# LAND FOR BIOMASS?

## SUSTAINABILITY PERFORMANCE OF BIO-BASED VALUE CHAINS UNDER CONTEXT-SPECIFIC CONDITIONS

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Sustainability performance of bio-based value chains under context-specific conditions

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# Land for biomass?

Sustainability performance of bio-based value chains under context-specific conditions

## Land voor biomassa?

Duurzaamheidsprestaties van bio-based waardeketens onder contextspecifieke omstandigheden

(met een samenvatting in het Nederlands)

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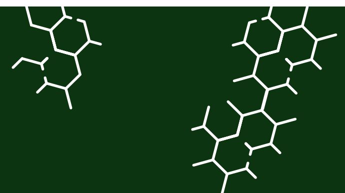
# 1

# Introduction



## 1.1 THE URGENCY TO MITIGATE ADVERSE IMPACTS OF CLIMATE CHANGE

It is unequivocal that human-induced greenhouse gas (GHG) emissions are the main driver of global warming[1]. In the last decades, atmospheric GHG concentrations have increased mainly driven by the combustion of fossil fuels, resulting in a global average surface temperature increase of 1°C since the pre-industrial era[2]. Unless deep GHG emission reductions are attained, temperature rise will continue in the upcoming years and possibly exceed 1.5°C to 2°C by the end of the 21st-century compared to pre-industrial levels[1,2]. The United Nations climate change conference of the parties (COP) 21 (2015) specified that it is crucial to avoid exceeding the 1.5°C thresholds to prevent large-scale irreversible climate change impacts (e.g., species loss, ecosystem degradation and sea-level rise)[3]. Meeting this target requires a significant reduction in anthropogenic GHG emissions. It is required to reduce 45% of global GHG emissions by 2030 and reach net-zero emissions by 2050[1]. Some regions are stepping up in the pursuit to meet these targets. For example, as part of the European Green Deal, the European Union has proposed cutting down GHG emissions by at least 55% by 2030, achieve net-zero emissions by 2050, and align their policies to such targets[4].

Achieving significant GHG emission reductions is challenging, especially in sectors with a high carbon footprint, such as electricity, heat, and transport, jointly accounting for 45.1% of global GHG emissions[5]. Decarbonizing these sectors requires large-scale uptake of renewable energy. Although several forms of renewable energy, such as wind and photovoltaic, are expected to contribute greatly to future energy supply, bioenergy (biomass-derived energy) is expected to play an important role[2,6,7]. Currently, renewable energy sources contribute 17.7% to the total primary energy supply, of which biomass supplies 70%[8]. Bioenergy demand is expected to increase and reach 20% of the total energy supply to meet net-zero emissions by 2050[8,9] (see Figure 1-1). Therefore, biomass supply for bioenergy production will remain essential to meet climate targets.

Biomass-derived products are also expected to reduce the dependency on fossil fuels in other sectors. Bio-based value chains can provide a wide range of alternatives to fossil-based services, such as liquid fuels for the transport sector and feedstock for non-energy carriers (e.g., plastics and chemicals). For example, the substitution of fossil-based products in different applications such as liquid fuels, bio-based chemicals, and bio-based materials is expected to be based on biomass-derived products[10]. Thus, the role of biomass will be more relevant in sectors that are difficult to decarbonize and/or electrify (e.g., marine, aviation and heavy-duty vehicle transport in the transport sector), or require a low-cost dispatchable source of renewable energy for industrial processes (e.g., heat requirement for cement production)[9,11]. These conditions are expected to shift





biomass from traditional uses (i.,e. use of wood and charcoal for cooking and heating) to one that targets biorefining biomass into bio-based materials, advanced biofuels and energy for industrial processes (see Figure 1-1)[9,12].

Policy developments have also promoted using (dedicated) biomass to increase renewable energy supply share and encouraged scaling up bio-based value chains. Currently, more than 40 national governments have introduced strategies to shift from a fossil-based to a bio-based economy[13]. For example, several European countries such as Germany, The Netherlands and Spain have adopted the European bio-based strategy and established action plans to strengthen and scale up different bio-based sectors[14]. In the EU, the renewable energy directive (REDII) establishes targets and sustainability criteria for the use of energy from renewable sources, with special targets for biomass fuels[15]. Furthermore, in the last decade, the EU has invested more than €80 billion of research funding to consolidate a biorefining sector[16]. In Brazil, the development of biofuels has been promoted by government regulations to reduce the dependency on fossil fuels and develop the country's economy[17].

Overall, the development of bio-based value chains can be a crucial strategy for achieving GHG emission reduction targets and reducing our dependency on fossil fuels[14]. In addition, bio-based value chains can provide additional sustainability benefits under adequate conditions and contribute to meeting the UN Sustainable Development Goals (SDGs)[18,19]. For example, biomass production for energy purposes in marginal land (e.g., degraded land) can contribute to land restoration by providing a vegetation cover to reduce soil erosion, increase soil organic carbon, improve soil quality and provide business alternatives for new markets that enhance rural development[20–23]. However, despite the benefits of bio-based value chains, there are still many sustainability concerns associated with the large-scale deployment of these systems, especially land-related impacts from biomass production[24,25].

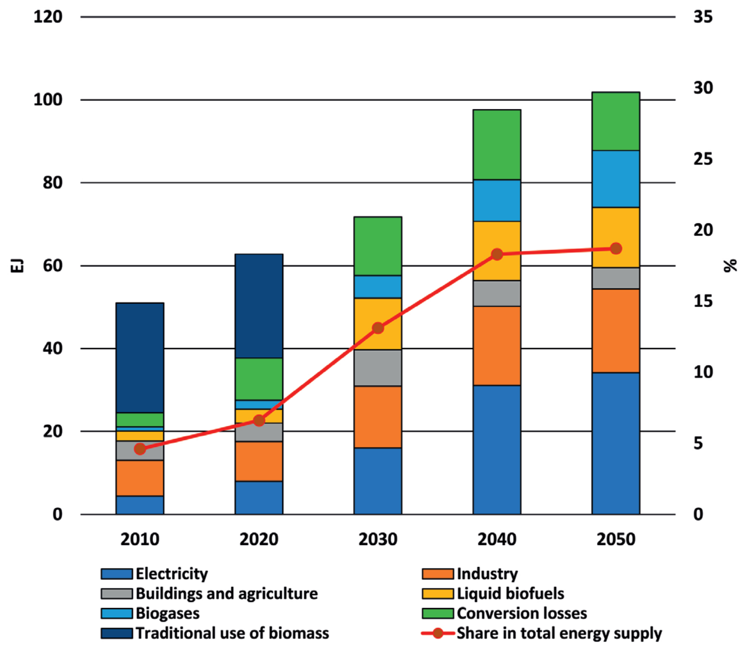



**Figure 1-1. Bioenergy demand to meet net-zero emissions by 2050. Derived from IEA (2021)[9]**

## 1.2 CONCERNS RELATED TO BIO-BASED VALUE CHAINS

The deployment of large-scale bio-based value chains to meet GHG emissions targets will require large volumes of biomass supply. It is expected that dedicated energy crops (non-food perennials) to become the predominant supply source, potentially reaching up to 68 percent of the total biomass supply by 2050 under current developments[26]. However, dedicating land for biomass production may compete with other land-based services such as conservation of natural habitats and the provision of food, feed, and fibre (i.e., agricultural production)[27–29]. Land is already considered a scarce resource in many regions of the world[30], and in the years to come, the demand for land-based services is expected to increase. Thus, increased implementation of biomass for bio-based value chains can result in the loss of natural areas (e.g., forests) or food production-related agricultural areas, negatively affecting biodiversity, losing important carbon sinks and decreasing food-related agricultural output [31–33]. Many of these impacts are associated with land use changes (LUC) that could be triggered (directly or indirectly) by additional land use for biomass production. Direct LUC is when biomass production replaces the use of the land where it is grown; indirect land use change (iLUC) occurs when the land use displaced by biomass production expands over other land uses[34].





The overall GHG emissions performance of bio-based value chains is highly affected when carbon stock changes from (i)LUC are accounted for[7]. In some instances, LUC-related carbon stock changes triggered by the cultivation of energy crops can lead to bio-based value chains with a net increase in GHG emissions[34] or that fail to comply with GHG emissions reduction targets. For example, biofuels deployed in the European market after 2021 must comply with at least 65% of GHG emissions savings in reference to the fossil fuel counterpart (accounting for LUC-related GHG emissions)[15]. GHG emissions benefits can also be considerably affected when additional carbon stock changes occur driven by displacement effects (iLUC) from energy crop cultivation[35]. However, it is also shown that for some bio-based value chains, the LUC-related carbon stock changes can also result in a net carbon sink and increase GHG savings[36]. Therefore, the suitability of bio-based value chains as climate change mitigation options will be largely determined by LUC-related GHG emissions and the capacity of these supply chains to avoid competition with other land-based services (avoid iLUC).



LUC processes triggered by energy crops production can also result in other environmental and socioeconomic impacts. For example, utilizing land for biomass production can substantially increase water withdrawals and intensify the pressure on water resources[32]. Although no irrigation is required in areas with favorable conditions, in some water-scarce regions, yields would be reduced if no irrigation is applied. As a result, more land would be required to achieve the same bioenergy production level, potentially leading to land competition. Land competition can negatively affect agricultural production and food supply and lead to increases in food prices[37]. LUC is also recognized as one of the main causes of biodiversity loss[38]. Without mitigation measures, land used for energy crops can lead to land competition with other land-based objectives, involving the loss of forest and deterioration of important ecosystems [31]. LUC processes (triggered by energy crops cultivation) can also increase soil erosion and soil losses[39]. However, under certain conditions, energy crops production has also been shown to result in positive environmental and socioeconomic impacts such as improvements in soil quality, water quality, biodiversity, expansion of new markets and enhancement of rural economies[40–43].

The specific LUC-related sustainability impacts from biomass production depend on a wide range of context-specific conditions that are spatially heterogeneous and can vary over time: previous land use (the type of land that is converted to energy crop productions e.g., cropland, pasture or forest), biophysical characteristics (e.g., climate and soil conditions), socioeconomic developments (e.g., food security), and crop management practices (see Figure 1-2)[44]. These conditions can determine the direction (positive or negative) and magnitude of sustainability LUC-related impacts. For example, producing energy crops to meet global bioenergy demand can result in a net loss of natural areas, negatively affect biodiversity and result in net carbon losses [45]. Instead, utilizing underutilized cropland or pastures for energy crop production can positively affect biodiversity (e.g., increasing pollinator communities) and provide GHG emissions savings[46,47]. Therefore, accounting for the effect of these conditions in bio-based value chains is paramount to avoid counterproductive systems. To illustrate, Bio-based value chains that fail to achieve GHG emissions savings or systems that, despite providing GHG emissions benefits, can result in trade-offs with other sustainability aspects.

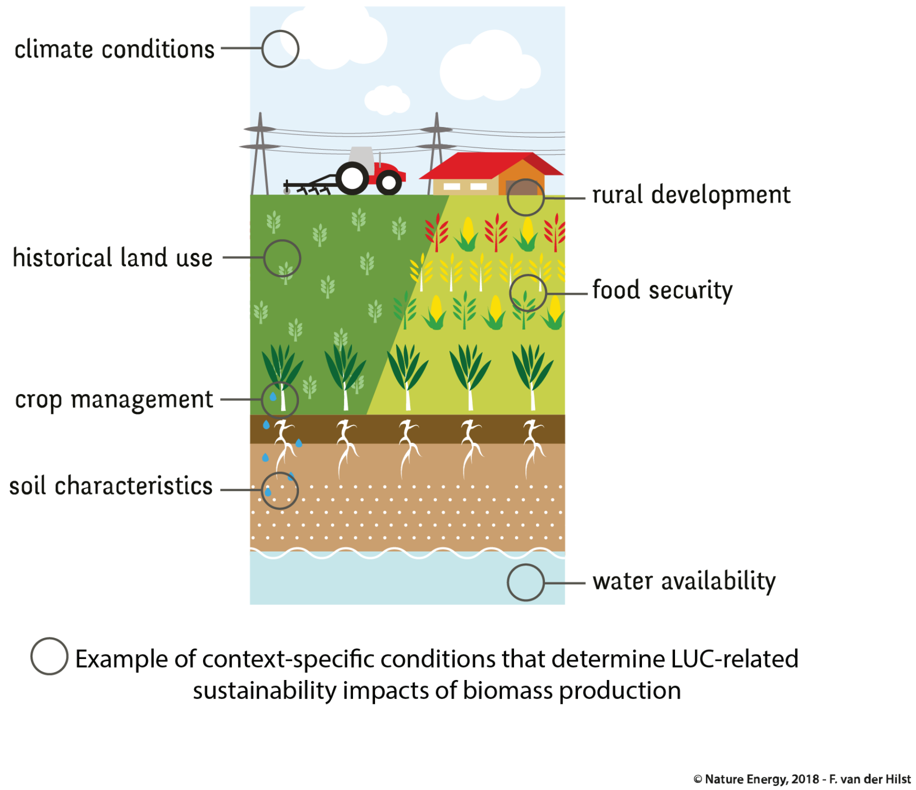



**Figure 1-2. Example of context-specific conditions. Adapted from van der Hilst (2018)[44]**

The potential of bio-based value chains to reduce GHG emissions and provide benefits also relies on the interaction between land availability, land use for biomass production and potential yields, efficient logistics, conversion, and use (supply chain configuration)[48]. Land availability for energy crops production is also limited by current and future land-use dynamics, which are steered by socioeconomic, demographic, and political drivers[49,50]. Thus, context-specific conditions will determine the amount of available land and the potential achievable yield on such land, both aspects are crucial prerequisites to understanding the overall sustainability of bio-based value chains. However, for





several locations, the domestic supply of and potential biomass production is limited[51]. Furthermore, biomass production can be more restricted when considering the sustainability criteria of voluntary schemes or national and international legislation such as REDII. To move biomass from the supply origin to conversion facilities cost-effectively and in compliance with stringent sustainability criteria will require increasingly complex mobilization strategies. This can lead to a broad domain of supply chain configuration options with an extensive range of environmental and economic performances, especially for novel value chains[26,52,53].



Overall, the supply chain configuration and context-specific conditions determine the magnitude and direction of (LUC-related) sustainability impacts and the sustainability performance of bio-based value chains. Therefore, it is highly relevant to understand the effect of supply-chain configuration and context-specific conditions on the sustainability performance of bio-based value chains. Moreover, when considering LUC-related impacts and the effect of context-specific conditions on determining such impacts. This process is vital to achieve bio-based value chains with GHG emissions savings that contribute towards climate change mitigation, provide synergies and avoid trade-offs between sustainability aspects.

## 1.3 GAPS IN KNOWLEDGE

In the last years, increasing relevance has been given to consider LUC-related sustainability impacts of bio-based value chains[27]. However, recent studies reveal that most studies related to bio-based value chain still focus on supply chain GHG emissions without accounting for LUC-related GHG emissions[54]. For example, only a few studies related to ethanol produced from lignocellulosic biomass such as perennial grasses and Short Rotation Coppice (SRC) account for LUC-related GHG emissions and other impacts[54,55]. In addition, most of these studies are carried under an aggregated level and fail to include the effect of context-specific conditions that determine essential spatially explicit parameters such as yields and carbon stocks of the previous and the following land uses.

Generally, the conversion facilities (e.g., bio-refineries) are located where biomass is available[56]. Therefore, most studies on GHG emissions supply chain performance and costs assume that the input biomass sources are domestic, readily available, close to the conversion site, and in sufficient quantities[57]. However, many future conversion sites will be located strategically to supply several commodities (e.g., bio-based materials, advanced biofuels, electricity, and heat) in multiple downstream markets that are well connected but often remote from biomass supply sources[14,56]. For example, when dedicated energy crops would be cultivated on marginal land. Therefore, for several novel bio-based value chains, the supply chain design can involve many logistics configurations (e.g., long-distance intermodal transport of biomass), feedstock types, biomass availability, end uses, and supply/demand locations. This can lead to a wide range of GHG emission performance and costs. Thus, it is increasingly relevant to assess different supply chain configuration options for novel bio-based value chains to maximize GHG emission savings, costs and adhere to sustainability criteria.

The GHG emissions performance of bio-based value chains is closely related to land availability, sustainable biomass production and supply chain configuration[58]. Several studies have quantified land availability and determined the current and future supply potential of lignocellulosic energy crop production under different sustainability criteria[34,59–62], and few studies have coupled those projections of energy crop production with bio-based value chains (e.i., advanced biofuels) to assess the overall GHG emission performance[63–65]. However, most of these studies have a limited time frame and fail to integrate updated sustainability criteria such as the one present in REDII, which are vital for current and future GHG emissions reduction strategy design. For example, REDII encourages land use that avoids displacement effects (e.g. marginal or degraded lands) and excludes land previously used for food, feed or fibre. In addition, most of these studies fall short of providing a high-resolution assessment that considers the spatial and temporal effects from context-specific conditions.

Achieving GHG emission reduction is the main objective of many bio-based value chains and thus has been the primary focus in sustainability assessments[66]. Nevertheless, in the last decades, assessing other (land-related) sustainability environmental and socioeconomic impacts has become increasingly relevant. For example, In Brazil, different sustainability indicators for sugarcane ethanol are assessed individually for biodiversity[67,68], water[69,70], soil[71–74] and carbon stock changes[75–81] under different criteria. Similarly, it occurs for novel bio-based value chains. This can be seen across countries and indicators such as water quality and use for SRC based bio-based value chains [82–85], biodiversity impacts of perennial grasses related bio-based value chains [43,86,87] and socioeconomic impacts related to bio-based value chains deployment (e.g., jobs creation, food security, and financial aspects such as costs)[88–91]. Most of these studies and the majority of the literature tend to focus on a single sustainability impact indicator[39]. Assessing bio-based value chains from an individual indicator can provide a distorted impression of the overall sustainability of bio-based value chains. Moreover, it can fail to recognize the likely synergies and trade-offs between sustainability aspects.

Identifying synergies and trade-offs between sustainability aspects of bio-based value chains are paramount to develop holistic strategies for sustainable land use





and sustainable development. Despite the relevance of this topic, the literature on it is limited[31]. Only a few studies have quantified synergies and trade-offs between sustainability aspects of bio-based value chains[31,92,93]. In addition, they lack to consider the context-specific conditions and thereby overlook valuable opportunities to understand the circumstances that shape the relations between sustainability aspects of bio-based value chains.



## 1.4 THESIS OBJECTIVE, RESEARCH QUESTION AND OUTLINE

Based on the gaps of knowledge identified in existing literature, the main aim of this thesis is to determine the sustainability performance of existing and novel bio-based value chains under context-specific conditions. In this thesis, novel bio-based value chains refer to bio-based supply chains in which non-conventional feedstock types such as perennial grasses, SRC, and Invasive Alien Plants (IAP) are utilized for electricity, advanced fuels or materials. In this thesis, conventional bio-based value chain refers to bio-based supply chains in which sugar feedstock types are used for advanced fuels or materials. Bio-based value chains can include all supply chain steps from biomass production, logistics, conversion and end-use. However, a strong focus is given to LUC-related processes from biomass production and use. The sustainability performance includes both environmental and socioeconomic impacts. This thesis considers the key context-specific conditions that affect the sustainability of bio-based value chains: previous land use, agricultural management practices, location-specific biophysical characteristics, and social and economic aspects.

The following research questions are addressed to meet the overall aim:

I. How can environmental and socioeconomic impacts of feedstock production and the rest of the supply chain for bio-based value chains be assessed?

II. What are the environmental and socioeconomic impacts of bio-based value chains and how do these impacts influence sustainable biomass potentials?

III. What are the synergies and trade-offs between environmental and socioeconomic impacts of bio-based value chains?

To answer these questions, several studies on the sustainability performance of selected bio-based value chains under different context-specific conditions are assessed (see Table 1-1). In chapter 2, a spatially explicit approach is applied to assess the spatial variation in LUC-related environmental impacts of sugarcane expansion and the trade-offs between them. In chapter 3, the effect of supply chain configurations on the overall GHG performance of bio-based value chains is assessed with a particular focus on long-distance transport of internationally sourced biomass. In chapter 4, the biomass potentials and GHG balance of advanced biofuels from energy crops produced on marginal lands are assessed, taking into account land-related carbon stock changes and supply chain emissions. In chapter 5, the sustainability performance of bio-based value chains is assessed, including both environmental and socioeconomic impacts related to LUC. In chapter 6, the synergies and trade-offs between sustainability aspects of using land for energy crop production are assessed.

**Table 1-1. Overview of the chapters of this thesis and the research questions that are addressed in them**

| Chapter | Topic | Research question | | |
|---|---|---|---|---|
| | | I | II | III |
| 2 | Spatially explicit assessment of environmental impacts of sugarcane expansion | X | X | X |
| 3 | GHG emission performance of biomass supply chains of multi-output biorefineries | X | X | |
| 4 | Biomass potentials and GHG emission performance of advanced biofuels | X | X | |
| 5 | Environmental and socio-economic impacts of using invasive alien plants for bioenergy purposes | X | X | X |
| 6 | Land use for bioenergy: synergies and trade-offs between Sustainable Development Goals | | | X |

**Chapter 2** assesses the LUC-related environmental impacts of sugarcane expansion in Sao Paulo state, Brazil. An integrated spatially and temporal explicit approach is applied to quantify the spatial and temporal variation in environmental impacts of sugar cane expansion between 2004 and 2015. GHG emissions, biodiversity, soil erosion, and water quantity are assessed and integrated into a single environmental performance index to determine synergies and trade-offs between impacts.

**Chapter 3** explores the GHG performance of bio-based value chains of the simultaneous production of lactide and ethanol in a biorefinery located in the Netherlands. Special attention is given to the variation in GHG emissions of different supply-chain configurations with internationally-sourced lignocellulosic biomass (pine, foresty residues and bagasse) from the USA, the Baltic States, and Brazil. The results are compared to a biorefinery that uses locally cultivated sugar beets. The GHG emissions from the supply-chain configurations for biofuels are compared with the minimum GHG saving requirements in the REDII. For bio- bio-based materials, supply chain GHG emissions are compared with GHG missions of three relevant fossil-based counterparts.





**Chapter 4** spatially explicitly quantifies marginal land availability in the EU, the biomass production potentials for eight lignocellulosic crop types, and the GHG performance of advanced biofuel supply chains. Available land is mapped based on land marginality and the REDII land-related sustainability criteria. Biomass potentials are assessed with a water-use-to-biomass-production approach while considering the available land, context-specific conditions, and crop-specific characteristics. The GHG balance of advanced biofuels from energy crops produced on marginal lands is assessed considering both land-related carbon stock changes and supply chain emissions.



**Chapter 5** assesses the environmental and socioeconomic impacts of using IAPs for electricity generation in South Africa. It also includes a scenario in which pellets are exported and used for electricity production in the Netherlands. Besides accounting for supply chain GHG emissions, a strong focus is given to LUC-related carbon stock changes under different plausible land-use transition scenarios. In addition, the impact on water availability from IAPs removal is considered. The socioeconomic impacts that are assessed are supply chain costs and job creation.

**Chapter 6** presents a state-of-the-art review on synergies and trade-offs between the impacts of using land for dedicated energy crops on the sustainable development goals. A pairwise comparison between GHG emission reduction (SDG 13) and other SDGs is used to identify the main synergies and trade-offs. An additional approach focusing on environmental-related SDGs is also applied to identify and classify the context-specific conditions (feedstock, previous land use, climate, soils and management) in which the synergies and trade-offs arise.

# 2

# Spatial Variation in Environmental Impacts of Sugarcane Expansion in Brazil

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## ABSTRACT

In the past decades, sugarcane production in Brazil has expanded rapidly to meet increasing ethanol demand. The large majority of this expansion occurred in Sao Paulo state. We used an integrated approach considering location-specific biophysical characteristics to determine the environmental impacts of sugarcane expansion and their spatial variation in Sao Paulo state (2004–2015). The included environmental impacts are greenhouse gas (GHG) emissions, biodiversity, soil erosion, and water quantity. All impacts were integrated into a single environmental performance index to determine trade-offs between impacts. Our results show a strong spatial variation in environmental impacts and trade-offs between them. The magnitude and direction of these impacts are mostly driven by the type of land use change and by the heterogeneity of the biophysical conditions. Areas where expansion of sugar cane has resulted in mostly negative environmental impacts are located in the center and east of the state (related to the change of shrublands, eucalyptus, and forest), while areas where sugar cane expansion has resulted in positive impacts are located in the center-west and north (related to the change of annual crops). Identifying areas with mainly positive and negative impacts enables the development of strategies to mitigate negative effects and enhance positive ones for future sugarcane expansion.





## 2.1 INTRODUCTION

The use of biomass for energy purposes is recognized as a key pillar for the reduction of greenhouse gas emissions (GHG) and meeting worldwide climate change mitigation targets [7,94,95]. In the past years, global biofuel production has increased from 37.5 thousand tons of oil equivalents (ktoe) in 2007 to 84.1 ktoe in 2017, and this trend is expected to continue[94]. Ethanol accounts for the largest share of biofuel production, with Brazil positioned as the world's second largest producer after the USA[96,97]. Brazil produces ethanol primarily from sugarcane and has developed an efficient model to produce sugar and ethanol in an integrated manner[98]. This development has led to an increase in sugarcane production and triggered more than 5.2 million hectares of land to be converted at the country level between 2000 and 2018[99].



The sustainability of the Brazilian ethanol sector has been the object of political and societal debate. Despite the potential social, economic, and environmental benefits[97] associated with biofuel production, there are also major concerns about the sector's sustainability performance[100]. Many of the concerns are related to the environmental impacts of land use change (LUC) directly or indirectly caused by sugarcane expansion, and include, e.g., deforestation, habitat loss, soil erosion, GHG emissions, and impacts on water availability and quality[100–102]. Therefore, in recent years, major attention has been given to monitor and assess the impacts of sugarcane expansion on biodiversity, soil, water, and GHG emissions [100,101,103].

Several studies have assessed the GHG balance of land conversion to sugarcane plantations. Some studies found a net decrease in soil carbon stocks when land, such as forest and pasture, is converted to sugarcane[75–79]. However, sugarcane expansion can also result in a net increase of soil carbon stocks, e.g., when sugarcane expands in (former) cropland area[80,81]. Additionally, for biomass carbon stocks, both losses and accumulation have been reported depending on the prior land uses[76,104,105].

LUC is recognized as one of the main causes of biodiversity loss[38]. In Brazil, habitat loss[106], ecosystem fragmentation[107], and deterioration of ecosystem services[108] are some of the main biodiversity impacts of sugarcane expansion. Sugarcane expansion has reduced the extent of important ecosystems, such as the Brazilian Atlantic forest [109] and the Cerrado[107]. Considering that Brazilian sugarcane plantations are generally homogeneous and with low species abundance[110], expansion of sugarcane plantations can result in biodiversity loss[67]. However, sugarcane expansion on degraded pasturelands has shown no negative impacts on biodiversity[68].

Soil degradation is identified as one of the major threats for the sustainability of the sugarcane sector[101,111]. The expansion of sugarcane cultivation has been reported to result in soil compaction, increases in soil erosion risk, and land degradation [71–73,75]. However, a slight increase in soil quality was reported when sugarcane expanded in degraded pasturelands[74].

Effects on water availability have been a significant concern for sugarcane[112]. A decline of water availability at the regional level can be prompted by the expansion of sugarcane plantations, particularly in water-stressed areas[69,71]. Sugarcane expansion can also alter the hydrological cycle[101]. For example, the precipitation regime in the Brazilian Cerrado was disrupted as consequence of changes in evapotranspiration rates from sugarcane expansion[70]. There are suggestions that impacts on the water balance from sugar expansion are more acute on a local level than on a regional level[113,114].

So far, the assessments of environmental impacts of sugarcane expansion have focused on single impact categories. In addition, the effect of location-specific biophysical conditions, which steer the magnitude and direction of LUC-related environmental impacts, is not commonly considered. Moreover, although there are likely trade-offs between the various environmental impacts, these have not been assessed up until now. It is expected that sugarcane expansion will increase, driven by ethanol demand. Brazil's annual ethanol production is projected to increase from 36.5 billion liters in 2019[115] to 43-54 billion liters by 2030[116,117]. Therefore, an integrated assessment of sugarcane LUC-related environmental impacts (and trade-offs between impacts) while considering location-specific biophysical conditions from historical sugarcane land use dynamics is necessary to mitigate future negative effects of sugarcane expansion, and enhance positive ones.

The objective of this study was to assess the spatial variation in environmental impacts of sugarcane expansion. Four key environmental impacts (i.e., $CO_2$ emissions, soil erosion, water shortage, and biodiversity) were taken into consideration. In addition, these indicators were integrated into one environmental performance index to identify sugarcane expansion areas that result in mainly positive or negative impacts. This assessment is demonstrated for sugarcane expansion in Sao Paulo state between 2004 and 2015. Sao Paulo state, located in the southeast of Brazil, was selected as a case study since it is responsible for the majority of sugarcane production in the country[99]. Sao Paulo state has experienced an expansion of approximately 1.8 million hectares of sugarcane in the last decade[99], and it is responsible for 64% and 48% of the country sugar and ethanol production, respectively[115]. This study can support the understanding of the direction, magnitude, and trade-offs between environmental impacts of sugarcane expansion, and





thereby contribute to developing and improving sound land use planning policies for sustainable sugarcane expansion in the future.

## 2.2 MATERIALS AND METHODS

The environmental impacts from sugarcane expansion in comparison to the land use/cover prior to conversion were assessed for each year within the time period 2004-2015 and then presented as cumulative results for the entire time period. The assessment was carried out considering the heterogeneity of biophysical conditions at a spatial resolution of 1 $km^2$ and was limited to sugarcane expansion areas. The spatial approach applied in this study was developed within a geographical information system (GIS). LUC-related $CO_2$ emissions were assessed following the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC 2006)[118], given the changes in the above and belowground biomass (AGB and BGB) and soil organic carbon (SOC). Impacts on biodiversity were assessed by estimating the difference in the mean species abundance (MSA) index of each land use/cover category[119]. Impacts on soil erosion were quantified with the revised universal soil loss equation (RUSLE)[120]. Impacts on water quantity were quantified using a water balance approach (difference between evapotranspiration rates and effective precipitation)[121]. These four environmental impacts were integrated by standardizing the results of each impact and combining the standardized scores with equal weights into an environmental performance index.



### 2.2.1 Land use change dynamics

Assessing the annual changes in land use/cover enables identification for each year of the location, the amount, and type of land use/cover that changes to sugarcane. This is a prerequisite to determine the LUC-related environmental impacts. The annual expansion of sugarcane was determined spatially explicitly for each year by identifying the areas that changed from any type of land use/cover in one year to sugarcane cultivation in the subsequent year. Land use/cover data was obtained from TUDelft[122]. The data set distinguishes 10 land use/cover categories: Urban, water, forest, mid vegetation, low vegetation, annual crops, sugarcane, sugarcane under renovation, eucalyptus, and harvested eucalyptus. The accuracy ranges between 70% and 90%, depending on the land use/cover type [122]. "Low vegetation" refers to mostly unmanaged grasslands and rangeland with predominant grass cover while "mid vegetation "refers to dense shrubland and woodland with low crown cover, including dense foliage/mid biomass rangeland but also fruit crops, such as citrus[122]. For the purpose of this study, the categories "low vegetation" and "mid vegetation" were considered as grasslands and shrublands, respectively. The "forest" category corresponds to dense foliage/high biomass species with high crown cover, with native forest being the most occurring case[122]. "Harvested eucalyptus" refers to eucalyptus plantations that are harvested but remain in use as eucalyptus plantations. In this study, “eucalyptus” and “harvested eucalyptus” were combined to a single land use/cover category. Sugarcane under renovation relates to the ratoon cycle of sugarcane and refers to the sugarcane areas that are replanted. “Sugarcane” and “sugarcane under renovation” were also combined into one land use/cover category.

### 2.2.2 LUC-related $CO_2$ emissions

LUC-related $CO_2$ emissions result from carbon stock changes in biomass (above and below ground), dead organic matter, litter, harvested wood products, and soils (SOC)[118]. These carbon stock changes are mainly driven by the conversion from one land use/cover to another and can lead to carbon accumulation or loss. To quantify the LUC-related $CO_2$ emissions from sugarcane expansion, the IPCC 2006 guidelines[118] were applied spatially explicitly. The stock difference method with a combined tier 1 and tier 2 approach was used to assess inter-annual carbon stock changes. Under tier 1, dead organic matter and litter stocks are assumed to be zero, and the carbon stock of harvested wood products is not relevant given the scope of this study. Therefore, only the biomass and SOC pools were considered in this assessment. The tier 2 approach was used only for the biomass carbon pool given the availability of specific biomass data. Equation 2-1 represents the stock difference method. For each year, the difference in carbon stock was calculated for all areas converted to sugarcane. Consistent with IPCC guidelines, an amortization period of 20 years was assumed for carbon pools. Therefore, effects on the carbon pools from LUC were calculated over a 20-year time horizon and are presented in $t\,CO_2\,ha^{-1}year^{-1}$

**Equation 2-1**

$$CO_{2\,LUC} = \frac{C_{t2} - C_{t1}}{T} * \left(\frac{44}{12}\right)$$

where:
$CO_{2\,LUC}$ = LUC-related $CO_2$ emissions from sugarcane expansion, $t\,CO_2\,ha^{-1}year^{-1}$ ;
$C_{t1}$ = Carbon stock in land use prior to conversion, $t\,ha^{-1}$ ;
$C_{t2}$ = Carbon stock in sugarcane land use after conversion, $t\,ha^{-1}$ ;
T = Amortization period, $years$ ;
44/12 = Conversion factor to convert C to $CO_2$.

#### 2.2.2.1 Biomass carbon stocks

To determine the $CO_2$ emissions from changes in the biomass carbon pool, the biomass carbon stock was quantified spatially explicitly for each land use/cover category. The AGB was estimated for sugarcane, grasslands, and shrublands land use/cover categories making use of agroecological suitability maps[123,124] and pan-tropical biomass maps for





forest[125]. BGB was derived as a function of the above ground biomass for every land use/cover category using IPCC climate-zone-dependent root-to-shoot ratios (R)[118]. Vegetation type-specific carbon fraction (CF) coefficients were employed to obtain biomass carbon stocks. For eucalyptus, no agro-ecological suitability or pan-tropical biomass maps were available to quantify biomass spatially explicitly. Therefore, the spatial variation in biomass carbon in eucalyptus was solely based on the IPCC default values for the four climate regions in Sao Palo State (i.e., tropical wet, tropical moist, tropical montane, and temperate). For annual crops, 4.7 t $ha^{-1}$ of carbon were assumed in the biomass carbon stock[118]. The employed parameters are summarized in Table S2-1 from the Supplementary Material.

2

#### 2.2.2.2 Soil organic carbon stocks

Changes in soil organic carbon were assessed by comparing the SOC levels of the land use/cover prior to conversion with the SOC levels of sugarcane after conversion. The IPCC 2006 default values for SOC were assigned to each land use/cover based on the stratification of climate regions and soil types. IPCC soil stock change factors were used to account for the effect of land use, management regime, and organic amendment (see Equation 2-2). The soil stock change factors were assigned considering Brazil’s land use and management conditions. Table S2-2 in the Supplementary Material summarizes the assigned stock change factors for each land use/cover category.

**Equation 2-2**

$$SOC_x = SOC_{ref} * F_{LU,x} * F_{MG,x} * F_{I,x}$$

where:
$SOC_x$ = Soil organic carbon stock for land under land use/cover type x, $t\,C\,ha^{-1}$;
$SOC_{ref}$ = The reference carbon stock, $t\,C\,ha^{-1}$;
$F_{LU}$ = Stock change factor for land use system x, unitless;
$F_{MG}$ = Stock change factor for management regime for land use/cover x, unitless;
$F_I$ = Stock change factor for input of organic matter for land use/cover x, unitless.

### 2.2.3 Mean species abundance

The expansion of agricultural land in natural areas can lead to habitat loss/fragmentation and consequently reduces the biodiversity in a region [126]. This is especially true for homogenous agricultural systems with low species diversity, such as most sugarcane plantations[110]. To estimate the impact of sugarcane expansion on species abundance, the mean species abundance (MSA) index was applied. The MSA index is defined as “*the remaining mean species abundance of original species, relative to their abundance in pristine or primary vegetation, which are assumed to be not disturbed by human* *activities for a prolonged period*"[119]. This index has been applied in previous studies to determine LUC-related impacts on species abundance[127,128]. The MSA index overlooks any species abundance distribution information; instead, it assumes a causal–effect relationship between environmental drivers and biodiversity impacts. Five environmental drivers (LUC, atmospheric nitrogen deposition, infrastructure, fragmentation, and climate change) are generally considered in the MSA index. However, the relative change in the MSA index can also be applied while considering only one environmental driver (LUC), as executed in this study. The MSA index of a specific land use/cover varies from 0 to 1. A land use/cover with the value of 1 refers to a pristine original ecosystem with species abundance not affected by human activities and 0 refers to the opposite[119]. MSA values were assigned to each land use/cover category based on previous studies (see Table S2-3 from the Supplementary Material)[119,129]. The impact on species abundance was assessed spatially explicitly for each year by comparing the MSA value of sugarcane plantations with the MSA value of the land use/cover prior to conversion.

### 2.2.4 Soil erosion

Soil erosion is identified as the main form of soil degradation and it is enhanced by LUC dynamics[130]. Soil degradation induced by soil erosion can limit root growth in sugarcane plantations, which in turn decreases yields[131]. Soil erosion is mainly determined by soil characteristics, terrain, land use, weather conditions, and management practices[132]. In Brazil, soil erosion is attributed to high-intensity rainfall and erosion-prone soils [133]. The revised universal soil loss equation (RUSLE) is the most frequently used and accepted method to estimate soil erosion[134]. Therefore, the RUSLE equation was applied to estimate the change in potential soil loss when land is converted to sugarcane[120].

The RUSLE equation includes 5 factors (Equation 2-3). The rainfall erosivity factor (R) considers the aggressiveness of the rain to provoke erosion[132]. The soil erodibility factor (K) is associated with the soil potential to erode in relation to its physical characteristics[135]. Topography factors are included by slope length (L) and slope gradient (S) on erosion[132]. The cover management factor (C) represents the effect of land use/cover on soil loss, and the support practice factor (P) represents erosion prevention practices[136]. For each year, the potential soil loss from sugarcane cultivation was compared to the potential soil loss of the reference land use/cover prior to conversion. The change in potential soil loss is expressed in $t\ ha^{-1}\ year^{-1}$. The methods to estimate each of the relevant factors from RUSLE are presented in the Supplementary Material (Section 4 soil erosion), including the C factor (see Table S2-4 from the Supplementary Material) and P factor (see Table S2-5 from the Supplementary Material) from each land use/cover.





**Equation 2-3**

$$A = R * K * LS * C * P$$

where:
$A$ = Soil loss for sugarcane, $t\ ha^{-1} year^{-1}$;
R = Rainfall-runoff erosivity factor, $MJ\ mm\ ha^{-1}\ h^{-1}\ year^{-1}$;
K = Soil erodibility factor $t\ ha\ h\ ha^{-1}\ MJ^{-1}\ mm^{-1}$;
$L$ = Slope-length factor, unitless;
$S$ = Slope steepness factor, unitless;
$C$ = Cover management factor, unitless;
$P$ = Conservation support practice factor, unitless.



### 2.2.5 Water shortage

The hydrological cycle is strongly affected by LUC. LUC affects the water balance in an area mainly through changes in evapotranspiration and percolation rates [137]. The amount of water that sugarcane plantations take up and release can lead to major changes in the water balance of a watershed and can potentially lead to local water depletion [69]. The evapotranspiration and percolation rates depend on biophysical conditions, such as soil characteristics, hydro-climatic regime, and plant growth stage [112]. To estimate the effect of sugarcane expansion on the local water balance, the approach from Brouwer and Heibloem was applied (Equation 2-4) [121]. The water shortage (WS) was determined on a monthly basis by comparing each land use/cover category's evapotranspiration rates during the length of the growing season with the effective precipitation over the same period. Then, the WS from sugarcane was compared to the WS of the land use/cover prior to conversion to determine the difference in WS between land uses/covers. Land use/cover evapotranspiration rates were determined by multiplying the reference evapotranspiration ($ET_0$) with land use/cover category-specific and growth stage-specific evapotranspiration coefficients (Kc, see Table S2-6 from the Supplementary Material). The methods to estimate each of the relevant factors from the WS approach are presented in the Supplementary Material (Section 5 water shortage).

**Equation 2-4**

$$WS_x = \sum_{i=1}^{12} ET_{0,i} * Kc_{i,x} - \sum_{i=1}^{12} EP_i$$

where:
$WS$ = Water shortage for land under land use/cover type x, $mm\ year^{-1}$;
$i$ = Month of the year;
$ET_0$ = Monthly reference evapotranspiration in month $i$, $mm\ month^{-1}$;
$Kc$ = Crop coefficient in month $i$ for land under land use/cover type x, unitless;
$EP$ = Effective precipitation in month $i$, $mm\ month^{-1}$.

### 2.2.6 Environmental perfomance index

To assess the trade-offs between LUC-related $CO_2$ emissions, soil erosion, species abundance, and water shortage, the results of each impact were integrated into one environmental performance index. The performance index allows a holistic identification of the areas where sugarcane expansion resulted mainly in negative impacts and areas where it resulted in mainly positive effects. Standardization was applied to homogenize the units and scale from all impacts into a common measure. For each impact result, maximum standardization was applied in which all positive effects from sugarcane expansion were converted into a -1 to 0 scale and all negative effects into a 0 to 1 scale. Positive effects are presented on a negative scale to be consistent with other environmental impact results, given that a positive LUC-related environmental impact is generally displayed by a negative score on each impact scale (e.g., carbon accumulation translates into negative $CO_2$ emissions). To avoid a skewed distribution, the maxima were set on two times the standard deviation from the average. All environmental impacts were assumed to have the same weighting factor, i.e., they were considered equally important. However, decision makers may prioritize certain impact categories over others, and alternative weighting factors could be applied.

## 2.3 RESULTS

### 2.3.1 Land use change dynamics

Between 2004 and 2015, approximately 23.6 thousand km$^2$ of land was converted to sugarcane cultivation in Sao Paulo state (See Figure 2-1). The largest share of replaced land use/cover corresponds to grassland (69%), followed by annual crops (17.5%) and shrubland (12.2%) (see Supplementary Material, Figure S 2-1). Most of the sugarcane expansion at the expense of grasslands occurred in the central and north-western part of the state. The conversion of annual crops is concentrated in the north and center-west part, while the conversion of shrubland occurred mostly in the eastern part. The conversion of forest and eucalyptus areas occurred sporadically, both accounting for less than 1% of the total converted area. The strong difference in sugarcane expansion between the north and the south of the state is associated with location-specific logistical and biophysical conditions. Most of the sugarcane mills are located in the center and northern part of the state. In addition, the south of the state is characterized by a hilly terrain with remnants of the Atlantic forest that pose slope-related natural constrains for sugarcane expansion.







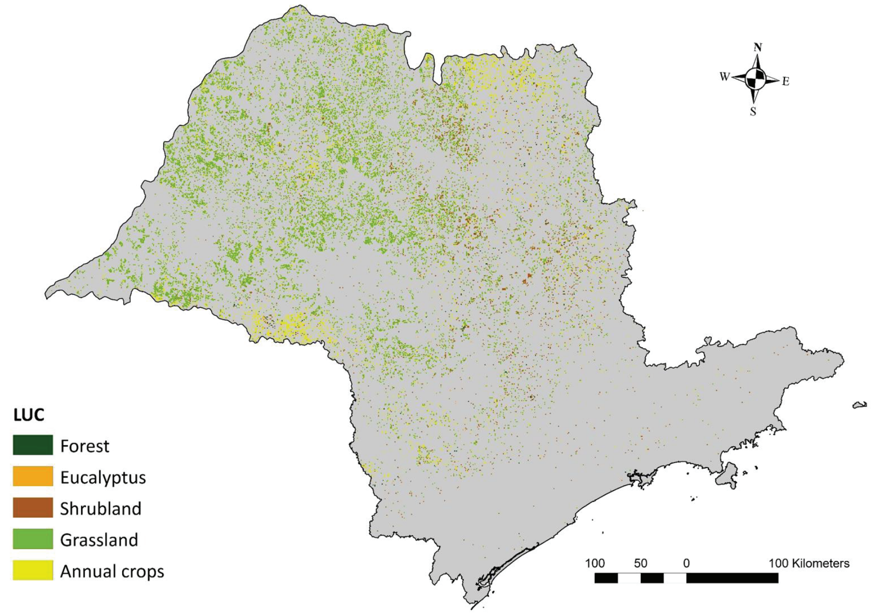


**Figure 2-1. Land use/cover types replaced by sugarcane between 2004 and 2015.**

### 2.3.2 LUC-related $CO_2$ emissions

LUC-related $CO_2$ emissions from sugarcane expansion vary considerably over space in Sao Paulo state; they are calculated to be between -8 t $CO_2$ ha$^{-1}$ year$^{-1}$ and > 6 t $CO_2$ ha$^{-1}$ year$^{-1}$ (see Figure 2-2). The $CO_2$ emissions are mostly driven by changes in the biomass carbon stock and to a lesser extent by changes in SOC. The conversion of grasslands and especially of annual crops to sugarcane in the center and northern part of the state results in carbon accumulation (i.e., negative emissions). When grasslands are converted to sugarcane, the carbon accumulation is mainly determined by the yield difference between both land uses/covers, in which sugarcane delivers higher yields in comparison to grassland. The biophysical conditions in the center-north part of the state are more suitable for sugarcane production than in the center part. Therefore, more carbon accumulation is achieved in the center-north part of the state. The conversion from annual crops to sugarcane results in the highest carbon accumulation. The negative $CO_2$ emissions are caused by the strong differences in biomass carbon, while there are almost no changes in the SOC pool. Generally, the conversion from shrublands to sugarcane results in carbon stock changes varying between 0 and 4 t $CO_2$ ha$^{-1}$ year$^{-1}$. Still, there are some regions (mainly in the east and south) where the loss of shrublands generates $CO_2$ emissions between 4 and 6 t $CO_2$ ha$^{-1}$ year$^{-1}$. These high $CO_2$ emissions are attributed to the biophysical conditions in the vicinity of the Atlantic forest, which are suitable for the development of dense shrublands. The conversion of forest and eucalyptus results in large losses in the biomass carbon stock and to a lesser extent in the SOC stock. The conversion of these two types of land uses to sugarcane shows the highest LUC-related $CO_2$ emissions. The loss of forest from sugarcane expansion generates $CO_2$ emissions between 4 and 10 t $CO_2$ $ha^{-1}$ $year^{-1}$. However, at some locations, the loss of forest can induce emissions of up to 12-14 t $CO_2$ $ha^{-1}$ $year^{-1}$.

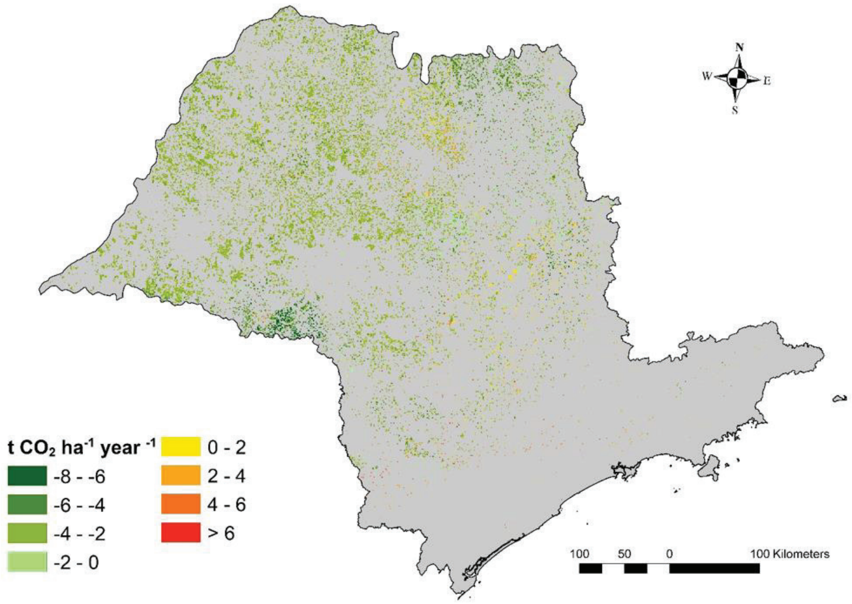


**Figure 2-2. Annual LUC-related CO2 emissions from sugarcane expansion between 2005 and 2015, considering an amortization period of 20 years. Negative values represent carbon accumulation.**

On average, the expansion of sugarcane in Sao Paulo state resulted in -2.8 t $CO_2$ $ha^{-1}$ $year^{-1}$ (see Supplementary Material, Figure S2-2). Most of the carbon uptake occurs in the sugarcane above ground biomass; approximately 70% of this carbon is accumulated in the harvestable section of the plant. However, little carbon is accumulated in below ground biomass or SOC. The annual variation (see Supplementary Material, Figure S2-2) in emissions is mainly caused by the location and type of land use change.

### 2.3.3 Mean species abundance

The change in the MSA index when land is converted to sugarcane in Sao Paulo state varies from -0.2 to 0.7 (see Figure 2-3). The abundance of original species declines for all land use types, except annual crops, when converted to sugarcane. When annual crops are converted to sugarcane, a relative increase in species abundance (-0.2) is reported. This occurs mainly in the north and in the center-west part of Sao Paulo state. The strongest decrease in species abundance (0.7) occurs when forest areas are converted to





sugarcane plantations. Despite that the conversion of forest areas induces the strongest decline in species abundance, the overall impact is small when compared to other land use categories given the relatively little sugarcane expansion that occurs at the expense of forest. The conversion of shrubland to sugarcane also results in a strong decrease (0.45) in species abundance. The conversion of grassland to sugarcane reduces the species abundance by 0.3. The impacts on biodiversity from the loss of grassland areas are concentrated in the center, north, and north-west parts of the state. The lowest decrease in species abundance is found when eucalyptus is converted to sugarcane (0.2). On average, sugarcane expansion resulted in a decrease of species abundance (0.23 $year^{-1}$) driven mostly by the conversion of grasslands to sugarcane (see Supplementary Material, Figure S2-3). However, the decrease in species abundance for the whole state is offset to some extent by the conversion of annual crops to sugarcane.



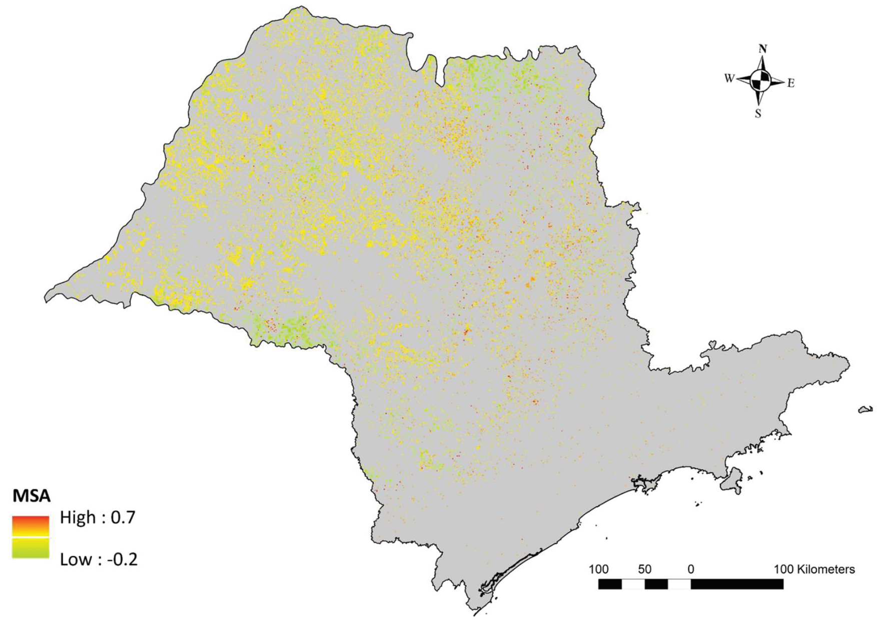


**Figure 2-3. Difference in the mean species abundance between sugarcane and land use/cover prior to conversion for the period of 2005 to 2015. Positive values indicate a loss of mean species abundance.**

### 2.3.4 Soil erosion

As shown in Figure 2-4, the change in soil loss resulting from land conversion to sugarcane varies from < -2 to > 12 t $ha^{-1}$ $year^{-1}$. For all land uses, except for annual crops, conversion to sugarcane resulted in an increase in soil loss. The variation in soil loss is attributed to the differences in the cover effect of the various land uses and to a lesser extent to the spatial heterogeneity in biophysical conditions (mainly terrain conditions). For example, the change from annual crops to sugarcane results in a decrease in soil loss given that annual crops provide less cover against rain impact (mainly at the north and center-west part of Sao Paulo state). A large increase in soil loss occurs when forest and shrubland are converted to sugarcane, as both forest and shrubland provide better soil cover. Given that the soil cover of grasslands is slightly better than the cover effect of sugarcane, soil losses increased to some extent when grasslands were converted to sugarcane. The expansion of sugarcane at the expense of eucalyptus resulted in soil losses between 2 and 6 t ha$^{-1}$ year$^{-1}$. Soil losses increase in areas with a higher rainfall intensity, such as in the western part of the state. In addition, as seen in the south of the state, steep slopes enhance soil loss when land is converted to sugarcane.

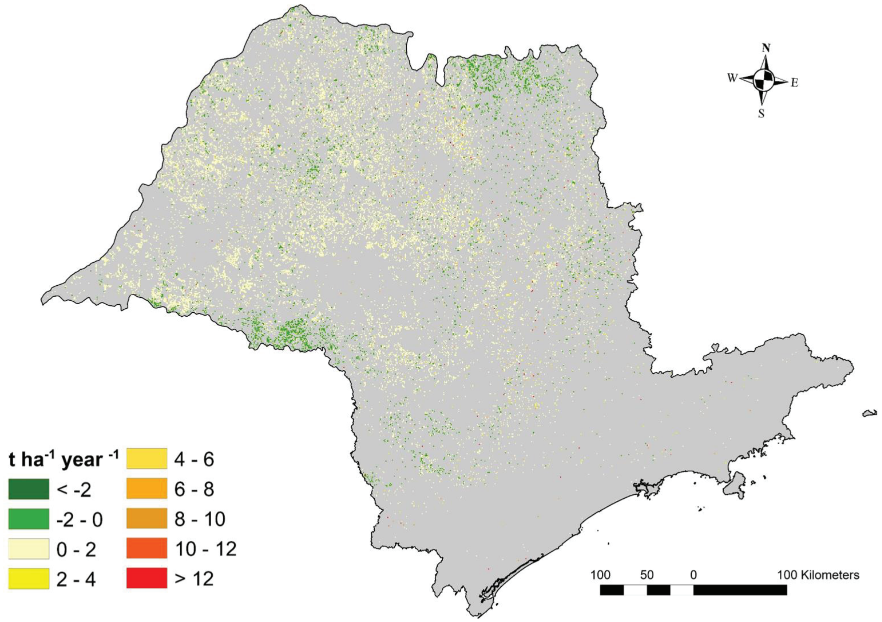


**Figure 2-4. Difference in soil loss between sugarcane and land use/cover prior to conversion in the time period of 2005 to 2015.**

On average, the sugarcane expansion in Sao Paulo state resulted in soil loss of 0.5 t ha$^{-1}$ year$^{-1}$ (see Supplementary Material, Figure S2-4). The variation in annual soil loss is mainly driven by the LUC dynamics, which are dominated by the conversion of grassland and annual crops. Both land use/cover categories provide a similar soil cover as sugarcane; grassland provides a slightly better cover and annual crops a slightly worse cover. Therefore, average soil losses are low. Average soil loss increased for the years 2009–2010, 2010–2011, and 2014–2015, in which the LUC dynamics were less dominated by the conversion of grasslands and the ratio of land use conversion was spread more equally among other land use/cover categories. In the same years, the conversion from





forest to sugarcane resulted in some areas in a soil loss of 12 t ha$^{-1}$ year$^{-1}$. For all years, the conversion of sugarcane resulted in an increase in soil loss except for 2012–2013; in this year, the expansion occurred only at the expense of annual crops.

### 2.3.5 Water shortage

Figure 2-5 shows the spatial variation in the difference in water shortage between sugarcane and the land use/cover prior to conversion. The difference in water shortage is reported to vary between -180 and 300 mm year$^{-1}$. The high evapotranspiration rates of sugarcane in comparison to other land types determine that the change of almost all land use/cover categories, with the exception of forest and eucalyptus, to sugarcane results in an increase in water shortage. When eucalyptus and forest areas are converted to sugarcane, the change in water deficit is negative (i.e., there is less water deficit); forests and eucalyptus areas use more water in comparison to sugarcane as a consequence of their high evapotranspiration rates. There is a strong increase in water deficit when sugarcane displaces annual crops. The growing season of sugarcane lasts 12 months while the growing season of annual crops is much shorter. Therefore, the average annual evapotranspiration of annual crops is lower. The strong difference in water use between both land use/covers is illustrated in the years 2012–2013 (see Supplementary Material, Figure S2-5) when sugarcane expanded only in annual crops and the water shortage reported to be the highest.The smallest increase in water deficits occurs in the center-south part of the state, when grasslands and shrublands are converted to sugarcane. The largest increases in deficits are found in the north when annual crops are converted. The strong variations in changes in the water balance over the state are mainly caused by varying climatic conditions, particularly the precipitation regimes, and by the land use/cover prior to conversion. For example, the north of the state is characterized as a dry area with less precipitation than in any other part of the state. These conditions in conjunction with the conversion of annual crops result, in some areas, in an increase in water shortage of up to 300 mm year$^{-1}$.

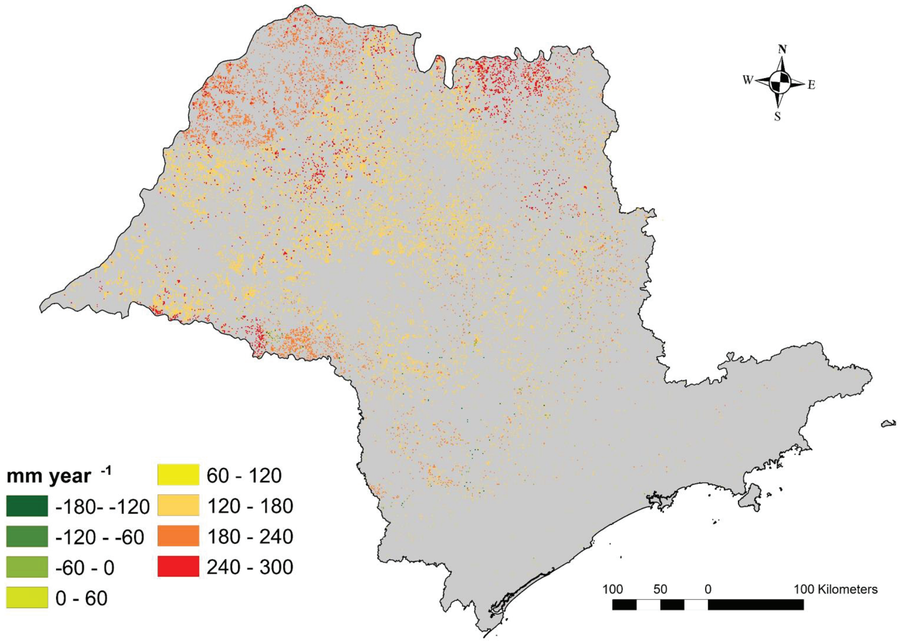


**Figure 2-5. Difference in the annual water shortage between sugarcane and land use/cover prior to conversion in the time period of 2005 to 2015.**

### 2.3.6 Environmental performance index

There is a strong spatial variation in the environmental performance of sugarcane expansion, with the environmental performance index varying between -2.5 and 2.5 at a scale from -4 to 4 (see Figure 2-6 ). Areas characterized by a net positive effect of sugarcane expansion (negative scores) occur mostly in the northeast and central west part of the state. However, there are also areas of poor performance that show medium to strong negative effects of sugarcane expansion, such as the northwest and central east parts of the state. Positive effects of sugarcane expansion generally occur when annual crops are converted to sugarcane. Two distinct clusters located in the center-west and north, related to the conversion of annual crops, show the highest positive effects of sugarcane expansion.





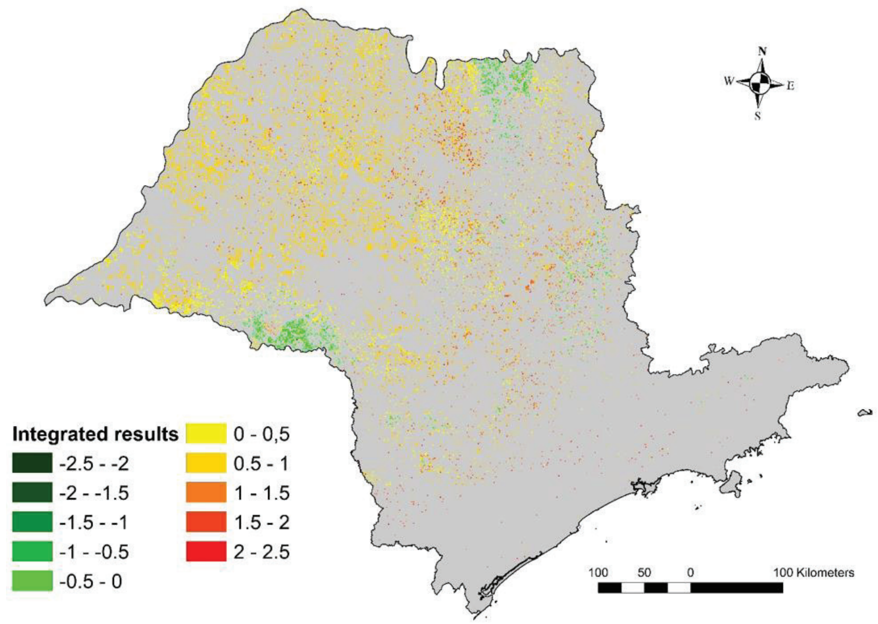


**Figure 2-6. Environmental performance index from sugarcane expansion in the period of 2005 to 2015. Negative values indicate reduced environmental impacts of sugarcane compared to the previous land use/cover.**

The magnitude of net positive or net negative effects is influenced by the trade-offs between the various environmental impacts; these tend to balance each other out at some locations. The expansion of sugarcane into annual crops generally results in carbon accumulation, increases species abundance, and decreases soil loss. However, all these positive effects are balanced out to some extent by an increase in the water deficit. The conversion of grasslands to sugarcane generally results in a large carbon sink; this positive effect is balanced out by the increase in soil loss, increase in water shortage, and a decrease in species abundance. Nevertheless, for some areas, the increase in soil loss and water shortage is minimal. Still, the overall effect is negative. The highest negative overall effects are reported when forest, shrublands, and eucalyptus are converted to sugarcane. The overall negative effect when forest is converted is mainly due to high $CO_2$ emissions, a large loss in mean species abundance, and a considerable increase in soil loss. However, the overall negative effect is counter-balanced to some extent by a decrease in water deficit. For eucalyptus, the negative environmental performance index is dominated by high $CO_2$ emissions. Similar to forests, the high $CO_2$ emissions are balanced out to some extent by the decrease in water deficit. Most of the worst-performing areas (orange and red) are characterized by high woody biomass volumes prior to conversion. These areas generally store more carbon, provide better cover to reduce the risk of soil erosion, and are more suitable for species abundance.

There are no sugarcane expansion areas characterized by only positive impacts as there are always trade-offs between positive and negative impacts for all categories. Moreover, sugarcane expansion in shrublands is characterized by only negative impacts, including an increase in soil loss, increase in water shortage, reduction in species abundance, and increase in LUC-related $CO_2$ emissions.

## 2.4 DISCUSSION

A strong spatial variation in magnitude and direction was found for all LUC-related environmental impacts of sugarcane expansion. These impacts are steered to a large extent by local biophysical conditions and by the type of land use transition. Generally, the direction of the impact is attributed to the land use transition and the magnitude of impacts is largely due to the local biophysical conditions. The observed land use dynamics of sugarcane expansion in our study follow the same trend as observed in other studies [76,102,104], with most of the sugarcane expansion occurring at the expense of grasslands and arable crops.

In general, the conversion of grasslands and annual crops to sugarcane leads to net carbon sequestration while the change from forest, eucalyptus, and shrubland leads to LUC-related $CO_2$ emissions. Similar results are described in other studies[76,79,80]. On average, sugarcane expansion sequestered 2.8 t $CO_2$ $ha^{-1}$ $year^{-1}$ in biomass and soils from 2004 to 2015. Another study reported higher LUC-related carbon sequestration from sugarcane expansion[76]. However, in this study, above or below ground biomass was not considered for the land use prior to conversion. Our assessment of LUC-related $CO_2$ emissions was based on a broad range of input parameters. As a result, the output is sensitive to the assumptions made on yields, land suitability maps, and soil stock change factors. For example, applying a different sugarcane yield could strongly steer the results in carbon sequestration or carbon loss in the biomass pool when land is converted. However, the assumptions were consistently applied while considering Sao Paulo state's heterogeneous biophysical conditions and management practices.

The conventional life cycle GHG emissions from the Brazilian sugarcane ethanol supply chain (including transportation and distribution in the EU) is estimated at 28.6 g $CO_2$-eq $MJ_{ethanol}^{-1}$ [138] of which 17.1 g $CO_2$-eq $MJ_{ethanol}^{-1}$ is from the cultivation phase (including the use of machinery, fertilizer, and pesticide application). The average annual 2.8 t $CO_2$ $ha^{-1}$ $year^{-1}$ savings from sugarcane expansion estimated in this study, which is equivalent to 20.1 g $CO_2$-eq $MJ_{ethanol}^{-1}$, offsets these life cycle emissions to a large extent. When considering the LUC-related $CO_2$ emissions from sugarcane expansion in shrublands and forests, total GHG emissions from sugarcane ethanol increase considerably from 43.1 to 57.5 g $CO_2$-eq $MJ_{ethanol}^{-1}$ for shrublands and from 57.5 to 100.7 g $CO_2$-eq $MJ_{ethanol}^{-1}$ for forests. The total GHG emissions from sugarcane ethanol, including LUC-related $CO_2$ emissions from the conversion of shrubland or forest, fail to comply with the 65% GHG emission reduction requirement of the new European Commission Renewable Energy Directive REDII[15] or the requirements for advanced fuels of the United States Renewable Fuels Standard. Nevertheless, most of the expansion occurs on grassland and annual crops. The expansion of sugarcane on annual crops resulted in only negative LUC-related $CO_2$ emissions, ranging from -49.1 to -0.7 g $CO_2$-eq $MJ_{ethanol}^{-1}$, while in grassland, emissions ranged from -35.8 to 10.7 g $CO_2$-eq $MJ_{ethanol}^{-1}$. The total GHG emissions of sugarcane ethanol produced on land converted from annual crops and grassland not only comply with international regulations, but for several locations, it results in overall negative GHG emissions. The effects of indirect land use change are outside the scope of our study and could increase $CO_2$ emissions.

The mean species abundance declined due to sugarcane expansion except from the conversion of annual crops. In line with[109,139,140], the strongest decrease in biodiversity was found when forest was converted. The MSA indicator omits information, such as species distribution, threatened species, or connectivity. Despite the changes in MSA from sugarcane expansion being assigned based on constant values that neglect the heterogeneity in spatial biophysical conditions, it is a good proxy indicator to estimate the relative impact on species abundance from LUCs. For example, the expansion of sugarcane in natural land uses/covers, such as forest, can potentially reduce species abundance by the loss of (connectivity) species habitats.

The conversion of land to sugarcane increased soil loss, and this is in line with other studies[100,141]. The conversion of any land use/cover, except for annual crops, increases soil losses in comparison to the conditions prior to conversion. Soil loss under sugarcane is slightly less than under annual crops. In Brazil, an average soil loss tolerance threshold of approximately 3 t $ha^{-1}$ year has been established for soils with unfavorable conditions (e.g., shallow non-permeable soils) and 12.5 t $ha^{-1}$ year for soils with favorable ones (e.g., deep well-drained soils)[141]. When considering the annual soil loss from sugarcane plantations, several locations surpassed these soil loss tolerance limits. Sugarcane land is relatively susceptible to erosion processes[142]. In this study, the change in soil loss is driven by the cover effect (C factor) of the different land uses/covers. Despite sugarcane C factors ranging significantly between 0.0012 and 0.58 depending on the growth stage and plant characteristics[143], 0.17 was used as it is generally applied for sugarcane in Brazil[144].

The impact on water shortage was assessed as the difference in the annual water deficit between sugarcane and the land use/cover category prior to conversion. Despite including important biophysical parameters, such as temperature and precipitation, it neglects others, such as soil characteristics and watershed dynamics, that can be relevant to determine direct water impacts from sugarcane expansion. Therefore, it describes particularly whether more or less water is used for sugarcane production than for the land use/cover prior to conversion. In addition, it neglects temporal shortages that can potentially lead to a decrease in crop productivity[145,146] and affect other environmental impacts.

Maximum standardization while considering the same weight for each impact was the selected approach to spatially integrate all environmental impacts into a single environmental performance index. This allowed comprehensive assessment of the environmental impacts from sugarcane expansion and identification of the trade-offs between impacts. However, the integration of results should be interpreted with care. The integration of distinctive scales from different impacts can lead to misinterpretations. A score in one specific indicator can signify a stronger/weaker actual effect in the environment than the same score for a different indicator. In addition, individual scores can balance out or enhance each other in the overall score. Thus, a detrimental effect in one category can be masked by a positive effect in a different category and vice-versa. For example, in the environmental performance index, a high carbon sink can offset a reduction in species abundance. However, in reality, these impacts fail to compensate each other. Therefore, additional expertise is required to estimate the actual environmental impact from a score in each specific indicator and to what extent these impacts interact with each other.

The applied methods allow negative performing areas to be flagged but fall short to some extent in indicating the key biophysical conditions that characterize areas of concern. For example, regardless of a precipitation regime (either wet or dry), a change in use from annual crops to sugarcane will result in an increase in water shortage. Despite the increase in water shortage (negative effect), the biophysical conditions can still be adequate to provide sugarcane plantations with enough water without generating negative impacts in the watershed.

Additional standardization methods can be applied. The integration and weights assigned to the environmental impacts could be established in line with policy objectives and based on engagement with relevant stakeholders. For example, if GHG emissions are prioritized, a higher weight can be assigned to this impact category than to others. However, this should be done with care. The prioritization of one impact can lead to neglecting other impacts and thereby induce higher negative effects in other environmental areas.





The assessment of environmental impacts required a wide range of input data and assumptions that steered the results and could lead to an under- or overestimation of environmental impacts. Some parameters lack a temporal and spatial attribute and were assumed to be constant over space and/or time. This is the case for parameters such as the MSA scores and the cover management (C) component from the soil erosion equation; it is suggested that soil erosion rates can vary considerably when considering the temporal variation of C[147] while the distribution of species abundance can vary in space and time as consequence of different drivers besides LUC[119]. A higher resolution and accuracy of land use/cover data (ranging between 70% and 90% depending on category), suitability maps, and other input data could potentially be more adequate to avoid the loss of details when estimating LUC-related environmental impacts. The land use/cover data set accuracy could result in an under/overestimation of impacts for some locations. However, a land use/cover data set with a higher accuracy degree was not available for the relevant period. Several input parameters, such as precipitation, that were available on a monthly basis were averaged for the studied period; this was done to avoid extreme results.



Retrospectively identifying areas with good and bad environmental performance from sugarcane expansion is a first step to define suitable locations for future expansion and avoid negative impacts. However, in this study, only the environmental impacts related to direct LUC were included. The expansion of sugarcane areas is very likely to result in indirect land use changes when cropland and pastures are converted[106,117,148,149]. The indirect land use changes will result in environmental impacts elsewhere, for example, if pasture area expands outside Sao Paulo state at the expense of forest, it will negatively affect carbon and biodiversity. Therefore, to identify environmentally sound sugarcane expansion strategies, indirect impacts should also be considered in conjunction with schemes that target the reduction of negative and/or enhancement of positive direct LUC-related environmental impacts. In addition, in order to assess all three pillars of sustainability, socioeconomic impacts should also be included in an integrated sustainability assessment.

