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Assessing the environmental impacts of cropping systems and cover crops: Life cycle assessment of FAST, a long-term arable farming field experiment

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A B S T R A C T

To reduce environmental impacts of cropping systems, various management strategies are being discussed. Long-term field experiments are particularly suitable to directly compare different management strategies and to perform a comprehensive impact assessment. To identify the key drivers of several environmental impacts, we analysed a six year crop rotation of the Farming System and Tillage Experiment (FAST) by means of the Swiss Agriculture Life Cycle Assessment method (SALCA). The following factors of the FAST experiment were considered: (1) cropping system (stockless conventional farming vs. organic farming), (2) tillage (intensive tillage vs. no or reduced tillage), and (3) cover crop. We analysed the effects of these three factors on the global warming potential (GWP), aquatic and terrestrial eutrophication, and aquatic ecotoxicity for two functional units, i.e. per product and per area. Potential impacts on biodiversity were also analysed. Our analysis revealed that there is not one superior cropping system, as the ranking depended on the environmental impact selected and on the functional unit. The cropping system had the strongest effect on most of the environmental impacts, and this was mainly driven by differences in N-fertilisation (amount and form) and yield. The global warming potential, for instance, was highest in both conventional systems compared to the organic systems, when emissions were calculated per area. In contrast, calculating emissions per product, there were no statistical differences between all four systems. On the other hand, due to higher nitrogen emissions related to the application of cattle slurry in the organic system, the terrestrial eutrophication of the organic systems was higher than the conventional systems, independent of the functional unit. The effects of tillage were much lower compared to the cropping system. No tillage, but not necessarily reduced tillage, and the cultivation of cover crops had the potential to reduce aquatic eutrophication. As N-fertilisation dominated many impact categories, we suggest improving the N-efficiency as a crucial leverage point to improve the environmental performance of arable farming systems.

1. Introduction

Agricultural production increased tremendously in the last decades (Tilman et al., 2002). However, agriculture also has negative impacts on the environment. Depending on the definition of the system boundaries, it is estimated that agriculture contributes between 13.5% and 30.0% to the total global anthropogenic greenhouse gas emissions (Bellarby et al., 2008; IPCC, 2007). Between 1961 and 2011, these emissions doubled, reflecting the pace of the continual agricultural intensification (FAOSTAT, 2014). Population growth, rising per capita caloric intake, changing dietary preferences, and limited resources, particularly agricultural land, are important drivers for the increasing intensification of the agricultural production and its emissions (Popp et al., 2010; Smith et al., 2007). Although animal husbandry is responsible for a vast amount of climate-relevant emissions (enteric CH_4), arable farming is particularly associated with volatile and aquatic nitrogen losses (Carpenter et al., 1998; Skinner et al., 1997). In fact, one of the most critical leverage points in agriculture is the excess of nitrogen in agricultural areas and associated N emissions (West et al., 2014). The excess of nitrogen is also responsible for the nutrient enrichment in terrestrial and, together with phosphorus, in aquatic ecosystems, the eutrophication, which can cause tremendous changes in the environmental conditions and thus species composition (Carpenter et al., 1998; Withers and Haygarth, 2007). Additionally, the land use and associated changes in natural habitats have a strong impact on the natural

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flora and fauna. Agriculture is known to be a key driver for above- and belowground biodiversity loss (Butler et al., 2007; Green et al., 2005; Hails, 2002; Bender et al., 2016). Furthermore, numerous other detrimental impacts on the environment, such as soil erosion or the contamination with toxic pollutants are related to agriculture, especially to arable farming (Aktar et al., 2009; Carvalho, 2006; Skinner et al., 1997).

Considering the numerous potential impacts of arable farming and the necessity to produce enough food, the question arises: How could arable farming be optimised for the lowest possible impacts on the environment? To address this question, several strategies are currently discussed, for example the cultivation of cover crops and an improved nutrient management to increase the nutrient efficiency (Dalgaard et al., 2011; Smith et al., 2008; Smith et al., 2007). In Switzerland, for instance, efforts in nutrient management were successful and nitrogen excess could be reduced by about one-third between 1990 and 2013 (BFS, 2015). Further suggested management strategies are the expansion of organic farming systems or conservation agriculture (Gattinger et al., 2012; IPCC, 2007; Johnson et al., 2007; Khaledian et al., 2010; Smith et al., 2008). Numerous empirical studies were performed to test these different management strategies: The tillage regime, for instance, was often tested for its effects on agricultural parameters, such as soil properties, weed abundance and yield potential (De Vita et al., 2007; Gronle et al., 2015; Pittelkow et al., 2015; Vakali et al., 2011). As the "sustainability" i.e. the emission reduction became a central topic in research on agricultural management strategies, there are many studies focusing on environmental parameters, such as ammonia losses and greenhouse gas emissions (Bacenetti et al., 2015; Bacenetti et al., 2016; Carozzi et al., 2013; Fusi et al., 2014; Hokazono and Hayashi, 2012; Niero et al., 2015; Wang and Dalal, 2015). Most of these studies applied a Life Cycle Assessment, a method to assess emissions and resource use occurring from "cradle-to-grave" (Finkbeiner et al., 2006). However, it has also been shown that the environmental performance of a farming system strongly depends on the perspective of the analysis. Due to the lower yield in organic farming, the ecological advantage often diminishes dramatically if emissions are analysed per product unit (yield) instead of area unit (ha; Tuomisto et al., 2012). For that reason, the intensification of agriculture in terms of emissions relative to the yield has been also suggested as a cost-effective greenhouse gas mitigation strategy (Burney et al., 2010; Cassman, 1999).

These examples illustrate clearly that the uncertainties regarding the environmental performance of management strategies are still high and that site-specific and empirical long-term verifications are essential to overcome this problem. Moreover, studies testing multiple management strategy simultaneously and taking site-, crop-, and year-specific interactions into account are lacking. Therefore, the Farming System and Tillage Experiment (FAST), a long-term arable field experiment, was established in Switzerland. The FAST investigates mainly how several important agricultural indicators, such as productivity, plant nutrition, nutrient cycling, as well as plant and soil biodiversity and their ecosystem functions are affected by cropping system (stockless conventional vs. organic management), tillage (intensive tillage vs. no or reduced tillage), and cover crops (Wittwer et al., 2017). Moreover, this experiment aims at investigating the long-term environmental performance of the examined cropping systems. To consider multiple environmental impacts simultaneously, a Life Cycle Assessment (LCA) was performed, which has become an established tool to assess emissions from complex processes, such as agricultural production. LCA is a quantitative assessment of the main emissions occurring throughout the whole value chain, from 'cradle to grave'. Considered processes include resource extraction, production of primary materials and infrastructure, processing, transport, and storage. The sums of all occurring emissions and used resources (life cycle inventory) are summarised in impact categories, such as the global warming potential (GWP; Guinée, 2001; Hellweg and I Canals, 2014; ISO, 2006a), and are generally expressed as equivalents of one contributing substance (e.g. kg CO_2 equivalent).

To reveal the environmental impacts of the various management strategies of the FAST, we analysed the experiment with the Swiss Agriculture Life Cycle Assessment tool (SALCA). The central aim of this study was to evaluate and compare the environmental impacts of the three experimental factors of the FAST experiment over a six-year crop rotation from 2009 to 2015: (1) cropping system (stockless conventional farming vs. organic farming), (2) tillage, and (3) cover crops (compared with bare fallow). Through this analysis, we aimed at identifying the key drivers for individual emissions and thus the leverage points for ecological improvements in arable farming.

