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Comparing the environmental performance of mixed and specialised dairy farms: the role of the system level analysed

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ABSTRACT

Mixed crop–livestock systems are often considered more environmental friendly compared to specialised systems, but due to the interactions between different farming activities, it is not trivial to quantify possible benefits. Using life cycle assessment (LCA), we tested different allocation procedures and system expansion through avoided burden to compare the environmental impact of milk from either specialised or mixed dairy production systems (product level). In a second approach, we compared the whole farming systems with additive system expansion, where the functional unit comprised milk, live animals sold for meat production and crops (farm level). On the product level, milk from the mixed farm had higher non-renewable cumulative energy demand, terrestrial ecotoxicity and phosphorus use, but lower aquatic eutrophication N, independently of the allocation method. For all other impact categories, differences were not significant. On the farm level, results were partially reversed. The mixed system had a lower energy demand and potassium use, while phosphorus use was higher. All other differences were not significant on farm level. The different rankings on product and on farm level were caused by the way manure was attributed to the farming activities. In order to avoid allocation, manure management was sub-divided into storage and application processes. Storage was attributed to dairy production, application to dairy production only if applied on grassland or feed crops, and to cash crops when applied to produce these crops. Manure applied on cash crop areas was thus out of the scope of the product approach, and mineral fertilisers that could be saved within the cash crop production were thus not attributed to milk production. We conclude that only system expansion was able to cope with the complexity of mixed farming systems in LCA. Based on our results with modelled farms, mixed farming showed the potential to reduce environmental impacts compared to specialised farming. Nevertheless, due to the complexity of the system regarding farm management and interactions between cropping and livestock activities, only an assessment with real farm data could reveal the actual benefits of such systems.

1. Introduction

Mixed farming systems combine cash crop and livestock production on the same farm. Such systems were very common in the past, but in industrialised countries increasingly specialised agricultural systems emerged (Ryschawy et al., 2012). With a rising concern about the environmental effects of agriculture, mixed farms are currently reconsidered, as they are assumed to be more efficient in nutrient cycling and to foster ecosystem services through an enhanced biodiversity (Lemaire et al., 2014). However, it is not evident to which extent these theoretical advantages are translated into effective environmental benefits. In a life cycle

Abbreviations: aqEN, Aquatic eutrophication N; FADN, Farm Accountancy Data Network; FE, Farm enterprise; FPCM, Fat and protein corrected milk; GWP100, Global warming potential over 100 years; IG, Input group; K use, Potassium use from mineral sources; nrCED, Cumulative energy demand from fossil and nuclear sources; LCA, Life cycle assessment; LCI, Life cycle inventory; LU, Livestock unit; P use, Phosphorus use from mineral sources; terrET, Terrestrial ecotoxicity; SE, System expansion.

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assessment (LCA) of Swiss dairy production, Alig et al. (2011) found no significant differences between specialised and mixed farms per kilogram of milk. Veysset et al. (2014) even calculated a higher N surplus per hectare and higher greenhouse gas emissions on mixed beef and crop farms compared to specialised beef farms in France. Both studies focussed on the livestock product, and did not compare the crop products from the mixed systems to those from specialised crop farms. However, the interactions between crops and livestock have benefits and drawbacks (Bell and Moore, 2012), and thus a focus on just one product category might not reflect the overall effect of mixed farming. For LCA studies, processes with co-products are challenging, as there are different approaches on how to allocate emissions to different outputs. Dairy production is a multi-output process per-se, with the outputs being milk and live animals sold for meat production. Various studies therefore used dairy production to illustrate the influence of different co-product handling methods on LCA results and interpretation (Bartl et al., 2011; Cederberg and Stadig, 2003; Flysjö et al., 2011; Thomassen et al., 2008). They showed that the choice of the co-product handling method has a significant influence on the absolute results, and some even showed that the ranking between production alternatives can change depending on the method chosen (Flysjö et al., 2012; Zehetmeier et al., 2011). All these studies were based on specialised dairy production systems, i.e. systems that only produced milk and meat. If the dairy production happens on a mixed crop–livestock farm, this adds further complexity to the system. The livestock system provides manure for the cropping system, and part of the cropping system produces feed for the livestock system. These interactions might have an influence on the environmental performance of the different products on the farm and the whole farming system. In order to identify the most suitable method to compare mixed and specialised farming systems in LCA, we therefore analysed the effect of different co-product handling methods as well as different system boundaries when comparing specialised and mixed dairy production systems.

2. Methods

In the present study, a specialised and a mixed dairy system were modelled. The LCA was performed on both product and farm level. On the product level, we focussed on milk as the primary product and tested different co-product handling methods between milk and its co-product meat. On the farm level, we considered all products of the farming systems, i.e. milk, meat and crops. The latter approach was more holistic and aimed at including all possible effects of mixed farming systems compared to specialised ones. The focus was put on the ranking of the different systems and not primarily on the absolute results.

2.1. Dairy production systems

Our analysis focussed on dairy production in the Swiss lowlands, where both specialised and mixed dairy farms can be found. In order to get representative farms for the two systems, we modelled the farms based on the average specialised and the average mixed dairy farms as obtained from the Swiss Farm Accountancy Data Network (FADN; Mouron and Schmid, 2011). Table 1 gives an overview about the main characteristics of the simulated farms. The average specialised lowland dairy farm kept 33.1 livestock units (LU) and had an agricultural area of 20.3 ha. Thereof 17.7 ha were grassland, 1.1 ha silage maize and the remaining part was used for other crops. The average mixed lowland dairy farm kept 29.9 LU and had an agricultural area of 26.8 ha, thereof 12.3 ha grassland. The cropping area was used for both, cash and feed crop production. On both farms, the LU consisted mainly of dairy cows and young stock, with some minor quantities of other animal categories, like fattening pigs. For the simulated farms, we presumed that the farms only kept dairy cattle, i.e. dairy cows and young stock. In order to cover the full dairy production cycle, i.e. from the birth of a female dairy calf until the end of the productive life of a dairy cow, we attributed the LU on the farms to the two animal categories young stock and dairy cows, based on a restocking rate of 0.29 cows per year and an age at first calving of 30 months (Boessinger et al., 2013). The mixed farm was assumed to produce less meat per kilogram milk, because the total milk production per cow was higher on this farm type (Mouron and Schmid, 2011). This higher milk yield on mixed farms was achieved with a higher amount of concentrate fed to the cows, which was produced in part on the farm.

