3.34E-05
Freshwater eutrophication	3.29E-03	6.51E-03	-4.13E-05
Marine eutrophication	5.22E-04	1.07E-03	1.20E-04
Freshwater ecotoxicity	4.12E-04	1.26E-03	7.09E-06
Land use	0.00E+00	0.00E+00	0.00E+00
Water use	1.88E-04	-1.01E-04	-2.35E-04
Abiotic depletion	-5.47E-05	-8.71E-05	-4.76E-05
Abiotic depletion (fossil fuels)	-4.52E-03	-1.98E-03	-1.92E-04
Sum	**4.29E-02**	**4.43E-02**	**6.18E-02**

Table 244. Weighted results for Case study 6: food packaging film – alternative

Weighted results	without toxicity		
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**
Climate change	3.61E-02	1.96E-02	6.08E-02
Ozone depletion	1.89E-03	-3.12E-05	4.06E-03
Human toxicity, non-cancer effects			
Human toxicity, cancer effects			
Particulate matter	-3.44E-04	2.82E-04	-4.39E-04
Ionizing radiation HH	2.10E-04	8.02E-04	-1.04E-04
Photochemical ozone formation	1.40E-03	2.39E-03	1.27E-03
Acidification	9.61E-04	2.45E-03	1.01E-05
Terrestrial eutrophication	7.93E-04	1.72E-03	3.52E-05
Freshwater eutrophication	3.47E-03	6.86E-03	-4.35E-05
Marine eutrophication	5.50E-04	1.13E-03	1.26E-04
Freshwater ecotoxicity			
Land use	0.00E+00	0.00E+00	0.00E+00
Water use	2.00E-04	-1.07E-04	-2.50E-04
Abiotic depletion	-5.85E-05	-9.32E-05	-5.10E-05
Abiotic depletion (fossil fuels)	-4.85E-03	-2.13E-03	-2.06E-04
Sum	**4.04E-02**	**3.29E-02**	**6.52E-02**

Case study 6: food packaging film – reference

Table 245. Weighted results for Case study 6: food packaging film – reference

Weighted results	with toxicity		
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**
Climate change	3.43E-02	1.86E-02	5.77E-02
Ozone depletion	1.77E-03	-2.92E-05	3.79E-03
Human toxicity, non-cancer effects	3.44E-04	5.57E-04	5.57E-05
Human toxicity, cancer effects	3.84E-03	1.13E-02	-1.14E-04
Particulate matter	-3.23E-04	2.64E-04	-4.12E-04
Ionizing radiation HH	1.96E-04	7.48E-04	-9.67E-05
Photochemical ozone formation	1.32E-03	2.24E-03	1.19E-03
Acidification	8.97E-04	2.29E-03	9.43E-06
Terrestrial eutrophication	7.52E-04	1.63E-03	3.34E-05
Freshwater eutrophication	3.29E-03	6.51E-03	-4.13E-05
Marine eutrophication	5.22E-04	1.07E-03	1.20E-04

Freshwater ecotoxicity	4.12E-04	1.26E-03	7.09E-06
Land use	0.00E+00	0.00E+00	0.00E+00
Water use	1.88E-04	-1.01E-04	-2.35E-04
Abiotic depletion	-5.47E-05	-8.71E-05	-4.76E-05
Abiotic depletion (fossil fuels)	-4.52E-03	-1.98E-03	-1.92E-04
Sum	**4.29E-02**	**4.43E-02**	**6.18E-02**

Table 246. Weighted results for Case study 6: food packaging film – reference

Weighted results	**without toxicity**		
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**
Climate change	3.61E-02	1.96E-02	6.08E-02
Ozone depletion	1.89E-03	-3.12E-05	4.06E-03
Human toxicity, non-cancer effects			
Human toxicity, cancer effects			
Particulate matter	-3.44E-04	2.82E-04	-4.39E-04
Ionizing radiation HH	2.10E-04	8.02E-04	-1.04E-04
Photochemical ozone formation	1.40E-03	2.39E-03	1.27E-03
Acidification	9.61E-04	2.45E-03	1.01E-05
Terrestrial eutrophication	7.93E-04	1.72E-03	3.52E-05
Freshwater eutrophication	3.47E-03	6.86E-03	-4.35E-05
Marine eutrophication	5.50E-04	1.13E-03	1.26E-04
Freshwater ecotoxicity			
Land use	0.00E+00	0.00E+00	0.00E+00
Water use	2.00E-04	-1.07E-04	-2.50E-04
Abiotic depletion	-5.85E-05	-9.32E-05	-5.10E-05
Abiotic depletion (fossil fuels)	-4.85E-03	-2.13E-03	-2.06E-04
Sum	**4.04E-02**	**3.29E-02**	**6.52E-02**

Case study 7: single-use plastic carrier bags – baseline

Table 247. Weighted results for Case study 7: single-use plastic carrier bags – baseline

Weighted results	**with toxicity**				**without toxicity**			
per 1 kg for each technology	**MSWI**	**Landfilling**	**Industiral composting**	**AD**	**MSWI**	**Landfilling**	**Industiral composting**	**AD**
Climate change	3.85E-03	7.12E-03	7.31E-04	1.41E-03	4.06E-03	7.50E-03	7.70E-04	1.48E-03
Ozone depletion	-1.06E-	3.78E-	-3.25E-	-	-1.14E-05	4.05E-	-3.47E-	-

	05	04	06	8.19E-06		04	06	8.76E-06
Human toxicity, non-cancer effects	-3.77E-05	-8.29E-06	1.17E-04	1.15E-04				
Human toxicity, cancer effects	-9.44E-05	-3.20E-05	-1.67E-04	-1.86E-04				
Particulate matter	-1.44E-04	-6.45E-05	-5.66E-05	-1.76E-04	-1.53E-04	-6.87E-05	-6.03E-05	-1.87E-04
Ionizing radiation HH	2.24E-04	-4.97E-05	6.49E-05	1.29E-04	2.40E-04	-5.33E-05	6.96E-05	1.38E-04
Photochemical ozone formation	1.27E-04	1.29E-04	6.42E-06	6.32E-05	1.36E-04	1.38E-04	6.85E-06	6.75E-05
Acidification	2.02E-04	-4.61E-05	3.27E-05	1.35E-04	2.16E-04	-4.93E-05	3.51E-05	1.45E-04
Terrestrial eutrophication	1.14E-04	-7.50E-06	1.32E-05	8.12E-05	1.20E-04	-7.91E-06	1.39E-05	8.55E-05
Freshwater eutrophication	1.42E-04	-1.48E-05	6.94E-06	4.23E-05	1.50E-04	-1.56E-05	7.31E-06	4.45E-05
Marine eutrophication	7.63E-05	1.18E-06	1.35E-05	7.64E-05	8.04E-05	1.24E-06	1.43E-05	8.05E-05
Freshwater ecotoxicity	-2.97E-05	-2.57E-06	-9.72E-06	-1.91E-05				
Land use	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	-1.36E-04	-7.96E-06	-4.04E-05	-8.77E-05	-1.44E-04	-8.44E-06	-4.29E-05	-9.31E-05
Abiotic depletion	-3.23E-05	-4.87E-06	-1.17E-05	-3.42E-05	-3.46E-05	-5.21E-06	-1.25E-05	-3.66E-05
Abiotic depletion (fossil fuels)	-8.83E-04	-1.69E-04	-2.55E-04	-5.14E-04	-9.47E-04	-1.82E-04	-2.74E-04	-5.51E-04
Sum	**3.37E-03**	**7.22E-03**	**4.42E-04**	**1.02E-03**	**3.71E-03**	**7.65E-03**	**5.24E-04**	**1.17E-03**