2. Materials and methods

2.1. Farming System and Tillage Experiment (FAST)

The FAST was established 2009 and is ongoing near the agricultural research institute Agroscope in Zurich-Reckenholz, Switzerland (latitude 47°26′N, longitude 8°31′E). General aims of this long-term experiment are to assess the agronomical performance, ecological services (e.g. plant and soil biodiversity and interactions, nutrient cycling, soil ecological functions), and economic viability of various production systems. Specific aims are a) a general comparison of four production systems of arable crops in Switzerland, b) the development of reduced tillage in organic farming, and c) assessing the role of cover crops in the examined systems (Wittwer et al., 2017). The three investigated factors with the corresponding factor levels in this study are:

I. Cropping system: stockless conventional farming (C) vs. organic farming (O)
II. Tillage: intensive tillage (IT) vs. no tillage (NT; for stockless conventional farming) or reduced tillage (RT; for organic farming)
III. Cover crop: non-legume (NL) vs. legume (L) vs. mixture (M) vs. control (C; fallow)

The cropping systems differed mainly in the type of fertilisation and the weed and pest control: In the stockless conventional systems (C), solely mineral fertilisers were used, weeds were controlled by synthetic herbicides, and insecticides or fungicides were applied when pest severity was above the incidence threshold. Generally, the fertilisation was done according to Swiss guidelines for fertilisation, which means that in the conventional systems, winter wheat and maize received 110 or 120 kg N/ha and 90 kg N/ha, respectively (see Table 1a). In the organic systems (O), fields were fertilised with cattle slurry at a target level of 1.4 livestock units ha^{-1} (on average 117 kg N_{total} ha^{-1} yr^{-1} or 51 kg $N_{mineral}$ ha^{-1} yr^{-1}). According to the Swiss organic rules, weed control was performed mechanically (hoeing, raking), and no pesticides were applied. The tillage regimes differed in the presence (IT) or absence (NT, RT) of conventional ploughing. As we wanted to represent the typical conservation tillage practice for each cropping system (stockless conventional vs. organic), reduced-tillage (RT) was performed in the organic system and no-tillage (NT) was performed in the stockless conventional system (Carr et al., 2012). Whereas in the reduced-tillage regime, soil tillage was performed to a target depth of 5 cm (with a disk or rotary harrow) primarily for weed control, no soil tillage at all occurred in the no-tillage regime and weed control was performed by additional use of the herbicide glyphosate. The combination of the two factors cropping system and tillage resulted in four investigated so-called production systems: C-IT (conventional intensive tillage), C-NT (conventional no tillage), O-IT (organic intensive tillage),

Table 1a
Overview of the field operation performed at the Farming System and Tillage Experiment (FAST) for the four arable crops wheat 1 and 2, maize and field bean.

	Wheat 1	Maize	Field bean	Wheat 2
	Machine / Product / Amount	Machine / Product / Amount	Machine / Product / Amount	Machine / Product / Amount
Cover crop	Sowing	Sowing		
Destruction (C-IT, O-IT, O-RT)	Mulching	Mulching		
Tillage				
Intensive tillage (C-IT, O-IT)	Mouldboard plough (20 cm) Rotary harrow (5 cm)	Mouldboard plough (20 cm) Rotary harrow (5 cm)	Mouldboard plough (20 cm) Rotary harrow (5 cm)	Mouldboard plough (20 cm) Rotary harrow (5 cm)
ORG reduced tillage (O-RT)	Disk harrow (5 cm)	Rotary harrow (5 cm)	Rotary harrow (5 cm)	Rotary harrow (5 cm)
CONV no tillage (C-NT)	Glyphosate (4 l/ha)	Glyphosate (4 l/ha)		Glyphosate (4 l/ha)
Sowing	400 seeds/m^2	9.5 seeds/m^2	45 seeds/m^2	400 seeds/m^2
Fertilisation				
CONV				
Total P/K		47 kg P_2O_5/ha	125 kg P_2O_5/ha 194 kg K_2O/ha	60 kg P_2O_5/ha 100 kg K_2O/ha
Total N	110 kg N/ha	90 kg N/ha		120 kg N/ha
Application 1	50 kg N/ha	30 kg N/ha		60 kg N/ha
Application 2	30 kg N/ha	60 kg N/ha		30 kg N/ha
Application 3	30 kg N/ha			30 kg N/ha
ORG				
Total N_T/N_{min}	126/50 kg N/ha	137/67 kg N/ha		88/35 kg N/ha
Application 1	30 m^3/ha	30 m^3/ha		30 m^3/ha
Application 2	30 m^3/ha	40 m^3/ha		30 m^3/ha
Weed control				
CONV	Herbizides: isoproturon, terbuthylazine, glyphosate (C-NT)	Herbizides:, isoproturon, terbuthylazine, glyphosate (C-NT)	Herbizides: isoproturon, terbuthylazine, glyphosate (C-NT)	Herbizides: isoproturon, terbuthylazine, glyphosate (C-NT)
ORG	Harrow	Hoing Hoing	Hoing Hoing	Harrow Harrow
Harvest	Combine	Combine	Combine	Combine

Table 1b
Overview of the field operation performed at the Farming System and Tillage Experiment (FAST) for the two years of grass-clover ley.

	Grass-clover 1st year	Grass-clover 2nd year
	Machine/Product/ Amount	Machine/Product/ Amount
Stubble cultivation	Mulcher	-
Tillage		
CONV intesive tillage, ORG intesive tillage and reduced (C-IT, O-IT, O-RT)	Rotary tiller (5 cm)	-
CONV no tillage (C-NT)	-	-
Sowing	33 kg/ha, 16.6 cm row distance	-
	Roller	-
Fertilisation		
CONV (mineral)		
Total P/K	90 kg P_2O_5/ha; 275 kg K_2O/ha	80 kg P_2O_5/ha; 240 kg K_2O/ha
Total N	130 kg N/ha	100 kg N/ha
Application 1	40 kg N/ha	40 kg N/ha
Application 2	30 kg N/ha	30 kg N/ha
Application 3	30 kg N/ha	30 kg N/ha
Application 4	30 kg N/ha	
ORG (Cattle slurry, 1.4 LU/ha)		
Total P/K	103 kg P_2O_5/ha; 451 kg K_2O/ha	93 kg P_2O_5/ha; 539 kg K_2O/ha
Total N_T/N_{min}	205/66 kg N/ha	219 / 93 kg N/ha
Application 1	30 m^3/ha	30 m^3/ha
Application 2	30 m^3/ha	30 m^3/ha
Application 3	30 m^3/ha	30 m^3/ha
Application 4	30 m^3/ha	30 m^3/ha
Harvest		
	1st cut	1st cut
	2nd cut	2nd cut
	3rd cut	3rd cut
	4th cut	4th cut
	5th cut	-

Table 2
Description of the eight impact groups to which emissions could be assigned.

Input group[1]	Description
N-fertiliser	Emissions from the production of N-fertilisers
Field emissions	Direct and indirect N_2O, NH_4, and NO_{3-} emissions, mainly during and after the application of fertilisers
Machines	Energy demand and emissions from the production of machinery infrastructure (all except soil cultivation devices)
Tillage	Energy demand and emissions from the production of used soil cultivation devices (proportional to application; harrow, plough)
Other fertilisers	Emissions from the production of all mineral fertilisers, such as phosphorus and potassium (except nitrogen)
Seeds	Emissions from the production of seeds (includes all relevant processes, such as machines and fertilisers)
Pesticides	Emissions from the production; impacts of pesticide application on toxicity
Other inputs	Emissions from the production of goods not belonging to one of the previous input groups

[1] The transports of the inputs (fertilisers, pesticides, seeds, Diesel) to the farm are included in each input group.