2.2. Life cycle assessment

In order to identify a suitable method to compare crop–livestock systems, two different LCA approaches were tested. These were a product approach and a farm approach. The product approach focussed on milk production, while the farm approach integrated all products obtained from the activities in the entire farming system, i.e. milk, live animals for meat production and cash crops.

2.2.1. Goal and scope

The goal of the product approach was to compare the environmental impact of milk production, while the farm approach aimed at comparing the impact of a basket of products generated by the farms. Both, the product and the farm approach, included all environmental impacts from cradle to farm gate. All inputs and outputs of the farm were considered and no cut-off criteria were applied. The farming system itself was sub-divided into two farm enterprises (FE), both with their own system boundaries: dairy and cash crops. The FE dairy produced milk and the co-product meat from culled animals and surplus calves. It included all processes related to the husbandry of dairy cows and young stock, such as direct emissions generated by the animals or the storage of its manure, forage and concentrate feed production on the farm including direct emissions of applied fertilisers and manure, external inputs and infrastructure for keeping the animals. The FE cash crops included all processes related to the production of sold crops, such as external inputs, machinery, and direct emissions from the application of fertilisers and manure (Fig. 1). The system boundaries of the product approach were limited to the FE dairy, while the farm approach included all FE on the farm. Both approaches had their own definition of the functional unit and different methods to cope with multiple outputs from the production systems.

Product approach: The functional unit was 1 kg FPCM at farm gate. As the dairy system had two outputs, milk and live animals for meat production, the environmental impact of the dairy system needed to be allocated between the two products. Previous studies have shown that different allocation methods may influence the results (e.g. Flysjö et al., 2012; Zehetmeier et al., 2011). Therefore, four different co-product handling methods were applied in the present study to evaluate their influence on the result: physical causality allocation, economic allocation, and two system expansion alternatives. We performed physical causality allocation based on the guidelines from the IDF (2010) and economic allocation based on price information from Boessinger et al. (2013). For system expansion, we assumed that the meat derived from the dairy system replaced an equal amount of meat from an alternative production system. The impact of the replaced meat was thus credited to the dairy system (system expansion through avoided burden). As

Table 1
Main characteristics of the simulated specialised and mixed dairy farms.

	Specialised dairy farm	Mixed dairy farm
Total LU	33.1	29.9
Dairy cows (LU)	25.7	23.2
Young stock (LU)	7.4	6.7
Milk yield (kg FPCM sold/cow)	6540	6940
Usable agricultural area (ha)	20.3	26.8
Grassland including leys (ha)	17.7	12.3
Maize, silage and grain (ha)	1.2	3.1
Cereals (ha)	1.0	7.3
Beets and potatoes (ha)	0.1	2.4
Other crops incl. perennials (ha)	0.2	1.7
Sold products		
Milk (kg FPCM)	168,142	161,009
Animals for meat production (kg LW)	6496	5863
Cereals (kg DM)	4547	37,963
Maize (kg DM)	671	2421
Beets and potatoes (kg DM)	1109	31,893
Other crops (kg DM)	1083	4947

DM: dry matter; FPCM: fat and protein corrected milk; LU: Livestock units; LW: live weight.

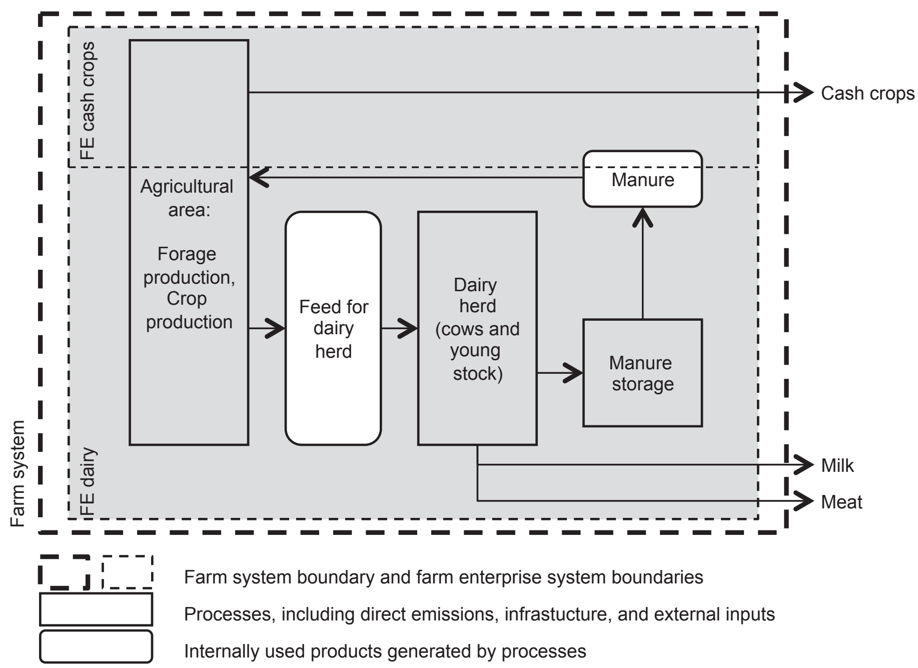

Fig. 1. System boundaries of a dairy farm and its farm enterprises (FE).

already discussed by Flysjö et al. (2011), quality and price of meat from the dairy system are different from those of suckler beef production systems and it is therefore not evident if consumers will actually replace meat from dairy systems with suckler beef. We therefore distinguished between two possible meat alternatives: meat from suckler beef systems and pork. Results obtained without allocation, where all impacts were attributed to milk, were used as a reference to which the influence of the different co-product handling methods was compared.