## 2.5 CONCLUSION

This study provided a detailed assessment of the spatial variation of environmental impacts of sugarcane expansion in Sao Paulo state in the period 2004–2015. The results show where positive and negative LUC-related impacts from sugarcane expansion occurred. The direction and magnitude of LUC-related environmental impacts are highly affected by the type of LUC transitions and the local biophysical conditions. In addition, there are trade-offs between impacts that need to be considered for sustainable development of the sugarcane sector. Ignoring the trade-offs between impacts from sugarcane expansion could result in improving the environmental performance in one dimension but simultaneously worsen the performance in the other dimensions and therefore fail to integrally maximize the environmental performance of the sector. The conversion from annual crops to sugarcane resulted in positive environmental impacts in all dimensions with the exception of water; for all the other land use conversion, there are trade-offs between impacts but the overall effect is negative. The latter suggests that annual crops is the most suitable land for future sugarcane expansion. However, such conversion can result in indirect land use change and related environmental impacts elsewhere and potentially diminish the overall positive effects. In order to assess the sustainability of sugar cane expansion, this assessment highlights the importance of performing integrated environmental impact assessments while considering the heterogeneity in biophysical conditions and LUC dynamics. It is an important step forward in the development of sound land use planning for sustainable sugarcane expansion.

## 2.6 ACKNOWLEDGMENTS

The authors would like to thank TU delft for providing Sao Paulo state land use/cover data, Walter Cervi for providing context about sugarcane cultivation in Brazil and Maria J. Santos for her support with spatial modelling.





## 2.7 SUPPLEMENTARY MATERIAL

### 2.7.1 Biomass carbon stocks

**Table S2-1. Parameters to estimate spatially explicit biomass carbon stock for each land use/cover category.**

| Land use/ cover | Above ground biomass (AGB), range of values t $ha^{-1}$ | Root to shoot ratios (R) | Carbon Fraction (CF) |
|---|---|---|---|
| Forest | 40 - 300. Established spatially explicitly[A] | Tropical – Wet = 0.221[B] | 0.5[B] |
| | | Tropical – Moist = 0.284[B] | |
| | | Tropical – Montane = 0.348 for ≤ AGB 125 t $ha^{-1}$ and 0.283 for > AGB 125 t $ha^{-1}$[B] | |
| | | Temperate = 0.46 for ≤ AGB 125 t $ha^{-1}$ and 0.19 for AGB 125 t $ha^{-1}$[B] | |
| Eucalyptus | Tropical – Wet = 200[B] | Tropical – Wet = 0.221[B] | 0.5[B] |
| | Tropical – Moist = 90[B] | Tropical – Moist = 0.284[B] | |
| | Tropical – Montane = 75[B] | Tropical – Mountain = 0.348[B] | |
| | Temperate = 60[B] | Temperate = 0.464[B] | |
| Shrubland | 26 - 78. Established spatially explicitly[A] | 1.80[C] | 0.5[B] |
| Grassland | 1.6-6.2. Established spatially explicitly[A] | Tropical – Wet & Moist = 1.6[B] | 0.47[B] |
| | | Warm Temperate – Wet = 4[B] | |
| Sugarcane | 15-62. Established spatially explicitly[A] | 0.15[D] | 0.43[D] |

[A] See supplementary material, section: Biomass carbon stocks
B 118
C 150
D 151

I. Forest

The improved pan-tropical map of aboveground woody biomass[125] was used to estimate AGB for forest. With an overlay assessment, AGB values were assigned from the pan-tropical map to each specific location categorized as forest.

II. Grassland

The FAO/IIASA suitability map for low-input level rain-fed alfalfa[123] was used to assess the spatial variation in biomass for grasslands. This approach was selected given the lack of biomass spatially explicit grassland data for Sao Paulo state. With an overlay assessment, alfalfa suitability values were assigned to each location categorized as grassland. The maximum alfalfa suitability value was assumed to be correlated with the IPCC climate zone specific peak AGB coefficients[118]. The AGB in grasslands was assessed for each location from the specific suitability value by considering the link between maximum alfalfa suitability and AGB peak biomass coefficients.

III. Shrubland

A similar process as the one used for grasslands was used to estimate AGB spatially explicit for shrublands. The FAO/IIASA suitability map for low-input level rain-fed alfalfa[123] was also used to assess the spatial variation in biomass for shrublands. Suitability values were assigned to each location categorized as shrubland. Then, it was assumed that the average suitability value was correlated with the average AGB in shrublands for the Brazilian savanna woodland[150]. AGB for shrublands was assessed for each location from the specific suitability value while considering the established link between average suitability and average AGB for shrublands in the Brazilian savanna woodland.

IV. Sugarcane

The FAO/IIASA suitability map for intermediate-input level rain-fed sugarcane was used to determine the spatial variation in sugarcane biomass. Sugarcane suitability values were assigned to each location with sugarcane land use. Then, it was assumed that the average suitability value was correlated with the sugarcane average yield (average sugarcane yield between 2004 and 2015, 80.6 t $ha^{-1}$). Sugarcane yields were estimated from each location-specific suitability value by considering the established link between average suitability and average yield. Sugarcane yields were retrieved from UNICA[99,152]. AGB was assessed considering a 29% dry matter content[151] for sugarcane and a yield to AGB ratio of 1.40[153].





### 2.7.2 Soil organic carbon stocks

**Table S2-2. Soil organic carbon (SOC) stock change factors valid for each land use/cover category, derived from IPCC 2006 guidelines118.**

| Land use/cover categories | Climate region | IPCC relative soil stock change factors | | |
|---|---|---|---|---|
| | | $F_{LU}$ | $F_{MG}$ | $F_I$ |
| **Forest** | All | 1[A] | 1[A] | 1[A] |
| Eucalyptus | Tropical—All | 1.01[B] | 1.10[c] | 1[D] |
| | Temperate—Wet | 0.72[B] | 1.10[c] | 1[D] |
| Shrublands[E] | All | 1[F] | 1[G] | 1[H] |
| Grassland | Tropical—Wet & Moist | 1[F] | 0.97[I] | 1[H] |
| | Tropical—Montane | | 0.96[I] | |
| | Temperate—Wet | | 0.95[I] | |
| Sugarcane | Tropical—Wet & Moist | 0.83[J] | 1.04[K] | 1.11[L] |
| | Tropical—Montane | 0.83[J] | 1.04[k] | 1.11[L] |
| | Temperate—Wet | 0.69[J] | 1.05[k] | 1.08[L] |
| Annual crops | Tropical—Wet & Moist | 0.83[J] | 1.04[M] | 1.11[L] |
| | Tropical—Montane | 0.83[J] | 1.04[M] | 1.11[L] |
| | Temperate—Wet | 0.69[J] | 1.05[M] | 1.08[L] |

[A] Forest is assumed to have the same soil carbon stock as the reference condition. Therefore, all stock change factors are 1
[B] Value for long-term perennial tree crops
[C] Value for management without primary tillage, with only minimal soil disturbance in the seeding zone. Eucalyptus is harvested every 7 years and very limited tillage practices are applied [154]
[D] Assumed for medium input with mineral fertilization
[E] All stock change factors for Shrubland are assigned from IPCC chapter 6 Grasslands given that there are no specific values for Shrubland
[F] Value for permanent grassland
[G] Value for grasslands without significant management improvements
[H] Value for grassland where no additional management practices have been used
[I] Value for overgrazed or moderately degraded grassland. The definition of grassland for this research includes rangelands
[J] Value for long-term cultivated area
[K] Value for primary and/or secondary tillage but with reduced soil disturbance. Sugarcane plantations have reduced till practices due its ratoon cycle and 6-year harvest cycle [155]
[L] Value for high-input crops with significantly larger crop residue input. In addition, sugarcane plantations are characterized by high inputs and mechanized harvesting, which leaves a high amount of residues on the land [156]
[M] Value for primary and/or secondary tillage but with reduced soil disturbance. In Brazil, approximately 50% of annual cropland is no-till and the rest incorporates tillage practices [157]

### 2.7.3 Mean species abundance

**Table S2-3. Mean species abundance index assigned to each land use/cover 119,129.**

| Land use/cover | MSA value |
|---|---|
| Forest | 1[A] |
| Eucalyptus | 0.5[B] |
| Shrubland | 0.75[C] |
| Grassland | 0.6[D] |
| Sugarcane | 0.3[E] |
| Annual crops | 0.1[F] |

[A] Assumed for forest land cover category and primary vegetation
[B] Assumed for secondary forest land cover category
[C] Assumed for mean value between grass or shrubland and agroforestry land cover categories
[D] Assumed for mean value between grass or shrubland, livestock grazing, and man-made pastures land cover categories
[E] Assumed for Low-input agriculture land cover category and perianal bioenergy crop
[F] Assumed for intensive agriculture land cover category

### 2.7.4 Soil erosion

I. Rainfall erosivity

The rainfall erosivity (R) factor is normally assessed by summing for each rainstorm the product of total storm energy and the maximum 30-min intensity [132]. However, this method requires pluviometry data on high spatial and temporal resolution, which is not available for Sao Paulo state. It is common to employ equations that determine rainfall erosivity values based on region-specific precipitation patterns [158]. Equation S 2-1 was applied to determine the rainfall erosivity spatially explicitly for Sao Paulo state. Equation S 2-1 is derived from Neto and Moldenhauer [146], which considered more than 20 years of precipitation data for Sao Paulo state. Monthly precipitation data between 2004 and 2015 were gathered from 23 meteorological stations from the Instituto Nacional de Metereologia (INMET). Monthly erosivity values were calculated for each station and summed over the year. The average erosivity value for the period 2004-2015 was determined for each station and spatially interpolated between stations to obtain average rainfall erosivity maps that cover the extent of Sao Paulo state for the time period 2004-2015.





**Equation S 2-1**

$$R = \sum_{i=1}^{12} 68.730 * \left(\frac{M_i^2}{P}\right)^{0.841}$$

where
R = Rainfall-runoff erosivity factor, $MJ\ mm\ ha^{-1}\ h^{-1}\ year^{-1}$;
*i*= Month of the year;
*M* = Average monthly precipitation in month i, $mm$;
*P* = Average annual precipitation, $mm$.

II. Soil erodibility

To determine the soil erodibility (K) factor, the soil map of Brazil[159] was used to assign the appropriate k value from the soil erodibility database[160] to each soil type within Sao Paulo state, based on soil characteristics.

III. Slope length and slope steepness

The slope length and slope steepness (LS) factor is commonly determined with empirical methods [132]. However, over the last decades, several algorithms have been developed to model this component in a GIS environment[161]. It is suggested that Equation S2-2[162,163] is an adequate alternative to empirical measurements to estimate the sloe length/steepness factor [163]. The flow accumulation and slope gradient were derived from digital elevation models (DEM); the Brazilian DEM was used in this study [164].

**Equation S2-2**

$$LS = \left(\frac{\lambda}{22.13}\right)^{0.6} * \left(\frac{\sin\theta}{0.0896}\right)^{1,3}$$

$$\lambda = Flow\ accumulation * Cell\ size$$

where
*LS* = combined slope length and slope steepness factor, unitless;
*Flow accumulation* = accumulated upslope contributing area for a given cell, unitless;
*Cell size* = size of grid cell, $meter$;
*ϑ* = slope gradient, $degree$.

IV. Cover management

The cover management (C) factor was assigned to each land use/cover category based on a literature review while considering biophysical conditions in Sao Paulo state (Table S2-4). The cover management factors are generally time dependent and vary in accordance to plant growth stages [132]. However, for this study, it was assumed that the cover management factor is constant over time.

**Table S2-4. Cover management (C) factor for the relevant land use/cover categories.**

| Land use/cover | C value |
|---|---|
| Forest | 0.03[A] |
| Eucalyptus | 0.12[B] |
| Shrubland | 0.0219[C] |
| Grassland | 0.16[D] |
| Sugarcane | 0.17[E] |
| Annual crops | 0.172[F] |

[A] Assumed for "native forest" category [165]
[B] Assumed for "eucalyptus plantations" category [165]
[C] Assumed for "transitional woodland-shrub" category [166]; this category was assumed from European conditions given the lack of data from Brazil
[D] Assumed for "pastures" category [167]
[E] Assumed for "sugarcane plantations" [143]
[F] Assumed for "annual crops", corn and soy bean fields [167]

V. Conservation support practice

The conservation support practice (P) factor for erosion control was assigned based on slope (%) thresholds [132] (Table S2-5). The slope % was calculated from the Brazilian DEM [164].

**Table S2-5. Conservation support practice factor according to slope thresholds 132.**

| Slope Threshold (%) | *p*-Value |
|---|---|
| 1–2 | 0.6 |
| 3–5 | 0.5 |
| 6–8 | 0.5 |
| 9–12 | 0.6 |
| 13–16 | 0.7 |
| 16–20 | 0.8 |
| 21–25 | 0.9 |

### 2.7.5 Water shortage

Given data availability, $ET_0$ was assessed with the Turc approach [168], described in Equation S2-3. Kc values were assigned to each month based on land use/cover growing cycles. For sugarcane, it was assumed that the sugarcane growth cycle is completed in 12 months and is harvested in April [169]. For annual cropland, an annual rotation cropping system (common in Brazil), including soy beans and wheat, was considered [170]. For eucalyptus, grasslands, shrublands, and forest, kc values were assigned based on Sao Paulo state characteristics [171]. The kc values for each land use/cover in line with specific land uses'/covers' growing stages can be found in Table S2-6.





Effective precipitation (Equation S2-4) is the share of precipitation that is stored in the soil and is available for the crop, and is derived from actual precipitation [121]. Precipitation, insolation, temperature, and humidity data from 2004 to 2015 were retrieved from 23 meteorological stations in Sao Paulo state from INMET [172] to determine $ET_0$ and effective precipitation. The selection of the stations was carried out based on the location of each station and the completeness of data. Evapotranspiration (ET) and effective precipitation (EP) were calculated on a monthly basis for each station and summed over the year for each year. The average evapotranspiration and effective precipitation for the time period 2004–2015 were calculated for each station and spatially interpolated to obtain coverage for the whole Sao Paulo state. The WS shortage difference between sugarcane and land use/cover prior to conversion is expressed in mm/year.



**Equation S2-3**

$$ET_0 = a_T\, 0.013 \frac{T°_{mean}}{T°_{mean} + 15}\, 23.8856\, R_s + 50 * 0.408$$

where:
$ET_0$ = Reference evapotranspiration, $mm\ day^{-1}$ ;
$a_T$ = 1 for RH ≥ 50, where RH is mean daily relative humidity, %. When RH < 50, then $a_T = 1 + 50 - RH/70$;
$T$ = mean daily air temperature, °C;
$R_s$ = Solar radiation, $MJ\ m^{2-1} day^{-1}$;
$0.408$ = Radiation conversion factor from $MJ\ m^{2-1} day^{-1}$ to $mm\ day^{-1}$ .

**Equation S2-4**

$$EP_i = P_i * \left(\frac{125 - 0.2 * P_i}{125}\right) for\ P_i < 250\ mm$$

or

$$EP_i = 125 + 0.1 * P_i\ for\ P_i > 250\ mm$$

where:
*EP* = Effective precipitation in month *i*, $mm\ month^{-1}$ ;
*P*= Precipitation in month i, $mm\ month^{-1}$ ;
*i*= *Month* of the year.

**Table S2-6. Evapotranspiration coefficients (Kc).**

| Land Use/ Cover | Evapotranspiration Coefficients (Kc) | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Jan. | Feb. | Mar. | Apr. | May | Jun. | Jul. | Aug. | Sep. | Oct. | Nov. | Dec. |
| Forest[A] | 1.15[B] | 1.11 | 1.07 | 1.11[C] | 1.15 | 1.25 | 1.21[D] | 1.17 | 1.16 | 1.15[B] | 1.15[B] | 1.15[B] |
| Eucalyptus[E] | 1.35[B] | 1.26 | 1.18 | 1.24[C] | 1.30 | 1.56 | 1.51[D] | 1.42 | 1.29 | 1.35[B] | 1.35[B] | 1.35[B] |
| Shrubland[F] | 0.75[B] | 0.62 | 0.63 | 0.64[C] | 0.65 | 0.94 | 0.88[D] | 0.81 | 0.61 | 0.75[B] | 0.75[B] | 0.75[B] |
| Grassland[F] | 0.75[B] | 0.62 | 0.63 | 0.64[C] | 0.65 | 0.94 | 0.88[D] | 0.81 | 0.61 | 0.75[B] | 0.75[B] | 0.75[B] |
| Sugarcane[G] | 0.75 | 0.75 | 0.75 | 0.40 | 1.25 | 1.25 | 1.25 | 1.25 | 1.25 | 1.25 | 1.25 | 1.25 |
| Annualcrops[H] | 1.15 | 1.15 | 0.5 | 0.3 | 1.2 | 1.2 | 1.2 | 0.6 | - | - | - | 0.4 |

[A] Assumed for forests in Sao Paulo state [171]
[B] Average from the months with measurement
[C] Average between March and May
[D] Average between June and Augustus
[E] Assumed for Eucalyptus plantations in Sao Paulo state [171]
[F] Assumed for pastures in Sao Paulo state [171]
[G] 173
[H] 173

## 2.7.6 Land use change dynamics

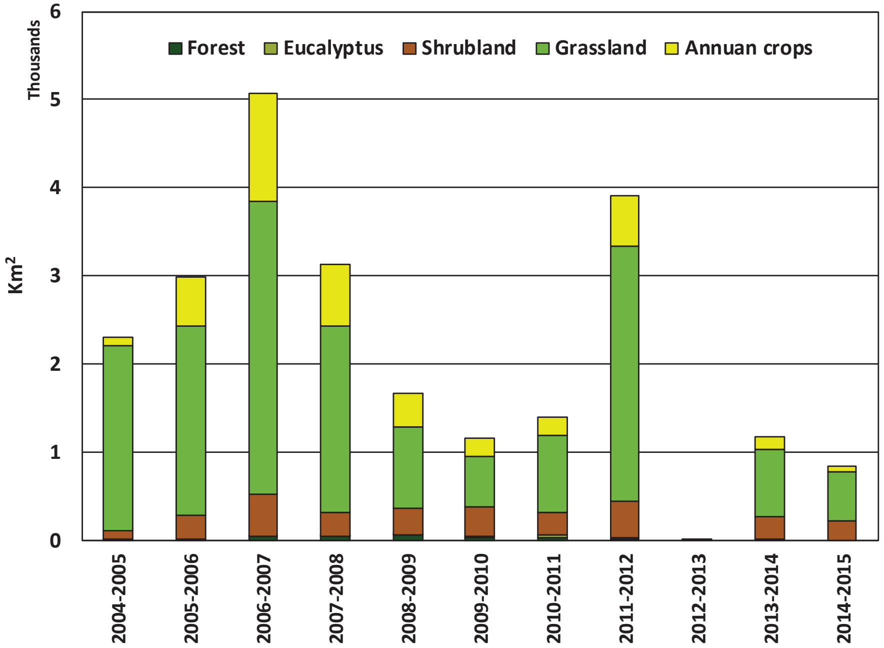


**Figure S 2-1. Share of land that was converted to sugarcane from 2004 to 2015.**





## 2.7.7 LUC-related $CO_2$ emissions

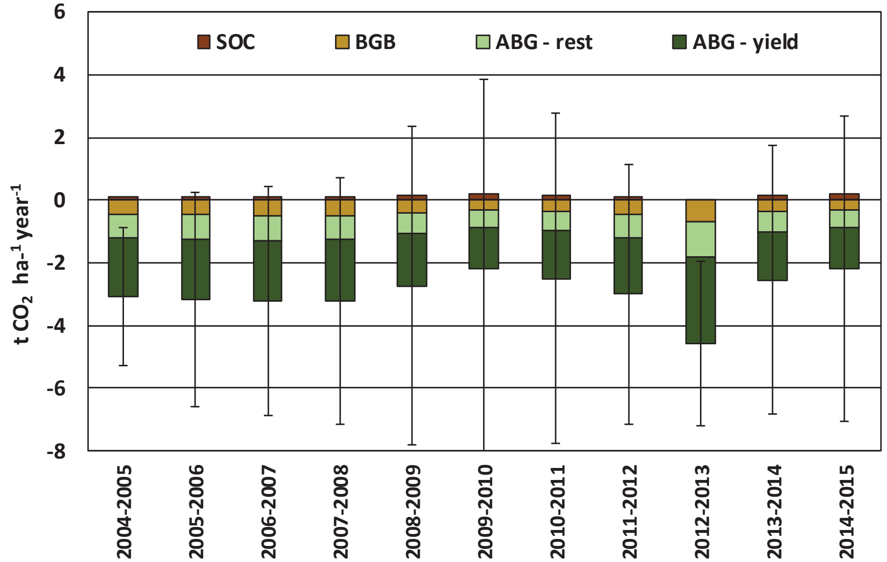


**Figure S2-2. Average LUC-related CO2 emissions with 2 standard deviations from the expansion of sugarcane from 2004 to 2015.**

## 2.7.8 Mean species abundance

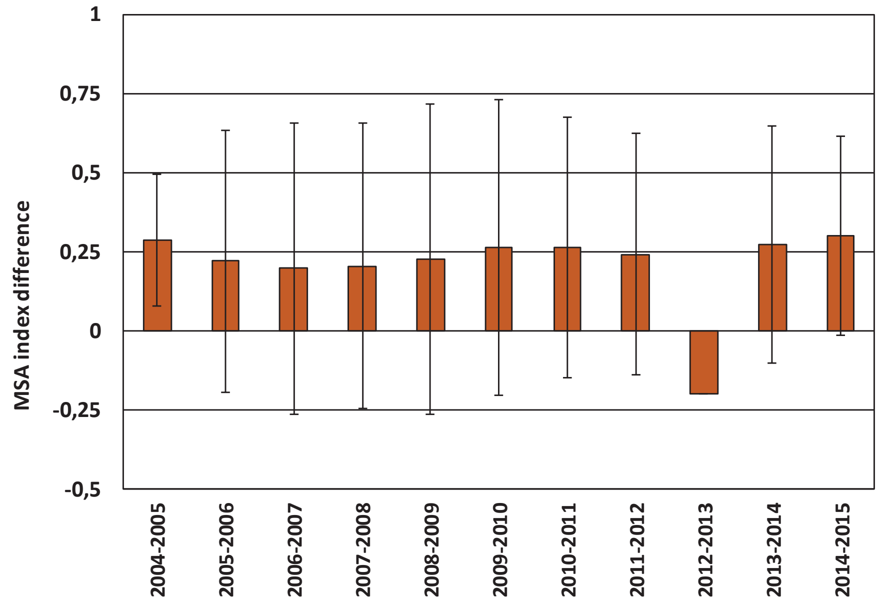


**Figure S2-3. Average annual difference in mean species abundance with 2 standard deviations from the expansion of sugarcane from 2004 to 2015.**

### 2.7.9 Soil erosion

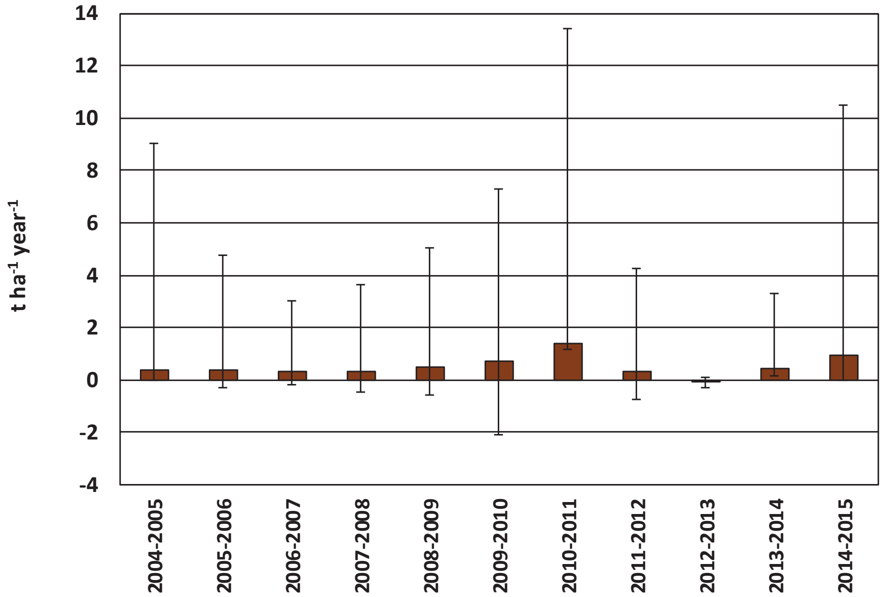


**Figure S2-4. Average annual difference in soil loss with 2 standard deviations from the expansion of sugarcane from 2004 to 2015.**





### 2.7.10 Water shortage

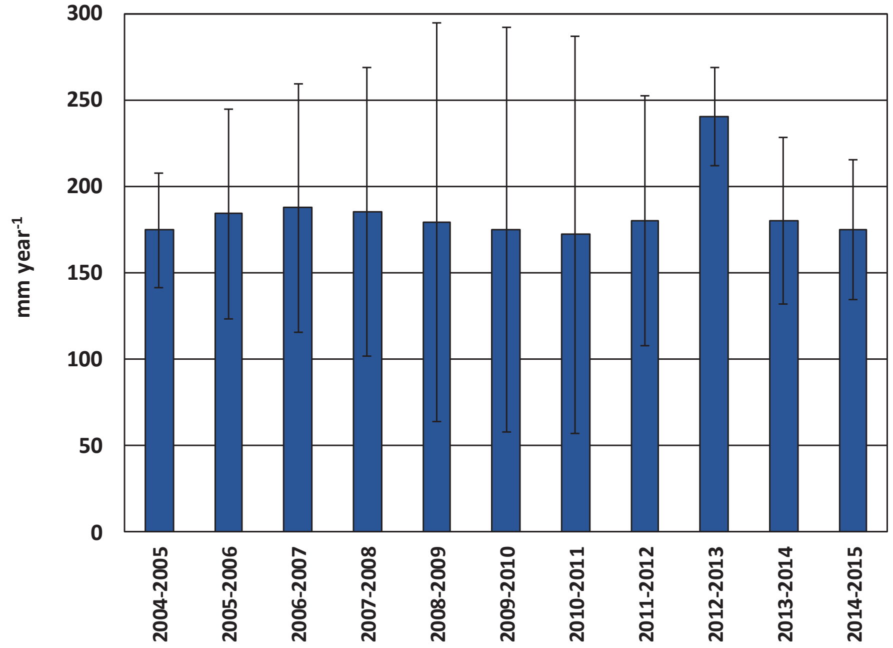


**Figure S2-5. Average annual water shortage with 2 standard deviations from the expansion of sugarcane from 2004 to 2015.**

# 3

# **A carbon footprint assessment of multi-output biorefineries with international biomass supply:** a case study for the Netherlands

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## ABSTRACT

The efficient use of lignocellulosic biomass for the production of advanced fuels and bio-based materials has become increasingly relevant. In the EU, regulatory developments are stimulating the mobilization and production of bio-based chemicals / materials and biofuels from lignocellulosic biomass. We used an attributional life-cycle assessment approach based on region-specific characteristics to determine the greenhouse gas emissions (GHG) performance of different supply-chain configurations with internationally sourced lignocellulosic biomass (stem wood, forest residues, sawmill residues, and sugarcane bagasse) from the USA, the Baltic States (BS), and Brazil (BR) for the simultaneous production of lactide and ethanol in a biorefinery located in the Netherlands (NL). The results are compared with a biorefinery that uses locally cultivated sugar beets. We also compared GHG emissions savings from the supply-chain configurations with the minimum GHG saving requirements in the revised Renewable Energy Directive (RED II) and relevant fossil-based counterparts for bio-based materials. The GHG emissions 'from cradle to factory gate' vary between 692 g $CO_{2eq}/kg_{lactide}$ (sawmill residues pellets from the BS) and 1002 g $CO_{2eq}/kg_{lactide}$ (sawmill chips from the USA) for lactide and between 15 g $CO_{2eq}/MJ_{ethanol}$ (sawmill residues pellets from the BS) and 28 g $CO_{2eq}/MJ_{ethanol}$ (bagasse pellets from BR) for ethanol. Upstream GHG emissions from the conversion routes have a relatively small impact compared with biomass conversion to lactide and ethanol. The use of woody biomass yields better GHG emissions performance for the conversion system than sugarcane bagasse or sugar beets as result of the higher lignin content that is used to generate electricity and heat internally for the system. Only the sugar beet from the NL production route is able to comply with RED II GHG savings criteria (65% by 2021). The GHG savings from polylactide acid (a derivate of lactic acid) are high and vary depending on choice of fossil-based counterpart, with the highest savings reported when compared to polystyrene (PS). These high savings are mostly attributed to the negative emission credit from the embedded carbon in the materials. Several improvement options along the conversion routes were explored. Efficient feedstock supply chains (including pelletization and large ocean vessels) also allow for long-distance transportation of biomass and conversion in large-scale biorefineries close to demand centers with similar GHG performance to biorefineries with a local biomass supply.





## 3.1 INTRODUCTION

It is crucial to keep the rise in global temperature to well below 2 °C, as specified by the 2015 United Nations Climate Change Conference (COP21) in 2015 (the Paris Agreement), to prevent dangerous impacts from climate change[3]. Meeting this target requires a significant reduction in anthropogenic greenhouse gas (GHG) emissions. These climate-change targets entail decreasing total GHG emissions between 50% and 55% for 2030 and between 80% and 95% for 2050 in developed countries[174]. The development of a bio-based economy is recognized as crucial for meeting such challenging targets. This will require structural changes across all sectors of the economy[175,176] in particular in countries with energy-intensive industry, such as the Netherlands[176]. The role of biomass for energy and materials is more essential in sectors that are difficult to decarbonize or have few alternatives to biomass. For example, the substitution from fossil-based products in materials such as plastics and chemicals will likely be based mostly on biomass-derived products[10]. These new conditions can shift the use of solid biomass feedstock from power generation and heat to one that targets bio-based materials and advanced fuel generation[177]. Multi-output biorefineries have emerged as key facilitators for the successful development of the bio-based economy[52].



Facilitating the development of sustainable biorefineries is one of the key actions in the EU strategy towards the development of a bio-based economy[14]. Lignocellulosic biorefineries are of particular importance for this. The use of lignocellulosic biomass has a crucial advantage over other biomass types by avoiding direct competition with food crops, which can have a detrimental effect on the dynamics of food supply[178]. Regulatory developments are also accelerating the mobilization of such lignocellulosic-based biorefining facilities; for instance, the EU has proposed ambitious targets to reduce the production of biofuels from food crops to 7% by 2030 and has invested close to €80 billion of research funding to consolidate a lignocellulosic biorefining sector[15,16]. As a result, biorefineries that process lignocellulosic biomass into bio-based materials, advanced biofuels, electricity, and heat are becoming increasingly relevant in different applications for liquid fuels, bio-based chemicals, and bio-based materials[12]. The multi-output / parallel production (co-production) characteristics of biorefineries result in biomass supply-chain optimization by reducing, recovering, or re-using waste, residues, and energy.

The potential benefits from biorefineries will depend to a large extent on the efficiency of the supply chain[53,179]. Most studies investigating the supply-chain design of biorefineries or bioenergy supply chains aim at optimization from an economic perspective, i.e., a minimum cost requirements approach[180–186]. Nevertheless, this approach can diminish potential environmental benefits from biorefineries[187]. Other studies assess supply chains efficiency from a GHG perspective, with recommendations generally targeting logistics[188–196]. There are also concerns regarding the large-range (and sometimes poor) performance in GHG emissions for bioenergy and bio-based materials supply chains[52]. The majority of these studies focus particularly on supply chains with local-sourced biomass, implicitly assuming that these supply chains will have clear GHG benefits over long-distance supply chains. However, in countries such as the Netherlands, the domestic supply potential of biomass available for bioenergy and bio-based chemicals is limited[197]. As a consequence, the Netherlands will depend, to a large extent, on intra-EU and extra-EU imports of sustainably sourced biomass to develop its bio-based economy and biorefining sector[198,199]. Such developments will also have to comply with key sustainability criteria of which GHG emissions savings in comparison to fossil counterparts are considered a crucial aspect[200], particularly for energy outputs that need to comply with GHG emissions-savings criteria as established in the revised Renewable Energy Directive (REDII).

The main goal of the study is to assess supply-chain options with internationally sourced lignocellulosic biomass for multi-output biorefineries to identify optimal supply-chain design from a GHG perspective; furthermore, a domestic sugar-beet supply chain is also assessed for comparison purposes. The secondary goal of the study is to indicate the supply chains' GHG emissions savings in comparison with relevant fossil counterparts. This type of assessment can help to maximize GHG savings for the bioenergy and bio-based material sectors in the EU and guarantee sustainable production and efficient use of biomass. It can also ensure a smooth transition from a fossil-fuel-based to a bio-based economy, strengthen the bio-based materials sector from a GHG savings perspective towards future production and demand, facilitate biomass trade, contribute to sound planning for European GHG emissions reduction targets, and avoid the devalorization of biomass streams.

The Netherlands has been selected as a case study given that biomass imports are expected to continue growing in the future to meet national climate targets and potential growth in the demand for bioenergy and bio-based materials[198,199]. The biorefinery is assumed to be located in an existing lounge refinery cluster in the port of Rotterdam. Three lignocellulosic international supply areas relevant for the Dutch context were considered: (1) Most of the European woody biomass imports, especially wood pellets, originate from the southeastern USA[201]. This trade stream, driven by European demand, has increased almost ten times in the last seven years[202] (2) In Europe, the Baltic states› wood exports have recently developed and this area is expected to play an important role in the biomass trade sector[203]. (3) Recently, sugarcane bagasse pellets from São Paulo state (Brazil) are considered to have a high export potential, which meets economic, social, and sustainability criteria[204]. A domestic sugar beets supply stream was also considered for comparison purposes (domestic sourced sugar crops versus internationally sourced lignocellulosic biomass). This crop type is considered relevant for different applications, as in bio-based materials, for the country's bio-based economy transition[205].





## 3.2 MATERIALS AND METHODS

### 3.2.1 Scope

The output of the biorefinery is to a large extent determined by the characteristics of the different biomass feedstocks. Lignocellulosic biomass consists mainly of lignin, cellulose, and hemi-cellulose[206]. Cellulose is a polysaccharide of glucose (C6 sugar) whereas hemi-cellulose consists mainly of C5 sugars (xylose and arabinose) and some C6 sugars (galactose, glucose, and mannose). In contrast, sugar beets (SB) only provide C6 sugars (sucrose), and beet pulp. Currently, industrial fermentation of biofuels and bio-based chemicals is mainly carried out by employing *Saccharomyces cerevisiae* to ferment C6 sugars[207–209]. However, modifications in *S. cerevisiae* have allowed this yeast to ferment C5 sugars on an industrial level[209–211]. The employment of such yeast for industrial / commercial purposes is still under development and mainly focuses on ethanol production[209,212]. To ensure the comparability of the results between the different supply chains it is critical that each of the product systems assessed serves the same function. The simultaneous production of bio-based chemicals from C6 sugars (lactide) and biofuels (ethanol) from C5 sugars is therefore considered the main function of the multi-output biorefinery. Accordingly, two functional units have been defined: 1kg of lactide and 1MJ of ethanol.

To determine the GHG emissions from the different supply chains and the emissions hotspots, an attributional life-cycle assessment (LCA) approach is used. A 'cradle-to-factory-gate' scope is adopted in this study, i.e. including all stages from biomass production to conversion of bio-based chemicals and biofuels. The use phase and end-of-life (waste management) are excluded from the carbon footprint. This scope is sufficient to identify the best performing supply-chain design from a GHG emission perspective in line with the study goal. Upstream emissions from fuels and chemical / agriculture inputs were included for every supply chain. The distinctive conditions from the conversion routes according to regional characteristics were considered. The technical scope of the study, system conversion to ethanol and lactide, is represented in Figure 3-1. It is assumed that the biorefinery will be operational by 2022. Greenhouse gas emissions other than $CO_2$ ($CH_4$ and $N_2O$) are expressed in $CO_2$ equivalent for a global-warming potential (GWP) impact calculated over 100 years (GWP100), consistent with the characterization factors used in REDII.

3

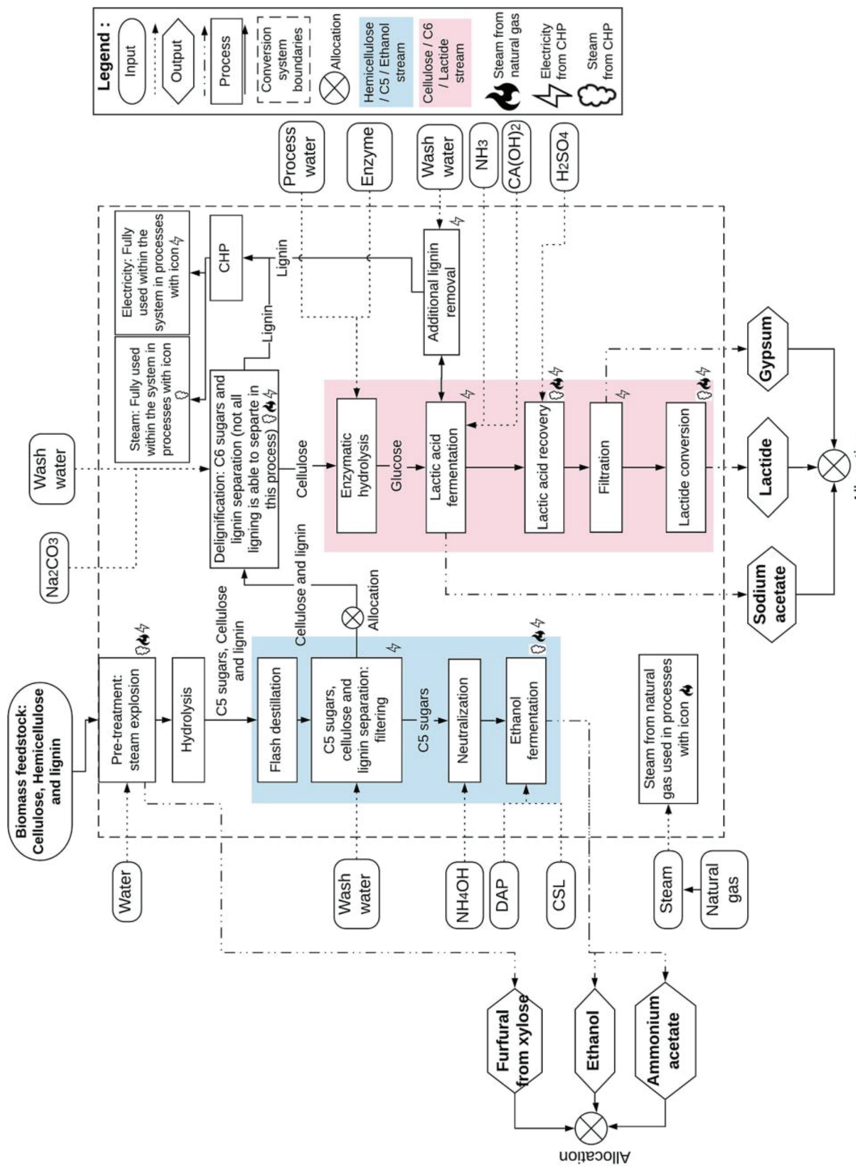


**Figure 3-1 Biorefinery conversion system from biomass to ethanol and lactide.**





The scope was extended to calculate GHG savings for ethanol and lactide. For ethanol, the use phase was included to allow a comparison with the fossil fuel counterpart as described in RED II (annex V). However, RED II assumes carbon neutrality for biofuels and $CO_2$ emissions released from the use phase are considered to be zero. For lactide, the scope is still up to factory gate but the impacts from the conversion from lactide to polylactic acid (PLA) were considered (as a proxy indicator) for the comparison with the relevant fossil-based counterparts. In addition, and for a consistent comparison with relevant fossil based counterparts, carbon emissions derived from biomass need to be addressed in terms of carbon neutrality[213]. The carbon uptake from plants and embedded in the materials is therefore considered and included as negative emission through the LCA as discussed in Kikuchi *et al.*[213,214].

### 3.2.2 Investigated supply chains

The main combinations of supply chains analyzed in this study are displayed in Figure 3-2 and summarized in Table 3-1. For the base-case supply chains, stem wood (SW) is harvested, seasoned, and collected at roadside or collected at the sugar cane mill in case of bagasse (BG) in the corresponding sourcing country. Lignocellulosic biomass is transported to a pellet plant for pelletization, domestically sourced SB is transported directly from the collection site to the biorefinery. After pelletization, pellets are transported to a terminal and shipped with a dry bulk carrier to the Netherlands. In the port of Rotterdam, pellets are processed in the biorefinery. First, pellets undergo a conversion process that separates hemicellulose, cellulose, and lignin. Hemicellulose and cellulose are processed in C5 and C6 sugars while the lignin is used as a fuel in a combined heat and power (CHP) plant to generate heat and electricity used by the biorefinery. All the generated heat and electricity are utilized within the biorefinery (no surplus). The C5 and C6 sugars are then processed separately into ethanol and lactide respectively. In the case of SB, ethanol and lactide are both produced from C6 sugars in two separate streams.

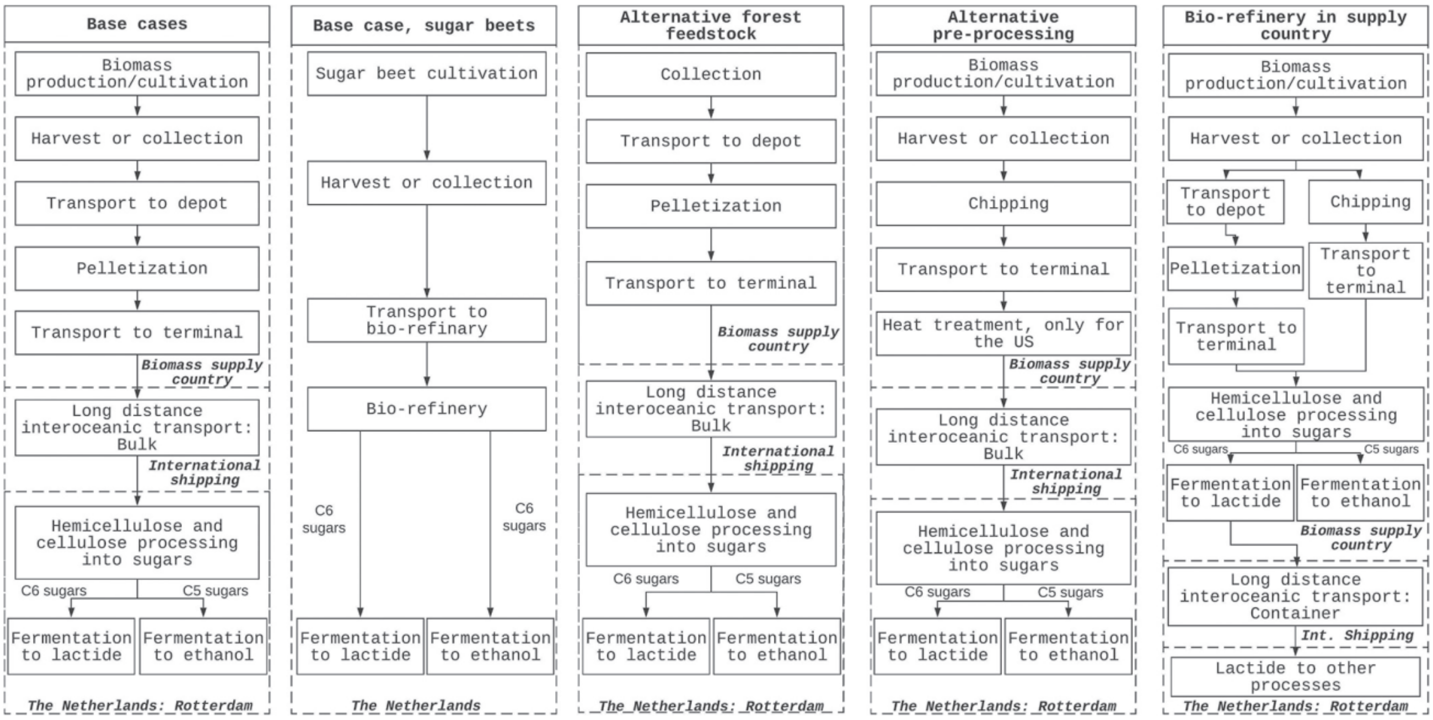


**Figure 3-2 Flow charts of internationally sourced lignocellulosic biomass and locally sourced sugar beets processed into lactide and ethanol in the Netherlands. Conversion to ethanol and lactide in the biomass exporting country is only assessed for the sensitivity scenarios.**





**Table 3-1 Overview of the base-case supply chains and supply-chain alternatives.**

| Description | Supply chain name[a] | Feedstock | Treatment | Biomass sourcing region | Bio-refinery location |
|---|---|---|---|---|---|
| Base cases | Pellets SW US - NL C | Pulp grade stem wood | Pelletization | US-Southeast (Georgia) | The Netherlands (Rotterdam) |
| | Pellets SW LV - NL C | Pulp grade stem wood | Pelletization | Baltic states (Latvia) | The Netherlands (Rotterdam) |
| | Pellets BG BR - NL C | Bagasse | Pelletization | Brazil (Sao Paulo) | The Netherlands (Rotterdam) |
| | SB NL C | Sugar beet | n/a | The Netherlands | The Netherlands (Rotterdam) |
| Alternative forest biomass feedstock | Pellets FR US - NL C | Primary forest residues | Pelletization | US-Southeast (Georgia) | The Netherlands (Rotterdam) |
| | Pellets FR LV - NL C | Primary forest residues | Pelletization | Baltic states (Latvia) | The Netherlands (Rotterdam) |
| | Pellets SR US - NL C | Sawmill residues | Pelletization | US-Southeast (Georgia) | The Netherlands (Rotterdam) |
| | Pellets SR LV - NL C | Sawmill residues | Pelletization | Baltic states (Latvia) | The Netherlands (Rotterdam) |
| Alternative pre-processing: chipping and heat | Chips SW US - NL C | Pulp grade stem wood | Chipping/heat | US-Southeast (Georgia) | The Netherlands (Rotterdam) |
| | Chips SW LV - NL C | Pulp grade stem wood | Chipping/heat | Baltic states (Latvia) | The Netherlands (Rotterdam) |
| | Chips FR US - NL C | Primary forest residues | Chipping/heat | US-Southeast (Georgia) | The Netherlands (Rotterdam) |
| | Chips FR LV - NL C | Primary forest residues | Chipping/heat | Baltic states (Latvia) | The Netherlands (Rotterdam) |

[a] Biomass acronyms: SW = stemwood , BG = Bagasse, SB = Sugar beets, FR = Forest residues, SR = Sawmill residues

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Primary forest residues (FR) and sawmill residues (SR) are considered as feedstock for the alternative forest biomass feedstock supply chains. These feedstock types are collected and transported to a pellet facility. The energy intensity of the pelletization process from this type of feedstock was considered. For instance, grinding of SR requires less energy given their structural composition.[215] After pelletization, the alternative forest biomass feedstock supply chains follow the same pathway and conversion to the lactide / ethanol route as the base cases.

For the alternative pre-processing supply chains, biomass is cultivated / collected at the biomass source location. Wood is debarked and chipped on site. The bark is assumed to be left on site while wood chips are transported directly to the terminal. When wood chips are exported from the USA to the EU, phytosanitary measures (heat treatment) are required[216]. Dry bulk carriers are used to transport wood chips from biomass sourcing country to the Netherlands. These supply chains follow the same conversion to the lactide / ethanol routes as the base cases. Note that these supply chains are technically not feasible for BG and SR. The physical characteristics and handling properties of, for example sawdust, make it very difficult to transport over long distances in its raw form.

For sensitivity scenarios with the biorefinery located in the country of biomass supply, the biorefinery is considered to be located at the terminal from which biomass is exported in other supply chains. This is carried out to indicate the difference in GHG performance from locating the biorefinery close to biomass supply but remote from existing refinery infrastructure and demand. For these scenarios, biomass is pelletized or chipped on site and converted to lactide / ethanol as described in the other cases. After conversion, lactide is transported to Rotterdam in containers and ethanol stays in the biomass supply country. In Rotterdam, lactide is assumed to be an intermediate used for other processes.

### 3.2.3 Inventory analysis

#### 3.2.3.1 Harvest and collection

For the southeast USA US-SE, wood was assumed to be supplied from a 25-year rotation softwood plantation with medium intensive forest management practices as described in Jonker *et al.*[217]. Forestry activities include raking, spot piling, bedding, fertilizer application, thinning, clear-cut harvest and forwarding. Wood from the Baltic states is assumed to come from natural managed forest. Only diesel consumption from forestry activities related to clear cut harvesting and forwarding were therefore included. For the Netherlands, it was considered the agricultural management practices for Dutch SB cultivation presented in Smit *et al.* [218]. In addition, agricultural equipment is used during





soil preparation, pesticides / fertilizers application, and SB collection. Fertilizer-induced $N_2O$ emissions were taken into account for all scenarios and calculated in accordance with IPCC, (2006) Tier 1 methods[118].

In accordance with RED II calculation methods, residues are considered to have zero GHG emissions before the process of collection[200]. Forest residues and sugarcane bagasse residues are therefore considered to have zero GHG emissions upstream of collection. The forest residues collection for the southeast USA and Baltic states contemplates the operations of forwarding, lifting, loading, and unloading[219]. Table 3-2 includes the data used for the collection and cultivation processes. Dry matter percentage losses were also included for each step and are included in Table S 3-1

**Table 3-2 Harvest and collection activity data.**

| Process | US southeast (US-SE) | Baltic states (BS) | Brazil (BR) | Netherlands (NL) |
|---|---|---|---|---|
| *Cultivation ( $kg/ha$ )* | | | | |
| N-fertilizer | 582[a] | - | - | 149[d] |
| $P_2O_5$ | 103[b] | - | - | 50[d] |
| $K_2O$ | - | - | - | 40[d] |
| CaO | - | - | - | 72[d] |
| Pesticides | - | - | - | 15[d] |
| Seeds | - | - | - | 3.6[d] |
| Diesel | 1105[a] | 1023[a] | - | 98[d] |
| Yield | 280000[c] | 280000[c] | - | 81390[d] |
| *Forest residues collection ( $kg/ha$ )* | | | | |
| Diesel | 0.0052[e] | 0.0052[e] | - | - |

[a] 220; N-fertilizer is composed by Diammonum Phospate (DAP) and urea
[b] 46% of DAP
[c] 217
[d] 218; N-fertilizer is composed by synthetic fertilizer and manure
[e] 219

3

#### 3.2.3.2 Road transport

A spatial explicit approach based on actual infrastructure and biomass availability was used to calculate aggregated road transport distances for all scenarios. This process was carried out on a geographic information system (GIS) tool. The weighted average distances are summarized in Table 3-3 . These distances from biomass supply locations to the pellet facilities and to the export port were calculated based on the optimized total delivery routes when considering the capacity of each pellet facility and surrounding biomass availability. Pellet facilities were therefore selected based on the priorities mentioned above (i.e. distance from biomass supply locations to pellet factory gate, which is cumulative for multiple transport loads) until total cumulative biomass to be utilized at the biorefinery is achieved. When pellet facilities are not within the system (i.e. wood chips and SB scenarios), then transport distances from the feedstock origins are optimized for and consider the local feedstock type capacity until total biorefinery demand is met.

**Table 3-3 Aggregated distances**

| Description | Supply chain name[ab] | Feedstock supply location to pellet mill (km) | Pellet mill to port (km) | Feedstock supply location to port (km) | Port |
|---|---|---|---|---|---|
| Base cases | Pellets SW US - NL C | 67.8 | 182.1 | - | Savannah |
| | Pellets SW LV - NL C | 59.3 | 119.2 | - | Riga |
| | Pellets BG BR - NL C | - | 364 | - | Santos |
| | SB NL C | - | - | 190 | Rotterdam |
| Alternative forest biomass feedstock | Pellets FR US - NL C | 62 | 182.1 | - | Savannah |
| | Pellets FR LV - NL C | 112.5 | 119.2 | - | Riga |
| | Pellets SR US - NL C | 73.7 | 182.1 | - | Savannah |
| | Pellets SR LV - NL C | 99.9 | 119.2 | - | Riga |
| Alternative pre-processing: chipping and heat | Chips SW US - NL C | - | - | 111.3 | Savannah |
| | Chips SW LV - NL C | - | - | 26 | Riga |
| | Chips FR US - NL C | - | - | 77.5 | Savannah |
| | Chips FR LV - NL C | - | - | 94.3 | Riga |

[a] Biomass acronyms: SW = stemwood , BG = Bagasse, SB = Sugar beets, FR = Forest residues, SR = Sawmill residues
[b] Country dependent emissions factors for heavy duty 30-32 ton truck for the US, Europe and Brazil were included.[138,221,222] In addition, empty return trips with no load were assumed.

Biomass potentials from stem wood and residues (including FR and SR) at the county level in the state of Georgia were obtained from Fingerman et al.[202] combined with the locations and capacities of actual pellet plants in Georgia.[223] For the Baltic states , sustainable forest biomass potentials were derived from Dees et al.[224] and combined with actual pellet plant locations and capacities of Graanul Invest, the largest pellet producer in the Baltic states.[225]

The sustainable potential of SB in the Netherlands and BG in Brazil were not available. For the Netherlands SB, the geographic potential was determined based on SB suitability maps for the Netherlands. A mean suitability value was calculated,[226] taking into account only SB land cover area in the Netherlands for 2015.[227] This value was combined with the average yield for SB in the Netherlands for 2015.[228] The average suitability/yield relation





from SB was used to derive SB yield from each corresponding suitability value for each production area. This approach resulted in a supply map (proxy indicator) with location and yield of SB spatially explicit defined for the Netherlands. Finally, an optimized service area was calculated that considered the distance required to supply the biomass demands at the biorefinery when considering the calculated yields. For the Brazil sugar cane BG, existing pellet facility locations were used.[229] Note however that sugarcane BG pellets in Sao Paulo state will likely be produced at the sugarcane mill sites itself.[230] Sugarcane BG availability (total supply) was assumed from the nearest sugar cane mill site to the existing pellet plant.[231] BG availability (a side product of the Sugar cane mills) was calculated based on the production statistics of sugar cane mills and a ratio of 26% sugarcane to BG.[232,233] Only 35% of the BG is estimated to be available to produce BG pellets. The remaining fraction is required to meet internal energy demand of the sugar cane mill (mostly for drying).[234] Thus, 9.1% of total raw feedstock that enters each sugar cane mill is available to produce BG pellets.



#### 3.2.3.3 Pre-treatment (chipping, drying, pelletization)

The pelletization process of round wood includes debarking, chipping, grinding, densification and cooling.[215] Pelletization of BG requires fewer processing steps compared to woody biomass. It includes conditioning, grinding, densification, cooling and some miscellaneous procedures.[235] The energy requirements of the individual pre-processing steps at the pellet plant are summarized in Table 3-4 .Biomass was assumed to be used as fuel for the drying process. A reduction of moisture content from 30% to 10% for wood pellets and from 50% to 10% for BG pellets was considered. The heat demand and associated biomass fuel is calculated from the required evaporation of water 0.28 $kg_{water}/kg_{pellets}$ for wood pellets and 0.8 $kg_{water}/kg_{pellets}$ for BG pellets and boiler efficiency of 85% based on the method and values from Thek and Obernberger,[215] and Edwards et al.[138].Country specific electricity mixes were considered to calculate the GHG emissions of electricity use. On-site chipping (diesel powered) for chips pathways includes debarking, chipping, and loading of feedstock.[236,237] For the phytosanitary measures, chips are dried at a temperature at which the organisms cannot survive. The drying is assumed to be carried out in the export port. For the drying process biomass is used as fuel and the same methodology as applied for pellets was used. A reduction of wood chips moisture content from 30% to 10% was considered.

**Table 3-4 Pelletization / chipping activity data.**

| Process | Unit | Wood | Bagasse |
|---|---|---|---|
| *Pelletization* | | | |
| Conditioning | $MJ_{electrity}/kg_{pellets}$ | - | 0.022[b] |
| Debarking | $MJ_{electrity}/kg_{pellets}$ | 0.038[a] | - |
| Chipping | $MJ_{electrity}/kg_{pellets}$ | 0.13[a] | - |
| Grinding | $MJ_{electrity}/kg_{pellets}$ | 0.067[a] | 0.076[b] |
| Densification | $MJ_{electrity}/kg_{pellets}$ | 0.14[a] | 0.14[b] |
| Cooling | $MJ_{electrity}/kg_{pellets}$ | 0.005[a] | 0.022[b] |
| Additional | $MJ_{electrity}/kg_{pellets}$ | - | 0.065[b] |
| Heat requirement | $MJ_{heat}/MJ_{pellets}$ | 0.066[a] | 0.207[b] |
| *Chipping and debarking on site* | | | |
| Chipping | $MJ_{diesel}/kg_{biomass}$ | 0.13[c] | - |
| Debarking | $MJ_{diesel}/kg_{biomass}$ | 0.065[d] | - |

a 215
b 235
[c] based on efficiency from drum and disc chippers [236]
[d] based on chain flail debarker [237]

### 3.2.3.4 Long-distance sea transportation

Methods from Edwards et al.[138] were used to calculate GHG emissions from maritime transport. Bulk carriers are used for oceanic transports of wood chips and wood pellets. Bulk carriers are constrained by the bulk density of the transported goods and the design load is volume limited.[138] The following bulk densities were assumed:

Wood chips: 223-328 $kg/m^3$
Wood and BG pellets: 650 $kg/m^3$

Supramax bulk carriers were considered for maritime transport with fuel (heavy fuel oil) efficiency depending on transported feedstock.[138] Additionally, return trips with 30% load were taken into account (30% of total distance under ballast/empty) . Electrical grab cranes are used for loading and unloading pellets/chips in Dutch terminals.[238] GHG emissions from electric grab cranes were calculated based on the work from Tilke et al.[239] and are dependable on feedstock bulk density (0.00097 $MJ_{electrity}/kg_{pellets}$, 0.0028 $MJ_{electrity}/kg_{\ chips}$). Distances between ports of export and the port of Rotterdam were based on actual maritime routes.[240]





### 3.2.3.5 Transportation of lactide in containers (only valid for sensitivity scenarios) from biomass sourcing country

Transportation of lactide in containers is valid only for the sensitivity cases when the biorefinery is located in the country of biomass supply. Lactide is assumed to be transported in a solid state in Twenty Food Equivalent (TEU) containers with a load volume of 33.2 $m^3$ per container. These containers are transported with 3000 - 4999 TEU type container ships. The bulk density of lactide was assumed to be similar to calcium lactate (700 $kg/m^3$). GHG emissions from container transport by ship are calculated following Equation 3-1.[241]

**Equation 3-1**

$$GHG\ emissions/_{container} = MCR * LF * AT * EF$$

Where,
MCR = Maximum Continuous Rating of the combustion engine in use, *kW*. Typically, this value corresponds to 80% of the installed engine power [242]
LF = Load Factors , $\left({}^{\text{actual speed}_{km/h}}/_{\text{Maximum speed}_{km/h}}\right)^3$ (unit less);
AT = Activity time, *h;*
EF = Power based emission factor for the greenhouse gas, $kg/kWh$;

Maximum speed, TEU capacity, and MCR data were gathered from Rickmers Holdings group.[243] The actual speed of 35 $km/h$ was assumed, which is the common navigation speed of these type of ships.[244] The activity time was calculated based on distance from port to port and the actual speed. Power based emissions factors were based on Veidenheimer.[241] It was assumed that 65% of the total capacity of the vessel was used for lactide transportation given that container ships carry different products at a time and 70% of container volume utilization (limited by container payload capacity). The containers are being loaded with electrically powered cranes at the biomass source country terminal and unloaded at Rotterdam biorefinery. Electrical cranes energy consumption (5,26 $kWh/TEU$) was calculated following Tran et al.[245].The GHG emissions of loading and unloading are country specific as a result of the variations in carbon intensity of electricity generation in each country.

### 3.2.3.6 Conversion to lactide and ethanol

To split lignocellulosic biomass into cellulose, hemicellulose and lignin at the biorefinery, biomass undergoes a steam explosion process (Figure 3-1). Furfural ($C_5H_4O_2$) is formed during the steam explosion as a result of small share of xylose degradation. A share of the hemicellulose is then processed into 5-carbon sugars by a hydrolysis process. A split-up process is carried out to separate xylose from lignin and cellulose. The liquid filtered part contains the xylose and the solid filtered part the cellulose and lignin. The filtered liquid is neutralized with ammonium hydroxide ($NH_4OH$). Ammonium acetate ($C_2H_7NO_2$) is formed from the neutralization process. Diammonium phosphate (DAP) and corn steep liquor (CSL) are then added into the neutralized liquid where xylose is fermented into ethanol. A chemical delignification process with sodium carbonate ($Na_2CO_3$) is used to remove the lignin from the filtered solid. Lignin is burned in a combined heat and power (CHP) plant to generate electricity and steam at the biorefinery. After another separation procedure, enzymes are used to process cellulose into glucose by means of enzymatic hydrolysis. Then, glucose is fermented into lactic acid.

For the lactic acid fermentation process, ammonia ($NH_3$) is added and calcium hydroxide ($Ca(OH)_2$) is applied to control the pH of the fermentation broth. An additional separation process is carried out to remove additional lignin, which is burned in the CHP plant. The steam and electricity from the CHP are fully used within the system. The steam from the CHP is assigned to each process with the relative share from the total steam demand. We assumed that the extra heat demand needed for each process is supplied from natural gas. The lactic acid fermentation process results in the formation of calcium lactate ($C_6H_{10}CaO_6$) and in low quantities of sodium acetate ($C_2H_3NaO_2$). $C_6H_{10}CaO_6$ is treated with sulfuric acid ($H_2SO_4$) to recover lactic acid. As a result of this process, lactic acid is formed and calcium sulfate / gypsum ($CaSO_4$) is precipitated. The $CaSO_4$ is removed by filtration. For the last step, water is removed from the lactic acid and processed into lactide. Note that the quantity of chemical inputs used for lactide production in the biorefinery is output dependent[246]. A schematic representation of the conversion process from lignocellulosic biomass to ethanol and lactide is depicted in Figure 3-1.

For the conversion of SB to lactide, it was assumed that the energy demand (power and heat) for the fermentation process of SB to lactide is similar to the SB-to-ethanol fermentation process presented in Edwards *et al.* [138]. First, SB is processed to extract sugars. The extracted sugar juice is pasteurized and fermented into lactide. As in other pathway conversion processes, $Ca(OH)_2$ is applied to control the pH of the fermentation broth. $C_6H_{10}CaO_6$ is formed and is treated with $H_2SO_4$ to recover lactic acid. Gypsum is formed from the lactic acid recovering process and is filtered. For the final process, purified lactic acid is processed into lactide. Sugar-beet pulp is an additional output generated from the fermentation process. Sugar-beet pulp undergoes a treatment process where it is pressed, dried and pelletized. For the ethanol stream, SB is processed and fermented with only sugar beet pulp as additional output[138]. Table **5** includes the inventory data for the different conversion routes.





**Table 3-5 Input data for the conversion to ethanol and lactide process.**

| Feedstock | Product | Mass flow (kg/year) | Mass allocation factor (%) | LHV (MJ/kg) | Energy allocation factor (%) | Price (€/kg) | Economic allocation factor (%) |
|---|---|---|---|---|---|---|---|
| Wood | Lactide | 255.987.048,9 | 45,97 | 18,72[a] | 71,25 | 2,47[d] | 89,65 |
| | Ethanol | 63.747.939,9 | 11,45 | 26,81[b] | 25,41 | 0,66[e] | 6,03 |
| | Sodium acetate | 11.689,6 | 0,00 | - | - | 0,79[f] | 0,00 |
| | Gypsum | 205.757.911,6 | 36,95 | - | - | 0,008[g] | 0,23 |
| | Ammonium acetate | 21.976.441,3 | 3,95 | - | - | 0,84[h] | 2,61 |
| | Furfural | 9.410.719,4 | 1,69 | 23,98[c] | 4,00 | 1,1[i] | 1,47 |
| | Total | 556.891.750,8 | 100 | - | 100 | - | 100 |
| Bagasse | Lactide | 255.987.048,9 | 41,00 | 18,72[a] | 56,35 | 2,47[d] | 83,00 |
| | Ethanol | 122.332.237,7 | 19,31 | 26,81[b] | 38,57 | 0,66[e] | 10,71 |
| | Sodium acetate | 16.692,7 | 0,00 | - | - | 0,79[f] | 0,00 |
| | Gypsum | 205.757.911,6 | 32,48 | - | - | 0,008[g] | 0,22 |
| | Ammonium acetate | 31.382.358,1 | 4,95 | - | - | 0,84[h] | 3,45 |
| | Furfural | 18.059.161,9 | 2,85 | 23,98[c] | 5,08 | 1,1[i] | 2,61 |
| | Total | 633.535.410,9 | 100 | - | 100 | - | 100 |
| Sugar beet | Lactide | 255.987.048,9 | 51,62 | 18,72[a] | 90,68 | 2,47[d] | 99,47 |
| | Sodium acetate | 16.692,7 | 0,00 | - | - | 0,79[f] | 0,00 |
| | Gypsum | 205.757.911,6 | 41,49 | - | - | 0,008[g] | 0,26 |
| | Sugar beet pulp | 34.111.554,2 | 6,88 | 14,43[b] | 9,32 | 0,05[j] | 0,27 |
| | Total | 495.873.207,5 | 100 | | 100 | | 100 |
| Sugar beet | Ethanol | 181.144.115,4 | 84,15 | 26,81 | 90,80 | 0,66[e] | 98,62 |
| | Sugar beet pulp | 34.111.554,2 | 15,85 | 14,43[b] | 9,20 | 0,05[j] | 1,38 |
| | Total | 215.255.669,6 | 100 | - | 100 | - | 100 |

[a] assumed from PLA [247]
[b] [138]
[c] [248]
[d] Price of raw lactide [249]
[e] Average price of ethanol between 2016-2018 [250]
[f] Average price from sodium acetate reported in e-commerce webpage [251]
[g] [252]
[h] Average price from ammonium acetate reported in e-commerce webpage [253]
[i] [254]
[j] Average price from sugar beet pulp for the Netherlands [255]

### 3.2.3.7 Multifunctionality

The approach chosen to deal with multi-input and multi-output systems in LCA has a strong influence on the results and is, therefore, a debated issue[256]. The lack of harmonization between standards and handbooks and their ambiguity generate inconsistencies when choosing a solution for multifunctionality problems. For instance, the LCA methodology standardized in ISO 14040/14044 requires that allocation should be avoided by subdivision or system expansion when feasible; when these are not possible, allocation should be in proportion to the physical properties of the products such as mass or energy, or economic value. Other standards, such as RED II, allocate the burden of impacts among products and by-products by energy content. However, this specification is frequently challenged because not all products and by-products of biorefineries are meant for energy purposes[257,258].

The inclusion of the functions of ethanol and lactide provision in a single functional unit would partly avoid allocation but was considered infeasible because the share of lactide and ethanol varies between the studied alternatives as a result of the differences in cellulose and hemicellulose content. This would lead to inconsistent reference flows. System boundary expansion or substitution is not adequate for the attributional model applied in this study. Mass allocation deemed the preferred allocation method because (1) the majority of the by-products such as $C_2H_7NO_2$ or $CaSO_4$ are not energy carriers; and (2) most of the biorefinery outputs, with the exception of ethanol, are not intended for energy purposes. Alternatively, economic allocation could have been applied as demonstrated in the sensitivity analyses (Figure 3-9). However, this method is sensitive to price fluctuations that are highly unstable[259]. There is no single correct approach to deal with multifunctionality and it is a major source of method-induced variability. Results vary substantially if alternative methods are used, as shown in Figure 3-9[260].

As illustrated in Figure 3-1, a multi-level allocation approach was used at three specific points in the system to isolate the individual production routes of ethanol and lactide that are produced from the shared process of lignocellulosic biomass conversion to fermentable sugars. Lignin is also an intermediate output of this shared process but is used entirely within the system to generate electricity and heat. The allocation factors are displayed in Table 3-6. First, mass allocation was applied between the C5 and C6 sugars (intermediate products) to share the burden of GHG emissions up to this conversion to sugars step and to allocate the heat generated by natural gas in the biorefinery and its associated GHG emissions to the individual downstream production routes of ethanol (C5) and lactide (C6). The share depends on feedstock-specific hemicellulose, cellulose, and lignin content (Table S 3-2). After this separation, two different process routes can be clearly distinguished: C5 to ethanol and C6 to lactide. Each process route produces by-products. $C_2H_3NaO_2$ and gypsum are by-products of lactide production from C6 sugars. $C_2H_3NaO_2$ is considered a by-product even if it is produced in almost negligible amounts compared to lactide. $C_5H_4O_2$ and $C_2H_7NO_2$ are by-products from ethanol production





from C5 sugars. $C_5H_4O_2$ is commonly used for bio-based chemicals applications and $C_2H_7NO_2$ (a salt) as food additive or buffering substance[261]. Mass allocation was applied at sub-level between the individual production outputs of lactide and ethanol and associated by-products.

**Table 3-6 Allocation factors for main results.**

| Feedstock | Process | Mass allocation factor (%) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | C6 sugars | C5 sugars | Lactide | $C_2H_3NaO_2$ | $CaSO_4$ | Ethanol | $C_2H_7NO_2$ | $C_5H_4O_2$ | Sugar beet pulp |
| Wood | *Sugars portioning[a] conversion* | 69.14 | 30.86 | - | - | - | - | - | - | - |
| | *To lactide* | - | - | 55.43 | 0.02 | 44.55 | - | - | - | - |
| | *To ethanol* | - | - | - | - | - | 67.01 | 23.10 | 9.89 | - |
| Bagasse | *Sugars portioning[a]* | 53.87 | 46.13 | - | - | - | - | - | - | - |
| | *To lactide* | - | - | 55.43 | 0.02 | 44.55 | - | - | - | - |
| | *To ethanol* | - | - | - | - | - | 71.22 | 18.27 | 10.51 | - |
| Sugar beet | *To lactide* | - | - | 51.62 | 0.02 | 41.49 | - | - | - | 6.87 |
| | *To ethanol* | - | - | - | - | - | 84.15 | - | - | 15.85 |

[a] Process generates C5/C6 sugars and lignin. The lignin is used in a CHP plant to generate heat and electricity which is used within the system boundaries (no surplus).





Although the biorefinery is a single facility with interconnected energy flows, the multi-level allocation approach avoids overestimation or underestimation of GHG emissions for the two functions of the system that were considered. The production of lactide from C6 sugars is more resource and energy intensive than the production of ethanol from C5 sugars. If the environmental burden (GHG emissions) would be distributed proportionally over the outputs of the total system, part of the environmental burden resulting from the lactide production process would be allocated to ethanol.

The ethanol results in the section headed 'GHG emission savings in comparison to fossil-based counter parts' are presented following RED II methods. For this section and to allow a consistent comparison with the fossil fuel counterpart presented in RED II, energy allocation has been applied considering all the system outputs (without multi-level allocation) and RED II calculation rules. The energy allocation factors applied for that section are present in Table 3-7.



**Table 3-7 Energy allocation factors.**

| Feedstock | Product | Energy allocation factor (%) |
|---|---|---|
| Wood | Lactide | 70,32 |
| | Ethanol | 26.23 |
| | Sodium acetate | - |
| | Gypsum | - |
| | Ammonium acetate | - |
| | Furfural | 3.45 |
| | Total | 100 |
| Bagasse | Lactide | 55.25 |
| | Ethanol | 39.54 |
| | Sodium acetate | - |
| | Gypsum | - |
| | Ammonium acetate | - |
| | Furfural | 5,21 |
| | Total | 100 |
| Sugar beet | Ethanol | 90,80 |
| | Sugar beet pulp | 9,20 |
| | Total | 100 |

### 3.2.4 GHG emissions saving criteria

The RED II has established minimum GHG emissions saving requirements for bioenergy in comparison with their fossil fuel counterparts. From October 2015 until December 2020, biofuels should comply with at least 60% of GHG emissions savings; from January 2021 onwards, biofuels should comply with at least 65% of GHG emissions savings. This signifies that biofuels conversion routes require to emit no more than 37.6 $g\ CO_{2eq}/MJ_{ethanol}$ from 2015 (after October) to 2020 and 32.9 $g\ CO_{2eq}/MJ_{ethanol}$ from 2021 onwards. For bio-based materials, there are no binding minimum GHG emissions saving requirements or other sustainability criteria at the EU level. Lactide is an intermediate for polylactic acid (PLA) production. The emissions from the ring opening polymerization of lactide for PLA production reported in Vink and Davies,[262] were used and added as a proxy indicator to have a more adequate comparison between the lactide supply chains and fossil-based counterparts. Three fossil-based polymers were considered for the comparison exercise; (1) polystyrene (PS) 2.25 $kg\ CO_{2eq}/kg$, for polypropylene (PP) 1.63 $kg\ CO_{2eq}/kg$ and for polyethylene terephthalate (PET) 2.2 $kg\ CO_{2eq}/kg$[263–265].