O-RT (organic reduced tillage). An overview of the most important field operations is given in Tables 1a & 1b.

All four production systems were combined with the four factor levels of cover crop, which led to 16 treatments in total. The experimental plots were arranged in a split plot design ($n = 4$).

All production systems (main plot) in FAST had the same six-year crop rotation: cover crop (CC1), winter wheat (WW1; *Triticum aestivum*), cover crop (CC2), maize (ZEA; *Zea mays*), faba bean (FAB; *Vicia faba*), winter wheat (WW2; *Triticum aestivum*), and two years of grass–clover ley (LEY; *Lolium multiflorum*, *Trifolium pratense*). Cultivated cover crop species were white mustard (non-legume; NL; *Sinapis alba*), vetch species (legume; L; *Vicia sativa* and *Vicia villosa)*, and mixtures (M) of various species. The mixture treatment differed in its composition with Phacelia (*Phacelia tanacetifolia*), Persian clover (*Trifolium resupinatum*), and Berseem clover (*Trifolium alexandrinum*) for CC1 and Phacelia, buckwheat (*Fagopyrum esculentum*), hairy vetch (*Vicia villosa*), and Camelina (*Camelina sativa)* for CC2.

Note that the stockless conventional systems were managed according to the 'proof of ecological performance' regulations of the direct payments system in Switzerland (FOAG, 2004) and represent a kind of integrated management that is already less intensive than typical conventional systems in Europe. Moreover, fertilisation was strictly mineral although the majority of conventional farms in Switzerland possess livestock (mixed farming) and use both organic and mineral fertilisers. The FAST provided all required data, such as the field operations, their dates and used machines, and capital goods to model the emissions with SALCA. The variable 'yield', necessary to calculate the emissions per product (see Functional units), was the only empirical characteristic used. More details about the FAST experiment, including experimental setup, machine use, and yield in 2010 and 2011, is given in Wittwer et al., 2017.

2.2. LCA method

The LCA was conducted by means of SALCA (Nemecek et al., 2010; Nemecek et al., 2011). SALCA includes the use of life cycle inventories from the ecoinvent database (version 2.2; ecoinvent Centre, 2010; Hischier et al., 2010, Nemecek and Kägi, 2007). Additionally, SALCA comprises the SALCA database and models for estimating direct field emissions. Input data and inventories necessary for the calculations are:

- Environmental data: such as soil type, climate, hydraulic balance.
- Infrastructure: buildings, machinery.
- Capital goods and inputs: fertilisers, pesticides, seeds, fossil fuels.
- Field operations: such as tillage, sowing, harvest.
- Crop rotation and type-of-use: such as crops, pastures.
- Dates of operations: to consider interactions between date (season), crop, operation and emissions.

The following emissions were estimated using these assumptions:

- The losses of ammonia (NH_3) from mineral N-fertilisers were estimated with constant emission factors according to Asman (1992) dependent on the type of fertiliser. For farmyard manure, the quantity of manure, the ammonium concentration, the time of the application, and the application technique were taken into account to estimate the emission.
- Direct and induced emissions of nitrous oxide (N_2O) were estimated according to IPCC (2006). Direct emissions came from the application of N-fertiliser (1% of applied nitrogen is assumed to be released as N_2O) and incorporation of crop residues (of all incorporated crop residue N, 1% is assumed to be released as N_2O). In addition to the direct emissions, induced emissions were considered from the following processes: deposition of volatilised ammonia and nitrate leaching losses to groundwater, once the groundwater appears at the surface. The respective factors were 1% for NH_3-N and 0.75% for NO_3^--N. More details are given in Nemecek et al. (2015b).
- Three paths of phosphorus emissions to water were included: run-off as phosphate, erosion as phosphorus to rivers, and leaching to ground water as phosphate (Prasuhn, 2006). Phosphorus emissions depend on land use category, the type of fertiliser, the quantity of phosphorus spread, and the characteristics and duration of soil cover (for erosion). Further information is given in Nemecek et al. (2015b).
- Nitrate (NO_3^-) leaching was estimated by the SALCA-NO_3 model. It is calculated on a monthly balance of N-mineralisation from soil organic matter and N-uptake by the vegetation, specific to each crop (Richner et al., 2014). N-mineralisation depended on the clay and humus contents, the intensity of soil tillage, and the preceding crop. For the crops in the FAST, the following factors were relevant: after the incorporation of residues from grain legumes, the mineralisation rate was increased by 21% for six months; after a cover crop, it was increased by 21% for four months (Richner et al., 2014). If mineralisation exceeded uptake by the crop, the excess nitrogen was assumed as leached. In addition, the risk of nitrate leaching from fertiliser application was calculated, taking into account the crop, the month of application, and the potential rooting depth. Further information is given in Nemecek et al. (2015b).
- Heavy metal emissions (Cd, Cr, Cu, Hg, Ni, Pb, and Zn) were assessed by an input–output balance (Freiermuth, 2006). The following inputs were considered: seeds, fertilisers, and pesticides. Outputs by harvested products, erosion, and leaching were included. Only part of the quantities lost to the aquatic environment by erosion or leaching was considered, because the farmer controls these processes to some extent to prevent the deposition of heavy metals. The allocation factor was derived from the share of agricultural inputs in the total inputs (including deposition). Further information is given in Nemecek et al. (2015b).
- Pesticide emissions were modelled in a simplified manner as 100% emission of the active ingredient into the agricultural soil. The further fate of the substances like degradation or transfer to water was included in the impact assessment method for aquatic ecotoxicity (see below).

A detailed description of the SALCA method can be found in Nemecek et al. (2010). To enhance the comprehensibility and reduce redundancy, we focused our detailed analysis on selected impact categories that are central for agriculture. Previous analyses have revealed strong correlations between several impact categories, which therefore show similar relative results between the analysed treatments (Nemecek et al., 2011). The selected impact categories for this paper were:

- Global warming potential (100 years; kg CO_2 eq.; IPCC, 2007)
- Aquatic eutrophication potential (kg N eq.; CML01)
- Terrestrial eutrophication potential (m^2; CML01)
- Aquatic ecotoxicity potential (1,4-dichlorobenzene eq.; CML01)

To understand the role of impact-relevant processes, the results include the contribution of eight so-called input groups. Each of these input groups has a different colour code in the presented figures. Processes comprised by the input group are described in Table 2. Due to the strong impact of agriculture on species diversity, the effects of the production systems on biodiversity were assessed additionally. For the biodiversity assessment, we modelled the suitability of the test fields for species diversity under the four production systems. For this purpose, we applied the SALCA biodiversity method (Jeanneret et al., 2014) that calculates effects of habitat quality, i.e. the crop type, and of management practices on eleven groups of indicator species (crop flora i.e. weeds, grassland flora, birds, small mammals, amphibians, snails, spiders, carabid beetles, butterflies, wild bees, and grasshoppers). Effects on biodiversity of detailed management practices of the four production systems and cover crops were estimated based on model calculations, and individual results were aggregated to an overall biodiversity score. Note that the estimated biodiversity values are based on indicator values obtained in other studies and in different fields, and the presented values represent model estimates and do not necessarily reflect the actual effects of the different production systems and cover crops on biodiversity in this experiment.