Farm approach: The farm approach considered all outputs generated by the simulated farms. Therefore, the functional unit was a basket of products containing milk, live animals for meat production, and crops. As the specialised and mixed farming systems compared did not produce these products in the same proportion (Table 1), comparability was achieved through system expansion. This means, if one system produced less crops than the other, this difference was balanced with crops produced on specialised crop farms. For meat, differences were balanced with the corresponding number of animals to produce the same amount of beef or pork. The total production amounts of the systems were divided by the amount of milk produced, resulting in a functional unit of 1 kg FPCM plus the respective amounts of co-products live animals and crops, which were thus equal for both systems (Table 2).

2.2.2. Inventory

For each of our simulated dairy farms, as well as for the specialised crop farm, the suckler beef farm and the pig farm occasionally needed for system expansion, the production within one year was simulated. Economic data and main farm characteristics, such as revenues from sales and production costs, or land use types, were taken from FADN (Mouron and Schmid, 2011), which were complemented with quantitative data on inputs and processes. Accordingly, the livestock system model considered forage

Table 2
Total amounts of milk, live animals sold for meat production and crops included in the functional unit and the amounts contributed by the dairy farms and by system expansion farms.

Contribution of single farms	Specialised system		Mixed system		Functional unit
	Dairy farm	System expansion	Dairy farm	System expansion	
Milk (kg FPCM)	1	–	1	–	1
Live animals (kg LW)[a]	0.0386	–	0.0364	0.0022	0.0386
Cereals (kg DM)	0.0270	0.2087	0.2358	–	0.2358
Maize (kg DM)	0.0040	0.0110	0.0150	–	0.0150
Beets and potatoes (kg DM)	0.0066	0.1915	0.1981	–	0.1981
Other cash crops (kg DM)	0.0064	0.0243	0.0307	–	0.0307

DM: dry matter; FPCM: fat and protein corrected milk; LW: live weight.
[a] For the substitution with pork, the higher ratio between boneless meat and LW compared to beef was considered.

requirements of the animals (Flisch et al., 2009), typical compositions of concentrate mixtures (Boessinger et al., 2013), infrastructure such as animal houses, feed and manure storage containers, manure production and management (Kupper et al., 2013), and requirements for other inputs such as energy (Dux et al., 2009). Simulations of crop and forage production included data on yields (Boessinger et al., 2013; Mack et al., 2013; Mouron and Schmid, 2011), a farm-gate nutrient balance to calculate fertiliser imports, nutrient requirements of the plants and the availability of manure (Flisch et al., 2009), pest management (FOAG, 2012; Nemecek et al., 2005) and machine usage (Boessinger et al., 2013). Wherever required, the models were complemented with inputs from experts or additional sources, e.g. concerning recommendations on crop rotation (Vullioud, 2005).

Out of these models, an inventory of all inputs was extracted and linked to the corresponding processes from ecoinvent v2.2 (ecoinvent Centre, 2010). Direct emissions from farming activities such as storage and spreading of manure, emissions from enteric fermentation of cattle and combustion of fuels were calculated with the method SALCAfarm v3.2 (Swiss Agricultural Life Cycle Assessment for farms; Nemecek et al., 2010). To facilitate a contribution analysis, SALCA assigns the different inputs and direct emissions to eleven input groups (IG): buildings, machinery, energy carriers, fertilisers & field emissions, pesticides, purchased seeds, purchased concentrate, purchased roughage, purchased animals, animal husbandry, and other inputs. The life cycle inventory (LCI) was calculated for both the whole farm, as well as for the different FE.

2.2.3. *Impact assessment*

For comparative LCA, the impact assessment should consider a comprehensive set of impact categories (ISO, 2006). Accordingly, SALCA provides a broad set of impact categories that are relevant for agricultural systems (Nemecek et al., 2010), which were all calculated. From these we selected cumulative energy demand from fossil and nuclear sources (nrCED; Frischknecht et al., 2007a), global warming potential over 100 years (GWP; IPCC, 2007), aquatic eutrophication N (aqEN) according to EDIP2003 (Hauschild and Potting, 2005), and terrestrial ecotoxicity (terrET) according to CML 2001 (Guinée et al., 2001) for further analysis. Other impact categories like ozone formation and depletion, eutrophication P, acidification, aquatic and human toxicity are not shown. They turned out to be closely related to at least one of the above mentioned categories, like for example the ozone formation potential which was related to non-renewable energy resources, as both are linked to the combustion of fuels. Thus, these other categories provided no additional information. Furthermore, we decided to exclude water use from the set of impact categories. In an initial assessment, more than 90% of water use was caused on farm, the remaining part was caused by external inputs. However, the on farm water consumption in the farm simulation was based on a very rough estimate, only considering water consumption within animal husbandry, where variability is very high and depending on various factors such as manure management, dilution factor of the slurry, or even the water source (tap water or collected rainwater). Water consumption linked to crop production, e.g. for irrigation or cleaning, was not simulated. Therefore, the inventory for water consumption was incomplete and too uncertain to provide meaningful results. As a supplement to the basal set of impact categories shown, we show the use of potassium (K use) and phosphorus (P use) from mineral sources based on the LCI. These two categories were added because nutrient use efficiency is often used as an argument in favour of mixed farming systems.

2.2.4. *Uncertainties*

In LCA, there are various sources of uncertainties, which can be categorised into parameter uncertainties, scenario uncertainties and model uncertainties (Huijbregts et al., 2003).

Parameter uncertainties come from the uncertainty of input data of LCI, e.g. uncertainty about the exact amount of slurry spread or energy used or uncertainty due to natural variability in crop yields. As for a majority of the LCI data uncertainty information was lacking, uncertainty was calculated based on the ecoinvent pedigree matrix included in the SALCA tools. This simplified method for quantifying uncertainty presumes a lognormal distribution and uses basic uncertainty factors for different input groups, and further factors for reliability, completeness, temporal, geographic or technological correlation as well as for the sample size to calculate the square of the geometric standard deviation σ_g^2 (Frischknecht et al., 2007b). The σ_g^2 calculated from our data varied from 1.07 for diesel combustion, to 1.11 for land occupation and 1.51 for nitrate leaching, and up to 5.00 for heavy metal emissions. Based on this uncertainty data, a Monte Carlo analysis was performed with the software SimaPro 7.3.3 for the main allocation scenario (physical causality according to IDF, 2010) and for all farm level results. In the present article, we only discussed differences in cases where at least 950 out of 1000 runs were in favour of one of the situations studied. All other differences were considered as not significant.