## 3.3 RESULTS

### 3.3.1 Carbon footprint of the supply chains

Figure 3-3 presents the total sum of 'cradle-to-factory-gate' GHG emissions for 1 kg of lactide production for the assessed supply chains. The GHG emissions are calculated to be between 692 g $CO_{2eq}/kg_{lactide}$ (SR pellets from the Baltic states) and 1002 g $CO_{2eq}/kg_{lactide}$ (SW chips from the USA). The conversion process of lignocellulosic biomass to lactide is the main contributor to the total GHG emissions. Those GHG emissions are mainly related to process energy demand supplied from natural gas and upstream emissions from chemicals used in the process. The use of $Ca(OH)_2$ for pH control during lactic acid fermentation is one of the main contributors. Conversion routes from the USA and Baltic states make use of the same feedstock (wood biomass) and therefore share the same conversion system and associated GHG emissions from chemicals and heat supply. Differences in conversion to lactide process between conversion routes are mainly attributed to each system›s heat demand and feedstock lignin content. Feedstocks with a higher lignin content (burned in the CHP) can supply net heat to a larger extent than feedstocks with a lower lignin content; this results in lower amounts of natural gas needed to meet total heat demand for lactide production. Thus, the higher lignin content from wood (US and Baltic states conversion routes), in comparison to BG (Brazil conversion route) and SB (no lignin present in SB, Netherland conversion route), results in better GHG emissions performance for the conversion to lactide step. For all conversion routes, with the exception of the SB one, the amount of electricity generated at the CHP from burning lignin is enough to cover the system›s electricity demand completely. The absence of lignin in SB determines that heat and electricity requirements are supplied entirely from natural gas and the grid, which results in an additional impact.





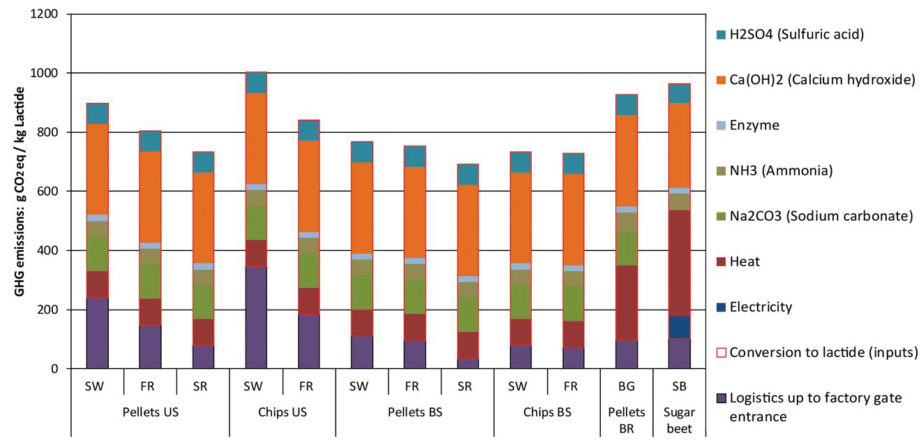


**Figure 3-3 'Cradle-to-factory-gate' GHG emissions of 1 kg lactide production.**

For ethanol, the total sum of 'cradle-to-factory-gate' GHG emissions varies between 15 g $CO_{2eq}/MJ_{ethanol}$ (SR pellets from the Baltic states) and 27 g $CO_{2eq}/MJ_{ethanol}$ (BG pellets from Brazil) as shown in Figure 3-4. The conversion process of lignocellulosic biomass to ethanol is the main contributor to GHG emissions. Those emissions are strongly related to the system's heat demand and the upstream emissions from $NH_4OH$ used in the neutralization process; it is worth noting that the SB conversion route omits the use of $NH_4OH$. The GHG emissions from the use of DAP fertilizer are also omitted.

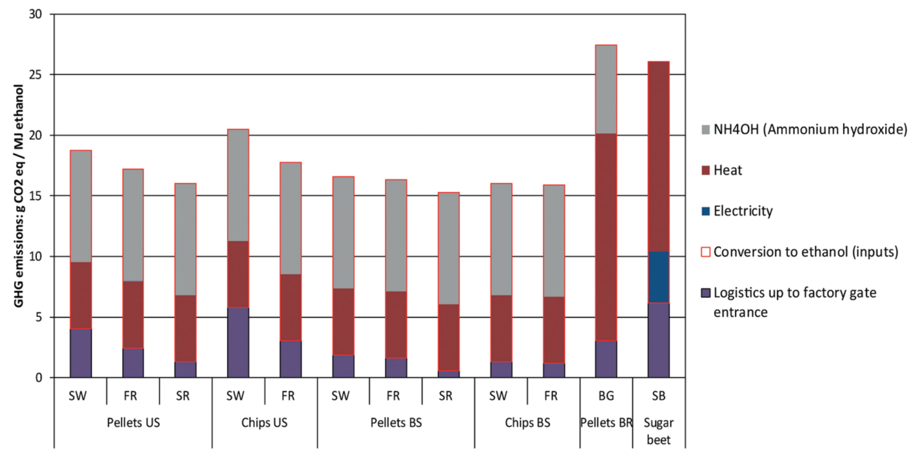


**Figure 3-4 'Cradle-to-factory-gate' GHG emissions of 1 MJ ethanol production.**

The large difference between second-generation (lignocellulosic biomass) production routes is attributed to the variance in biochemical composition between wood biomass pellets and sugarcane BG pellets (see above). These differences mean that the burden of GHG emissions from the Brazil pathway is shared more uniformly between lactide and ethanol production than for other biomass types. Although ethanol results follow a similar trend to lactide, the biophysical composition that determines the share of the burden between ethanol and lactide for the sugarcane BG pathway also results in a slightly higher GHG emissions footprint than the SB conversion route; this is contrary to the results trend for lactide. The first-generation ethanol production route (ethanol from SB in the Netherlands) emits one-third more (fossil) GHG emissions compared to second-generation ethanol from wood biomass. However, compared to the sugarcane BG, the impact is almost identical. High emissions from the SB conversion to ethanol process are related to steam and electricity demand, which are provided from natural gas and grid electricity.

### 3.3.2 GHG emissions upstream of feedstock supply up to the factory gate

Figure 3-5 presents the total sum of upstream GHG emissions of feedstock supply up to the factory entrance gate in the Netherlands for 1kg of lactide production. Greenhouse gas emissions are lowest for SR pellets from the Baltic states (36 g $CO_{2eq}/kg_{lactide}$) and highest for SW chips from the USA (346 g $CO_{2eq}/kg_{lactide}$). The GHG emissions from the logistics in the conversion routes are considerably lower than the GHG emissions from the conversion to lactide step (Figure 3-3), with the exception of SW chips from the USA. For this conversion route, the logistics emissions account for one-third from the total. The higher GHG emissions performance of wood residues (FR and SR versus SW) pathways can be accredited to the lack of a cultivation phase and the feedstock features advantages for pelletization. For instance, there are no requirements for chipping or grinding SR when pelletization is carried out. There is a tradeoff between benefits from wood on-site processing (debarking and chipping) and pelletization. On-site processing has a slightly lower impact than pelletizing. There is also a shorter transport distance as the travel from pellet facilities is omitted. However, bulk oceanic transport of (heat-treated) chips is less efficient. The lower bulk density of chips in comparison to wood pellets results in higher maritime transport emissions. Differences in GHG emissions from the pelletization process between locations arise mostly from regional characteristics in electricity production. To illustrate, Brazil's electricity carbon mix intensity is lower than the one from Latvia (Baltic states) or the USA. The availability of feedstock types can vary depending on local conditions and it is a key aspect of GHG emissions performance. For example, the availability of forest residues is comparatively lower than that of other wood biomass sources in the Baltic states.





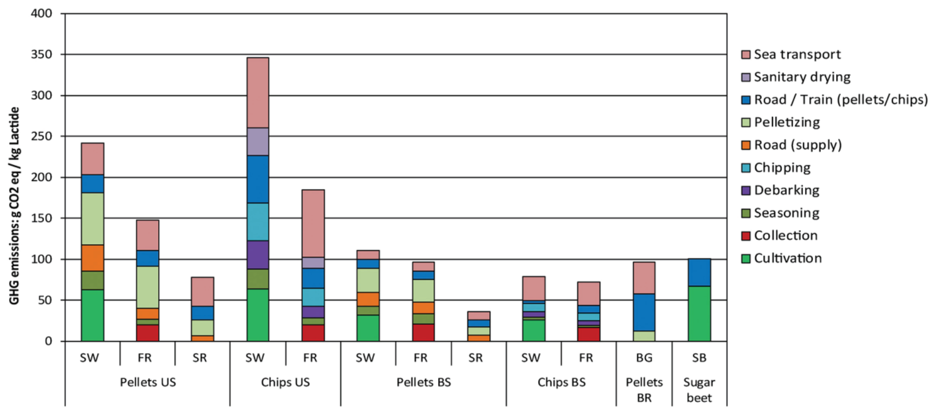


**Figure 3-5 'Cradle-to-factory-gate' entrance (logistics) GHG emissions of 1 kg lactide production.**

The GHG emissions from SB are accredited to relatively large agricultural inputs including N-fertilizers, and transport of wet SB (76% moist) from the supply area to the biorefinery. Large amounts of agricultural land area are required to supply the necessary quantity of SB to meet annual feedstock demand for lactide production in the Netherlands. For the Brazil conversion route, there is a high impact for transport given the long distance between pellet plants located inland and the port of Santos. Overall, the superior GHG performance of the Baltic states' pathways is related to shorter maritime transport distances between ports (Riga and Rotterdam), biomass availability, and shorter transportation distances between biomass supply areas and pellet plants.

In terms of upstream GHG emissions before lignocellulosic biomass conversion to ethanol, emissions vary between 0.6 g $CO_{2eq}/MJ_{ethanol}$ (SR pellets from Baltic states) and 6.5 g $CO_{2eq}/MJ_{ethanol}$ (SB from the Netherlands) as shown in Figure 3-6. The upstream GHG emissions before the conversion to ethanol are comparably higher for the Netherlands pathway than for any other conversion route. Like the results for lactide, high emissions from the Netherlands scenario are attributed mainly to high input of N-fertilizers for the SB production. The impact from N-fertilizers includes the direct and indirect $N_2O$ emissions from managed soils and fossil inputs for fertilizer production. There is also no division of the burden between ethanol and lactide logistics for SB as in other conversion routes. The system design mentioned above results in considerably larger GHG upstream emissions for the SB conversion. The rest of the conversion routes results follow a similar trend to lactide.

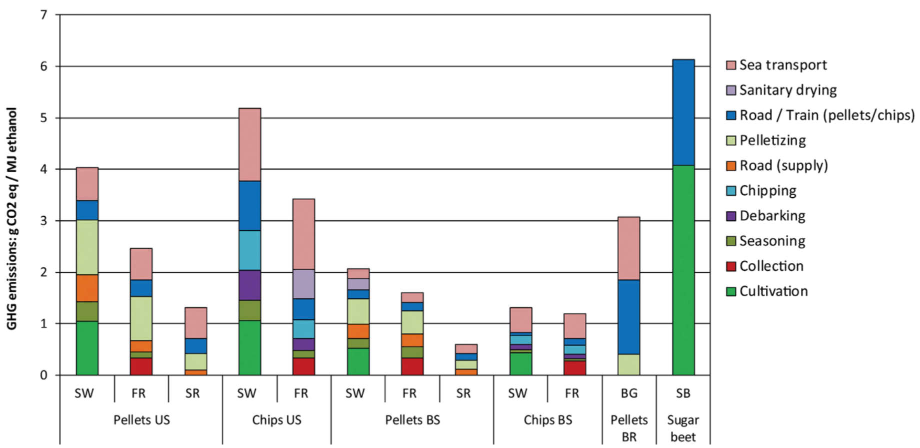


**Figure 3-6 'Cradle-to-factory-gate' entrance (logistics) GHG emissions of 1 MJ ethanol production.**

### 3.3.3 GHG emission savings in comparison to fossil-based counter parts

Figure 3-7 compares 1 kg lactide production with fossil-fuel counterparts. The error bars for the US and Baltic states supply chains represent the range of GHG emission between the different supply chains composition valid for each conversion route (only valid for woody supply chains). All conversion routes report high GHG savings in comparison with the selected fossil-based polymers. On average, the highest GHG emissions savings are reported when the conversion routes are compared with fossil-based PS; GHG emissions savings are reported to be between 2916 g $CO_{2eq}/kg_{lactide}$ (Netherlands supply chain) and 3151 g $CO_{2eq}/kg_{lactide}$ (Baltic states supply chains). The high GHG emissions savings in comparison with fossil-based counterparts are mainly attributed to the carbon neutrality advantage that are accounted as negative emissions from the embedded carbon in the materials. All conversion routes nevertheless report GHG emission savings if the carbon neutrality characteristic is omitted. The performance of the conversion routes is also strongly related to the GHG emissions generated to supply heat and electricity demand from these conversion routes. However, the impact from heat and electricity is bounded to the fossil inputs used to generate them. This impact can therefore be reduced significantly in the future with the reduction of fossil inputs. To illustrate, in 2014, 70% of the electricity used in the Netherlands was produced with fossil inputs; for 2035 it is expected that more than 50% of the electricity will be generated with renewable sources[266]. These changes can reduce considerably the impact from the conversion routes that have a higher electricity and heat demand (Brazil and Netherlands routes).





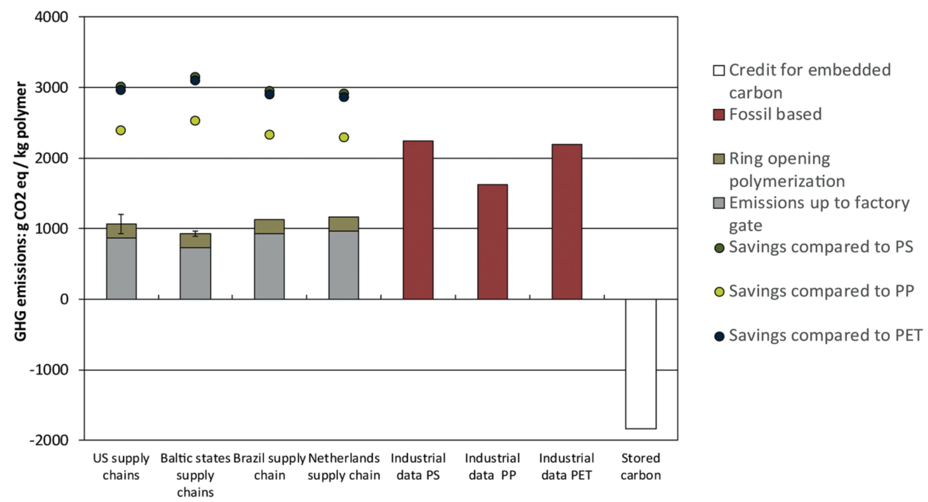


**Figure 3-7 'Cradle-to-factory-gate' GHG emissions of 1 kg lactide production in comparison with fossil-fuel counterparts.**

Figure 3-8 displays the comparison of 1 MJ ethanol production in accordance with RED II GHG calculation rules and the fossil-fuel counterpart. All conversion routes report GHG emissions savings in comparison with the fossil fuel reference established in RED II. Unlike bio-based materials, there are already established mandatory savings criteria for GHG emissions in biofuels. None of the conversion routes with the exception of SB from the Netherlands are able to comply with RED II GHG savings criteria for 2015 and 2021. The difference in performance between conversion routes is strongly related to the energy allocation rules in agreement with RED II and the different system designs of the SB and lignocellulosic conversion routes. For lignocellulosic conversion routes the burden between lactide, ethanol and by-products is spread with energy allocation. However, several by-products are allocated zero emissions as they are not energy carriers. This characteristic in combination with the high energy content of ethanol (in comparison with other outputs such as lactide) dictates that a large share of the total generated GHG emissions are allocated to ethanol. Considerable large improvements along the lignocellulosic biomass conversion routes are needed to reduce GHG emissions and meet the GHG emissions savings criteria from RED II.

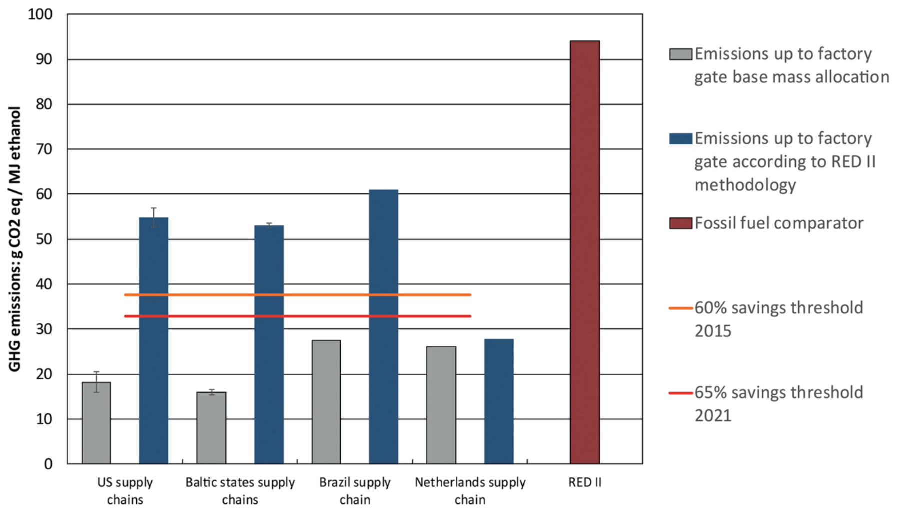


**Figure 3-8 'Cradle-to-factory-gate' GHG emissions of 1MJ ethanol production (in accordance with RED II GHG calculation rules) in comparison with fossil-fuel counterpart**

## 3.4 DISCUSSION

For illustrative purposes, the figures present in the discussion section exclude the net GHG emissions reported for lactide.

### 3.4.1 Implications of methodological choices

The LCA attributional approach adopted in this study is static, independent of a policy and economic context[267]. This approach is unsuitable to assess possible consequences of policy choices including the potential GHG mitigation impact of developing a biorefinery in the Netherlands. Important aspects that are not included are a counterfactual and marked-mediated impacts including emission from (indirect) land use change and possible carbon debt[34,268] (Indirect) land use change is most relevant for production systems that are land intensive such as the SB supply chain. Carbon debt and payback time are most relevant for the systems that use slow-growing biomass (SW and FR).

The fossil-based counterpart comparison exercise was based on cradle-to-factory gate emissions per kg of polymer. Nevertheless, the different density and the end-of-life of plastic materials shall be also incorporated in such exercise. Different material densities can lead to different weights within the same plastic application (e.g. a plastic cup). It can be more significant to compare bio-based materials using a volume reference; and account for plastic items produced by volume filling techniques such as injection molding. In terms of end-of-life, the incorporation of this stage in such comparator





exercises should be in line with policy legislation targets. An adequate comparator should include at least the GHG emissions from incineration to account for the carbon neutrality advantage of the bio-based polymer as done in Kikuchi et al.[214] To illustrate, if a policy has a target in terms of GHG emissions savings for 2030 and the percentage of plastic materials incinerated is forecasted to be e.g. 25% in the same year, the fossil fuel comparator should be: cradle-to-gate emissions plus 25% of the emissions that would be released by burning that polymer. This suggestion can be an adequate proxy even if it does not consider other possible end-of-life options; but the following consideration must be accounted for; (1) The near future infrastructure may not allow PLA recycling[269]; (2) there are environmental trade-offs in shifting from a recyclable (i.e. PP,PS and PET) to a compostable polymer[270]; and (3) there are large uncertainties about the landfilling emissions of PLA[271,272].

### 3.4.2 Impact of allocation methods

Different allocation procedures can be applicable to deal with multifunctionality and could significantly impact the results[268,273]. In this study, a mass balance allocation with an intermediate allocation step approach was determined as the most suitable allocation method for the system composition and is defined as base allocation (see above). To understand the impact of allocation procedures in multifunctional systems and determine the robustness of the results, a sensitivity analysis was carried out. For this approach, the burden of GHG emissions was allocated to the biorefinery outputs based on physical or other types of relationship between products; considering the conversion to lactide / ethanol as a unified process and not as separated streams (after C5 and C6 sugars division) as illustrated in Figure 3-1. Allocation was applied based on (1) the total mass outputs from the main products and by-products (mass allocation); (2) the energy content of products and by-products as suggested in RED II (energy allocation); and (3) the market value of the products and by-products (economic allocation). For illustrative purposes, the sensitivity allocation analysis was only carried out for the base cases displayed in Figure 3-2. The resulting allocation factors are reported in Table 3-8.

**Table 3-8 Mass, energy and economic allocation factors.**

| Feedstock | Product | Mass flow (kg/year) | Mass allocation factor (%) | LHV (MJ/kg) | Energy allocation factor (%) | Price (€/kg) | Economic allocation factor (%) |
|---|---|---|---|---|---|---|---|
| Wood | Lactide | 255.987.048,9 | 45,97 | 18,72[a] | 70.32 | 2,47[d] | 89,65 |
| | Ethanol | 63.747.939,9 | 11,45 | 26,81[b] | 26.23 | 0,66[e] | 6,03 |
| | Sodium acetate | 11.689,6 | 0,00 | - | - | 0,79[f] | 0,00 |
| | Gypsum | 205.757.911,6 | 36,95 | - | - | 0,008[g] | 0,23 |
| | Ammonium acetate | 21.976.441,3 | 3,95 | - | - | 0,84[h] | 2,61 |
| | Furfural | 9.410.719,4 | 1,69 | 23,98[c] | 3.45 | 1,1[i] | 1,47 |
| | Total | 556.891.750,8 | 100 | - | 100 | - | 100 |
| Bagasse | Lactide | 255.987.048,9 | 41,00 | 18,72[a] | 55.25 | 2,47[d] | 83,00 |
| | Ethanol | 122.332.237,7 | 19,31 | 26,81[b] | 39.54 | 0,66[e] | 10,71 |
| | Sodium acetate | 16.692,7 | 0,00 | - | - | 0,79[f] | 0,00 |
| | Gypsum | 205.757.911,6 | 32,48 | - | - | 0,008[g] | 0,22 |
| | Ammonium acetate | 31.382.358,1 | 4,95 | - | - | 0,84[h] | 3,45 |
| | Furfural | 18.059.161,9 | 2,85 | 23,98[c] | 5.21 | 1,1[i] | 2,61 |
| | Total | 633.535.410,9 | 100 | - | 100 | - | 100 |
| Sugar beet | Lactide | 255.987.048,9 | 51,62 | 18,72[a] | 90.29 | 2,47[d] | 99,47 |
| | Sodium acetate | 16.692,7 | 0,00 | - | - | 0,79[f] | 0,00 |
| | Gypsum | 205.757.911,6 | 41,49 | - | - | 0,008[g] | 0,26 |
| | Sugar beet pulp | 34.111.554,2 | 6,88 | 14,43[b] | 9.71 | 0,05[j] | 0,27 |
| | Total | 495.873.207,5 | 100 | | 100 | | 100 |
| Sugar beet | Ethanol | 181.144.115,4 | 84,15 | 26,81 | 90,80 | 0,66[e] | 98,62 |
| | Sugar beet pulp | 34.111.554,2 | 15,85 | 14,43[b] | 9,20 | 0,05[j] | 1,38 |
| | Total | 215.255.669,6 | 100 | - | 100 | - | 100 |

[a] assumed from PLA [247]
[b] [138]
[c] [248]
[d] Price of raw lactide [249]
[e] Average price of ethanol between 2016-2018 [250]
[f] Average price from sodium acetate reported in e-commerce webpage [251]
[g] [252]
[h] Average price from ammonium acetate reported in e-commerce webpage [253]
[i] [254]
[j] Average price from sugar beet pulp for the Netherlands [255]

Figure 3-9 displays the 'cradle-to-factory-gate' GHG emissions for 1kg of lactide and 1MJ of ethanol production for the base conversion routes when applying mass, energy, and economic allocation. The results show that the chosen allocation method has a strong impact on the GHG emissions of both lactide and ethanol production. The use





of mass allocation leads to a better GHG emissions performance for lactide and slightly worse performance for ethanol in comparison with the base approach used in the base cases. When mass allocation is applied, a larger share of GHG emissions is allocated to gypsum. Large quantities of gypsum are produced from the lactic acid recovery process, which results in a considerable mass output of gypsum from the system. This allocation procedure entails that the burden of GHG emissions is spread out more evenly between gypsum and lactide, and, to a lesser extent, to ethanol. This characteristic also applies for the base allocation method. However, with the base approach, emissions upstream from the factory gate are only allocated to the main products of the system. The production of gypsum is not considered a primary function of the system. Mass allocation can lead to inappropriate conclusions as large part of the GHG emissions burden from the lactide production and before conversion are allocated to gypsum.

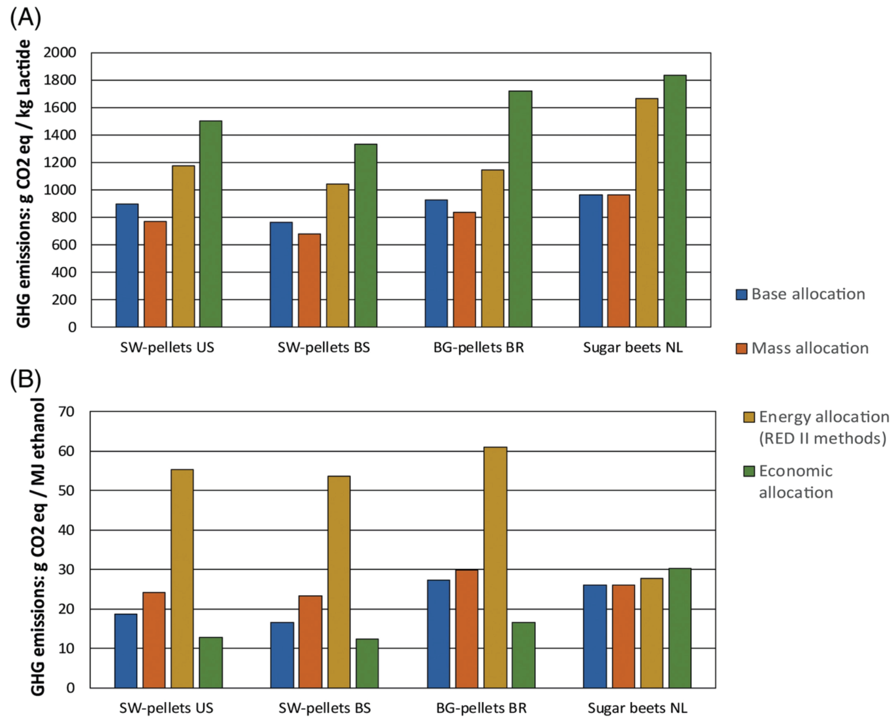


**Figure 3-9 Sensitivity analysis results for (A) lactide and (B) ethanol between different allocation procedures.**

Applying energy allocation as suggested by RED II leads to considerably higher GHG emissions for ethanol in all scenarios with the exception of the Netherlands SB case. When energy allocation is used for lignocellulosic biomass conversion routes, the burden of GHG emissions is spread only between lactide, ethanol, and furfural. The higher energy content of ethanol dictates that a larger share of GHG emissions is allocated to ethanol compared with other allocation methods. The relatively larger share of hemicellulose in BG entails that larger quantities of ethanol are produced in comparison to other conversion routes. This results in higher GHG emissions for ethanol in this conversion route as a result of the higher energy content of ethanol compared to lactide. A smaller share of emissions is allocated to lactide when energy allocation is applied compared to the base cases. Nevertheless, most of the GHG emissions from chemicals inputs at the conversion step and heat demand derive from lactide production. Thus, energy allocation over all outputs of the biorefinery without an intermediate allocation step leads to a considerable burden of GHG emissions from lactide production being allocated to ethanol. Still, RED II has no clear suggestions and solutions of how to avoid such inaccuracy in burden share for such multi-output systems and the results of such exercise with energy allocation are a shortcoming that can lead to inadequate conclusions.

The use of economic allocation shifts the burden to lactide and away from ethanol compared to the other allocation methods; with the exception of ethanol from the Netherlands conversion route. The higher price from lactide determines that a large share of the GHG emissions burden from the lactide / ethanol production are allocated to lactide. However, economic allocation is bound to the market volatility that leads to instability and price fluctuations. To illustrate, the market value for ethanol used in this study is a two-year average from 2016 to 2018 (0.66 €/kg); in this same period of time ethanol prices fluctuated between 0.83 €/kg and of 0.55 €/kg. Still, economic allocation reflects a higher degree of relatedness in terms of outputs and the system's primary function.

### 3.4.3 Inventory data limitations

No foreground data for the conversion from calcium lactide to lactide, electricity and heat demand was available. A stoichiometry and mass balance equation, as suggested by Vink and Davies, [262] was applied in order to calculate the amount of $Ca(OH)_2$ and $H_2SO_4$ required for the conversion process. Heat and electricity demand from the lactic acid to lactide conversion was approximated with a back end calculation following the results from Vink et al.[274] Vink and Davies.[262]The biorefinery for this study was assumed to produce 272 $kt_{lactide}/year$ However, current PLA/lactide facilities have a much lower capacity. NatureWorks corn based facility in Blair, Nebraska has a capacity of 150 $kt_{PLA}/year$. Meanwhile, Corbion recently finalized their sugarcane lactide plant expansion (100 $kt_{lactide}/year$) and a 75 $kt_{PLA}/year$ plant in Thailand .[275]





### 3.4.4 Improving hot-spots in GHG emissions performance

The assessment of the conversion routes is affected by the choice of input parameters such as transport modes and distances, choice of heat supply, yields and other assumptions about the life-cycle inventory. These parameters can strongly steer the magnitude of the results and the direction of the analysis. To understand the impact of these parameters, a sensitivity analysis was carried out with the base-case scenarios displayed in Figure 3-2. These conversion routes were selected for illustrative purposes. In addition, the variation of assumptions for the sensitivity analysis base-case scenarios has an equivalent impact if applied to the other conversion routes (e.g. alternative forest feedstock). The relevant parameters for the sensitivity analysis were selected based on their influence on the results (GHG hotspots in the supply chains). As a result, three processes along the conversion routes were selected for this analysis: (1) the conversion step; (2) transport of feedstock; and (3) cultivation (higher yields). The variations in inputs are applied with a view to superior GHG emissions performance for the selected processes and are based on plausible scenarios. The sensitivity conversion routes are defined as best cases.



For the conversion step (1), the majority of the GHG emissions result from the use of chemicals in the process. However, no distinct improvement potential could be identified with the limited information and data available. Improvements were therefore limited to alternative sources of heat supply. Different transport modes and shorter feedstock transportation distances were assumed for the feedstock transportation process (2), and higher yields were considered mainly for the cultivation process in the SB conversion route (3). Table 3-9 includes a summary of the parameters and description adopted for the sensitivity analysis base cases.

**Table 3-9 Sensitivity analysis parameters for best case scenarios.**

| Base cases | Parameter | Value | Unit | Description/assumption |
|---|---|---|---|---|
| US pellets – SW – NL C | Heat supply | 1.07[a] | $kg_{dry\ pellets}/kg_{dry\ pellets}$ | Heat demand supplied by burning additional (wood or bagasse) pellets at the CHP instead of natural gas |
| BS pellets – SW – NL C | | 1.07[a] | $kg_{dry\ pellets}/kg_{dry\ pellets}$ | |
| BR pellets – BG – NL C | | 1.20[a] | $kg_{dry\ pellets}/kg_{dry\ pellets}$ | |
| BR pellets – BG – NL C | Transport | 100 km by truck and 264 km by train | $km$ | Large scale bagasse pellets production could use the railway system in Sao Paulo state to transport pellets to port (COSAN, 2018). Sugarcane bagasse pellets are transported from pellet facilities by road to a storage/load terminal (2) from the terminal, pellets are loaded in freight trains and transported to the port of Santos. |
| SB NL C | Heat supply | 3.58[b] | $MJ_{steam}/kg_{lactide}$ | It is possible that sugar beet conversion facilities/refineries include a biogas plant to process waste slop and use the heat from this procedure (biogas) for the conversion process to lactide and ethanol |
| | Transport | 50 | $km$ | Sugar beet cultivation can increase in the proximity of the plant driven by the bio-refinery biomass demand. Therefore, it was considered a lower average transportation distance from biomass supply area to bio-refinery |
| | Yield | 120 | $t_{sugar\ beets}/ha$ | The suitability method applied in this paper to calculate spatially explicit sugar beet yields can achieve values as high as 120 t/ha (ideal conditions). Hence, this yield was considered for the cultivations stage. |

[a] calculation based on values from [246]
[b] calculation based on values from [138]





Figure 3-10 displays the comparison between the base supply chains with the best performing alternatives assessed in the sensitivity analysis. As shown in Figure 3-10(A), the best performance conversion routes have a reduction of 15 g $CO_{2eq}/kg_{lactide}$ (SW pellets from the US), 52 g $CO_{2eq}/kg_{lactide}$ (SW pellets from Baltic states), 276 g $CO_{2eq}/kg_{lactide}$ (BG pellets from Brazil) and 156 g $CO_{2eq}/kg_{lactide}$ (SG from the Netherlands) when compared with the base scenarios. There is a tradeoff between GHG emissions from logistics and conversion to lactide for the woody lignocellulosic biomass scenarios. The use of biomass to cover system heat demand as a natural gas replacement results in a reduction of GHG emissions at the conversion stage. However, the mobilization of additional biomass to supply heat demand generates an increase in logistics GHG emissions (Figure 3-10(B)). To illustrate, the reduction of 89 g $CO_{2eq}/kg_{lactide}$ in conversion to lactide GHG emissions from biomass heat supply for the SW pellets from the USA is offset by 74 g $CO_{2eq}/kg_{lactide}$ due to the increase of GHG emissions in logistics; the same GHG emissions tradeoff occurs for the SW pellets from the Baltic states.

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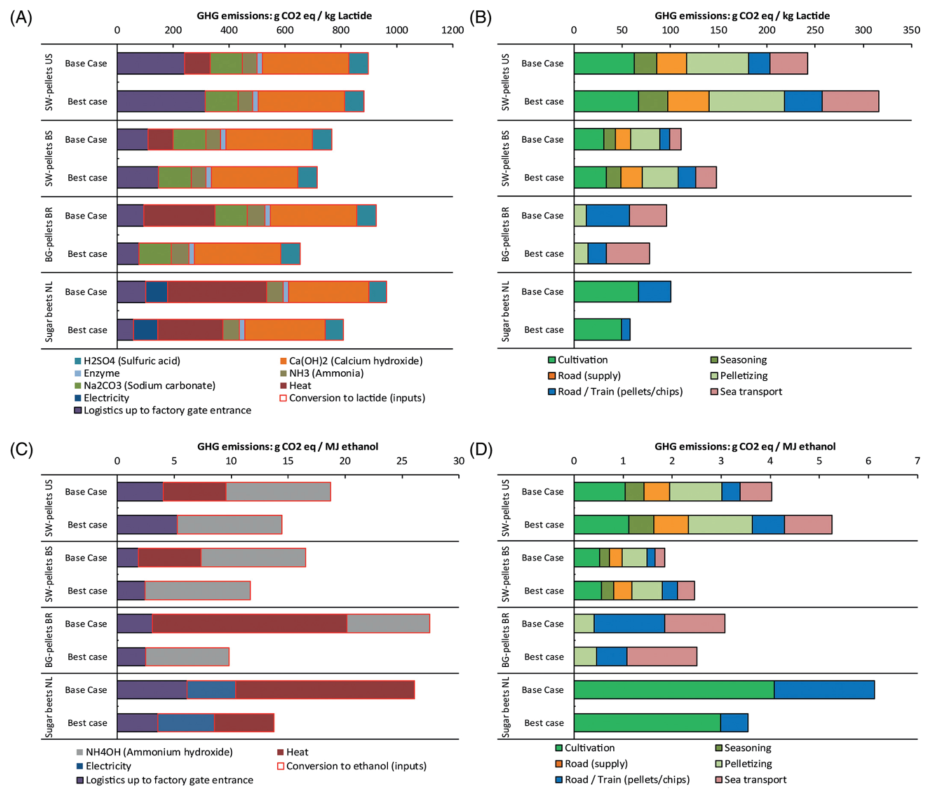


**Figure 3-10 Sensitivity analysis results for lactide with (A) and without conversion stage (B), and for ethanol with (C) and without (D) conversion stage. Best case assumptions include higher feedstock yields, more efficient transport chains and replacement of natural gas with biomass.**

For the best BG pellets from the Brazil scenario, the increase in emissions due to additional supplied biomass is counterbalanced by the use of trains for pellets transport. Transporting pellets by trains is more efficient than trucks and generates lower GHG emissions. To ensure this advantage, the location of the cargo terminal is of paramount importance for the structured transport of pellets from inner São Paulo state to the port of Santos. This requires important infrastructure developments. For the best SB from the Netherlands scenario, higher SB yields and shorter supply distance results in a reduction of 42 g $CO_{2eq}/kg_{lactide}$ in logistics up to the factory-gate entrance. To obtain a higher yield, besides adequate climate and soil conditions, an increase in agricultural inputs can be expected. An increase in such inputs could result in higher GHG emissions from SB production. The heat supply from a biogas plant can only cover to an extent the heat demand from the conversion to the lactide stage. However, the heat supply from the biogas plant reduces by 122 g $CO_{2eq}/kg_{lactide}$ the impact at this stage. The majority of GHG savings from the best case scenarios are related to the use of additional biomass for heat supply at the conversion stage with the highest improvement on the BG pellet from the Brazil conversion route.

For ethanol (Figure 3-10(C)), the GHG emissions from the best case scenarios are reduced between 6 g $CO_{2eq}/MJ_{ethanol}$ (SW pellets from the USA) and 21 g $CO_{2eq}/MJ_{ethanol}$ (BG pellets from Brazil). As with the lactide results, the replacement of natural gas with biomass to generate process heat induces the largest influence for GHG emissions reduction for the best case scenarios. It is worth noting that the worst performing base-case scenario (BG pellet from Brazil) becomes the best performing conversion route in terms of GHG emissions. Delivering heat with additional biomass for the ethanol conversion process for the Brazil supply chains results in a reduction of 17 g $CO_{2eq}/MJ_{ethanol}$. Likewise, the heat supply from a biogas plant reduces by 10 g $CO_{2eq}/MJ_{ethanol}$ the impact generated at the conversion to ethanol process from the SB supply chain. Overall, several of the considered improvements need to be assessed more extensively due to the possible cascade effects that they could induce. For instance, an increase of SB cultivation in the vicinity of the biorefinery can induce the displacements of other land-use types.

### 3.4.5 Impact of the choice of biorefinery location

The total GHG emissions 'from cradle to factory gate' can differ between conversion routes depending on the biorefinery location. To understand the impacts of the biorefinery location choice, alternative scenarios for the location of the biorefinery in the country of biomass supply were assessed with the base-case scenarios displayed in Figure 3-2 (only results from lactide are displayed for illustrative purposes). The biorefinery is considered to be located at the terminal from which biomass is exported in other supply chains. Lactide is transported by containers ships to Rotterdam and ethanol stays





in biomass supply country Figure 3-11 displays the comparison between the base-case conversion routes and routes when the biorefinery is located in the country of biomass supply. In comparison with lactide conversion in the Netherlands, the 'cradle-to-factory-gate' emissions for 1 kg of lactide production in the USA are 75 g $CO_{2eq}$ less. Greenhouse emissions from chemicals supply chains are relative similar with the exception of sulfuric acid and sodium carbonate. Mining of sulfur and production of sulfuric acid and sodium carbonate in the USA are less energy intensive than in Europe[276]; this results in lower GHG emissions from the conversion to lactide step in the USA. There are small variations in GHG emissions from the use of natural gas to provide heat demand and upstream emissions from other used chemicals. When the biorefinery is located in the Baltic states (Latvia), there is almost no difference between GHG performance. When the biorefinery is located in Brazil and BG is used as feedstock, higher GHG emissions are generated from the use of natural gas for heat supply (the extraction of natural gas in Brazil is more energy intensive). Overall, the 'cradle-to-factory-gate' emissions for 1 kg of lactide increase by 95 g $CO_{2eq}$ between locating the biorefinery in Brazil or the Netherlands.



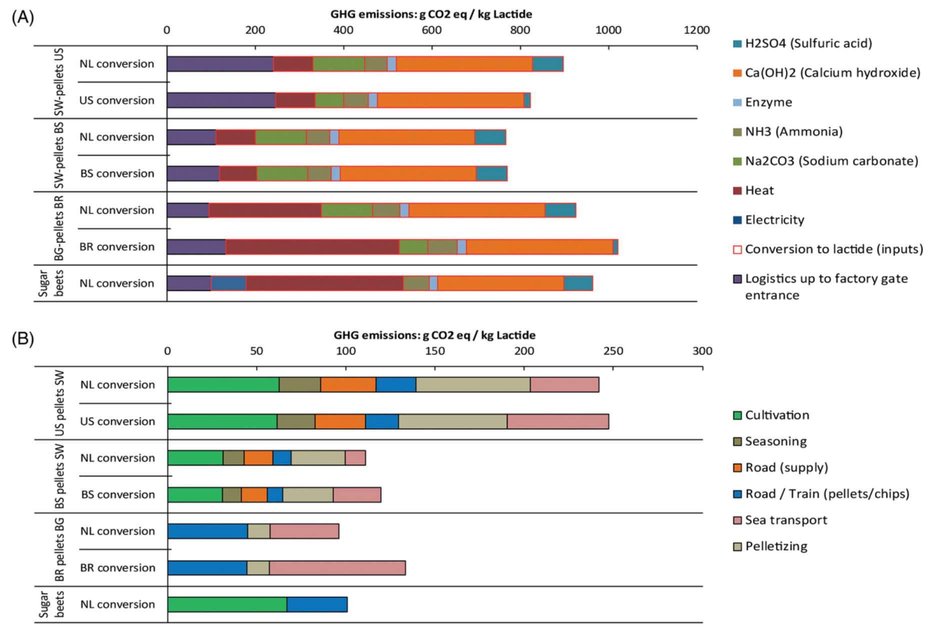


**Figure 3-11 Comparison between GHG emissions when locating the biorefinery and producing lactide at the country of biomass source to the Netherlands with (A) and without (B) conversion stage.**

For upstream emissions (Figure 3-11(B)), the main difference between locating the biorefinery in the country of biomass source or in the Netherlands is the mode of sea transport. Sea transport of pellets has a better GHG performance as a result of bulk transportation mode than lactide transportation in containers. Transporting lactide from biomass supply countries results in an increase of 18 g $CO_{2eq}/kg_{lactide}$ for the US conversion route, 15 g $CO_{2eq}/kg_{lactide}$ for the Baltic states conversion route and 38 for the Brazil conversion route. However, on average, differences in sea transportation mode are small and have little impact when compared to the conversion step. Overall, it is concluded that the location of the biorefinery in either the country of biomass source or the Netherlands has little impact on overall GHG emissions performance when considering the total 'cradle-to-factory-gate' GHG emissions from the conversion routes.

### 3.4.6 System comparison

The differences in scope, system design, and functional unit poses limitations for comparison with other multi-output biorefinery studies such as Mandegari et al.[277] and Farzad et al.[278] Nevertheless, the GHG footprint of the individual outputs of the biorefinery to lactide and ethanol are compared with single-output processes as show in Table 3-10. For comparison purposes, the system boundaries were extended with the conversion of lactide to PLA by adding the emissions from the ring opening polymerization of lactide for (PLA) production as reported in Vink and Davies.[262] On general basis, in literature, different results are reported for lactide and similar for ethanol (see: Literature comparison in the supplementary material)





**Table 3-10 Literature comparison.**

| Feedstock | Supply country | End product | GHG performance | Unit | Source |
|---|---|---|---|---|---|
| Wood biomass | US | Lactide | -1096 – -828 (+200)[a] | $g\ CO_{2eq}/kg_{lactide}$ | This study |
| Wood biomass | Baltic states | Lactide | -1138 – -1063 (+200)[a] | $g\ CO_{2eq}/kg_{lactide}$ | This study |
| Sugarcane bagasse | Brazil | Lactide | -904 (+200)[a] | $g\ CO_{2eq}/kg_{lactide}$ | This study |
| Sugar beets | Netherlands | Lactide | -866 (+200)[a] | $g\ CO_{2eq}/kg_{lactide}$ | This study |
| Corn | US | PLA | 2023 | $g\ CO_{2eq}/kg_{PLA}$ | [274] |
| Corn | US | PLA | 620 | $g\ CO_{2eq}/kg_{PLA}$ | [262] |
| Corn | US | PLA | 1303 | $g\ CO_{2eq}/kg_{PLA}$ | [279] |
| Sugarcane | Thailand | PLA | 500-800 | $g\ CO_{2eq}/kg_{PLA}$ | [280] |
| Sugarcane/corn | Thailand/US | PLA | 590[b] | $g\ CO_{2eq}/kg_{PLA}$ | [272] |
| Wood biomass | US | Ethanol | 16 – 20.4 | $CO_{2eq}/MJ_{ethanol}$ | This study |
| Wood biomass | Baltic states | Ethanol | 15.3 – 16.5 | $CO_{2eq}/MJ_{ethanol}$ | This study |
| Sugarcane bagasse | Brazil | Ethanol | 27.4 | $CO_{2eq}/MJ_{ethanol}$ | This study |
| Sugar beets | Netherlands | Ethanol | 26.1 | $CO_{2eq}/MJ_{ethanol}$ | This study |
| Eucalyptus | Spain | Ethanol | 208.9[c] | $CO_{2eq}/MJ_{ethanol}$ | [281] |
| Black locust | Spain | Ethanol | 155.51[c] | $CO_{2eq}/MJ_{ethanol}$ | [281] |
| Poplar | Italy | Ethanol | 182.76[c] | $CO_{2eq}/MJ_{ethanol}$ | [281] |
| Sugar beet | - | Ethanol | 30.8 | $CO_{2eq}/MJ_{ethanol}$ | [138] |
| Sugarcane | - | Ethanol | 28.1 | $CO_{2eq}/MJ_{ethanol}$ | [138] |
| Eucalyptus | - | Ethanol | 37.31 | $CO_{2eq}/MJ_{ethanol}$ | [138] |
| Forest residue | - | Ethanol | 26.62 | $CO_{2eq}/MJ_{ethanol}$ | [138] |
| Straw | - | Ethanol | 13.7 | $CO_{2eq}/MJ_{ethanol}$ | [138] |

[a] 200 g $g\ CO_{2eq}/kg_{PLA}$ for ring opening polymerization
[b] Average from corn and sugarcane
[c] Cradle to gate values. Converted from $CO_{2eq}/MJ_{ethanol}$. These values omit for comparison purposes the $CO_2$ binding in the dry matter while feedstock is grown which is deducted.

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The multi-output/parallel production characteristic from biorefineries results in considerable lower GHG emissions over stand-alone conversion systems of biomass to lactide (or similar) and stand-alone conversion systems to ethanol; Biomass utilization is optimized for the production of different marketable products with a lower carbon footprint for each product in comparison to values reported in literature. The multi-output design determines that the burden of emissions is divided accordingly to a feedstock biochemical composition (mainly between lactide and ethanol); instead, for one main product output biorefineries the whole burden of GHG emissions is assigned to one product as ethanol production from wood biomass as reported in González-García et al.[281] or PLA production reported in Vink and Davies.[262] Nevertheless, it is suggested that for such advantage of multi-output biorefineries to be in place, the use of lignocellulosic biomass feedstocks with high lignin content is crucial, as this is typically used to cover internal heat demand.

## 3.5 CONCLUSION

This study compared the GHG footprint of different supply chains design for a biorefinery that uses either imported lignocellulosic biomass or SB from the domestic supply for the production of bio-based chemicals (lactide) and biofuels (ethanol) in the Netherlands. The highest performance of the pathways considered corresponds with SR pellets from the Baltic states with 692 g $CO_{2eq}/kg_{lactide}$ when not considering the embedded carbon up to the factory gate and net GHG emissions of -1138 g $CO_{2eq}/kg_{lactide}$ and 15 g $CO_{2eq}/MJ_{ethanol}$ . In terms of feedstock GHG emissions, SR and sugarcane BG outperform other feedstock types. Other feedstock types are characterized by higher GHG emissions from agricultural inputs and fuel-use for agricultural machinery. Pelletization is preferable over long-distance overseas transport of wood chips due to the increased bulk density and improved handling characteristics further downstream in the supply chain. In terms of supply regions, it has been shown that in the Baltic states (Latvia), adequate biomass supply, the relatively short transport distance between biomass sources and pelletization facilities, and the shorter overseas transport distance between the port of Riga and Rotterdam result in a lower GHG footprint compared with the intercontinental supply chains (Brazil, US southeast) and the national (Netherlands) supply chain. This output challenges the general assumption that local-sourced biomass attains higher GHG emissions by default; it has been shown that the explicit GHG performance depends largely on specific feedstock characteristics and supply-chain configurations. For the conversion stage to the lactide and ethanol stage, process heat requirements and upstream emissions from the production of chemicals used in the conversion process contribute substantially to the overall GHG footprint. Lignin is used to generate process heat that is otherwise generated from natural gas. The implication of this supply chain





design assumption is that wood supply chains, with a higher lignin content compared to bagasse, have a better GHG emissions performance at the conversion stage; this is valid for the analyzed supply chains designs. Processes from the upstream conversion stage have been demonstrated to have a relatively small impact share on the total GHG emissions conversion routes.

All conversion routes, with the exception of the SB Netherlands, are unable to comply with the strict GHG emissions criteria for biofuels in RED II. However, the GHG emissions calculations in line with RED II do not consider the challenges of burden allocation in multi-output systems. This shortcoming largely affects ethanol GHG emissions outcomes when calculated with RED II criteria. Overlooking the challenges of burden allocation can discourage theproduction of renewable forms of energy in multi-output systems. In addition,it can constrain the incorporation of highly efficient systems and conversion routes that can support to maximize energy security and reduce GHG emissions. For lactide, all conversion routes report high savings when compared with relevant fossil-based counterparts. The high savings are mainly attributed to negative emissions from the imbedded carbon in the materials. The end-of-life phase from bio-based and fossil-based materials or the embedded carbon in bio-based materials needs to be accounted for to give an adequate comparison. The end-of-life phase from the products subjected to comparison between bio-based and fossil-based can have major repercussions for the total GHG emissions performance and possible policy outcomes. Therefore future research should focus on exploring the complexity from this stage related to plausible end-of-life scenarios.



The results have been tested against alternative supply-chain configurations with the biorefinery located in the country of biomass supply, improvements along the supply chains' GHG hotspots, and different allocation choices. Locating the biorefinery in the USA results in a better GHG performance for the conversion stage than in Brazil or the Baltic States due to the upstream emissions from chemicals supply. However, the net advantage over the total life cycle ('cradle to factory gate') is small. It can therefore be concluded that the development of an integrated biorefinery in the Netherlands, instead of importing final commodities such as ethanol or chemicals, is a viable strategy. The GHG emissions performance from the supply chains can be further improved mainly by utilizing biomass as fuel to supply heat demand from conversion to the lactide / ethanol process instead of natural gas.

Multifunctional systems are highly sensitive to allocation procedures. Next to variations from actual supply chain design variations, the chosen method to allocate GHG emissions over the multiple outputs of the biorefinery results in large variations in the total GHG footprint. For multi-output systems we recommend using a multi-level allocation method as a base and apply different allocation methods to explicate the impact of methodological choices on the results.

Future research should focus on technologies / processes that allow for a reduction in the consumption of process chemicals in the production of lactide. Given the level of maturity, biorefinery systems and bio-based chemicals still have large development potential in terms of environmental performance by technological development. Allocation guidelines thatconsider the challenges in allocating the burden of impacts for multifunctionalsystems should be developed.

## 3.6 ACKNOWLEDGEMENTS

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## 3.7 SUPPLEMENTARY MATERIAL

**Table S 3-1 Dry matter losses**

| Process | Dry matter loss (%) |
|---|---|
| *Wood* | |
| Losses at the landing site | 0.5[a] |
| Losses at seasoning | 5[a] |
| Losses during chipping or debarking | 2[b] |
| Losses transport forest to pellet mills | 1[c] |
| Losses transport sawmill to pellet mills | 1[c] |
| Losses during pelleting | 2[c] |
| Losses transport pellet mill to ports | 1[c] |
| Losses trans-ocean shipping | 2[c] |
| *Sugarcane bagasse* | |
| Losses recollecting | 0.5[d] |
| Losses during transportation | 1[d] |
| Losses during pelleting | 2[d] |
| Losses trans-ocean shipping | 2[d] |
| *Sugar beets* | |
| Losses recollecting | 0.5[d] |
| Losses during transportation | 1[d] |

a 282
b 283
c 284
[d] Assumed as a proxy indicator from wood scenarios

**Table S 3-2 Feedstock biophysical composition**

| Feature | Woody biomass (% dry basis) | Sugarcane bagasse (% dry basis) | Sugar beet (% dry basis) |
|---|---|---|---|
| Cellulose | 48[a] | 33.6[b] | - |
| Hemicellulose | 21.6[a] | 29[b] | - |
| Lignin | 26.8[a] | 18.5[b] | - |
| Sucrose | - | - | 64[c] |
| Other | 3.6[a] | 18.9[b] | 36[c] |

a 246
b 285
[c] assuming 0.75 of water content [138]

### 3.7.1 Literature comparison

The use of lignocellulosic biomass for conversion to lactide process has an advantage over comparison with corn and sugarcane systems (lignin burning in the CHP). When corn and sugarcane are utilized, the demand for steam is supplied by natural gas and electricity from the grid. [262,272,280] For ethanol, Edwards et al.[138] reports similar results to the values from this study for the ethanol from woody biomass. However, that report assumes eucalyptus and forest residues from poplar plantations for the conversion process. Sugar beet conversion pathway GHG emissions performance is similar to that reported in Edwards et al.[138] Differences are attributed primarily to the calculation of $N_2O$ field emissions from the use of fertilizers; these emissions were calculated with the IPCC tier 1 methods instead of the Global Nitrous Oxide Calculator (GNOC) from the European Commission.

Fermentable sugars that are extracted from sugar cane and sugar beet have similar characteristics. Therefore, it was presumed that the SB scenario conversion stage to lactide GHG performance was in line with Groot and Borén,[280] and COWI A/S and Utrecht University.[272] However, there is a difference in GHG performance that can be explained by the methodological choices made for the sugar beet conversion system. The energy demand for pre-processing and sugar extraction of sugar beet was assumed to be the similar to those for the sugar beet to ethanol production system in Edwards et al.[138] Differences in emissions occur at the pre-processing and sugar extraction steps within the conversion to lactide process. Improved processes that are less energy intensive for the pre-processing and sugar extraction of this type of feedstock adopted in Edwards et al.[138] can explain the variation. In addition, IfBB,[286] reports a lower weight ratio of sugarcane/lactide to SB/lactide. This ratio difference entails that more resources are used at pre-processing stage to produce the same amount of lactide from sugarcane.



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# 4

## **Supply potential of lignocellulosic energy crops grown on marginal land and greenhouse gas footprint of advanced biofuels** - A spatially explicit assessment under the sustainability criteria of the Renewable Energy Directive Recast

**Published as:** *Vera, I., Hoefnagels, R., Junginger, M., & van der Hilst, F. (2021). Supply potential of lignocellulosic energy crops grown on marginal land and greenhouse gas footprint of advanced biofuels - A spatially explicit assessment under the sustainability criteria of the Renewable Energy Directive Recast. GCB Bioenergy, 13(9), 1425-1447.*

## ABSTRACT

Advanced biofuels produced from lignocellulosic crops grown on marginal lands can become an important part of the European Union (EU) climate change mitigation strategy to reduce $CO_2$ emissions and meet biofuel demand. This study quantifies spatially explicit the availability of marginal land in the EU, its production biomass potentials for eight different crops, and the greenhouse gas (GHG) performance of advanced biofuel supply chains. Available land is mapped based on land marginality and Renewable Energy Directive Recast (REDII) land-related sustainability criteria. Biomass potentials are assessed with a water-use-to-biomass-production approach while considering the available land, location-specific biophysical conditions and crop-specific phenological characteristics. The GHG balance of advanced biofuels from energy crops produced on marginal lands is assessed considering both land-related carbon stock changes and supply chain emissions with the carbon footprint approach from the REDII. Available marginal land that meets REDII criteria is projected at 20.5–21 Mha 2030 and 2050, respectively. Due to biophysical limitations, not all available land is suitable for energy crop production. The maximum biomass potential of lignocellulosic energy crops (optimal crop choice with maximum yield for each available location) varies between 1951 PJ year$^{-1}$ in 2030 and 2265 PJ year$^{-1}$ in 2050. The GHG emission performance (net emissions) of different advanced biofuel supply chains varies on average between -32 g $CO_2$-eq $MJ_{fuel}^{-1}$ for poplar/willow diesel to 38 g $CO_2$-eq $MJ_{fuel}^{-1}$ for reed canary grass renewable jet fuel. The large variability in GHG performance is strongly determined by the spatial heterogeneity, which dictates the type of feedstock produced under specific local biophysical conditions, the crop characteristics, and the best conversion pathway. Negative GHG emissions are related to increased carbon stocks for the biomass and soil organic carbon pools compared to the land prior to conversion. When for each location, the advanced biofuel supply chain with the highest GHG performance (lowest net GHG emissions) is selected, 618 PJ year$^{-1}$ of advanced biofuels can be produced by 2030. Under REDII GHG emission criteria, slightly less (552 PJ year$^{-1}$) is viable. Smart choices on location, crop type and supply chain design are paramount to achieve maximum benefits of bioenergy systems.






## 4.1 INTRODUCTION

The EU has set ambitious GHG emissions reduction targets to mitigate climate change impacts. Becoming climate-neutral by 2050 will require the deployment of various renewable energy sources [287]. Although biomass for energy purposes has been constantly debated [288], it is expected that biomass will contribute considerably to the decarbonization of the energy system in the years to come [26]. In some sectors, such as heavy transport and aviation, biomass's role as a decarbonization strategy is more relevant given the few alternatives [289]. In Europe, REDII has established a minimum 14% share of renewable energy within the final consumption of energy in the transport sector for 2030 (9% in 2019) and included a 7% cap on food and feed crop-based biofuels [6,15]. These conditions are set with sights on a double objective. First, phase out food and feed crop-based biofuels, currently dominating the biofuels sector [290]. Second, stimulate the development of lignocellulosic biomass-based advanced biofuels [291]. By 2030, at least 3.5% of renewable energy in the transport sector must be derived from advanced biofuels and biogas (0.2% in 2019) [15,290]. It is projected that between 9 to 29 Mha will be used to produce lignocellulosic energy crops [289]. Therefore, a significant increase in biomass production is expected [292]. However, current biomass production volumes are limited and far from meeting bioenergy requirements [293]. The current land dedicated for lignocellulosic energy crop production stands at 0.09 Mha [289].

There are still many sustainability concerns associated with large-scale biomass production for energy purposes. One of the main concerns is related to the net carbon fluxes when land is dedicated to biomass production [294]. It is crucial for biomass production systems to avoid competition with other land-based services such as food production and nature conservation [15,295]. Generally, lignocellulosic energy crops require fewer inputs (e.g., fertilizers) than food-based energy crops and can deliver higher yields in less suitable conditions (e.g., marginal lands) [296,297]. There is also a lower risk for biomass to compete with other land-based services [298]. In addition, lignocellulosic energy crops production on marginal lands could contribute to carbon sequestration, land restoration, limit soil erosion and enhance rural development [20–22]. Accordingly, utilizing marginal lands for lignocellulosic energy crops can be a promising option to source sustainable biomass. It can minimize competition for land and reduce negative environmental impacts such as Land Use Change (LUC)-related GHG emissions [23]. Therefore, quantifying lignocellulosic biomass produced in marginal lands can help understand the actual potential available for bioenergy systems, their contribution to renewable energy targets, and their environmental performance.

The bioenergy sector's potential to reduce GHG emissions depends on the interaction between land availability, sustainable biomass production, efficient logistics, conversion, and use [48,58]. Land availability for dedicated biomass production is limited by current and future land-use dynamics, which are steered by economic, demographic, and political drivers [49]. In addition, the suitability of such land for biomass production is conditioned by a wide range of local biophysical conditions (e.g., soil type, climate and previous land use) that can vary across space and time [44]. In combination with supply chain characteristics such as feedstock-technology conversion efficiency, these conditions determine the GHG performance of bioenergy systems to a large extent. For example, LUC-related GHG emissions are determined by location-specific land-use transitions and related changes in carbon stocks [25]. Therefore, GHG emissions are location-specific and vary across feedstock types and conversion technologies. These parameters should be considered when assessing these systems' performance to enable smart choices on locations and specific conversion technologies. This configuration can upgrade advanced biofuels production towards GHG emission reduction targets.

Several studies have quantified land availability and projected the potential of lignocellulosic energy crop production for Europe under different sustainability criteria [34,59–62], and few have coupled those projections with bioenergy supply chains to assess GHG emissions performance [63–65]. However, most of these studies have a limited time frame towards 2030. It is crucial to extend these projections up to 2050 with sights on EU climate change mitigation targets. Also, they are restricted to some extent when considering the effect of local biophysical conditions in biomass potentials and GHG emissions performance [299]. For example, several of these studies consider (and some lack) the spatial difference between biophysical conditions on a NUTS2/3 level and fall short of providing a high-resolution assessment. In addition, several studies do not cover the updated sustainability criteria entirely set in REDII and focus on surplus/unused agricultural land. This surplus/unused land is still productive and can potentially be re-used for agriculture purposes or other land-based services. Therefore, to understand the actual contribution of advanced biofuels in climate change mitigation targets, a high-resolution assessment that combines REDII sustainability criteria, location and temporal specific biophysical characteristics, marginal land, relevant lignocellulosic biomass types and conversion technologies is required.

This paper aims to assess the potential of woody and herbaceous energy crops grown on marginal land under REDII sustainability criteria and assess the GHG footprint of advanced biofuels produced from these crops in Europe. To that end, a three-step spatially explicit approach is applied. Land availability is quantified based on the available marginal land that meets REDII sustainability criteria and could therefore be used for energy crop production. Biomass potentials are determined for lignocellulosic energy crops that can be cultivated on the available marginal land, considering the location-specific agro-ecological suitability. The GHG performance of advanced biofuel supply chains is assessed considering the LUC-related GHG emissions and the specific characteristics of the advanced biofuel supply chains.





## 4.2 MATERIALS AND METHODS

### 4.2.1 Lignocellulosic energy crops and advanced biofuel supply chains

In this study, we include eight different lignocellulosic energy crops: Miscanthus, switchgrass, giant reed, reed canary grass, cardoon, poplar, willow and eucalyptus. These feedstock types are selected because they are considered the most representative energy crops in Europe [50]. Information on the assumed cultivation regime and crop-specific characteristics are found in Table S 4-1 in the supplementary material.

Eight advanced biofuel pathways to produce ethanol, Renewable Jet Fuel (RJF), gasoline, diesel, methanol and dimethyl ether (DME) are considered. The conversion technologies are selected since they are already available or close to commercialization [300,301]. Not all lignocellulosic energy crops are directly suitable to be combined with all conversion technologies. The biomass feedstock types' chemical and physical characteristics determine the suitability for (bio) chemical and thermal conversion. The relatively high chlorine content and low calorific value of herbaceous crops result in less suitable feedstock for thermal conversion than woody crops without pre-processing [302]. Therefore, in this study, it is assumed that herbaceous crops are only used for (bio) chemical conversion systems. Both herbaceous and woody lignocellulosic energy crops can be converted to ethanol by fractionation and hydrolysis. Ethanol can also be upgraded to produce RJF. Woody lignocellulosic energy crops can undergo a pyrolysis process followed by upgrading to produce a wide range of pyrolysis-oil-derived advanced biofuels as RJF, heavy fuel oil (HFO), gasoline and diesel. In this study, gasoline is defined as the main output of pyrolysis upgrading. Woody lignocellulosic energy crops can also be gasified. The derived syngas can go through Fischer-Tropsch synthesis to produce diesel or through a chemical synthesis to generate either methanol or dimethyl ether (DME). Figure 4-1 shows the considered advanced biofuel pathways for a total of 28 supply chains:

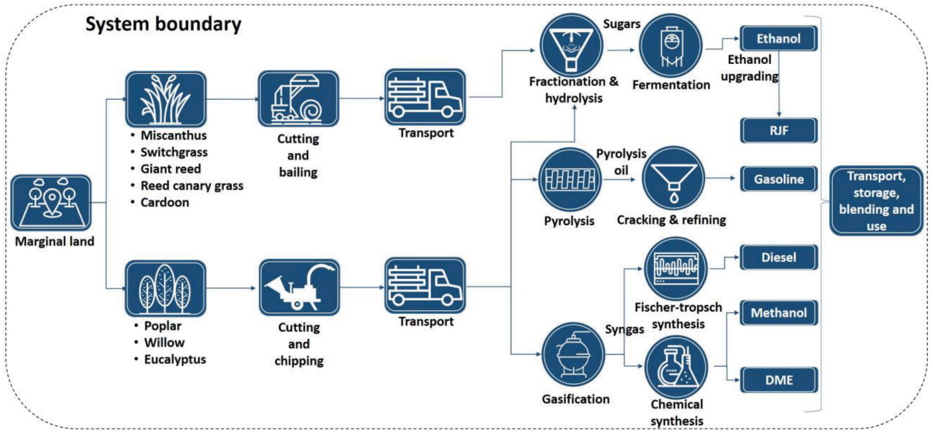


**Figure 4-1. Advanced biofuel pathways from lignocellulosic energy crops included in this study**

### 4.2.2 General approach

The available land and biomass potentials of lignocellulosic energy crops on marginal lands in Europe are assessed for 2030, 2040 and 2050. Three time steps are selected to analyze the developments over time. The GHG balance of advanced biofuel production pathways assessment in Europe is limited for 2030. This process is carried out to show the combined effect of the spatial variation in biomass potential, carbon stocks and the supply chain characteristics on the overall GHG balance of advanced biofuels. For biofuel production beyond 2030, it would require a detailed assessment of future technology change beyond the article's scope. The geographical scope is limited to the EU and the United Kingdom, and excluding the island countries Cyprus and Malta that have a relatively small domestic bioenergy crop potential [224]. The assessment is conducted at 1-km$^2$ spatial resolution while considering the spatiotemporal heterogeneity in biophysical conditions. R version 3.5.2 (2018-12-20) was used to carry out all the assessments and ArcMap 10.6.1 for map development. A three-step approach was applied, and it is depicted in Figure 4-2:

I. Land availability for lignocellulosic energy crops' potential production is mapped according to land use/cover data [303,304], land marginality [305] and REDII sustainability criteria [15]. All marginal land that meets REDII sustainability criteria and is not restricted by artificial or natural constraints is considered available (see section 4.2.3).

II. Lignocellulosic energy crops biomass potentials are assessed for eight different lignocellulosic energy crops considering the available land, location-specific biophysical conditions and crop-specific phenological characteristics. Crop biomass potentials are assessed following the methods in Ramirez-Almeyda et al. (2017) and considering the location-specific suitability of each crop (Perpiña Castillo et al. 2015). According to climate change scenarios, the change in climatic parameters is considered to simulate future biomass potentials (see section 4.2.3).

III. The GHG balance of advanced biofuels from lignocellulosic energy crops produced on marginal lands is assessed considering both LUC emissions and supply chain emissions. The GHG emissions are assessed following the methods in European Commission (2018b) for the eight lignocellulosic energy crops in combination with six different types of conversion technologies to produce advanced biofuels. Also, it was determined whether these advanced biofuels comply with the REDII GHG savings criteria (see section 4.3).

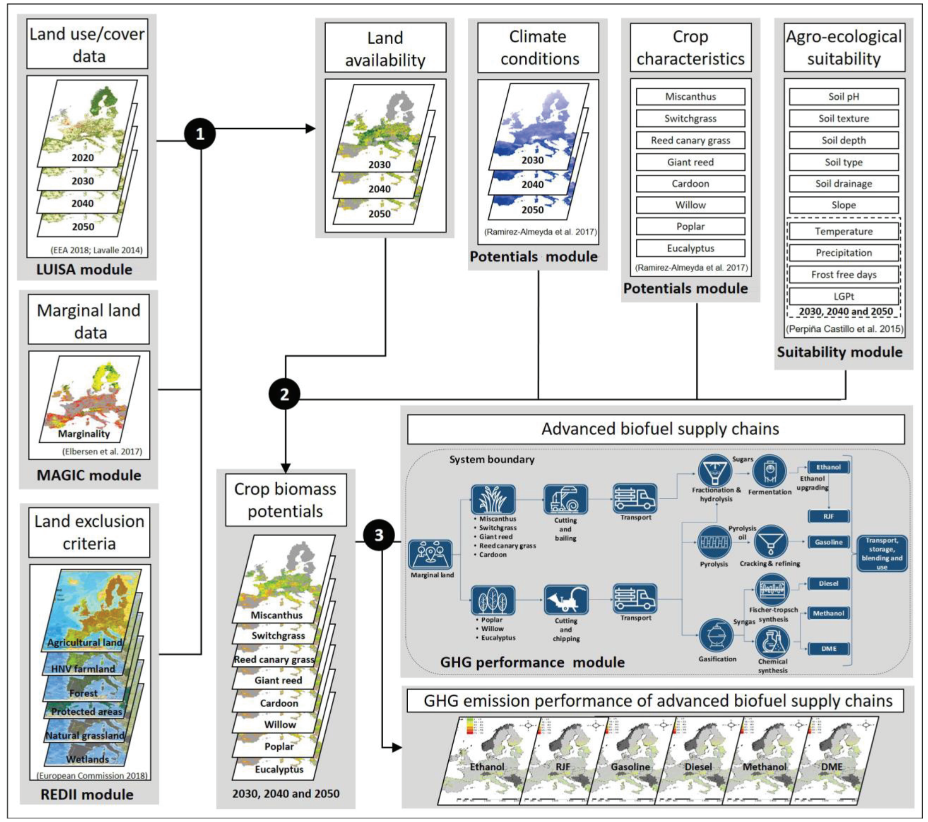


**Figure 4-2. Three-step methods approach**

### 4.2.3 Land availability

The amount of land dedicated to lignocellulosic energy crop production is determined for 2030, 2040 and 2050. Available land was selected based on two different criteria, REDII sustainability criteria and land marginality. Table 4-1 summarizes the REDII sustainability criteria as laid down in the directive and how these criteria are translated into land exclusion parameters. It also describes each of these land exclusion parameters and their data sources. Marginal land data is retrieved from the Marginal Lands for Growing Industrial Crops (MAGIC) project [305]. In this dataset, marginal lands are mapped with a two-step approach based on biophysical constraints and land management factors. Then, it classifies them based on socio-economic constraints, ecosystem services and threats. For this study, the classification of marginal land is not relevant. In addition, the land management factors applied to marginal land mapping are tailor to agricultural land. However, agricultural land is excluded in this study for the potential production of lignocellulosic crops. Therefore, the dataset applied in this study corresponds to the marginal land map based on six different biophysical constraints such as adverse climate or adverse chemical conditions, as established in Elbersen et al. (2017). Besides land marginality and REDII sustainability criteria, land use/cover with natural or artificial constraints such as water, urban areas and bare rock is also excluded.