2.3. System boundaries

The system boundaries were defined according to Nemecek et al. (2005): The spatial boundaries of the agricultural production system were defined by the borders of a field. Upstream emissions and resource use for the provision of infrastructure and the production of commodities were also included. The respective life cycle inventories were taken from the ecoinvent database (version 2.2; ecoinvent Centre, 2010; Nemecek and Kägi, 2007) or the SALCA database (Nemecek et al., 2010). Emissions from manure storage were fully assigned to the animal production system and therefore not included (cut-off approach). The temporal assessment period for a single crop started after the harvest of the previous crop and ended with its own harvest. Corresponding to previous LCA studies for the evaluation of whole crop rotations, the start and the end of the complete crop rotation defined the temporal boundaries. In our case, we had a six-year crop rotation. As empirical data of soil carbon alteration were not available and current estimation methods are still under critical debate, potential changes in the soil carbon content were not considered in this analysis (see also Discussion section).

2.4. Functional units

Agriculture is multifunctional (Nemecek et al., 2011). In brief, the main functions are the production of commodities (productive function), the maintenance of a sustainable land use (land management function), and the generation of income (financial function). The productive function is the main reason for agricultural production, but the land management and financial functions are also important functions. From these functions, the functional units can be deduced, which are I) per product (e.g. kg), II) per area and time (i.e. ha and year), and III) per income (e.g. CHF). As the financial evaluation was not a main task of the study, it was not considered in the following. The first and second functional units are linked to two foci: a) products with low environmental impacts and b) land use with low environmental impacts. As the cover crops were not harvested, they had no physical product. Their impacts were thus allocated to the succeeding main crops winter wheat and maize (only per product).

Evaluation of a crop rotation with various crops requires a product unit, which allows the comparison of crops with differing agricultural and nutritional functions. Therefore, we used the cereal unit (CU), a common unit in German agricultural statistics, as the per product unit. According to a standardised method, the CU of a product expresses the nutrition value for pig fattening relative to 100 kg barley, which is defined as the reference with a CU of 1. Brankatschk and Finkbeiner (2015) recommended the use of the CU for the analysis of crop rotations. To allow a comparison of our results with other studies, we expressed the final results per year by dividing the results by the duration of the complete crop rotation. In summary, we analysed the data per ha and year and per CU.

2.5. Statistical analysis

To test treatment effects on the impact categories, linear mixed-effects models (Pinheiro et al., 2012; Table 3) were used for each impact category and functional unit (per ha and year, per CU), with production system (C-IT, C-NT, O-IT, O-RT) and cover crop (C, NL, L, M) as fixed effects (main factors). As we wanted to investigate the effects of the agricultural practice, i.e. the treatments, independent of the crops and the crop rotation, the crops were used as replicates and set as random effect in the model.

If one of the fixed factors was significant (p-value < 0.05), a post hoc test (Tukey) with multiple comparisons was performed (Table 4). Note that the data of the statistical analysis are slightly different from data shown in the figures: Whereas the figures represent the emissions per crop rotation and year (sum divided by six years), the statistical test used the crops as replicates, and thus the calculated means and standard deviations represent average annual management of an ‘average crop’. As the biodiversity scores are dimensionless, the biodiversity data as well as its statistical test were not assigned to a functional unit.

Moreover, we want to note that the statistical test is based on the output of a model, which uses to large extents emissions inventories (ecoinvent data). Such an inventory can be included multiple times. Here, this was the case for the calculations of the different treatments. In terms of statistical analyses, such data show strong co-variance, in general.

3. Results

The environmental impact of different cropping systems can be assessed per unit food produced or it can be based on a comparison of land use and management where the area and year is the functional unit. In our analysis below we present both measures, first the land management function (per ha and year), followed by the productive function. The reason is that this presentation allows to better interpret the results, as the effects of different inputs or management per area and the different yield can be clearly distinguished. In contrast, results “per functional unit food produced (CU)” includes the effect of both, management and yield. The relative contribution of the input groups to the total impact is the same “per ha” and “per CU” and therefore does not need to be repeated.

Table 3
Effects of the (fixed) factors production system (ps) and cover crop (cc) on the four impact categories global warming potential (GWP), aquatic eutrophication (nitrogen), terrestrial eutrophication, aquatic ecotoxicity and biodiversity scores. Effects were tested by an *F*-test referring to a linear mixed-effects model. The crops ($n = 7$) within the crop rotation were specified as random effect and are considered replicates. Significant differences at a *p*-value < 0.05 are printed in bold.

	Impact category	GWP		Aquatic eutrophication		Terrestrial eutrophication		Aquatic ecotoxicity		Biodiversiy scores
	Functional unit	ha and year	CU	ha and year	CU	ha and year	CU	ha and year	CU	–
Factors	ps	**< 0.001**	**< 0.001**	0.185	**0.006**	**< 0.001**	**< 0.001**	**< 0.001**	**< 0.001**	**0.034**
	cc	0.531	0.830	**0.020**	0.808	0.999	0.958	1	0.995	< 0.001
	ps: cc	1	0.997	1	0.999	1	0.997	1	1	1

Table 4
Multiple comparisons within the two fixed factors production system (C-IT: conventional – intensive tillage, C-NT: conventional – no tillage, O-IT: organic – intensive tillage, O-RT: organic – reduced tillage) and cover crop (C: control, NL: non-legume, L: legume, M: mixed). The variables were tested with a post hoc (Tukey) test based on a linear mixed-effects model (Table 3). Different letters indicate significant differences (*p*-value < 0.05).

Impact category		GWP		Aquatic eutrophication		Terrestrial eutrophication		Aquatic ecotoxicity		Biodiversity scores
Functional unit		ha and year	CU	ha and year	CU	ha and year	CU	ha and year	CU	–
Production system	C-IT	a	a	a	a	a	a	a	a	a
	C-NT	a	a	a	a	a	a	a	a	a
	O-IT	b	a	a	a	b	ab	b	b	a
	O-RT	b	a	a	a	b	b	b	b	a
Cover crop	C	a	a	a	a	a	a	a	a	a
	NL	a	a	ab	a	a	a	a	a	a
	L	a	a	b	a	a	a	a	a	a
	M	a	a	ab	a	a	a	a	a	a

3.1. Global warming potential

The production system (C-IT, C-NT, O-IT, O-RT) had a significant effect (*p*-value < 0.001) on global warming potential (GWP) per ha and year (Tables 3 and 4). The two stockless conventional production systems (C-IT, C-NT) had about 80% higher emissions than the two organic systems (Fig. 1; mean conventional: 2375.8 kg CO_2 eq. ha^{-1} yr^{-1}, SD: ± 577.0; mean organic: 1298.1 kg CO_2 eq. ha^{-1} yr^{-1}, SD: ± 516.9; mean impact categories of the four different production systems are in Table 1 of the Supplementary material). The difference was mainly caused by the higher emissions from the mineral fertilisers (input groups N-fertiliser and other fertilisers), with the highest contribution by the N-fertiliser (~50% of the overall emissions). As no mineral fertilisers were used in the organic systems (O-IT; O-RT), these emissions were absent. In contrast, in the organic production systems, field emissions originating mainly from the application of manure had the highest contribution to the overall emissions (> 50%). Field emissions of the organic production systems were roughly two-fold of those of the stockless conventional production systems.

Due to the higher energy demand for tillage operations (input group tillage), production with tillage (IT: intensive tillage) had always slightly higher emissions (~10%; not significant) than the production without ploughing (NT: no-tillage, RT: reduced tillage), independent of the cropping system.

The factor cover crop had no significant influence (*p*-value = 0.531) on the GWP per ha and year. Generally, the cultivation of cover crops always had a slightly higher GWP than the control (fallow), caused by the additional N_2O emissions from crop residues and the additional energy demand for management (seeding, mulching) necessary for the cultivation of cover crops. Among the three cover crops (without control), the legume treatment always showed the highest GWP, followed by the non-legume and the mix treatment, which had similar effects on the emissions (independent of the production system). Emissions from the seed production (input group seeds) and the high N_2O emissions from the nitrogen-rich crop residues of the legumes (input group field emissions) were responsible for this difference.