Scenario uncertainties are due to normative choices such as the method of co-product handling, the definition of the system boundaries or the functional unit (Huijbregts et al., 2003). To study the effects caused by such uncertainties was the main goal of the present study, and it was covered by the different ways of allocation in the product approach and by the two possible meat substitutes considered in the farm level approach. On farm level, the system was not only expanded by alternative meat production systems, but also by alternative crop production systems. We assumed that crops that were not produced on dairy farms will be substituted by crops from specialised crop farms in Switzerland. An alternative way of substituting crops would be to import them from abroad, but this scenario was not considered for two reasons: (1) Differences between the same crops produced in different countries (Bystricky et al., 2014) are not as large as differences between beef and pork production (de Vries and de Boer, 2010), and are therefore not expected to cause significant changes. (2) Our farm modelling system was based on Swiss FADN and therefore only operates with Swiss farms. We did not have access to production data of imported crops of the same quality, i.e. with the same system boundaries and the same models for the calculation of direct emissions. Taking data from another source would therefore have added another level of uncertainty and would not have improved the model.

Model uncertainties in the present study are due to two main reasons: (1) The models used to calculate direct emissions, such as those from fertiliser application or enteric fermentation, as well as models to simulate the whole farming systems are simplified versions of the real world. (2) Characterisation factors in impact assessment methods are based on models with their own inherent uncertainties. This can be illustrated by the methane conversion factor for the global warming potential for a time horizon of 100 years, which was changed from 21 to 25 kg CO_2eq between the IPCC reports from 2001 to 2007, and was corrected to 28 (without inclusion of climate–carbon feedbacks) or 34 (with inclusion of climate–carbon feedbacks) in the most recent IPCC report (Myhre et al., 2013). Such uncertainties not only affect the metrics behind the global warming potential, they might be even higher for other environmental impact methods such as ecotoxicity (Rosenbaum, 2015). Model uncertainties due to the first reason are partially covered in the present study by the pedigree matrix used for parameter uncertainties, e.g. by attributing higher basic uncertainty factors to direct emissions. Furthermore, some model uncertainties that might not be sufficiently covered by the uncertainty factors were discussed in a qualitative way. Model uncertainties due to the second reason were not addressed here.

3. Results

3.1. Product approach

Compared to dairy production on a specialised farm, milk produced on a mixed farm had significantly higher nrCED, terrET and P use, while aqEN was lower (Fig. 2). The different co-product handling methods influenced the absolute values of the impacts. Under allocation, the proportion of impact allocated to milk was the same for all impact categories, i.e. 77.7% for milk from specialised farms and 79.0% for milk from mixed farms in case of physical allocation, and 84.1% and 84.9% in case of economic allocation. Under system expansion, the proportion allocated to milk depended on the impact category. In some cases, namely nrCED, GWP, aqEN, and P use, system expansion with beef resulted in the smallest impacts. For terrET and K use, the situation was different. Here system expansion with pork led to the smallest impacts of milk production. The ranking between dairy production systems was the same under all co-product handling methods studied, except for K use. For this impact category, although not on a significant level, the mixed dairy farm had a lower impact with allocation and system expansion with the avoided burden of beef production, and a slightly higher impact under system expansion with the avoided burden of pork production.

For nrCED, aqEN, terrET and P use, the impact categories significantly differing between specialised and mixed farms. The contribution of the different IG under physical causality allocation is shown in Fig. 3. Compared to milk produced on specialised farms, nrCED was higher for milk from mixed farms. This was mainly caused within IG machinery, IG energy carriers and IG fertilisers & field emissions, all linked to feed production on farm. As the proportion of feed produced on farm was higher in mixed farms, the contribution of the IG purchased concentrates decreased. A similar effect was observed for terrET and P use. Emissions related to on-farm feed production were higher on mixed farms, namely the emissions from IG fertilisers & field emissions, IG pesticides and IG purchased seeds. The contribution of IG purchased concentrates decreased in comparison to the milk production on specialised farms. For the impact category terrET, the differences were caused by the changed feed composition. On mixed farms, a part of the potatoes produced on farm were fed to the animals, and due to the high pesticide usage in potato production, this led to the increase in terrET of dairy production. The aqEN was mainly reduced in IG fertilisers and field emissions, due to different N sources used on mixed farms, with a higher proportion of mineral fertilisers applied on fields designated for feed production.