**Table 4-1 Translation of REDII sustainability criteria into land exclusion parameters**

| **REDII Sustainability criteria[A]** | **Translation of REDII sustainability criteria into land categories to be excluded** |
|---|---|
| REDII Article 2, definition 37 | |
| Reducing the risk of indirect land-use change:<br>"low indirect land-use change-risk biofuels, bioliquids and biomass fuels' means biofuels, bioliquids and biomass fuels, the feedstock of which was produced within schemes which avoid displacement effects of food and feed-crop based biofuels, bioliquids and biomass fuels through improved agricultural practices as well as through the cultivation of crops on areas which were previously not used for cultivation of crops, and which were produced in accordance with the sustainability criteria for biofuels, bioliquids and biomass fuels laid down in Article 29" | Exclusion of all land dedicated to food, feed and fibre production to avoid the risk of indirect land-use change. All land categorized as "Agriculture" in the land cover data set is excluded, including[B]:<br>• Non-irrigated arable land<br>• Permanently irrigated land<br>• Rice fields<br>• Vineyards<br>• Fruit trees and berry plantations<br>• Olive groves<br>• Pastures<br>• Annual crops associated with permanent crops<br>• Complex cultivation patterns<br>• Land principally occupied by agriculture, with significant areas of natural vegetation<br>• Agro-forestry areas |
| REDII Article 29: Sustainability and greenhouse gas emissions saving criteria for biofuels, bioliquids and biomass fuels | |
| Paragraph 2:<br>"Biofuels, bioliquids and biomass fuels produced from agricultural biomass... shall not be made from raw material<br>obtained from land with a high biodiversity value..." | Exclusion of land with a High Nature Value, targeted to HNV farmland[C]:<br>• Farmland with a high proportion of semi-natural vegetation.<br>• Farmland with a mosaic of low-intensity agriculture and natural and structural elements.<br>• Farmland supporting rare species or a high proportion of European or World populations. |
| Paragraph 3, subsection a and b:<br>"...primary forest and other wooded land, namely forest and other wooded land of native species... "<br>"...highly biodiverse forest and other wooded land which is species-rich and not degraded, or has been identified as being highly biodiverse by the relevant competent authority..." | Exclusion of land categorized as "Forest" in the land cover data set, including[B]:<br>• Broad-leaved forest<br>• Coniferous forest<br>• Mixed forest |

Table 4-1 Translation of REDII sustainability criteria into land exclusion parameters (continued)

| REDII Sustainability criteria[A] | Translation of REDII sustainability criteria into land categories to be excluded |
|---|---|
| Paragraph 3, subsection c:<br>"...areas designated by law or by the relevant competent authority for nature protection purposes..."<br>"...For the protection of rare, threatened or endangered ecosystems or species recognised by international agreements or included in lists drawn up by intergovernmental organisations or the International Union for the Conservation of Nature..." | Exclusion of all areas dedicated to achieve long-term conservation of nature with its associated ecosystem services and cultural values. Exclusion of all areas designated with a protected status of protected under European policy, mainly[D]:<br>• The Natura 200 network<br>• The Emerald network of areas of special conservation interest<br>• Wetlands under the Ramsar convention |
| Paragraph 3, subsection d:<br>"...highly biodiverse grassland...namely grassland that would remain grassland in the absence of human intervention..." | Exclusion of land categorized as "Natural grasslands" in the land cover data set[B] |
| Paragraph 4, subsection a, b and c:<br>"...land with high-carbon stock, namely land that had one of the following status...wetlands, namely land that is covered with or saturated by water permanently,..."<br>"continuously forested areas" | Exclusion of land categorized as "Wetlands" and "Forest" in the land cover data set, including[B]:<br>• Inland marshes<br>• Peat bogs<br>• Salt marshes<br>• Salines<br>• Intertidal flats<br>• Broad-leaved forest<br>• Coniferous forest<br>• Mixed forest |
| Paragraph 5:<br>"...shall not be made from raw material obtained from land that was peatland in January 2008..." | Exclusion of land categorized as "Wetlands" in the land cover data set, mainly[B]:<br>• Peat bogs |

A 15
B 303,304
C All farmland is excluded for every time step with the land cover data set category "Agriculture". This includes all farmland, either with an HNV status or not.
D 307,308





Land-use/cover projections are crucial to determine future land availability for lignocellulosic energy crop production. The amount and location of land that meets REDII sustainability criteria vary over time due to LUC dynamics. Land use/cover projections are obtained from the Land-Use-based Integrated Sustainability Assessment modelling platform (LUISA) [304]. This dataset provides spatially explicitly land use/cover projections at a European level between 2020 and 2050 at a 10-year time step while considering economic, demographic and political drivers [49]. However, the LUISA dataset's land use/cover classification is not directly suitable for the assessment. In LUISA, several land use/cover categories are aggregated. For example, the category '*Forest/Transitional woodland-shrubland'* includes several land uses/covers. From this category, forests are to be excluded in line with REDII criteria, but shrubland is assumed to be available. In addition, LUC-related GHG emissions can vary considerably between different land use/cover categories [118].

For this study, several land use/cover categories from LUISA projections are disaggregated/aggregated into a new classification using the most recent version (2018) of the CORINE Land Cover (CLC) dataset [303]. The 2018 CLC dataset is selected for this process as its categories classification is well aligned with the LUISA dataset. Also, CLC categories are not aggregated and provide greater detail. These characteristics allow separating to some extent the LUISA dataset based on CLC. For example, this study's category 'shrubland' is established by overlapping the '*Forest/Transitional woodland-shrubland'* category in LUISA with 'broad-leaved', 'coniferous' and 'mixed forest' categories in CLC. It is assumed that forest areas in CLC will remain constant over time until 2050. The overlapping process is carried out for each projection starting in 2020 as a base year. Then, all locations that fail to overlap between '*Forest/Transitional woodland-shrubland'* in LUISA with 'broad-leaved', 'coniferous' and 'mixed forest' in CLC is considered 'shrubland'. Similar processes are carried for other land use/cover categories in the LUISA dataset. The overall process results in determining 11 land use/cover categories for this study. For a more detailed explanation of the disaggregation/aggregation process and land use/cover classification used in this study, see Land use/cover classificaton
Table S 4-2 in the supplementary material.

The land use/cover dataset distinguished in this study was filtered out in line with REDII sustainability criteria for each time step. Then, it was overlapped with the marginal land data to obtain available land projections that meet both land marginality and REDII sustainability criteria. Note that 2020 is used as a base year but not presented in the results. In addition, the marginal land data is assumed to be constant over time.

### 4.2.4 Lignocellulosic energy crops biomass potentials

Biomass potentials are quantified for each lignocellulosic energy crop assuming all available land is dedicated to a single crop. The (theoretical) gross biomass potential is assessed considering the spatial variation in evapotranspiration rates of lignocellulosic energy crops according to climate conditions. This potential refers to the maximum amount of biomass produced annually, given the water use efficiency of biomass production in relation to water loss from evapotranspiration. Then, agro-ecological suitability maps are employed to include the effect of other biophysical characteristics such as soil characteristics and terrain conditions on yield. Finally, crop-specific harvest indices are applied to estimate the amount of biomass that could potentially be harvested. A 1% annual increase in the theoretical gross biomass potential is considered for all energy crops to reflect productivity increases from improved crop management practices [49,309]. In addition to crop-specific biomass potentials (i.e., all available land is dedicated to a single energy crop), the 'maximum-yield' biomass potential is estimated. The maximum-yield biomass potential is quantified by selecting for each location the lignocellulosic energy crop with the highest attainable yield (t $ha^{-1}$). The crop-specific and maximum-yield biomass potentials are quantified for 2030, 2040 and 2050 and expressed in PJ biomass.

#### 4.2.4.1 Gross biomass potential

Biomass production is assessed with crop-specific parameters on length of the growing season, crop growth stage, cumulative evapotranspiration for each crop growth stage and water use efficiency (see Table S 4-1 in the supplementary material) following the approach in Dees et al. (2017) and Ramirez-Almeyda et al. (2017). Equation 4-1 represents the relation between crop yield on the one hand and crop-specific phenological characteristics and climate conditions on the other. This approach was applied to simulate crop yields into the future while considering the effects of climate change. It was assumed that the crops' growth is not limited by rain-fed conditions and water is not a limiting factor. However, even under no water limiting conditions and applying irrigation, crops can reach 90% of the gross biomass potential [306]. Daily evapotranspiration is calculated spatially explicit for 2030, 2040, 2050 using the Penman-Monteith equation [310]. Climatic parameters are derived from the HadGEM2-ES [311] global climatic model under the Representative Concentration Pathway (RCP) 4.5 [312]. Spatiotemporal projections on temperature, precipitation, relative humidity, shortwave radiation and wind speed are collected from the Inter-Sectoral Impact Model Intercomparison Project (ISIMIP2b) [313].





**Equation 4-1**

$$AB_{t,i} = \sum_j ET_j * Kc_{i,j} * WP_i$$

Where:
$AB_t$ = Theoretical gross above ground biomass potential, $t_{dry}$ $ha^{-1}$ $year^{-1}$
$i$= Crop type
$j$= Crop growth stage
$ET$ = Reference evapotranspiration, $m^3$
$Kc$ = Crop coefficient, *dimensionless*
$WP$ = Water use efficiency, $t_{dry}$ $m^{3^{-1}}$ $year^{-1}$

#### 4.2.4.2 Biomass potential based on agro-ecological suitability

The effect of location-specific biophysical conditions on crop growth is determined using crop-specific agro-ecological suitability maps. The crop-specific agro-ecological suitability maps are based on ten biophysical parameters. The agro-ecological suitability is expressed as a percentage of the theoretical gross maximum obtainable yield, ranging from 0 (unsuitable conditions) to 100 (highly suitable conditions), see Equation 4-2 . The suitability maps are calculated for each crop and for each point in time following the methods from Perpiña Castillo et al. (2015). Soil pH, soil texture, soil depth, soil type, soil drainage and slope are considered constant over time. Temperature, precipitation, frost-free days (FFD) and temperature growing periods (LGPt) vary following climate change RCP 4.5 projections. Section 4.8.3 and Table S 4-3 in the supplementary material contain the methods and crop-specific scores for each suitability parameter.

**Equation 4-2**

$$AB_{s,i} = AB_{t,i} * S_i$$

Where:
$AB_s$ = Above ground biomass potential considering biophysical factors, $t_{dry}$ $ha^{-1}$ $year^{-1}$
$i$= Crop type
$AB_t$ = Theoretical gross above ground biomass potential, $t_{dry}$ $ha^{-1}$ $year^{-1}$
$S$ = Suitability index for specific location, *%*

#### 4.2.4.3 Harvestable biomass yields

Crop-specific harvest indices (see Equation 4-3) were applied to estimate the location-specific harvestable yields (see Table S 4-1 in the supplementary material). The harvest index refers to the crop ratio between yield and biomass. This index varies between crops, given that some crops encounter more losses at harvest than others [306].

**Equation 4-3**

$$Y_i = AB_{s,i} * HI_i$$

Where:
$Y_{i}$ = Harvestable Yield, $t_{dry}$ $ha^{-1}$ $year^{-1}$
$i$= Crop type
$AB_{s,i}$ = Above ground biomass potential considering biophysical factors, $t_{dry}$ $ha^{-1}$ $year^{-1}$
HI = Harvest index, *dimensionless*

## 4.3 GHG EMISSION PERFORMANCE OF ADVANCED BIOFUEL SUPPLY CHAINS

All stages from lignocellulosic energy crop cultivation up to fuel use were included to quantify the GHG emission performance of advanced biofuel production from lignocellulosic energy crops in 2030. GHG emissions of advanced biofuels were assessed following the GHG emission calculation method indicated in the REDII, see Equation 4-4 [15] and the system boundary is defined in Figure 4-1.

**Equation 4-4**

$$E = e_{ec} + e_l + e_p + e_{td} + e_u - e_{sca} - e_{ccs} - e_{ccr}$$

Where:
E = Total emissions from the use of the fuel, $g\ CO_{2\text{-}eq}\ MJ^{-1}$
$e_{ec}$ = Emissions from the extraction or cultivation of raw materials, $g\ CO_{2\text{-}eq}\ MJ^{-1}$
$e_l$ = Annualised emissions from carbon stock changes caused by land-use change, $g\ CO_{2\text{-}eq}\ MJ^{-1}$
$e_p$ = Emissions from processing, $g\ CO_{2\text{-}eq}\ MJ^{-1}$
$e_{td}$ = Emissions from transport and distribution, $g\ CO_{2\text{-}eq}\ MJ^{-1}$
$e_u$ = Emissions from the fuel in use, $g\ CO_{2\text{-}eq}\ MJ^{-1}$
$e_{sca}$ = Emission savings from soil carbon accumulation via improved agricultural management, $g\ CO_{2\text{-}eq}\ MJ^{-1}$
$e_{ccs}$ = Emission savings from CO2 capture and geological storage, $g\ CO_{2\text{-}eq}\ MJ^{-1}$
$e_{ccr}$ = Emission savings from CO2 capture and replacement, $g\ CO_{2\text{-}eq}\ MJ^{-1}$

$CO_2$ emissions from biogenic sources are assumed to be zero for biofuels in line with the calculation rules in ANNEX V of the REDII [15]. Non-$CO_2$ GHG emissions, including $CH_4$ and $N_2O$ from end-uses, are assumed to be zero, consistent with the default and typical values set in the REDII as calculated by JRC [314]. Therefore, emissions from fuel in use ($e_u$) are set to zero. In this study, no carbon accumulation via improved agricultural management ($e_{sca}$) is considered. However, carbon accumulation related to land use and related land management changes is included in the emissions from carbon stock changes ($e_l$). $CO_2$ capture and geological storage ($e_{ccs}$); and $CO_2$ capture and replacement ($e_{ccr}$) fall outside the scope of this study. REDII grants a bonus of 29 g $CO_{2\text{-}eq}$ $MJ^{-1}$ when severely degraded land is restored, including a sizable reduction in erosion phenomena





[15]. However, this bonus is not applicable in this study, given the lack of evidence in soil erosion processes. For all advanced biofuel pathways, it is assessed whether they are able to comply with the 65% GHG emissions savings criteria of the REDII. In addition, it is also evaluated for each location the advanced biofuel pathway with the best GHG performance. The spatial results on GHG emissions are also used to derive emissions curves for each advanced biofuel. These curves are used to assess advanced biofuels production potential (PJ) under REDII GHG emissions savings criteria. GHG emissions other than $CO_2$ ($CH_4$ and $N_2O$) are expressed in $CO_2$equivalents using a global warming potential (GWP) impact calculated over 100 years (GWP100), consistent with the REDII.

**Multioutputs**

Most of the included conversion pathways produce multiple outputs, such as surplus electricity sold to the grid. The method to deal with multi outputs has a significant impact on the results [315,316]. Although substitution is seen as the most appropriate method to assess the impact of policy decisions [267], allocation is seen as more suitable for a comparative assessment of individual pathways or regulation of individual economic operators [15]. Following REDII requirements and to allow a consistent comparison between pathways, energy allocation is applied for co-products. In the case of co-generation of heat and power (CHP), as is the case in the ethanol pathways, exergy allocation is applied. The detailed exergy allocation method is described in more detail by [314] and [317].

### 4.3.1 Emissions from the extraction or cultivation of raw materials (eec)

The GHG emissions from the cultivation of energy crops are directly related to diesel usage for field activities, including pesticides and fertilizer application. Field operations include soil preparation, seeding/planting, pesticides/fertilizers application and harvesting. For herbaceous lignocellulosic energy crops, harvesting includes cutting and bailing, while for woody energy crops, it consists of cutting and chipping. Annual pesticide application and diesel usage per energy crop are included in Table S 4-4 from the supplementary material. Emission factors of diesel, pesticides and fertilizers are derived from Prussi et al. (2020). In this study, balanced fertilization is considered to estimate the emissions from fertilizers use. Therefore, the input rate is directly proportional to what is removed by harvesting the crop. An additional 15% is accounted for all inputs for potential losses from minerals uptake and terrain conditions. All equations and input data to calculate fertilizers inputs and the direct and indirect $N_2O$ emissions from lignocellulosic crop cultivation are found in section 4.8.5 and Table S 4-5 of the supplementary material.

### 4.3.2 Annualised emissions from carbon stock changes caused by land-use change (el)

Annualised LUC emissions from lignocellulosic energy crop production are calculated following Equation 4-5[15]. The method to assess GHG emissions from carbon stock changes resulting from LUC proposed in the REDII builds upon the stock difference approach from the IPCC Guidelines for National Greenhouse Gas Inventories Volume 4 [118]. Carbon stocks include above and below ground biomass, dead organic matter, litter, harvested wood product and soil [118]. However, REDII emphasizes the accounting of carbon stocks present in biomass and soils. Also, it is indicated that other carbon stocks are primarily relevant when land is converted to/from forest [118]. Therefore, in this study, only the biomass and soil carbon stocks are considered. Carbon stocks are assessed spatially explicitly for the land use/cover of available marginal land and if this land is dedicated to lignocellulosic energy crops. An amortization period of 20 years for carbon stocks to reach equilibrium is assumed in line with REDII and IPCC guidelines.

**Equation 4-5**

$$e_l = (CS_r - CS_A) * 3.664 * \frac{1}{20} * \frac{1}{P}$$

Where:
$e_l$ = Annualised emissions from carbon stock changes caused by land-use change, *g $CO_{2-eq}$ $MJ^{-1}$*
$CS_r$ = Carbon stock of marginal land before conversion, *t C $ha^{-1}$*
$CS_A$ = Potential carbon stock of marginal land when it is used for lignocellulosic energy crop production, *t C $ha^{-1}$*
*3.664 = Conversion factor to convert C to $CO_2$*
*1/20* = Factor to annualize emissions
P = Annual biofuel production from lignocellulosic energy crop, *MJ $ha^{-1}$*

#### 4.3.2.1 Biomass carbon stock

The method to assess the above ground biomass potentials of lignocellulosic energy crops was already described in section 2.2.2. The below ground biomass is calculated using crop-specific root-to-shoot ratios (see Table S 4-6). Willow, Poplar and Eucalyptus are not harvested annually, and it is assumed that the biomass carbon stock accumulates for 3 to 7 years [224].

For the land cover of available land prior to conversion, the spatially explicit above and below-ground biomass was assessed using a similar method as for the lignocellulosic energy crops. The above ground biomass of available land was quantified using the land-use and climate-zone specific default values of the IPCC on the maximum above ground biomass and the spatially explicit data on soil productivity (the degree to which the soil carries out its biomass production service) [319]. Below ground biomass was estimated as a function of above ground biomass using the IPCC's crop-specific and climate-dependent root-to-shoot ratios (IPCC 2006). Carbon fractions were employed to obtain the crop-specific biomass carbon stock. For a more detailed description of the spatially explicit calculation of each land use/cover category's biomass carbons stock, see section 4.8.7 in the supplementary material.





#### 4.3.2.2 Soil organic carbon stocks

The SOC stocks of the land use/cover before and after conversion are quantified with Equation 4-6. The IPCC default values for reference SOC levels for mineral soils are used considering soil type and climate zones stratification. The IPCC SOC stock change factors were employed to consider the effect of land use, management regime and input of organic amendments (IPCC 2006). See Table S 4-7 in the supplementary material for all input data to calculate the spatially explicit SOC levels.

**Equation 4-6**

$$SOC_{ref\ x\ or\ i} * F_{LU\ x\ or\ i} * F_{MG\ x\ or\ i} * F_{I\ x\ or\ i}$$

Where:
$SOC_{x\ or\ i}$ = Soil organic carbon stock for marginal land under land use/cover type x or s energy crop i, *t C $ha^{-1}$*
$SOC_{ref}$ = The reference carbon stock, *t C $ha^{-1}$*
$F_{LU}$ = Stock change factor for land use system, *dimensionless*
$F_{MG}$ = Stock change factor for management regime, *dimensionless*
$F_I$ = Stock change factor for carbon input levels, *dimensionless*

### 4.3.3 Emissions from processing

In this study, the selected conversion processes are:

- Production of ethanol from lignocellulosic biomass: The biomass is pre-treated by steam explosion to split the lignocellulosic into cellulose, hemicellulose and lignin, followed by an enzymatic hydrolysis step to break it down into fermentable xylose and glucose sugars. The sugars are fermented to ethanol in a simultaneous saccharification and fermentation process (SSF). The lignin is used in a CHP plant to generate process heat and electricity. The generated heat is used entirely for internal processes. Surplus electricity is sold to the grid. The input data and assumptions are based on JRC [314] and the associated references [320,321].

- Production of RFJ from ethanol (Alcohol-to-Jet, ATJ): The ATJ process converts alcohols into RJF, diesel and naphtha through dehydration, oligomerization and hydrogenation. Ethanol from lignocellulosic biomass is used as a feedstock. This ethanol process upgrading is the most commonly applied [322,323]. For the hydrogenation process, hydrogen is required to remove the double bonds. The source of hydrogen supply is described in section 4.3.3.1. The process assumptions are based on de Jong et al. [196] and Staples et al. [324].

- Production of hydrocarbon fuels through fast pyrolysis and upgrading**:** Biomass is first dried to a moisture content below 10% before entering the reactor using heat from char combustion. The fast pyrolysis produces bio-oil, char and non-condensable off-gases. The bio-oil cannot be used as a drop-in biofuel mainly due to the high oxygen content, low pH and instability. Bio-oil can be catalytically converted into drop-in fuels by hydrodeoxidation and hydrocracking and yield a mixture of hydrocarbon fuels with different chain lengths. The ratio of HFO, gasoline, diesel and RJF depends on the conditions in the upgrading process. The hydrogen required for upgrading is from external sources, as described in 4.3.3.1. Fuel output ratios are based on de Jong et al. ( 2017). The gasoline output ratio is the highest one among all outputs and is thus considered the main one. The amount of surplus electricity sold to the grid depends on the moisture content of biomass feedstock and hydrogen supply.

- Production of syn diesel from woody crops through gasification and Fischer-Tropsch synthesis (FT): Biomass is gasified to produce syngas (CO and $H_2$). The syngas is catalytically converted in a range of hydrocarbons in the Fischer-Tropsch reactor. The ratio depends on the CO/H ratio of the syngas, the type of catalyst and process conditions in the reactor. Commonly, the reactors involve mainly gasoline and diesel production [325]. However, diesel is selected as it shows the highest output ratio. The required H2 is produced from the syngas using the water gas shift reaction. Excess heat is used to generate electricity and sold to the grid. Data from JRC [314] is applied.

- Production of methanol through the gasification of woody crops and methanol synthesis: Syngas is produced in a pressurized fluidised-bed steam/$O_2$-blown gasifier and catalytically converted into methanol (Lücking, 2017). The process results in excess heat, which can be used for pre-heating the syngas and electricity generation. All excess heat is used to generate electricity and sold to the grid. Data is applied from Case MeOH-1 described in Hannula and Kurkela [326].

- Production of dimethyl ether (DME) through the gasification of woody crops and DME synthesis: DME is either produced simultaneously with methanol over advanced catalysts that are not commercially available yet or by methanol synthesis and a dehydration process [326]. In this study, methanol synthesis followed by dehydration to DME in a 2-step process is considered. Input data are based on Case DME-1 as described in Hannula and Kurkela ( 2013). Excess heat is used to generate electricity and sold to the grid.

- The input data and main assumptions for all conversion processes included are summarized in Table S 4-8 in the supplementary material





#### 4.3.3.1. Hydrogen supply

Hydrogen is used in multiple conversion processes. Previous studies have already demonstrated that the source of hydrogen from fossil or renewable sources has a significant impact on the total GHG footprint of biofuel supply chains [196]. However, in this study, it is assumed that hydrogen is supplied via steam methane reforming (SMR) of natural gas, where natural gas is processed with steam to produce syngas and additional $H_2$ with a water shift reaction. The SMR method is the most commonly applied [327]. Input data and assumptions are present in Table S 4-9 in the supplementary materials

### 4.3.4 Emissions from transport and distribution (etd)

It is assumed that the conversion plant is close to the source of biomass. Biomass is considered to be transported by truck to the conversion plant. More complex feedstock supply chains that include multimodal transport (for example, trains or ships) or pre-processing (such as pelletizing) could be feasible but not assessed. Transport of advanced biofuels from the processing plant to a blending depot and from a blending depot to a filling station are assumed similar between all advanced biofuel pathways. The assumptions of transport for all advanced biofuel pathways are largely consistent with the advanced biofuels pathway calculations in the REDII as described in Edwards et al. (2017). These assumptions represent typical feedstock supply chain conditions for Europe [328,329]. Transport distances and assumptions for inputs at the blending depot and filling station can be found in Table S 4-10 in the supplementary material.



## 4.4 RESULTS

### 4.4.1 Land availability

The total amount of available marginal land that meets REDII sustainability criteria varies from approximately 20.5 Mha in 2030 to 21 Mha in 2050 (see Figure 4-3). The largest share of available land corresponds to shrubland, followed by open space suitable. The share of both categories remains relatively constant over time. The share of abandoned land and established energy crops is considerably lower than other land categories. In general, there is a slight variation over time in land availability. The small variation over time in land availability is ascribed to the marginal land criteria and LUC dynamics.

The total amount and location of marginal lands in Europe are considered to remain constant over time. Thus, the changes in land availability driven by LUC dynamics are permanently restricted to marginal land areas. Over time, the release of available land is directly related to the LUC dynamics, which dictate the land use/cover changes. The exogenous land-use baseline scenario determines these LUC dynamics. For example, the exogenous model projects that for 2050, previously (2040) abandoned land areas shift to established energy crops land. Table S 4-11 in the supplementary material contains the available land LUC dynamics over time.

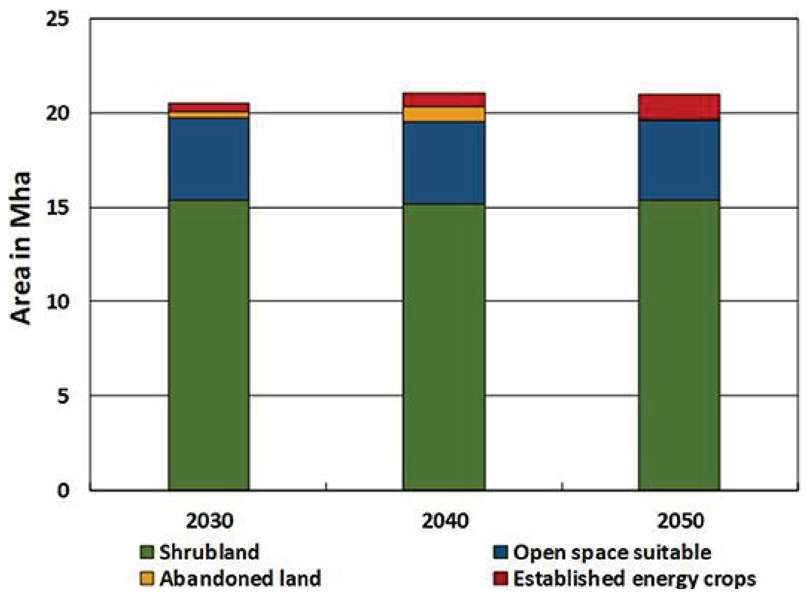


**Figure 4-3 Area of marginal land (in 1,000,000 ha) that meets the REDII sustainability criteria in 2030, 2040 and 2050**

The type of land available for energy crops and its location varies considerably across Europe (see Figure 4-4). A large share of this land is located in remote areas or areas with extreme biophysical conditions unsuitable for growing energy crops at a commercial scale (e.g., North of Scandinavia, North of the UK and South of Greece). There are some distinctive patterns for certain land types in Scandinavia and north of the Iberian Peninsula. Regardless of the land use/cover, a large extent of Scandinavia is categorized as marginal due to adverse biophysical conditions for crop production. Besides, a vast area of this region is covered by shrubland. Therefore, the Scandinavian region projects a considerable amount of available marginal land covered with shrubland that complies with REDII criteria. Likewise occurs for the north of the Iberian Peninsula.





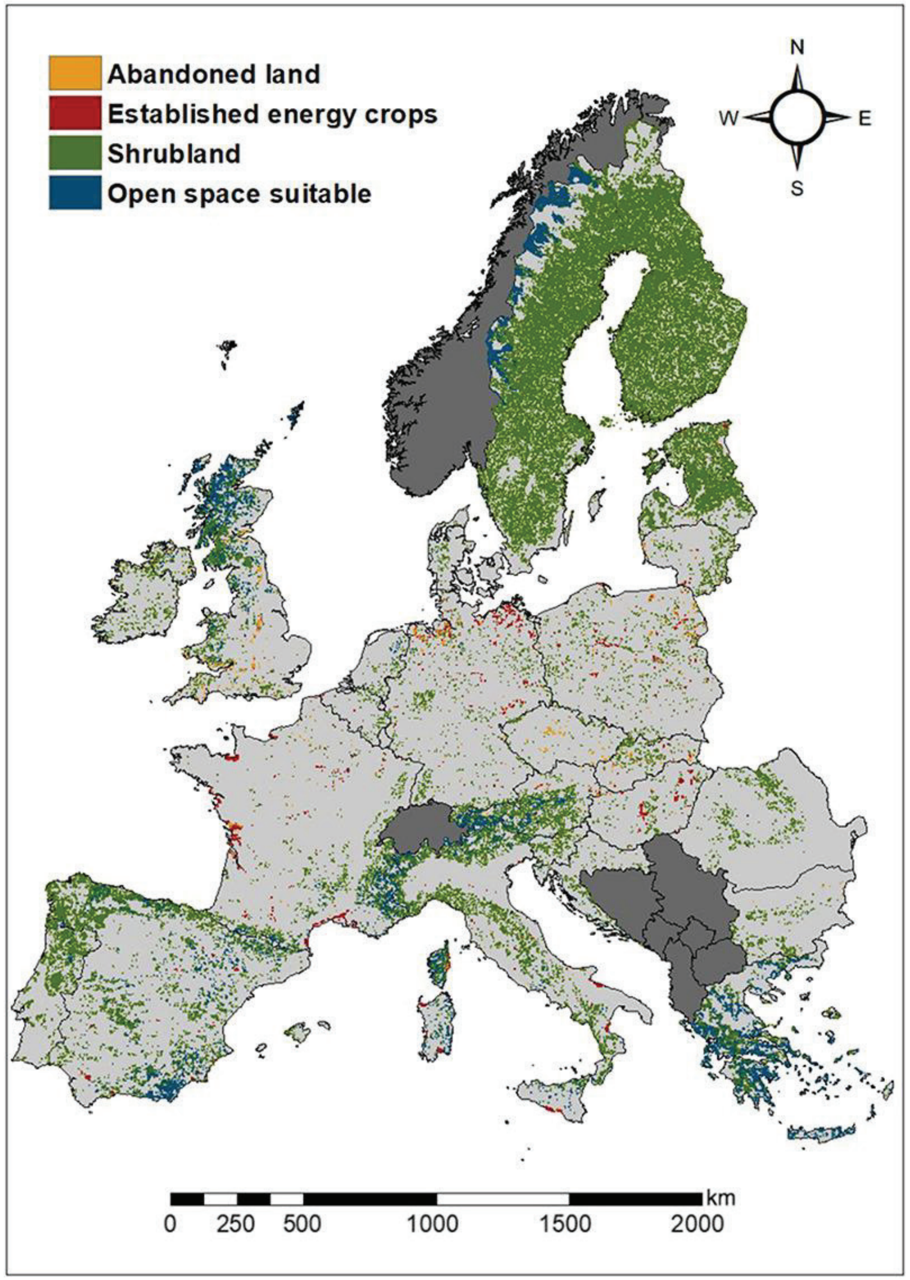


**Figure 4-4 Spatial distribution and land-use/cover of marginal land that meets REDII sustainability criteria in 2030. Maps for 2040 and 2050 are available in Figure S 4-1. The pixel size is enhanced for displaying purposes**

Open space suitable land is located in regions where temperature and precipitation regimes limit the development of vegetation. For example, in the North or South regions of Europe, such as the south of Greece/Spain (see Figure 4-4), high temperatures and low precipitation regimes limit vegetation development. This biophysical constraint results in open space suitable land characterized by sparely herbaceous vegetated areas. For mountainous regions (e.g., Alps), marginal land that meets REDII criteria is projected to be dominated by shrubland and open space suitable categories. These regions are generally characterized by a forest or shrubland cover. However, some biophysical characteristics common in these regions, such as high slope and low temperature, also limit vegetation growth.

All land categories are projected to be available, even in relatively small batches for most countries. There is a strong spatial variation in countries such as Germany, France, Poland and Spain. For example, in Germany's north section, a more significant degree of abandoned land and established energy crops are projected to be available. The exogenous land-use baseline scenario projects a change from agricultural production areas to abandoned or established energy crops. At the same time, the south is characterized by the availability of marginal shrubland. LUC dynamics and biophysical conditions describe the difference in the spatial distribution of available land for Germany. Countries such as France project less available marginal land that meets REDII criteria than others. For France, a large extent of the country is projected to be dedicated to agriculture. In some countries (e.g., Spain, Poland, and Hungary), the exogenous land-use scenario projects that several agricultural areas in 2030 become abandoned by 2040 (see Figure S 4-1 in the supplementary material).

### 4.4.2 Lignocellulosic energy crop potentials

There is a substantial variation in biomass potentials. In 2030, the biomass potentials vary between 386 PJ $yr^{-1}$ for willow to 1367 PJ $yr^{-1}$ for miscanthus (see Figure 4-5). The difference in biomass potentials between crops is mainly attributed to the adaptability of each crop to different biophysical conditions and the potential yield that it can deliver under those specific circumstances. For example, reed canary grass is characterized by the lowest water use efficiency and shortest growing period. As a consequence, reed canary grass delivers on average the lowest potential yields. However, the high tolerance of reed canary grass to a wider range of biophysical conditions (e.g., low temperatures and reduced precipitation) allows the production of this crop in locations that are not suitable for others. The opposite occurs for giant reed. Giant Reed is characterized by the highest water use efficiency and delivers on average the largest potential yields. Nevertheless, the adaptability of giant reed to a wide range of biophysical conditions is limited. To illustrate, giant reed is poorly adapted to cold temperatures and can only thrive in warmer regions. The adaptability difference results in a lower overall biomass potential for giant reed than for reed canary grass. Willow and poplar report similar biomass potentials to each other and low in comparison to other crops. Despite that both crops have similar phenological characteristics, willow delivers on average slightly higher yields. However, the adaptability of poplar is slightly higher. Poplar is partially better adapted to cold temperatures than willow. This allows for the potential production of poplar in more areas and overall higher biomass potential. Medium to low precipitation rates also constrains the potential production of both crops. As a result, the distribution of willow and poplar is reduced compared to other crops.

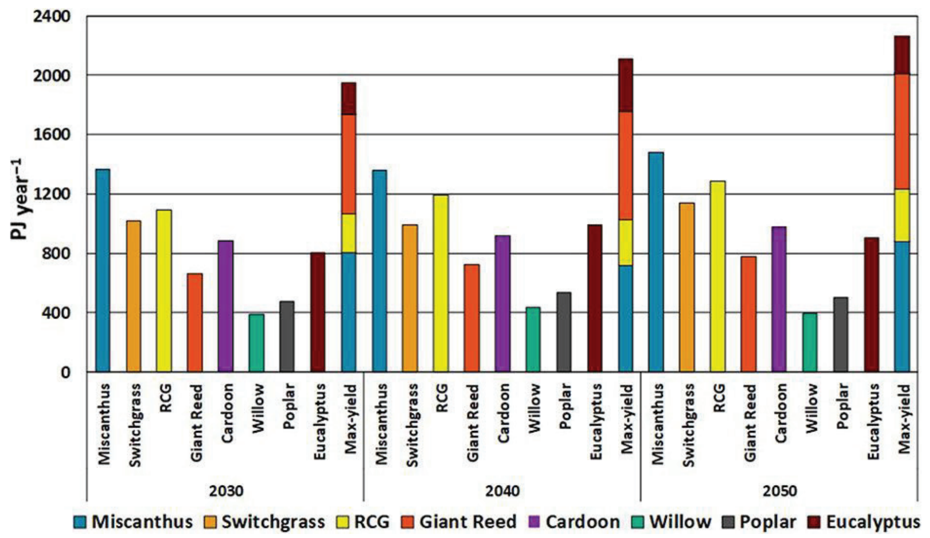


**Figure 4-5 Biomass potentials of eight lignocellulosic energy crops cultivated at available marginal land that complies with sustainability criteria of the REDII. Max-yield refers to the maximum yield biomass potential for which for each location the lignocellulosic energy crop with the highest attainable yield is selected. RCG, Reed Canary Grass**

Biomass potentials are projected to increase over time as a result of LUC dynamics (determined by the exogenous model), variation in climate conditions and (assumed) 1% annual yield increase. The LUC dynamics determine, for each point in time, the availability of marginal land that meets REDII sustainability criteria. Climate variations dictate the extent to which a crop is constrained to grow in a specific location. For example, eucalyptus biomass potential increases by 186 PJ $yr^{-1}$ between 2030 and 2040. The rise in average daily temperatures results in fewer annual frost-free days in some areas from the Mediterranean region. These climatic changes allow for the production of eucalyptus in several locations that were not suitable before (see figure Figure S 4-2 in the supplementary material), increasing the crop distribution. The opposite occurs for miscanthus. The overall biomass potential between 2030 and 2040 for miscanthus is reduced by 10 PJ $yr^{-1}$. The reduction in annual precipitation in some Medeterrian locations such as south of Spain reduces the suitability to produce miscanthus by 2040 in previously suitable areas. The impact of these climatic changes is highly localized and depends strongly on each crop's adaptability. While lower annual precipitation affects the distribution and production of miscanthus in some regions, other crops such as eucalyptus, giant reed and reed canary grass are less affected given their higher tolerance to drier conditions.

The maximum-yield biomass potential increases over time from 1951 PJ $yr^{-1}$ in 2030 to 2265 PJ $yr^{-1}$ in 2050 (see Figure 4-5). The maximum-yield biomass potential is composed mainly of miscanthus and giant reed, followed by reed canary grass and eucalyptus. The contributions of other crops are insignificant. Giant reed has the highest water use efficiencies and therefore delivers on average the highest potential yields. Accordingly, giant reed is selected over all other crops at locations where several crops can be produced. Miscanthus and switchgrass share similar phenological characteristics and adaptability to biophysical conditions. However, miscanthus delivers higher potential yields and thus, miscanthus is selected over switchgrass for all locations where both crops can potentially be cultivated. Willow and poplar are not selected given that for most of the sites suitable for their production, other crops deliver higher yields (e.g., giant reed and miscanthus). Reed canary grass is selected only in the regions where the biophysical conditions prevent the production of any other feedstock.

There is a strong spatial variation in the feedstock type and yield for the maximum-yield potential (see Figure 4-6). The lowest biomass yields are located in Scandinavia, the north of the Iberian Peninsula close to the Pyrenees, and in mountainous regions (e.g., the Alps). The extreme biophysical conditions (such as low temperature, acidic soils, low annual precipitation and few frost-free days) in these areas limit biomass production to a large extent. Mainly reed canary grass and miscanthus are marginally adapted to such conditions and deliver yields in the range of 0 to 100 GJ $ha^{-1}$ $yr^{-1}$. In the south of Europe, particularly the Iberian Peninsula and Greece, high temperatures and moderate/low precipitation constrain the biomass production of several feedstock types. However, giant reed and eucalyptus are adapted to these conditions and potentially deliver yields between 200 to 300 GJ $ha^{-1}$ $yr^{-1}$. The areas with the highest biomass potentials are located in Spain, Greece and Hungary. These areas feature favourable biophysical conditions for giant reed production, which by 2030 result in potential biomass yields up to 600 $ha^{-1}$ $yr^{-1}$.





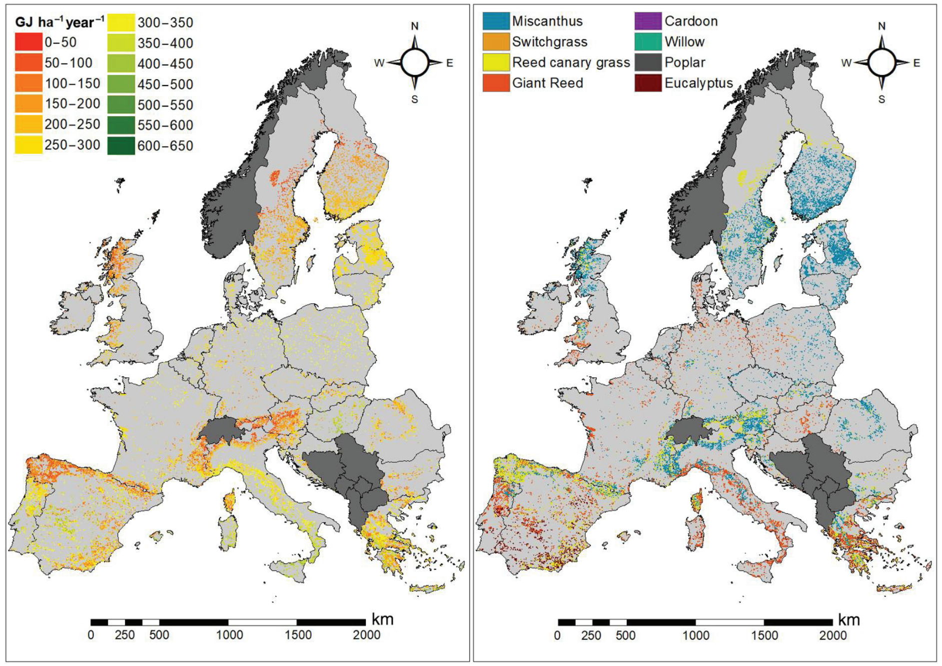


**Figure 4-6 Maximum–yield biomass potential (for each location, the lignocellulosic energy crop with the highest yield is selected) in 2030. Left-hand panel: biomass potential in GJ ha-1 year-1. Right-hand panel: the corresponding lignocellulosic energy crop selected for each location. Maps for 2040 and 2050 are available in Figure S 4-2 and Figure S 4-3**

The change in climatic conditions over time shifts the crop selection for some places for the maximum-yield potential. For example, in 2030, most of the maximum biomass potential in Germany and Poland is composed of miscanthus and giant reed. By 2050 there is an increase in frost-free days for these regions. This increase results in unsuitable conditions for giant reed production. However, miscanthus is better adapted to a rise in frost-free days. Thus, areas for giant reed production in 2030 shift to miscanthus in 2050 (see Figure S 4-3 in the supplementary material). A similar process occurs in Spain, driven by the change in precipitation. By 2050, several areas previously selected to produce eucalyptus, shift to giant reed. The shift between eucalyptus and giant reed is driven by the capacity of giant reed to deliver higher yields than eucalyptus in locations suitable for both crops. In other regions, the climatic conditions change results in newly suitable areas to produce lignocellulosic energy crops, e.g., production of reed canary grass in northern Scandinavia (see Figure S 4-2 and Figure S 4-3 in the supplementary material).

### 4.4.3 GHG emission performance of advanced biofuel supply chains

Total GHG emissions of the different advanced biofuel pathways vary on average between -32 g $CO_{2\text{-eq}}$ $MJ_{fuel}^{-1}$ for poplar/willow diesel to 38 g $CO_{2\text{-eq}}$ $MJ_{fuel}^{-1}$ for reed canary grass RJF (see Figure 4-7). However, total GHG emissions for some locations can increase up to 75 g $CO_{2-eq}$ $MJ_{fuel}^{-1}$ for reed canary grass RJF. The advanced biofuel pathways that undergo a gasification process to produce methanol, DME and diesel lead to the lowest GHG footprint. The conversion systems from these pathways are largely self-sufficient. Off gases from the reactors are used internally as energy fuel. In addition, small quantities of chemicals are required. These characteristics result in considerable low GHG emissions from upstream processes. Pathways that produce diesel perform slightly better than those that produce methanol and DME. The burden of GHG emissions is spread out more evenly between fuel and co-products (electricity). Similarly occurs for pathways that produce pyrolysis gasoline. Regardless of the upstream GHG emissions generated from hydrogen use in the conversion process, GHG emissions are spread more evenly between pyrolysis gasoline and co-products, resulting in overall low GHG emissions. The production of RJF followed by ethanol leads to the highest GHG emissions. These high GHG emissions from ethanol and RJF pathways are mainly caused by the upstream emissions related to the chemicals used in the fermentation process and the hydrogen use when ethanol is upgraded to RJF. In addition, these two pathways have a lower conversion efficiency, which results in higher GHG emissions along the supply chains. On average, all supply chains can comply with the 65% GHG emissions savings requirement of the REDII for new installations. However, there are several locations where the production of ethanol and RFJ from herbaceous energy crops will surpass the REDII GHG savings criteria.





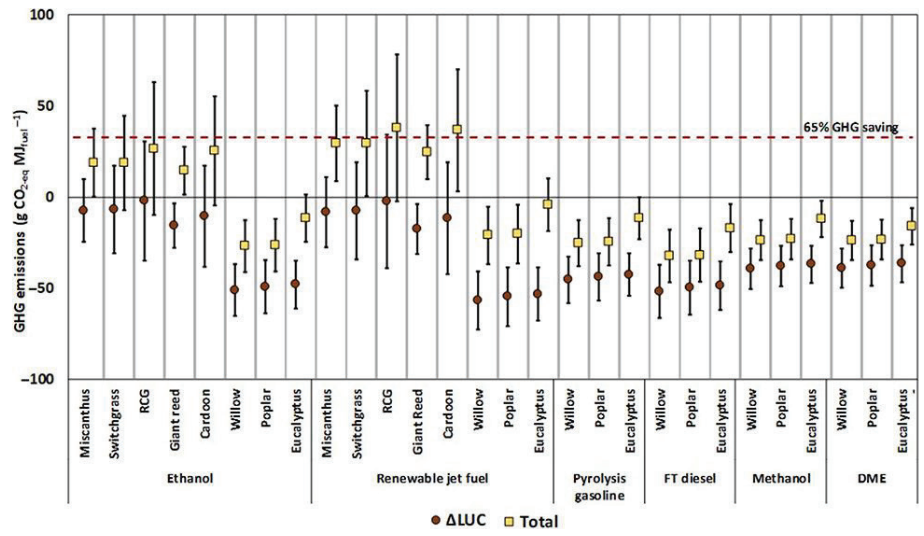


**Figure 4-7 Average greenhouse gas (GHG) emissions of advanced biofuel pathways with two standard deviations, including land-userelated net changes in carbon stocks in Europe for 2030. The ranges indicate the values within two times the standard deviation from the average in both directions (positive and negative). Thus, these ranges reflect spatial variability of GHG emissions due to the heterogeneity in biophysical conditions**

Total supply chain GHG emissions are driven largely by LUC-related net carbon stock changes (see Figure 4-7) and $N_2O$ field emissions. The impact of other stages of the supply chain, such as the conversion stage, becomes more relevant when LUC GHG emissions are low. Despite that, on average net carbon sequestration is achieved when marginal lands are used for energy crop cultivation. For some locations, LUC GHG emissions are considerably high. Generally, there is a SOC loss when land is dedicated to the production of herbaceous crops. However, this carbon loss is balanced out by the carbon accumulation in the biomass carbon pool. On average, the production of herbaceous crops results in more biomass per unit of area than the land prior to conversion. Nevertheless, for some regions, biomass production is limited by extreme biophysical conditions. In these regions, herbaceous crops deliver lower biomass, resulting in an additional loss of carbon in the biomass pool. For example, RJF from reed canary grass cultivated in the north of Scandinavia can result in emissions up to 75 g $CO_{2-eq}$ $MJ_{fuel}^{-1}$, from which LUC-related carbon stock changes cause approximately 50%. (spatially explicit results for all advanced biofuel supply chains are present in 4.8.12 of the supplementary material)

LUC-related and $N_2O$ field emissions vary considerably between crop types and location. Woody energy crop pathways perform better than herbaceous energy crop pathways. Generally, woody crops such as eucalyptus store more carbon in the biomass and SOC pools due to high yields and carbon accumulation over the crop harvesting cycles. Unlike herbaceous crops, woody crops (on average) increase the SOC compared to the land prior to conversion. However, for eucalyptus, the net LUC-related carbon storage is balanced out by $N_2O$ field emissions. Nitrogen inputs for eucalyptus are high. Similarly occurs for giant reed. Giant reed high carbon accumulation benefit in the biomass pool is counterbalanced by the high upstream emissions from nitrogen use and $N_2O$ field emissions. Conversely, willow and poplar require fewer inputs and, in combination with high carbon accumulation over the crops harvesting cycles, results in overall low GHG emissions. The GHG emissions difference between supply chains that use willow or poplar is negligible as both crops require similar inputs, deliver similar yields and share similar characteristics. On the contrary, there is more variation in the performance of herbaceous crop advanced biofuel supply chains. This variation is attributed to the different potential yields and the feedstock-specific conversion efficiencies.

As shown in Figure 4-8, the largest advanced biofuel potential is for eucalyptus methanol (455 PJ $yr^{-1}$) and the lowest for willow RJF (62 PJ $yr^{-1}$). For all poplar and willow advanced biofuels, the potential can be achieved with negative GHG emissions and substantial GHG savings. On the contrary, for all herbaceous crops advanced biofuels (ethanol and RJF), less than 3% can be achieved with negative GHG emissions. Approximately 99% of eucalyptus ethanol and 75% of eucalyptus RFJ can be achieved with negative emissions. For all herbaceous crops advanced ethanol between 70% to 98% can be achieved while meeting the 65% GHG savings REDII criteria (<32.9 g $CO_2$-eq $MJ_{fuel}^{-1}$). For all herbaceous crops advanced RJF between 44% to 90% can be achieved under REDII GHG savings criteria. The conversion efficiency and potential yield primarily determine the difference in advanced biofuel potentials. For example, the conversion efficiency between eucalyptus and methanol is considerably higher than between eucalyptus and diesel. This difference dictates that more methanol (455 PJ $yr^{-1}$) can be produced than diesel (223 PJ $yr^{-1}$) despite dedicating the same amount of land for eucalyptus production. Similarly occurs for reed canary derived ethanol and miscanthus derived ethanol. The difference in yields and conversion efficiency, both higher for miscanthus, dictates that more miscanthus ethanol (373 PJ $yr^{-1}$) can be produced than reed canary grass ethanol (326 PJ $yr^{-1}$). Despite that, more land is dedicated to reed canary grass production.





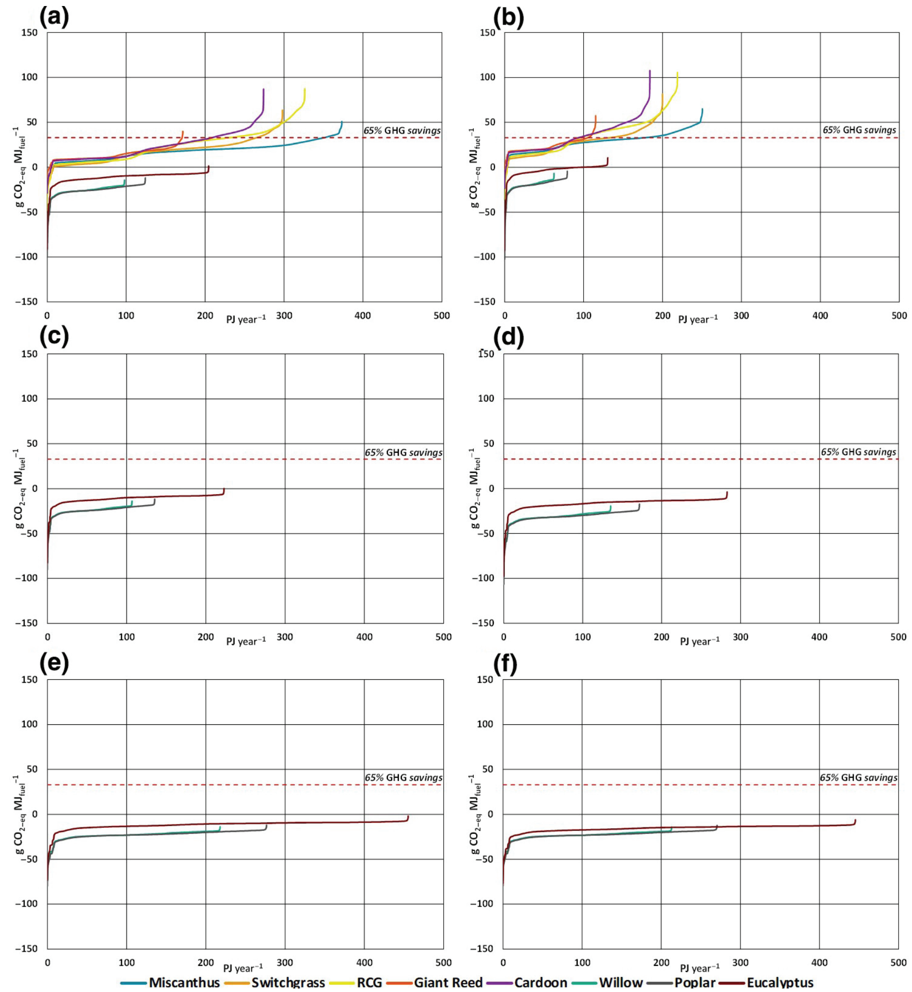


**Figure 4-8 GHG emissions curves for advanced biofuels production potential in 2030. The curves account for the total potential production if all available land is dedicated to a single crop and advanced biofuel conversion. Thus, they cannot be summed as the projected potential obeys a specific crop production to advanced biofuel pathway. (a) ethanol; (b) RJF, (c) pyrolysis gasoline, (d) diesel, (e) methanol, (f) DME. DME, dimethyl ether; GHG, greenhouse gas; RCG, Reed canary grass; RJF, renewable jet fuel**

The spatial heterogeneity in biophysical conditions determines to a large extent the overall GHG savings from a specific pathway. For example, on average, giant reed RJF results in lower GHG emissions than miscanthus RJF (see Figure 4-6). However, the potential of miscanthus RJF (178 PJ $yr^{-1}$) that can be achieved under REDII GHG emissions criteria is considerably larger than the potential of ethanol from giant reed (105 PJ $yr^{-1}$). Similarly occurs when miscanthus RJF is compared to other pathways such as eucalyptus

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RJF. Thus, overall, larger potentials can be achieved with additional GHG savings in the miscanthus to RJF supply chains compared to the giant reed RJF or other ones.

There is a spatial variation in the performance of the conversion routes. In the case of ethanol produced from miscanthus (Figure 4-9), supply chain emissions vary from 0.2 to 37 g $CO_{2\text{-eq}}$ $MJ_{fuel}^{-1}$. Most locations can comply with REDII GHG savings criteria. However, there are several regions where, despite meeting REDII land-related suitability criteria, the GHG savings criteria are not met. For example, in Scandinavia and Austria, the supply chain emissions of miscanthus ethanol surpass the REDII threshold. The relatively low performance in these locations is driven to a large extent by the LUC GHG emissions. The biophysical conditions from these areas constrain yield development and result in net carbon losses for the biomass carbon pool and less in soil.





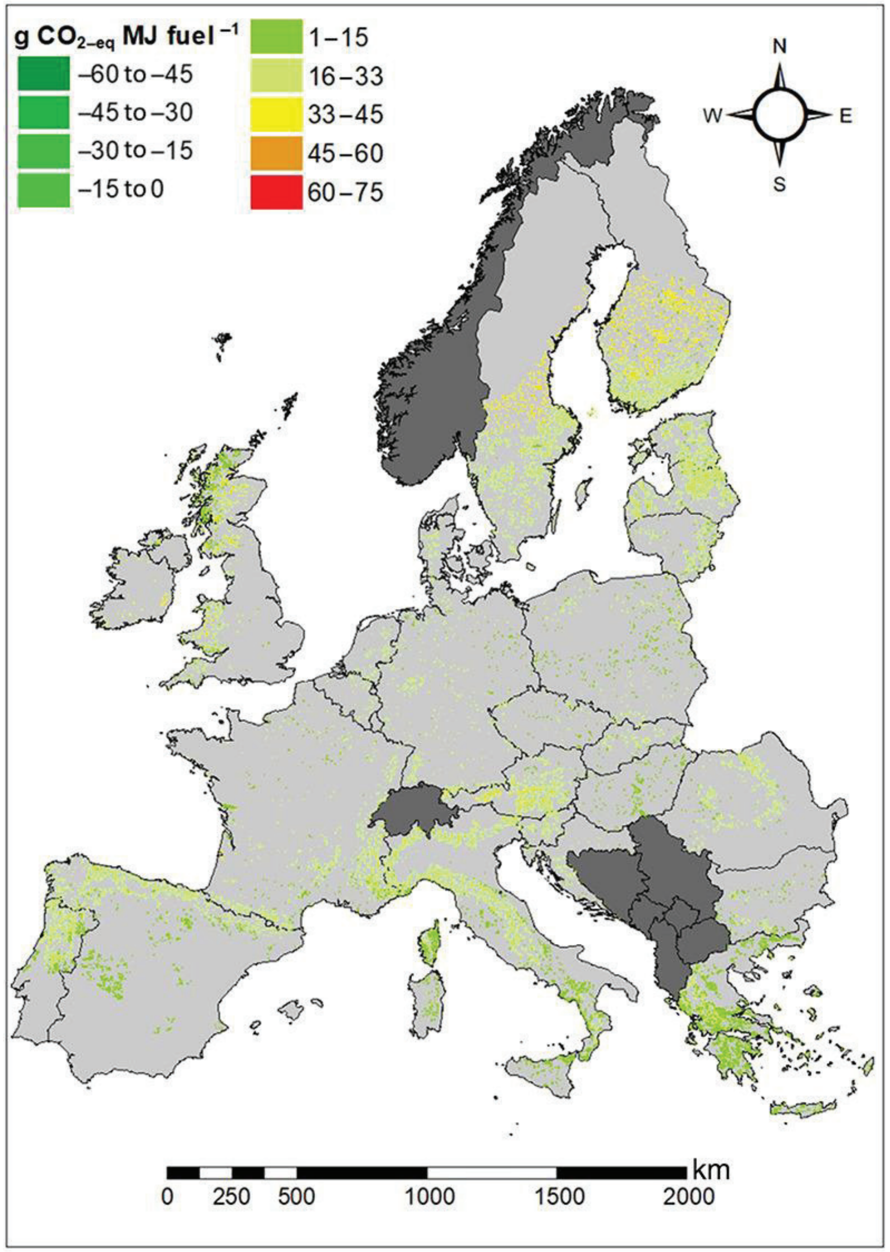


**Figure 4-9 Spatially explicit supply chain greenhouse gas (GHG) emissions of miscanthus ethanol production in Europe marginal lands by 2030. Green colours represent the locations that meet REDII GHG emissions savings criteria, while yellow and red represent the locations where the criteria are not met. Extreme maximum and minimum values are represented with the maximum and minimum values of two times the standard deviation. The maps showing the spatial variation of the GHG performance of each supply chain are found in the Section S12. For display purposes, the GHG emissions scale is set the same for all maps. REDII, Renewable Energy Directive Recast**

Woody crop advanced biofuel supply chains result in negative GHG emissions for all locations, except for some areas for eucalyptus ethanol and RJF pathways (see section 4.8.12 for all advanced biofuel supply chains in the supplementary material). The highest performance of woody crop advanced biofuels is generally located in Spain, Italy, or Greece, characterized by beneficial conditions for woody crop production. Other supply chains RJF derived from reed canary grass show a large spatial variation between -2 g $CO_{2\text{-eq}}$ $MJ_{fuel}^{-1}$ to 75 g $CO_{2\text{-eq}}$ $MJ_{fuel}^{-1}$. The strong variation for reed canary grass advanced biofuels is attributed mainly to the adaptability of the crop. Reed canary grass can be produced in a wide range of regions from Spain to north Scandanivaia (see section 12 in the supplementary material for spatially explicit results for all advanced biofuel supply chains). Generally, these regions present opposing biophysical characteristics. Thus, producing reed canary grass results in a diverse range of LUC GHG emissions. The opposite occurs for advanced biofuels derived from crops with lower adaptability, such as giant reed. There is less variability in GHG emissions given that this crop production is limited to regions with relatively similar characteristics.

The results vary strongly over space when the supply chain with the best GHG performance is selected (see Figure 4-10). The best GHG performance varies from -46 g $CO_{2\text{-eq}}$ $MJ_{fuel}^{-1}$ to 75 g $CO_{2\text{-eq}}$ $MJ_{fuel}^{-1}$. Several locations are still not able to comply with REDII GHG savings criteria. For most sites, the choice of the supply chain is already limited by the type of feedstock that can be produced under the local biophysical conditions. Therefore, the location-specific conditions determine the most suitable crop and, consequently, based on the crop adaptability, the best conversion pathway. For example, reed canary grass ethanol is selected for all locations where the biophysical conditions allow only its production. Herbaceous crops RJF supply chains are omitted given that the GHG performance is always lower compared to ethanol.





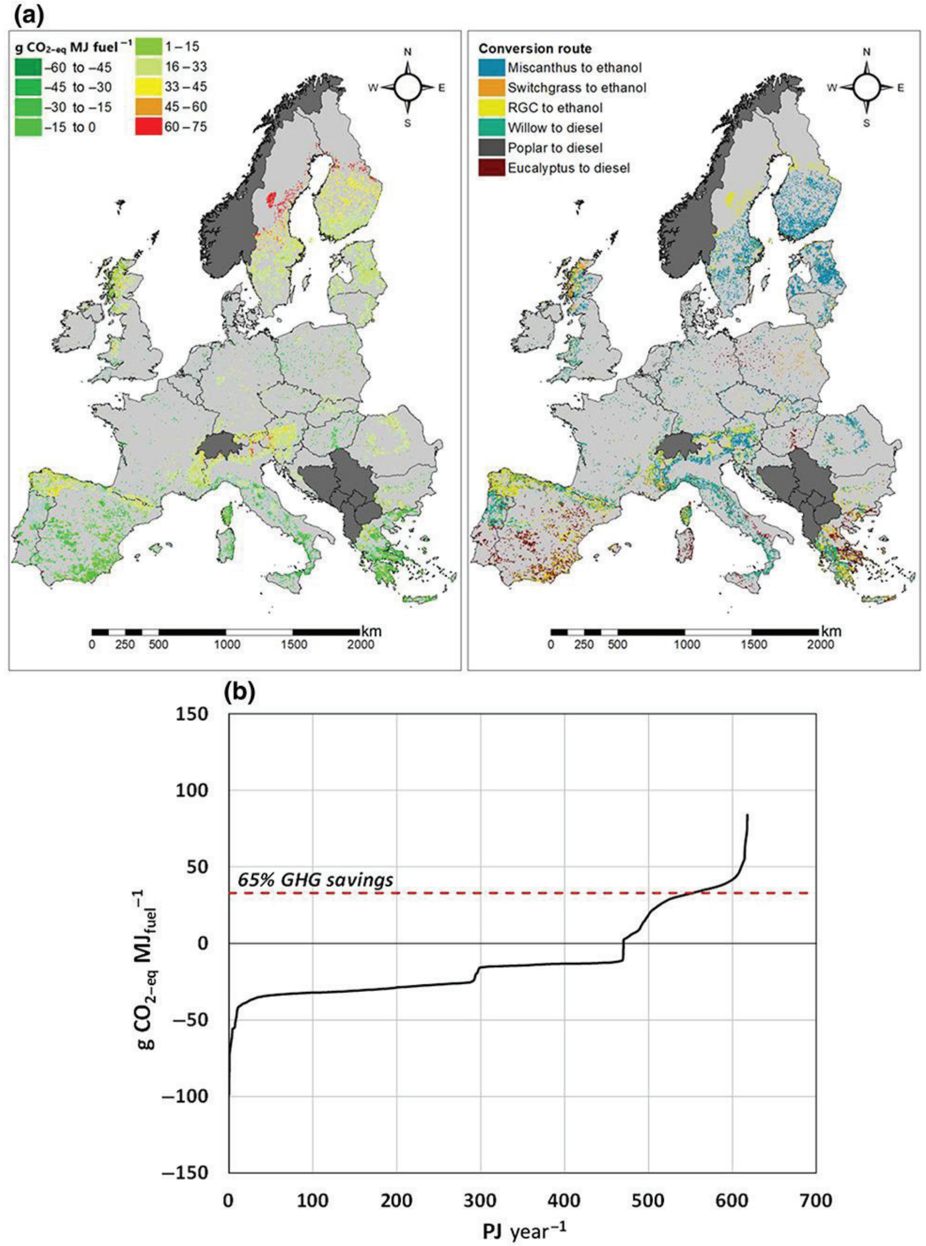


**Figure 4-10 Spatially explicit supply chain greenhouse gas (GHG) emissions and GHG emissions curve when the best GHG performance supply chain is selected for each location. (a) Supply chain GHG emissions and correspondent supply chain configuration in Europe marginal lands by 2030. Green colours represent the locations that meet REDII GHG emissions savings criteria, while yellow and red represent the locations where the criteria are not met. (b) GHG emissions curve for advanced biofuels production potential in 2030. The curve accounts for the total potential production if all available land is utilized when combining the lowest supply chain GHG emissions for each location. REDII, Renewable Energy Directive Recast**

The most suitable crop needs to be considered combined with the conversion pathway collectively. Otherwise, the GHG emissions savings potential from the advanced biofuel pathways can be diminished. To illustrate, if giant reed is selected for a specific location (considering the highest yield performance) without accounting for the conversion pathway, it can result in supply chains with lower GHG emissions performance. Other supply chains can achieve higher GHG emissions savings while utilizing crops with lower yields (e.g., woody crops). In locations where several crops can be produced, diesel from woody crops is generally selected as it presents the highest GHG emission savings.

Approximately 618 PJ $yr^{-1}$ can be achieved when selecting for each location the best GHG performance supply chain (see Figure 4-10). A large share of the potential is composed of woody crops diesel (76%). Within woody crops diesel share, 60% is obtained from eucalyptus and willow (30% each) and 16% from poplar. The contribution of ethanol from switchgrass is almost negligible, while ethanol derived from miscanthus contributes 15% and from reed canary grass 9%. A large extent of the advanced biofuel potential (470 PJ $yr^{-1}$) can be achieved with negative GHG emissions and 552 PJ $yr^{-1}$ can be produced under REDII GHG emissions savings criteria. However, despite that the lowest GHG emissions are selected for each location, there are still 66 PJ $yr^{-1}$ of advanced biofuels that fail to meet REDII GHG savings criteria and could not be rolled out into the market.

## 4.5 DISCUSSION

This study assessed spatially explicit land availability, biomass potentials and GHG performance of advanced biofuel pathways produced on marginal lands in Europe and the UK under REDII sustainability constraints.

Approximately half (I.e., 7.9-8.9 Mha) of the projected available marginal land that meets REDII criteria shows suitable conditions for energy crop cultivation. These projections are lower than the 13-15 Mha of available land estimated by the JRC-EU-TIMES model under the low scenario. Despite that, the JRC-EU-TIMES model applied similar REDII restrictive criteria [62]. The JRC-EU-TIMES land availability projections are not limited to marginal land.

The estimated biomass potentials are considerably lower than the conservative projections found by Hoefnagels and Germer (2018) on domestic biomass potentials in the EU-28 (from 4800 PJ in 2020 to 8160 PJ in 2050). The differences in scope, parameters and approach limit the comparison exercise. Nevertheless, projections for 2050 are in the same order of magnitude as the JRC-EU-TIMES model low (1515 PJ $y^{-1}$; high sustainability constraints) and middle (2063 PJ $yr^{-1}$ moderate sustainability constraints) scenarios [62], accounting only for lignocellulosic crops. Applying a similar approach and sustainability constraints, Dees et al. (2017) projected 2661 PJ of biomass available for 2030. However, in Dees et al. (2017), biomass production is not limited to marginal lands. There is a considerable difference in biomass potentials with other studies, such as Bogaert et al. (2017). The high biomass potentials projected for 2030 (4731 PJ) in Bogaert et al. (2017) are only achievable if agricultural land is dedicated to energy crop production.





The estimated biomass potentials were explicitly assessed under REDII sustainability criteria. Therefore, other sustainability considerations such as water use efficiency and impacts on biodiversity were not accounted for. Biomass potentials can reduce to a large extent under water-limited conditions [306]. For example, miscanthus biomass potential stands at 1367 PJ for 2030. Under water-limited conditions (no irrigation), the potential of miscanthus reduces to 759 PJ for 2030 (see section 4.8.13 in the supplementary material). Thus, accounting for water-limited conditions on each location can result in an overall 44% reduction of biomass potential for every crop. The projected biomass potential relies to a large extent upon the conversion of shrublands. However, shrublands contribute to several ecosystem services, such as maintaining biodiversity and nutrient provision[331]. Considering further environmental constraints to avoid the conversion of shrublands would reduce biomass potentials between 248 PJ $yr^{-1}$ to 1491 PJ $yr^{-1}$ depending on crop type and relevant point in time (see section 4.8.13 sensitivity analysis in the supplementary material). Additional expertise is required to categorize and convert shrubland areas with a low biodiversity impact.



Economic and non-economic barriers such as competition with other domestic and imported biomass sources, lack of skill and infrastructure and poor cash flow for farmers [332] were not accounted for in the assessment. The remoteness of some areas can limit accessibility. The supply of large biomass volumes from remote regions to the conversion facilities with inadequate infrastructure can be costly and inefficient [333]. It will also require several years to scale up logistics and processing capacity [291]. These variables could lead to lower biomass potentials and additional GHG emissions along supply chains. The biomass production cost and the competitiveness of the different crop types will vary and influence farmers' crop selection. Typically, marginal land crop yields are low, and thus, feedstock costs are relatively high [334]. Accordingly, from an economic point of view, the assessed biomass potentials and conversion routes can vary in the future in the function of end-use-markets, costs and competitiveness. Competing uses of biomass were not accounted for and could result in different biomass and fuel potentials. Other biomass value chains for heat, power and materials can provide a better end-use from an economic and environmental perspective [11]. These economic drivers and competing uses should be considered for future assessments.

The spatial heterogeneity drives the variability in GHG emissions performance. The location-specific biophysical conditions dictate the type of feedstock produced and, based on the crop characteristics, the best conversion pathway. Under water-limited conditions (see section 4.8.13 sensitivity analysis in the supplementary material), supply chain average GHG emissions increase between 7 to 25 g $CO_{2\text{-eq}}$ $MJ_{fuel}^{-1}$ for herbaceous crop pathways, and most surpass REDII GHG thresholds. A yield reduction leads to less carbon stored in the biomass section (for several locations) than the land prior to conversion. For woody crop pathways, the effect of water-limited conditions on GHG emissions is less significant. For these pathways, a large share of the carbon accumulation occurs in the SOC pool. Accordingly, average GHG emissions present a low variation as the SOC accumulation remains constant for both yield scenarios. Herbaceous pathways appear to be more sensitive to yield changes and woody pathways to SOC changes. However, it is still highly uncertain whether the soil will reach the expected state of carbon equilibrium [335]. Failing to reach a SOC equilibrium state can potentially result in a release of carbon [118] and result in considerably less locations with negative GHG emissions.

The overall net GHG emissions balance is negative for all advanced biofuels when considering the GHG emission savings from replacing fossil fuels (see Table S 4-12). Between -7.1 Mt $CO_{2\text{-eq}}$ $yr^{-1}$ for willow RJF to -48.8 Mt $CO_{2\text{-eq}}$ $yr^{-1}$ for eucalyptus DME. The GHG savings suggest that implementing any of the pathways can lead to a GHG emissions mitigation strategy. However, these savings can reduce if water-limited conditions are accounted for, or the SOC pool fails to reach equilibrium. Approximately 65.4 Mt $CO_{2\text{-eq}}$ $yr^{-1}$ can be mitigated (compared to fossil fuel use) when selecting for each location the pathway with the lowest GHG performance. This mitigation potential could reduce in 3% the projected GHG emissions from the transport sector, including aviation and maritime, for 2030 (2168 Mt $CO_{2\text{-eq}}$ $yr^{-1}$) [336]. Despite that this reduction seems minimal, GHG emissions are estimated to fall only by 3% between 2020 and 2030 if the current trend continues [336].

By 2030, approximately 552 PJ $yr^{-1}$ of advanced biofuels can be produced from feedstock originated solely in marginal lands under the REDII GHG emissions threshold. With a final energy consumption in the EU transport sector projected at 10467 PJ $yr^{-1}$ for 2030 [6], the advanced biofuels potential of marginal lands can contribute 5.2% to the final energy consumption. This supply can support to meet the 14% minimum share of renewable energy within the final consumption of energy in the transport sector (currently standing at 9%) and the sub-target of 3.5% for advanced biofuels and biogas (currently +/- 0.4%) established by REDII [6,337]. However, these potentials can increase between 6.2 to 10.4% (depending on final use) if credits from REDII are accounted for. A joint effort between different alternatives is anticipated to meet renewable energy targets [293]. The GHG emissions performance and economic potential of utilizing non-marginal abandoned





arable land for biomass production are highly uncertain [23]. There could be insufficient available land with a high level of sustainability constraints. Therefore, the role of biomass imports can be of paramount importance to meet future demand in Europe [338].

The results of this study should be interpreted with care as there are many uncertainties regarding projections of future land availability and applied methods. The land use/cover input dataset is based on a supply-demand module that considers agricultural markets, demographic and macroeconomic trends [309]. However, the ability to reproduce historical trends in combination with a diverse set of drivers does not necessarily assure that future land use/cover conditions will be adequately represented [339]. The location of marginal lands was assumed to be constant over time. However, some parameters related to climate conditions (temperature, precipitation, and humidity) that define land marginality to some extent [305] are not constant. Including the temporal variation in climate conditions in line with the RCP 4.5 projections could affect the location and extent of marginal lands. RCP 4.5 climate projections were used as inputs to determine biomass potentials and GHG performance. The results are likely to be different for different RCP's. However, no big differences are expected, as in the short term there is no significant variation between scenarios[340]. At longer times scales (after 2050), the differences between climate projections become considerable and could lead to different results.