Case study 7: single-use carrier bags – alternative

Table 248. Weighted results for Case study 7: single-use carrier bags – alternative

Weighted results	with toxicity			without toxicity		
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**	**Recycling**	**Incineration**	**Landfilling**
Climate change	-1.76E-04	5.13E-03	5.49E-04	-1.85E-04	5.40E-03	5.78E-04
Ozone depletion	-9.42E-06	-1.82E-05	3.19E-05	-1.01E-05	-1.95E-05	3.42E-05
Human toxicity, non-cancer effects	7.54E-05	6.66E-05	1.22E-05			

Human toxicity, cancer effects	-1.52E-04	1.04E-03	2.17E-06			
Particulate matter	-4.24E-04	-2.84E-04	2.82E-06	-4.51E-04	-3.02E-04	3.00E-06
Ionizing radiation HH	3.84E-05	3.76E-04	-1.12E-07	4.12E-05	4.03E-04	-1.20E-07
Photochemical ozone formation	-3.81E-04	1.15E-04	1.64E-05	-4.06E-04	1.22E-04	1.75E-05
Acidification	-1.88E-04	2.50E-04	5.01E-06	-2.01E-04	2.67E-04	5.36E-06
Terrestrial eutrophication	-6.24E-05	1.22E-04	3.48E-06	-6.58E-05	1.29E-04	3.67E-06
Freshwater eutrophication	-2.76E-05	1.69E-04	7.57E-08	-2.91E-05	1.78E-04	7.98E-08
Marine eutrophication	-5.22E-05	8.35E-05	1.00E-05	-5.51E-05	8.80E-05	1.06E-05
Freshwater ecotoxicity	-9.94E-05	1.84E-04	1.79E-06			
Land use	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	7.61E-05	-2.42E-04	5.58E-06	8.07E-05	-2.57E-04	5.92E-06
Abiotic depletion	-9.26E-06	-5.55E-05	3.03E-08	-9.91E-06	-5.94E-05	3.24E-08
Abiotic depletion (fossil fuels)	-4.08E-03	-1.52E-03	1.82E-05	-4.37E-03	-1.63E-03	1.95E-05
Sum	**-5.47E-03**	**5.42E-03**	**6.59E-04**	**-5.66E-03**	**4.33E-03**	**6.78E-04**

Case study 7: single-use carrier bags – reference

Table 249. Weighted results for Case study 7: single-use carrier bags – reference

Weighted results	**with toxicity**			**without toxicity**		
per 1 kg for each technology	**Recycling**	**Incineration**	**Landfilling**	**Recycling**	**Incineration**	**Landfilling**
Climate change	-1.76E-04	5.13E-03	5.49E-04	-1.85E-04	5.40E-03	5.78E-04
Ozone depletion	-9.42E-06	-1.82E-05	3.19E-05	-1.01E-05	-1.95E-05	3.42E-05
Human toxicity, non-cancer effects	7.54E-05	6.66E-05	1.22E-05			
Human toxicity, cancer effects	-1.52E-04	1.04E-03	2.17E-06			
Particulate matter	-4.24E-04	-2.84E-04	2.82E-06	-4.51E-04	-3.02E-04	3.00E-06
Ionizing radiation HH	3.84E-05	3.76E-04	-1.12E-07	4.12E-05	4.03E-04	-1.20E-07
Photochemical ozone formation	-3.81E-04	1.15E-04	1.64E-05	-4.06E-04	1.22E-04	1.75E-05
Acidification	-1.88E-04	2.50E-04	5.01E-06	-2.01E-04	2.67E-04	5.36E-06
Terrestrial eutrophication	-6.24E-05	1.22E-04	3.48E-06	-6.58E-05	1.29E-04	3.67E-06
Freshwater eutrophication	-2.76E-05	1.69E-04	7.57E-08	-2.91E-05	1.78E-04	7.98E-08

Marine eutrophication	-5.22E-05	8.35E-05	1.00E-05	-5.51E-05	8.80E-05	1.06E-05
Freshwater ecotoxicity	-9.94E-05	1.84E-04	1.79E-06			
Land use	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Water use	7.61E-05	-2.42E-04	5.58E-06	8.07E-05	-2.57E-04	5.92E-06
Abiotic depletion	-9.26E-06	-5.55E-05	3.03E-08	-9.91E-06	-5.94E-05	3.24E-08
Abiotic depletion (fossil fuels)	-4.08E-03	-1.52E-03	1.82E-05	-4.37E-03	-1.63E-03	1.95E-05
Sum	**-5.47E-03**	**5.42E-03**	**6.59E-04**	**-5.66E-03**	**4.33E-03**	**6.78E-04**

Case 7: Carrier bag reuse scenario

1 Primary resuse

The primary reuse occurs a number of times, which each time will avoid the production, use and EoL of a carrier bag, see Figure 328. The primary reuse has been modelled for all possible EoL technologies. In the separate product systems, the carrier bag avoided is of the same type, e.g. a bio-based starch plastic carrier bag will avoid a starch plastic carrier bag and the LDPE bags will avoid a petrochemical or bio-based carrier bag.

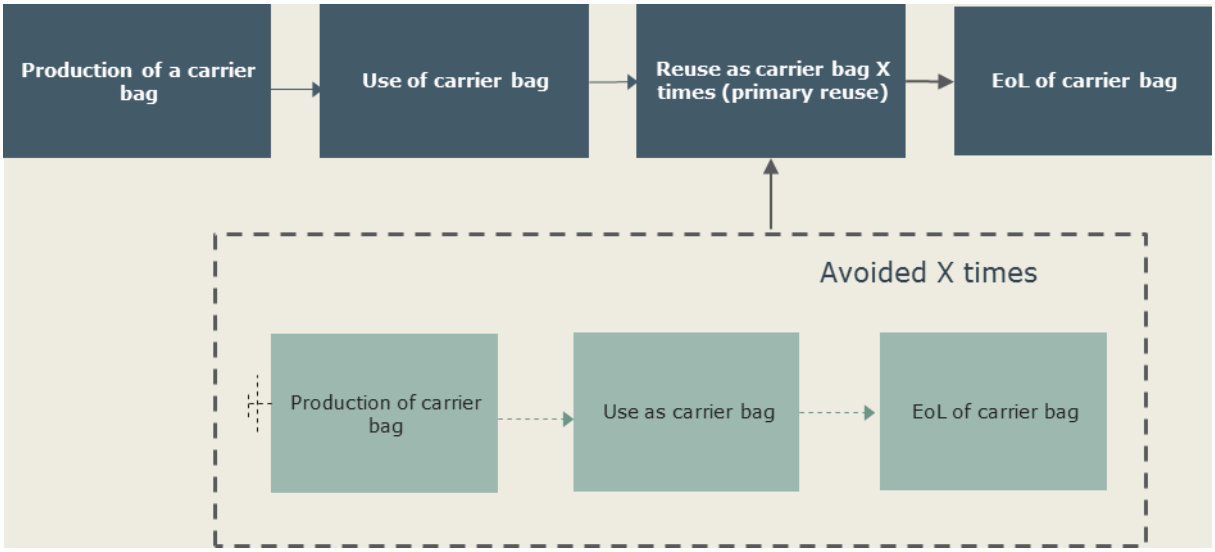

Figure 328. Flow chart for primary reuse as carrier bag.