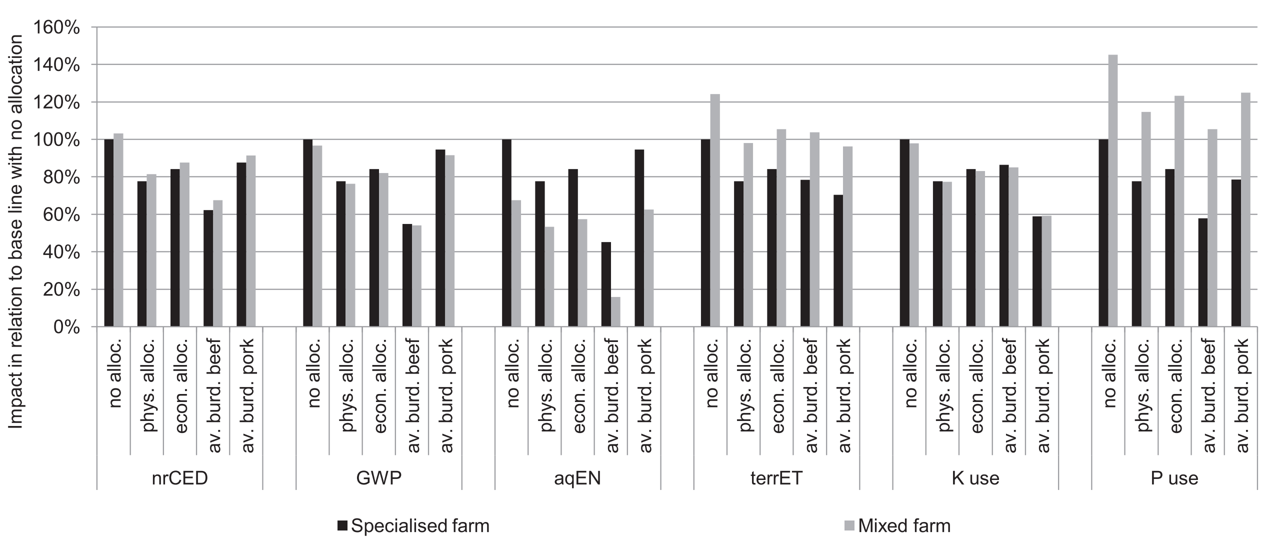

Fig. 2. Comparison of milk production on specialised and mixed farms under the different co-product handling methods (no alloc. = reference with no allocation; phys. alloc. = physical causality allocation; econ. alloc. = economic allocation; av. burd. beef = system expansion with avoided burden from beef production; av. burd. pork = system expansion with avoided burden from pork production) for the impact categories nrCED (cumulative energy demand from fossil and nuclear sources), GWP (global warming potential over 100 years), aqEN (aquatic eutrophication N), terrET (terrestrial ecotoxicity), K use (potassium use), and P use (phosphorus use) applying the product approach.

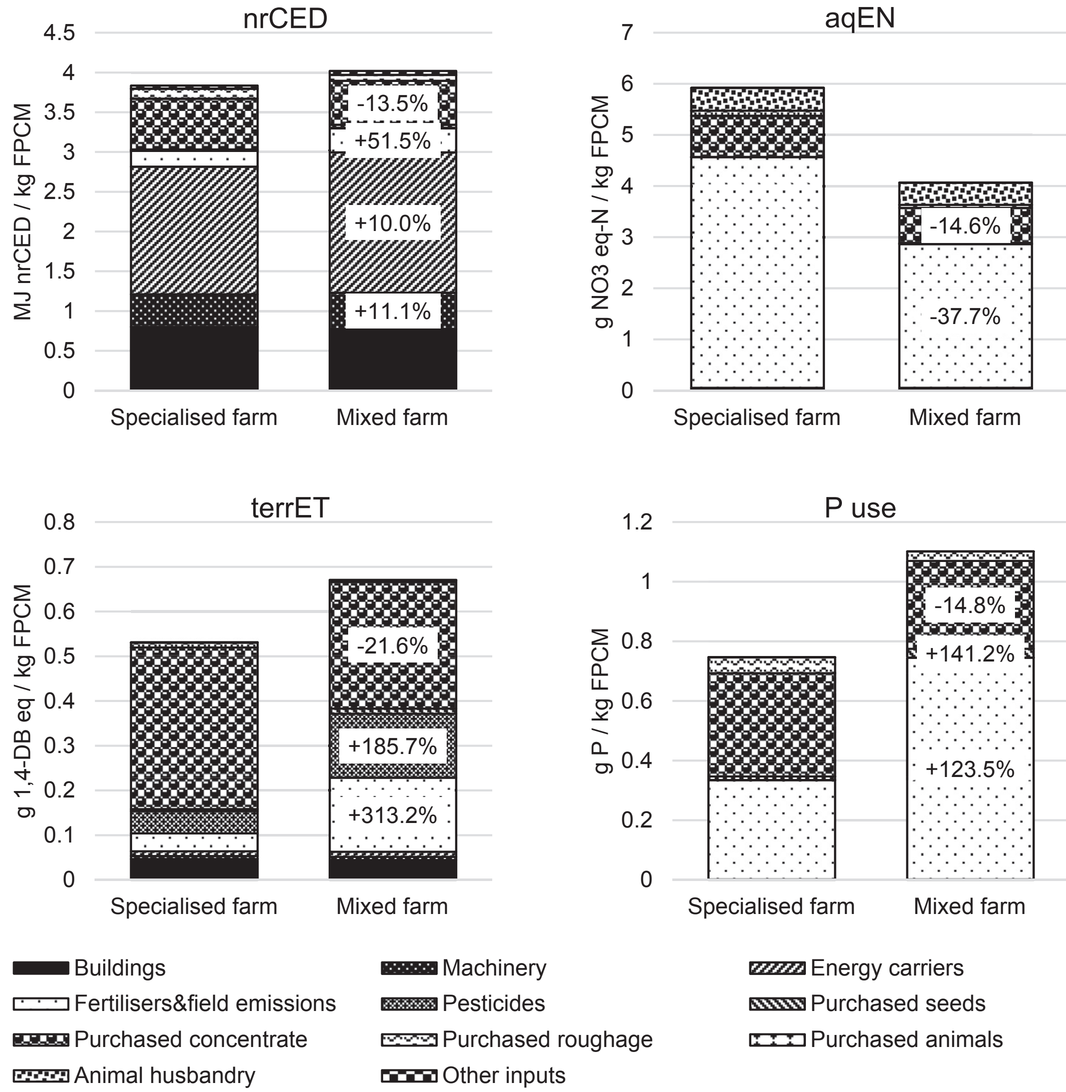

Fig. 3. Contribution analysis of milk production for milk production on specialised and mixed farms for the impact categories nrCED (cumulative energy demand from fossil and nuclear sources), aqEN (aquatic eutrophication N), terrET (terrestrial ecotoxicity), and P use (phosphorus use) per kilogram of FPCM (fat and protein corrected milk) when using physical causality allocation in the product approach.

3.2. Farm approach

Considering all farm outputs, i.e. milk, meat and cash crops, differences between specialised and mixed dairy systems were significant for nrCED, K use, and P use (Fig. 4). The choice of the meat production system for system expansion had no influence on the ranking and only a small influence on the absolute values. For nrCED, GWP, aqEN, and P use, the expansion with suckler beef added more to the mixed production system than the expansion with pork, while the opposite was the case for terrET and K use. Different from the product approach, nrCED was lower on mixed farms and the difference in terrET was no longer significant when applying the farm approach. This was mainly due to reductions within the FE cash crops. These reductions were high enough to more than compensate the increased nrCED of FE dairy and FE suckler beef and compensate terrET from these two FE (Fig. 5). For aqEN there was still a trend for a lower impact from the mixed system that was, due to high uncertainties linked to nitrate leaching, no longer significant. The FE cash crops was also responsible for the significant lower K use within the mixed system compared to the specialised production. Most of the reduction was achieved in IG fertilisers and field emissions. Phosphorus use was also reduced within FE cash crops, but in this case, the reduction was not high enough to compensate for the increased P use within FE dairy cows and the P use from system expansion to compensate the lower meat production on the mixed dairy farm (Fig. 6).