More expertise is required to translate and understand the relationship between suitability scores and yield. Despite that, yields are based on widely recognized crop suitability parameters [50]. These are not calibrated with empirical results. In reality, yields are often lower as a result of less optimal conditions such as nutrient shortage or incomplete plant cover [306]. The translation of suitability scores into indices that determine yield is uncertain. In addition, there is little knowledge of the cause-effect relationship between suitability parameters and yields. This relationship could be linear, quadratic, or exponential. Accounting for such processes requires an additional understanding of physical and biological relationships. Nevertheless, the yields estimated in this study fall in the range as the ones reported in Dees et al. (2017), based on a literature review.

The GHG performance of advanced biofuels is based on modelling work that is inherently uncertain due to the assumptions made and input data. For example, choices on allocation procedures highly affect the results, as highlighted/demonstrated by [273,329]. This study is also limited to specific advanced biofuel conversion pathways. Other system designs with the same final product will perform different. To illustrate, hydrogen is assumed to be supplied from steam methane reforming of natural gas. However, it has been demonstrated that using different sources of hydrogen could have a considerable impact on the GHG performance [341]. Other technologies, including iso-butanol and hydrothermal liquefaction (HTL), could improve GHG performances compared to ethanol and pyrolysis fuels [341,342]. More advanced pre-processing techniques such as pelletizing or steam explosion could substantially reduce transport GHG emissions for long-distance supply chains [343]. In addition, $CO_2$ capture and storage combined with advanced biofuel production (BECCS) could be a valuable carbon mitigation technology. Carbon savings of advanced biofuels can increase due to technological progress and should be included in future assessments, such as more efficient and upgraded conversion processes [344].

The assumption of balanced fertilization considers a linear relationship between fertilizer use and crop yield, resulting in equivalent GHG emissions from fertilizer use per unit of fuel for every location. However, the amount of fertilizers applied and related GHG emissions can vary according to country/region farming characteristics [345]. Indirect $N_2O$ emissions from leaching were not considered. However, in regions characterized by a wet regime, such as in Scandinavia [346], leaching can occur. Still, the GHG emissions from leaching are minimal compared to the overall GHG emissions of the supply chain.

## 4.6 CONCLUSION

The production of advanced biofuels from marginal land sourced energy crops can rise as a valuable EU climate change mitigation strategy to reduce $CO_2$ emissions and support to meet EU biofuel demand. Smart choices on location, crop type and supply chain design are of paramount importance to release the maximum benefits of such strategy. However, other environmental, social and economic impacts should be considered in the decision making to avoid negative trade-offs. The use of marginal lands could be acknowledged in policy as a supplementary climate change mitigation strategy, given that other sources will be required to meet GHG emissions reduction targets and biofuel demand. In addition, it needs to be designated in line with policy targets as there can be trade-offs between maximizing the mitigation potential of $CO_2$ emissions or biofuel supply. It is highly relevant to account for the spatial heterogeneity in biophysical conditions when assessing the climate effects in bioenergy systems. The omission of such characteristics can lead to inadequate assessments and, consequently, defective policy recommendations.

## 4.7 ACKNOWLEDGEMENTS

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## 4.8 SUPPLEMENTARY MATERIAL

### 4.8.1 Energy crop cultivation

**Table S 4-1 Crop specific length of the growing season, length of crop growth stage, evapotranspiration per crop growth stage, water use efficiency and harvest index of lignocellulosic energy crops (Dees et al. 2017)**

| Lignocellulosic energy crop | Length of the growing season (days) | Start day of growing season (day) | Length of growth stage (cumulative fraction of length of season) | | | | Crop coefficient per growth stage (Kc) | | | | Water use efficiency $(t_{dry}\ m^{3-1}\ ha^{-1})$ | Harvest index |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Initial | Development | Mid | Late | Initial | Development | Mid | Late | | |
| Miscanthus | 210 | 80 | 0.21 | 0.34 | 0.84 | 1 | 0.48 | 1.05 | 1.41 | 0.95 | 33 | 0.7 |
| Switchgrass | 210 | 80 | 0.18 | 0.31 | 0.80 | 1 | 0.50 | 0.99 | 1.30 | 0.80 | 30 | 0.6 |
| Giant reed | 220 | 90 | 0.21 | 0.32 | 0.78 | 1 | 0.54 | 1.01 | 1.74 | 1.10 | 31 | 0.7 |
| Reed canary grass | 190 | 80 | 0.20 | 0.30 | 0.80 | 1 | 0.50 | 1.00 | 1.40 | 1.00 | 22 | 0.6 |
| Cardoon | 250 | 90 | 0.10 | 0.20 | 0.80 | 1 | 0.50 | 0.70 | 1.00 | 0.95 | 31.3 | 0.6 |
| Willow | 300 | 80 | 0.16 | 0.39 | 0.84 | 1 | 0.40 | 1.00 | 1.50 | 0.50 | 30 | 0.65 |
| Poplar | 300 | 80 | 0.16 | 0.39 | 0.84 | 1 | 0.40 | 1.00 | 1.50 | 0.40 | 29 | 0.6 |
| Eucalyptus | 300 | 90 | 0.16 | 0.39 | 0.84 | 1 | 0.40 | 1.00 | 1.50 | 0.40 | 27 | 0.65 |

## 4.8.2 Land use/cover classificaton

**Table S 4-2 Land use/cover classification in this study based on land use/cover projections of LUISA database and disaggregation based on CLC**

| Land use/cover categories classification in this study | LUISA land use/cover categories | Corine land cover land use/cover categories used for aggregation/ disaggregation | Description of aggregation/disaggregation |
|---|---|---|---|
| 1 Artificial | 1 Urban<br>2 Industry, Commercial and Services<br>7 Infrastructures<br>10 Urban green leisure | | LUISA land use/cover categories *Urban; Industry, Commercial and Services; Infrastructures and Urban green leisure* are aggregated to a single land use/cover category *'Artificial'* |
| 2 Agriculture | 3 Agriculture | | No aggregation / disaggregation |
| 3 Forest | 4 Forest/Transitional woodland-shrubland | 3.1.1 Broad leaved forest<br>3.1.2 Coniferous forest<br>3.1.3 Mixed forest | LUISA land use/cover category *'Forest/Transitional woodland-shrubland'* is disaggregated to *'Forest'* and '*Shrubland*'. All areas classified as forest in CLC 2018 (categories 3.1.1, 3.1.2 and 3.1.3) that overlap with '*Forest/Transitional woodland-shrubland*' in LUISA are classified as *'Forest'*. [A]<br>All areas that on a previous time step are categorized as *'Forest'* and on the subsequent time step as *"new energy crops"* are classified as *'Forest'*.[B] |
| 4 Natural grassland | 6 Natural land | | LUISA land use/cover category '*Natural land'* is disaggregated to '*Natural grassland*'; '*Open space suitable*' and '*Open space unsuitable*'.<br>All areas of '*Natural land*' not classified as *'Open space suitable' or 'Open space unsuitable*' are classified as *'Natural grassland'*.<br>All areas that on a previous time step are categorized as *'Natural grassland'* and on the subsequent time step as *"new energy crops"* are classified as *'Natural grasslands'*.[B] |
| 5 Established energy crops | 5 New energy crops | | LUISA land use/cover category "5. New energy crops"<br>All areas that remain "new energy crops" over each subsequent time step. |
| 6 Wetlands | Wetlands | | No aggregation / disaggregation |
| 7 Water bodies | Water | | No aggregation / disaggregation |





**Table S 4-2 Land use/cover classification in this study based on land use/cover projections of LUISA database and disaggregation based on CLC (continued)**

| Land use/cover categories classification in this study | LUISA land use/cover categories | Corine land cover land use/cover categories used for aggregation/ disaggregation | Description of aggregation/disaggregation |
|---|---|---|---|
| 8 Shrublands | 4 Forest/Transitional woodland-shrubland | | LUISA land use/cover category *'Forest/Transitional woodland-shrubland'* is disaggregated to *'Forest'* and '*Shrubland*'. The area not categorised as *'Forest'* (3) is classified as 'Shrubland'.<br>All areas that on a previous time step are categorized as '*Shrubland*' and on the subsequent time step as *"new energy crops"* are classified as '*Shrubland*.[B] |
| 9 Open space unsuitable | 6 Natural land | 3.3.1 Beaches-dunes-sands<br>3.3.2 Bare rocks<br>3.3.5 Glaciers and perpetual snow | LUISA land use/cover category '*Natural land'* is disaggregated to '*Natural grassland*'; '*Open space suitable*' and '*Open space unsuitable*'<br>The CLC 2018 land use/cover categories *'Beaches-dunes-sands*'; *'Bare rocks*'; and *'Glaciers and perpetual snow'* that spatially overlap with '*Natural land*' of the LUISA projections are classified as *'Open space unsuitable'*.[C]<br>All areas that on a previous time step are categorized as 'Open space unsuitable ' and on the subsequent time step as *"new energy crops"* are classified as *'Open space unsuitable'*.[B] |
| 10 Open space suitable | 6 Natural land | 3.2.2 Moors and heathland<br>3.2.3 Sclerophyllous vegetation<br>3.3.3 Sparsely vegetated areas | LUISA land use/cover category '*Natural land*' is disaggregated to '*Natural grassland*'; '*Open space suitable*' and '*Open space unsuitable*'<br>The CLC 2018 land use/cover categories '*Moors and heathland*'; '*Sclerophyllous vegetation*', '*Sparsely vegetated areas*' and 'Burnt areas' that spatially overlap with '*Natural land*' of the LUISA projections are classified as *'Open space suitable'*[D]<br>All areas that on a previous time step are categorized as 'Open space suitable ' and on the subsequent time step as *"new energy crops"* are classified as *'Open space suitable'*.[B] |
| 11 Abandoned land | 5 New energy crops | | LUISA land use/cover category *'New energy crops'* which was previously agricultural land, is reclassified as abandoned agricultural land. |

[A] It is assumed areas classified as *'Forest'* in CLC 2018 remain constant over time until 2050.
[B] the disaggregation of "New energy crops" is required to assure that the LUC dynamics and therefore land availability meets REDII sustainability and marginal land criteria for all categories and each point in time.
[C] land with physical constraints for lignocellulosic energy crops production
[D] i.e. land with no-physical constraints for lignocellulosic energy crop production

### 4.8.3 Agro-ecological suitability maps

The suitability scores for each crop under the ten biophysical variables are retrieved from Perpiña Castillo et al. (2015). These scores are estimated through an extensive literature review and defined within six classes: very suitable (highest adaptability), suitable, moderately suitable, low suitability, poorly suitable (low adaptability) and not suitable. Each suitability class corresponds to a numerical score between 0 to 100, reflecting the suitability class in numerical values (see Table SM3). A suitability map was developed for each crop individually for all biophysical variables. Then, to determine the overall suitability for each crop, all biophysical variables maps were combined with a weighted sum. A score of 0 in any of the biophysical variables results in a location where it is not feasible to grow the crop. All biophysical variables were assigned the same weight except for temperature and precipitation. These two variables were assigned double the weight to reflect the importance of these variables in crop growth [50].





**Table S 4-3 Crops suitability scores for each biophysical variable class adapted from Perpiña Castillo et al. (2015).**

| Biophysical variables | Classes | Lignocellulosic energy crops suitability to biophysical variables (%) | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Miscanthus | Switchgrass | Reed canary grass | Giant Reed | Cardoon | Willow | Poplar | Eucalyptus |
| Soil pH | 0-4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | 4-5 | 40 | 40 | 40 | 20 | 20 | 20 | 20 | 20 |
| | 5-6 | 80 | 60 | 60 | 80 | 60 | 60 | 60 | 60 |
| | 6-7 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| | 7-8 | 60 | 40 | 80 | 60 | 80 | 80 | 80 | 80 |
| LGPt (days) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | 1-59 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | 60-149 | 100 | 100 | 100 | 100 | 100 | 0 | 0 | 0 |
| | 150-209 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| | 210-269 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| | 270-365 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| FFD (days) | 0-100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | 100-200 | 80 | 80 | 80 | 0 | 80 | 0 | 0 | 0 |
| | 200-300 | 80 | 80 | 100 | 80 | 100 | 100 | 100 | 100 |
| | >300 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| Soil texture | Coarse | 60 | 60 | 20 | 60 | 40 | 40 | 60 | 60 |
| | Medium | 80 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| | Medium fine | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| | Fine | 40 | 60 | 60 | 80 | 60 | 60 | 40 | 60 |
| | Very fine | 20 | 0 | 20 | 60 | 0 | 0 | 0 | 60 |
| | No mineral soils | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

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Table S 4-3 Crops suitability scores for each biophysical variable class adapted from Perpiña Castillo et al. (2015). (continued)

| Biophysical variables | Classes | Lignocellulosic energy crops suitability to biophysical variables (%) | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Miscanthus | Switchgrass | Reed canary grass | Giant Reed | Cardoon | Willow | Poplar | Eucalyptus |
| Soil depth (cm) | <10 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | 10-20 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | 20-40 | 0 | 0 | 20 | 0 | 0 | 0 | 0 | 0 |
| | 40-60 | 20 | 20 | 40 | 0 | 20 | 0 | 40 | 40 |
| | 60-80 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
| | 80-100 | 60 | 60 | 80 | 80 | 60 | 60 | 60 | 60 |
| | 100-120 | 80 | 80 | 100 | 100 | 80 | 80 | 80 | 80 |
| | 120-150 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| Soil type | Alluvial deposits | 80 | 80 | 80 | 80 | 80 | 80 | 80 | 80 |
| | Other rocks | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Sandy materials | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| | Clayey materials | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| | Crystalline rocks | 40 | 40 | 40 | 40 | 40 | 40 | 40 | 40 |
| | Volcanic rocks | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Loamy materials | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| | Calcareous rocks | 60 | 60 | 80 | 60 | 60 | 60 | 60 | 60 |
| | Detrital formations | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
| Soil drainage | Excessively well drained | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| | Imperfectly drained | 0 | 0 | 0 | 0 | 0 | 40 | 0 | 0 |
| | Moderately drained | 60 | 60 | 60 | 80 | 60 | 60 | 60 | 60 |
| | Poor drained | 0 | 0 | 0 | 0 | 0 | 40 | 20 | 0 |
| | Temporary drained | 40 | 40 | 40 | 40 | 40 | 40 | 40 | 40 |
| | Very Poor drained | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Well drained | 80 | 80 | 80 | 80 | 80 | 80 | 80 | 80 |





Table S 4-3 Crops suitability scores for each biophysical variable class adapted from Perpiña Castillo et al. (2015). (continued)

| Biophysical variables | Classes | Lignocellulosic energy crops suitability to biophysical variables (%) | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Miscanthus | Switchgrass | Reed canary grass | Giant Reed | Cardoon | Willow | Poplar | Eucalyptus |
| Temperature (°C) | 0- (-)15 | 0 | 0 | 40 | 0 | 0 | 0 | 40 | 0 |
| | 0-4 | 0 | 0 | 60 | 0 | 0 | 20 | 40 | 0 |
| | 4-6 | 20 | 20 | 60 | 0 | 0 | 20 | 60 | 0 |
| | 6-8 | 100 | 60 | 60 | 0 | 0 | 80 | 100 | 0 |
| | 8-10 | 100 | 100 | 80 | 20 | 20 | 100 | 100 | 20 |
| | 10-15 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| | 15-20 | 20 | 20 | 40 | 100 | 100 | 40 | 40 | 100 |
| | >20 | 0 | 0 | 40 | 100 | 100 | 0 | 20 | 100 |
| Precipitation (mm) | 0-200 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | 200-400 | 0 | 0 | 60 | 0 | 40 | 0 | 0 | 40 |
| | 400-500 | 0 | 60 | 60 | 40 | 60 | 0 | 0 | 60 |
| | 500-600 | 40 | 60 | 60 | 60 | 60 | 0 | 0 | 60 |
| | 600-800 | 60 | 80 | 80 | 80 | 80 | 80 | 80 | 80 |
| | 800-1000 | 80 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| | >1000 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| Slope (%) | 0-2 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| | 2-5 | 80 | 80 | 80 | 80 | 80 | 80 | 80 | 80 |
| | 5-8 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
| | 8-16 | 40 | 40 | 40 | 40 | 40 | 40 | 40 | 40 |
| | 16-30 | 20 | 20 | 20 | 20 | 20 | 20 | 20 | 20 |
| | >30 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

### 4.8.4 Cultivation stage data

**Table S 4-4 Diesel usage and pesticide application for all lignocellulosic energy crops and emission factors**

| Lignocellulosic energy crop | Pesticides (kg ha$^{-1}$ year$^{-1}$) | Diesel for cultivation and harvesting (l ha$^{-1}$ year$^{-1}$) |
|---|---|---|
| Miscanthus | 0.3$^{A}$ | 38.3$^{D}$ |
| Switchgrass | 0.2$^{A}$ | 31$^{D}$ |
| Giant Reed | 0.2$^{A}$ | 38$^{D}$ |
| Reed Canary grass | 0.2$^{A}$ | 87$^{E}$ |
| Cardoon | 0.2$^{A}$ | 43$^{F}$ |
| Willow | 4$^{B}$ | 25$^{D}$ |
| Poplar | 4$^{C}$ | 30$^{D}$ |
| Eucalyptus | 1.6$^{C}$ | 41$^{C}$ |

A 347

B Based on data for Poplar [317]

C 317

D 348

E 349

F 350

### 4.8.5 Fertilizers inputs, direct and indirect N2O emissions methods and input data

#### Direct and indirect N2O emissions

Direct $N_2O$ emissions occur from the application of synthetic fertilizers and manure, crop residues, and from N mineralization associated with LUC induced loss of soil organic matter, see Equation S 4-1 [118]

**Equation S 4-1**

$$N_2O_{Direct} = (F_{SN} + F_{CR} + F_{SOM}) * EF_1 * 44/12$$

Where:
$N_2O_{Direct}$ = Annualized direct $N_2O$ emissions, *kg $N_2O$ year$^{-1}$*
$F_{SN}$ = Annual amount of applied N synthetic fertilizer, *kg N year$^{-1}$*
$F_{CR}$ = Annual amount of N in crop residues, *kg N year$^{-1}$*
$F_{SOM}$ = Annual amount of N in mineral soils that is mineralized in association with LUC, *kg N year$^{-1}$*
$EF_1$ = Emission factor for $N_2O$ emissions from N inputs, *kg $N_2O$–N kg N$^{-1}$*
*44/12* = Conversion factor from $N_2O$–N to $N_2O$

To simplify, in this study, it is assumed that the application of synthetic fertilizers meets crop nitrogen requirements. Balanced fertilization is considered to estimate the emissions from fertilizers use. Therefore, the nitrogen input rate is directly proportional to what is removed by harvesting the crop. An additional 15% is accounted for potential losses.





Table S 4-5 displays the crop-specific nitrogen content. The crop yield is location-specific. Default emissions factors (0.01, $EF_1$) are assumed [118].

The annual amount of N in crop residues includes above and below ground biomass residues (see Equation S 4-2). The above and below ground biomass residues depend on crop yield, root-to-shoot ratio, and harvest index. Therefore, it is location-specific. It is considered that crop residues are left on the field, in line with IPCC (2006). For perennial crops, the below-ground residues are only removed when the field is renewed (every 15 years). Default data of N content in residues are used for herbaceous energy crops [118]. For woody crops, there are no default values of N content in residues. Therefore, it was assumed that the amount of N content in woody crop residues was the same as the N content in the yield (see Table S 4-5).

**Equation S 4-2**

$$F_{CR\,i} = (AGR_i * N_{AGR\,i}) + (BGR_i * N_{BGR\,i})$$

$$AGR_i = yield_i / HI_i$$

$$BGR_i = (Yield_i + AGR_i) * R_i * Frac_{renew\,i}$$

Where:
$F_{CR}$ = Annual amount of N in crop residues, *kg N year$^{-1}$*
*i* = Crop type,
*AGR* = Annual amount of above ground crop residues, *kg AGR year$^{-1}$*
$N_{AGR}$ = N content in above ground residues, *kg N kg AGR$^{-1}$*
*BGR* = Annual amount of below-ground crop residues, *kg BGR year$^{-1}$*
$N_{BGR}$ = N content in below ground residues, *kg N kg BGR$^{-1}$*
*Yield*= Annual amount of harvested biomass, *kg ha$^{-1}$ year$^{-1}$*
*HI* = Harvest index, *%*
*R* = Ratio of below ground biomass to above ground biomass,
$Frac_{renew}$ = fraction of total area under crop i that is renewed. For lignocellulosic crops, which are renewed on average every 15 years, $Frac_{renew}$ = 1/15

The assessment of mineralized N is based on LUC-related soil organic matter (SOM) losses (see Equation S 4-3) [118]. The C:N ratio of SOM is applied to derive mineralized N from LUC changes in SOC. The calculation of the LUC-related SOC changes and related annualised carbon emissions are described in section 2.3.2. However, LUC can also result in SOM accumulation depending on the type of conversion. Potential N sequestration associated with SOM accumulation due to LUC is not accounted for in line with IPCC (2006). A default C:N ratio of 15 in line with the type of land use/covers in this study is applied [118].

**Equation S 4-3**

$$F_{SOM\,i} = (C_{mineral\,i} * 1/R_{CN}) * 1000$$

Where
$F_{SOM}$ = Annual amount of N in mineral soils that is mineralized in association with LUC, *kg N year*$^{-1}$
$i$ = Crop type,
$C_{mineral}$ = Average annual loss of soil carbon, *t C ha*$^{-1}$
$R_{CN}$ = C:N ratio of the soil organic matter,
*1000* = Conversion factor to convert t to kg

Indirect $N_2O$ emissions occur from N volatilization/deposition and leaching from managed soils. Leaching only occurs in very wet areas characterized by strong precipitation regimes or mainly when irrigation is used [118]. GHG emissions from leaching were not considered. $N_2O$ emissions from atmospheric deposition occur from the volatilization of N as $NH_3$ and $NO_x$ from managed soils, see Equation S 4-4[118]. Default values for the fraction of synthetic fertilizers that volatilizes (0.11, $Frac_{GASF}$) and default emissions factors (0.01, $EF_4$) are assumed [118].

**Equation S 4-4**

$$N_2O_{ATD} = F_{SN} * Frac_{GASF} * EF_4 * 44/12$$

Where:
$N_2O_{ATD}$ = Annual amount of $N_2O$ emissions from atmospheric deposition of volatilized N, *kg* $N_2O$ *year*$^{-1}$
$F_{SN}$ = Annual amount of applied N synthetic fertilizer, *kg N year*$^{-1}$
$Frac_{GASF}$ = Fraction of synthetic fertilizer N that volatilizes as $NH_3$ and $NO_x$, *kg* $(NH_3–N + NO_x–N)_{volatilized}$ *kg N*$^{-1}$
$EF_4$ = Emission factor for $N_2O$ emissions from atmospheric deposition of N on soils, *kg* $N_2O–N$ *kg* $(NH_3–N + NO_x–N)_{volatilized}^{-1}$
*44/12* = Conversion factor from $N_2O–N$ to $N_2O$

**Table S 4-5 Crop specific yield mineral content, used as input for calculating GHG emissions from fertilizers use, adapted from Dees et al. (2017).**

| Lignocellulosic energy crop | N content (kg $t_{dry}^{-1}$) | $P_2O$ content (kg $t_{dry}^{-1}$) | $K_2O$ content (kg $t_{dry}^{-1}$) | CaO content (kg $t_{dry}^{-1}$) |
|---|---|---|---|---|
| Miscanthus | 6.3 | 2 | 8.1 | 5.7 |
| Switchgrass | 4.7 | 2 | 11.8 | 1.2 |
| Giant Reed | 9.9 | 1.1 | 8.8 | 20.5 |
| Reed Canary grass | 5 | 0.3 | 2.9 | 7.4 |
| Cardoon | 13 | 2.3 | 2.4 | 3.1 |
| Willow | 4 | 0.4 | 3.5 | 7.7 |
| Poplar | 3 | 0.1 | 1.4 | 5.3 |
| Eucalyptus | 11 | 0.6 | 24.1 | 3.2 |





### 4.8.6 Root to shoot ratios

**Table S 4-6 Crops specific root-to-shoot ratios**

| Crop | **Root-to-shoot ratio** ($BG_r$) |
|---|---|
| Miscanthus | 0.49[A] |
| Switchgrass | 0.48[B] |
| Giant reed | 0.56[C] |
| Reed Canary grass | 0.48[D] |
| Cardoon | 0.80[E] |
| Willow | 0.48[F] |
| Poplar | 0.30[G] |
| Eucalyptus | For $AB_s$ < 50, then $BG_r$ 0.44. For $AB_s$ > 50, then $BG_r$ 0.28[H] |

A 351–354
B354–357
C358–360
D361
E362,363
F364–367
G368,369
H118



### 4.8.7 Spatially explicit calculation of biomass carbons stock of each land use/cover category

#### Established energy crops

It is assumed that land remaining in the same use/cover category has a biomass carbon net balance of 0 [118]. In this case, Established energy crops are categorized as land that remains in use for the potential production of the 8 lignocellulosic energy crop over each subsequent time step.

#### Shrubland

The soil productivity map for grasslands [319] and the climate-zone-specific default values of grassland biomass yield in the IPCC (2006) were used as proxies to estimate shrublands above ground biomass spatially explicitly. With an overlay assessment and based on proximity, soil productivity values were assigned to the shrubland area. Then it is assumed that the maximum soil productivity values are correlated with the IPCC climate-zone-specific peak above ground biomass coefficients. The above ground biomass of shrubland was determined from each location-specific soil productivity value by considering the established relation between maximum soil productivity and peak biomass coefficients. The below-ground biomass was obtained by applying a 2.8 ratio between below and above ground biomass [118]. Biomass carbon stocks were derived using a default 0.5 carbon fraction coefficient for woody biomass [118].

### Open space suitable

For Open space suitable, the soil productivity map of grassland [319] and climate-zone dependent default values for peak above ground biomass of grasslands of the IPCC (2006) were used as a proxy to estimate above ground biomass for this land use/cover category spatially explicitly. The same approach as for shrubland was applied. Unlike the approach for shrubland , the above to below ground biomass ratios for this category are climate-zone dependent (IPCC 2006); 4 for Boreal/Cold temperate/Warm temperate – Wet and 2.8 for Cold temperate/Warm temperate/Tropical – Dry climate zones. Biomass carbon stocks were derived by applying a default 0.47 carbon fraction coefficient for herbaceous biomass [118].

### Abandoned land

Abandoned agricultural land is assumed to come from annual cropland. Energy crops are placed in this land recently after the annual cropland harvest. Therefore, it was considered that little biomass was present in this category prior to conversion and the biomass carbon stock pool can be assumed as 0 [118].





## 4.8.8 SOC stock change factors

**Table S 4-7 Relative SOC stock change factors valid for each land use/cover category 118**

| Land use/cover categories | Climate region | Relative stock change factors in accordance to IPCC | | |
|---|---|---|---|---|
| | | $F_{LU}$ | $F_{MG}$ | $F_{I}$ |
| 8. Shrublands [A] | | 1[B] | 1[C] | 1[D] |
| 10. Open space suitable [E] | Temperate/Boreal climates | 1[B] | 0.95[F] | 1[D] |
| | Tropical | | 0.97[F] | |
| 11. Abandoned land [G] | Temperate/Boreal dry | 0.80[H] | 1[I] | 1.04[J] |
| | Temperate/Boreal moist / tropical wet | 0.69[H] | | 1.11[J] |
| | Tropical dry | 0.58[H] | | 1.04[J] |
| Lignocellulosic energy crops (herbaceous) | Temperate/Boreal dry | 0.93[k] | 1.02[L] | 1[M] |
| Temperate/Boreal and tropical wet | | 0.82[k] | 1.08[L] | |
| | Tropical dry | 0.93[k] | 1.09[L] | |
| Lignocellulosic energy crops (woody) | Temperate/Boreal dry | 1[N] | 1.02[L] | 1[M] |
| | Temperate/Boreal wet | | 1.08[L] | |
| | Tropical dry | | 1.09[L] | |

[A] All stock change factors for Shrubland are assigned from IPCC chapter 6 Grasslands, given that there are no specific values for Shrubland
[B] Value for all permanent grasslands
[C] Value for non-degraded without significant management improvements
[D] Value for grasslands where no additional management inputs have been used
[E] All stock change factors for Open space suitable are assigned from IPCC chapter 6 Grasslands given that this land use/cover definition includes a like habitat such as moors or and heathland
[F] This category includes burnt areas and areas where grazing has occurred. Therefore, the value for moderately degraded grassland was applied. The Climate zone tropical is very limited to a few areas in the south of Europe.
[G] This category is assumed to be abandoned agricultural land that has been under the use of annual cropland (for long term) and to be available for energy crops allocation recently after harvest. Therefore, stock change factors for this category are assigned based on IPCC annual cropland default values.
[H] Values for areas that have been continuously managed to predominantly annual crops
[I] Tillage practices are assumed to take place to produce annual crops
[J] Given the difficulty to cover the yearly different crop types and used inputs for each of them. It was assumed a high input factor.
[K] Value for perennial grasses
[L] Given these crops managements characteristics, full till is not needed [50] and therefore, reduce till with little soils disturbance is considered
[M] It was considered that after harvesting, the residues are left on the field and no additional organic matter is needed (in line with the biomass section). Value for crops with medium input
[N] Value for Long-term perennial tree crops

### 4.8.9 Input data for conversion processes

**Table S 4-8 Input data and assumptions for conversion processes, adapted from Vera and Hoefnagels (2020).**

| Process | Unit | Ethanol | ATJ | Pyrolysis + upgrading | Gasification | | |
|---|---|---|---|---|---|---|---|
| Sub-process | | Dilute acid, hydrolysis | | Hydrogen supply | FT synthesis | Methanol synthesis | DME synthesis |
| **Main output** | | | | | | | |
| Type | | Ethanol | RJF | Pyrolysis - gasoline | Syn diesel (BtL) | Methanol | DME |
| Conversion efficiency | $MJ_{main\ output}\ MJ_{feedstock}^{-1}$ | 0.26 - 0.34[A] | 0.67 | 0.30 | 0.38 | 0.61 | 0.60 |
| **Co-production**[B] | $MJ\ MJ_{main\ output}^{-1}$ | Electricity: 0.4 | Diesel: 0.12<br>Naphtha: 0.21 | Diesel: 0.37<br>RJF: 0.13<br>HFO: 0.28<br>Electricity: 0.06 | Electricity: 0.24 | Electricity: 0.01 | Electricity: 0.04 |
| **Inputs (without allocation)** | | | | | | | |
| **Feedstock** | | | | | | | |
| Feedstock type | | Herbaceous and woody energy crops | Ethanol | Woody energy crops | Woody energy crops | Woody energy crops | Woody energy crops |
| **Utilities** | | | | | | | |
| Electricity[C] | $MJ\ MJ_{main\ output}^{-1}$ | | 0,03 | 0 | | | |
| Hydrogen | $MJ\ MJ_{main\ output}^{-1}$ | | 0,08 | 0,69 | | | |
| **Chemicals:** | | A detailed list of chemicals used in the conversion processes is provided in [370] | | | | | |
| **Main data source** | | 314 | 196 | 196 | 314 | 326 | 326 |

[A] Conversion efficiency varies between feedstock types depending on the cellulose and hemicellulose content
[B] Electricity demand excluding electricity for hydrogen production through electrolysis
[C] Co-production is normalized per main unit output.





**Table S 4-9 Input data and assumptions for hydrogen production 317**

| Input | Output | Unit | SMR of natural gas |
|---|---|---|---|
| Natural gas[A] | | $MJ_{NG}\ MJ_{hydrogen}^{-1}$ | 1.375 |
| Electricity[A] | | $MJ_{NG}\ MJ_{hydrogen}^{-1}$ | 0.033 |
| | Hydrogen | MJ | 1 |
| | GHG emissions | $g\ CO_{2\text{-}eq}\ MJ_{hydrogen\text{-}1}$ | 95.31 |

[A] Natural gas, 2500, EU Mix quality: 68.89 g $CO_{2\text{-}eq}\ MJ_{NG}^{-1}$
[B] Current EU mix, EU MV: 141.1 g $CO_{2\text{-}eq}\ MJ_e^{-1}$

**Table S 4-10 Input data and assumptions for transport, blending depot and filling station 314**

| Assumption | Transport mode | Payload (t) | Distance (km, one way) | Share (%) | Fuel use | Electricity ($MJ_e\ MJ_{fuel}^{-1}$) |
|---|---|---|---|---|---|---|
| *Transport from biomass source to the conversion plant* | | | | | | |
| Herbaceous lignocellulosic energy crops (bales) | Truck | 40 | 50 | - | Diesel | - |
| Woody lignocellulosic energy crops (chips) | Truck | 40 | 50 | - | Diesel | - |
| *Transport from processing plant to a blending depot* | | | | | | |
| Advance biofuel<br>Truck<br>40<br>305 | | | | 13.2 | Diesel | - |
| | Product tanker | 15000 | 1118 | 31.6 | HFO | - |
| | Inland ship/barge | 1200 | 153 | 50.8 | Diesel | - |
| | Train | - | 381 | 4.4 | Diesel | - |
| Energy consumption at blending depot | Blending depot | - | - | - | - | 0.00084 |
| *Transport from blending depot to filling station* | | | | | | |
| Advance biofuel | Truck | 40 | 150 | 100 | Diesel | - |
| Energy consumption at filling station | - | - | - | - | - | 0.0034 |

4

## 4.8.10 Land availability

**Table S 4-11 Amount of marginal land available for each time step.**

| Land use/cover | Time step (ha) | | | |
|---|---|---|---|---|
| | 2020 | 2030 | 2040 | 2050 |
| Abandoned land | 501,200 | 290,200 | 826,000 | 43,900 |
| Established energy crops | 0 | 514,800 | 713,000 | 1,382,800 |
| Shrubland | 15,936,900 | 15,376,900 | 15,197,400 | 15,355,600 |
| Open space suitable | 4,378,000 | 4,361,100 | 4,295,800 | 4,233,600 |

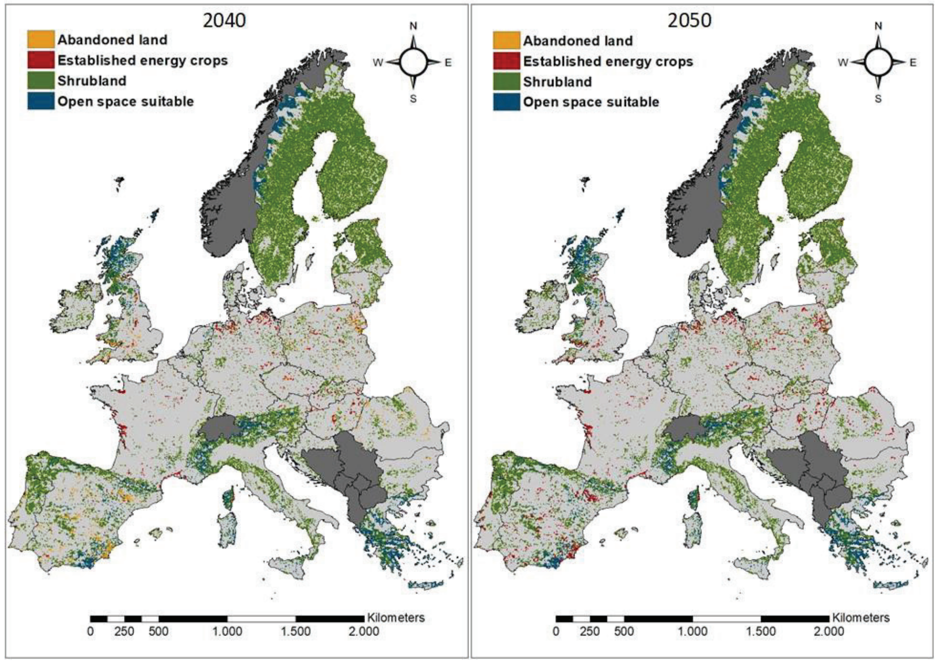



**Figure S 4-1 Spatial distribution and land-use/cover of marginal land that meets REDII sustainability criteria in 2040 and 2050. The pixel size is enhanced for displaying purposes**





## 4.8.11 Biomass potentials

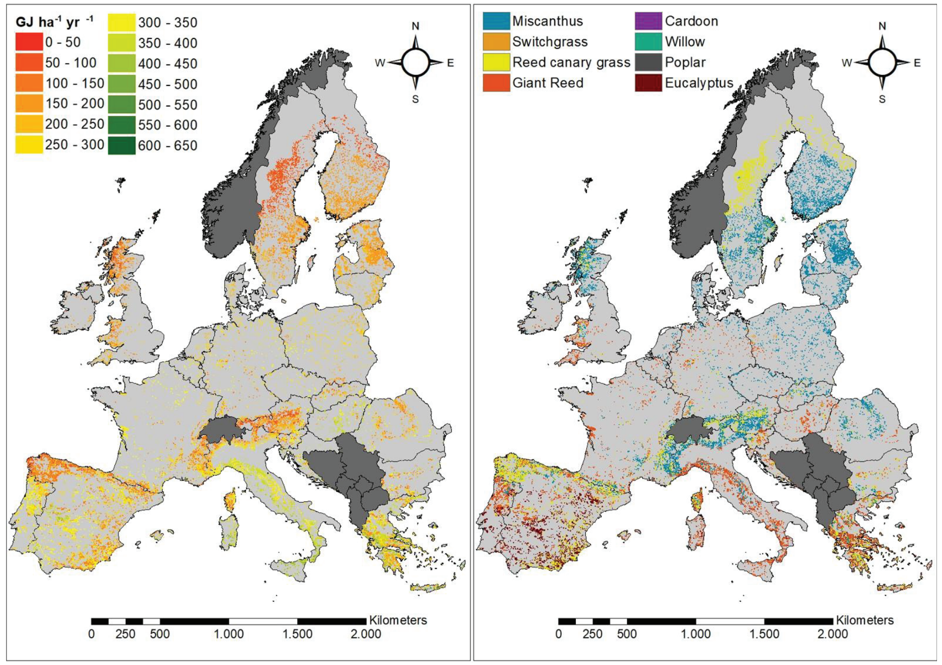



**Figure S 4-2 Maximum–yield biomass potential (for each location, the lignocellulosic energy crop with the highest yield is selected) in 2040. Left-hand panel: biomass potential in GJ ha–1 year–1. Right-hand panel: the corresponding lignocellulosic energy crop selected for each location.**

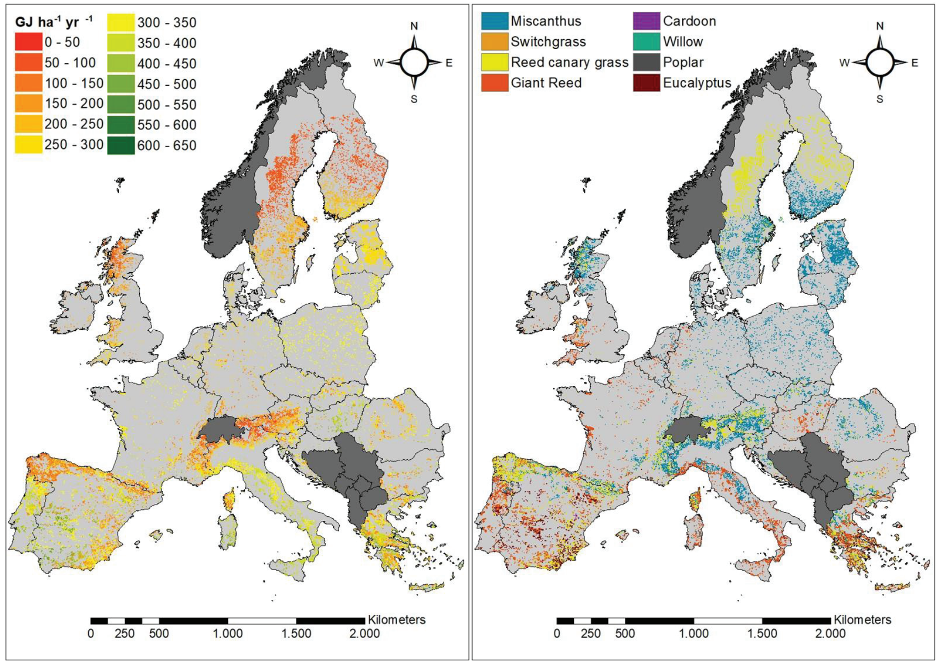



**Figure S 4-3 Maximum–yield biomass potential (for each location, the lignocellulosic energy crop with the highest yield is selected) in 2050. Left-hand panel: biomass potential in GJ ha-1 year-1. Right-hand panel: the corresponding lignocellulosic energy crop selected for each location.**

### 4.8.12 Spatially explicit results for all advanced biofuel supply chains

The 28 different maps can be seen here: https://onlinelibrary.wiley.com/action/downloadSupplement?doi=10.1111%2Fgcbb.12867&file=gcbb12867-sup-0008-Supinfo.docx

### 4.8.13 Sensitivity analysis

The assessment of biomass potentials and GHG emissions is affected by the assumptions and approaches selected in this study. The assessment was performed explicitly under REDII sustainability criteria. Other relevant environmental aspects that are not mentioned in REDII were not considered. However, bioenergy systems can have severe implications for biodiversity and water quantity [371,372]. To understand the impact of accounting for these environmental aspects, a sensitivity analysis was carried out. The choice to assess the results while considering water use and biodiversity aspects was selected based on their influence on the results. Accordingly, two scenarios were developed. First, it was assumed that crop production was carried out under water-limited conditions. Therefore, no irrigation is applied to avoid depleting water resources. Under these conditions, yields can reach 50% of the gross biomass potential and crop growth is determined by rain-fed conditions [306]. Second, it was assumed that land categorized as shrublands could not be dedicated to producing energy crops. Shrublands contribute to several ecosystem services, such as maintaining biodiversity [331], and thus, should be conserved. For the water-limited conditions scenario, results are shown for biomass potentials and GHG emissions. For the biodiversity scenario, results are only shown for biomass potentials.





Biomass potentials reduce significantly when crops are grown under water-limited conditions (see Figure S 4-4). All crop potentials have a significant reduction proportional to the amount delivered under no water-limited conditions. Under these conditions, there is a 44% reduction of biomass potentials for every crop. For maximum-yield biomass potential, this reduction translates into 867 PJ produced less for 2030 and up to 1006 PJ for 2050. The reduction varies between 172 PJ $yr^{-1}$ and 659 PJ $yr^{-1}$, depending on the point in time for other crops. Not applying irrigation to avoid depleting water resources has a significant impact on the results. A reduction in biomass potentials will also lead to less production of advanced biofuels in the same order of magnitude. On average, GHG emissions increase between 7 to 25 g $CO_{2\text{-eq}}$ $MJ_{fuel}^{-1}$ for herbaceous crop pathways, and most surpass REDII GHG thresholds (see Figure SM4). However, there are some locations where emissions can rise to 135 g $CO_{2\text{-eq}}$ $MJ_{fuel}^{-1}$ (reed canary grass RJF). A reduction in yield translates into more than 50 g $CO_{2\text{-eq}}$ $MJ_{fuel}^{-1}$ for the reed canary grass RJF pathway (for some locations) compared to GHG emissions under no water-limited conditions. A yield reduction leads to less carbon stored in the biomass section (for several locations) than the land prior to conversion. The effect is more acute in regions where the extreme biophysical conditions already constrain biomass growth. Therefore, higher GHG emissions are reached in those locations, especially for the herbaceous crops produced in such sites. For woody crop pathways, the effect of water-limited conditions on GHG emissions is less significant. For these pathways, a large share of the carbon accumulation occurs in the SOC pool.

There is a strong reduction in biomass potentials between 248 PJ $yr^{-1}$ to 1491 PJ $yr^{-1}$ depending on crop type and relevant point in time when the conversion of shrublands is avoided (see Figure S 4-5). Preventing the conversion of shrublands to maintain biodiversity at those locations results in a reduction of available land. Approximately 15 Mha of shrublands are dedicated to producing energy crops (see Table S 4-11) when this land category is considered available. The projected biomass potential relies to a large extent upon the conversion of shrublands. Consequently, considering the environmental aspect of biodiversity to maintain shrublands will also reduce significantly advanced biofuels production.

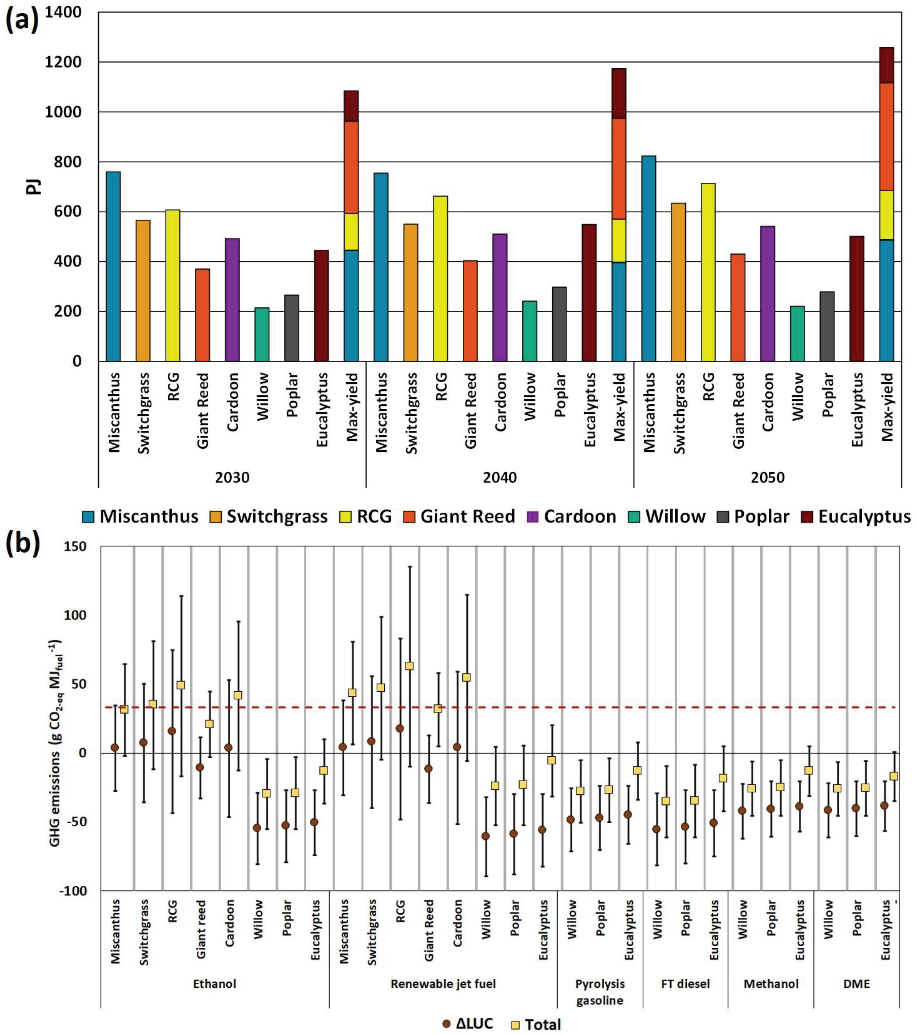



**Figure S 4-4 Biomass potentials and GHG emissions of the water-limited conditions scenario. (a) GHG emissions of advanced biofuel pathways with two standard deviations, including land-us-erelated net changes in carbon stocks in Europe for 2030. The ranges indicate the values within two times the standard deviation from the average in both directions (positive and negative). Thus, these ranges reflect spatial variability of GHG emissions due to the heterogeneity in bio-physical conditions. (b) Biomass potentials of eight lignocellulosic energy crops cultivated at available marginal land that complies with sustainability criteria of the REDII. Max-yield refers to the maximum yield biomass potential for which for each location the lignocellulosic energy crop with the highest attainable yield is selected. RCG, Reed Canary Grass**





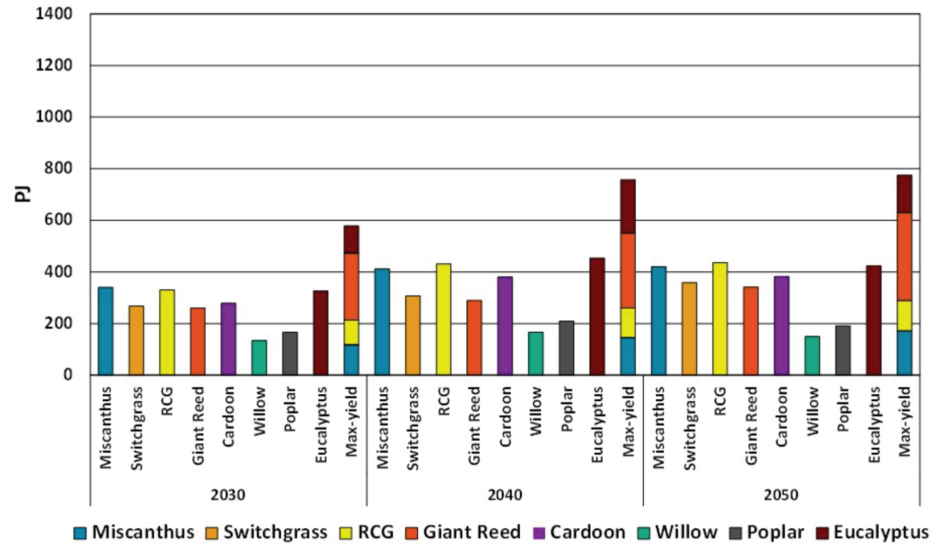


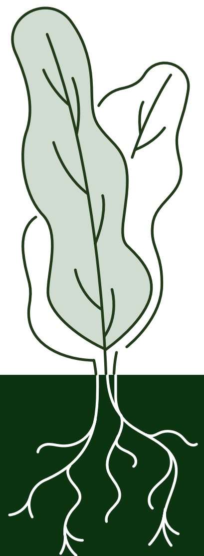

**Figure S 4-5 Biomass potentials of eight lignocellulosic energy crops cultivated at available marginal land that complies with sustainability criteria of the REDII for the biodiversity sce-nario. Max-yield refers to the maximum yield biomass potential for which for each location the lignocellulosic energy crop with the highest attainable yield is selected. RCG, Reed Canary Grass**

### 4.8.14 Supply chains GHG mitigation potentials

**Table S 4-12 Supply chains GHG mitigation potential**

| Advanced biofuel pathway | Overall GHG emissions mitigation potential (Mt $CO_{2\text{-eq}}$ $yr^{-1}$) |
|---|---|
| Miscanthus to ethanol | -28.3 |
| Switchgrass to ethanol | -22.8 |
| Reed canary grass to ethanol | -22.8 |
| Giant Reed to ethanol | -13.7 |
| Cardoon to ethanol | -19.6 |
| Willow to ethanol | -11.8 |
| Poplar to ethanol | -14.8 |
| Eucalyptus to ethanol | -21.5 |
| Miscanthus to RJF | -16.3 |
| Switchgrass to RJF | -13.2 |
| Reed canary grass to RJF | -12.8 |
| Giant Reed to RJF | -8.1 |
| Cardoon to RJF | -11.1 |
| Willow to RJF | -7.1 |
| Poplar to RJF | -9 |
| Eucalyptus to RJF | -12.7 |
| Willow to pyrolysis gasoline | -12.6 |
| Poplar to pyrolysis gasoline | -16 |
| Eucalyptus to pyrolysis gasoline | -23.4 |
| Willow to FT-diesel | -17 |
| Poplar to FT-diesel | -21.5 |
| Eucalyptus to FT-diesel | -31.2 |
| Willow to methanol | -25.6 |
| Poplar to methanol | -32.2 |
| Eucalyptus to methanol | -48.1 |
| Willow to DME | -25 |
| Poplar to DME | -31.6 |
| Eucalyptus to DME | -48.8 |
| each location the advanced biofuel supply chain with the best GHG performance | -65.4 |

4



# 5

## **Bioenergy potential from Invasive alien plants:** analysis of environmental and socioeconomic impacts in Eastern Cape, South Africa

**Published as:**. *Vera, I., Goosen, N., Batidzirai, B., Hoefnagels, R., & van der Hilst, F. (2022). Bioenergy potential from invasive alien plants: Environmental and socio-economic impacts in Eastern Cape, South Africa. Biomass and Bioenergy, 158, 106340.*

## ABSTRACT

South Africa's natural resources and ecosystems are negatively affected by Invasive Alien Plants (IAPs). We used a life-cycle approach to assess the environmental and socio-economic impacts of using IAPs for electricity generation in South Africa or exported and used for electricity generation in the Netherlands. Supply chain greenhouse gas (GHG) emissions of electricity from IAPs pellets, excluding land use change-related GHG emissions, are 31.5 $gCO_2eq\ MJ^{-1}$ for electricity generation in South Africa and 31.2 $gCO_2eq\ MJ^{-1}$ for electricity generation in the Netherlands. An additional 3.9 $gCO_2eq\ MJ^{-1}$ is accounted for if emissions of land use change are included and land is rehabilitated to its natural state. The removal of IAPs results in water savings when considering any potential land use transition, ranging between 1,263 mm $year^{-1}$ for annual cropland to 12 mm $year^{-1}$ for dense forest. The supply chain costs of pellets are 5,344 ZAR $Mg^{-1}$ (285 € $Mg^{-1}$) delivered at the power plant in South Africa and 2,535 ZAR $Mg^{-1}$ (159 € $Mg^{-1}$) delivered at Rotterdam port. Direct full-time jobs generated from removing IAPs up to the conversion-factory-gate are 604 FTE $year^{-1}$ for South Africa and 525 FTE $year^{-1}$ for the Netherlands. There are clear trade-offs between environmental and social benefits and costs. There are generally net carbon losses when considering the land use transitions after IAP removal, even when land is rehabilitated to its natural state. Using IAPs for electricity can be a valuable strategy for South Africa to generate employment, conserve water resources and reduce GHG emissions.





## 5.1 INTRODUCTION

To reduce the risk of serious impacts from climate change and avoid a temperature rise of more than 2°C compared to preindustrial levels, deep greenhouse gas (GHG) emissions reductions are required [3]. For developing countries, such as South Africa (SA), reducing GHG emissions is challenging, as these countries also face other urgent development goals, e.g. reducing poverty and inequality [373]. South African energy supply is still dominated by coal for electricity generation, contributing approximately 85% to national GHG emissions [374]. In addition, SA is the 12th largest GHG emitter globally, with 434.5 Tg of $CO_2$ emitted in 2020 (from the consumption of oil, gas and coal for combustion), equivalent to 7.6 Tg $CO_2$ per capita (37th globally) [375]. Nevertheless, SA aims to maintain current GHG emission levels and not surpass the limit of 614 Tg of $CO_2eq$ $year^{-1}$ before 2030; after 2030, SA has planned to reduce GHG emissions to 212 Tg - 428 Tg of $CO_2eq$ $year^{-1}$ by 2050 [373]. To meet SA's and global GHG emission reduction targets, the development and deployment of renewable energy is crucial. Despite that photovoltaics and wind are expected to dominate the contribution to renewably energy [374,376], biomass is also expected to play an essential role in reducing the country's high coal dependency and meeting GHG emissions reduction targets [377].

The potential biomass supply for energy purposes in SA is currently limited and consists mainly of residues from agriculture and forestry [378]. Other biomass sources, such as IAPs, are recognized as promising feedstock. The use of IAPs for bioenergy could potentially result in carbon savings and, at the same time, deliver additional environmental and socio-economic advantages [379,380]. Invasive alien tree species such as pine and wattle (acacias) limit water availability [10]. Compared to the natural landscape (e.g., savannah/grasslands), these IAPs increase transpiration and evaporation losses and are characterized by a deep-rooted system that allows them to access deeper stored soil moisture [381]. It is estimated that IAPs reduce the country's water availability by 4% and without eradication measures, this can potentially increase up to 16% [382] and thereby aggravate droughts. For SA, this is a significant impact as the country has experienced an increase in the intensity of drought spells, leading to water crisis episodes in the last years [383]. In addition, the intensity and frequency of these extreme droughts are expected to increase further, driven by climate change [1]. Eradicating IAPs also benefits biodiversity as it helps protect and restore natural areas of endemic ecosystems such as the fynbos shrublands [384]. Removing IAPs has been a key priority since the introduction of the 'Working for Water' program (1995), which led to the treatment of more than 25,000 $km^2$, creating more than 250,000 direct jobs in the process (up until 2017) [385]. Despite the potential benefits of job creation, water source protection, and other ecosystem services, only a relatively small proportion of the estimated invaded area has been treated [384]. Approximately 114,000 ha of invasive alien trees have been submitted to initial treatment. However, it is estimated that invasive trees have invaded more than 1 million ha, and many species are now entering a phase of exponential growth [385,386]. It is estimated that IAPs could supply 11.3 Tg of solid biomass annually in SA [378]. However, this biomass source is currently largely underutilized as a result of logistical limitations [387]. After clearing, most of the IAP biomass is left on-site and therefore, potential valuable opportunities to use IAP biomass are wasted [380].

Using IAPs for energy purposes has emerged as an important sustainable development strategy to mitigate GHG emissions from non-renewable energy sources, protect the country's water resources, create job opportunities and protect biodiversity. Yet, such a strategy's success will rely largely on biomass availability, supply chain costs (including IAPs clearing and transport), benefits for water availability, and GHG emission savings. So far, IAPs assessments are generally targeted to biomass availability or biomass utilization costs (mainly focused on IAP's clearing) [377,378,388–391]. Few studies analyze the entire supply chain on utilizing IAPs for energy purposes [378,380]. Generally, the costs of using IAPs for bioenergy are high because of challenging logistics. Therefore, using IAPs is only feasible when subsidies are provided [387]. However, using IAPs for electricity generation can result in significant GHG savings. Net GHG savings between 69 to 250 g $CO_2$eq $MJ^{-1}$ can be achieved compared to fossil counterparts (coal-dominated electricity) and excluding carbon stock changes from Land Use Change (LUC) [378]. Nevertheless, when LUC-related GHG emissions are accounted for, these net GHG emissions savings can be substantially reduced if a net carbon loss occurs, given the difference in carbon storage between IAPs and subsequent potential land uses (e,g., rangeland). Studies on the effect of IAPs on water availability have been limited to water savings/runoff from IAPs eradication [381,391–393]. For employment generation, only direct jobs related to the eradication process are generally accounted for. Therefore, a more integrated approach that includes important aspects such as LUC, water impacts, job creation, and supply chain costs is necessary to understand the overall environmental and socio-economic impacts and trade-offs from using IAPs for energy production. Such integrated assessment can support the development of the bioenergy sector in SA and assist efficient biomass use. Furthermore, it can contribute to meeting SAs GHG emissions reduction targets, job creation (poverty alleviation) and protection of water resources.

The main goal of this study is to assess the environmental and socio-economic impacts of using biomass from IAPs for electricity generation in SA. An international supply chain was also considered for comparison purposes, in which biomass is exported and used for electricity generation in the Netherlands (NL). The Netherlands was selected as it is one of the main importing countries of wood pellets [204]. Furthermore, Dutch biomass imports are projected to continue growing to meet domestic bioenergy demand





[198,199]. This study assesses the socio-economic and environmental impacts of different landscape restoration scenarios. This study focuses on the Eastern Cape province of South Africa, as this province has the highest IAP biomass potentials in SA [378].

## 5.2 MATERIAL AND METHODS

The environmental and socio-economic impacts of using IAPs for bioenergy were assessed following an integrated modelling approach that considered the social, economic and environmental context of the Eastern Cape province in SA. The following processes were included in the assessment: biomass harvesting/eradication, logistics and conversion, and land restoration after IAPs removal (taking into account local biophysical characteristics). This approach considers the potential environmental and socio-economic impacts that differ considerably for different IAPs species and successive land uses.

### 5.2.1 Scope and scenarios

To quantify the environmental and socio-economic impacts of the biomass supply chains and ensure the comparability of results, each assessed system must serve the same function. Electricity generation is considered the main end-use function. We divided the assessment into three steps (see Figure 5-2).



Scope 1: Currently, the eradication of IAPs is limited. When land is cleared, biomass is left on-site, used for fuelwood by the local communities, or sold to timber companies [380]. Regardless of the final use, IAP clearing is set to continue in line with the Working for Water program. Therefore, the first step of the assessment focuses on assessing the impacts of land use transitions when IAPs are cleared and the land is rehabilitated. In this step, the scope is limited to only LUC-related impacts. Different successive land use scenarios are included after IAPs are removed (see Figure 5-2 and section 5.2.1.4). This is done to consider and compare the potential impacts of different scenarios for landscape restoration. Indirect effects along the supply chain, such as indirect LUC-related GHG emissions are not considered. For example, nitrous oxide emissions from fertilizers when land is dedicated to agriculture after IAPs eradication.

Scope 2: In the second step, the scope is expanded to include the impacts of using biomass to produce wood pellets. For this step, the system boundaries cover all stages: biomass extraction (IAPs clearing and collection), debarking/chipping on-site, forwarding and transportation to the pellet plant, and pelletization. Therefore, the second step focuses on the impacts of producing wood pellets, excluding LUC impacts.

Scope 3: For the third and final step, the conversion to electricity is considered. This step compares the implications of generating electricity from IAPs pellets in SA or exporting pellets for electricity generation in the Netherlands. The analysis of water savings is limited to scope 1 (LUC-related impacts), as water use in the rest of the supply chain (steps 2 and 3) is marginal [394]. The overall results of the supply chain GHG emissions (scope 3) are presented with and without LUC-related emissions. For the supply chain costs and job creation, LUC effects are excluded.

#### 5.2.1.1 Geographical scope

For the analysis, we focus on Port Elizabeth and its surroundings. We selected Port Elizabeth as it has an existing wood pelletization plant with a 120,000 Mg year-1 capacity. It also has access to two large operational harbours: Port Elizabeth and the recently built Port of Coega. Both have the required facilities for bulk export. The geographical scope was established by accounting for the service area (see section 5.2.1.3) required to meet the annual biomass demand of the pellet plant (biomass availability). This service area partially covers the municipalities of Kouga, Sundays river valley, and Nelson Mandela bay (see Figure 5-1).

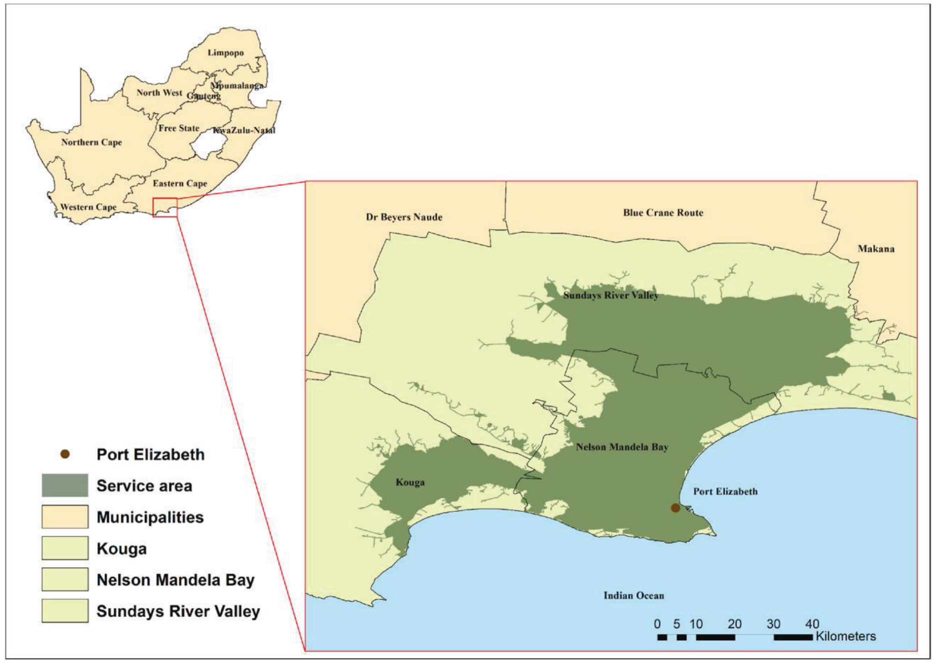



**Figure 5-1 The geographical scope of the study. Map of South Africa with the selected focus area in the zoom-in section.**





#### 5.2.1.2 Temporal scope for impacts

GHG emissions other than $CO_2$ ($CH_4$ and $N_2O$) are expressed in $CO_2$ equivalent ($CO_2$eq) for a global warming potential (GWP) impact calculated over 100 years (GWP100) consistent with the characterization factors used in the REDII [15]. In addition, it is required to account for the different time horizons related to each environmental impact or process. For example, annual soil carbon fluxes from LUC are assessed for a 20-year horizon as it is assumed that it takes 20 years for carbon pools to reach equilibrium [118]. For water and socio-economic analysis, all parameters are assessed on an annual basis. The temporal effect of the different parameters is presented in each subsection.

#### 5.2.1.3 Availability, type and distribution of IAPs

The availability and type of IAP that is eradicated and used for bioenergy largely affect the environmental and socio-economic impacts. This study focused on invasive trees since they offer higher biomass potentials than other invasive plants types in SA [378]. Woody biomass potentials were derived from South African Bioenergy Atlas [378]. This data set provides spatially explicitly the annual amount of standing (i.e. total) and exploitable woody biomass available per polygon (without species composition) for a 20-year time period. We used the exploitable woody-biomass potentials for this study, which refers to the amount of available biomass suitable for pelletization given its typical mass, age, and ease of access. Exploitable woody-biomass potentials are given in reference to the area of each polygon and available biomass is relative to the biophysical characteristics of each polygon (e.g. terrain conditions such as slope). The service area was estimated through a spatially explicit approach (GIS tools) based on exploitable woody biomass potentials, local infrastructure (i.e., roads) [378] and annual pellet demand. Therefore, the service area (Figure 5-1) represents the optimized aggregated distance required to supply sufficient biomass to meet the annual demand of the pellet plant located in Port Elizabeth.

Given the lack of high-resolution data, the tree species composition was estimated by overlapping the inventory data on cleared sites in SA [395] with the service area. Accordingly, in previously cleared sites within the service area, the invaded areas were dominated mainly by acacia (68%), eucalyptus (13%) and pine (12%), and other species (7%). Given their minimal share, these other species are not considered in the assessment. To determine the cover dominance, we assumed that one unit of area is dominated only by IAPs. This approach is generally followed by similar studies in which one unit of area (e.g. hectare) is assumed to be covered 100% by IAPs ("condensed" area) [396–398]. Therefore, we assumed that one unit of the ("condensed") area covered by IAPs is dominated only by acacia, eucalyptus and pines. We extrapolated the dominance shares found with the overlapping exercise to assume an IAPs distribution of 73% acacia, 14% eucalyptus and 13% pine.

#### 5.2.1.4 Post-removal land use scenarios

The different land use scenarios after IAP removal are based on [380,399,400]. It is suggested that after IAP removal, the cleared land could be restored to its natural state or used for agriculture and, therefore, recover and/or provide different ecosystem services [380]. It is assumed that after eradication, there is no regrowth of IAPs. We carried out two overlay exercises to identify the potential land use transitions after removing IAPs. First, we overlayed the most recent (2018) SA land cover data [399] with the service area. Second, we overlayed the SA vegetation map [400] with the service area. The vegetation data set provides the spatial distribution of vegetation classes in line with the country's bioregions and provides vegetation groups with similar biotic and abiotic features.

The composition of land cover within the service area is predominately characterized as dense forest & woodland (26%), natural grasslands (20%), fynbos (7.8%), contiguous low forest & thicket (7.1%), annual crops (7%), cultivated orchards (3.8%) and fallow land (3.1%) used as pastures [399]. Accordingly, seven post-removal land use scenarios were considered for the assessment: Dense forest, natural grasslands, fynbos, low forest and thicket, annual cropland, orchards and pastures. Citrus crops, grains/cereals (dominated by barley) and grasses are the main crops produced in Kouga, Sundays river valley and Nelson Mandela bay localities [401]. Hence, for the orchards land use scenario, we assume the production of citrus crops and for the annual cropland scenario we assume the production of grains/cereals (barley). For simplicity reasons and a lack of high-resolution data on the exact location of invaded areas, we assumed that any land use transition scenario could occur after IAP removal if the local biophysical conditions are adequate. All the scenarios are assessed individually.

The composition of vegetation classes according to biotic and abiotic features within the service area is mainly characterized by the thicket (56%) and fynbos (33%) [400]. The remaining share is a combination of vegetation with large shares of natural grasslands. Therefore, the remaining share (11%) is assumed to be natural grassland. We used the thicket, fynbos, and natural grassland shares for the natural restoration land use scenario (8th post-removal scenario). Thus, the natural restoration land use scenario combines the most relevant vegetation classes present in the region. In reality, land restoration will be limited by the local social and biophysical characteristics.

#### 5.2.1.5 Supply chain scenarios

Figure 5-2 shows the different assessment steps and supply chain stages. In SA, the eradication of IAPs involves the harvesting of trees mainly with (chain) saws and chemical treatments; followed in the course of the next five years by several treatments (1-3 years' intervals depending on plant species) to remove (potential) new growth [402]. In this study,





it is assumed that trees are harvested manually with chain saws. After clearing, biomass is collected and loaded manually into trailers and transported with tractors to the roadside. At the roadside, biomass is debarked/chipped, loaded into trucks and transported to the pellet plant in Port Elizabeth. In the pellet plant, biomass is pelletized. For biomass pelletization, additional biomass is required for the drying process. For the final stage of the supply chain (electricity production), we considered two scenarios (1) transporting the wood pellets to a local power plant for electricity production in SA (Electricity in SA scenario) or (2) exporting the wood pellets to the Netherlands and generating electricity at a facility in the port of Rotterdam (Electricity in NL scenario).

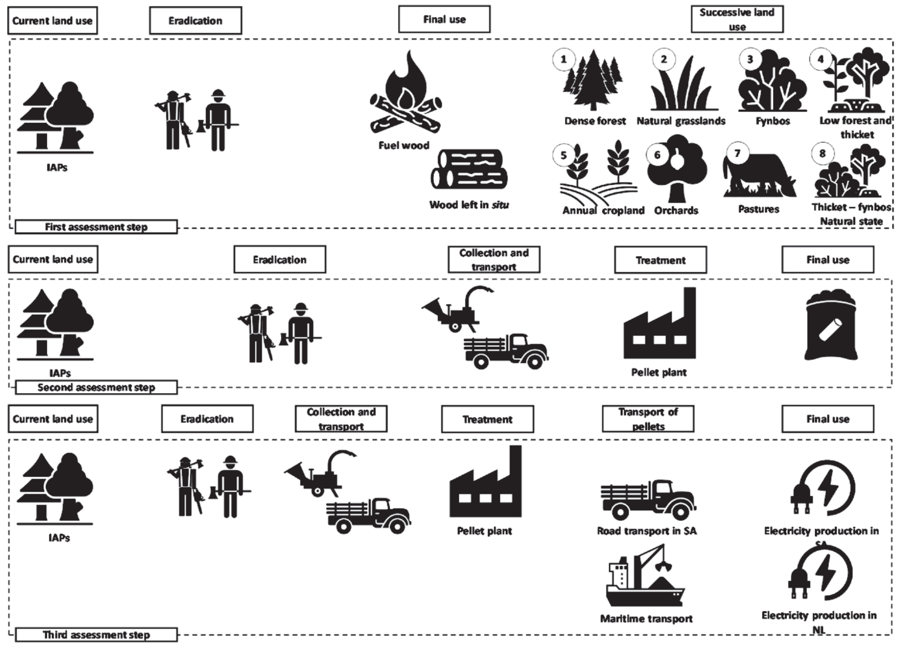



**Figure 5-2 IAP Supply chain scenarios and assessment stages.**

### 5.2.2 GHG emission calculation method

Supply chain GHG emissions, including LUC-related GHG emissions, were calculated following a life cycle assessment (LCA) approach using methods in line with the directive on the promotion of the use of energy from renewable sources recast (REDII). The REDII methods are applied for consistency reasons given the scope of the assessment (one conversion route involves biomass use in Europe). For the first assessment step (scope 1), in which annualized LUC-related GHG emissions are calculated (Equation 5-1), biomass and soil organic carbon (SOC) pools are included [15]. An amortization period of 20 years is assumed for carbon pools to reach equilibrium after IAPs are removed. Therefore, effects of LUC from IAPs to the different subsequent land use scenarios on the carbon pools are calculated over a 20-year time horizon, in line with IPCC and REDII standards [15,118]. The methods to calculate the biomass and SOC pools of each IAP type and land use scenario are present in section 5.7.1 of the supplementary material.