2 Secondary reuse

Secondary reuse as a waste bag is modelled as follows and are depicted in Figure 329. The carrier bag is used one time as a carrier bag and afterwards used as a waste bag. This secondary reuse will avoid one waste bag. A waste bag is defined as a thinner plastic bag (amounting to 9 grams (DTU, 2017)). This modelling has been done on all possible technologies, with the following EoL scenario for each product system;

Bio-based starch: avoids a thin starch plastic waste bag:

- The carrier bag is used as organic waste bag and followed by 100 % industrial composting.

Petrochemical LDPE: avoids a thin LDPE waste bag:

- 30 % Recycling scenario; the carrier bag is used for residual waste and hence cannot be recycled. Instead the EoL is changed to European mix of incineration (53 %) and landfilling (47 %).
- 39 % incineration: the carrier bag is used as residual waste bag and followed by incineration (100 %).
- 31 % landfill: the carrier bag is used for residual waste and followed by landfilling (100 %).

The residual and organic waste the waste bag holds is not a part of the EoL-reference flow and is not included in the modelling of secondary reuse.

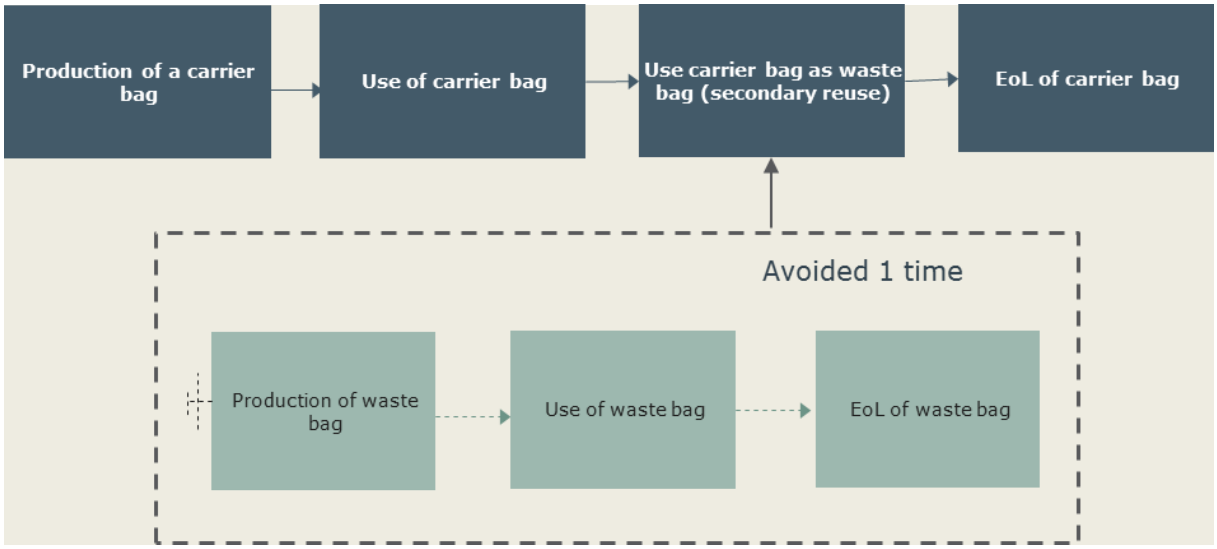

Figure 329. Flow chart for primary reuse as carrier bag.

3. Results of the reuse scenarios

3.1. Comparing the EoL technologies when reusing the carrier bags

The bio-based LDPE has a negative climate change impact from the cradle-to-gate results, due to plant uptake of carbon, and therefore the reuse will lead to a higher climate change. It is important to note that this is only the case for climate change, in all other impact categories the results for cradle-to-gate are positive and there will be savings in these impacts when reusing the bio-based LDPE carrier bag. The starch based and the petrochemical carrier bags both have a climate change impact in the cardle-to-gate results and therefore show savings in climate change when the carrier bag is reused.

The EoL technologies industrial composting and anaerobic digestion show a very similar trend and save the most in climate change after reuse of 20 times. These two EoL technologies show savings after reuse of two times. All EoL technologies of bio-based starch and petrochemical LDPE have savings in climate change after reusing twice, except incineration of petrochemical LDPE which gives saving after reusing the carrier bag 3 times.

The incineration of both LDPE product systems have the highest equal climate change impact in the start, the reuse of the bio-based starch however begins to save more after the 4th reuse as the cradle-to-gate results for bio-based starch are higher. Landfilling of both starch and petrochemical LDPE give the same amount of savings, if reused 5 times before sent to landfill the climate change savings are 0.17 kg CO_2-eq/FU.

Generally, the starch based carrier bag has a higher cradle-to-gate climate change impact per functional unit and therefore the savings become bigger every time the starch carrier bag is reused. Reusing the carrier bag up to five time the petrochemical LDPE bag followed by recycling is the most preferred choice. Industrial composting saves the most in climate change impact potentials when reusing the carrier bag 12 times or more.