4. Discussion

4.1. Co-product handling

Physical causality allocation, economic allocation and system expansion through avoided burden with either beef or pork were the investigated co-product handling methods on product level. Although the absolute values were strongly influenced by the different methods, the ranking between milk produced on specialised or mixed farms was not affected for most impact categories. The ranking only changed for K use, but this on an insignificant level. Similarly, no influence of the co-product handling method on the ranking among different production alternatives was observed in a study comparing dairy production in Sweden and New Zealand, where allocation and system expansion led to the same ranking

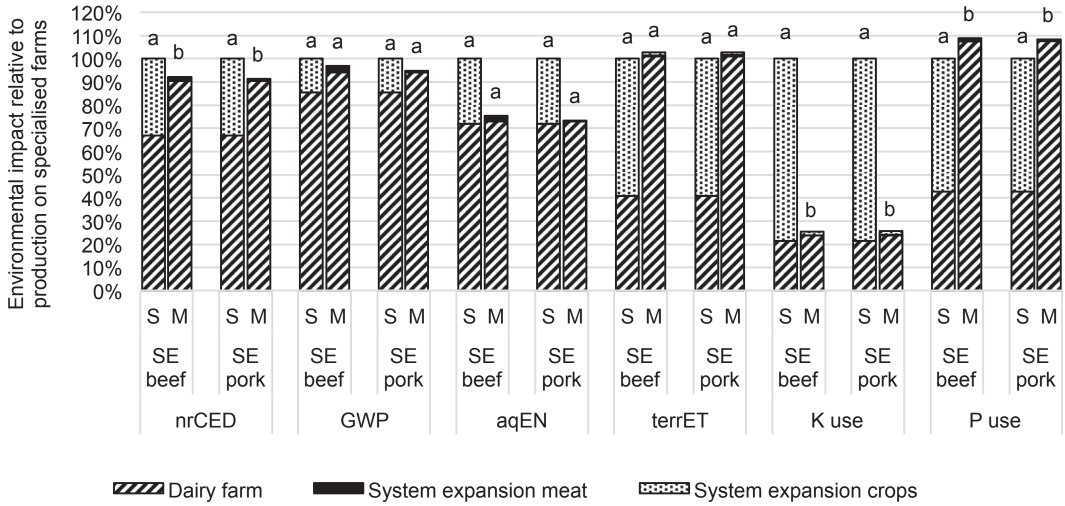

Fig. 4. Comparison of the different dairy production systems and the contribution of the dairy farms and the expansion systems for meat and crops using the farm approach. S = specialised farming system, M = mixed farming system; SE beef = system expansion with beef, SE pork = system expansion with pork; nrCED = cumulative energy demand from fossil and nuclear sources, GWP = global warming potential over 100 years, aqEN = aquatic eutrophication N, terrET = terrestrial ecotoxicity, K use = potassium use, P use = phosphorus use. a,b = differing letters indicate significant differences.

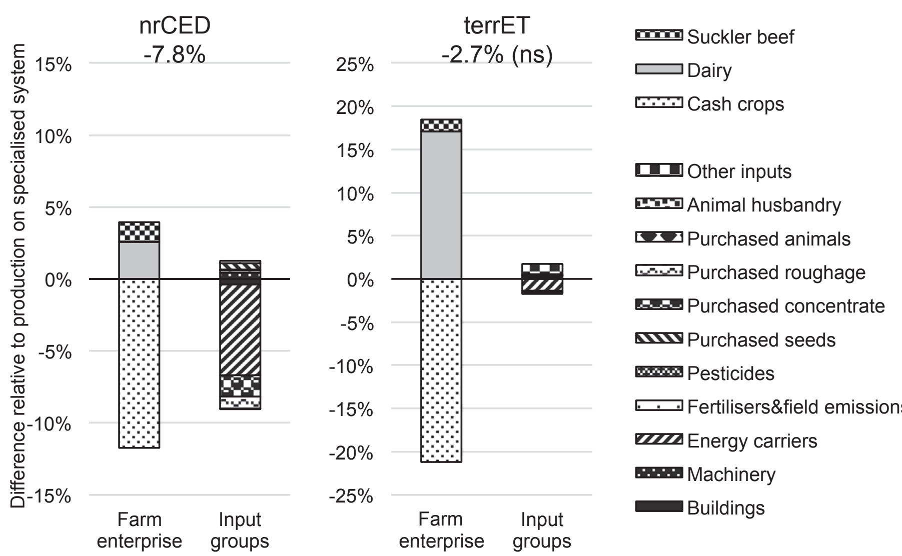

Fig. 5. Comparison of nrCED (cumulative energy demand from fossil and nuclear sources) and terrET (terrestrial ecotoxicity) found in the mixed system compared to the specialised system within the input groups and farm enterprises under the farm approach. Differences were not significant (ns) in the case of terrET.

between the two systems (Flysjö et al., 2011). However, there are cases where the choice of co-product handling methods influenced the ranking. Flysjö et al. (2012) compared organic and conventional dairy production, and found that conventional milk production had a lower environmental impact when no allocation was performed, whereas system expansion resulted in a lower impact for organic milk production. This reversed ranking was mainly caused by the higher meat production per kilogram milk in the organic system and thus a higher avoided burden from beef production. In a study comparing dairy production systems with different milk yields per cow, Zehetmeier et al. (2011) found that performing no allocation and economic allocation resulted in the lowest greenhouse gas emissions for the system with the highest yield, while system expansion led to the opposite conclusion. The latter two studies compared systems with relatively large changes in meat production per unit of milk (17% less meat on the conventional system compared to the organic system; Flysjö et al., 2012; 60% less meat in the high compared to low milk yield systems; Zehetmeier et al., 2011). In our study, the milk-to-meat ratio was also affected by the production system, as the milk yield per cow on mixed farms was higher. However, the difference was small; the mixed farm produced only 6% less meat per kilogram FPCM. This change was too small to have a significant effect on the ranking.