Nitrous oxide and CO_2 were the most important greenhouse gases

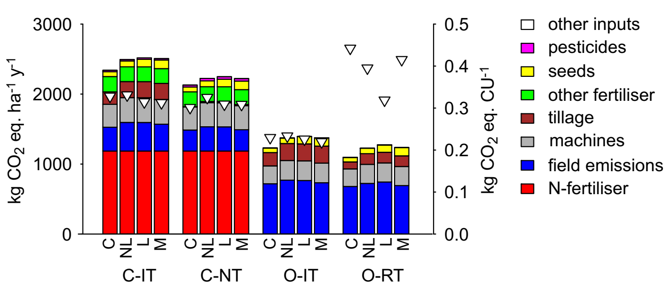

Fig. 1. Global warming potential (GWP) for the 16 treatments of the FAST with the factors production system (C-IT: conventional – intensive tillage, C-NT: conventional – no tillage, O-IT: organic – intensive tillage, O-RT: organic – reduced tillage) and cover crop (C: control, NL: non-legume, L: legume, M: mixed). The columns represent the emissions (kg CO_2 eq.) per ha and year (left y-axis). Open triangles represent the total emissions per cereal unit (CU; right y-axis).

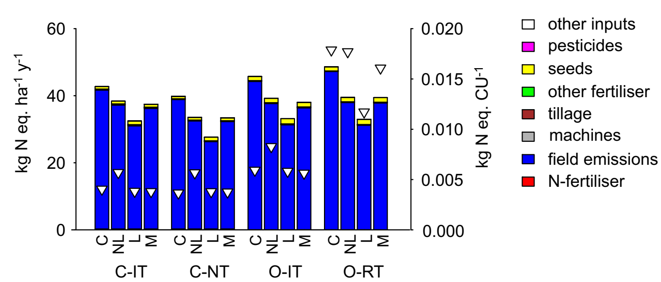

Fig. 2. Aquatic eutrophication potential (EDIP2003, Hauschild and Potting, 2005) for the 16 treatments of the FAST with the factors production system (C-IT: conventional – intensive tillage, C-NT: conventional – no tillage, O-IT: organic – intensive tillage, O-RT: organic – reduced tillage) and cover crop (C: control, NL: non-legume, L: legume, M: mixed). The columns represent the emissions (kg N eq.) per ha and year (left y-axis). Open triangles represent the total emissions per cereal unit (CU; right y-axis).

contributing to the GWP, with an average contribution (of all treatments) of 58.5% and 39.6%, respectively. In the stockless conventional production systems, main sources for N_2O emissions were the production of mineral N-fertiliser and field emissions, which together contributed almost 50%. In the organic production systems, the N_2O emissions mainly occurred in the field during/after manure application (> 90%; input group field emissions). CO_2 mainly stemmed from the fuel combustion for mechanical field management and from the production of mineral fertilisers (~40% of GWP), present only in the stockless conventional farming systems. As the animal husbandry was outside the system boundary, biogenic CH_4 was not relevant in this analysis.

Considering the emissions 'per CU' (Fig. 1, open triangles, right y-axis), the results were different: The production system also affected the GWP (p-value < 0.001). Contrary to the analysis per ha and year, organic-reduced tillage (O-RT) showed the highest GWP per CU, which was due to very low yields in this system. Almost no difference existed between the two stockless conventional systems, and each showed a smaller difference (31% more emissions) compared to O-IT per CU than per ha and year. The factor cover crop had no effect on the GWP per CU (p-value = 0.830). The relative contribution of the input groups is identical for the two functional units by definition.

3.2. Aquatic eutrophication

The SALCA model was also used to estimate aquatic eutrophication per ha and year. Production systems had no significant effect on aquatic eutrophication (p-value = 0.185, Tables 3 and 4, mean stockless conventional: 36.3 kg N eq. ha^{-1} yr^{-1}, SD: ± 14.4; mean organic: 40.3 kg N eq. ha^{-1} yr^{-1}, SD: ± 17.6; Fig. 2). The higher emissions were due to the application of slurry, with higher emission rates for ammonia. The absence of ploughing reduced aquatic eutrophication on average by about 10% (not significant) in the stockless conventional production systems because mineralisation is usually reduced under no-tillage regimes (Doran, 1980; Kandeler and Böhm, 1996). In the organic production systems, reduced tillage showed on average a slightly higher aquatic eutrophication (~3%; not significant) than production with intensive tillage. This result was strongly influenced by the low yield of O-RT with consequent low N-uptake and, hence, higher nitrate leaching risk.

Aquatic eutrophication was dominated by leaching of nitrate (originating mainly from mineral and organic N-fertilisers as well as mineralised soil organic matter) into ground water (included in field emissions). Furthermore, nitrate leaching occurred also upstream during the production of seeds that are later used in the FAST (input group seeds).

The factor cover crop influenced aquatic eutrophication per ha and year (p-value = 0.020). In comparison with the control, the inclusion of cover crops led always to a reduction in aquatic eutrophication, which was significant only for the legume treatment (p-value < 0.001; Table 4), because the temporal N-uptake by legumes offset best the temporal course of the mineralisation. The non-legume and the mix treatments had a similar reduction effect on aquatic eutrophication, which was not significant compared with the control (non-legume, p-value = 0.124; mix, p-value = 0.077).

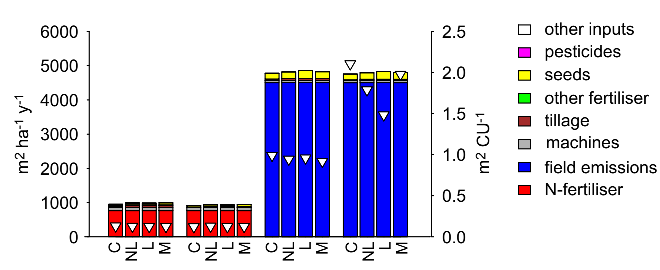

Fig. 3. Terrestrial eutrophication potential (EDIP2003, Hauschild and Potting, 2005) for the 16 treatments of the FAST with the factors production system (C-IT: conventional – intensive tillage, C-NT: conventional – no tillage, O-IT: organic – intensive tillage, O-RT: organic – reduced tillage) and cover crop (C: control, NL: non-legume, L: legume, M: mixed). The columns represent the potential area (m^2) per ha and year (left y-axis). Open triangles represent the total area per cereal unit (CU; right y-axis).

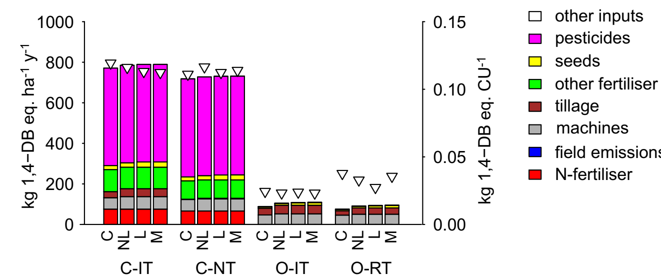

Fig. 4. Aquatic ecotoxicity (CML01) for the 16 treatments of the FAST with the factors production system (C-IT: conventional – intensive tillage, C-NT: conventional – no tillage, O-IT: organic – intensive tillage, O-RT: organic – reduced tillage) and cover crop (C: control, NL: non-legume, L: legume, M: mixed). The columns represent the emissions (kg 1,4-dichlorobenzene eq.) per ha and year (left y-axis). Open triangles represent the total emissions per cereal unit (CU; right y-axis).