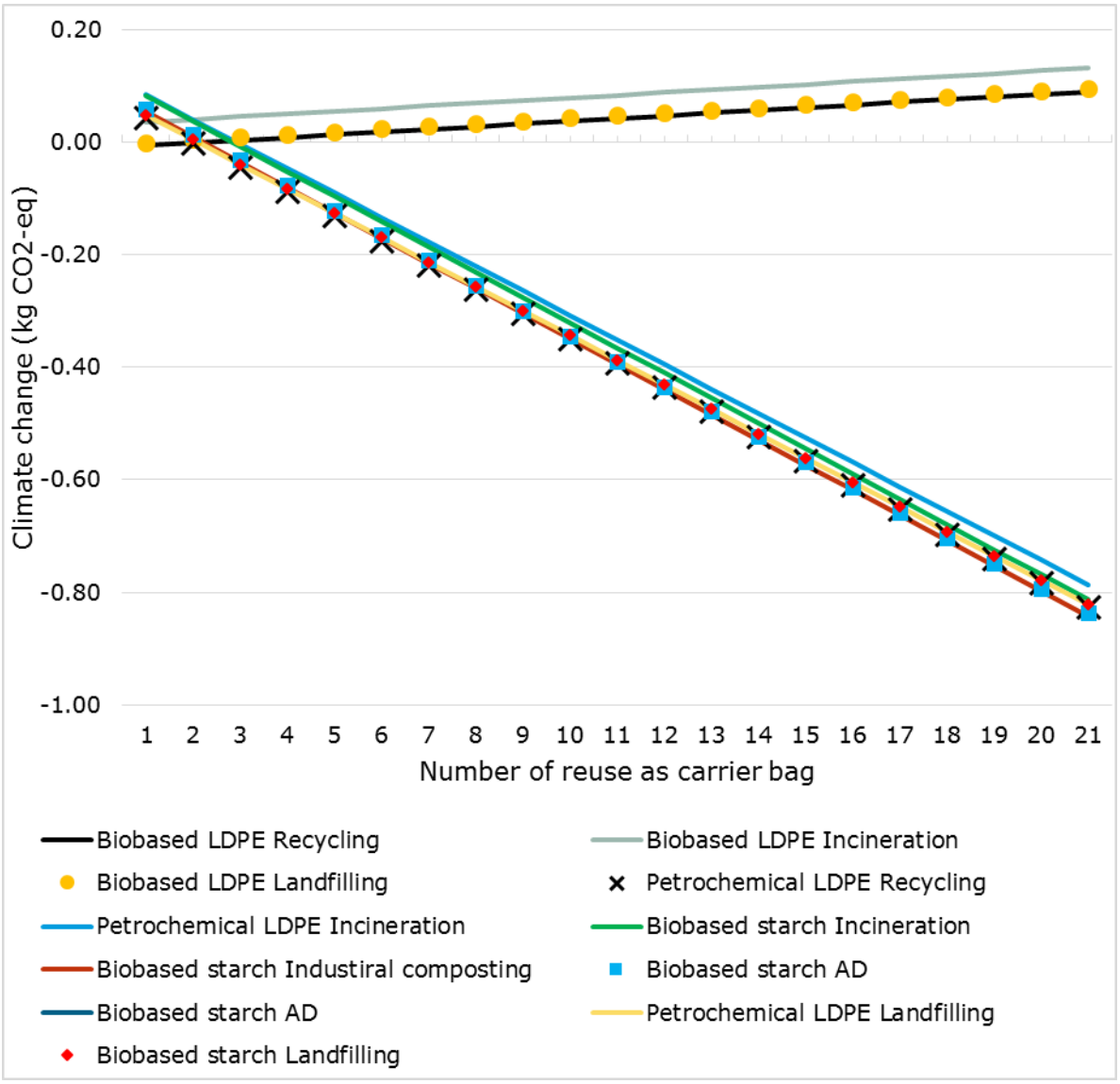

Figure 330. Climate change savings per functional unit for primary reuse of the carrier bags followed by secondary reuse as waste bag with the EoL technology.

3.2. End of Life mix - reuse results

Table 250, Table 251 and Table 252 show the results of the reuse analysis for each product system. The reuse scenarios are given based on a EoL mix when the carrier bag is thrown away. The scenarios show the impact from each product system with several different reuse times and either with or without using the carrier bag as a waste bag.

For the starch based carrier bag, using it once and throwing it away will have an impact in all categories with the EoL mix chosen. The EoL mix consists of industrial composting, incineration and landfilling, when the starch bag is used as a waste bag for industrial composting it is assumed to be used for organic waste (replace a thin starch waste bag), otherwise for residual waste (replace a thin LDPE waste bag). Reusing the starch carrier bag once and using it as a waste bag will change all impact categories to negative values. After reusing the starch carrier bag 10 times, it still saves from 1-15 % depending impact category (15 % for climate change) when the bag is also used as a waste bag. The percentage change when the number of reuses is increased is largest for ozone depletion, abitotic depletion and marine eutrophication.

Table 250. Reuse scenarios for bio-based starch carrier bags for EoL mix (30 % recycling, 39 % incineration and 31 % landfill. When reused as waste bag for the recycling , the waste is divided to 53 % incineration and 47 % landfill).

Impact category	Unit	Use 1 time and throw away	Reuse 1 time and throw away	Reuse 1 time and use as waste bag	Reuse 10 times and throw away	Reuse 10 times and use as waste bag
Climate change - total	kg CO_2 eq / FU	8.38E-02	3.90E-02	-1.39E-02	-3.64E-01	-4.17E-01
Ozone depletion	kg CFC-11 eq / FU	2.31E-08	8.60E-09	6.90E-10	-1.20E-07	-1.30E-07
Human toxicity, non-cancer effects	CTUh / FU	1.24E-08	1.25E-10	-2.71E-09	-1.11E-07	-1.14E-07
Human toxicity, cancer effects	CTUh / FU	3.01E-09	-3.75E-11	-7.68E-10	-2.75E-08	-2.82E-08
Particulate matter	kg PM2.5 eq / FU	2.72E-05	-8.41E-07	-6.47E-06	-2.53E-04	-2.58E-04
Ionizing radiation HH	kBq U235 eq / FU	1.24E-02	4.62E-04	-3.84E-03	-1.07E-01	-1.11E-01
Photochemical ozone formation	kg NMVOC eq / FU	1.96E-04	1.43E-05	-7.78E-05	-1.62E-03	-1.71E-03
Acidification	molc H+ eq / FU	3.63E-04	1.36E-05	-9.62E-05	-3.13E-03	-3.24E-03
Terrestrial eutrophication	molc N eq / FU	6.84E-04	5.25E-05	-1.91E-04	-5.64E-03	-5.88E-03
Freshwater eutrophication	kg P eq / FU	2.21E-05	6.26E-07	-5.84E-06	-1.93E-04	-1.99E-04
Marine eutrophication	kg N eq / FU	1.20E-04	4.62E-06	-2.50E-05	-1.03E-03	-1.06E-03
Freshwater ecotoxicity	CTUe / FU	3.21E-01	-1.55E-03	-8.00E-02	-2.90E+00	-2.98E+00
Land use	kg C deficit / FU					
Water use	m^3 / FU	3.85E-02	-1.83E-03	-2.19E-02	-3.64E-01	-3.85E-01
Abiotic depletion	kg Sb eq / FU	5.66E-08	-3.02E-09	-8.34E-09	-5.39E-07	-5.45E-07
Abiotic depletion (fossil fuels)	MJ / FU	1.15E+00	-7.82E-02	-6.41E-01	-1.11E+01	-1.17E+01
NREU	MJ / FU	1.33E+00	-8.61E-02	-7.04E-01	-1.28E+01	-1.34E+01

For the bio-based LDPE carrier bag using it once and throwing it away will have an impact in all categories with the EoL mix chosen. Reusing the carrier bag once and using it as a waste bag all impact categories become negative, except climate change. After reusing the bio-based LDPE bag 10 times, it only saves 5-6 % depending impact category when the bag is also used as a waste bag, but increases the climate change impact with 4 %. Increasing the number of reuses of the bio-based LDPE bag the highest percentage change is in the impact categories of ionizing radiation, human toxicity (cancer effects) and freshwater eutrophication.

Table 251. Reuse scenarios for bio-based LDPE carrier bags for EoL mix (30 % recycling, 39 % incineration and 31 % landfill. When reused as waste bag for the recycling , the waste is divided to 53 % incineration and 47 % landfill).