Using the farm approach, where all FE were included, instead of the product approach influenced the ranking of the systems. For terrET and aqEN, the difference between specialised and mixed production systems found with the product approach was no longer significant, and for nrCED the ranking was even reversed. For K use, mixed farms had a significant lower impact, a difference not apparent with the product approach. Bell and Moore (2012) stated that many practices in mixed farming systems can have positive or negative effects on other farming activities, which was clearly the case for nrCED, terrET, P use, and K use on the present mixed farm. For these impact categories a reduction was achieved in FE cash

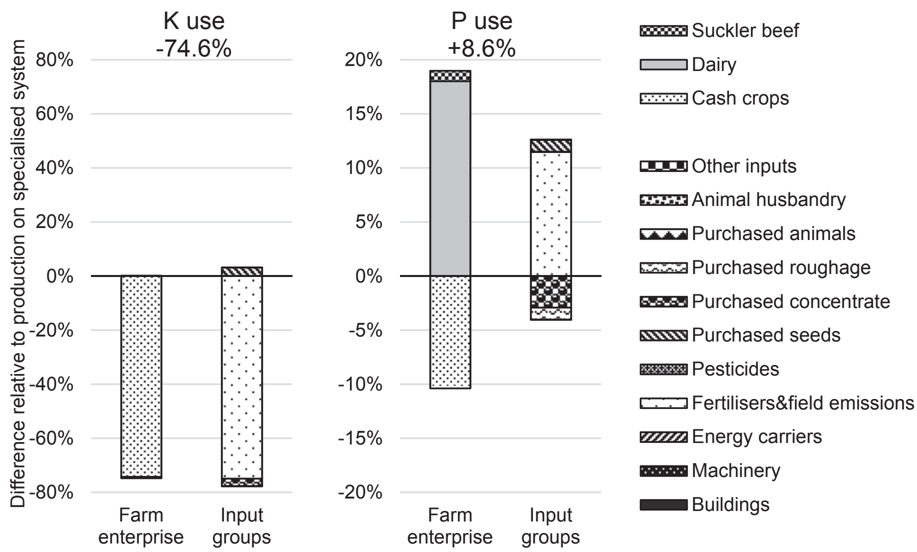

Fig. 6. Difference in K use (potassium use) and P use (phosphorus use) found in the mixed system compared to the specialised system within the input groups and farm enterprises under the farm approach.

crops, and this FE was out of the scope of the product approach. The reason for this lies in our definition of the system boundaries between the FE and the way manure was attributed to the different FE. We sub-divided manure handling into two processes: storage and application. Storage was fully attributed to the animals that produced the manure, while application was attributed to the crop where it was applied and its distinction (feed crops or forage for livestock vs. cash crops). This is the standard procedure within the SALCA method, and is rather commonly used in agricultural LCA, both in the context of dairy (Cederberg and Flysjö, 2004) and crop production (Nemecek and Kägi, 2007; Willmann et al., 2014). As mixed farms produced more crops, a larger share of the total available manure was used for cash crops, and subsequently this manure was no longer available for feed production. This increased the use of mineral fertilisers within feed production, and thus within the FE dairy. On the other side, the manure applied on the cash crops reduced the use of mineral fertilisers within the FE cash crops.

These results show that the division of processes into sub-processes might mask some environmental effects, although it is stated as the first option in the ISO 14044 standard to avoid allocation (ISO, 2006). Therefore, it should only be applied if sub-processes are independent and do not cause any rebound effects, like the ones we observed in the case of manure, where manure applied outside of the dairy system led to an import of mineral fertiliser within the dairy system. The second option recommended by ISO 14044 to avoid allocation is system expansion. For manure, this is also the approach recommended by IDF (2010). Yet, other than for the allocation between meat and milk, where IDF (2010) offers a ready-to-use formula, there is no clear recommendation on how to account for the nutrients exported with manure. It is also not trivial to define the effective amount of mineral fertilisers this exported manure can displace. Firstly, not all nutrients in manure, especially nitrogen, are directly available to the plants. Secondly, direct emissions from manure and mineral fertiliser application differ, and they depend on the application time and technique (Flisch et al., 2009). Dalgaard and Halberg (2007) as well as Weiss and Leip (2012) argue that all extra emissions from the application of manure instead of mineral fertilisers should be attributed to the livestock system, while the displaced mineral fertiliser should be credited. However, to define the amount of displaced mineral fertiliser and the amount of extra emissions properly, the crops where the manure is applied and the application technique should be known. For exported manure, this is rarely the case. Alternatively, direct emissions can be approximated based on national standards as done by Dalgaard and Halberg (2007) as well as Mogensen et al. (2014). The third option recommended by the ISO standard 14044 is allocation. Physical causality allocation is no alternative, because the approaches used either by IDF (2010) or by other LCA attempts are based only on the net energy requirements to produce meat and milk (Basset-Mens et al., 2009; Cederberg and Stadig, 2003). The last option would be an economic allocation. LCA using economic allocation approaches usually do not attribute any or only a very small environmental burden to manure because it is, from an economical point of view, rather considered as waste (Bartl et al., 2011; van der Werf et al., 2009). This also applies to Switzerland, where market prices for manure are zero or even negative (Gerwig, 2008). Therefore, this procedure would not change our results if applied to the product approach. System expansion is thus the only method suitable to consider the effects of manure application outside of the dairy system appropriately.