In the second and third steps (scope 2 and 3), GHG emissions are calculated for every step of the supply chain (Equation 5-2) [15]. Upstream emissions from the production of fuels and products (e.g., diesel) are considered, but emissions involved in constructing facilities, buildings and vehicles are not included. The use stage is also considered. However, wood pellets used for electricity generation are considered carbon-neutral, and $CO_2$ emissions released during this stage are considered zero [15]. Different GHG emission reporting units are considered in line with each assessment step. For the first step, LUC-related GHG emissions are reported in Mg $CO_2$ $ha^{-1}$ $year^{-1}$. For the second step (pellet production), the results are shown in g $CO_2eq$ $kg^{-1}$; and for the final step, electricity production, results are presented in g $CO_2eq$ $MJ^{-1}$. The productivity of IAPs input for electricity output (MJ $ha^{-1}$) is considered in the final assessment step to include the LUC-related GHG emissions in the overall supply chain emissions. The inventory data and main assumptions for the assessment of the GHG emissions of electricity production of IAPs are presented in section 5.7.4 of the supplementary material.

**Equation 5-1**

$$e_l = (CS_r - CS_A) * \frac{44}{12} * \frac{1}{20}$$

Where:
$e_l$ = Annualised emissions from carbon stock changes caused by land use change, *Mg $CO_2eq$ $ha^{-1}$ $year^{-1}$*
$CS_r$ = Carbon stock (C) in land associated with the IAPs, *Mg $ha^{-1}$*
$CS_A$ = Potential carbon stock (C) in the successive land use scenarios, *Mg $ha^{-1}$*
*44/12 = Conversion factor to convert C to $CO_2$*
*1 year /20 year* = Factor to annualize emissions

**Equation 5-2**

$$E = e_{ec} + e_l + e_p + e_{td} + e_u$$

Where:
E = Total emissions from the use of electricity, *g $CO_2eq$ $MJ^{-1}$*
$e_{ec}$ = Emissions from the extraction or cultivation of raw materials, *g $CO_2eq$ $MJ^{-1}$*
$e_l$ = Annualised emissions from carbon stock changes caused by land use change, *g $CO_2eq$ $MJ^{-1}$*
$e_p$ = Emissions from processing, *g $CO_2eq$ $MJ^{-1}$*
$e_{td}$ = Emissions from transport and distribution, *g $CO_2eq$ $MJ^{-1}$*
$e_u$ = Emissions from the fuel in use, *g $CO_2eq$ $MJ^{-1}$*





### 5.2.3 Water shortage

IAP's high evapotranspiration rates can lead to significant changes in a region's water balance, reduced stream flows and groundwater levels, and can potentially lead to local water depletion [403]. Within IAPs, acacia, eucalyptus and pine are responsible for the most significant impact on the country's water resources [392,393,404,405]. The effect of IAPs on the local water balance was assessed with Equation 5-3 [121]. Despite that this equation lacks a direct indicator to determine the potential water depletion in the entire region, it provides an adequate estimate of the amount of water that different vegetation types require and use. The spatially explicit daily water shortage was assessed by comparing the evapotranspiration rates from IAPs during the growing season's length with the effective precipitation over the same period. The water shortage was also assessed for each land use scenario. Then, the water shortage from IAPs was compared to the water shortage of the different land use scenarios to determine the net reduction in water shortage. The water shortage was assessed individually for acacia, eucalyptus and pine. However, the results are presented while considering a weighted average in line with IAPs distribution shares (see section 5.2.1.3). A similar process was done for the natural state land use scenario with thicket, fynbos and natural grasslands assessed individually but results are presented for the corresponding shares. The growth cycle of each vegetation is considered over one year and each corresponding crop/vegetation coefficient (Kc) is applied based on the development stage (see Table s 5-3 in the supplementary material). This is done to obtain land use and location-specific evapotranspiration rates. The methods to assess evapotranspiration and effective precipitation are presented in section 5.7.2 of the supplementary material. Water shortage is expressed in mm $year^{-1}$.

5

**Equation 5-3**

$$WS_i = \sum_{GC} ET_0 * Kc_{i,j} - \sum_{GC} EP$$

Where:
*WS* = Water shortage, *mm $year^{-1}$*
*GC*= Grow cycle
*i*= Vegetation type
*j* = Crop/vegetation growing stage
$ET_0$ = Reference evapotranspiration, *mm $day^{-1}$*
*Kc* = Crop/vegetation coefficient
*EP* = Effective precipitation, *mm $day^{-1}$*

### 5.2.4 Socio-economic impacts

The socio-economic impacts are assessed under two performance indicators; supply chain costs and full-time jobs created. The supply chain costs are investigated up to conversion-facility-gate. Therefore, the cost analysis is limited to delivering the biomass in the power plant in SA or in the port of Rotterdam, and biomass conversion costs are excluded. Different sector-based wages are considered in line with regional characteristics. The costs and employment generation related to land rehabilitation after IAP's are removed are discussed but not quantified in the main results. Supply chain costs are presented on a dry basis, ZAR $Mg^{-1}$ and per unit of energy ZAR $GJ^{-1}$. Full-time jobs are assessed based on a 40-hour workweek and presented on an annual basis. One full-time job equivalent (FTE) represents 2080 working hours. FTE's are presented up to the conversion-facility-gate. The total annual biomass output delivered at the conversion-facility-gate is considered to estimate the total amount of jobs created in one year. Direct jobs generated at the conversion facility are excluded. The parameters assumed in the GHG calculations presented in the inventory section, such as dry matter losses and biomass Moisture Content (MC), are applied consistently in the socio-economic analysis. The main regional cost parameters are summarized in Table 5-1. The specific parameters related to costs and job creation of each step in the supply chain are presented in section 5.7.3 of the supplementary material.

**Table 5-1 Main cost parameters according to regional characteristics**

| Parameter | Unit | Value |
|---|---|---|
| Exchange rate | ZAR : € | 18.77[A] |
| Hourly labor costs agriculture/forestry | ZAR $h^{-1}$ | 18.68[B] |
| Hourly labor costs transport | ZAR $h^{-1}$ | 99[C] |
| Hourly labor costs manufacturing | ZAR $h^{-1}$ | 115[D] |
| Electricity price industry | ZAR $MJ^{-1}$ | 7.74[E] |
| Diesel price | ZAR $Mg^{-1}$ | 14,470[F] |
| HFO price | ZAR $Mg^{-1}$ | 304[G] |
| Annual interest rate | % | 7 |

[A] Exchange rate average for 2020 between ZAR and € [406]
[B] According to the new National Minimum Wage (NMW) rate for farm/forestry workers [407]. In SA agricultural and forestry sector, minimum wages are common practice [408].
[C] Average heavy truck driver salary in SA [409].
[D] [410]
[E] Electricity price for large business electricity consumers (6600V and above). Price includes average between winter and summer rate. Basic and maximum demand charges are also included. [411]
[F] Average price of 2020 [412]
[G] [413]





## 5.3 RESULTS

The results are divided into two main parts. Section 5.3.1 presents the GHG and water impacts of land use transitions when IAPs are cleared and the land is rehabilitated to the selected land use scenarios. Section 5.3.2 presents the impacts of mobilising and delivering IAP biomass at power plants on GHG emissions, costs and employment.

### 5.3.1 Environmental impacts of IAP removal and land rehabilitation

#### 5.3.1.1 LUC-related GHG emissions

On average, the removal of IAPs results in a carbon loss of approximately 11.9 Mg $CO_2$ $ha^{-1}year^{-1}$ (Figure 5-3). The carbon loss mainly results from carbon losses stored in the above ground biomass and only to a limited extent from changes in the SOC pool. For all successive land uses, a carbon uptake is projected after IAPs are cleared. However, the net carbon flux is positive for almost all successive land use scenarios, i.e. net $CO_2$ emissions. IAPs thrive under local biophysical conditions and produce more biomass (carbon) than other vegetation types. IAPs are fast-growing tree species and can develop large amounts of above ground biomass in short periods compared to the other land uses and therefore store more carbon in the biomass carbon pool.



The highest LUC-related net carbon emissions are projected for annual cropland and pastures. The establishment of these land uses after the land is cleared from IAPs results in annual emissions of 7.6 Mg $CO_2$ $ha^{-1}year^{-1}$ for cropland and 7.3 Mg $CO_2$ $ha^{-1}year^{-1}$ for pastures. Annual cropland is characterized by low carbon storage, given its annual harvest cycles. Most of the carbon stored in biomass during the growing cycle is lost when the crops are harvested, resulting in an almost balanced net carbon flux for biomass carbon. When natural grasslands are reestablished, 6.9 Mg $CO_2$ $ha^{-1}year^{-1}$ is potentially emitted. The dry semi-arid conditions and relatively shallow root systems result in little biomass development for grasslands. Therefore, the high carbon loss is mainly related to the difference in biomass between IAPs and natural grasslands. Net carbon emissions also occur when IAPs are replaced with fynbos (2.9 Mg $CO_2$ $ha^{-1}year^{-1}$). However, this carbon loss is small compared to the other land use transitions. The SOC pool changes are almost insignificant given that for a 20-year horizon, the carbon loss from LUC is restored almost entirely. Only two scenarios result in a negative carbon flux ($CO_2$ accumulation). Carbon is accumulated when IAPs are replaced by dense forests (-2.9 Mg $CO_2$ $ha^{-1}year^{-1}$) or low forest and thicket (-1.4 Mg $CO_2$ $ha^{-1}year^{-1}$). Both of these land uses can develop more biomass under local biophysical conditions than IAP's and thus, accumulate more carbon in the biomass carbon pool. However, these land uses represent natural ecosystems where little to no degradation/deterioration has occurred. Still, the carbon accumulation in these scenarios is low given that these vegetation types are adapted to the local conditions and haves biomass yields similar to IAPs. The net carbon flux is positive when the land is restored to its natural state. Nevertheless, the net carbon emissions are relatively small 0.9 Mg $CO_2$ $ha^{-1}year^{-1}$. Most of the region is within the thicket biome. The land use transition between both uses generally results in a net carbon accumulation.

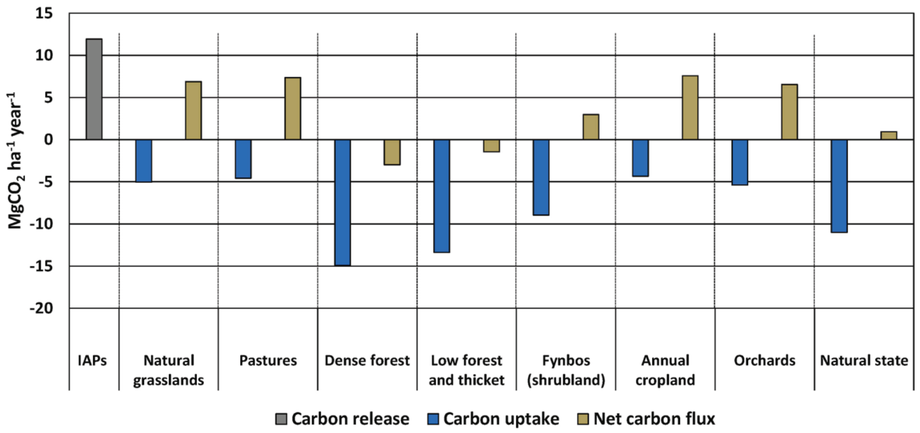



**Figure 5-3. Carbon loss of IAPs removal, carbon uptake from land use scenarios and net carbon flux (carbon release IAP – carbon uptake) in Mg CO2 ha-1year-1 (20 year amortization period).**

#### 5.3.1.2 Water shortage

Figure 5-4 shows the average water balance of IAPs and the corresponding water balances of the different land use scenarios. On average, IAPs show the highest water shortage. The considerably high evapotranspiration rates combined with low annual precipitation result in a water deficit of 1,222 mm $year^{-1}$ for IAPs. This signifies that, on average, an additional 12,221 $m^3$ $ha^{-1}$ is potentially withdrawn from different water sources (besides precipitation) by IAPs for their growing cycle. To illustrate, IAPs are characterized by a deep root system that allows them to access deeply stored groundwater. The use of these water sources can disturb water tables and potentially results in local water depletion. The water balance for all other land uses, except for annual cropland, is positive. Precipitation is sufficient for annual cropland (barley) to meet the crop water requirements. Thus there is no need for other water sources (e.g., irrigation). Conversely, for other land uses, the water supply from precipitation is insufficient for their vegetation development and other water sources are potentially utilized.

The difference in water demand between IAPs and other potential land uses varies between -1,263 and -12 mm $year^{-1}$. On average, the removal of IAPs results in water savings when considering any potential land use transition. The highest water savings





are projected for annual croplands, followed by pastures. The growth cycle of annual croplands (barley) is considerably shorter (5 to 6 months) than the growth cycle of IAPs (12 months) or any other potential land use. The lowest water savings are achieved when IAPs are replaced with dense forest (12 mm $year^{-1}$). The high evapotranspiration rates of dense forests result in a water deficit similar to IAPs. However, different from IAPs, dense forest areas are generally limited to mostly riparian areas. Water savings are also projected when the land is replaced with low forest and thicket (244 mm $year^{-1}$) or fynbos (472 mm $year^{-1}$). Both land uses are well adapted to local biophysical conditions and report lower evapotranspiration rates and less water use than IAPs. The potential land use transition from IAPs to orchards (citrus) results in 427 mm $year^{-1}$ water savings. Although the evapotranspiration rates of orchards are high for several months, orchards result in less water deficit than IAPs, as the high evapotranspiration rates coincide with the rainy season. A water savings potential of 361 mm $year^{-1}$ (3,610 $m^3$ $ha^{-1}$) is projected if the IAPs area is restored to its natural state.

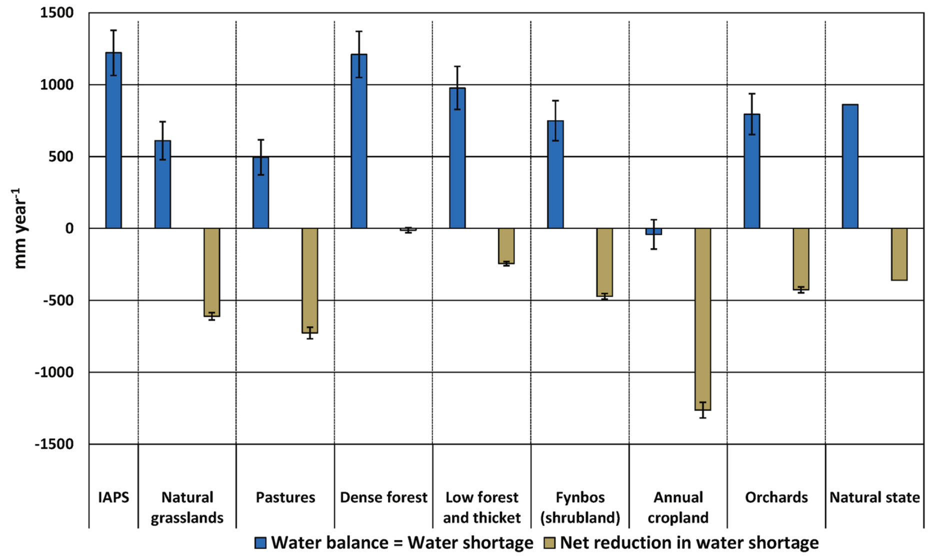



**Figure 5-4. Water shortage (evapotranspiration - effective precipitation) of IAPs compared to other land use scenarios (in mm year-1). The ranges indicate two standard deviations of the spatial variability of the water balance due to the heterogeneity in the biophysical conditions.**

### 5.3.2 Impacts of using IAPs for bioenergy

#### 5.3.2.1 GHG emissions of pellet production

As shown in Figure 5-5, the sum of GHG emissions from IAP biomass supply up to the pellet plant gate is 150 g $CO_2eq$ $kg^{-1}$. The pelletization step is the process with the most

5

significant impact along the entire supply chain (67 g $CO_2$eq $kg^{-1}$). Two factors mainly cause this: first, the electricity demand of the pellet mill (e.g., grinding, densification) and the associated upstream emission of fossil-dominated electricity supply in SA, and second, the additional wood fuel demand for drying. Approximately 0.5 kg of wood is required to reduce the moisture content from 50% to 10% in the drying process for each kg of pellets output. This additional wood supply results in an increase in GHG emissions for the upstream processes of pelletization. The impact of transport on GHG emissions is lower than the pelletization process. The biomass required to meet the pellet plant's annual biomass demand can be sourced relatively close to the pellet plant location. Within an 82 km service area, there is enough biomass to supply the pellet plant requirements, including the additional wood required for drying. Mobilizing IAP biomass from clearing site to road has an impact of 18 g $CO_2$eq $kg^{-1}$. This impact is lower than other processes but is relatively high considering that biomass is only transported for 8 km on average. The impact of wood chipping is similar to the transport to the roadside stage given the intensive use of diesel to kg wood chipped ratio. This ratio is almost half for the debarking process, resulting in nearly half of the emissions of the chipping step. The GHG emissions from the IAPs harvesting/removal are comparatively low and contribute marginally to the overall supply chain emissions.

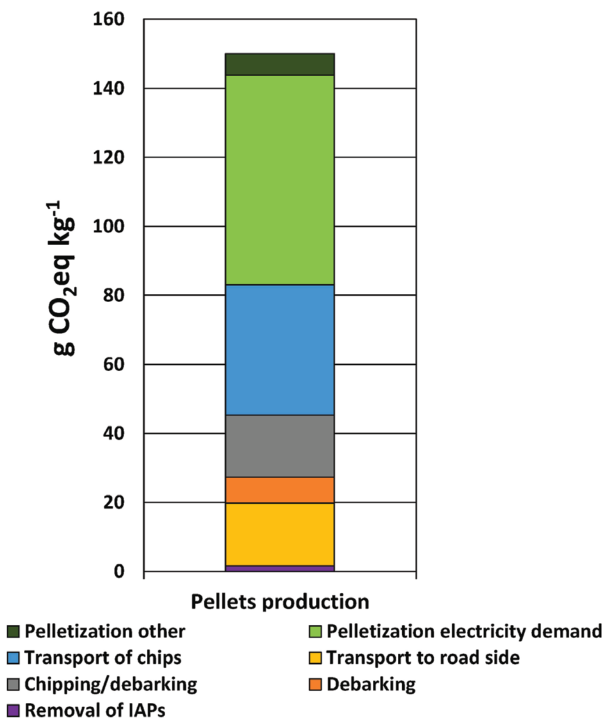



**Figure 5-5 GHG emission from IAPs used for pellet production in Port Elizabeth.**





### 5.3.2.2 GHG emissions for electricity from IAP pellets

The aggregate supply chain GHG emissions of electricity from IAP pellets (excluding LUC-related GHG emissions) are 31.5 g $CO_2$eq $MJ^{-1}$ for electricity generation in SA and 31.2 g $CO_2$eq $MJ^{-1}$ for electricity generation in NL (Figure 5-6-A). The main difference between the two options is in transportation, i.e., distance and mode of transport. However, the impact of transport-related emissions between the two options is minimal. In SA, pellets are transported by truck over a long distance (1,000 km) to the closest power plant. When pellets are exported, bulk carriers deliver the pellets from the pellet plant location in Port Elizabeth to Rotterdam. Despite the long maritime distance between ports, bulk carrier's transport efficiency is considerably higher compared to transport by truck over long distances.

The impact of LUC-related GHG emissions (3.9 g $CO_2$eq $MJ^{-1}$) is relatively low compared to the emissions in the supply chain itself (Figure 5-6-B). The net carbon fluxes that result from the carbon release from the eradication of IAPs followed by the carbon uptake of rehabilitating the land to its natural state is relatively small (see section 4.1.1). However, when the land use is replaced with annual cropland, total supply chain GHG emissions can increase by 32 g $CO_2$eq $MJ^{-1}$. It could, however, also decrease by 12 g $CO_2$eq $MJ^{-1}$ when IAPs are replaced by dense forests. Both supply chains, including LUC-related GHG emissions, can comply with REDII 70% GHG savings requirement for 2021 and 80% GHG savings requirement after 2025. However, complying with REDII requirements is only feasible when the land is restored entirely to its natural state(i.e. thicket or dense forest). Conversely, if the land is dedicated to natural grasslands, pastures, fynbos, orchards, or annual croplands, the supply chains would fail to comply with the REDII GHG emissions savings targets.

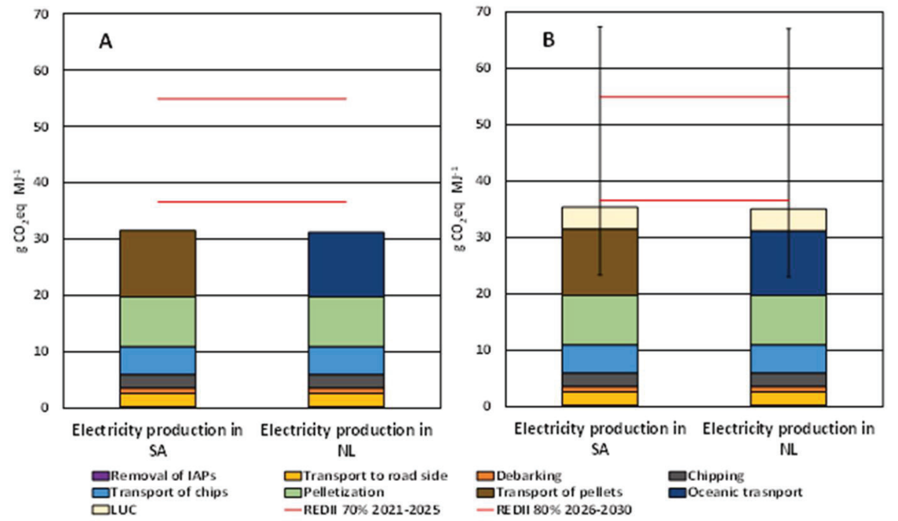



Figure 5-6. GHG emission from IAP pellets used for electricity generation in South Africa or the Netherlands (in g CO2eq MJ-1). Figure A excludes the carbon stock changes induced by IAP removal and land rehabilitation to its natural state. Figure B includes the carbon stock changes induced by IAP removal and land rehabilitation to its natural state. The ranges indicate the carbon stock changes from other land use transitions.

### 5.3.2.3 Supply chain costs of using IAPs for bioenergy

The supply chain cost of pellets is 5,344 ZAR $Mg^{-1}$ (284.7 € $Mg^{-1}$) delivered at the power plant in SA and 2,535 ZAR $Mg^{-1}$ (159.1 € $Mg^{-1}$) delivered at Rotterdam port as shown in Figure 5-7-A. Results are also shown on an energy basis (Figure 5-7-B). The cost of pellets in SA is considerably higher due to the long distance between Port Elizabeth to the closest power facility (1,000 km) and the lack of railway infrastructure to enable efficient transport. The transport of pellets accounts for approximately 52% of the total supply chain costs in SA. The largest share of the pellets transport cost corresponds to the truck kilometer rate. Conversely, the cost of delivering pellets to NL is significantly lower. Despite that port and voyage costs can be perceived as high, the overall costs when considering the ship cargo capacity are low. The cost of pelletization corresponds to 1,200.4 ZAR $Mg^{-1}$ (64 € $Mg^{-1}$). The biggest share of pelletization costs is the operating costs (80%), mainly the additional biomass costs required for drying (56%). Drying with biomass requires that additional feedstock is mobilized from the harvesting location to the pellet plant. This drying biomass demand generates a cost increase in all logistics up to pelletization. Removing IAPs corresponds to 10% of the SA supply chain's total costs and 19% for the NL supply chain, 561 ZAR $Mg^{-1}$ (29.9 € $Mg^{-1}$) for both scenarios. The removal costs are higher than for chipping (49.5 ZAR $Mg^{-1}$ – 2.6 € $Mg^{-1}$) and transport to roadside (295.6 ZAR $Mg^{-1}$ – 15.7 € $Mg^{-1}$), due to the high number of workers involved in IAP removal. The transport of chips from the roadside to the pellet plant is estimated at 427.4 ZAR $Mg^{-1}$ (22.7 € $Mg^{-1}$).





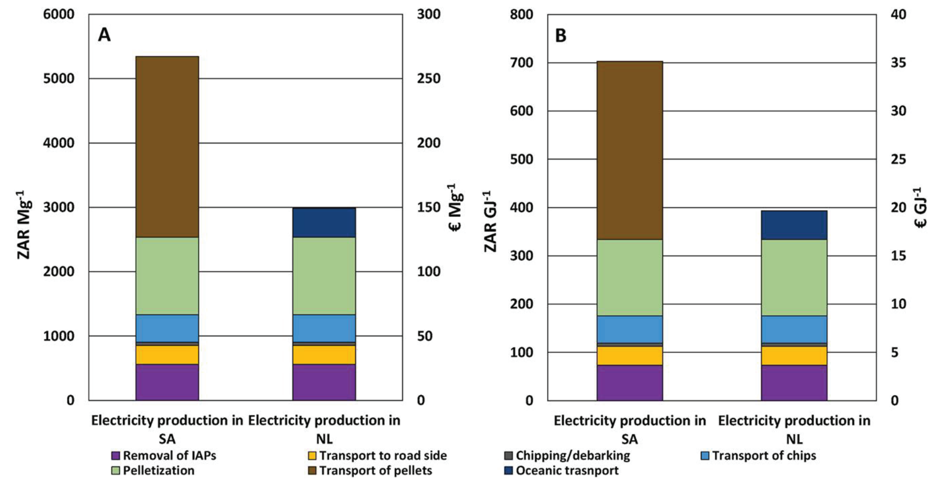



Figure 5-7 Supply chain costs of IAP pellets used for electricity generation in South Africa or the Netherland, delivered at conversion-factory-gate. Results are expressed on a mass and energy basis.

### 5.3.2.4 Supply chain employment of using IAPs for bioenergy

The total annual direct full-time jobs generated from sourcing IAPs up to the conversion-factory-gate is 604 for SA and 525 for the NL (Figure 5-8). This indicates that running operations over a year for both supply chains would generate direct employment full-time for 604 people if electricity is generated in SA and 525 if electricity is produced in the NL. For both supply chains, most of the jobs are created in SA. However, when pellets are shipped to the NL, it is uncertain under which country ship members are employed. Therefore, crew jobs are not considered. The most significant number of jobs are created in the stages that require more manual labor, such as IAP removal and transport to the roadside. Removing IAPs to meet annual biomass pellet demand requires approximately employing 351 people on a full-time basis. Most of these positions are related to chain saw operations for tree cutting/removal. Transporting biomass to the roadside also requires several workers. About 135 full-time jobs are required for this stage; most of these jobs are related to manually loading the biomass in the tractor transport trailers. It takes approximately 1 hour to collect and load manually 1 $m^3$ of IAPs biomass. The transport of chips from the roadside to the pellets plant requires a small number of full-time employees (10), while the transport of pellets to the power plant requires a more significant number (78). The large difference in employment of both stages is caused by the difference in distances and consequently working time. To illustrate, it takes 1.5 hours to transport biomass from the roadside to the pellet plant, while it takes 18.2 hours to .travel from the pellet plant to the power plant. Few jobs are generated at the pellet plant. Approximately 7 workers are required to run operations annually due to the more mechanized systems operations than in other supply chain stages.

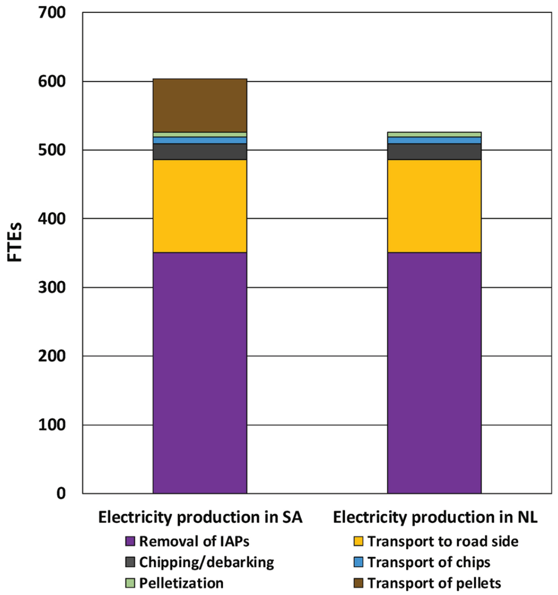



**Figure 5-8 Supply chain employment from IAPs pellets used for electricity generation in South Africa or the Netherlands, delivered at conversion-factory-gate.**

## 5.4 DISCUSSION

### 5.4.1 GHG emissions from landscape restoration and using IAPs for bioenergy

IAPs are a significant carbon sink, and removing these will result in a net loss of carbon (depending on land use transition). The carbon fluxes resulting from LUC fromIAPs to the natural state will rely mainly on the type of vegetation that is actually restored and to what degree it can be regenerated to the original condition. For example, the carbon stock(biomass + SOC) of intact Baviaans Spekboom Thicket land is estimated at 61.13 Mg ha$^{-1}$ [414]. If IAPs are restored to Baviaans Spekboom Thicket land instead of Fish Spekboom Thicket land (assumed in this study, 73 Mg ha-1, see section 1 in the supplementary material), the net carbon flux would be positive, resulting in an overall carbon release. Furthermore, the carbon release can be considerably higher if an intact (without degradation) natural state is not reached. Net carbon fluxes will also depend on the vegetation status of IAPs prior to conversion and designated land use (vegetation) after restoration, and both can vary considerably. To illustrate, in this study, IAP's average yields are set at 65 Mg ha$^{-1}$. However, in similar biophysical conditions, other studies have reported yields of approximately 66-68 Mg ha$^{-1}$ for eucalyptus and 78 Mg ha$^{-1}$ for Acacia





plantations [118,415]. In addition, it is still uncertain whether, after the removal of IAPs, the soil and biomass will reach the expected state of carbon equilibrium [389]. If such carbon equilibrium states are not reached, potentially more carbon will be released [118]. Dedicating the treated land to annual crops or pastures for cattle can result in higher GHG emissions from agricultural-related activities such as fertilizer application or methane-derived enteric fermentation [416]. Other studies have reported similar net carbon losses varying between 20-70 Mg ha$^{-1}$ (depending on land use transition) [389,417,418].

Significant GHG savings are possible by restoring lands with natural vegetation, mainly thicket/fynbos for the Port Elizabeth region. However, if other land uses such as annual croplands and pastures are implemented after IAPs are cleared, the LUC-related GHG emissions from such transitions can offset the benefits achieved of using biomass for electricity production [378,380,389]. The question is if these LUC-related emissions should be allocated to bioenergy. The calculation rules of REDII require to account for LUC-related GHG emissions unless the biomass is categorized as a residue. According to REDII article 2, residues are defined as "*... substance that is not the end product(s) that a production process directly seeks to produce; it is not a primary aim of the production process and the process has not been deliberately modified to produce it*" [15]. Independent of the final biomass use, IAPs are removed with the primary objective of restoring the land and enhancing water and biodiversity protection services [402]. Therefore, after the land is treated, the use of IAPs does not intervene with the primary objective of land restoration. Currently, IAPs are not grown and removed for the purpose of bioenergy, or other end uses. Since their introduction, IAPs have spread across the country over time, as they are highly adapted to SAs biophysical conditions [419]. Therefore, utilizing IAPs as a feedstock for any purpose is not the primary aim of the land restoration process and does not seek to produce additional IAPs. The use of IAPs could be considered carbon neutral upstream of collection if they are considered a residue in line with REDII. However, regardless of the feedstock classification and associated calculation rules, removing IAPs will result in net carbon losses in almost all cases.

Currently, biomass is left on site unused or used on a small scale by the local communities as fuelwood [380]. Nevertheless, using IAPs for electricity can result in indirect effects as it could displace current IAPs feedstock uses (fuelwood) or generate additional pressure to produce IAPs for economic purposes. However, producing IAP feedstock types requires multiple permits and is not permitted in treated areas [385]. The applied model is unsuitable for assessing the possible consequences of replacing different end uses as it is static and independent of an economic context [267]. Other types of frameworks (i.e., consequential LCA) can be applied to account for such effects.

IAPs as feedstock for electricity production in SA can help to reduce SA GHG emissions and to decarbonize the energy sector. Approximately 226 g $CO_2$eq $MJ^{-1}$ are saved when pellets derived from IAPs feedstock are used for electricity production and replace conventional electricity production in SA. Similar results were found in other studies with GHG emission savings ranging between 70 – 250 g $CO_2$eq $MJ^{-1}$ [378,380]. However, in these studies, LUC-related GHG emissions were not accounted for, and biomass is not pelletized before co-firing at the power plant. Dedicating the whole pellet production to electricity generation in SA and restoring the land to its natural state can lead to GHG emission savings of 206,112 Mg $CO_2$eq $year^{-1}$. In SA, 48% of annual GHG emissions are from electricity and heat production [420]. Therefore, dedicating the whole pellet production can reduce on a cumulative basis 1.7%. GHG emissions from electricity and heat production. In addition, it can provide electricity to 67,000 people annually since the annual average per capita electricity consumption in SA is 3,759 kWh [421].

The GHG savings from electricity production in NL with IAPs derived pellets from SA are considerably lower. The electricity mix in NL has a lower share of fossil-based sources and thus, GHG emissions from conventional electricity production are lower. About 112 g $CO_2$eq $MJ^{-1}$ GHG savings are accounted for in the NL supply chain, equivalent to 102,144 Mg $CO_2$eq $year^{-1}$. However, there could be limitations to introducing IAPs pellets in the European market as the wood properties from these tree species might not meet European standards for industrial pellets. For example, for large installations, the ash content of the feedstock should be lower than or equal to 2% [422]. This threshold can potentially not be met according to the properties of some species included in our study. A study of wood properties in the Western Cape reported a 2.1% ash content for acacia and 2.3% ash content for eucalyptus [423]. Additional research is required to assess the wood properties from the IAPs in the study area to estimate better the potential to export IAPs pellets to the European market.

The selected supply chain configuration for electricity production in SA poses several limitations. All coal power plants in SA are located inland and remote from seaports. The closest power plant is located 1,000 km away from Port Elizabeth and the mobilization of biomass by road transport can be costly and inefficient [180]. It is suggested that transport costs are one of the main barriers to biomass exploitation [377]. In addition, pelletization is not required for co-firing biomass at the power plant. Wood chips could potentially be transported directly from the supply area to the power plant. Still, the IAPs supply location is far from the nearest power plant location. The benefits of biomass pelletization are valuable for long-distance intermodal biomass supply chains in which there are clear GHG emissions and cost benefits of condensed bulk transport [180,329]. Thus, from a logistic perceptive, the export of pellets seems more suitable for a supply chain





configuration with a pellet plant located in Port Elizabeth. However, the export of pellets for electricity production in other countries such as in the NL, as assessed in the study, would considerably diminish the GHG emissions savings in SA. Nevertheless, coal power plants in SA are located near coal mines, and many will be decommissioned in the coming decade. New plants could also be developed near port regions, such as Port Elizabeth, to provide carbon benefits for biomass supply chains. Other final uses of pellets should also be explored. Final uses such as pellets for the residential market could offer higher benefits for this supply chain and SA GHG emission mitigation targets. In addition, other supply chain configurations could be assessed, such as drying the wood at the harvesting site could reduce MC content and GHG emissions from logistics.

### 5.4.2 Water savings

Clearing the land of IAPs will result in water savings independently of the succeeding land use. Water savings are particularly important because only 9% of the country's annual precipitation ends up as water in rivers or aquifers (Bailey and Pitman 2015). It can also reduce the pressure on irrigation systems and overall water consumption in the country. To illustrate, on average 7,659 $m^3$ $ha^{-1}$ are used for irrigation purposes in SA [424]. This signifies that if the land is restored to its natural state, 47% of water demand for irrigation (ha basis) could be covered by allocating the additional water previously utilized by IAPs compared to the natural landscape. Despite that irrigation supports only 25 to 30% of SA agricultural production, 90% of the country's high-value crops such as potatoes and fruits are irrigated [424]. High water savings are reported when IAPs are replaced with annual crops (Barley), pastures and orchards. However, the overall savings can be lower when considering other water uses such as water for livestock production [425], irrigation for orchards and using the land for other purposes after harvesting annual crops.

The water savings are estimated based on the difference between the annual water deficit in IAPs and the successive land use scenarios after IAP removal. Therefore, it describes particularly whether IAPs use more or less water than the potential land use scenarios according to local specific biophysical characteristics. Also, despite that the water balance includes important parameters such as temperature and precipitation, it neglects others such as soil characteristics. Crop/vegetation coefficient values were not available for the thicket vegetation type. Crop/vegetation coefficient values for thicket were assumed as an average between dense forest and fynbos, as thicket is considered to be a transition ecosystem between shrubs and forest [426]. Thicket-specific coefficient values could lead to different water savings results. However, it is widely reported that IAPs generally use more water than thicket vegetation [392,397,427,428]. Climate data for 2020, instead of long-term averages, was used in line with the study's temporal scope to represent current conditions. This data set is already corrected to account for extreme events such as prolonged dry spells. Thus, it is expected that using long-term averages would not considerably affect the results. The results are likely to be affected when climate change is considered, given that drought episodes are projected to occur more frequently in the future [1].

### 5.4.3 Supply chains costs of using IAPs for bioenergy

The supply chain costs are considerably higher for electricity production in SA than for the export of IAPs pellets to NL. The high logistics cost of delivering pellets from Port Elizbeth to the closest power plant makes it infeasible to compete with electricity market prices without subsidies. Average market electricity prices stand at 0.44 ZAR $MJ^{-1}$ [377]; this is approximately 63% less than the costs estimated in this study for the SA supply chain (0.7 ZAR $MJ^{-1}$). Furthermore, conversion costs are not included. Thus, from a market perspective, electricity production from IAPs sourced in the Port Elizabeth region is expensive. However, the cost assessment disregards the overall hidden benefits from using IAPs and rehabilitating ecosystem services. From a cost-savings perspective, the overall benefits from removing IAPs and carbon emissions reduction from displacing fossil fuels could sum up to 69,682,555 ZAR $year^{-1}$ for water services and 24,076,800 ZAR $year^{-1}$ for carbon services when allocating the whole pellet output to electricity production in SA. These savings are obtained based on a water value of 1.5 ZAR $m^{3\ -1}$ and a carbon tax of 1 ZAR $Mg^{-1}(CO_2eq)$ [377,429]. These cost savings externalities translate into 0.11 ZAR $MJ^{-1}$ that could be allocated to reduce the overall electricity price from IAPs. However, the overall cost savings could decrease if the land use after IAP removal is considered. In addition, replacing IAPs with land uses that provide an economic activity for the region, such as citrus orchards, can result in additional cost benefits.

In recent years, pellet imports to the Netherlands were widely sourced from the United States (US) and Canada [203]. The market price of pellets delivered at Rotterdam can vary between spot prices and contract prices. Between 2012 and 2018, CIF ARA spot prices from the US and Canada (delivered at Rotterdam) varied between 107 € $Mg^{-1}$ to 137 € $Mg^{-1}$ and contract prices varied between 131 € $Mg^{-1}$ and 182 € $Mg^{-1}$ [430]. Compared to CIF ARA spot prices of wood pellets, the calculated costs of SA wood pellets (159.1 € $Mg^{-1}$) are high. However, note that most of the pellet imports from these countries to the Netherlands are traded under long-term price contracts [343]. Therefore, comparing the costs to contract prices could offer a more realistic benchmark. Hence, these pellets could potentially compete with other international pellet markets. However, it highly depends on external factors such as exchange rates, shipping rates, and market conditions. The estimated costs also appear to align with the costs of pellets found in literature ranging between 88 € $Mg^{-1}$ to 279 € $Mg^{-1}$ [343].





The cost estimates are based on a desk study and are subject to high uncertainty. For example, average 2020 time charter rates (8,150 USD $day^{-1}$) were applied to estimate maritime shipping costs. However, in the last two years, supramax charter rates have surpassed the 14,000 USD $day^{-1}$ barrier for several months [431]. A substantial increase in time charter rates would lead to an additional 4 € $Mg^{-1}$. The export of pellets is submitted to market conditions that are considerably affected by the exchange rate between Euros and South African rand. The whole supply chain operates on a South African rand basis except for ocean freight. Profits of wood pellets exports to NL are made on a Euro basis. An adverse and volatile exchange rate (e.g., depreciation of the ZAR) can considerably affect the cost margins along the whole supply chain and result in an unsustainable pellets-export business. Preliminary costs of rehabilitating the land are estimated between 6,100 ZAR $ha^{-1}$ to 12,200 ZAR $ha^{-1}$ (corrected for inflation) for fynbos and thicket [432,433]., Including these costs would increase the overall supply chains costs by approximately 0.027 ZAR $MJ^{-1}$ to 0.55 ZAR $MJ^{-1}$. These costs could be addressed through a public-private partnership with clear incentives for both sides, given that in some cases, the costs of rehabilitating can drastically reduce the cost of follow-up treatments [428]. In addition, ecosystem services and biodiversity conservation can provide additional incentives for land restoration.

### 5.4.4 Supply chain employment of using IAPs for bioenergy

Job creation is crucial for social development and poverty alleviation, both at the top of the agenda for SA [373].The Working for Water programme generated almost 25,000 full-time jobs on a country level in 2017. However, most of these jobs were seasonal and not stable over more extended periods [385]. Instead, the supply chains investigated in this study potentially could keep workers on a full-time basis over the pellet plant expected productive lifetime (15 years). Biomass potentials are given on an annual basis over a 20-year time horizon of availability. Thus, biomass supply can be carried out in the elected region beyond the pellet plant lifetime. In addition, Working for Water is defined as an Extended Public Works Programme in the Department of Public Works. Therefore, the remuneration for workers is lower than in a private project. For example, the current minimum wage for workers employed on an expanded public works program is11.42 ZAR $h^{-1}$; this is almost 8 ZAR $h^{-1}$ less than employees in the agriculture and forestry sectors [407]. Therefore, a pellet plant project in the Port Elizabeth region could generate more and better-paid employment. However, it must be highlighted that Working for Water targets underprivileged communities in order to contribute to poverty alleviation at the national level. Additional jobs could be created if restoring the land is accounted for in the pellet plant project. To meet the input requirements of the pellet plant with a capacity of 120,000 Mg $year^{-1}$ pellets, 2940 ha of IAPs need to be cleared annually. It is suggested that rehabilitating 1 ha under a native thicket ecosystem requires 50 working days [432]. Therefore, restoring the total treated land could lead to an additional 403 workers employed full-time annually.

## 5.5 CONCLUSION

The eradication of IAPs results in trade-offs between GHG emissions, water savings, and socio-economic impacts. The land use transition dictates to a large extent the magnitude and direction (positive or negative) environmental effects resulting from IAPs removal. The eradication of IAPs could reduce water shortages with 120 $m^3$ $ha^{-1}$ $year^{-1}$ if replaced with dense forests, to up to 12,630 $m^3$ $ha^{-1}$ $year^{-1}$ if replaced with annual cropland without irrigation. However, replacing IAPs with annual cropland will also result in the highest net carbon losses from LUC (7.3 Mg $ha^{-1}$ $year^{-1}$). Generally, net carbon losses will occur when considering the land use transitions after IAPS removal, even when land is rehabilitated to its natural state (3.3 Mg $ha^{-1}$ $year^{-1}$). However, independent of the land use transition, removing IAPs results in water savings and job creation. These benefits can also amplify other ecosystem services, such as the conservation of biodiversity and socio-economic development. Trade-offs of using IAPs for bioenergy need to be considered for the sustainable development of the biomass sector.

The use of IAPs for electricity generation can generate employment and reduce GHG emissions when fossil electricity is replaced. However, the reported GHG savings depend on whether IAPs are classified as a residue or not. This classification will determine whether LUC-related GHG emissions should be allocated to bioenergy or allocated to the eradication program itself. This study explored both options (with and without allocation) separately to provide insights into the effect on the performance of the supply chain and possible trade-offs between impacts.

If pellets are exported to the NL, both 2021 and 2025 REDII GHG emission reduction criteria can be met. However, if IAPs are not classified as a residue, meeting the REDII criteria will rely mostly on rehabilitating the land to its natural state. There are clear trade-offs between environmental and social benefits with costs. The costs of producing electricity in SA from IAPs sourced in the Port Elizabeth region are high (5,344 ZAR $Mg^{-1}$, 284.7 € $Mg^{-1}$), due to high logistical costs. However, it will employ 604 workers on a full-time basis. A public-private partnership is essential to share electricity production costs and unleash the environmental and social benefits from such supply chains. From an economic perspective, exporting the pellets to NL seems a more viable strategy than electricity production in SA. However, the GHG emission savings from using IAP pellets would be accounted for in NL. Therefore, other IAP end-uses in SA can be more adequate to avoid long-distance transport given economic constraints. This study is an important step forward in developing sound land use planning for IAP›s removal and use.





## 5.6 ACKNOWLEDGMENTS

We would like to sincerely thank Hayden Wilson for providing input data and context about biomass supply chains in SA. We also want to thank Werner Euler for providing primary (IAPs) costs/employment data of harvesting and logistics operations carried out in the Port Elizabeth region. Finally, we thank Kees Kwant for providing assistance and proofreading the manuscript.

## 5.7 SUPPLEMENTARY MATERIAL

### 5.7.1 Biomass and soil organic carbon pools

#### Biomass carbon stocks

Biomass carbon stocks were assessed for the IAPs land cover and each land use scenario. For all categories, biomass carbon stocks are assumed to be in equilibrium. Above ground biomass (AGB) or above ground carbon (AGC) were assessed making use of the IPCC climatic-zone specific default AGB values and studies carried out for different ecosystems/land uses in SA [118,414,434,435]. The values are directly given in AGC or aggregated as total above and below ground biomass carbon for some land uses. For some categories, below ground biomass (BGB) was estimated as a function of the AGB using IPCC climate-zone-dependent root-to-shoot ratios (R) [118] or directly retrieved from the mentioned studies (see Table s 5-1). Vegetation type-specific carbon fraction (CF) coefficients were employed to obtain biomass carbon stocks. For IAPs, AGB is estimated by overlapping the Above Ground Woody Biomass database [434] with the service area established in the geographical scope given in section 5.2.1.1. In addition, the IAPs sites [395] were used as a proxy to increase the level of accuracy in AGB estimation and obtain an average value for IAPs for the studied area. This process was applied given that the Above Ground Woody Biomass database lacks a distinction between tree species. A similar overlay assessment was carried out to identify the AGB of dense forest land use. The Above Ground Woody Biomass database was overlapped with the land cover data set [399] to obtain a proxy indicator of AGB from dense forest land use within the study area. AGB was assumed from studies carried out within the study area (i.e., thicket) or in places with similar biophysical characteristics (e.g., fynbos or pastures) for the rest of the land use categories. For the natural state land use scenario, carbon stocks were assessed individually for thicket fynbos and natural grasslands, but results account for the correspondent shares.

**Table s 5-1 parameters used to estimate biomass carbon pools**

| Land use/cover | Above ground biomass (AGB) Mg ha$^{-1}$ | Above ground carbon (ABC) Mg ha$^{-1}$ | Root to shoot ratios (R) | Carbon Fraction (CF) | Below ground carbon Mg ha$^{-1}$ | Above and below ground carbon Mg ha$^{-1}$ | Reference |
|---|---|---|---|---|---|---|---|
| IAPs[A] | 65 | 31.9 | 0.29 | 0.49 | 9.2 | 41.1 | 118,378,395,434 |
| Natural grasslands[B] | 1.9 | 0.91 | 2.8 | 0.47 | 2.57 | 3.48 | 118 |
| Pastures[C] | - | 1.4 | - | - | 2 | 3.4 | 435 |
| Dense forest | 87.8 | 43.1 | 0.332 | 0.49 | 14.28 | 57.4 | 118,399,434 |
| Low forest and thicket[D] | 76.7 | 36.8 | 0.332 | 0.48 | 12.22 | 49.02 | 118,414 |
| Fynbos (shrubland)[E] | - | 11.6 | - | - | 13.3 | 24.9 | 435 |
| Annual cropland | - | - | - | - | - | 4.7 | 118 |
| Orchards | - | 8.5 | 0.37 | - | 3.14 | 11.64 | 118 |

[A] the IAPs AGB is assumed to be composed of 73% acacia, 14% eucalyptus and 13% pine.
[B] Value for semi-arid dry grasslands
[C] Value from grazed pastures in Agulhas Plain, South Africa
[D] Value from intact Fish Spekboom Thicket
[E] Value from Overberg Sandstone Fynbos





### Soil organic carbon stocks

Soil carbon stocks (Equation S 5-1) were quantified by comparing the amount of SOC present when land is under IAPs to the amount of SOC after IAPs removal for each land use scenario. The IPCC 2006 default values for SOC and soil stock change factors (to account for the effect of land use, management regime, and organic amendment) were assigned to each land use and applied based on the stratification of climate regions and soil types. Table s 5-2 summarizes the assigned stock change factors for each land use/cover category

**Equation S 5-1**

$$SOC_x = SOC_{ref} * F_{LU,x} * F_{MG,x} * F_{I,x}$$

where:
$SOC_x$ = Soil organic carbon stock (C) for land under land use type x, *Mg ha$^{-1}$*
$SOC_{ref}$ = The reference carbon stock (C), *Mg ha$^{-1}$*
$F_{LU}$ = Stock change factor for land use system x, unitless
$F_{MG}$ = Stock change factor for management regime for land use x, unitless
$F_I$ = Stock change factor for input of organic matter for land use x, unitless

**Table s 5-2 IPCC relative soil stock change factors118**

| Land use/cover | IPCC relative soil stock change factors | | |
|---|---|---|---|
| | $F_{LU}$ | $F_{MG}$ | $F_I$ |
| IAPs | 1 | 1 | 1 |
| Natural grasslands | 1 | 1 | 1 |
| Pastures | 1 | 0.90 | 1 |
| Dense forest | 1 | 1 | 1 |
| Low forest and thicket | 1 | 1 | 1 |
| Shrubland-fynbos | 1 | 1 | 1 |
| Annual cropland | 0.76 | 1 | 1.04 |
| Orchards | 0.72 | 0.99 | 1.04 |

### 5.7.2 Evapotranspiration and effective precipitation

Evapotranspiration rates are highly dependable on local biophysical conditions such as hydro-climatic regime and plant growth stage, which are considered in line with the assessment's geographical scope. Reference evapotranspiration ($ET_0$) was determined using the widely applied Penman-Monteith equation (Equation S 5-2) [310]. Effective precipitation is the share of precipitation stored in the soil and is usable for the vegetation; effective precipitation is derived from actual precipitation (Equation S 5-3) [121]. Spatially explicit daily data on temperature, precipitation, relative humidity, shortwave radiation and wind speed for 2020 were collected from the Inter-Sectoral Impact Model Intercomparison Project (ISIMIP2b) database [313] and clipped to the extent of the geographical scope.

**Equation S 5-2**

$$ET_0 = \frac{0.408\Delta(R_n - G) + \gamma \frac{900}{T+273} U_2(e_s - e_a)}{\Delta + \gamma(1 + 0.34U_2)}$$

Where:
$ET_0$ = Reference evapotranspiration, *mm day$^{-1}$*
$R_n$ = Net radiation at the crop surface, *MJ m$^{-2}$ $^{-1}$ day$^{-1}$*
$G$ = Soil heat flux density *MJ m$^{-2}$ $^{-1}$ day$^{-1}$*
$T$ = Temperature, °C
$U_2$ = wind speed, *m s$^{-1}$*
$e_s$ = saturation vapour pressure, *kPa*
$e_a$ = actual vapour pressure, *kPa*
Δ = slope vapour pressure curve, *kPa* °C$^{-1}$
γ = psychometric constant, *kPa* °C$^{-1}$

**Equation S 5-3**

$$EP = P * \left(\frac{125 - 0.2 * P}{125}\right) for\ P_{month} < 250\ mm$$

Or

$$EP = 125 + 0.1 * P\ for\ P_{month} > 250\ mm$$

Where:
$EP$ = Effective precipitation, *mm day$^{-1}$*
$P$= Precipitation, *mm day$^{-1}$*





**Table s 5-3 Land uses crop/vegetation coefficients**

| Land use | Crop/vegetation coefficient (Kc) | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Oct | Nov | Dec |
| Acacia[A] | 1.8 | 0.48 | 0.48 | 0.48[B] | 0.48[B] | 0.48 | 0.48 | 1.2 | 1.2 | 1.8 | 1.8 | 1.8 |
| Eucalyptus[C] | 1.5 | 0.4 | 0.4 | 0.4[B] | 0.4[B] | 0.4 | 0.4 | 1 | 1 | 1.5 | 1.5 | 1.5 |
| Pine[D] | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Natural grasslands[E] | 0.73 | 0.73 | 0.73 | 0.73 | 0.73 | 0.73 | 0.73 | 0.73 | 0.73 | 0.73 | 0.73 | 0.73 |
| Pastures[F] | 0.75 | 0.75 | 0.75 | - | 0.3 | 0.3 | 0.75 | 0.75 | 0.75 | 0.75 | 0.75 | 0.7 |
| Dense forest[G] | 1.15 | 1.11 | 1.07 | 1.11 | 1.15 | 1.25 | 1.21 | 1.17 | 1.16 | 1.15 | 1.15 | 1.15 |
| Low forest and thicket[E] | 0.96 | 0.93 | 0.94 | 0.98 | 1.03 | 1.1 | 1.06 | 1.02 | 1.01 | 0.99 | 0.97 | 0.95 |
| Fynbos[F] | 0.78 | 0.75 | 0.81 | 0.84 | 0.92 | 0.98 | 0.92 | 0.86 | 0.85 | 0.84 | 0.80 | 0.75 |
| Annual cropland[G] | - | - | - | 0.3 | 0.3 | 1.15 | 1.15 | 1.15 | 0.25 | - | - | - |
| Orchards[H] | 0.80 | 0.80 | 1.28 | 1.28 | 1.28 | 0.79 | 0.79 | 0.79 | 0.62 | 0.62 | 0.62 | 0.80 |

[A] Monthly Kc values for acacia are extrapolated for each month based on eucalyptus growing cycle and a maximum Kc value of 1.8 reported in Western Cape [436].
[B] Values for March and April were assumed from the initial development stage.
[C] Assuming a growing cycle starting in June for South Africa [437–439] and Kc values based on [224].
[D] FAO Kc values for conifers [173].
[E] Average from FAO grasses categories [173].
[F] Values from FAO extensive grazing pasture [173] and assuming a growing cycle of ryegrass in Western Cape [440]
[G] Values assumed from dense forest sub-tropical forest [171].
[E] Average between fynbos and dense forest values. Assuming thicket as an intermediate ecosystem between fynbos and dense forest [426].
[F] Average between sandstone fynbos and sandplain fynbos [441]
[G] Values from barley [173] and assuming a growing cycle starting in April for South Africa [442]
[H] Values for citrus plantations in South Africa [443]

5

### 5.7.3 Costs and job creation

**Removal of trees, transporting to roadside and chipping.**

The capital costs involved in the removal process are related to acquiring equipment such as chain saws, vehicles, and tools. Operating costs refer to wages, salaries, and machinery running expenses (e.g., fuels and lubricants). The removal team comprises 24 persons (day $ha^{-1}$), including chain saw operators, helpers, and management. Cost and number of workers data are based on primary information from harvesting operations in the Port Elizabeth region [444]. Transport of biomass to the roadside is performed with tractors and an attached 10-ton capacity trailer. The capital costs include purchasing equipment, depreciation, maintenance, and interest rates [445]. The fixed costs are corrected for inflation (2020) based on the World Bank consumer price index for SA [446]. Operating costs include mainly wages and fuel use. Biomass is assumed to be loaded manually and tractors travel from the harvesting site to the roadside [444]. Empty return trips are also accounted for. The capital and operating costs of chipping are estimated based on cost/capacities curves [378]. The ratio between the systems' reference biomass output and input at the biomass chipping stage is considered. Chipping total costs are also corrected for inflation. Two workers are assumed to be required to run chipping operations. The main cost and job-related parameters of clearing, transporting to roadside and chipping are summarized in Table s 5-4.

**Table s 5-4 Parameters of land clearing, transport to roadside and chipping**

| Parameter | Unit | Value |
|---|---|---|
| Clearing cost | ZAR $Mg^{-1}$ | 330[A] |
| Tractor power | MJ | 0.16[B] |
| Tractor fixed cost | ZAR $h^{-1}$ | 38.9[B] |
| Trailer fixed cost | ZAR $h^{-1}$ | 52.5[B] |
| Tractor fuel consumption | $m^3 h^{-1}$ | 0.0079[B] |
| Average distance | Km | 8[C] |
| Average speed | Km $h^{-1}$ | 10[D] |
| Load rate | $m^3 h^{-1}$ | 1[E] |
| Wood density | Mg $m^{3-1}$ | 0.7[F] |
| Chipping costs | ZAR $Mg^{-1}$ | 45.6[G] |
| Chipper productivity | Mg $h^{-1}$ | 8[H] |

[A] 35% MC content [444]
[B] 445
[C] Assumed average distance between harvesting location and roadside [444]
[D] Assumed average tractor speed given terrain conditions
[E] 447
[F] Wood density for Acacia dry basis [448]
[G] Based on dry basis [378]
[H] Based on BR-6161 mobile chipper with six knives





**Road Transport**

Capital costs of road transport are estimated for 32-ton heavy-duty trucks with a 27-Mg payload with costs/truckload size curves [378,449]. The capital costs over time are corrected for inflation and empty return trips are considered. Operating costs include mainly wages and fuel use. Two supply chain stages involve trucks for transportation; transporting chips from roadside to pellet plant and pellets to power plant. The main difference between the two transport stages is the travel distances and feedstock type moisture content. Trip distances and moisture content are presented in section 5.7.4 in the supplementary material. Empty returns trips are considered. Total travel times are considered for direct job creation. The main cost parameters of road transport summarized in Table s 5-5.

**Table s 5-5 Cost parameter of road transport**

| Parameter | Unit | Value |
|---|---|---|
| Labor cost | ZAR $Mg^{-1}$ $km^{-1}$ | 0.066 |
| Kilometer rate | ZAR $Mg^{-1}$ $km^{-1}$ | 1.05[A] |
| Truck fuel consumption | $m^3$ $Mg^{-1}$ $km^{-1}$ | 0.000011[B] |
| Speed | Km $h^{-1}$ | 55[C] |
| Loading/unloading cost | ZAR $Mg^{-1}$ | 76.14[D] |

[A] 378,449
[B] 314
[C] Assumed average truck average speed
[D] 180



**Pelletization**

The wood pellet production covers raw material handling and storage, drying, grinding, pelletization, cooling and handling, and pellets storage. Capital investment costs are calculated for a 120,000 Mg $year^{1}$ output plant based on a state-of-the-art pellet production plant [215]. The investment costs are adjusted based on the Chemical Engineering Plant Cost Index (CEPCI). These indices are widely accepted to update investment costs [450]. The updated investment costs (Table SM6) are calculated following Equation S 5-4 based on the capacities of the reference pellet plant's costs and the CEPCI index for 2008 and 2019 (most updated value). Investment costs were annualized using the capital recovery factor (CRF) based on the pellet plant's lifetime and the interest rate. Operation costs such as consumables are assessed as a share of the capital investment costs [215]. Other operating costs such as labor, electricity and diesel use are estimated based on the total required inputs to produce 1 Mg of pellets. The costs of additional biomass necessary for the drying process are also considered. The main cost parameters and required working hours of pelletization are summarized in Table s 5-6.

**Equation S 5-4**

$$Investment\ costs = Referecne\ investment\ costs * ({}^{Cost\ index\ 2019}/_{Cost\ index\ 2008})$$

**Table s 5-6 Pellet production, adapted from 215**

| Parameter | Unit | Value |
|---|---|---|
| Pelletes annual production | Mg year$^{-1}$ | 120,000 |
| Pellets production rate | Mg h$^{-1}$ | 15 |
| Annual operating hours | h | 8,000 |
| Lifetime | year | 15 |
| Total investment | M ZAR | 223.6 |
| CRF | | 0.11 |
| Replacement parts and consumables (dies and rollers and spare parts for the hammer mill) | ZAR Mg$^{-1}$ | 68 |
| Service, maintenance and other costs as a share of the total investments costs for the plant per year | % | 6.2 |
| Labor requirement (considering three shifts) | h year$^{-1}$ | 13,870 |
| Simultaneity factor for electric consumptions | % | 0.85 |
| Energy-e | MJ year$^{-1}$ | 25,142,400 |
| | MJ Mg$^{-1}$ | 209.5 |
| Diesel use | m$^3$ Mg$^{-1}$ | 0.0011 |
| Drying | Rotary drum dryer | |
| Energy-h to evaporate water | MJ | 3960 |
| Boiler efficiency | % | 85 |

## Long-distance maritime transport

Maritime transport of pellets between Port Elizabeth and Rotterdam is performed with Supramax dry bulk carriers. The main characteristics of these vessels are presented in Table s 5-7. The cost of marine shipping can be divided into three main cost components: time charter rates, voyage cost and port costs. Time charter rates are susceptible to macro-economic developments, the linked trade activities, ship supply and demand. Time charters rates are essentially the cost to hire a ship and were assumed of the average 2020 time-charter rates for supramax carriers [431]. The cost of marine fuel (bunker fuel cost) is one of the essential voyage cost components and it was retrieved from the Bunker Index [413]. Port costs are port-location and ship size dependable. Updated tariffs from Port Elizabeth and Rotterdam ports were applied [451–453] considering the vessel dimensions, including the Gross Ton (GT) (Unit of measurement for the gross content of a Seagoing Vessel as referred to in the International Convention on Tonnage Measurement of Ships, London 1969 Treaties journal 1979, no 122 and 194). The amount





of cargo transported in bulk ocean carriers depends on the stowage factor (the volume of space in m$^3$ occupied by a Mg of cargo) and the ship's design. Pellets are stowed at 1.5 m$^3$ Mg$^{-1}$ (bulk density is 0.65 Mg m$^{3-1}$). With a typical design stowage ratio of 1.33 m$^3$ Mg$^{-1}$ the pellet load is volume-limited [454].

**Table s 5-7 Maritime transport costs**

| Parameter | Unit | Value |
|---|---|---|
| Deadweight tonnes (DWT) | Mg | 57,000$^A$ |
| Cargo Capacity | Mg | 54,000$^A$ |
| Total displacement when under ballast | Mg | 29,650$^A$ |
| Ballast | Mg | 14,250$^A$ |
| Total displacement when fully loaded | Mg | 70,000$^A$ |
| Hold volume for cargo | m$^3$ | 72,000$^A$ |
| Speed | Km h$^{-1}$ | 27$^A$ |
| Distance | km | 12169$^B$ |
| Fuel consumption (HFO) at full load | kg km$^{-1}$ | 58.9$^A$ |
| Fuel consumption (HFO) when empty (at ballast) | kg km$^{-1}$ | 44.2$^A$ |
| Empty trip factor | Distance empty total distance$^{-1}$ | 0.43$^A$ |
| Time charter rate Supramax | ZAR day$^{-1}$ | 132,030$^C$ |
| Port costs – Port Elizabeth$^D$ | | |
| Port rates | ZAR ship$^{-1}$ | 54,448.1 |
| VTS charges | ZAR ship$^{-1}$ | 9,595.5 |
| Pilotage | ZAR ship$^{-1}$ | 10,559.1 |
| Tugs vessels asstiance | ZAR ship$^{-1}$ | 39,696.7 |
| Berthing/mooring | ZAR ship$^{-1}$ | 11,283.7 |
| Running of vessel lines | ZAR ship$^{-1}$ | 3,757.4 |
| Exports of biomass | ZAR Mg$^{-1}$ | 7.56 |
| Port costs – Port of Rotterdam | | |
| Port rate | ZAR ship$^{-1}$ | 394,024.6$^E$ |
| Towage | ZAR ship$^{-1}$ | 43,640.3$^E$ |
| Mooring | ZAR ship$^{-1}$ | 20,309.1$^E$ |
| Pilotage | ZAR ship$^{-1}$ | 122,455.5$^F$ |
| Loading/unloading | ZAR Mg$^{-1}$ | 84.5$^G$ |
| Storage (in port) | ZAR Mg$^{-1}$ | 44.9$^G$ |

A 454
B 240
C 431
D 451
E 452
F 453
G 180

5

### 5.7.4 Inventory data for GHG emissions

#### Removal of trees, transporting to roadside and chipping.

Table s 5-8 shows the input data used to calculate the GHG emissions harvesting, transporting to the roadside, debarking and chipping process. Trees are assumed to be removed with chain saws (power sawing, without catalytic converter). Then biomass is transported with tractors to the roadside. An average 8 km distance is assumed between the harvesting site and the roadside [444]. At the roadside, biomass is debarked and chipped [444]. The bark is assumed to be left at the roadside. GHG emissions upstream emissions from fuel production and use were considered as well as dry matter losses of each process. GHG emissions coefficients from fuel are derived from Edwards et al., (2017) [314]. Dry matter losses and moisture content are based on Jonker el., (2014) [217] and primary data [444]. 1 % of dry matter losses are considered for the removal stage, 1% for the transport to the roadside stage and 2 % for the debarking and chipping process. A moisture content of 50% is assumed for this stage [217,444].

**Table s 5-8 overview of input data**

| Process | Unit | Value |
|---|---|---|
| Removal of trees with chain saws | Kg $CO_2$eq $Mg^{-1A}$ | 1.027 |
| Transport to roadside | Kg $CO_2$eq $Mg^{-1}km^{-1B}$ | 358 |
| Debarking (diesel) | Kg $Mg^{-1C}$ | 1.6 |
| Chipping (diesel) | Kg $Mg^{-1D}$ | 3 |

[A] Ecoinvent 3.6, based on 0.13h use to cut 1 t of wood (dry basis) and a GHG coefficient of 7.9 kg $CO_2$eq/h
[B] Ecoinvent 3.6, represents a tractor and trailer transport including empty return.
C 237
D 236

#### Road transport

A spatially explicit approach (GIS tools) based on the available infrastructure and IAP's exploitable biomass potential was used to calculate the distance between the biomass supply locations and the pellet facility located in Port Elizabeth and the transport to a coal-fired power plant. Aggregated distances were based on the optimized total-delivery routes considering the spatial distribution of biomass availability, the road network [378] in the surroundings of Port Elizabeth, the capacity of the pellet facility (120,000 Mg $year^{-1}$) and power plant location. An optimized service area was calculated to consider the distance required to supply the wood demands at the pellet facility. Following the methods in Edwards et al., (2017) [314], 32-Mg heavy-duty trucks with a 27-Mg payload were assumed to transport the wood chips and pellets. Empty return trips were





considered and 1% dry matter losses for each transport process. Table s 5-9 summarizes the aggregated distances.

**Table s 5-9 Transport distances**

| Process | Unit | Value |
|---|---|---|
| Feedstock supply location to pellet facility | km | 82 |
| Pellet facility to the nearest power plant in South Africa[A] | km | 1,000 |

A 455

#### Pelletization

Table s 5-10 summarizes the process parameters of the pelletization facility. The boiler heat demand and fuel requirement were calculated for the drying process based on [314]. Biomass was assumed to be utilized as fuel. Heat demand (MJ $Mg^{-1}$ of water evaporated) considering a reduction of moisture content from 50% to 10% for wood pellets [215]. About 2% of dry matter losses were assumed during pelletization. Country-specific (average grid) electricity GHG emission factor for SA was considered [456].

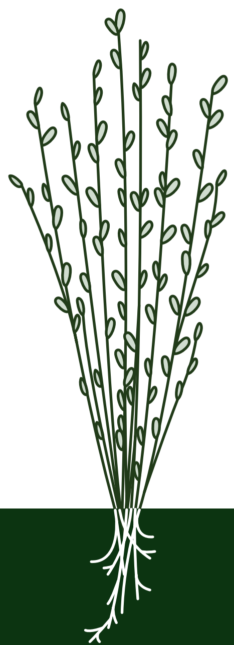
**Table s 5-10 Energy consumption of the pelletization process 215**

| Process | Unit | Coefficient |
|---|---|---|
| Grinding | MJ $Mg^{-1}$ | 67.3 |
| Densification | MJ $Mg^{-1}$ | 136.38 |
| Cooling | MJ $Mg^{-1}$ | 5.4 |
| Diesel use | MJ $Mg^{-1}$ | 39 |
| Heat requirement for drying | MJ $Mg^{-1}$ | 190 |



#### Long-distance maritime transport

Pellets are assumed to be transported with Supramax carriers (57,000 DWT) with a cargo hold volume of 72000 $m^3$. Empty return trips were included and 2% of dry matter losses were accounted for during oceanic transport. The fuel consumption of supramax carriers for pellets with a 0.65 Mg $m^{3\ -1}$ bulk density is approximately 0.066 MJ $Mg^{-1}km^{-1}$ [454]. The impact of using electric grab cranes was also included and based on the work from [239] and are dependable on feedstock bulk density (0.97 MJ $Mg^{-1}$). Local electricity mixes were used for SA and the NL. Data from the maritime distance between ports (port Elizabeth to Rotterdam) was based on actual maritime routes (12,160 km)



# 6

# Land use for bioenergy: synergies and trade-offs between SDGs

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## ABSTRACT

Bioenergy aims to reduce greenhouse gas (GHG) emissions and contribute to meeting global climate change mitigation targets. Nevertheless, several sustainability concerns are associated with bioenergy, especially related to the impacts of using land for dedicated energy crop production. Cultivating energy crops can result in synergies or trade-offs between GHG emission reductions and other sustainability effects depending on context-specific conditions. Using the United Nations Sustainable Development Goals (SDGs) framework, the main synergies and trade-offs associated with land use for dedicated energy crop production were identified. Furthermore, the context-specific conditions (i.e., biomass feedstock, previous land use, climate, soil type and agricultural management) which affect those synergies and trade-offs were also identifed. The most recent literature was reviewed and a pairwise comparison between GHG emission reduction (SDG 13) and other SDGs was carried out. A total of 427 observations were classified as either synergy (170), trade-off (176), or no effect (81). Most synergies with environmentally-related SDGs, such as water quality and biodiversity conservation, were observed when perennial crops were produced on arable land, pasture or marginal land in the 'cool temperate moist' climate zone and 'high activity clay' soils. Most trade-offs were related to food security and water availability. Previous land use and feedstock type are more impactful in determining synergies and trade-offs than climatic zone and soil type. This study highlights the importance of considering context-specific conditions in evaluating synergies and trade-offs and their relevance for developing appropriate policies and practices to meet worldwide demand for bioenergy in a sustainable manner.





## 6.1 INTRODUCTION

Reducing greenhouse gas (GHG) emissions to limit global temperature rise to well below 2°C above pre-industrial levels is crucial for avoiding serious impacts from climate change [1,2]. Meeting GHG emission reduction targets will require various forms of renewable energy. Bioenergy is expected to play an essential role in future energy supply [2,6,7] and could exceed 20% of global (gross) final energy consumption by 2050 [8,26,457]. However, in recent years, the sustainability of large-scale bioenergy deployment has been the subject of fierce debate, with a strong focus on the impacts of using land for dedicated energy crops [24,25,30,458,459].

By reducing GHG emissions, bioenergy can contribute directly to the United Nations Sustainable Development Goals (SDGs) 7 (Affordable and clean energy) and 13 (Climate action) [460]. Although bioenergy aims to reduce emissions, it does not always lead to emission reductions [461]. Furthermore, allocating land to dedicated energy crops can compete with other ecosystem services, such as the provision of food, feed, and fibre [27,462]. In addition, growing dedicated energy crops may compete for land with other climate change mitigation options (e.g., afforestation and solar energy) or the conservation of natural habitats [28,29,463,464]. Competition for land will increase hand in hand with the increase in demand for these services [30]. This competition can lead to trade-offs by diminishing biodiversity and water resources or by causing adverse socio-economic impacts (e.g., increasing the risk in food security) [46]. Conversely, relative to arable land, utilizing land for dedicated energy crops can result in synergies with an increase in, among other things, biodiversity, water conservation, soil carbon stocks and employment (SDG 6 - Clean water and sanitation, SDG 8 - Decent work and economic growth and 15 - Life on Land) [465–467].

Whether land use for dedicated energy crops leads to synergies or trade-offs depends on a wide range of context-specific biophysical and socio-economic conditions [44,468]. Not accounting for these conditions can lead to sustainability impacts and counterproductive land use planning. For example, afforestation programs in China with non-native tree species have exacerbated water shortages [469,470]. Similar effects can occur for certain dedicated energy crops [471]. Despite potentially providing GHG mitigation benefits, the production of dedicated energy crops can simultaneously worsen other sustainability aspects when not considering context-specific conditions, e.g., increased food prices driven by land use change [37]. Therefore, identifying the context-specific conditions that maximize synergies and minimize trade-offs between the impacts of using land for dedicated energy can better inform decision-making on sustainable bioenergy systems.

Some studies have quantified synergies and trade-offs between different sustainability aspects of land use for dedicated energy crops [31,93,472,473]. Nevertheless, the understanding of this topic is still limited [31]. In addition, a synthesis that highlights the context-specific conditions underpinning the synergies and trade-offs between sustainability effects is missing. This lack of understanding hinders the development of sustainable land use strategies in which context-specific biophysical and socio-economic conditions are explicitly considered to prevent undesirable effects, or opportunities to deliver win-win outcomes are increased. Therefore, a synthesis to highlight the effect of context-specific conditions in shaping sustainability is paramount to developing coherent land use strategies.