Impact category	Unit	Use 1 time and throw away	Reuse 1 time and throw away	Reuse 1 time and use as waste bag	Reuse 10 times and throw away	Reuse 10 times and use as waste bag

Climate change - total	kg CO_2 eq / FU	1.18E-02	1.66E-02	1.90E-02	5.98E-02	6.22E-02
Ozone depletion	kg CFC-11 eq / FU	8.02E-09	-2.39E-12	-4.03E-09	-7.22E-08	-7.62E-08
Human toxicity, non-cancer effects	CTUh / FU	5.50E-08	2.72E-10	-2.73E-08	-4.93E-07	-5.20E-07
Human toxicity, cancer effects	CTUh / FU	2.05E-09	1.12E-10	-8.60E-10	-1.73E-08	-1.83E-08
Particulate matter	kg PM2.5 eq / FU	7.59E-04	-1.80E-06	-3.84E-04	-6.85E-03	-7.23E-03
Ionizing radiation HH	kBq U235 eq / FU	7.58E-03	6.38E-04	-2.85E-03	-6.19E-02	-6.54E-02
Photochemical ozone formation	kg NMVOC eq / FU	4.72E-04	-7.66E-06	-2.49E-04	-4.33E-03	-4.57E-03
Acidification	molc H+ eq / FU	5.80E-04	5.80E-06	-2.83E-04	-5.17E-03	-5.45E-03
Terrestrial eutrophication	molc N eq / FU	2.25E-03	2.55E-05	-1.09E-03	-2.00E-02	-2.11E-02
Freshwater eutrophication	kg P eq / FU	1.65E-05	5.46E-07	-7.46E-06	-1.43E-04	-1.51E-04
Marine eutrophication	kg N eq / FU	1.47E-04	2.04E-06	-7.08E-05	-1.30E-03	-1.37E-03
Freshwater ecotoxicity	CTUe / FU	3.06E-01	3.46E-03	-1.49E-01	-2.72E+00	-2.87E+00
Land use	kg C deficit / FU					
Water use	m^3 / FU	2.29E-01	-1.68E-03	-1.18E-01	-2.08E+00	-2.20E+00
Abiotic depletion	kg Sb eq / FU	7.02E-08	-3.35E-09	-4.03E-08	-6.65E-07	-7.02E-07
Abiotic depletion (fossil fuels)	MJ / FU	2.92E-01	-2.54E-01	-5.28E-01	-5.17E+00	-5.44E+00
NREU	MJ / FU	3.86E-01	-2.71E-01	-6.01E-01	-6.18E+00	-6.51E+00

For the petrochemical LDPE carrier bag all impacts categories are positive when the bag is only used one time and then thrown away. When the bag is reused once and then used as a waste bag, saving are shon in all impact categories. Reusing the petrochemcal LDPE carrier bag will give substantial savings in ozone depletion, particulate matter abiotic depletion and more. If the LDPE carrier bag has been reused 10 times, using the carrier bag as a waste bag will give additional 4-12 % in impact savings, most in climate change savings. But if the carrier bag is only reused once, using it as a waste bag afterwards will have a great impact on the savings (up to 1700 % and 131703 % for ozone depletion). When the number of reuses of petrochemical LDPE carrier bag is increased, the perceptual change is highest for human toxicity (non-cancer), abiotic depletion and water use.

Table 252. Reuse scenarios for petrochemical LDPE carrier bags for EoL mix (30 % industrial composting, 39 % incineration and 31 % landfill).

Impact category	Unit	Use 1 time and throw away	Reuse 1 time and throw away	Reuse 1 time and use as waste bag	Reuse 10 times and throw away	Reuse 10 times and use as waste bag
Climate change - total	kg CO_2 eq / FU	6.01E-02	1.66E-02	-2.86E-02	-3.75E-01	-4.20E-01
Ozone depletion	kg CFC-11 eq / FU	5.60E-09	-2.39E-12	-3.15E-09	-5.04E-08	-5.36E-08
Human toxicity, non-cancer effects	CTUh / FU	3.82E-09	2.72E-10	-1.72E-09	-3.16E-08	-3.36E-08
Human toxicity, cancer	CTUh / FU	8.25E-10	1.12E-10	-4.18E-10	-6.30E-09	-6.83E-09

effects						
Particulate matter	kg PM2.5 eq / FU	7.72E-06	-1.80E-06	-5.45E-06	-8.75E-05	-9.12E-05
Ionizing radiation HH	kBq U235 eq / FU	6.76E-03	6.38E-04	-3.24E-03	-5.45E-02	-5.83E-02
Photochemical ozone formation	kg NMVOC eq / FU	1.59E-04	-7.66E-06	-9.98E-05	-1.51E-03	-1.60E-03
Acidification	molc H+ eq / FU	1.49E-04	5.80E-06	-8.46E-05	-1.28E-03	-1.37E-03
Terrestrial eutrophication	molc N eq / FU	3.37E-04	2.55E-05	-1.87E-04	-2.78E-03	-2.99E-03
Freshwater eutrophication	kg P eq / FU	9.19E-06	5.46E-07	-4.65E-06	-7.73E-05	-8.25E-05
Marine eutrophication	kg N eq / FU	3.23E-05	2.04E-06	-1.82E-05	-2.70E-04	-2.90E-04
Freshwater ecotoxicity	CTUe / FU	1.02E-01	3.46E-03	-5.39E-02	-8.79E-01	-9.37E-01
Land use	kg C deficit / FU					
Water use	m^3 / FU	4.95E-02	-1.68E-03	-2.44E-02	-4.62E-01	-4.85E-01
Abiotic depletion	kg Sb eq / FU	2.20E-09	-3.35E-09	-2.11E-09	-5.33E-08	-5.20E-08
Abiotic depletion (fossil fuels)	MJ / FU	1.23E+00	-2.54E-01	-8.87E-01	-1.36E+01	-1.42E+01
NREU	MJ / FU	1.34E+00	-2.71E-01	-9.53E-01	-1.48E+01	-1.55E+01

It is clear that the more often the carrier bag is used the more environmental savings are achieved. Reusing the carrier bag gives greater savings than using it as a waste bag. The best scenario is however to reuse it as many times as possible and then secondary reuse, i.e. use it as a waste bag, for bio-based starch and petrochemical LDPE product system.

ANNEX 5 BIODIVERSITY

1 Biodiversity assessment with ReCiPe 2016

1.2 Characterisation model

For predicting biodiversity loss caused by land use the ReCiPe 2016 methodology employ the most commonly used Classical SAR model to describe the relationship between species and area. See section 7.2.1 for a more detailed discussion of this model. The species loss predicted with this characterisation model is local and no spatial differentiation is considered (Hujbregts et al., 2016; 2017) [84].

1.3 Characterisation factors

The ReCiPe 2016 methodology provides characterisation factors expressing the relative local species loss resulting from 6 different land use types (annual crops, permanent crops, mosaic agriculture, urban land, pasture and meadow, and used forest). The characterisation factors are based on species richness data for 4 taxonomic groups: mammals, birds, arthropods and vascular plants. (Huijbregts et al., 2016).