4.2. Advantage of the farm approach

Taking system expansion on product level one step further and including the effects of manure application on the FE cash crops through an avoided burden approach would generate results with the same ranking as the results from the farm approach. Thus, both approaches would be suitable to identify the most environmental friendly production system. However, the goal of LCA is not only to identify the best solution, but also to identify possibilities for optimisation (Hellweg and Milà i Canals, 2014). In this context, the limitation to the product level makes it difficult to identify optimisation potentials outside of the dairy production system even under system expansion with avoided burden. This can be illustrated by the example of manure. System expansion on a product approach credits the displaced mineral fertiliser to the determining product, in our case milk, but it also attributes all extra emissions from manure application (compared to mineral fertilisation) to milk. As the maintenance of the mass balance is a principle that needs to be respected under system expansion (Weidema and Schmidt, 2010), crops produced with manure from the dairy system are thus treated as if they were produced only with the help of mineral fertilisers. In consequence, if emissions from manure application are reduced through better timing and emission reducing application techniques, these reductions will be attributed to the milk and not to the crops. This might be counter-intuitive for the farmer, as he would expect that such reductions were attributed to the crops where the actual reduction happens. The situation becomes even more contradictory in cases where manure is not used on the same farm but is exported. A crop farmer who imports manure from a dairy farm has no incentive to reduce direct emissions from manure application, as his products will be treated as if they were produced with mineral fertiliser anyway, and his efforts to reduce emissions would only reduce the impact of milk. By contrast, in combining all products within one composite functional unit using additive system expansion, all effects are attributed to the whole production system, and thus all involved parties can profit from an optimisation. In addition, the distinction between different farm enterprises within the whole system gives additional information about possible trade-offs and illustrates hot spots for further optimisation.

4.3. Identifying the best meat substitutes

The scenarios studied did not differ much in the amount of meat produced as a co-product. Thus, the different expansion systems for meat investigated had no significant effect on the ranking in either the product or the farm approach. Nevertheless, in systems with more pronounced changes in the amount of meat produced as a co-product, the choice of the displaced product can be decisive due to the large differences in the environmental impact of different meat production systems. For meat from the dairy system, beef seems to be the most obvious choice, and many studies use suckler beef systems as a substitute for meat produced in dairy production systems (Cederberg and Stadig, 2003; Flysjö et al., 2012; Zehetmeier et al., 2011). In order to account for the different meat qualities and characteristics, Flysjö et al. (2011) recommended using different meat substitutes for different meat types from the dairy system, such as a mix of pork and suckler beef for meat from culled cows, suckler beef for meat from fattened surplus calves and chicken for meat from bobby calves (slaughtered at an age of four days). Another approach was proposed by Weidema (2003), who recommended using market information for the identification of the most appropriate substitute. For Switzerland, domestic production of pork and beef both increased in the last 10 years but, at the same time, the market demand only increased for beef. Veal, which is one of the meat types closely linked to dairy production, had a decrease in both production and demand (Proviande, 2013). Thus, from the domestic market trends, beef currently seems to be the most appropriate substitute, as both supply and demand are increasing. When including meat import into the considerations, pork is appropriate as a substitute as well. In Switzerland, import is regulated through quotas, and increased imports are currently only possible for pork (Proviande, 2013). To identify the most probable substitute or a ratio between them, a more profound market study or a study on preferences of consumers would be necessary. If this is not possible, at least a scenario analysis as the one performed in the present study is required.

4.4. Differences between mixed and specialised farming systems and the limits of farm simulations

In our example of Swiss dairy farms, there was a tendency for lower emissions on mixed farms when using the farm approach. However, for nutrient efficiency, which is one of the presumed advantages of mixed farming (Hendrickson et al., 2008; Ryschawy et al., 2012), results were contradictory. Although there was a trend towards a more efficient use of N on mixed farms, N eutrophication was not significantly lower in mixed farms, with 938 of 1000 Monte Carlo runs in favour of the mixed system it was just slightly below the threshold of 950 runs. For the other two main nutrients, K and P, the differences between the specialised and the mixed production systems were significant, with lower K use and higher P use on mixed farms. This seemingly contrasting result might have been a result of the way we modelled crop production. Nutrient application per hectare was assumed to be the same for both, specialised and mixed farms, as it was based on fertiliser recommendations (Flisch et al., 2009), while yields were based on data from FADN and thus on effective crop yields. As these yields were higher on the specialised farms, this led to a higher P use efficiency on these farms, and thus to an advantage of the specialised system. This higher efficiency might either be an artefact of our simulation, or the result of a possibly better crop management on specialised farms. If the latter is the case, this result would illustrate one of the major challenges of mixed farming systems, namely the skills of the farmer (Bell and Moore, 2012). If managing a mixed dairy farm, the farmer needs to be a generalist, with knowledge about cropping and dairying, while on a specialised farm the farmer can focus on only one activity. Possible benefits from mixed farming systems can therefore only be achieved if the farmer manages to perform livestock and cropping activities on the same level of professionalism as a specialised farmer. The mixed dairy farm in the Swiss lowlands simulated from real data had a slightly lower crop yield than the specialised crop farm, but a higher milk yield per cow compared to the specialised dairy farm. On average, the mixed dairy farms from FADN seem to manage the balancing act between the two activities quite well, but there might be room for improvement within the cropping activity.

Another presumed advantage of mixed farming is a reduced use of pesticides on crops due to improved crop rotation and benefits from ecosystem services (Lemaire et al., 2014). In the present study pesticide use was modelled by crop type independently of the farm type where the crop was grown. Therefore, no difference in application on the two farm types was assumed, which impedes final conclusions about benefits from mixed farming regarding pesticide use. Only an assessment of real farms would reveal the presence or absence of positive and negative side-effects of mixed or specialised farming systems.

5. Conclusion

Mixed farms are complex systems, and optimisation within one enterprise can have negative or positive effects on other enterprises. Performing LCA at the product level was not suitable to cover trade-offs to a full extent in the example analysed in the present study due to the way the manure management process was handled. Although being the prime solution according to ISO 14044, the sub-division of processes should be conducted with care. In our product approach, the sub-division of the manure management process into storage and application processes led to the exclusion of side-effects that were caused by manure application onto cash crops. System expansion was thus the only way to integrate the benefits and trade-offs of manure application outside of the dairy system. A system can be expanded in two ways, either through substitution (avoided burden), applied in the product approach, or additive, applied in the farm approach. The latter turned out to be more holistic and suitable for farmers who intend to identify further optimisation potential. Based on our results from the farm approach applied to the modelled farms, we conclude that mixed farming has the potential to reduce environmental impacts. Certain possible benefits of mixed farming, such as a potentially reduced use of pesticides due to ecosystem services, were even not covered by our models. However, due to the complexity of the system the success depends on the individual skills of the farmer. The question whether the theoretical benefits of mixed farming can be translated into a real advantage over specialised farming needs to be tested with real farm data.

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