Using the United Nations SDGs as a framework, this study synthesizes research on synergies and trade-offs between SDGs associated with land use for dedicated energy crops. This is done by reviewing the literature through a pairwise comparison between GHG emission reduction and other sustainability impacts of land use for dedicated energy crop production. Furthermore, the context-specific conditions under which sustainability synergies or trade-offs are found by examining biomass feedstock type, previous land use, climate, soil type and agricultural management leading to each outcome are assessed . Based on this analysis, strategies that can maximize synergies and minimize trade-offs, providing insights and identifying challenges for policy-making and future research are discussed.

## 6.2 MATERIALS AND METHODS

The scope of the study is land use for dedicated energy crops. Hence, bioenergy systems that use, for example, agricultural or forest residues as main feedstock are not considered. The residues definition of the European Commission's Renewable Energy Directive (REDII) to exclude residues studies was applied [15]. Synergies and trade-offs between the effects of land use for dedicated energy crops on SDGs were analyzed. As the objective of many policies promoting bioenergy is climate change mitigation, it was specifically addressed the synergies and trade-offs related to GHG emission savings. Therefore, synergies and trade-offs are analyzed through a pairwise comparison between GHG emission savings and effects in other SDGs. Studies that explicitly define and quantify GHG emission savings, and those that implicitly assume these savings were included. For example, studies that assess the effects on sustainability aspects (e.g., water availability) driven by dedicated energy crop production (biomass potentials) to meet bioenergy demand or climate change targets (implicitly assuming GHG savings).





Given the different definitions of synergies and trade-offs in literature, these terms were explicitly defined in accordance with the definitions provided by the IPCC. Positive connections between mitigation options (in this case, use of land for dedicated energy crop production to reduce GHG emissions) and SDGs are presented as synergies and negative connections as trade-offs [2]. Therefore, a synergy was considered when in addition to mitigating GHG emissions, land use for dedicated energy crops results in a positive effect on another SDG. For example, growing poplar for bioenergy on land previously in use for pasture can mitigate GHG emissions (SDG 13 Climate action) and reduce the risk of soil erosion (SDG 15 life on land) [37]. This positive connection is thus considered a synergy. Conversely, a trade-off occurs when land use for dedicated energy crops results in GHG emission reduction (SDG 13) while adversely affecting another SDG. For example, converting pasture to sugarcane for ethanol production can reduce GHG emissions (SDG 13) but also increase water scarcity (SDG 6 clean water and sanitation) [474]. This negative connection is, therefore, considered a trade-off. A connection was classified as ´no effect´ when no positive or negative effects on SDGs besides GHG emission reduction were reported from using the land for growing dedicated energy crops. Unless explicitly specified, on every occasion where synergies and trade-offs are addressed in the following sections, it refers to those between GHG emission reduction (SDG 13) and other SDGs of using land for dedicated energy crop production for bioenergy.

The review was conducted in two stages, (1) synergies and trade-offs were identified, and (2) the context-specific conditions of these connections were characterized. For the first stage, international peer-reviewed journal papers were analyzed, seeking to extract reported synergies and trade-offs. An integrated search string in Scopus (TITLE-ABS-KEY (bioenergy AND land use AND trade-offs OR synergies/co-benefits) AND DOCTYPE (ar), where "ar" refers to articles) was applied. The search words were based on relevant terms that capture the most important literature regarding synergies (or 'co-benefits'´), trade-offs and bioenergy. Only studies that include both terms "bioenergy" and "land use" were considered. Only articles published from 2010 onwards were included to focus on the most recent developments and ensure a state-of-the-art review. Review style papers were not considered to avoid double counting. The search was complemented with additional relevant peer-reviewed publications suggested by experts from the International Energy Agency Bioenergy Task 45: Climate and Sustainability Effects of Bioenergy within the broader Bioeconomy. An overview of the studies included in the sample is presented in Table S1 of the supplementary material and in section 3 on the observations of synergies and trade-offs of land use for bioenergy section.

For this first stage, a multi-step approach was followed in which first the sustainability effects presented in each study and their direction (positive or negative) were identified.

No threshold was applied on the severity of the sustainability effects; the focus was on the direction of the sustainability effect and not its intensity. Second, these sustainability effects were classified according to the SDGs. This was based on previously defined linkages between sustainability indicators for bioenergy (here taken from the Global Bioenergy Partnership, (GBEP)) and the SDGs [475]. The GBEP has developed a set of indicators for assessing bioenergy sustainability [476]. These indicators are developed for the three pillars of sustainability. Frtischce et al., [475] has developed a framework to link SDGs and GBEP sustainability indicators for bioenergy based on the relevant criteria of each indicator and SDG descriptions. For example, effects on water quality from biomass production were linked with SDG 6 (clean water and sanitation), more specifically with target 6.3 water quality [475]. Thus, effects on water quality were classified within SDG 6. For this first stage of the literature review, sustainability effects were classified on an SDG level, and thus SDG targets and indicators were not specified. Third, based on the SDG classification and the direction of the relation, it was established whether the connection between GHG emission reduction (SDG 13) and other SDGs was a synergy or a trade-off. Fourth, the number of "observed" synergies and trade-offs present in the literature sample were counted. For this study, one "observation" refers to one individual synergy or trade-off found in the literature sample. To illustrate, an identified synergy or trade-off between GHG emission reduction (SDG 13) and life on land (SDG 15) represents one observation.

For the second stage of the review, the scope of the analysis was narrowed to the environmental SDGs and included SDG indicators. The focus was on the environmental SDGs because of the primary relevance of the natural environment for human well-being [477]. Synergies and trade-offs between GHG emission reduction (SDG 13) and SDG 6 (Clean water and sanitation), SDG 15 (Life on land), and other indicators of SDG 13 (Climate action, e.g., albedo effect) are covered. The following context-specific conditions that can determine synergies and trade-offs are considered: previous land use, biophysical characteristics (mainly climate conditions and soil characteristics), management practices and feedstock type. The IPCC climate zone and soil type classes (IPCC 2006) were used to classify the biophysical characteristics of the area of each reviewed study. This was done to harmonize terminology used by different studies to describe biophysical characteristics and ascribe conditions for studies that did not report soil or climate data. The designation 'scale' was used to classify observations in which the geographical scope of the study extended beyond the boundaries of one particular climate zone or soil type. The context-specific conditions for each study were identified in order to link them to the synergies and trade-offs. This allowed determining under which conditions land use for energy crops led to synergies or trade-offs between GHG emission reduction and other SDGs.





## 6.3 OBSERVATIONS OF SYNERGIES AND TRADE-OFFS OF LAND USE FOR BIOENERGY

A total of 134 peer-reviewed studies were found through the systematic literature search. However, studies that focused on agricultural residues were excluded [478–480], land use mapping [481–484], stakeholders' perspectives and decision making [485–488], farmland services (e.g., dairy and poultry) [489–491] and review style studies [492–495]. Thus, in total only 59 studies fell within the scope of the study.

The geographical scope of the sample was mainly limited to North America (39%) and Europe (29%). 7% of studies focused on South America and the Caribbean, and 2% focused on Africa. The geographical scope of the rest of the studies included in the review was worldwide (23%). The study area (scale) of each study varied considerably within the sample, from a few hectares (8%) to a global scale (25%). The scale from a few studies was limited to a continent (7%) or country (7%), while the majority applied a regional scale (53%). Nevertheless, the scale of the regional studies varied considerably in extent. For some studies, a small area was defined by a county, while others comprised an entire region like the Southeastern US. Regarding the approach used by these studies, 73% of studies present in the sample were modelling exercises with a wide range of model types such as input-output models, partial equilibrium models, integrated assessment models and hydrological models (e.g., SWAT). In addition, several studies combined the modelling exercise with other particular methods such as ecosystem service assessment (7%) or a life cycle assessment approach (LCA) (2%). Few studies carried out field trials (8%). The rest of the studies were carried out with an LCA approach (2%), ecosystem service assessment (5%) or a descriptive assessment (3%). A large proportion (74%) of the relevant studies were published in the last years (from 2015), highlighting increasing interest in the topic. Table s 6-1 in the supplementary material presents the overview of the studies in the sample.

A total of 427 observations from the 59 retained studies were classified as either synergies (170), trade-offs (176), or no effect (81) (Figure 6-1). Most of the observations were found between GHG emission reduction (SDG 13) and SDGs 6 (Clean water and sanitation) with 45 synergies, 72 trade-offs and 32 no effects; and SDG 15 (Life on land) with 74 synergies, 56 trade-offs and 31 no effects. Fewer observations were found with SDG 2 (Zero hunger with 1 synergy, 34 trade-offs and 9 no effects), other indicators of SDG 13 (Climate action; 21 synergies, 4 trade-offs and 2 no effects), SDG 8 (Decent work and economic growth; 6 synergies, 10 trade-offs and 2 no effects) and SDG 3 (Good health and well-being; 18 synergies, 0 trade-offs and 4 no effects). One synergy was observed related to SDG 14 (Life below water).

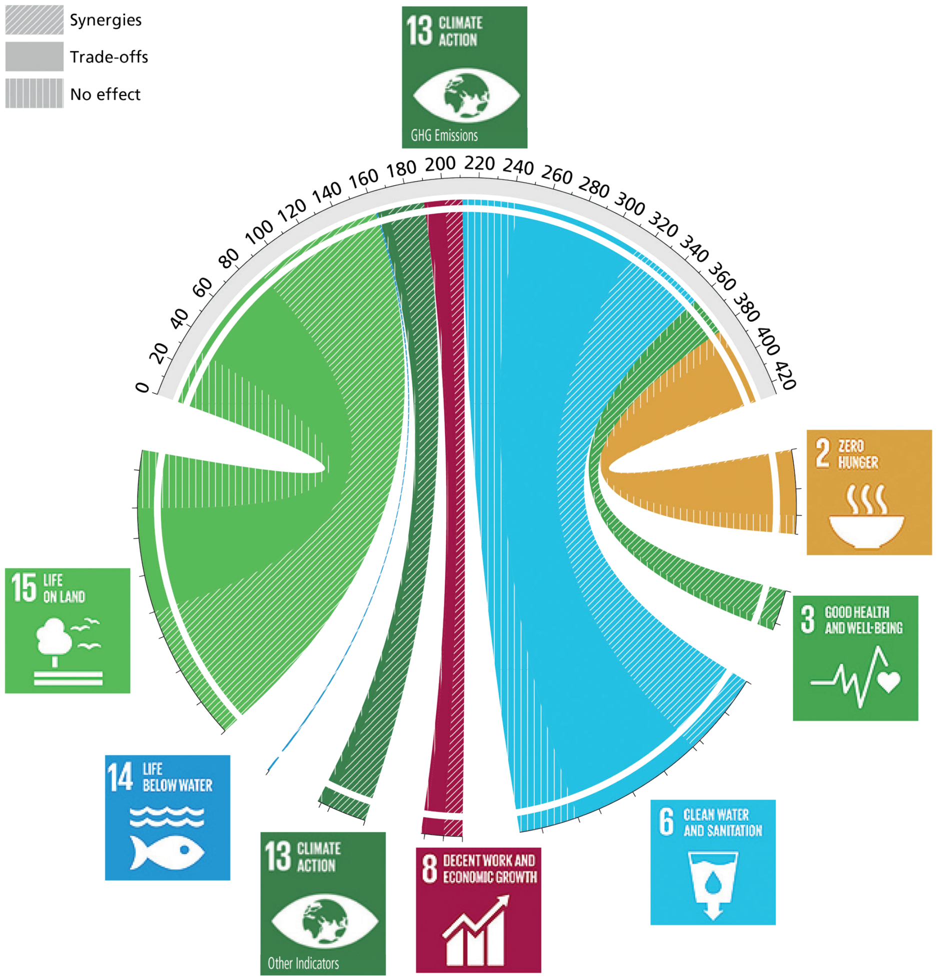


**Figure 6-1. Synergies and trade-offs between effects of growing dedicated energy crops on GHG emission reductions (SDG 13 Climate action) and other SDGs. The width of each connection refers to the number of observations recorded as synergies, trade-offs and no effect found in the literature.**

### 6.3.1 Synergies and trade-offs with SGD 15 – Life on land

Compared to other SDGs, more synergies and trade-offs were found in the sample between GHG emission reductions (SDG 13) and life on land (SDG 15) (Figure 6-1) . SDG 15 aims to protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, halt and reverse land degradation, and halt biodiversity loss. Most synergies are observed with soil quality and biodiversity conservation, while most trade-offs are with the sustainable management of forests. Production of dedicated perennial energy crops can increase soil organic carbon





(SOC) levels compared to the previous land uses such as arable land and pasture. In addition, it can also increase sediment retention, and improve the overall quality of soils, thus contributing to restoring degraded land (SDG 15) [37,40,496–500]. Because perennial crops are not replanted every year, soil disturbance is less frequent than annual crops. Nevertheless, it is also reported that the initial cultivation of dedicated energy crops when non-agricultural land is used can aggravate soil erosion, negatively affect soil quality, and contribute to overall soil degradation [474,501–503].

For some locations and feedstocks, the production of dedicated energy crops was reported to increase species abundances and improve habitat for insects, birds and mammals (SDG 15) [37,497,504–506]. For example, it has been observed that using land for dedicated energy crops can enhance ecosystem conservation and promote pollinator communities, depending on the land use transition [47,507,508]. Other studies reported a trade-off between growing dedicated energy crops and biodiversity. Specifically, habitat deterioration was reported for lands with high ecological and biodiversity values [36,509]. Negative effects on species abundance were also reported in former natural areas [499,510].

Only trade-offs were reported for producing dedicated energy crops and the sustainable management of forests (SDG 15). Without mitigation measures, land used for dedicated energy crops can lead to competition for land with other land-based objectives, which can imply the loss of forest and other important biodiversity areas [511–514]. The majority of studies that presented these trade-offs followed a global geographical scope with an ex-ante type of assessment, in which forest loss was traced as an indirect effect of land competition under different bioenergy, food demand and socio-economic scenarios [46,515–517]. Therefore, trade-offs with sustainable management of forests depended upon additional dynamics such as the development of future food demand [31].

### 6.3.2 Synergies and trade-offs with SGD 13 – Water and sanitation

Fewer synergies and trade-offs were reported between the effects of land used for dedicated energy crops on GHG emission reductions (SDG 13) and clean water and sanitation (SDG 6). Most trade-offs were related to a region (mainly a watershed or a basin), hydrological cycle changes, or the effect of irrigation water withdrawals on water availability. Using land for dedicated energy crops can result in changes in run-off or streamflow, disturbances of evapotranspiration balances, and reductions in water storage [36,502,518–520]. To illustrate, using land for dedicated energy crops such as eucalyptus or oil palm is shown to reduce the streamflow of a watershed as a consequence of higher evapotranspiration rates compared to the previous land uses [83,521]. Reducing GHG emissions and meeting bioenergy targets by the end of the century can substantially increase water withdrawals, intensify the pressure on water resources, and exacerbate water stress [32,82,522–524]. In many regions, the cultivation of dedicated energy crops does not require irrigation. However, in some water-scarce regions, yields would be reduced if no irrigation is applied. Consequently, more land would be required to achieve the same bioenergy production. This can indirectly affect forests (SDG 15) and decrease food-related agricultural production (SDG 2) [31–33]

Most synergies between using land for dedicated energy crops and clean water and sanitation (SDG 6) are related to water quality. Generally, an overabundance of nutrients (mainly phosphorous and nitrogen) in water bodies can deteriorate water quality and affect aquatic life. However, it is shown that the production of some dedicated energy crops on agricultural landscapes can help to reduce these impacts by intercepting nutrient run-off to water bodies and benefit downstream ecosystems [525,526]. Therefore, these are synergies with water quality (SDG 6) and life below water (SDG 14). Lower fertilization and pesticides rates characterize the production of some dedicated energy crops so that the introduction of these crops into agricultural landscapes with previously high fertilization inputs can improve water quality and reduce salinization of water bodies [37,498,499,507,527].

### 6.3.3 Synergies and trade-offs with SDG 2 – Zero hunger

There are considerably more trade-offs than synergies between land use impacts for dedicated energy crops on GHG emission reduction (SDG 13) and zero hunger (SDG 2). Literature has highlighted that food security risks increase when land dedicated to food production is limited by dedicated grown energy crops [46,495,510,513,518,528–530]. Land competition can negatively affect agricultural production and food supply, and lead to increases in food prices [37,511]. Furthermore, it can potentially drive the conversion of natural land to managed systems and affect other SDGs such as SDG 15 [509]. Meeting global bioenergy demand while simultaneously assuring sufficient land for agricultural production for a continuously growing and changing food demand is a challenge [46,514,518]. However, shifting from 1st to 2nd generation crops for bioenergy (i.e. annual crops to lignocellulosic energy crops) can reduce food security risks and can even provide synergies with SDG 2 (Figure 6-1). For example, in the Brandenburg region (Germany), the regional biogas and GHG emissions reduction target can be achieved using miscanthus on marginal land instead of corn on arable land. Due to the comparatively higher yields, less area is required. This results in the release of land previously occupied by corn cultivation for biogas to additional agricultural production [497].

### 6.3.4 Synergies and trade-offs with SDG 8 – Decent work and economic growth

Most of the trade-offs between using land for dedicated energy crops for GHG emission reduction and decent work and economic growth (SDG 8) were related to farmers' revenue. Studies reported negative effects on farmers' revenue when they shifted from 1st to 2nd generation crops as bioenergy feedstock. In most cases, the shift reduced GHG emissions and decreased revenue for farmers [47,502,507,531]. Unfavorable economic conditions, such as low biomass prices and high logistical costs, resulted in lower revenues for dedicated energy crops than food crops [502]. For example, in Illinois (USA), corn for bioethanol is more competitive in term of farmers' revenue than switchgrass under current market conditions and policy incentives [47]. However, in other locations (e.g., East Tennessee), it has been shown that under certain conditions 2nd generation bioenergy crops can provide additional sources of income [526]. For example, farmers can benefit from additional income from harvesting switchgrass at times of the year when there is no other agricultural work available [42]. Still, bioenergy production costs are too high to compete with (conventional) energy costs in several world regions [503]. This disincentivizes farmers to use the land for dedicated energy crops. However, subsidies can promote increasing land use for dedicated energy crops for bioenergy and positively affect income and revenues (SDG 8) [532]. Furthermore, in some cases, wood pellet production from short-rotation coppice (SRC) is cost-competitive for heat and electricity, such as reported for Mexico [533]. Several synergies are also reported between GHG emission reduction and job creation (SDG 8). Using land for dedicated energy crops for bioenergy can create jobs for local communities [36,42,497].





### 6.3.5 Synergies and trade-offs with other indicators from SDG 13 – climate action and SDG 3 – Good health and well-being

Utilizing land for dedicated energy crops shows synergies with climate change adaptation indicators of SDG 13. It has been reported that using land for such purposes can help to mitigate floods, especially when integrating perennials in flood-prone areas in agricultural landscapes [498,534]. This effect is not only crucial for the adaptation to climate-related hazards and natural disasters (SDG 13), which can become more frequent in the upcoming years, but it also contributes to avoiding the high costs of other potential flood mitigation efforts [496]. In addition, integrating perennial crops into arable areas can provide cooling effects through albedo changes and improved air quality [42,473,518].

For good health and well-being (SDG 3), mostly synergies were reported with dedicated energy crop production. Most of these synergies are related to ecosystem cultural services. For example, in Denmark, managing land to grow poplar or oak for bioenergy positively affected recreation and aesthetics services, whereas miscanthus reported neither a positive nor negative effect [535]. Synergies were also reported for other recreational services and wildlife habitats. For example, in Illinois, dedicating land to grow switchgrass instead of corn for bioenergy production was reported to enhance water-based recreation, wildlife viewing and pheasant hunting [47].

## 6.4 CONTEXT-SPECIFIC CONDITIONS THAT SHAPE SYNERGIES AND TRADE-OFFS

Previous land use, feedstock type, soil and climate zone influence the synergies and trade-offs between GHG emissions reduction (SDG 13) and other environmentally-related SDGs. The analysis of these context-specific conditions and how they affect synergies and trade-offs show that previous land use and feedstock type appear to have more influence than other context-specific conditions.

### 6.4.1 Previous land use

Previous land use is an important biophysical attribute that defines trade-offs and synergies. The analysis showed more synergies than trade-offs between environmentally-related SDGs (6, 13, 14 and 15) (Figure 6-2) when agricultural land was used for the production of dedicated energy crops. However, this did not hold true when potential effects of indirect land use change on other SDGs were considered (SDG 2, 3, and 8). Dedicating arable land to the production of dedicated energy crops can result in GHG emission reductions (SDG 13) and simultaneously improve water quality (SDG 6.3); help to strengthen resilience and adaptive capacity to climate-related hazards and natural disasters (SDG 13.1); increase biodiversity (SDG 15.1); and improve soil quality (SGD 15.3). The synergies that are reported for a land use transition from arable land to dedicated energy crops are linked to the feedstock change from annual food crops (e.g., corn and soybean) to perennial energy crops (e.g., perennial grasses and SRC) (Figure 6-3). Conversely, when arable land is dedicated to growing perennial energy crops, it generally results in trade-offs with water-use efficiency (SDG 6.4).

Converting arable land to producing dedicated perennial energy crops has been reported to have synergies with several SDG indicators (Figure 6-2). For example, it was reported that growing perennials on land previously in use for (intensively) managed corn and soy bean decreased nutrient loading at the watershed/basin level (SDG 6.3) and helped to prevent downstream hypoxia episodes, thus benefitting aquatic life (14.1) [525]. Synergies with biodiversity conservation were reported for several locations (e.g., the USA, Germany and the Netherlands) [37,47,497,506,507]. For example, allocating less-profitable areas of arable land to switchgrass has been shown to increase bird species richness [43]. It was also reported that perennial crops provided better habitat conditions compared to annual crops in arable land, which increased the abundance of species such as birds, mammals, and pollinators (e.g., [504]). In the Netherlands, the introduction of miscanthus in intensively managed arable areas positively affected biodiversity [501]. Another synergy relates to improving soil quality by accumulating organic matter in deeper soil layers, providing better conditions to increase sediment retention and decreasing soil erosion





(SDG 15.3) (e.g. [499]). Integrating deep-rooted perennial crops as buffers into agricultural landscapes can help to increase resilience to more extreme climate conditions [534] and to provide mitigation benefits on flood events [496] (SDG 13.1). Conversely, Introducing dedicated energy crops (e.g., eucalyptus) can lead to changes in the hydrological system of a region and affect water availability by increasing water use for irrigation compared to previously arable land (e.g., [83]).

Synergies with water quality (SDG 6.3) from land previously in use as pastures (including intensively managed) generally involved a feedstock change to perennial crops as they require lower fertilization rates than annual crops such as corn. Applying fewer inputs can result in less nutrient run-off and overall water quality improvement [47,501,535]. However, meeting global bioenergy targets with dedicated energy crops produced in areas previously used as pasture (and forest) could double agricultural water withdrawal, increase pressure on water resources and lead to ecosystem degradation [32]. In addition, it can also reduce the soil water storage capacity and reduce water streamflow [36]. For example, the production of oil palm in tabasco on pastures increased evapotranspiration considerably and reduced the water shed stream flow[521]. Nevetheless, for some locations, it is suggested that that the production of poplar in pasture areas can have minimal hydrological effects [82]. When sugarcane was produced on land previously in use for pastures, it provided a slightly worst cover against the impacts of rain, thereby increasing soil loss to a limited extent [474].

Allocating natural or semi-natural areas, including forests, to the production of dedicated energy crops generally led to trade-offs affecting different targets related to life on land (SDG 15), especially target 15.1 biodiversity conservation and 15.2 sustainable use of forest (end deforestation and restore degraded forests) [511,515,536]. Meeting future global bioenergy demand while preserving natural and semi-natural areas and avoiding effects in other ecosystem services can be a significant challenge [45]. In addition, conversion from natural vegetation to dedicated energy crops can reduce carbon stocks in biomass and soil and can therefore result in negative effects on GHG emissions (SDG 13) [46]. Nevertheless, restoring natural landscapes, such as natural grasslands or forests, can provide a valuable source of biomass for bioenergy and (depending on the species grown) could enhance biodiversity conservation by providing species habitats. However, land restoration is undertaken to enhance conservation values and will not necessarily maximize biomass production[508]. In the USA, converting grassland to arable land (corn/soybean rotation for bioenergy purposes) decreased bird communities[505]. It has also been shown that compared to grasslands, introducing switchgrass affects the hydrological cycle by reducing transpiration and increasing evaporation [537].

Although there are fewer observations compared to other land use transitions, using marginal land for dedicated energy crops provided more synergies than trade-offs (Figure 6-2). These synergies were mainly observed between GHG emission reduction (SDG 13) and water quality (SDG 6.3), and to a lesser extent with soil quality (15.3). For example, allocating marginal lands previously used for cropland to dedicated energy crop production reduced nutrient loading to water bodies [527]. In addition, growing perennial crops such as miscanthus and poplar on marginal lands reduced sediment loss and the overall risk of soil erosion [519]. For land already in use for the production of dedicated energy crops, trade-offs are mainly with water-use efficiency (SDG 6.4). In South America and the Caribbean, many countries will have to introduce irrigation schemes to mitigate climate change-induced yield losses in sugar crop plantations to meet bioethanol demand and avoid significant expansion. These irrigation schemes can have impacts on local water resources [523].

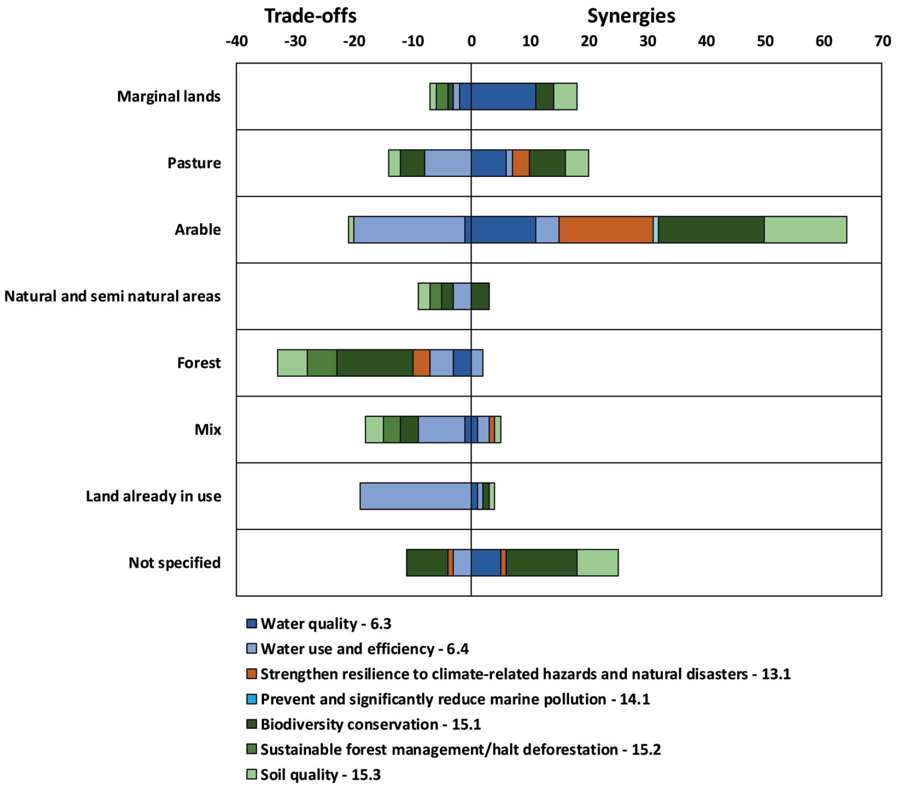


**Figure 6-2. Number of observations recorded as synergies and trade-offs between the effects of using land for dedicated energy crops on GHG emission reductions (SDG 13 Climate action) and other environment-related SDG targets classified according to the type of previous land use. A description of the land use categories can be found in Table S2 of the supplementary material.**





### 6.4.2 Feedstock type

Based on the studies in the sample, utilizing perennial crops and forestry feedstock provides more synergies than trade-offs, compared to annual crops (Figure 6-3). Most of the synergies for perennial crops were related to water quality (SDG 6.3), strengthening resilience and adaptive capacity to climate-related hazards and natural disasters (13.1), and biodiversity conservation (15.1), while the majority of trade-offs are with water use and water-use efficiency (SDG 6.4). Generally, perennial crops require fewer inputs such as fertilizers and pesticides. Therefore, for several locations, nitrogen and phosphorus losses to water consistently decreased when perennials were planted, resulting in an overall water quality improvement [42]. However, the reduction of nutrient losses is relative to the type of previous land use. In most of these cases, land previously in use for the cultivation of annual crops with high inputs was used for dedicated energy crop production [37,47,507,527]. The positive effect of perennial crops on biodiversity, and on resilience and adaptive capacity to climate-related hazards and natural disasters also depends on the previous land use. For example, including perennial crops in agricultural areas with annual crops can mitigate impacts from floods [498,534] and enhance biodiversity [505]. Furthermore, this feedstock transition can also lead to regional cooling via changes in albedo [538]. Conversely, the production of perennials to meet global bioenergy demand can considerably affect biodiversity if carried out in natural areas [509]. Still, perennial crops provide better habitat for species than annual crops [42].

Although most observations with water use and water use efficiency (SDG 6.4) for perennial crops are trade-offs associated with the inclusion of irrigation schemes for high yields and hydrological changes, there are also synergies. Perennial crops are characterized by higher water-use efficiency than annual crops, and can capture more water, i.e., increase interception of lateral flow. For example, under the same conditions, miscanthus is reported to have a higher water-use efficiency than corn [537]. In addition, deep-rooted perennials can have a reduced need for irrigation. However, watershed models show that replacing annual with perennial crops consistently reduces stream flow because of increased evapotranspiration [83,519]. Nevertheless, these effects can also vary across seasons. To illustrate, the production of poplar has minimal effects on annual stream flows. However, evapotranspiration from poplar substantially exceeds that of the previous land use during summer months, decreasing seasonal streamflow considerably [82]. Synergies are also found with soil quality (SDG 15.3) as perennials can accumulate more SOC than annual crops leading to an overall improvement in soil fertility.

For forestry feedstocks such as oak and spruce, mostly synergies have been shown between GHG emissions reduction and biodiversity conservation (SDG 15.1). In Denmark, it is reported that under local biophysical conditions, forestry feedstock types provide better habitat for species and thus a higher conservation potential relative to perennial grasses and SRC [535]. Similar outcomes are reported for other locations when poplar is managed under a short rotation forestry scheme [40]. Forestry feedstocks are also shown to increase soil quality (SDG 15.3) and provide pollination services (SDG 15.1) [535]. They are also reported to have less impact on local water availability than other feedstocks such as miscanthus. These positive effects also depend upon the previous land use, such as arable land or grassland.

For sugar crops, oil palm and annual crops, mostly trade-offs are reported. These feedstock types were reported to negatively affect water availability (SDG 6.4), biodiversity (SDG 15.1) and soil quality (15.3). However, these trade-offs also depend considerably on the local biophysical conditions and previous land use. For example, in Brazil, water availability can be diminished if sugarcane is produced in areas where precipitation conditions are not adequate for plant development and irrigation is required [522]. A similar outcome is reported for annual crops (e.g., maize, wheat and sorghum) at a regional and global scale where developing irrigation schemes in water-scarce areas can affect water availability [33,524]. Local water availability could also be affected in several countries in South America and the Caribbean, where climate change-induced precipitation changes could result in irrigation being required for sugar crops [523]. Producing sugar crops can affect biodiversity and soil quality, e.g., reducing species abundance and increasing soil erosion risk [474,501]. Nevertheless, this effect og sugar crops production relies strongly on the previous land use. In some cases, sugarcane production shows synergies as it increases species abundance and reduces the risk of soil erosion, mainly when carried out on previously arable land under annual crop regimes. Oil palm production has also been shown to have a negative effect on sustainable forest management, and it is suggested that developing oil palm in the Congo Basin can negatively affect the country's sustainable forest management (SDG 15.3) [511].





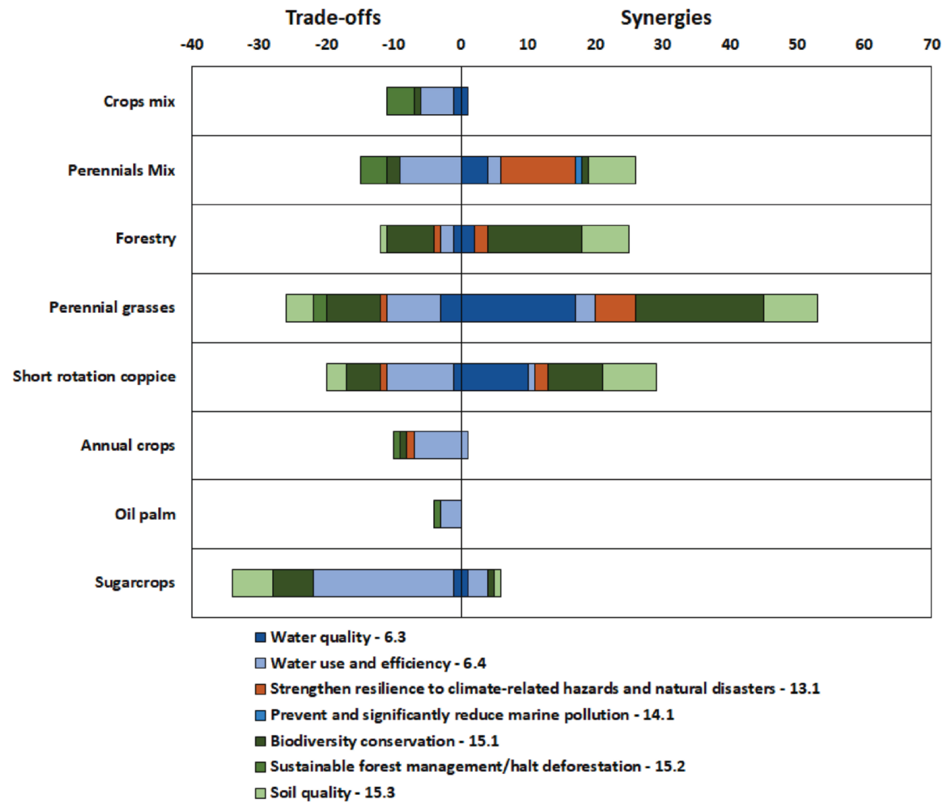


**Figure 6-3 Number of observations recorded as synergies or trade-offs between GHG emission reductions (SDG 13 Climate action) and other environment-related SDG targets for using land for dedicated energy crops, classified according to feedstock type. A description of the feedstock types can be found in Table S3 of the supplementary material.**

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### 6.4.3 Biophysical conditions

For dedicated energy crops grown in a 'cool temperate moist' climate more synergies than trade-offs were reported between GHG emission reduction and other SDGs (see Figure S 6-1 in the supplementary material). Under these climate conditions, more synergies were reported between GHG emission reduction (SDG 13) and water quality (SDG 6.3), strengthening resilience to climate-related hazards and natural disasters (SDG 13.1), biodiversity conversation (SDG 15.1) and soil quality (SDG 15.3). In contrast, more trade-offs were reported with water use and efficiency (SDG 6.4). Most synergies reported for a 'cool temperate moist' climate were also characterized by 'high activity clay' soils (see Figure S 6-2 in the supplementary material). In addition, most synergies were related to the production of perennial crops accompanied by a transition away from arable land, which generally produced more synergies than trade-offs. For example, the production of poplar on land previously in use as arable land in Mulde, Germany, which has a 'cool temperate moist' climate and has 'high activity clay' soils, resulted in positive effects on water quality (SDG 6.3), biodiversity conservation (SDG 15.1) and soil quality (SDG 15.3) [37]. Similar outcomes are reported for other locations with comparable biophysical characteristics. For example, the production of miscanthus in Ireland and switchgrass in Wisconsin (USA) on land previously in use for arable cropping with a 'cool temperate moist' climate and 'high activity clay' soils, positively affected biodiversity [504,505].

Also, more synergies than trade-offs were reported for the 'warm temperate moist' climate, especially related to water quality (SDG 6.3) and soil quality (15.3). Most of these synergies are classified under either 'high activity clay' soils or 'low activity clay' soils. For 'low activity clay' soils, more synergies than trade-offs were reported (see figure SM2 in the supplementary material). These synergies were also related to the production of perennial crops on land previously in use as pasture or cropland [47,526]. However, under this combination of biophysical characteristics, switchgrass production also reported trade-offs by reducing water storage (SDG 6.4), negatively affecting biodiversity (SDG 15.1) and hampering soil quality (SDG 15.3) [36]. Nevertheless, these trade-offs were also traced to a change in a combination of land use categories to produce dedicated energy crops, i.e., savannah, grasslands, pasture and barren land (see category Mix in Figure 2). It was also reported that allocating land to the production of dedicated energy crops under 'warm temperate moist' and 'low activity clay soil' negatively affected the sustainable use and managament of forest via indirect land use change (SDG 15.2) [539]

Other climate zones and soil types were not well represented in the literature. However, studies in the 'tropical' (moist and wet) climate zone reported more trade-offs than synergies between GHG emission reduction and other SDGs, especially with water use and efficiency (SDG 6.4). For example, oil palm production in Mexico reduced watershed streamflow due to an increase in evapotranspiration rates [521]. In addition, in the same climate zone, it was shown that the production of oil palm can affect the sustainable use of forests and biodiversity (SDG 15.1 and 15.2) [511]. Water availability trade-offs were also reported for sugarcane production under a 'tropical moist' climate zone and 'low activity clay' soil [474]. The production of eucalyptus was also shown to lead to streamflow reduction and affect water quantity (SDG 6.4) under a 'tropical moist' climate zone and 'high activity clay' soils [83].

The large majority of synergies and trade-offs were reported under the 'scale' classification for climate zone and soil type (studies in which the geographical scope extended beyond the boundaries of one particular climate zone or soil type). Most of the studies classified under 'scale' had a global, national or regional focus. Most of the trade-offs were related to water use and efficiency (SDG 6.4) and, to a lesser extent, to





biodiversity conservation (15.1). Regardless of the feedstock type and previous land use, it was generally shown that deploying large-scale bioenergy systems to meet global and national bioenergy demand can lead to unsustainable water withdrawals in the form of irrigation [523,524,540]. These trade-offs were generally reported through the nexus between land and water and its effects on biodiversity conservation (SDG 15.1) [509,514] and the sustainable use of forest (SDG 15.2) [46,516,541].

## 6.5 DISCUSSION

### 6.5.1 Key findings

This study compiled state-of-the-art knowledge on synergies and trade-offs between GHG emission reduction from utilizing land for dedicated energy crops production (SDG 13) and other SDGs. Context-specific conditions under which sustainability synergies or trade-offs occur, particularly for environmentally-related SDGs were identified. The findings suggest that using land for dedicated energy crops results overall in almost an equivalent number of synergies and trade-offs between SDG 13 and other SDGs, just a few more trade-offs were reported. Note that more synergies were found for SDGs 3, 13, 14 and 15, and more trade-offs for SDGs 2, 6 and 8. Dedicated energy crops can compete for land with food production and therefore (directly or indirectly) affect food supply and food price [542]. In addition, the pressure on food markets can intensify when annual crops such as corn are allocated to bioenergy [543]. However, this effect is not exclusively negative: increased food prices can harm consumers, especially low-income households, but can simultaneously benefit farmers through higher profits [544]. Still, the degree to which an increase in food prices is directly driven by using land to produce dedicated energy crops is difficult to determine, as food prices depend on interactions among many variables [545].

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In general, the findings show that trade-offs between SDG 13 and clean water and sanitation (SDG 6) are mostly related to water use and efficiency (SDG 6.4). In contrast, most synergies are related to water quality (SDG 6.3). Water demand for dedicated energy crops can increase water withdrawal through irrigation and increase water scarcity. However, carefully choosing locations to produce dedicated energy crops for bioenergy and choosing feedstocks adapted to local conditions can minimize pressures on water availability [464]. Synergies and trade-offs with SDG 15 are both consistently reported. Depending on context-specific conditions, it is shown that producing dedicated energy crops can positively or negatively affect biodiversity (SDG 15.1). Also, using arable land to grow dedicated energy crops can improve soil quality (15.3). Generally, perennials provide better protection to soil erosion and increase soil quality [546]. However, dedicating land to produce dedicated energy crops can also negatively affect sustainable forest management (15.2) without concrete mitigation measures. Utilizing land for dedicated energy crops without economic incentives results in trade-offs with farmers' revenues and income (SDG 8) but also in synergies with good health and well-being (SDG 3).

A combination of context-specific conditions influences synergies and trade-offs. Most synergies were observed when perennial crops were produced on marginal land, previous arable land or pasture under a 'cool temperate moist' climate zone and 'high activity clay' soils. However, synergies arising from the production of perennials on land previously in use as cropland were also reported for other climate zones such as 'cool temperate dry' and 'warm temperate moist'. Other feedstock types such as sugarcane also present synergies across different climatic zones and soil types. However, these synergies are limited to specific land use transitions. In addition, it is also shown that regardless of the feedstock type, allocating natural or semi-natural areas, including forests, to the production of dedicated energy crops generally leads to trade-offs. Therefore, these findings suggest that previous land use and feedstock type appear to be more relevant in modulating synergies and trade-offs than climatic zones and soil types.

### 6.5.2 Strategies and challenges to reduce trade-offs

This literature review showed that using abandoned arable land or marginal lands to produce perennial crops as bioenergy feedstock is an option that may maximize synergies and minimize trade-offs between GHG emission reduction and SDGs. On marginal land, perennial crop production is less likely to compete with other land-based services (e.g., food security SDG 2) and to generate displacement effects [298]. In addition, it is recognized that utilizing marginal lands for perennial crop production can have advantageous effects on biodiversity and contribute to land restoration [27]. For example, utilizing marginal lands to produce perennials can enhance biodiversity (SDG 15) by avoiding directly or indirectly the conversion of natural ecosystems for bioenergy production [547]. Furthermore, compared to annual crops, perennial crops can sequester more carbon in biomass and soil, improve soil quality, and reduce soil erosion (SDG 15.3) [21,34]. However, the total (current and future) marginal land area that can be dedicated to producing dedicated energy crops is highly uncertain [27]. In addition, the performance (e.g., yield) of perennials on marginal lands also remains uncertain [25]. So far, only a limited number of trials have been carried out with perennial feedstocks on marginal lands and these perennial crops are also not commonly planted by farmers [548,549]. In addition, feedstock production costs in marginal conditions are relatively high [334]. These conditions, including market uncertainties and lack of knowledge, affect the farmer's willingness to adopt perennial crop production systems in general. Nevertheless, there are new practices, techniques and technologies such as precision farming, reduced tillage and soil sampling that could improve the cultivation of perennial crops on marginal lands, achieve higher





yields and reduce costs. For example, carefully selecting well-adapted perennial crop species to local biophysical conditions, considering the crop's chemical and physical characteristics for specific end-uses applying a life cycle perspective [550]. This selection process can avoid additional pre-processing steps before the conversion process and provide more competitive production costs. Yields can be enhanced by identifying the morphological or physiological traits that allow plants to thrive in marginal conditions and enhance these traits in perennial crops through new breeding technologies[551]. Shifting from rhizome-based to seed-based or stem-based establishment practices has shown to reduce perennial grasses production costs [552]. The implementation of perennial crop production systems should focus on reducing or overcoming negative farmer income and revenue (SDG 8) and promoting the sustainability benefits of other SDGs.

To overcome barriers of changing to perennial bioenergy production systems, changes in the rewarding systems, policies and practices are needed. Introducing perennial crop systems should include long-term contracts with fixed prices and tax credits [553] to help guarantee farmers' income. In addition, policy can focus on internalizing the value of positive externalities. For example, by creating schemes to reward farmers for sequestering carbon, restoring ecosystems, enhancing biodiversity, increasing soil quality and improving water quality. Smart feedstock choices within perennials can minimize adverse effects on water availability while considering location-specific biophysical characteristics, thus resulting in more water-efficient biomass production systems. Irrigation schemes could be applied to obtain higher yields and reduce overall costs. However, irrigation schemes for bioenergy systems are controversial and should be avoided in water-stressed regions. Biomass production costs, dominated by stable costs related to land rent, are largely fixed per unit of land, and thus, overall production costs can decrease with higher yields [548].



### 6.5.3 Uncertainties of the study

The results of this review should be interpreted with care. The number of observations is limited to the sample size of relevant studies (59 relevant studies were included). Studies that failed to meet the search string or were not suggested by the board of experts were excluded. For example, the sample did not include studies that assessed the impacts of biomass production under different indicators that did not include the terms "synergy" or "trade-off". In addition, a focus was given to reporting and linking the number of synergies and trade-offs between GHG emission reduction (SDG 13) and other SDGs found in the literature. Although this allowed us to quantify how often a synergy or trade-off is studied, it falls short of indicating the strength of the connection. For example, more synergies related to biodiversity conversation were generally encountered for perennial grasses than for SRC (SDG 15.1). However, in some cases, growing SRC could be more beneficial for biodiversity than growing perennial grasses. For example, Sántha and Bentsen (2020) suggest that managing land for poplar rather than miscanthus production has a stronger positive effect on biodiversity. The magnitude of synergies and trade-offs depends on the context and therefore needs to be considered when designing bioenergy systems.

The analysis intended to serve as a basis to expand the current knowledge on synergies and trade-offs between SDG 13 and other SDGs. Most of the observations presented in the study are based on modeling assessments limited to the "Global North". In addition, a large part of the observations' study areas extended extensively beyond the boundaries of one particular climate zone or soil type. Therefore, other geographical regions, climate and soil types are under-represented. More observations from the "Global South", different climate zones and soil types are required to better understand synergies, trade-offs and potential trends in sustainability effects of using land for dedicated energy crop production. In addition, there are still relatively few trials to study the actual effects of growing novel feedstocks such as perennials for bioenergy systems on SDGs [554]. Furthermore, synergies and trade-offs between SDGs are also directly related to the scale of bioenergy implementation [27]. For example, regarding biodiversity, possible synergies are mainly reported on the field level, while the negative effects are reported across field, region, continent or global scales (Immerzeel et al., 2014). Thus, the effect of synergies and trade-offs also depends upon scale.

The scope of this review is limited to analyzing context-specific conditions through direct observations reported primarily around environmentally-related SDGs. However, bioenergy-induced synergies and trade-offs with socio-economic SDGs, such as SDG2 'zero hunger', also depend on context-specific conditions [44] and were not considered. The study provides a first impression of the relevance of context-specific conditions and how they affect synergies and trade-offs. Other analytical approaches such as regression analysis could be more suitable to identify the hierarchical relevance of context-specific conditions in determining specific positive or negative effects [555]. In addition, the included studies specifically mentioned or assumed that growing dedicated energy crops leads to GHG emissions reduction (progressing SDG 13) and thus, bioenergy crop production provides synergies when it leads to positive effects for another SDG. Nevertheless, land use transitions can also result in additional GHG emissions by disturbing carbon stocks in biomass and soils [118].

Determining actual synergies related to GHG emissions reductions from bioenergy requires a full supply chain perspective within the entire energy system as GHG performance of bioenergy value chains depends on supply chain designs, logistics, and end uses, and not just land management [144,550,556]. Similar positive and negative pairwise correlations between SDGs can also occur at later stages of the supply chain. For example, synergies exist between GHG emissions reduction and job creation (SDG 8) in the bioenergy sector (e.g., processing and transporting biomass) [380]. Furthermore, significant indirect effects across connected systems, induced by market perturbations, may require a broader consequential life cycle perspective to be captured [557].





Although synergies and trade-offs between GHG emissions (SDG 13) and other environmental SDGs were assessed through a pairwise comparison, effects on SDGs are shown to be interconnected. For example, utilizing arable land for dedicated energy crops can displace food production, expanding agricultural activities into natural areas [522]. This process of indirect land use change (iLUC) can simultaneously affect biodiversity (SDG 15) and global food prices (SDG 2) [509]. In contrast, interconnected effects can also result in synergies. For example, it can result in monetary benefits (SDG 8) from nitrate (SDG 6) and sediment retention (SDG 15), and increase the recreational and aesthetic value of the landscape [47,535]. Sediment retention can also contribute to reducing the adverse effects of floods (SDG 13) and the costs of flood mitigation projects (SDG 8 and SDG 13) [496,534]. Understanding these interconnected effects can expand the knowledge on synergies and trade-offs towards more sustainable land systems and be addressed in future research as well as to include other land-based feedstock types such as residues.

Although management practices are an important part of the context that determines whether synergies or trade-offs will occur, it was not possible to identify the effects of management practices in this study. The variety of management practices connected to feedstock types, scopes, geographical and temporal scales prohibited us from discerning clear patterns. However, synergies with water quality are shown for perennial crops partially as a result of fewer inputs (e.g., fertilizers) than annual crops [47,507,527,558]. Therefore, from a management practice perspective, the low requirements of perennial crops for chemical inputs are a characteristic that can potentially lead to more synergies with various SDGs than other feedstock types.



The differences in scope and approach with similar studies result in a challenge for comparison purposes. For example, this study focused on synergies and trade-offs between GHG emissions reduction (SDG 13) from using land for dedicated energy crops and six other SDGs that were found in the literature sample. However, bioenergy is a potential option to achieve "Affordable and clean energy" (SDG 7). In addition, more interlinkages between SDGs such as "Responsible consumption and production" (SDG 12) could be established under a different scope. To illustrate, producing dedicated energy crops can synergize with SDG 12 as it provides a more responsible way of producing and consuming energy than fossil fuels. For example, Nerini et al. (2018) identified that 16 SDGs are related to achieving affordable and clean energy. However, Nerini et al. (2018) assessed the whole energy sector and not specifically dedicated energy crop production. Studies with other biomass types have also found that developing bioenergy systems can indirectly result in synergies with societal related SDGs such as "No poverty" (SDG 1), "Quality education" (SDG 4) "Gender inequality" (SDG 5) and "Reduced inequalities" (SDG 10) [460,560]. Also, as shown in this study, Blair et al. (2021) reported that bioenergy systems have several synergies and trade-offs with SDG 6 and 15. It has also been reported that sustainable soil management is intrinsically related to 11 SDGs [561]. As allocating land for bioenergy crops can improve soil quality, it can also be related to additional SDGs like the ones shown in Lal et al. (2021).

## 6.6 CONCLUSION

This literature review synthesizes the current understanding of the synergies and trade-offs between the impacts of land use for dedicated energy crops to reduce GHG emissions (SDG 13) and other SDGs, and identifies the context-specific conditions that determine these relationships. Overall, an almost equal number of synergies and trade-offs were found between GHG emission reduction and SDGs 2 (Zero hunger), 3 (Good health and well-being), 6 (Clean water and sanitation), 8 (Decent work and economic growth), 13 (Climate action-other indicators), 14 (Life below water) and 15 (Life on land). However, more synergies were found related to SDGs 3, 13, 14 and 15, while more trade-offs were found related to SDGs 2, 6 and 8. Most synergies related to environmental SDGs were observed when perennial crops (SRC and perennial grasses) were produced on arable, pasture or marginal land in a 'cool temperate moist' climate zone and 'high activity clay' soils. Utilizing marginal land to produce perennial crops is a key strategy as it can avoid trade-offs with other land-based services such as food, feed, and fiber production (SDG 2). To minimize trade-offs, the findings suggest that it is of paramount importance to consider context-specific conditions, with priority given in the order of first land use transitions and second feedstock types, while both parameters need to be considered in line with local biophysical conditions to contribute to multiple SDGs. The magnitude of synergies and trade-offs between GHG emission reduction and other SDGs must be accounted for in decision making and from a supply chain perspective approach. Otherwise, it can lead to suboptimal implementation strategies and negative effects. This study highlights the importance of considering context-specific conditions to analyze synergies and trade-offs, informing appropriate policies and practices to meet worldwide demand for bioenergy, especially in regions with strongly competing needs for land for various land-based ecosystem services.





## 6.7 SUPPLEMENTARY MATERIAL

**Table s 6-1 overview of the studies accruing to geographical scope, study type and study area**

| Study | Year | Geographical scope | Study type | Study area |
|---|---|---|---|---|
| Water-land tradeoffs to meet future demands for sugar crops in Latin America and the Caribbean: A bio-physical and socio-economic nexus perspective | 2021 | The Caribbean and South America | Modelling exercise | Continent |
| Spatially variable hydrologic impact and biomass production tradeoffs associated with Eucalyptus (E. grandis) cultivation for biofuel production in Entre Rios, Argentina | 2021 | The Caribbean and South America | Modelling exercise | Region |
| Growing grasses in unprofitable areas of US Midwest croplands could increase species richness | 2021 | North America | Modelling exercise | Region |
| Combining mitigation strategies to increase co-benefits for biodiversity and food security | 2020 | Worldwide | Modelling exercise | Global |
| Hydrologic impacts and trade-offs associated with developing oil palm for bioenergy in Tabasco, Mexico | 2020 | North america | Modelling exercise | Region |
| Potential bioenergy production from miscanthus x giganteus in brandenburg: Producing bioenergy and fostering other ecosystem services while ensuring food self-sufficiency in the Berlin-Brandenburg region | 2020 | Europe | Modelling exercise | Region |
| Bioenergy with Carbon Capture and Storage (BECCS): Finding the win–wins for energy negative emissions and ecosystem services—size matters | 2020 | Europe | Ecosystem service assessment | Region |
| Land management and climate change determine second-generation bioenergy potential of the US Northern Great Plains | 2020 | North america | Modelling exercise | Region |
| Ecosystem service benefits and trade-offs-selecting tree species in Denmark for bioenergy production | 2020 | Europe | Ecosystem service assessment | Country |
| Ant biodiversity and ecosystem services in bioenergy landscapes | 2020 | North america | Field | ha |
| Beneficial land use change: Strategic expansion of new biomass plantations can reduce environmental impacts from EU agriculture | 2020 | Europe | Ecosystem service assessment | Continent |
| Spatial Variation in Environmental Impacts of Sugarcane Expansion in Brazil | 2020 | The Caribbean and South America | Modelling exercise | Region |
| Biomass Production with Conservation Practices for Two Iowa Watersheds | 2020 | North America | Modelling exercise | Region |

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**Table s 6-1 overview of the studies accruing to geographical scope, study type and study area (continued)**

| Study | Year | Geographical scope | Study type | Study area |
|---|---|---|---|---|
| Perennials in Flood-Prone Areas of Agricultural Landscapes: A Climate Adaptation Strategy | 2020 | North America | Descriptive assessment | Region |
| Natural climate solutions versus bioenergy: Can carbon benefits of natural succession compete with bioenergy from short rotation coppice? | 2019 | Worldwide | Modelling exercise | Global |
| Identifying trade-offs between socio-economic and environmental factors for bioenergy crop production: A case study from northern Kentucky | 2019 | North America | Modelling exercise | Region |
| Hydrologic impacts and trade-offs associated with forest-based bioenergy development practices in a snow-dominated watershed, Wisconsin, USA | 2019 | North America | Modelling exercise | Region |
| Valuation of ecosystem services in alternative bioenergy landscape scenarios | 2019 | North America | Modelling exercise and Ecosystem service assessment | Region |
| The land-water nexus of biofuel production in Brazil: Analysis of synergies and trade-offs using a multiregional input-output model | 2019 | The Caribbean and South America | Modelling exercise | Country |
| Global advanced bioenergy potential under environmental protection policies and societal transformation measures | 2019 | Worldwide | Modelling exercise | Global |
| Bioenergy and ecosystem services trade-offs and synergies in marginal agricultural lands: A remote-sensing-based assessment method | 2019 | Europe | Modelling exercise | Region |
| Estimating water–food–ecosystem trade-offs for the global negative emission scenario (IPCC-RCP2.6) | 2018 | Worldwide | Modelling exercise | Global |
| Large-scale bioenergy production: How to resolve sustainability trade-offs? | 2018 | Worldwide | Modelling exercise | Global |
| Exploring SSP land-use dynamics using the IMAGE model: Regional and gridded scenarios of land-use change and land-based climate change mitigation | 2018 | Worldwide | Modelling exercise | Global |
| Can upstream biofuel production increase the flow of downstream ecosystem goods and services? | 2018 | North America | Descriptive assessment | Region |
| Evaluation of bioenergy crop growth and the impacts of bioenergy crops on streamflow, tile drain flow and nutrient losses in an extensively tile-drained watershed using SWAT | 2018 | North America | Modelling exercise | Region |





**Table s 6-1 overview of the studies accruing to geographical scope, study type and study area (continued)**

| Study | Year | Geographical scope | Study type | Study area |
|---|---|---|---|---|
| Global consequences of afforestation and bioenergy cultivation on ecosystem service indicators | 2017 | Worldwide | Modelling exercise | Global |
| Bioenergy production and forest landscape change in the southeastern United States | 2017 | North america | Modelling exercise | Region |
| Synergies and trade-offs in renewable energy landscapes: Balancing energy production with economics and ecosystem services | 2017 | North america | Modelling exercise | Region |
| The bioenergy potential of Natura 2000 – a synergy between climate change mitigation and biodiversity protection | 2016 | Europe | LCA approach | Continent |
| Candidate perennial bioenergy grasses have a higher albedo than annual row crops | 2016 | North america | Field | ha |
| Assessing regional-scale impacts of short rotation coppices on ecosystem services by modeling land-use decisions | 2016 | Europe | Modelling exercise | Region |
| Potential impacts on ecosystem services of land use transitions to second-generation bioenergy crops in GB | 2016 | Europe | Modelling exercise and Ecosystem service assessment | Region |
| Competition between food, feed, and (bio)fuel: A supply-side model based assessment at the European scale | 2016 | Europe | Modelling exercise | Continent |
| Alternative scenarios of bioenergy crop production in an agricultural landscape and implications for bird communities | 2016 | North america | Modelling exercise | Region |
| Trade-offs between land and water requirements for large-scale bioenergy production | 2016 | Worldwide | Modelling exercise | Global |
| Perennial species mixtures for multifunctional production of biomass on marginal land | 2016 | Europe | Field | ha |
| Land-use change to bioenergy: Grassland to short rotation coppice willow has an improved carbon balance | 2016 | Europe | Field | ha |
| Spatial land use trade-offs for maintenance of biodiversity, biofuel and agriculture | 2015 | North america | Modelling exercise | Region |
| Water-energy Nexus: A case of biogas production from energy crops evaluated by Water Footprint and Life Cycle Assessment (LCA) methods | 2015 | Europe | Modelling exercise and LCA approach | Region |
| Yield-biodiversity trade-off in patchy fields of Miscanthus × giganteus | 2015 | Europe | Field | ha |

**Table s 6-1 overview of the studies accruing to geographical scope, study type and study area (continued)**

| Study | Year | Geographical scope | Study type | Study area |
|---|---|---|---|---|
| Comparing bioenergy production sites in the southeastern US regarding ecosystem service supply and demand | 2015 | North america | Modelling exercise and Ecosystem service assessment | Region |
| Sustainable bioenergy options for Mexico: GHG mitigation and costs | 2015 | North america | Modelling exercise | Country |
| Assessing multimetric aspects of sustainability: Application to a bioenergy crop production system in East Tennessee | 2015 | North America | Modelling exercise | Region |
| Trade-offs of different land and bioenergy policies on the path to achieving climate targets | 2014 | Worldwide | Modelling exercise | Global |
| Modeling Impact of Development Trajectories and a Global Agreement on Reducing Emissions from Deforestation on Congo Basin Forests by 2030 | 2014 | Africa | Modelling exercise | Region |
| Forecasting changes in water quality in rivers associated with growing biofuels in the Arkansas-White-Red river drainage, USA | 2014 | North America | Modelling exercise | Region |
| Global bioenergy scenarios - Future forest development, land use implications and trade-offs | 2013 | Worldwide | Modelling exercise | Global |
| Optimization-based trade-off analysis of biodiesel crop production for managing an agricultural catchment | 2013 | Europe | Modelling exercise | Global |
| Implications of agricultural bioenergy crop production in a land constrained economy - The example of Austria | 2013 | Europe | Modelling exercise | Country |
| Ecosystem-service tradeoffs associated with switching from annual to perennial energy crops in Riparian zones of the US Midwest | 2013 | North America | Modelling exercise and Ecosystem service assessment | Region |
| A regional comparison of water use efficiency for miscanthus, switchgrass and maize | 2012 | North america | Modelling exercise | Region |
| Dependency of global primary bioenergy crop potentials in 2050 on food systems, yields, biodiversity conservation and political stability | 2012 | Worldwide | Modelling exercise | Global |
| Spatial variation of environmental impacts of regional biomass chains | 2012 | Europe | Modelling exercise | Region |





**Table s 6-1 overview of the studies accruing to geographical scope, study type and study area (continued)**

| Study | Year | Geographical scope | Study type | Study area |
|---|---|---|---|---|
| Multimetric spatial optimization of switchgrass plantings across a watershed | 2012 | North America | Modelling exercise | Region |
| Projected water consumption in future global agriculture: Scenarios and related impacts | 2011 | Worldwide | Modelling exercise | Global |
| Environmental impacts of water use in global crop production: Hotspots and trade-offs with land use | 2011 | Worldwide | Modelling exercise | Global |
| Biomass energy production in agriculture: A weighted goal programming analysis | 2011 | Europe | Modelling exercise | Region |
| The economic potential of bioenergy for climate change mitigation with special attention given to implications for the land system | 2011 | Worldwide | Modelling exercise | Global |

**Table s 6-2 classification of land use categories used in Figure 6-2**

| Land use | Description |
|---|---|
| Land in use | Land already used for dedicated bioenergy crops |
| Forest | Natural forest |
| Natural (excluding forest) and semi-natural areas | Natural areas such as protected areas with dominant vegetation of shrublands or grasslands |
| Pasture | Managed grasslands, including categories such as rangeland, grazed pasture and ungrazed pasture |
| Arable land | Combination of land management categories such as annual croplands, and orchards |
| Marginal land | Combination of land use categories that include a level of marginality such as marginal pastures, marginal grassland and marginal agricultural land |
| Mix land use categories | Combination of indistinguishable land management categories. Therefore, assessments are carried out on a general level for different land use categories. For example, pasture, cropland and forest |
| Not specified | Land use or land use transition is not involved or mentioned |

**Table s 6-3 classification of land use categories used in Figure 6-3**

| Feedstock | Description |
|---|---|
| Sugar crops | Sugarcane and sugar beet |
| Oil palm | Oil palm |
| Annual crops | Corn, corn-soybean rotation, rapeseed, sorghum and wheat |
| Perennial mix | Combination of feedstocks involving short-rotation crops and perennial grasses |
| Short rotation coppice | Short rotation coppice species such as poplar, willow eucalyptus and American sycamore |
| Perennial grasses | Miscanthus, switchgrass and grasses |
| Forestry | Short rotation forest and long rotation forest including categories such as norway spruce, sitka spruce, European beech and pedunculate oak |
| Crop mix | Combination of different feedstocks grown includes both annual and perennial crops |





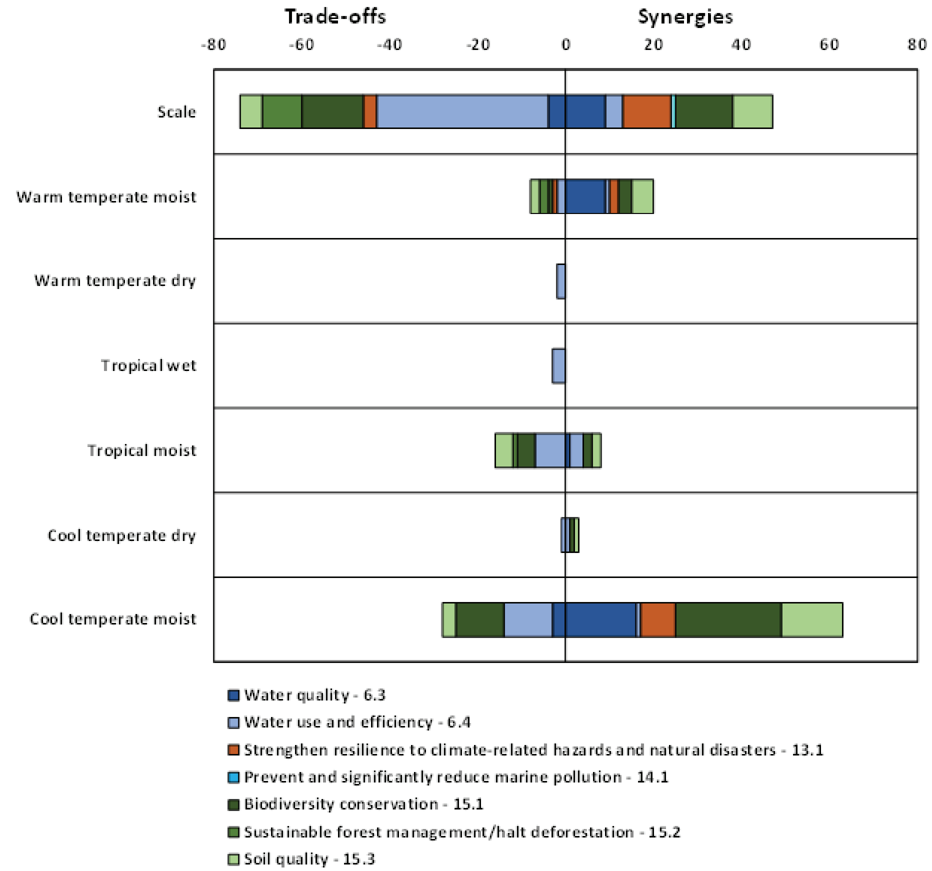


**Figure S 6-1 Number of observations recorded as synergies and trade-offs between effects of using land for dedicated energy crops on GHG emission reductions (SDG 13 Climate action) and other environment-related SDGs classified according to climate zone.**

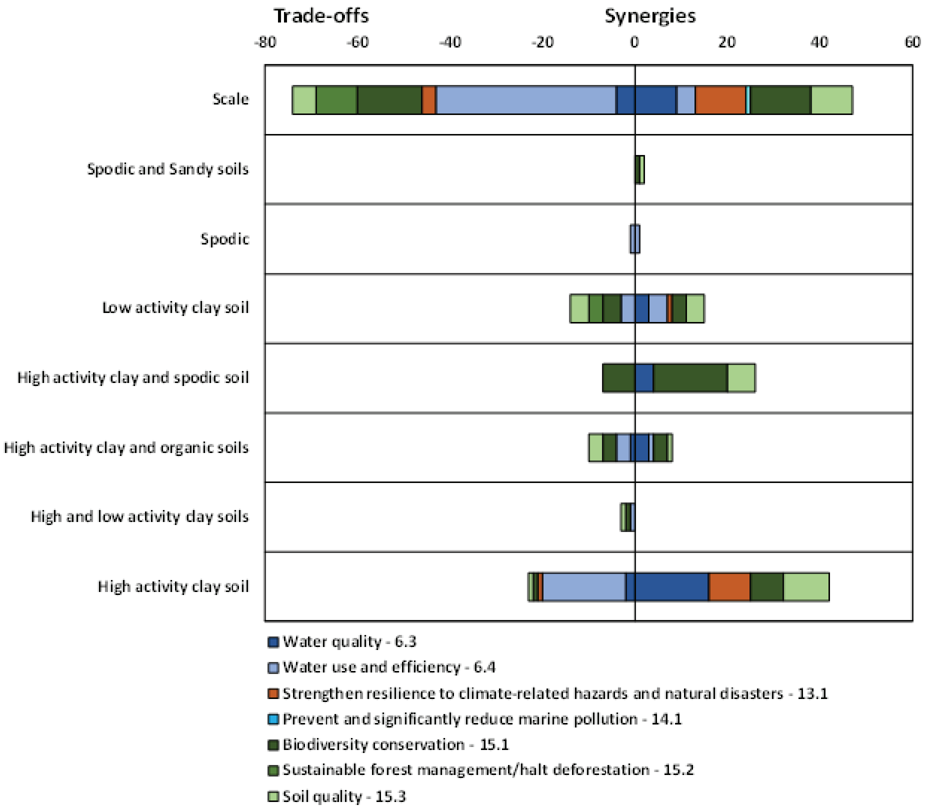


**Figure S 6-2 Number of observations recorded as synergies and trade-offs between effects of using land for dedicated energy crops on GHG emission reductions (SDG 13 Climate action) and other environment-related SDGs classified according to soil type.**

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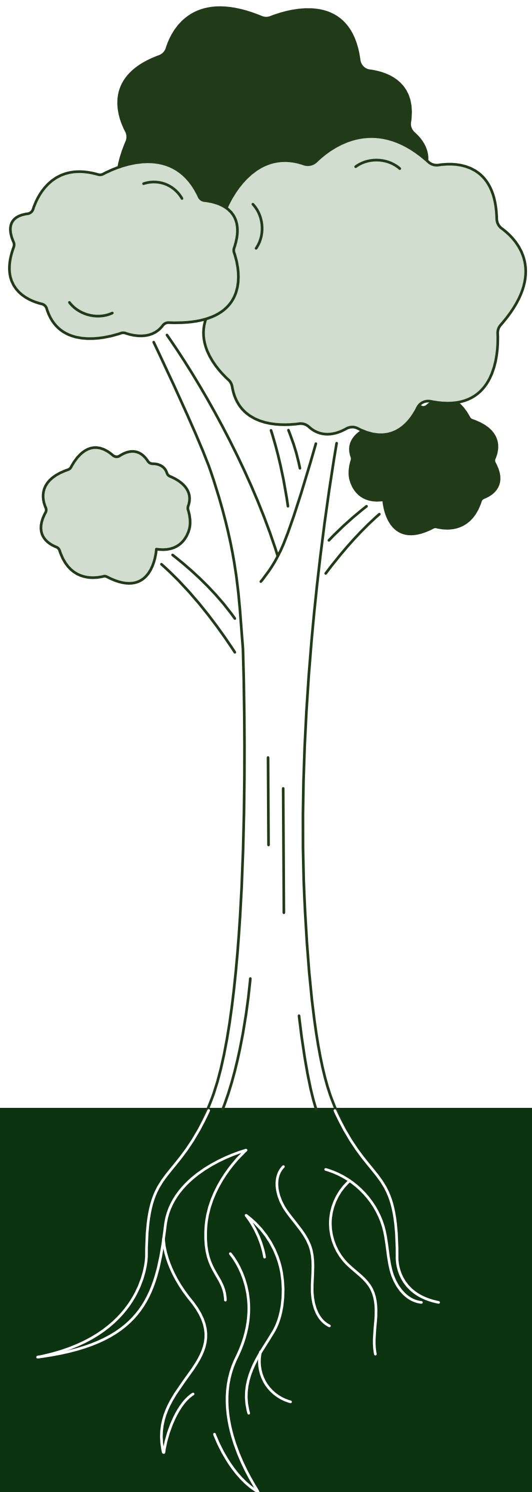

# 7

# Synthesis and conclusions

## 7.1 RESEARCH CONTEXT

Biomass supply for bioenergy production is recognized as a crucial strategy to meet mid-term and long-term climate targets, avoid adverse impacts from climate change and contribute to the transition to a more sustainable energy system[14]. In addition, the role of biomass is essential to reduce the dependency on fossil fuels and fossil-based products, and it can help to diversify markets and develop economies[52]. For example, fossil-based products, such as plastics and chemicals, will likely be substituted with bio-based alternatives[11]. Developing and consolidating a bio-based economy will require structural changes across all sectors of the economy and a rapid increase in volumes of biomass supply. It is expected that dedicated energy crops (non-food perennials) will become the main source of biomass for the bio-based economy, reaching up to 68 percent of the total biomass supply by 2050[26].

Despite that dedicated energy crops do not directly compete with agriculture commodity markets (food, feed and fiber), they can still trigger directly or indirectly LUCs. Many of the risks associated with biomass production for the bio-based economy are directly related to LUC[34]. Deploying large-scale bio-based value chains can pose direct competition with other land-based services and result in a wide range of LUC-related environmental and socioeconomic impacts. For example, although GHG emissions reduction is the main target of bio-based value chains, LUC-related carbon stock changes can lead to systems with a net increase in GHG emissions[34]. LUC is also recognized as one of the main drivers of biodiversity loss[38]. Furthermore, LUC can increase the risk of soil erosion, deteriorate ecosystems and intensify the pressure on water resources[32]. However, under certain conditions, LUC driven by dedicated energy crop production can also contribute to, e.g., carbon sequestration, land restoration, limit soil erosion and enhance rural development[295]. The potential of bio-based value chains to reduce GHG emissions and provide sustainability synergies relies on the interaction between land use for biomass production efficient logistics, conversion, and use (supply chain configuration)[48]. Therefore, the sustainability of bio-based value chains depends on context-specific conditions and the supply chain configuration.