Both midpoint and endpoint characterisation factors are made available in the methodology, see Table 253. Midpoint characterisation factors are expressed as annual crop equivalents while endpoint characterisation factors are expressed in PDF/m^2. The endpoint characterisation factors take into account an approximated average species density for terrestrial ecosystems. The actual difference between the two sets of characterisation factors is a conversion factor of $8.88*10^{-9}$ (PDF/m^2 annual crop eq). (Huijbregts et al., 2016).

In this study the impact assessment is carried out with endpoint characterisation factors in order to obtain a result expressed in a similar unit to that of the impact assessment carried out with the Chaudhary 2015 methodology applied in Chapter 7.

Table 253: Midpoint and endpoint characterisation factors for the impact of land occupation on species richness. (Huijbregts et al., 2016).

	Midpoint	**Endpoint**
Land use type	CFm_{occ} (annual crop equivalents)	CFe_{ooc} (PDF/m^2 land)
Used forest	0.30	$2.66*10^{-9}$
Pasture and meadow	0.55	$4.88*10^{-9}$
Annual crops	1.00	$8.88*10^{-9}$
Permanent crops	0.70	$6.22*10^{-9}$
Mosaic agriculture	0.33	$2.93*10^{-9}$

[84] When referring to the recipe 2016 methodology in this annex it is to the part of the methodology concerning impacts of land use on biodiversity.

Artificial areas	0.73	$6.48*10^{-9}$

The characterisation factor for annual crops is approximately 1.5 times greater than for permanent crops. This is likely to be due to the relatively low affinity of species to annual crops cultivation areas as alternative habitats compared to permanent crops. A similar effect is reflected in the ecoregion characterisation factors of the Chaudhary 2015 method.

1.3 Inventory and impact assessment

The characterisation factors for annual crops and permanent crops presented in Table 253 were multiplied with the respective land use requirements associated with each case study baseline feedstock resulting in a prediction of potential species loss. Table 254 summarises land use requirements (from Table 47) and local potential species loss.

Table 254. Inventory of case study baseline feedstocks and result of ReCiPe impact assessment.

Feedstock	Origin	Land use type	**Inventory** Land occupation (m^2*year/t feedstock fw)	**Impact assessment** Local relative species loss (PDF*yr/t feedstock fw)
Sugar cane	Thailand	Permanent crop	138.9	$8.64*10^{-7}$
Sugar cane	Brazil	Permanent crop	129.0	$8.02*10^{-7}$
Maize	USA	Annual crop	720.5	$6.40*10^{-6}$
Maize	Italy	Annual crop	1060	$9.42*10^{-6}$
Potato	Germany	Annual crop	233.3	$2.07*10^{-6}$

Figure 331 visualises the local potential species loss assessed for the baseline feedstocks with ReCiPe 2016. The impact assessment predicted the highest potential disappeared fraction of species for maize cultivation in Italy, followed by maize cultivation in USA, potato cultivation in Germany and sugar cane cultivation in Thailand and Brazil.

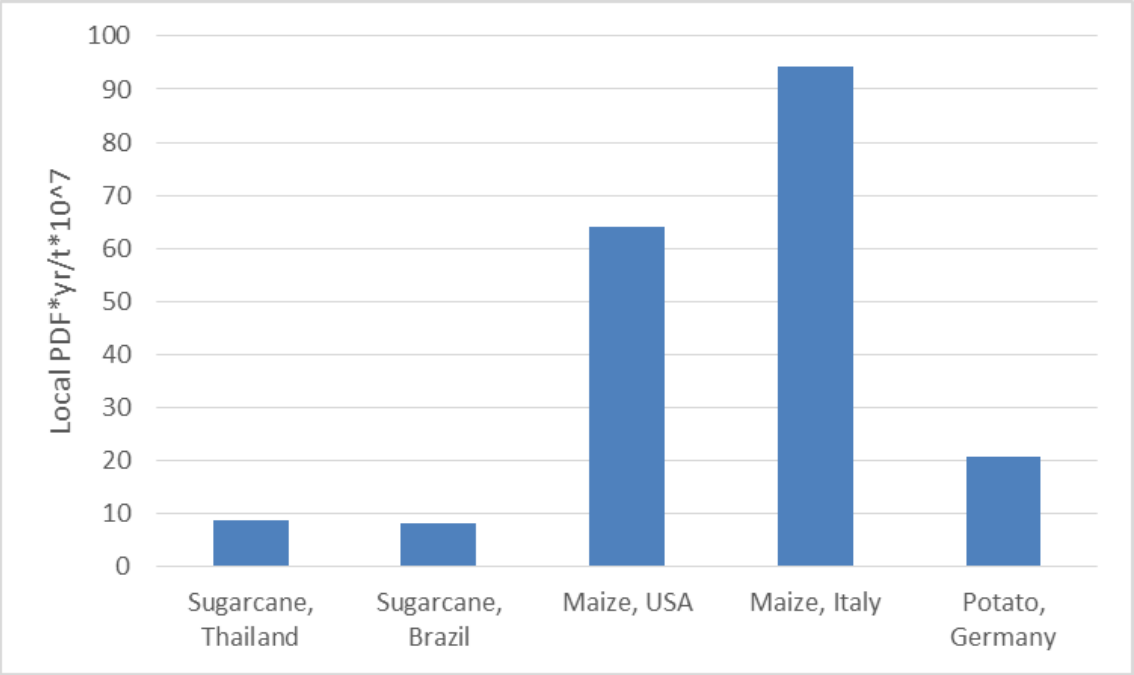

Figure 331. Local species loss per tonne fresh weight feedstock expressed as Local PDF*yr/t. Calculated using ReCiPe 2016 endpoint CFs for land occupation.

Due to the lack of consideration for spatial differentiation in the ReCiPe 2016 methodology the assessed biodiversity impacts only differentiates for cultivation of different feedstocks due variation in land use requirements and land use type. Hence, location of cultivation has no influence on the biodiversity loss other than what is reflected in the assessed land use requirements through yield. As a result, the order of feedstocks in terms of magnitude of impacts (seen in Figure 331) is dictated by the extent of land use requirements and influenced by the lower affinity of species to annual crop cultivation areas and use type (permanent crops associated with lower impact).

Consequently, when comparing the impacts related to maize cultivated in either Italy or USA and sugar cane cultivated in either Thailand or Brazil, a very little difference in impacts associated with sugar cane cultivation in Brazil and Thailand is found, while a much larger difference between impacts associated with maize cultivation in Italy and USA is found.

For the two PLA feedstocks the same conditions take effect. The impact assessment predicts a much higher local species loss associated with cultivation of maize in the USA compared cultivation of sugar cane in Thailand due to the larger area requirements of maize and the generally lower affinity of species to annual crop areas compared to permanent crops.

2 Comparison of results obtained with the ReCiPe 2016 and Chaudhary 2015 methodologies

Biodiversity assessment results obtained with ReCiPe 2016 and Chaudhary 2015 methodologies are not directly comparable due to the different underlying characterisation models. The ReCiPe methodology applies Classical SAR while the Chaudhary 2015 methodology applies Countryside SAR (see discussion of these in section 7.2.1). The main methodological elements of the two methodologies are compared in Table 255.