In contrast, the effects per CU were significant for the production systems (p-value = 0.006) but not for the cover crops (p-value = 0.808). Due to the high uncertainties, the multiple comparison (Tukey) did not find any significance between the systems.

Whereas in the stockless conventional systems the two tillage regimes showed no difference, the organic system with reduced tillage had a higher aquatic eutrophication (~56%) than the organic system with intensive tillage.

3.3. Terrestrial eutrophication

The terrestrial eutrophication potential per ha and year was affected by the production systems (p-value < 0.001, Tables 3 and 4). In the organic systems, terrestrial eutrophication was about five times higher (mean: 4891 m^2 ha^{-1} yr^{-1}, SD: ± 4073) compared with the stockless conventional systems (mean: 972 m^2 ha^{-1} yr^{-1}, SD: ± 415; Fig. 3). This effect was mainly due to the high ammonia emissions from the manure application (input group field emissions), not present in the stockless conventional production systems. For the latter, the mineral N-fertilisers was the major source of ammonia field emissions, but the emission rates are much lower than from animal manure. There was no difference between the two tillage regimes, as this impact category was driven by fertiliser applications. The factor cover crop had no effect on terrestrial eutrophication (p-value = 0.999) and showed only minimal differences between the cover crops.

Considering terrestrial eutrophication per CU, the effects of the production systems were significant (p-value < 0.001), whereas the effects of the cover crops were not (p-value = 0.958). The O-RT system showed the highest terrestrial eutrophication, which was significantly and roughly eight times higher than in the two stockless conventional systems, but not different from the organic system with intensive tillage (Table 4). Similar to the two previous impact categories, terrestrial eutrophication per CU reflected strongly the differences in yield.

3.4. Aquatic ecotoxicity

The production systems had an effect on the aquatic ecotoxicity potential, measured in 1,4-dichlorobenzene equivalents (1,4-DB eq.), per ha and year (p-value < 0.001, Tables 3 and 4). The emissions of the two conventional systems were on average (768.6 kg 1,4-DB eq. ha^{-1} yr^{-1}, SD: ± 401.1) about six times higher compared with the two organic systems (126.8 kg 1,4-DB eq. ha^{-1} yr^{-1}, SD: ± 56.3; Fig. 4; Table 4). The responsible drivers for that difference were the use of pesticides and mineral fertilisers (input groups pesticides, N-fertiliser,

Table 5
Effects of the cropping systems on the diversity scores of 11 species groups. Numbers represent diversity scores calculated with the SALCA biodiversity model, where higher numbers mean higher diversity. The treatments were production system (C-IT: conventional – intensive tillage, C-NT: conventional – no tillage, O-IT: organic – intensive tillage, O-RT: organic – reduced tillage) and cover crop (C: control, NL: non-legume, L: legume, M: mixed).

	C-IT				C-NT				O-IT				O-RT			
	C	NL	L	M	C	NL	L	M	C	NL	L	M	C	NL	L	M
Aggregated	6.8	7.9	7.7	7.9	6.9	8	7.8	8	7.3	8.3	8.1	8.3	7.4	8.4	8.2	8.4
Crop flora	13.7	12.9	13.1	12.9	13.6	12.8	12.9	12.8	15.9	14.7	14.8	14.7	15.9	14.6	14.7	14.6
Grassland flora	1.5	2.8	2.5	2.8	1.5	2.8	2.6	2.8	1.5	2.8	2.5	2.8	1.5	2.8	2.6	2.8
Birds	12	14.2	14.4	14.2	12	14.2	14.4	14.2	12.8	14.7	15	14.7	12.8	14.7	15	14.7
Small mammals	6.1	6.1	6.5	6.1	6.3	6.2	6.6	6.2	6.1	6.1	6.4	6.1	6.1	6.1	6.5	6.1
Amphibians	2.1	2.1	2.1	2.1	2.2	2.2	2.2	2.2	2.2	2.2	2.2	2.2	2.2	2.2	2.2	2.2
Snails	2.6	2.7	2.7	2.7	2.6	2.7	2.7	2.7	2.6	2.7	2.7	2.7	2.7	2.8	2.8	2.8
Spiders	10	10.7	10.7	10.7	10.3	11	11	11	10.2	10.9	10.9	10.9	10.4	11.2	11.2	11.2
Carabid beetles	12.7	13.9	13.9	13.9	12.9	14	14	14	13.1	14.2	14.2	14.2	13.9	15	15	15
Butterflies	1.5	4.6	3.8	4.6	1.6	4.7	4	4.7	1.5	4.5	3.7	4.5	1.5	4.7	3.9	4.7
Wild bees	4.4	5.6	4.4	5.6	4.4	5.6	4.4	5.6	4.6	5.8	4.5	5.8	4.6	5.8	4.5	5.8
Grasshoppers	2.2	5.8	5.3	5.8	2.3	5.9	5.5	5.9	2.2	5.7	5.3	5.7	2.2	5.8	5.5	5.8

and other fertilisers), both of which were generally not used in organic production systems. The impact was dominated by the two herbicides isoproturon (52%) and terbuthylazine (44%). Moreover, the production, mining, and application for mineral fertiliser (impurities) causes emissions of toxic elements, such as nickel and copper, which was reflected in the input groups N-fertiliser and other fertilisers.

Within the conventional systems, the use of a plough led to a slightly higher (not significant) aquatic ecotoxicity (input group tillage) than the no-tillage regime. This difference originated from the higher energy demand for ploughing and from toxic metals emitted mainly during the production of agricultural equipment. Likewise in the organic production systems, the use of a plough led to a higher aquatic ecotoxicity than the reduced-tillage regime. The use of glyphosate in C-NT had no marked effect on aquatic ecotoxicity according to the method applied.

The factor cover crop had no influence on aquatic ecotoxicity (p-value = 1), even though the ranking of cover crops was always C < NL < L < M, independent of the production system. This ranking was due to: a) less soil cultivation in the control (absence of sowing and mulching), and b) according to the cover crops due to a different amount and type of seeds with differing emissions in their production.

Similar to the results per ha and year, the production systems showed an effect (p-value < 0.001) on aquatic ecotoxicity per CU, whereas the factor cover crop had no effect (p-value = 0.995). The two conventional systems did not differ from each other and showed on average a four times higher aquatic ecotoxicity (0.12 kg 1,4-DB eq. CU^{-1}, SD: ± 0.08) than the two organic systems (0.03 kg 1,4-DB eq. CU^{-1}, SD: ± 0.02). Within the organic production systems, the reduced-tillage regime had, due to its lower yield, a higher (not significant) aquatic ecotoxicity than the plough regime.

3.5. Biodiversity

The biodiversity indicators revealed consistently higher overall biodiversity scores for the organic than for the conventional systems (p-value 0.034, Tables 3 and 5). This was mainly due to the absence of herbicides in the organic systems, which was of clear benefit for four of the eleven indicator species groups: crop flora, birds, carabid beetles and wild bees. Tillage was not reflected in distinct differences between the scores. However, the factor cover crop was significant in the linear mixed-effects model (p-value < 0.001) for the overall biodiversity scores. Independent of the production system, the four variables showed a constant ranking: C < L < NL = M.