Integrated environmental and socioeconomic assessments under context-specific conditions are required to identify holistically negative and positive impacts of bio-based value chains. Only then an accurate impression of the overall sustainability of bio-based value chains can be identified. Moreover, this process can help to identify synergies and trade-offs of bio-based value chains related to biomass production. This will support the development of sound strategies of sustainable biomass production and good governance of biomass use for current and novel bio-based value chains. It can





also lead to strategies to minimize negative environmental and socioeconomic impacts of bio-based value chains and enhance positive ones.

## 7.2 AIM AND RESEARCH QUESTIONS

This thesis aimed to determine the sustainability performance of existing and novel bio-based value chains under context-specific conditions. The following research questions were addressed:

I. How can environmental and socioeconomic impacts of feedstock production and the rest of the supply chain for bio-based value chains be assessed?

II. What are the environmental and socioeconomic impacts of bio-based value chains and how do these impacts influence sustainable biomass potentials?

III. What are the synergies and trade-offs between environmental and socioeconomic impacts of bio-based value chains?

**Table 7-1. Overview of the chapters of this thesis and the research questions that are addressed in them**

| Chapter | Topic | Research question | | |
|---|---|---|---|---|
| | | I | II | III |
| 2 | Spatially explicit assessment of environmental impacts of sugarcane expansion | X | X | X |
| 3 | GHG emission performance of biomass supply chains of multi-output biorefineries | X | X | |
| 4 | Biomass potentials and GHG emission performance of advanced biofuels | X | X | |
| 5 | Environmental and socio-economic impacts of using invasive alien plants for bioenergy purposes | X | X | X |
| 6 | Land use for bioenergy: synergies and trade-offs between Sustainable Development Goals | | | X |

## 7.3 SUMMARY OF THE RESULTS

**Chapter 2** provides a spatially explicit assessment of the LUC-related environmental impacts of sugar expansion in Sao Paulo state (Brazil) between 2004-2015. An integrated interannual approach considering location-specific biophysical characteristics and previous land use is applied to quantify the impact of sugar cane expansion on GHG emissions, water availability, biodiversity and soil erosion. LUC-related GHG emissions are addressed following the IPCC Guidelines for National Greenhouse Gas Inventories.

Impacts on biodiversity are assessed using the mean species abundance (MSA). The revised universal soil loss equation (RUSLE) is applied to assess impacts on soil erosion. Impacts on water quantity are quantified using a water balance approach. In addition, the four environmental impacts are integrated into an environmental performance index.

The results show a substantial spatial variation in environmental impacts. LUC-related GHG emissions vary between -8 to 6 t $CO_2$ $ha^{-1}$ $year^{-1}$. Generally, the conversion of grassland and annual crops to sugarcane results in a carbon sink given the larger share of sugarcane biomass stock, while the conversion of eucalyptus, forest and shrublands result in a carbon loss. The change in MSA varies from -0.2 to 0.7. When converted to sugarcane, the abundance of original species declines for all land use types, except annual crops. All land use categories, except annual crops, provide better conditions for species development than sugarcane. The change in soil loss from land conversion varies from -2 to 12 t $ha^{-1}$ $year^{-1}$. Except for annual crops, conversion to sugarcane increased soil loss for all land uses. The variation in soil loss is attributed to the differences in the cover effect against rain impact of the various land uses and the spatial heterogeneity terrain conditions (e.g., slope). The difference in water shortage varies between -180 and 300 mm $year^{-1}$. The high evapotranspiration rates of sugarcane in comparison to other land use types determine that the change of almost all land use categories, except for forest and eucalyptus, to sugarcane results in an increase in water shortage. The environmental performance index of sugarcane expansion varies between -2.5 and 2.5 at a scale from -4 to 4. There are no sugarcane expansion areas characterized by only positive impacts as there are always trade-offs between positive and negative impacts. The environmental impacts are primarily steered by land use transition type and local biophysical conditions. Generally, the direction of the impact is caused by the land use transition, and the magnitude of impacts is mainly due to the local biophysical conditions.

**Chapter 3** addresses the GHG performance of multiple supply-chain configurations using internationally-sourced lignocellulosic biomass (stem wood, forest residues, sawmill residues, and sugarcane bagasse) from the USA, the Baltic States (BS), and Brazil (BR) for the simultaneous production of lactide and ethanol in a biorefinery located in the Netherlands (NL). The results are compared with a biorefinery that uses locally cultivated sugar beets. An attributional life-cycle assessment (LCA) approach with a 'cradle-to-factory-gate' scope based on REDII accounting methods and region-specific characteristics is applied to determine the GHG emissions of the supply chains. GHG emission savings from the supply-chains are compared to the minimum GHG saving requirements in the REDII.





The GHG emissions of the biomass supply chain configurations for lactide vary between 692 g $CO_2$eq $kg_{lactide}^{-1}$ for sawmill residues pellets from the BS and 1002 g $CO_2$eq $kg_{lactide}^{-1}$ for sawmill chips from the USA. For ethanol, results vary between 15 g $CO_2$eq $MJ_{ethanol}^{-1}$ for sawmill residues pellets from the BS and 28 g $CO_2$eq $MJ_{ethanol}^{-1}$ for bagasse pellets from BR for ethanol. Upstream GHG emissions from feedstock supply to the conversion plant are low compared to the GHG emissions related to conversion. The use of woody biomass results in higher GHG emission reduction than sugarcane bagasse or sugar beets. Woody biomass provides a higher lignin content which is used to generate electricity and heat internally for the system and reduces the use of other fossil-based energy sources. Pelletization of woody biomass is preferred for long-distance overseas transport over wood chips due to the increased bulk density and improved handling characteristics further downstream in the supply chain. In terms of biomass supply regions, the Baltic states, sufficient biomass supply, relatively short transport distances between biomass sources and pelletization facilities, and the shorter overseas transport to the NL results in a lower GHG footprint than the other supply chains. The REDII GHG emission methods do not consider the challenges of burden allocation in multi-output systems. This affects the outcomes of the GHG emissions of ethanol as several by-products are allocated zero emissions as they are not energy carriers. This characteristic, in combination with the high energy content of ethanol (in comparison with other outputs such as lactide) dictates that a large share of the total generated GHG emissions are allocated to ethanol. Thus, under REDII criteria, only the production route using sugar beets from NL (separate streams for ethanol and lactide) can comply with REDII GHG savings criteria (65% by 2021). The challenge of burden allocation can discourage the production of renewable forms of energy in multi-output systems. The GHG savings from polylactide acid (a derivate of lactic acid) are high and vary depending on the choice of the fossil-based counterpart. These high savings are mainly caused to the negative emission credit from the embedded carbon in the materials. This case study shows that the explicit GHG performance depends mostly on specific feedstock characteristics and supply-chain configurations.

**Chapter 4** quantifies spatially explicit the availability of marginal land in the EU, its biomass potentials for eight different lignocellulosic energy crops, and the greenhouse gas (GHG) performance of different advanced biofuels for road and aviation produced from these crops. The available marginal land and biomass potentials of lignocellulosic energy crops on marginal lands in Europe are assessed for 2030, 2040, and 2050. The reference year for the advanced biofuel production pathways in the assessment is limited to 2030. Available land is mapped based on land marginality and REDII land-related sustainability criteria. Biomass potentials are assessed with a water-use-to-biomass-production equation while considering the available land, location-specific biophysical

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conditions and crop-specific phenological characteristics. The GHG balance of advanced biofuels from energy crops produced on marginal lands is assessed considering both land-related carbon stock changes and supply chain emissions with the carbon footprint approach from the REDII.

Available marginal land that meets REDII criteria is projected at 20.5–21 Mha 2030 and 2050, respectively. Due to biophysical limitations, not all available land is suitable for energy crop production. There is a strong variation in biomass potentials. In 2030, biomass potentials vary between 386 PJ $year^{-1}$ for willow to 1367 PJ $year^{-1}$ for miscanthus. The adaptability of each crop to different biophysical conditions and the potential yield that it can deliver under those specific circumstances drive the difference in biomass potentials. The maximum biomass potential of lignocellulosic energy crops (optimal crop choice for each available location) varies between 1951 PJ $year^{-1}$ in 2030 and 2265 PJ $year^{-1}$ in 2050. Biomass potentials are projected to increase over time as a result of LUC dynamics, variation in climate conditions and (assumed) 1% annual yield increase. The GHG emission performance of the advanced biofuel supply chains varies on average between -32 g $CO_2$eq $MJ_{fuel}^{-1}$ for poplar/willow diesel to 38 g $CO_2$eq $MJ_{fuel}^{-1}$ for reed canary grass renewable jet fuel (RJF). On average, all supply chains can comply with the 65% GHG emission savings requirement of the REDII. However, there are several locations where the production of ethanol and RFJ from herbaceous energy crops will not meet the REDII GHG savings criteria. Total supply chain GHG emissions are driven largely by LUC-related carbon stock changes and $N_2O$ field emissions. The large variability in GHG performance is strongly determined by the spatial heterogeneity, which dictates the type of feedstock produced under specific local biophysical conditions, the crop characteristics, and the best conversion pathway. Negative GHG emissions are related to increased soil and biomass carbon stocks compared to the land before conversion. When for each location the advanced biofuel supply chain with the best GHG performance is selected, 618 PJ $year^{-1}$ of advanced biofuels can be produced by 2030. Under REDII GHG emission criteria, slightly less (552 PJ $year^{-1}$) is feasible. The production of advanced biofuels from marginal land-sourced energy crops can rise as a valuable EU climate change mitigation strategy to reduce $CO_2$ emissions and support to meet EU biofuel demand.

**Chapter 5** addresses the environmental and socioeconomic impacts of using Invasive Alien Plants (IAPs) for electricity generation in South Africa or exporting this biomass source for electricity generation in the Netherlands. The assessment considers the social, economic and environmental context of the Eastern Cape province in South Africa. A special focus is given to the impacts of land use transitions when IAPs are cleared and the land is rehabilitated. Eight post-removal land-use scenarios were considered for the assessment based on biophysical and socioeconomic conditions of the study area.





Supply chain GHG emissions, including LUC-related GHG emissions, were calculated following an LCA approach using methods according to REDII. Impacts on water quantity were quantified using a water balance approach. The socioeconomic impacts are assessed for two performance indicators; supply chain costs and full-time jobs created.

Supply chain GHG emissions of electricity from IAPs, excluding LUC-related GHG emissions, are 31.5 g $CO_2$eq $MJ^{-1}$ for electricity generation in South Africa and 31.2 g $CO_2$eq $MJ^{-1}$ for electricity generation in the Netherlands. The main difference between the two supply chains is in transport mode to conversion plant. An additional 3.9 g $CO_2$eq $MJ^{-1}$ is accounted for if LUC emissions are included and the land is rehabilitated to its natural state. Both supply chains, including LUC-related GHG emissions, can comply with REDII 70% and 80% GHG savings requirements for 2021 and 2025. However, complying with REDII requirements relies strictly on restoring the land to its natural state, i.e., thicket or dense forest. The removal of IAPs results in water savings when considering any potential land-use transition, ranging between 1,263 mm $year^{-1}$ for annual cropland to 12 mm $year^{-1}$ for dense forest. The supply chain cost of pellets is 5,344 ZAR $t^{-1}$ (285 € $t^{-1}$) delivered at the power plant in South Africa and 2,535 ZAR $t^{-1}$ (159 € $t^{-1}$) delivered at Rotterdam port. The cost of pellets in SA is considerably higher, given the long-distance (1,000 km) between Port Elizabeth to the closest power facility. Annual direct full-time jobs (considering a 120,000 $t_{pellet}$ $year^{-1}$ output) generated from sourcing IAPs up to the conversion-factory-gate is 604 for South Africa and 525 for the Netherlands. The eradication of IAPs results in trade-offs between GHG emissions, water savings, and socioeconomic impacts. Generally, removing IAPs leads to net carbon losses but also leads to water savings and job creation. These benefits can also amplify other ecosystem services such as biodiversity conservation and socioeconomic development. Still, the supply chain's high cost poses a barrier to competing with the local electricity markets. Trade-offs of using IAPs for bioenergy need to be considered for the sustainable development of the biomass sector. However, other potential end uses such as heat for local households or fuel for cooking can improve the supply chain economic performance, increase GHG emissions savings and provide additional social benefits.



**Chapter 6** reviews with a pairwise comparison the synergies and trade-offs between GHG emission reduction (SDG 13) and other SDGs when land is used for dedicated energy crop production under context-specific conditions. In addition, the context-specific conditions (feedstock, previous land use, climate, soils and management) in which synergies and trade-offs arise are also specified.

The review showed a total of 427 pairwise observations that were classified as either synergies (170), trade-offs (176), or no effect (81). ´no effect´ observations represent when no positive or negative effects on SDGs besides GHG emission reduction were reported from using the land for growing dedicated energy crops. Most of the observations were found between GHG emission reduction (SDG 13) and SDGs 6 (Clean water and sanitation) with 45 synergies, 72 trade-offs and 32 no effects; and SDG 15 (Life on land) with 74 synergies, 56 trade-offs and 31 no effects. Fewer observations were found between GHG emission reduction (SDG 13) and SDG 2 (Zero hunger) with 1 synergy, 34 trade-offs and 9 no effects; other indicators of SDG 13 (Climate action) with 21 synergies, 4 trade-offs and 2 no effects; SDG 8 (Decent work and economic growth) with 6 synergies, 10 trade-offs and 2 no effects; and SDG 3 (Good health and well-being; 18 synergies, 0 trade-offs and 4 no effects. One synergy was observed related to SDG 14 (Life below water). A combination of context-specific conditions influences synergies and trade-offs. Most synergies were observed when perennial crops were produced on marginal land, previous arable land or pasture under a 'cool temperate moist' climate zone and 'high activity clay' soils. However, synergies arising from the production of perennials on land previously in use as cropland were also reported for other climate zones such as 'cool temperate dry' and 'warm temperate moist'. In addition, it is also shown that regardless of the feedstock type, allocating natural or semi-natural areas, including forests, to the production of dedicated energy crops generally leads to trade-offs.

To minimize trade-offs and provide synergies, the findings suggest that it is of paramount importance to consider context-specific conditions, with priority given in the following order: first land use transitions and second adequate feedstock types in line with (third) local biophysical conditions. In addition, it is shown that utilizing marginal land is a key strategy to avoid trade-offs with other land-based services such as food, feed, and fiber production (SDG 2). The magnitude of synergies and trade-offs between GHG emission reduction and other SDGs must be accounted for in decision making and from a supply chain perspective approach. Otherwise, it can lead to suboptimal implementation strategies and negative effects.

## 7.4 MAIN FINDINGS AND CONCLUSIONS

### RQ1: How can environmental and socioeconomic impacts of feedstock production and the rest of the supply chains for bio-based value chains be assessed?

In this thesis approaches have been developed to assess ex-ante and ex-post spatially explicit LUC dynamics, biomass potentials, and environmental and socioeconomic impacts of dedicated feedstock production and other steps of the supply chain. LUC dynamics and biomass potentials are considered as an essential pre-step to assess





sustainability impacts. In addition, there is a strong focus on LUC for bio-based value chains under different context-specific conditions.

The environmental and socioeconomic impacts of feedstock production for bio-based value chains depend on a wide range of key parameters. Mainly, previous land use (the type of land that is converted to energy crop productions), biophysical characteristics (e.g., climate and soil conditions), socioeconomic developments (e.g., food security), and crop management practices[44]. These parameters are spatially and temporarily heterogeneous. Land availability for dedicated biomass production is related to land use dynamics, which are driven by socioeconomic and political drivers that steer competing uses for land-based services (e.g., agriculture, conservation, etc.)[49]. These land use dynamics determine the amount, location and type of land use that can potentially be used for biomass production. All of these are essential factors to determine the LUC-related sustainability impacts of bio-based value chains. For example, in chapter 2, remote sensing data is used to map historic LUC dynamics by identifying the annual LUC to sugarcane. Estimating future land availability for bio-based value chains depends on the development of the other land use services as well on the sustainability criteria for bio-based value chains. Integrating sustainability criteria with land use projections is relevant to avoid biomass cultivation on land that leads to significant environmental and socioeconomic impacts and comply with policy requirements. In chapter 4, spatially explicit projections of LUC dynamics were used. These projections are based on an economic model that considers agricultural markets, demographic and macroeconomic trends[304]. These projections are combined with maps of marginal land (land with biophysical constraints for conventional agricultural use)[305] to identify the marginal land area that complies with REDII criteria. This integration is carried out as there is a lower risk for biomass under marginal conditions to compete with other land-based services[298]. LUC dynamics can also be assessed by determining plausible land use transition scenarios based on a region's relevant environmental and socioeconomic characteristics. For example, in chapter 5, successive land use scenarios after IAPs are eradicated for bioenergy purposes are defined while considering the region's land cover, agricultural practices and vegetation classes in line with local bioregions.

The sustainability of feedstock production depends on the productivity of the dedicated crop production system compared to the previous land use[31]. Biomass productivity of dedicated energy crops and the biomass productivity of other land uses are conditioned by local biophysical characteristics (e.g., soil type and climate), management practices and the characteristics of the plant species[44]. The productivity of energy crops and other land uses can be estimated based on data from sectorial representative entities (e.g., UNICA - Brazilian Sugarcane Industry Association) or international widely accepted guidelines such as the IPCC guidelines for LUC. To illustrate, IPCC provides climate-zone-specific default production values of maximum above-ground biomass and root-to-shoot ratios of different land use. However, these default values lack the local spatial variation in biomass productivity, vital for a more accurate assessment. Agroecological suitability maps or soil productivity maps of different land uses can be integrated with IPCC values to determine the spatial variation in biomass productivity as done in chapter 2. A correlation between the maximum suitability value and the maximum above-ground biomass production values can be assumed. Based on this correlation, biomass production values are derived from each suitability value. Energy crop yields can also be assessed with a water balance approach while considering land availability, location-specific biophysical conditions, and crop-specific phenological characteristics. These energy crop yields can be used to obtain regional biomass potentials, which are vital to estimating regional LUC-related sustainability impacts of bio-based value chains, as shown in chapter 4.

Sustainable bio-based value chains are expected to contribute to GHG emissions reduction and reduce our dependency on fossil-based energy[1]. However, under certain context-conditions, the expansion of dedicated feedstock production can potentially lead to negative environmental impacts and detrimental effects on the overall sustainability of bio-based value chains. The context-specific condition's role is paramount as the outcome of environmental impacts from feedstock production vary depending on context-specific conditions in which previous land use, local biophysical conditions and feedstock type play a decisive role[39]. Thus, the environmental impacts of feedstock production vary over space and time. In recent years, several studies have identified the main environmental areas of concern related to bio-based value chains, GHG emissions, water quantity and quality, biodiversity, and soil erosion, all of which are related to LUC's[34,39,103,458]. Ideally, environmental impacts are assessed with field measurements before and after land use conversion. For example, measuring the species richness of an area before and after land use is converted for dedicated feedstock production or measuring the soil loss relative to each land use before and after conversion. However, carrying out this type of assessment can be costly, take an extended period of time and can only be done ex-post. Therefore, in this thesis, the environmental impacts are estimated spatially explicit by adapting widely accepted methodologies in line with the scope of each case study based on the identified areas of concern. LUC dynamics and spatially explicit biomass productivity are used as input to estimate impacts from dedicated feedstock production.

Land conversion for dedicated feedstock production leads to carbon loss or gain in different carbon stocks. Generally, LUC-related GHG emissions cover carbon stock changes in biomass (above and below ground), dead organic matter, litter, harvested





wood products, and soils (SOC)[118]. LUC-related GHG emissions can be quantified by calculating the carbon content difference between the different carbon stocks before and after conversion to dedicated energy crops. In this thesis, LUC-related GHG emissions are quantified based on the REDII methods, built upon the stock difference approach from the IPCC [118]. However, REDII accounts only for carbon stocks present in biomass and soils. Biomass carbon stocks for dedicated energy crops and land use prior to conversion were derived from biomass productivity values assessed in previous steps with crop-specific factors of biomass to carbon fraction. SOC was estimated with default values for reference SOC levels of mineral soils, considering soil type, management, inputs and climate zones stratification. Land conversion for dedicated feedstock production also changes water availability mainly through changes in evapotranspiration[137]. The difference in water that dedicated feedstock crops take up and release in reference to the previous land use can lead to disturbances in water tables. Impacts on water are assessed with a water shortage approach based on spatial heterogeneous biophysical conditions (e.g., effective precipitation) and potential evapotranspiration rates dependent on the growing cycle of each vegetation type. The conversion of land for dedicated feedstock production also leads to changes in soil erosion. Soil erosion is mainly driven by wind and precipitation. However, precipitation-driven soil erosion is commonly the most accounted process[134]. Under similar biophysical conditions, soil erosion is determined mainly by the difference in cover effects against precipitation provided by the land uses before and after conversion. The impacts on soil erosion caused by precipitation are quantified with the widely applied revised universal soil loss equation (RUSLE) while accounting for precipitation, soil and terrain conditions, vegetation characteristics and support management practices to control soil erosion. The expansion of dedicated feedstock production leads to habitat changes and can affect biodiversity. LUC is recognized as the most important factor leading to biodiversity changes and depends largely on the initial land use condition, feedstock type and landscape configuration[472]. The impacts on biodiversity are assessed through the mean species abundance (MSA), in which a value between 0 and 1 is assigned to each land use based on its vegetation cover characteristics, with 1 referring to a pristine original ecosystem with species abundance not affected by human activities and 0 to the opposite. Assessing the environmental impacts of feedstock production in the identified areas of concern under context-specific conditions provides an integrated assessment and a step forward to develop more sustainable biomass systems. Integrated approaches are necessary to flag potential negative effects that arise from biomass production systems and develop strategies to reduce them.

The potential of bio-based value chains as a GHG emissions reduction strategy depends upon the interaction between land availability, sustainable biomass production (LUC), efficient logistics, conversion, and use [48,58]. Therefore, besides feedstock production related to LUCs, other supply chain steps can also affect the overall sustainability performance of bio-based value chains. Despite the different methods available to assess supply chains GHG emissions (e.g., PAS 2050), REDII methods should be applied for consistency purposes when bio-based value chains are expected to be rolled out into the European market. The GHG emissions from other supply chain steps can be based on default data and applied according to the biomass feedstock types' chemical and physical characteristics. However, this overlooks potential variation in GHG emissions arising from different spatial context-specific conditions such as GHG emissions transport given different distances, infrastructure and biomass availability. In this thesis, GHG emissions from biomass harvesting and collection are based on biomass availability, terrain conditions and local harvesting practices. A spatially explicit approach based on actual infrastructure, biomass availability, the capacity of the pre-treatments plants (e.g., pellet plants) and the conversion facilities are used to determine optimized delivery routes and transport distance from biomass production site to the conversion site. GHG emissions from maritime transport are assessed under actual international port-to-port routes and transport modes for biomass and biomaterials. GHG emissions from biomass pre-treatment and conversion to material and/or energy are assessed while considering the biomass physical and chemical properties, the conversion facilities' industrial features and their multi-input/output characteristics.

Bio-based value chains can also lead to different socioeconomic effects. It is suggested that bio-based value chains can significantly contribute to socioeconomic developments by expanding and diversifying (new) markets and employment generation[562]. This is more vital for developing nations that face urgent development goals, e.g., reducing poverty and inequality. In some countries, bio-based value chains have provided households with incomes, livelihood activities, and employment[563]. However, on some occasions, the costs of bio-based value chains can be high because of challenging logistics and result in supply chains that cannot compete with traditional energy/products market prices. Omitting to assess overall costs can lead to deploying (economically) non-competitive bio-based value chains that will eventually fail. This can have a detrimental effect on the employment generation along the supply chain. Thus, estimating bio-based value chain costs and employment generation are relevant parameters to understand bio-based supply chains' overall feasibility and socioeconomic effects. Employment generation and economic feasibility have been identified as two main categories for bio-based value chains socioeconomic impacts[564]. This thesis assessed the socioeconomic impacts under two indicators: supply chain costs and full-time jobs created. For costs, it is important to include capital and operational costs for each supply chain step based on the location context-specific conditions. Different sector-based wages are considered in line with





regional characteristics and job types. Investment costs for conversion facilities are required to be adjusted based on the widely applied Chemical Engineering Plant Cost Index (CEPCI). Full-time jobs are assessed based on a 40-hour workweek based on primary data of harvesting, processing and transport activities.

The approaches presented in this thesis are subject to several uncertainties and limitations. The interrelationship between LUC dynamics, biomass productivity and sustainability impacts and the wide range of input parameters can lead to uncertainty throughout the modeling chain. Spatial (input) data on land use can have quality issues related to accuracy and classification. For example, the applied remote sensing data from historical land use accuracy ranges between 70% and 90%[122]. Comparably, land use projections are characterized by a high level of aggregation in which some land uses are combined into one category. To illustrate in the LUISA data set, categories such as shrubland, transitional forest and forest are merged into one single category. In addition, the ability to reproduce historical trends in combination with diverse drivers does not necessarily assure that land use projections will be accurate [339]. Therefore, the future exact location and classification of specific land uses can be uncertain. The accuracy and aggregation level can affect identifying specific land uses and result in an under or overestimation of biomass productivity and subsequent sustainability impacts. More accurate spatial data could reduce the uncertainty level. Nevertheless, the most updated and accurate spatial data available was applied at the time of the assessment. Likewise, updated spatial data sets were used to disaggregate land use projections and minimize the overestimation of sustainability impacts.

Assessing biomass productivity combines a broad range of statistical and spatial input parameters, which are not necessarily consistent with each other and can be a source of uncertainty. For example, sugarcane yields are established assuming a linear correlation between average yield and average agroecological suitability value. However, it is uncertain whether this relation is linear. The relation between yield and suitability value could be exponential or quadratic. Accounting for such relations requires an additional understanding of physical and biological parameters and more accurate spatial and statistical data. Similarly, dedicated energy crop yields are estimated using agroecological suitability parameters in combination with a statistical approach that represents the relation between crop yield on the one hand and crop-specific phenological characteristics and climate conditions on the other. However, these are not calibrated with empirical results. In reality, yields are affected given less optimal conditions such as nutrient shortage[306]. More expertise is required to translate and understand the relationship between suitability parameters and yield.

Sustainability impacts could be better modeled when more accurate data is available instead of using default values. For example, the mean species richness indicator provides a score based on the land use category and it overlooks any species abundance distribution, information on threatened species, or connectivity. Similarly occurs for SOC, default data based on climate stratification and soil type, management and input is applied. Nevertheless, SOC is driven by complex physical and chemical processes such as hydrological processes that can be highly spatially heterogeneous[18]. In addition, evaluating the significance of sustainability impacts and translating them into actual impacts is challenging, given that a score on a specific indicator does not necessarily reflect the actual damage. Additional understanding of physical and biological parameters is required to translate sustainability scores into actual impacts.

The methods adapted, developed and applied in this thesis present different approaches that allow assessing the sustainability of bio-based value chains with a strong level of detail on land-related sustainability impacts. The methodological framework (Figure 7-1) shows the integrated steps required to evaluate the sustainability impacts of bio-based value chains, which can be applied both ways, ex-ante and ex-post. This framework is the result of integrating the various methods applied in different chapters. The connections strictly reflect the scope taken through different chapters, which does not necessarily signify that specific impacts are limited to a certain supply chain stage. First, land use dynamics are assessed to determine the amount, location and type of land use that can potentially be used for biomass production or the land already converted for dedicated biomass production. Second, biomass productivity is assessed for the available locations and the previous land uses while considering location-specific biophysical characteristics, management practices and dedicated energy crops characteristics. Third, both land use dynamics and biomass productivity are used as input to estimate sustainability impacts under relevant sustainability areas of concern. This approach allows combining spatial data on land use transitions, biophysical characteristics and other contexts-specific conditions to determine sustainability impacts of bio-based value chains at various regional or national scales. Therefore, the demonstrated approach is a significant step forward in understanding and assessing the sustainability impacts of feedstock production and other steps of the supply chains for bio-based value chains. It offers the capacity to identify areas integrally with negative or even positive impacts at various scales and trade-offs between sustainability impacts. This is highly relevant to identify and design policies to mitigate negative impacts under a holistic perspective.





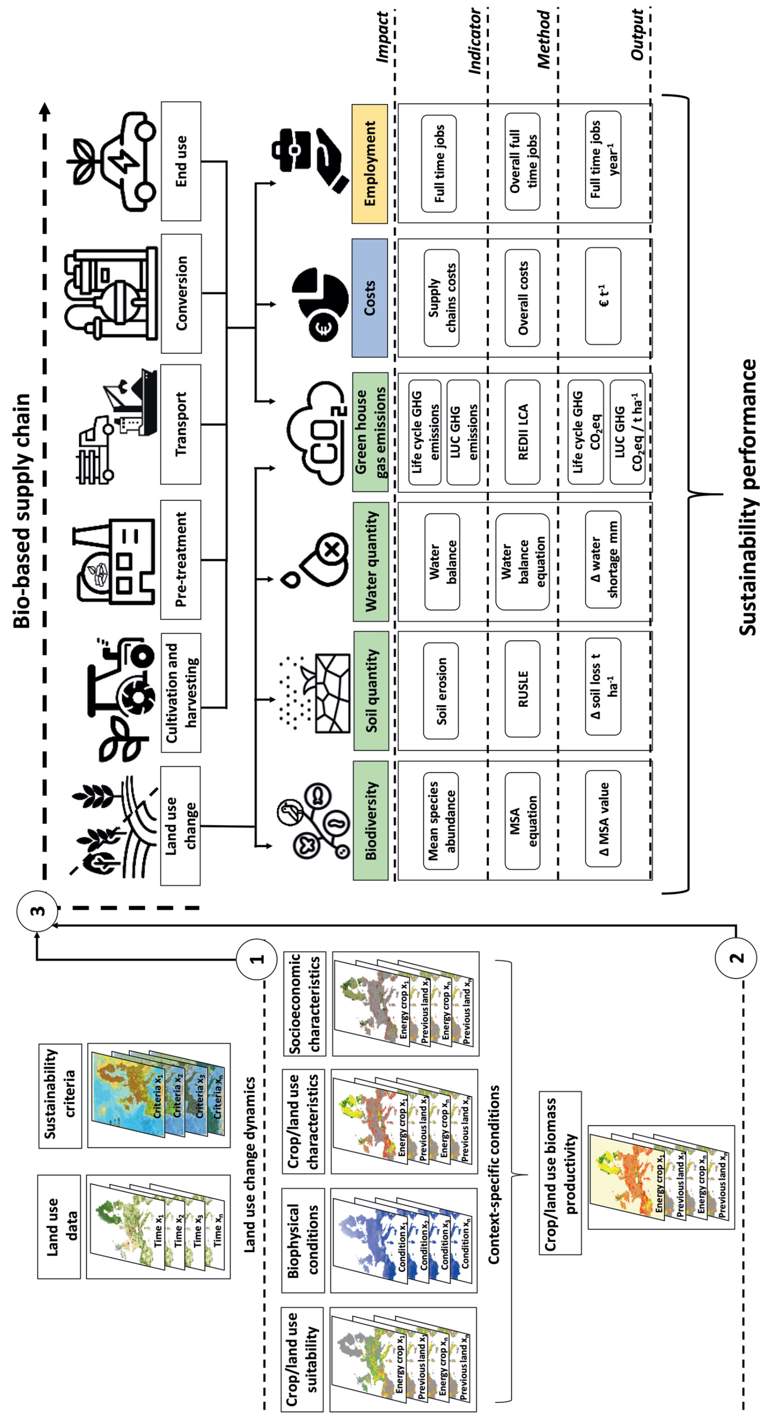


Figure 7-1 Methodical framework of the integrated approach between the main parameters included in the assessments.

**RQ2: What are the environmental and socioeconomic impacts of bio-based value chains and how do these impacts influence sustainable biomass potentials?**

The overall biomass supply potential for bio-based value chains covers forest biomass & residues, agricultural residues, dedicated energy crops and organic waste. In this thesis, the focus is given to dedicated energy crops. Generally, the amount of biomass potentially produced or dedicated for bio-based value chains depends upon different factors. Some of the most relevant are the developments in demand for other land-based services and the suitability of the land that becomes available for biomass production[18]. These factors are closely associated with context-specific conditions. In addition, biomass potentials are highly sensitive to the restrictions involved to avoid negative sustainability impacts of biomass production[224,292,330,567].

In the EU, land used for dedicated energy crops production must comply with REDII land-related sustainability criteria. For instance, lands with a high carbon stock are considered unavailable[15]. The sustainability of biomass production can be enhanced if the production is limited to marginal lands to reduce competition effects with other land-based services and, therefore, reduces the risk of ILUC and potentially provide restoration services[23,305]. In chapter 4, the biomass potential and marginal land availability under REDII criteria were assessed. In the EU, 7.9 to 8.9 Mha of marginal land are projected to be available in 2030 and 2050 that meet REDII criteria and is suitable for dedicated energy crop production (i.e. perennial grasses and SRC). By 2050, a biomass potential of 2265 PJ year$^{-1}$ can be reached, assuming optimal crop choice (i.e., crop with highest biomass yield) for each location. The biomass potentials are determined by the crop requirements and local biophysical conditions, which drive the potential yield that it can deliver under those circumstances. For example, in Scandinavia, the crop requirements of reed canary grass allow its production in locations with extreme conditions in which other dedicated energy crops cannot grow. However, only low yields can be obtained in these areas. Converting these low-yield areas, where low SOC accumulation occurs in reference to the previous land use, leads to high LUC-related GHG emissions. Even without accounting for other supply chain stages, LUC-related GHG emissions surpass the REDII GHG emissions savings criteria in such LUC scenarios. Therefore, lands in which the conversion results in high LUC-related GHG emissions are not suitable for dedicated energy crop production. Biomass potentials are also affected by constraints to avoid potential negative impacts on water and biodiversity. To illustrate, accounting for water-limited conditions to avoid irrigation for dedicated energy crops in the EU leads to a 44% reduction in biomass potential for every dedicated energy crop. Avoiding the conversion of shrubland to conserve biodiversity reduces biomass potentials by 47% in 2050.





The sustainability impacts of bio-based value chains are determined by the supply chain configurations (including the selection of feedstock) and, to a large extent, whether LUC is involved. The impact of LUC GHG emissions largely depends on the biomass productivity of each dedicated energy crop (or biomass type) according to local biophysical conditions and SOC accumulation in reference to the land prior to conversion. For example, using annual cropland for sugarcane production in Brazil leads to carbon accumulation (-8 - -4 t $CO_2$ ha$^{-1}$ year$^{-1}$), while using forest and eucalyptus areas results in high carbon losses (>6 t $CO_2$ ha$^{-1}$ year$^{-1}$). Average supply chainl GHG emissions from conventional sugarcane-derived ethanol without LUC are 28.6 g $CO_2$eq MJfuel$^{-1}$[314]. Total supply chain GHG emissions from sugarcane-derived ethanol, including LUC GHG emissions from forest conversion, can go as high as 100.7 g $CO_2$eq MJfuel$^{-1}$. As shown in chapter 5, in SA, carbon accumulates when IAPs (e.g., acacia) are replaced by dense forests (–2.9 t $CO_2$ ha$^{-1}$ year$^{-1}$), but there are high carbon losses when annual cropland replaces IAPs (7.6 t $CO_2$ ha$^{-1}$ year$^{-1}$). LUC-related GHG emissions can account for 50% of total GHG emissions when utilizing IAPs for electricity and land for annual crops after IAPs eradication. In the EU, up to 60% of GHG emissions of advanced biofuel pathways from dedicated energy crops produced in marginal lands are caused by LUC. For example, GHG emissions of ethanol from poplar are on average –29 g $CO_2$eq MJfuel$^{-1}$, while LUC-related GHG emissions are -53 g $CO_2$eq MJfuel$^{-1}$. However, these emissions can vary depending on location, type of LUC and conversion pathway. Therefore, LUC-related GHG emissions are highly relevant for supply GHG emissions when there Is a high carbon stock difference between land uses (before and after conversion). This difference can considerably steer bio-based value chains' negative or positive performance.

Usually, the direction of land-related sustainability impact is caused by the land use transition in which the type of feedstock and previous land use are decisive. This relation between land use conversion and feedstock type can considerably affect the overall sustainability of bio-based value chains. For example, In Sao Paulo state, sugarcane expansion declines mean species abundance for all land use types, except annual crops (–0.2 to 0.7). This relation between land use transition, feedstock type and land-related sustainability impact direction is also shown for other environmental impacts. In SA, IAPs are characterized by high evapotranspiration rates and a deep root system that allows them to access deeply stored groundwater, potentially leading to local water depletion. All land use transitions provide water savings, vital for a country with water scarcity issues. Removing IAPs could reduce water shortages between 12 and 12,630 m$^3$ ha$^{-1}$ * year$^{-1}$, depending on land use transition.

While the type of LUC and feedstock type often determines the direction of the environmental impact, i.e., whether it results in positive or negative environmental impact, the magnitude of the impacts is determined by the local biophysical conditions. Sugarcane expansion over annual cropland increases water shortage. Nevertheless, the impact magnitude of this land use transition shows a substantial spatial variation. For example, in Sao Paulo, the north is dry with less precipitation than in the rest of the state. In this region, the conversion of annual crops to sugarcane results in water shortage up to 300 mm year$^{-1}$. In the south of the state, the same land use transition results in a water shortage of 120 mm year$^{-1}$ because of higher precipitation rates. Similarly, sugarcane expansion increased soil loss compared to all land uses except annual cropland (-2 to 12 t ha$^{-1}$ year$^{-1}$), given that sugarcane provides less cover against the impact of rain than other land uses. In locations with higher precipitation rates, the increase in soil loss is higher.

The impact of other supply chain stages, such as transport and logistics, becomes more relevant for GHG emissions when land uses before and after conversion have a similar carbon stock or when no LUC processes are involved. The supply chain configuration of bio-based value chains can highly affect the overall GHG emissions. For example, supply chain GHG emissions of electricity production in SA from IAPs pellets are on average 31.5 g $CO_2$eq MJ$^{-1}$ with an additional 3.9 g $CO_2$eq MJ$^{-1}$ accounted for if the land is rehabilitated to its natural state (similar carbon stock between land uses). Transport-related GHG emissions are 12 g $CO_2$eq MJ$^{-1}$, accounting for 34% of total emissions. Transporting pellets from supply location to conversion site is logistically challenging given the long distance between locations.

Other chapters also show the effect of supply chain configuration on the overall GHG emissions of bio-based value chains. In chapter 3, for the biorefinery (simultaneous production of ethanol and lactide) in the Netherlands with international sourced lignocellulosic biomass (e.g., forestry residues and sugarcane bagasse) and locally sources sugar beets, it is shown that processes upstream from the conversion stage have a relatively small impact on the total GHG emissions. To illustrate, total GHG emissions of lactide production from forest residues pellets with conversion in the Netherlands are 1244 g $CO_2$eq kg$^{-1}$, of which 88% correspond to conversion emission related to process energy demand (use of natural gas) and upstream emissions from chemicals use. Still, net GHG emissions of lactide are negative when considering the embedded carbon in the material.

Different context-specific socioeconomic impacts also determine the sustainability of bio-based value chains. These impacts can determine the feasibility of deploying bio-based value chains even when delivering positive environmental effects. The case study investigated in chapter 5 shows positive environmental effects from using IAPs for electricity. In addition, the investigated bio-based value chain generates a total of





600 direct full-time jobs annually from sourcing IAPs up to the conversion-factory-gate, potentially increasing up to 1000 considering land restoration jobs. Nevertheless, the overall cost of delivering the biomass at the conversion facility (284.7 € t$^{-1}$) is not competitive. Mobilizing biomass by road transport can be costly and inefficient [180]. The transport of pellets accounts for approximately 52% of the total supply chain costs. The high logistics costs make it unfeasible for this bio-based value chain to compete with traditional electricity prices. Thus, deploying such a supply chain require incentives or public-private partnerships to share and reduce production costs or investigate other end-uses with less logistics costs. The overall cost will determine whether bio-based supply chains are (economically) viable.

In conclusion, the results in this thesis show that biomass potentials, environmental and socioeconomic impacts of bio-based value chains depend on context-specific conditions and are determined largely by LUC. Furthermore, sustainable biomass potentials are affected by sustainability constraints. Biomass potentials and the number of bio-based value chains deployed into the market will be determined by competing land uses, biomass production capacity (spatially heterogeneous) and sustainability criteria. Therefore, land can be the constraining factor for sustainable biomass production and deploying large-scale bio-based value chains to meet sustainability targets. However, land use for biomass production can only be developed while considering competing uses, demand for other services and products and sustainability impacts. It is possible to conclude bio-based value chains show positive and negative environmental and socioeconomic impacts, and those depend on the context conditions of each case study. In addition, it is shown that integrated sustainability assessments of bio-based value chains are vital to develop sound land use strategies and deployment feasibility (Figure 7-1). Especially, when combining integrated assessments with the most relevant context-specific conditions, conversion routes and supply chains configurations.



**RQ3: What are the synergies and trade-offs between environmental and socioeconomic impacts of bio-based value chains?**

Reducing GHG emissions to meet climate change targets is a key driver the development of bio-based value chains. Deploying bio-based value chains to reduce GHG emissions can result in other environmental and socioeconomic impacts, potentially deteriorating or improving the overall sustainability of these systems. This thesis identifies important trade-offs and synergies effects on the SDGs that arise from bio-based value chains, focusing on feedstock production and use. Synergies and trade-offs are shown to vary from place to place depending on the context-specific conditions.

Using land for dedicated energy crops for GHG emission reduction (SDG 13) shows both synergies and trade-offs with decent work and economic growth (SDG 8). Trade-offs are related to economic viability and synergies with employment. Studies reported a negative effect on farmers' revenue when they shifted from 1st to 2nd generation crops as bioenergy feedstock[47,502,507,531]. Unfavorable economic conditions, such as low biomass prices and high logistical costs, resulted in lower revenues for dedicated energy crops than food crops [502]. High logistics costs can also result in trade-offs between GHG emissions and economic growth along the entire supply chain. In SA, high logistics costs limit electricity production from IAPs, despite obtaining GHG emissions reduction. This bio-based value chain cannot compete with electricity market prices. However, subsidies for bioenergy can promote increasing land use for dedicated energy crops and positively affect income and revenues (SDG 8) [532]. Several synergies are reported between GHG emission reduction and job creation (SDG 8). Using land for dedicated energy crops for bioenergy can create jobs for local communities [36,42,497]. For example, more than 600 full-time annual jobs are created in SA when using IAPs for bioenergy purposes, of which 351 are related to harvesting activities. Deploying bio-based value chains will generally lead to employment generation and wealth creation[568]. There are considerably more trade-offs than synergies between the impacts of land use for dedicated energy crops on GHG emission reduction (SDG 13) and zero hunger (SDG 2). Literature has highlighted that food security risks increase when land dedicated to food production is limited by the production of dedicated energy crops [46,495,510,513,518,528–530]. Land competition can negatively affect agricultural production and food supply, and lead to increases in food prices [37,511].

Previous land use and feedstock type are more impactful in determining synergies and trade-offs than other context-specific conditions such as climatic zone and soil type. Most synergies are shown when previously arable land and marginal land are used to produce dedicated energy crops and a feedstock change from annual food crops (e.g., corn) to perennial energy crops (e.g., perennial grasses). This land use transition and feedstock change is shown to reduce GHG emissions (SDG 13) and simultaneously improve water quality (SDG 6.3); help to strengthen resilience and adaptive capacity to climate-related hazards and natural disasters (SDG 13.1); increase biodiversity (SDG 15.1); and improve soil quality (SGD 15.3). In Brazil, sugarcane expansion over annual crops increases carbon accumulation and provides better habitat for species, increasing the species abundance (SDG 15.1). It is also reported that (in several locations) perennial crops provide better habitat conditions than annual crops, which increase the abundance of species such as birds, mammals, and pollinators [43,501,504]. The conversion of annual crops to sugarcane also provides a better cover against precipitation, decreasing soil loss (SDG 15.3). Growing perennials accumulates organic matter in deeper soil layers, providing better conditions to increase sediment retention and decreasing soil erosion (SDG 15.3)[499].





For example, producing miscanthus and poplar on marginal lands reduces sediment loss and the overall risk of soil erosion [519]. Growing perennials on land previously in use for (intensively) managed corn and soybean decreased nutrient loading in the watershed (SDG 6.3) and helped to prevent downstream hypoxia episodes, benefitting aquatic life (14.1). Furthermore, it can increase the resilience to more extreme climate conditions [534] and provide mitigation benefits on flood events [496] (SDG 13.1).

In general, the trade-offs between GHG emissions reduction (SDG 13) and clean water and sanitation are mostly related to water use and water use efficiency (SDG 6.4). Water demand for dedicated energy crops can increase water withdrawal through irrigation and increase water scarcity. In Brazil, the expansion of sugarcane on annual crops, grasslands and shrublands shows to increase water shortage. However, carefully choosing locations to produce dedicated energy crops for bioenergy and choosing feedstocks adapted to local conditions can minimize pressures on water availability [464]. On the other hand, in SA, it is shown that eradication of IAPs leads to water savings, independent of the subsequent land use. However, this process follows a different order, as the land use before conversion is the one dedicated to bioenergy purposes. Trade-offs are also shown consistently for other types of previous land uses. For example, dedicating natural or semi-natural areas, including forests, to the production of dedicated energy crops negatively affects biodiversity conservation (SDG 15.1) and sustainable use of forest (SDG 15.2)[511,515,536].

The results show that synergies and trade-offs depend on context-specific conditions. It can be concluded that to minimize trade-offs it is of paramount importance to consider context-specific conditions, with priority given in the order of first land use transitions and second feedstock types, while both parameters need to be considered in line with local biophysical conditions. Thus, other context-specific conditions such as agricultural management practices, location-specific biophysical characteristics, and social and economic aspects are also relevant but appear to be less decisive. Synergies and trade-offs also depend on each study's applied indicators and scope. For example, in chapter 2, only environmental indicators were included and thus, synergies and trade-offs are limited to the environmental dimension. However, sugarcane production in Brazil has several positive effects that show synergistic relations with socioeconomic aspects. It has led to job creation, revenue for farmers, reduced the dependency on fossil fuels and promoted the country's economy[100]. Objectively identifying synergies and trade-offs is a difficult task as positive or negative effects of biomass production for bio-based value chains can be perceived through different lenses. To illustrate, utilizing IAPs show synergies between GHG emissions savings, water availability and job creation. However, these synergies are established while establishing a bio-based value chain for IAPs, while local community stakeholders may have different priorities and perspectives. For local communities, utilizing the wood directly for building houses or for fuel conversion could prove better synergies between GHG emission reduction and social aspects.

## 7.5 RECOMMENDATIONS FOR FUTURE RESEARCH

The methods adapted, developed and applied in this thesis present different approaches to assess the sustainability of bio-based value chains with a high level of detail on land-related sustainability impacts. The results of this thesis provide useful information about biomass potentials and the environmental and socioeconomic impacts of bio-based value chains. In addition, it helps to understand environmental and socioeconomic synergies and trade-offs of bio-based value chains. In all case studies, context-specific conditions were considered with a strong focus on LUC processes. There are limitations to the present research in several key topics that are recommended to be covered in future research.

- Biomass implementation potentials need to be assessed while considering economic and non-economic barriers such as competition with other domestic and imported biomass sources, infrastructure, and cash flow for poor farmers[332,333]. The supply of large biomass volumes from remote regions to the conversion facilities with inadequate infrastructure can be costly and inefficient. In addition, the biomass production cost and the competitiveness between different crop types will vary and influence farmers' crop selection. Typically, crop yields on marginal land are low, and thus, feedstock costs are relatively high. Considering economic factors could lead to lower biomass potentials. These factors should be integrated into future modelling exercises of bio-based value chains.

- The quantification of the environmental and socioeconomic impacts of bio-based value chains is based on modeling work that is inherently uncertain due to the assumptions made and input data. For example, choices on allocation procedures highly affect the results on GHG emissions attributed to biobased products. Furthermore, not all impacts are suitable/effective to assess at the product/supply chain level. Better quality data with a higher resolution could result in a more accurate assessment and reduce the loss of details and potential under/overestimation of LUC-related environmental impacts. Especially data related to current and future land use, projections on climatic variables, market trends, soil characteristics and agroecological suitability maps. However, the most recent data available was used at the time of the assessment. In addition, some input data was assumed to be constant over time, while in practice, they are expected to change over time. For instance, the location of marginal lands in chapter 4 are assumed to remain constant, while some parameters related to climate conditions that define land marginality vary over time. Future research should better address temporal variability to improve estimations of potential environmental and socio-economic impacts.

- The results should be interpreted with care given the uncertainties and shortcomings in bio-based value chains' environmental and socioeconomic impacts assessments. For example, LUC-related GHG emissions are estimated based on modeling exercises assuming that carbon pools reach equilibrium. However, changes in carbon stocks are driven by complex physical and chemical processes that can be highly heterogeneous, providing uncertainty about whether and how fast carbon pools can actually reach such equilibrium. In addition, land-related disturbances could also affect achieving equilibrium stages. Thus, net GHG emissions can differ if such carbon equilibrium states are not reached and provide a wide range of GHG emissions performance for bio-based value chains. This uncertainty is also present with other applied indicators related to water and biodiversity. To illustrate, the water shortage indicator neglects important parameters such as soil characteristics and water flow dynamics. The mean species abundance indicator omits information on species distribution, threatened species, or connectivity. These parameters (e.g., carbon equilibrium stages, water flow dynamics and species distribution) can be relevant to determine environmental impacts from biomass production and use and could be considered in future bio-based value chain sustainability assessments.

- The translation of environmental indicator values into actual impacts is challenging and requires further research. For example, more validation is required to understand how a relative score in the mean species abundance index translates into actual damage or improvement for biodiversity. Additional validation is also required to translate relative land suitability scores into actual yield, especially when considering the cause-effect relationship between suitability parameters and yields. This relationship could be linear, quadratic, or exponential. Validation processes require an additional understanding of physical and biological relationships. Both environmental impacts and yields should be calibrated with empirical results for a better assessment, given that, in reality, for instance, yields are often lower as a result of less optimal conditions such as nutrient shortage.

- There is a wide range of conversion options for bio-based value chains. This thesis focused on specific systems designs for conversion pathways for conventional and advanced biofuels, biobased materials, and electricity. In this thesis, different system configurations have been assessed. However, there are also other system designs for the same end product, which may perform differently. For example, more advanced pre-processing close to the sourcing area could substantially reduce transport GHG emissions. In addition, optimisation of logistics for residues and crops from marginal lands can minimize costs and GHG emissions. Furthermore, other end uses and technological progress for specific pathways could improve environmental and socioeconomic performance. For example, different end uses for IAPs such as pellets for the residential market could offer higher social and GHG benefits than electricity generation.

- Determining synergies and trade-offs between environmental and socioeconomic impacts of bio-based value chains requires a full supply chain perspective considering the entire energy system. This perspective is required as the sustainability performance of bio-based value chains depends not just land management but also on supply chain designs and end-uses. This thesis assessed environmental and socioeconomic impacts as well as synergies and trade-offs with the main focus on direct LUC. Future research should also include ILUC and the optimizations of logistics and conversion processes and end-uses for a holistic sustainability assessment. Furthermore, holistically assessing synergies and trade-offs can still be improved, especially under regionally specific settings. For example, exploring how to incorporate local stakeholders' preferences and their consequences for synergies and trade-offs between environmental and socioeconomic impacts.

## 7.6 POLICY RECOMMENDATIONS

- Integrated environmental and socioeconomic assessments are required to be implemented in the decision support on the development of bio-based value chains. This is important given that assessing a single sustainability indicator can fall short in providing a holistic picture of the sustainability performance of bio-based value chains. Thus, omitting an integrated perspective can lead to the development of bio-based value chains with positive impacts on one sustainability objective and detrimental effects on others. In addition, integrated sustainability assessments can promote developing strategies to mitigate negative sustainability effects of biobased value chains, enhance positive ones, and avoid leading to cross-sectorial detrimental sustainability effects.

- Policies on the sustainability of biobased value chains should consider context-specific conditions (especially LUC, feedstock types and biophysical characteristics) conditions and the other way around. In addition, bio-based value chains are part of bigger socioeconomic systems with different demands for land-based services that should be acknowledged. Policy on bio-based value chains needs to be aligned with policy on other land-based services such as agriculture and nature protection. Failing to align policies and to include context-specific conditions can lead to inadequate and counterproductive policy development.





- The methodological framework presented in this study (Figure 7-1) can provide valuable insights for policymaking and voluntary certification schemes. It can help develop pathways of sustainable land use for biomass production and market roll-out of bio-based value chains. This framework allows assessing the sustainability of biobased value chains from biomass production up to end-use while considering land availability (competition between land-based services), biomass productivity, and environmental and socioeconomic impacts, based on context-specific conditions. It also allows identifying synergies and trade-offs between environmental and socioeconomic impacts and identifying risks that could pose sustainability barriers to roll out bio-based value chains. Thus, the framework could also help to certify the sustainability of bio-based value chains and their contribution to SDGs. Furthermore, the integrated characteristic of the framework allows counterbalancing the potential inconsistencies present in some generic policies, allowing the deployment of high-performance bio-based value chains and avoiding the poor-performing ones. To illustrate, potentially, all marginal land locations that meet REDII land-related sustainability criteria can carry out biomass production. However, not all biomass production locations are suitable for bio-based value chains as the REDII GHG emissions savings threshold is not achieved due to LUC-related GHG emissions. Thus, all locations could be dedicated to sustainable biomass production and potentially restoring ecosystems. Still, only the high-performance areas in terms of LUC-related GHG emissions should be selected for bio-based value chains and the poor performance ones for other end-uses or sustainable biomass grow (without harvesting) as a restoration process. Developing this approach is an attempt to provide an integrated framework that could be generally applied while considering the most relevant steps in bio-based value chains with a strong focus on LUC. However, it remains challenging to balance generic approaches with location-specific conditions and regional land use impacts (e.g., changes in biodiversity) with product-specific ones.

- The use of marginal lands for biomass production and supply bio-based value chains could be further acknowledged in policy as a relevant strategy to reduce GHG emissions, support to meet the demand for bio-based products, provide synergies and reduce trade-offs with other SDGs. Especially through the production of perennial crops, which are more adapted to marginal conditions and are less likely to compete with other land-based services and thus, generate fewer displacement effects. In addition, perennial crop production in marginal land can have advantageous effects on biodiversity and contribute to land restoration. However, marginal lands are generally located in remote areas of difficult access. In order to develop these areas, more investment is required in infrastructure and the entire supply chain. This investment can provide adequate conditions to develop new markets and promote sustainable biomass production as a potential economic alternative for farmers.

- Sustainable biomass production should contribute to multiple SDGs and policies should target this multi-purpose character by internalizing the value of positive externalities of biomass production for bio-based value chains. For example, internalizing the value of carbon sequestration, ecosystem restoration, biodiversity improvement, increasing soil quality and improving water quality. Valuing positive externalities can promote sustainable biomass systems and thus become more economically viable. In addition, this value internalization process can provide an economic basis to develop mitigation strategies for negative externalities and target key bottlenecks such as farmers' income.

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Nederlandse samenvatting

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## NEDERLANDSE SAMENVATTING

### Context van het onderzoek

De voorziening van biomassa voor de productie van bio-energie wordt erkend als een cruciale strategie om de klimaatdoelstellingen op middellange en lange termijn te halen, negatieve effecten van klimaatverandering te voorkomen en bij te dragen aan de overgang naar een duurzamer energiesysteem. Daarnaast is de rol van biomassa essentieel om de afhankelijkheid van fossiele brandstoffen en fossiele producten te verminderen. Ook kan biomassa helpen om markten te diversifiëren en economieën te ontwikkelen. Zo zullen fossiele producten als kunststoffen en chemicaliën waarschijnlijk worden vervangen door biogebaseerde alternatieven. Het ontwikkelen en consolideren van een biogebaseerde economie vereist structurele veranderingen in alle economische sectoren en een snelle toename van het aanbod van biomassa. De verwachting is dat specifieke energiegewassen (meerjarige gewassen die niet tot voedsel dienen) de belangrijkste bron van biomassa voor de biogebaseerde economie zullen worden, tot 68 procent van de totale biomassavoorziening in 2050.

Hoewel specifieke energiegewassen niet direct concurreren met landbouwgrondstoffen (voedsel, voer en vezels), kunnen ze wel direct of indirect landgebruiksverandering veroorzaken. Veel van de risico's van biomassaproductie (voor de biogebaseerde economie) houden rechtstreeks verband met landgebruiksverandering. Het inzetten van grootschalige biogebaseerde waardeketens kan leiden tot directe concurrentie met andere grondgebonden diensten en resulteren in een breed scala aan milieu- en sociaaleconomische effecten gerelateerd aan landgebruiksverandering. Hoewel bijvoorbeeld vermindering van de broeikasgasemissies het belangrijkste doel is van biogebaseerde waardeketens, kunnen aan landgebruiksverandering gerelateerde veranderingen in de koolstofvoorraad leiden tot systemen met een netto toename van de broeikasgasemissies. Landgebruiksverandering wordt ook erkend als een van de belangrijkste oorzaken van biodiversiteitsverlies. Bovendien kan landgebruiksverandering het risico op bodemerosie vergroten, ecosystemen verslechteren en de druk op watervoorraden vergroten. Onder bepaalde omstandigheden kan landgebruiksverandering, aangedreven door de productie van specifieke energiegewassen, echter ook bijdragen aan bijvoorbeeld koolstofvastlegging, landherstel, beperking van bodemerosie beperken en verbetering van plattelandsontwikkeling. Het potentieel van biogebaseerde waardeketens om de uitstoot van broeikasgassen te verminderen en duurzaamheidssynergieën te bieden is afhankelijk van de interactie tussen landgebruik voor efficiënte logistiek, conversie en gebruik van biomassa (toeleveringsketenconfiguratie). Daarom hangt de duurzaamheid van biogebaseerde waardeketens af van contextspecifieke omstandigheden en de configuratie van de toeleveringsketen.





Geïntegreerde milieu- en sociaaleconomische beoordelingen onder contextspecifieke omstandigheden zijn nodig om holistisch negatieve en positieve effecten van biogebaseerde waardeketens te identificeren. Alleen dan kan een juist beeld worden gegeven van de algehele duurzaamheid van biogebaseerde waardeketens. Bovendien kan dit proces helpen bij het identificeren van synergieën en uitruilen (*trade-offs*) in biogebaseerde waardeketens die verband houden met de productie van biomassa. Dit zal de ontwikkeling ondersteunen van degelijke strategieën voor duurzame biomassaproductie en goed bestuur van het gebruik van biomassa voor huidige en nieuwe biogebaseerde waardeketens. Het kan ook leiden tot strategieën om de negatieve milieu- en sociaaleconomische effecten van biogebaseerde waardeketens te minimaliseren en positieve effecten te versterken.

### Doel en onderzoeksvragen

Dit proefschrift heeft zich gericht op het bepalen van de duurzaamheidsprestaties van bestaande en nieuwe biogebaseerde waardeketens onder contextspecifieke omstandigheden. De volgende onderzoeksvragen kwamen aan bod:

I. Hoe kunnen de milieu- en sociaaleconomische effecten van de productie van grondstoffen en de rest van de toeleveringsketen voor biogebaseerde waardeketens worden beoordeeld?

II. Wat zijn de milieu- en sociaaleconomische effecten van biogebaseerde waardeketens en hoe beïnvloeden deze effecten het duurzame biomassapotentieel?

III. III. Wat zijn de synergieën en afwegingen tussen milieu- en sociaaleconomische effecten van biogebaseerde waardeketens?

**Table 8-1. Overzicht van de hoofdstukken van dit proefschrift en de onderzoeksvragen die aan bod komem**

| Hoofdstuk | Onderwerp | Onderzoeksvraag | | |
|---|---|---|---|---|
| | | I | II | III |
| 2 | Ruimtelijk expliciete beoordeling van de milieueffecten van de uitbreiding van suikerriet | X | X | X |
| 3 | Broeikasgasemissieprestaties van biomassatoeleveringsketens van bioraffinaderijen met meerdere outputs | X | X | |
| 4 | Biomassapotentieel en broeikasgasemissieprestaties van geavanceerde biobrandstoffen | X | X | |
| 5 | Milieu- en sociaaleconomische effecten van het gebruik van invasieve uitheemse planten voor bio-energiedoeleinden | X | X | X |
| 6 | Landgebruik voor bio-energie: synergieën en uitruilen tussen Duurzame Ontwikkelingsdoelen (SDG's) | | | X |

## Samenvatting van de resultaten

**Hoofdstuk 2** geeft een ruimtelijk expliciete beoordeling van de landgebruiksveranderingsgerelateerde milieueffecten van suikerrietuitbreiding in de staat Sao Paulo (Brazilië) tussen 2004-2015. Een geïntegreerde, meerjarige benadering waarbij rekening wordt gehouden met locatiespecifieke biofysische kenmerken en eerder landgebruik wordt toegepast om de impact van de uitbreiding van suikerriet op de uitstoot van broeikasgassen, de beschikbaarheid van water, de biodiversiteit en bodemerosie te kwantificeren. Landgebruiksveranderingsgerelateerde broeikasgasemissies worden aangepakt volgens de IPCC-richtlijnen voor nationale broeikasgasinventarissen. Effecten op de biodiversiteit worden beoordeeld met behulp van de gemiddelde soortenabundantie (MSA). De herziene universele bodemverliesvergelijking (*Revised Universal Soil Loss Equation*, RUSLE) wordt toegepast om de effecten op bodemerosie te beoordelen. Effecten op de waterkwantiteit worden gekwantificeerd met behulp van een waterbalansbenadering. Daarnaast zijn de vier milieu-impacts geïntegreerd in een milieuprestatie-index.

**Hoofdstuk 3** behandelt de broeikasgasemissieprestaties van meerdere toeleveringsketenconfiguraties die gebruik maken van internationaal geproduceerde lignocellulosische biomassa (stamhout, bosresiduen, zagerijresiduen en suikerrietbagasse) uit de VS, de Baltische staten (BS) en Brazilië (BR) voor de gelijktijdige productie van lactide en ethanol in een bioraffinaderij in Nederland (NL). De resultaten worden vergeleken met een bioraffinaderij die lokaal geteelde suikerbieten gebruikt. Een attributionele levenscyclusanalysebenadering (LCA-benadering) met een systeemafbakening van "wieg tot fabriekspoort" op basis van REDII-methoden en regiospecifieke kenmerken wordt toegepast om de broeikasgasemissies van de toeleveringsketens te bepalen. Broeikasgasemissiebesparingen van de toeleveringsketens worden vergeleken met de minimale broeikasgasbesparingseisen in de REDII.





**Hoofdstuk 4** kwantificeert ruimtelijk expliciet de beschikbaarheid van marginaal land in de EU, het biomassapotentieel voor acht verschillende lignocellulose-energiegewassen en de broeikasgasprestaties van verschillende geavanceerde biobrandstoffen voor wegtransport en de luchtvaart die uit deze gewassen worden geproduceerd. Het beschikbare marginale land- en biomassapotentieel van lignocellulose-energiegewassen op marginale gronden in Europa worden beoordeeld voor 2030, 2040 en 2050. Het referentiejaar voor de geavanceerde productieroutes voor biobrandstoffen in de beoordeling is beperkt tot 2030. Het beschikbare land wordt in kaart gebracht op basis van landmarginaliteit en REDII landgerelateerde duurzaamheidscriteria. Biomassapotentialen worden beoordeeld met een watergebruik-naar-biomassa-productievergelijking, rekening houdend met het beschikbare land, locatiespecifieke biofysische omstandigheden en gewasspecifieke fenologische kenmerken. De broeikasgasbalans van geavanceerde biobrandstoffen uit energiegewassen die op marginale gronden worden geproduceerd, wordt beoordeeld rekening houdend met zowel landgerelateerde koolstofvoorraadveranderingen als emissies in de toeleveringsketen met de koolstofvoetafdrukbenadering van de REDII.

**Hoofdstuk 5** behandelt de milieu- en sociaaleconomische effecten van het gebruik van invasieve uitheemse plantensoorten voor elektriciteitsopwekking in Zuid-Afrika of het exporteren van deze biomassabron voor elektriciteitsopwekking in Nederland. De beoordeling houdt rekening met de sociale, economische en ecologische context van de provincie Oost-Kaap in Zuid-Afrika. Er wordt speciale aandacht besteed aan de effecten van landgebruiktransities wanneer invasieve uitheemse plantensoorten worden verwijderd en het land wordt hersteld. Acht scenario's voor landgebruik na verwijdering werden overwogen voor de beoordeling op basis van biofysische en sociaaleconomische omstandigheden van het studiegebied. De broeikasgasemissies van de toeleveringsketen, inclusief landgebruiksveranderingsgerelateerde broeikasgasemissies, werden berekend volgens een LCA-benadering met behulp van methoden volgens REDII. De effecten op de waterkwantiteit werden gekwantificeerd met behulp van een waterbalansbenadering. De sociaaleconomische effecten worden beoordeeld voor twee prestatie-indicatoren; kosten van de toeleveringsketen en gecreëerde voltijdbanen.

**Hoofdstuk 6** bespreekt met een paarsgewijze vergelijking de synergieën en uitruilen (*trade-offs*) tussen broeikasgasemissiereductie (SDG 13) en andere Duurzame Ontwikkelingsdoelen (SDG's) wanneer land wordt gebruikt voor de productie van specifieke energiegewassen onder contextspecifieke omstandigheden. Daarnaast worden ook de contextspecifieke omstandigheden (grondstof, voormalig landgebruik, klimaat, bodems en beheer) waarin synergieën en uitruilen ontstaan, gespecificeerd.

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**Beleidsaanbevelingen**

- Geïntegreerde milieu- en sociaaleconomische beoordelingen moeten worden geïmplementeerd in de beslissingsondersteuning voor de ontwikkeling van biogebaseerde waardeketens. Dit is belangrijk omdat het beoordelen van één enkele duurzaamheidsindicator niet voldoende kan zijn om een holistisch beeld te geven van de duurzaamheidsprestaties van biogebaseerde waardeketens. Het achterwege laten van een geïntegreerd perspectief kan dus leiden tot de ontwikkeling van biogebaseerde waardeketens met positieve effecten op de ene duurzaamheidsdoelstelling en nadelige effecten op andere. Daarnaast kunnen geïntegreerde duurzaamheidsbeoordelingen de ontwikkeling van strategieën bevorderen om negatieve duurzaamheidseffecten van biogebaseerde waardeketens te verminderen, positieve effecten te versterken en te voorkomen dat dit leidt tot sectoroverschrijdende nadelige duurzaamheidseffecten.

- Beleid inzake de duurzaamheid van biogebaseerde waardeketens zou rekening moeten houden met contextspecifieke omstandigheden (vooral landgebruiksverandering, grondstofsoort en biofysische kenmerken) en omgekeerd. Bovendien maken biogebaseerde waardeketens deel uit van grotere sociaaleconomische systemen met verschillende eisen voor grondgebonden diensten die erkend moeten worden. Beleid voor biogebaseerde waardeketens moet worden afgestemd op beleid voor andere grondgebonden diensten, zoals landbouw en natuurbescherming. Het niet afstemmen van beleid en het niet opnemen van contextspecifieke voorwaarden kan leiden tot gebrekkige en contraproductieve beleidsontwikkeling.

- Het methodologisch kader dat in deze studie wordt gepresenteerd (Figure 7-1) kan waardevolle inzichten opleveren voor beleidsvorming en vrijwillige certificeringsregelingen. Het kan helpen bij het ontwikkelen van routes voor duurzaam landgebruik voor biomassaproductie en het op de markt brengen van biogebaseerde waardeketens. Dit kader maakt het mogelijk de duurzaamheid van biogebaseerde waardeketens te beoordelen, van biomassaproductie tot eindgebruik, rekening houdend met de beschikbaarheid van land (concurrentie tussen grondgebonden diensten), biomassaproductiviteit en milieu- en sociaaleconomische effecten, op basis van contextspecifieke omstandigheden. Het kader maakt het ook mogelijk synergieën en afwegingen tussen milieu- en sociaaleconomische effecten en risico's te identificeren die duurzaamheidsbelemmeringen kunnen vormen voor de ontwikkeling van biogebaseerde waardeketens. Het kader zou dus ook kunnen helpen om de duurzaamheid van biogebaseerde waardeketens en hun bijdrage aan SDG's te certificeren. Bovendien biedt het geïntegreerde kenmerk van het kader een tegenwicht voor de mogelijke inconsistenties die aanwezig zijn in sommige generieke beleidsmaatregelen, waardoor de inzet van hoogwaardige biogebaseerde waardeketens mogelijk wordt en de slecht presterende ketens worden vermeden. Ter illustratie kunnen alle marginale land locaties die voldoen aan REDII landgerelateerde duurzaamheidscriteria mogelijk biomassa produceren. Niet alle productielocaties voor biomassa zijn echter geschikt voor biogebaseerde waardeketens, aangezien de REDII-drempel voor broeikasgasemissiereductie niet wordt gehaald vanwege landgebruiksveranderingsgerelateerde broeikasgasemissies. Zo zouden alle locaties kunnen worden gewijd aan duurzame biomassaproductie en mogelijk herstel van ecosystemen. Toch moeten alleen de gebieden met hoge prestaties in termen van landgebruiksveranderingsgerelateerde broeikasgasemissies worden geselecteerd voor biogebaseerde waardeketens en de slecht presterende gebieden voor ander eindgebruik of duurzame biomassagroei (zonder oogst) als herstelproces. Het ontwikkelen van deze benadering is een poging om een geïntegreerd raamwerk te bieden dat algemeen kan worden toegepast, waarbij rekening wordt gehouden met de meest relevante stappen in biogebaseerde waardeketens met een sterke focus op landgebruiksverandering. Het blijft echter een uitdaging om generieke benaderingen met locatiespecifieke omstandigheden en regionale effecten op landgebruik (bijv. veranderingen in biodiversiteit) in evenwicht te brengen met productspecifieke benaderingen.

- Het gebruik van marginale gronden voor de productie van biomassa en de levering van biogebaseerde waardeketens zou verder in het beleid kunnen worden erkend als een relevante strategie om de uitstoot van broeikasgassen te verminderen, ondersteuning te bieden om aan de vraag naar biogebaseerde producten te voldoen, synergieën te bieden en de compromissen met andere SDG's. Vooral door de productie van meerjarige gewassen die beter zijn aangepast aan marginale omstandigheden en minder snel concurreren met andere grondgebonden diensten en dus minder verdringingseffecten genereren. Bovendien kan de productie van meerjarige gewassen in marginaal land gunstige effecten hebben op de biodiversiteit en bijdragen aan landherstel. Marginale gronden bevinden zich echter over het algemeen in afgelegen moeilijk toegankelijke gebieden. Om deze gebieden te ontwikkelen zijn er meer investeringen nodig in infrastructuur en de gehele toeleveringsketen. Deze investeringen kunnen adequate voorwaarden scheppen voor het ontwikkelen van nieuwe markten en het bevorderen van duurzame biomassaproductie als een mogelijk economisch alternatief voor boeren.

- Duurzame biomassaproductie moet bijdragen aan meerdere SDG's en beleid moet gericht zijn op dit veelzijdige karakter door de waarde van positieve externe effecten van biomassaproductie voor biogebaseerde waardeketens te internaliseren. Bijvoorbeeld het internaliseren van de waarde van koolstofvastlegging, herstel van ecosystemen, verbetering van de biodiversiteit, verhoging van de bodemkwaliteit en verbetering van de waterkwaliteit. Het waarderen van positieve externe effecten kan duurzame biomassasystemen bevorderen en zo economisch levensvatbaarder worden. Bovendien kan dit waarde-internaliseringsproces een economische basis bieden voor het ontwikkelen van mitigatiestrategieën voor negatieve externe effecten en het aanpakken van belangrijke knelpunten zoals het inkomen van boeren.

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After graduation, Ivan worked as a data analyst in ScopeInsight, a consultuncy company dedicated to professionalizing farmers' organizations worldwide. Afther that, Ivan began his Ph.D. and junior researcher position at Utrecht University. His research focused on assessing spatially explicit environmental impacts from biomass production, the sustainability of bio-based value chains, and impacts from land use. He contributed to several projects, such as the ADVANCEFUEL Horizon 2020 project. His publications involve several sustainability topics in different locations such as South Africa, Brazil, and Europe.

Currently, Ivan is working as a scientific innovatior at TNO (Nederlandse Organisatie voor Toegepast Natuurwetenschappelijk Onderzoek). His work focuses on the environmental impacts of bio-based value chains and plastics.