Table 255: Comparison of main methodological elements of the ReCiPe 2016 and Chaudhary 2015 methodologies. (Hujbregts et al., 2016, 2017; Goedkoop et al., 2009) (Chaudhary et al., 2015).

Methodological element	ReCiPe 2016	Chaudhary 2015
Underlying characterisation model	Classical SAR	Country side SAR
Spatial differentiation/regionalisation of impacts	No	Yes
Geographical unit of inventory	None	Ecoregion, country, continental
Impact categories	Local species loss	Regional species loss Global species loss
Land use types	Annual crops, permanent crops, mosaic agriculture, urban land, pasture and meadow, and used forest	Annual crops, permanent crops, pasture, urban extensive forestry, intensive forestry
Taxonomic groups	Mammals, birds, vascular plants and arthropods	Mammals, birds, vascular plants reptiles and amphibians

The land use type definitions for the two types of crops, annual and permanent, relevant for this study, are similar for the methodologies. The two methodologies does not consider species richness of entirely the same taxonomic groups in the characterisation factors. The Chaudhary 2015 methodology additionally includes 'reptiles' and 'amphibians', but not 'arthropods'. Both methodologies is likely to be influenced by the fact that data on the taxonomic groups 'plants' and 'birds' is more extensive than for the other groups.

The starting point for both methodologies is the work of de Baan et al. (2013) and Elshout et al. (2014). The Chaudhary 2016 methodology thus builds on 'local species loss', just as ReCiPe 2016, but whereas the ReCiPe 2016 methodology only cover local impacts of land use, the Chaudhary methodology models all the way to global impacts on biodiversity via regional impacts (See Figure 19).

A direct comparison of results obtained with the two different methodologies will thus be a comparison of local and global impacts, which would make little sense. Instead, a comparison between results in terms of which hot spots are pointed out in the respective biodiversity assessments of the case study baseline feedstocks is carried out in the next section.

2.1 Hot spot analysis comparison

The main hot spot, 'global potential species loss' and 'local species loss', predicted by the impact assessments performed using Chaudhary 2015 (Figure 21) and ReCiPe 2016 methodologies (Figure 331) respectively, is 'cultivation of maize in Italy' in both assessments.

The Chaudhary 2015 (country level) impact assessment pointed out 'maize cultivation in USA' and 'sugar cane cultivation in Thailand' as more less equal second largest impacts, whereas the ReCiPe 2016 impact assessment pointed out 'maize cultivation in USA' as the second largest impact. Due to the emphasis on land requirements, potato cultivation became more dominant in the ReCipe results.

It is difficult to draw general conclusions about the similarity of the results obtained with the two methodologies based on the limited data material. It is important to take into account that the comparison of hot spots is made on the basis of the country level assessment carried out with the Chaudhary methodology, where the effects of the vulnerability weighting is averaged out in the aggregated country characterisation factors. Had the comparison been with assessments carried out at ecoregion level for all feedstocks it could perhaps have changed the result if particularly sensitive ecoregions had been affected. There were some indications in this biodiversity study that the generally high land requirements for annual crops lessened the significance of vulnerability weighting but it is conjecture.

ANNEX 6 CONFERENCE PROGRAM AND OUTCOMES

Innovative bio-based products: Investment, Environmental Impacts and Future Perspectives

Results of a tender study

06 June 2018 | Crowne Plaza Brussels - Le Palace | Rue Gineste 3, 1210, Brussels, Belgium

Programme

Time	Programme
09:00	REGISTRATION AND COFFEE
09:30	**Welcome, high-level statement** John Bell, Director of Bioeconomy, DG Research & Innovation, European Commission
09:40	**Opening Session** Moderator: Peter Woodward, Quest Associates
09:50	**The circular economy and the bioeconomy - Partners in sustainability** Mieke De Schoenmakere, Project manager, Green Economy, European Environment Agency
10:05	**Life Cycle Assessment (LCA) of seven innovative bio-based products** Li Shen, Assistant Professor, Energy & Resources, Utrecht University
10:20	**Some key methodological aspects related to the LCA case studies** Rana Pant, Team Leader, European Commission, Joint Research Centre, Directorate Sustainable Resources - Bio-Economy Unit (D1)
10:30	**Mini-panel with:** Li Shen, Assistant Professor, Energy & Resources, Utrecht University Rana Pant, Team Leader, European Commission, Joint Research Centre, Directorate Sustainable Resources - Bio-Economy Unit (D1) Rob van der Meij, Investment Manager, Capricorn Venture Partners Udo Felten, Manager Product Related Global Environmental Sustainability and Affairs, SIG Gerfried Jungmeier, Expert on Life Cycle (Sustainability) Assessment and Biorefineries, Joanneum Research Forschungsgesellschaft
10:50	COFFEE BREAK
11:20	**Success stories of bio-based products in Europe, with insights on key enablers, challenges and the investment potentials offered** Malene Sand Jespersen, Market Director, COWI
11:35	**The Winding Road to Success for Bio-based Products: Journey of the ins and outs for CelluComp and its owners** Christian Kemp-Griffin, Executive Director and CEO, CelluComp
11:45	**Mini-panel with:** Malene Sand Jespersen, Market Director, COWI Christian Kemp-Griffin, Executive Director and CEO of CelluComp Rob van der Meij, Investment Manager, Capricorn Venture Partners Udo Felten, Manager Product Related Global Environmental Sustainability and Affairs, SIG Gerfried Jungmeier, Expert on Life Cycle (Sustainability) Assessment and Biorefineries, Joanneum Research Forschungsgesellschaft

Time	Programme		
12:05	**Top emerging bio-based products, their properties and industrial applications** **Paola Fabbri**, Associate Professor, University of Bologna		
12:15	**The Avantium approach for bio-based chemicals, from ideation towards flagships** **Ed de Jong**, VP Development at Avantium		
12:30	**Mini-panel with:** **Paola Fabbri**, Associate Professor, University of Bologna **Ed de Jong**, VP Development at Avantium **Rob van der Meij**, Investment Manager, Capricorn Venture Partners **Udo Felten**, Manager Product Related Global Environmental Sustainability and Affairs, SIG **Gerfried Jungmeier**, Expert on Life Cycle (Sustainability) Assessment and Biorefineries, Joanneum Research Forschungsgesellschaft		
12:50	BUFFET LUNCH		
14:00	**Session I: Environmental impact of innovative bio-based products** Moderator: **Alex Williamson**, Bio-Based World News In this session, the focus will be on the environmental performance of bio-based products through environmental Life Cycle Assessments (LCA). The novelty aspects of the LCA methodology applied in the project, e.g. land-use change impacts, end of life modelling and comparison with fossil reference products will be discussed.	**Session II: Successes in mobilising finance for the development and commercialisation of bio-based products** Moderator: **Luke Upton**, Bio-Based World News In this session we will look at some of the different ways in which markets can be reached, private finance can be mobilized or profitability can be achieved and hear from companies and organisations with direct experience in these crucial dimensions of success.	**Session III: Prospecting the Bioeconomy: What products are most promising?** Moderator: **Paola Fabbri**, University of Bologna This session will present and discuss the top 20 innovative bio-based products that have been identified as having the greatest potential for deployment in the next 5-10 years, with focus on the most relevant value chains from key biomass intermediates.
15:30	COFFEE BREAK		
16:00	**Closing Session** *Report from Parallel Sessions and Panel Discussion with:* **Rob van der Meij**, Investment Manager, Capricorn Venture Partners **Udo Felten**, Manager Product Related Global Environmental Sustainability and Affairs, SIG **Gerfried Jungmeier**, Expert on Life Cycle (Sustainability) Assessment and Biorefineries, Joanneum Research Forschungsgesellschaft **Malene Sand Jespersen**, Market Director, COWI		
16:50	**Closing remarks** **Gaia Fantechi**, Head of Unit, DG Research & Innovation, F2 Bio-based Products and Processing		