3.6. Overall effects of the three experimental factors (results summary)

The factor production system had the strongest impact on the GWP, terrestrial eutrophication and aquatic ecotoxicity and effects of tillage and cover crops were much lower. The functional unit had a strong impact on the results as well. As the CU strongly reflects the yield, yield differences between the production systems became relevant and increased the environmental impacts of the organic systems. For instance, O-RT, which had the lowest GWP per ha and year, showed the highest GWP per CU. Depending on the functional unit, the organic system had a slightly to distinctly higher aquatic and terrestrial eutrophication. In both production systems, the GWP and terrestrial eutrophication were strongly driven by the mineral and organic N fertilisers. The use of pesticides, mainly of herbicides, was responsible for a six times higher aquatic ecotoxicity of the conventional systems and a lower biodiversity compared with the organic systems. Tillage affected the impact categories to a much lower extend than cropping system: For the functional unit per ha conservation tillage (C-NT, O-RT) showed only a slightly, not significant reduction for some impact categories than under intensive tillage (GWP, aquatic ecotoxicity). For the functional unit per CU, aquatic and terrestrial eutrophication was highest for the reduced tillage (O-RT). Compared with the control, the cultivation of cover crops led to an increase in the GWP (not significant) but reduced aquatic eutrophication potential significantly. The factor cover crops had a minor effect on the biodiversity. In summary, the overall effects of the three tested factors on the investigated impact categories can be ranked as follows: cropping system > tillage > cover crop.

4. Discussion

4.1. The role of environmental impacts and functional units

Based on the experimental set-up of the FAST, we were able to directly compare the environmental impact of the different production systems and the contributing factors. Our life cycle assessment revealed that the cropping system (stockless conventional vs. organic) had the strongest effect on most of the environmental impacts, and this was mainly driven by differences in N-fertilisation. The effects of the tillage were much lower compared to the farming system. Moreover, the results revealed that there is not one overall superior production system, as the ranking depended on the environmental impact selected and on the functional unit. Thus, a focus on only one impact, such as the carbon footprint, bears the risk of ignoring potential environmental trade-offs, as seen for eutrophication in our study.

Furthermore, whereas the results per ha and year were mainly input driven and thus reflects best the management, the results per CU were driven by the ratio emissions/output and therefore strongly depended on the yield. For that reason, the differences between the production systems are mainly discussed in "per ha" (see Results section). The generally lower yields (dry matter) in the two organic systems reduced the advantages of organic farming for various environmental impacts. The GWP of the organic treatment with tillage was almost 50% lower compared to the conventional treatment with tillage, when calculated per hectare while it was about 25% lower, when calculated per CU. Moreover, the O-RT had, depending on the functional unit, either the lowest (per ha and year) or the highest GWP (per CU; Fig. 1). It is important to mention that crop yield in the O-RT treatment was very low (on average about 30% lower than the O-IT treatment) due to weed problems and results of this treatment should be interpreted with caution and not seen as representative for organic production in Switzerland (Wittwer et al., 2017). The relatively lower yields are a drawback for the environmental performance of organic production systems in general (Brentrup et al., 2004a; Hokazono and Hayashi, 2012; Nemecek et al., 2011; Tuomisto et al., 2012). But also for conventional farming systems, the lower yields under global climate change can increase the environmental impacts significantly (Niero et al., 2015). Organic farming is often seen as a more environment-friendly system, especially regarding soil quality and biodiversity but also regarding the GWP (Gattinger et al., 2012; Lazzerini et al., 2014; Mäder et al., 2002; Schader et al., 2012; Skinner et al., 2014). Our results agree with this view regarding biodiversity, aquatic ecotoxicity and GWP, when calculated per ha. By contrast, regarding the GWP "per CU", the organic systems do not have an ecological gain, underlining the difficulty to make such general statements.

We need to point out that the farming system called conventional in the FAST represents a mineral or a stockless system with solely mineral fertilisation, while the organic system used manure and is representative for mixed farming with livestock. Furthermore, the straw and the clover–grass are removed in both farming systems, whereas farmyard manure is returned to the soil only in the organic system. This creates an important carbon leakage in the conventional system. The results for the conventional systems are valid for stockless farms, where the clover-grass and the straw is sold to other farms, but no manure is imported. This does not correspond to a typical conventional management in Switzerland.

4.2. The role of N-fertilisation

The factor cropping system (as a component of the production system) had the most evident effect on most of the analysed impact categories, independent of the functional unit (Tables 3 and 4). Our analysis revealed that the N-fertilisation in general but also the type, i.e. mineral fertiliser vs. slurry application, was a key driver particularly affecting the GWP and terrestrial eutrophication (Figs. 1 and 3). The energy-intensive production of mineral N-fertiliser consumes large amounts of fossil fuels, whose impact represented about 50% of the GWP of the conventional systems. However, we have to bear in mind that the GHG emissions from N fertiliser production have been drastically reduced in the last years (Brentrup et al., 2016), so that the impact of mineral N fertilisers would decrease, if the most recent data would be used. Even though the slurry fertilisation led to a lower absolute GWP, the subsequent higher field emissions were crucial for the GWP (> 50% of the total GWP; mainly N_2O). For terrestrial eutrophication, the N-fertilisation was also crucial. In contrast to the GWP, the fertiliser effect was reversed, with roughly a five times higher terrestrial eutrophication for slurry application in organically managed systems compared to the conventionally managed systems with only mineral N-fertiliser. Our results are in line with previous studies showing that N-fertilisation is a key factor for several environmental impacts and at the same time a key leverage point to mitigate environmental burdens (Goglio et al., 2014; Hillier et al., 2009; Nemecek et al., 2015a; West et al., 2014). A LCA study on Australian wheat production showed that N-related greenhouse gas emissions can contribute up to 60–95% of the production system (Wang and Dalal, 2015). Additionally, it was shown that the eco-efficiency of agricultural production strongly depends on the 'correct' amount of N-fertilisers (Brentrup et al., 2004b; Charles et al., 2006; Hülsbergen et al., 2002). Note that due to the system boundaries and non-existing in the FAST, animal husbandry and thus emissions from manure production (enteric CH_4) and storage were not included in our analysis. Impacts from animal husbandry are substantial, but the primary purpose of animal production is to provide meat, milk and other animal products and animal manure is regarded as a waste to be recycled in Swiss agriculture. Consistently with other LCA studies of farming system, a cut-off approach between animal husbandry and crop production was applied.

4.3. Tillage regime

Conservation tillage showed tendentially a reduction in aquatic eutrophication (not significant), but only in the stockless conventional system. As the mineralisation is generally reduced under conservation tillage, this contrasting result was driven by the very low yield and a consequent low N-uptake in O-RT. Conservation tillage reduced also the GWP by about 10% (not significant), independent of the system. As ploughing is an energy-intensive process, this effect was mainly due to the reduced demand for fossil fuels and the reduced field emissions. However, comparing the two tillage regimes within a cropping system, the relatively lower reduction of the GWP by the input groups tillage and field emissions for O-RT is striking. It implies that machines used for reduced tillage are still energy demanding compared with conventional ploughs and thus have a lower potential to save fossil fuels than a no-tillage regime. The lower reduction in field emissions at O-RT was caused by a relatively lower reduction in nitrate leaching, which is the precursor for increased N_2O emissions.

Interestingly, the additional use of glyphosate in the stockless conventional no-tillage system (C-NT) had no visible effect on aquatic ecotoxicity (Fig. 4). This is because glyphosate has in comparison with other pesticides relatively low toxicity levels (according to the CML01 method). Further work is necessary to verify this and to assess the toxicity of glyphosate in comparison to other pesticides (Busse et al., 2001; Fritschi et al., 2015; Tsui and Chu, 2003). Furthermore, the stockless conventional no-tillage system showed a lower ecotoxicity than the stockless conventional system with ploughing, because the no-tillage system had a lower demand for fossil fuels and for agricultural equipment with a subsequent lower demand for raw materials and thereby a reduced emission of toxic pollutants (such as nickel). These results imply that the toxicity of glyphosate has to be compared with the toxic pollutants emitted upstream in a system with ploughing. Furthermore, if fertilisers are applied (mineral and organic), other production factors such as the energy demand for tillage become less relevant (Busse et al., 2001; Fritschi et al., 2015; Tsui and Chu, 2003).

A further process often discussed in the context of conservation tillage, but also of organic farming, is the potential of soils to sequester carbon. In our study, this process was not considered for the GWP for several reasons: According to IPCC (2006), no tillage and manure application are classified as methods that increase the carbon stock of soils. In relation to stockless conventional farming with ploughing but without organic fertilisers, which is set as a reference (in our case C-IT), C-NT, O-IT, and O-RT are said to increase their carbon stock. This would imply a reduction of CO_2 emissions annually (1/20 years) by about 1100, 2200, and 3000 kg CO_2 eq. ha^{-1} yr^{-1}, in C-NT, O-IT, and O-RT respectively. In total, this increase would lead to a 50% lower GWP for the C-NT, and the two organic systems would represent carbon sinks (roughly −800 and −1800 kg CO_2 eq. ha^{-1} yr^{-1} for O-IT and O-RT, respectively). However, there is rising evidence that the potential to sequester carbon via management (in contrast to land use change), especially with non-tillage, is likely overestimated or even not existing in temperate zones (Baker et al., 2007; Dimassi et al., 2014; Hermle et al., 2008; Luo et al., 2010; Powlson and Jenkinson, 1981). In general, the effects of management on the soil carbon dynamics are believed to depend on an equilibrium reaction (soil carbon loss and gain), whose balance, if disturbed, can be regained after decades (Guo and Gifford, 2002). Therefore, the timeframe is fundamental if carbon sequestration is considered in an LCA and is recommended to be at least 10 years for site-specific LCA studies (Goglio et al., 2015). Furthermore, the potential to sequester carbon is nil if the soil carbon equilibrium is in balance. Thus, the function of a carbon sink has a temporal limit, whose magnitude is not known and often not considered at all in calculations (infinite sequestration). Considering the high uncertainties of short-term alteration in the soil carbon stock and the recommended minimum timeframe of 10 years, we decide to not regard soil carbon dynamics in our calculations.

4.4. Cover crops

Generally, cover crops showed only small effects, except for aquatic eutrophication and biodiversity. Aquatic eutrophication was strongly driven by the cover crops, which is in line with previous studies showing their mitigation potential for nitrate leaching (Brandi-Dohrn et al., 1997; Di and Cameron, 2002). It is surprising that the legumes showed the lowest aquatic eutrophication and this contrasts with a number of studies which showed that legumes enhance N leaching (Scherer-Lorenzen et al., 2003; Bouman et al., 2010). However, these publications are focusing on legumes within grassland and grass-clover-ley and not on leguminous cover crops. The observed effect is due to the temporal course of the legumes' N-uptake, which coincides best with the temporal course of the mineralisation in the SALCA model.

On the other hand, cultivation of cover crops led to a higher GWP due to additional management (sowing, mulching) and additional N_2O emissions from the crop residues. This is especially surprising for the legumes because their cultivation is a common measure to benefit from their symbiotic nitrogen fixation and subsequent reduction in N-fertilisation. In the FAST however, all cover crop treatments received the same amount of fertilisation and nitrogen fixation was not taken into account when fertilisation levels were determined. Thus, the potential to reduce the GWP was not tested in the FAST, and therefore the drawbacks of legume cultivation were more pronounced in this analysis. Nevertheless, the yield differences as a consequence of different cover crops could indirectly influence the GWP: Legumes, with the highest yield at O-RT, showed the lowest GWP per CU (not significant).

For biodiversity, the scores of the SALCA model revealed the importance of the presence of cover crops (non-legume, legume and mix) as potential food resources and shelter for indicator species groups such as birds, spiders, carabid beetles, butterflies, wild bees and grasshoppers. For insects feeding on nectar and/or pollen (butterflies and wild bees), scores of legume were lower than of non-legume and mix because the former did not come into flower. In contrast to all other indicator species groups, crop flora (weeds) was not favoured by cover crops as those do not provide suitable habitat conditions, and can even be used to control weeds (Hiltbrunner et al., 2007). For the aggregated biodiversity score, the scores of the eleven indicator groups are weighted taking into account their general species richness and their position in the food web. So crop flora contributed much to the overall biodiversity score and keeps the overall benefit of cover crops low.

In general, it is challenging to gather the effects of cover crops entirely as they have numerous indirect and very diverse effects. Cover crops represent an important carbon input for soils and have the potential to enhance soil organic carbon in the long-term. Thus, cover crops represent a potential management option for carbon sequestration, which could have, if taken into account, affected the results on GWP of our study (Lal, 2004; Poeplau and Don, 2015; Robertson et al., 2000). However, to draw final conclusions on the effect of cover crops on the GWP, both, carbon sequestration and N_2O emissions have to be taken also into account. As shown in our study, cover crops can also increase the greenhouse gas emissions in the field (N_2O) and due to increased machine use (CO_2).

4.5. Leverage points (conclusion)

There are several leverage points for arable farming to improve the environmental performance. Increasing the yield by keeping impacts constant or even decreasing them is a critical task for all production systems but especially for organic systems. Nevertheless, we could identify that one of the most crucial leverage points of arable farming is the N-fertilisation, independent of the system. Therefore, the improvement of the N-efficiency and of N-management is a critical task for improving the environmental performance of arable systems. Reducing the quantities of mineral N-fertilisers and using application techniques that reduce emissions seem to be most effective strategies to reduce many environmental impacts (Bacenetti et al., 2016). Conservation tillage offers a potential strategy to improve the N-efficiency and decrease aquatic eutrophication because this practice can reduce nitrate leaching. However, there is rising evidence that under conservation tillage soil-borne N_2O emissions tend to increase. Such emission should be considered in future studies, even if N_2O emissions are highly variable and not completely understood (Krauss et al., 2016).

Additionally, GWP and aquatic ecotoxicity can be reduced by conservation tillage, as it needs less fossil fuel and a lower equipment demand than intensive tillage. Consequently, the magnitude of both impact categories can be reduced by reducing the energy demand and the number of management steps in general without increasing the agricultural equipment pool. Furthermore, no tillage, but not necessarily reduced tillage, and the cultivation of cover crops have the potential to reduce aquatic eutrophication and increase biodiversity. Generally, it has to be considered that the different cover crops possess different functions, such as nutrient binding, weed control, or promoting beneficial organisms, and that these functions usually cannot be taken on simultaneously by one cover crop.

As the different areas of the agricultural production (farm level) are strongly interrelated, for instance via fodder-animal husbandry-manure-fertiliser, an isolated analysis of the environmental impacts of arable farming might give only an incomplete picture. Therefore, interactions between arable farming and the rest of the farm, in particular, the animal husbandry (manure), should be investigated in future studies (Marton et al., 2016). Furthermore it is fundamental to understand to which extent soils possess carbon sequestration under intensive agriculture and how it is affected by tillage, fertiliser type and crop residues in respect to different time horizons (balance). Subsequently, the improved integration of soil-related processes in LCA studies is fundamental to complete the holistic approach of carbon foot printing and LCA. Finally, as cropland properties and thus the management have a strong spatial variability, testing the interactions between arable cropping strategies and various sites (climatic conditions) could generate important insights into local and site-specific mitigation strategies.

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