Parallel Session I
Environmental impact of innovative bio-based products

Moderator: **Alex Williamson**, Bio-Based World News (BBWN)

Time	Programme
14:00 - 14:15	**Introduction to the main modelling approach, impact categories and initial cradle-to-gate results illustrated for the case study of single-use drinking cups for cold drinks** **Christian Moretti**, Utrecht University
14:15 - 14:30	**Determination of the fossil reference product and comparison of impacts** **Li Shen**, Utrecht University
14:30 - 14:45	**Modelling of direct and indirect land-use impacts and other non-PEF impact categories** **Lorie Hamelin**, Hamelinlab Consulting / COWI
14:45 - 15:00	**End-of-Life scenarios for bio-based and fossil products - advanced modelling and effect on impact categories** **Linda Høibye**, COWI
15:00 - 15:25	Panel discussion with speakers
15:25 - 15:30	Wrap up and closing

Parallel Session II
Successes in mobilising finance for the development and commercialisation of bio-based products

Moderator: **Luke Upton**, Bio-Based World News (BBWN)

Time	Programme
14:00 - 14:15	**Presentation of study: key observations regarding risks and drivers on a) financing, b) commercialisation** **Lucas Bossard**, COWI
14:15 - 14:25	**Financing bio-based industries through partnerships** **Suzy Renckens**, Bio-Based Industries Consortium
14:25 - 14:40	**Deciding on investments - the process** **Rob van der Meij**, Investment Manager, Capricorn Venture Partners
14:40 - 15:15	Panel discussion with: ▪ **Tuomas Mustonen**, Paptic® bags ▪ **Pieter Imhof**, BioBTX B.V. ▪ **Giulia Gregori**, Novamont S.p.A. ▪ **Tuomas Hänninen**, Lumir Oy ▪ **Jeremiah Dutton**, Trifilon AB ▪ **Lucas Bossard**, COWI ▪ **Suzy Renckens**, Bio-Based Industries Consortium
15:15 - 15:30	Q&A

Parallel Session III
Prospecting the Bioeconomy: What products are most promising?

Moderator: **Paola Fabbri**, University of Bologna

Time	Programme
14:00 - 14:20	**Is Lignin the new oil?** Panel discussion with: ▪ **Paola Fabbri**, University of Bologna ▪ **Piret Kukk Fischer**, Fraunhofer Institute for Systems and Innovation Research ISI ▪ **Ludo Diels**, VITO
14:20 - 14:30	Q&A
14:30 - 14:50	**Urban wastes: mainly a need of reduction or a great value?** panel discussion with: ▪ **Paola Fabbri**, University of Bologna ▪ **Martijn Vis**, BTG Biomass Technology Group ▪ **Martin Junginger**, Utrecht University
14:50 - 15:00	**Emerging innovative bio-based products from BBI JU projects** **Paloma Mallorquin**, Bio-Based Industries Joint Undertaking
15:00 - 15:20	**Let's look into the future of bio-based products: what's up?** Open discussion with: ▪ **Davide Viaggi**, University of Bologna ▪ **Fabio Fava**, University of Bologna ▪ **Paloma Mallorquin**, Bio-Based Industries Joint Undertaking
15:20-15:30	Q&A

Parallel Session I: Environmental impacts of innovative bio-based products

In this session, the focus was on the environmental performance of bio-based products through environmental Life Cycle Assessments (LCA). The novelty aspects of the LCA methodology applied in the project, e.g. land use change impacts, EoL modelling and comparison with fossil reference products were discussed.

Christian Moretti from Utrecht University initiated the session with an introduction to the main modelling approach, impact categories and initial cradle-to-gate results illustrated for the case study of single-use drinking cups for cold drinks. One of the main conclusions was that early positive gains are the potential environmental benefits from bio-based plastics (polypropylene) made from used cooking oil.

Next, Li Shen from Utrecht University, presented the main aspects to be considered when using LCA data for fossil-based plastics in a comparison with LCA data for bio-based products. Li revealed that the bulk part of the LCA data for fossil-based plastics do not comply with the ILCD requirements. Furthermore, many of the impact categories in the latest PEF cannot be assessed when using the currently available LCA data for fossil-based plastics.

The third presenter was Lorie Hamelin (from Hamelinlab Consulting, a subcontractor to COWI) who presented the applied approach for modelling direct and indirect land use change impacts in the project. Lorie presented the overall assumptions, data sources and methods for the assessment. The results presented showed that land use change impacts are significant from a global warming perspective, although they do not dominant the cradle-to-grave environmental impacts.

Lastly, Linda Høibye from COWI presented the EoL of the bio-based and fossil-based products in the project. Included in the presentation was an explanation of the chosen system boundaries and main assumptions. The results showed that the EoL is typically smaller – but none-the-less important to the full life cycle of the case study including single-use cups. In general, it was concluded that recycling if the best EoL scenario.

After the presentations there was a panel discussion with the speakers. There was a discussion of the very complex task of assessing the environmental aspect from bio-based versus fossil-based plastics. Thus, this comparative project can be considered as a first step in using LCA to make such an assessment. It was concluded that LCA must be supplemented by other assessment methodologies, tools etc. to include more aspects that the LCA can do. It is expected that this will enable a more fair comparison between bio- and fossil based products.

Another conclusion was that a clear, simple and transparent way of evaluating land use change is needed.

Finally, it was concluded that the project forms the very early and brave start of evaluating the gains and the drawbacks when using LCA for the comparison of bio-based and fossil based products. This first steps on a long journey is needed to make progress towards sustainable products both fossil-and bio based.

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The BIOSPRI Study supports the development of the revised EU Bioeconomy Strategy and consists of three work packages on 1) LCAs and environmental footprint of bio-based products, 2) Commercial and Financial Success stories of bio-based products and 3) Identification of Top 20 innovative bio-based products. This report delivers on the first package and provides science-based facts and evidences on the environmental impacts of innovative bio-based products, to support the future bio-economy policy and decision-making at the EU level, and to support the implementation of the plastics strategy. It concerns mostly plastic products and compares these with petrochemical plastic counterparts. Seven LCA (life cycle assessment) case studies were carried out covering three major commercialised bio-based polymers, namely i) bio-based PET (polyethylene terephthalate, case study "Beverage Bottles"), ii) PLA (polylactic acid, case studies "Single-use cups", "Single-use Cutlery" and "Packaging films") and iii) starch plastics (case studies "Clips", "Mulch films" and "Carrier bags").

Studies and reports

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