# Higher accuracy in N modeling makes a difference

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**ABSTRACT**

Simplified modeling of fertilizer N emissions in inventories of agricultural products may lead to considerable deviations of N emissions from actual N surpluses present in farming system. We compared impacts for global warming, eutrophication, and acidification of four crops either produced in organic or integrated (IP) production using ecoinvent inventories (v2.2). We remodeled $N_2O$, $NO_3$, and $NH_3$ emissions considering N input from fertilizers, N deposition, N fixation, management induced changes in the soil C-N-pool, and N withdrawal by yields. Global warming on a product basis of organic crops was up to 24% lower compared to IP after remodeling. Eutrophication of organic crops was comparable to IP crops and acidification initially up to 3.5 times higher in organic crops was only 1.9 times higher after remodeling. By taking total nitrogen turnover into account in the N modeling impacts of crops from different farming systems can be differentiated with higher accuracy.

Keywords: N emission modeling, fertilization, organic, conventional, total nitrogen turnover

## 1. Introduction

In the past 50 years agricultural intensification has led to a global increase in nitrogen fertilizer use by over 800%, which has led to a massive increase in food production (Foley et al., 2011). On a global scale food production and consumption contribute the highest share of reactive nitrogen compounds released to the environment (Leach *et al.*, 2012). This has environmental impacts on different levels: Nitrogen emissions contribute to global warming, eutrophication, acidification, and negatively impact biodiversity and human health (Galloway *et al.*, 2003). A more sustainable food production should therefore seek to reduce excessive fertilizer use and reduce nitrogen losses to the environment using a set of different strategies (Galloway *et al.*, 2008). One strategy often referred to in the scientific literature is the improvement of the N efficiency in animal and crop production (Galloway *et al.*, 2008; Foley *et al.*, 2011; Sutton *et al.*, 2011).

By prohibiting mineral nitrogen fertilization, organic agriculture is usually lower in its nitrogen intensity than conventional agriculture. In fact often in organic farming nitrogen is a yield limiting factor leading to lower N surpluses per ha. Analyses comparing N fluxes in different farming systems showed a higher N efficiency in organic crop rotations when taking into account the changes in soil organic nitrogen stocks (Küstermann *et al.*, 2010).

However, a recent review of comparative LCAs of organic and conventional products revealed often higher impacts for eutrophication, acidification and global warming for organic products (Meier *et al.*, 2014). A detailed analysis of several LCAs comparing organic with conventional products, though, showed that N emissions from fertilization as modelled in inventories may deviate considerably from the actual N-surplus present in the farming system. It was concluded that higher accuracy in N modeling would better differentiate between products from different farming systems.

A major problem in contemporary N emission modeling in LCA inventories of agricultural products seems to be that emissions models do not represent N emissions from organic fertilizers very well. This is particularly the case for soil born $N_2O$ emissions and $NO_3$ leaching. Soil borne $N_2O$ emissions from fertilizer input in LCA inventories are often modeled using the IPCC model (IPCC, 2006b), which estimates $N_2O$ emissions based on the fertilizer N input only. By doing so the model considers the soil as black box not taking into account N fluxes in and out of the soil. However, especially for organic fertilizers where only a fraction of the total N is readily available for plants microbial processes within the soil are important for their mode of action (Gutser *et al.*, 2005) and in consequence determine their emission pattern. If high fractions of organic fertilizers are applied (as in organic farming were exclusively organic fertilizers are used) the amount of applied total ammoniacal nitrogen (TAN) is usually not enough to meet the crop plants' demand. The additional nitrogen needed for plant growth is supplied by readily available nitrogen mineralized from the soil C-N-pool. A simple $N_2O$ emissions

model, which takes into account N mineralization from as well as N immobilization into the soil C-N-pool, has been proposed by Meier *et al.* (2012).
$NO_3$ leaching strongly depends on local climatic factors (rainfall), soil parameters and vegetation cover. Existing $NO_3$ emission models used in LCA inventories are usually too simplistic to cover the complex interactions between these factors. Accurate nitrate leaching requires the simulation of soil hydrological and biogeochemical processes (Li *et al.*, 2006). However, even with relatively complex models it is difficult to simulate nitrate dynamics accurate enough and a detailed parameterization is needed for satisfying results (Pedersen *et al.*, 2007). In part this is also due to the still poor understanding of the long-term fate of fertilizer-derived nitrogen in the plant-soil-water system (Sebilo *et al.*, 2013). Since $NO_3$ emissions from fertilization seem to be the most difficult ones to model an alternative solution is its calculation from the nitrogen balance by subtracting the modeled $NH_3$ and $N_2O$ emissions from the N surplus. This in turn requires all N inputs and outputs such as fertilizer input, symbiotic N fixation, atmospheric N deposition, management induced changes in the soil C-N-pool and the N withdrawal by the yield. By doing so the law of mass conservation for N fertilization is satisfied, which is also the main requirement for calculating N use efficiencies (Godinot *et al.*, 2014).

The objectives of this study were to assess the chances in impact assessment for global warming, eutrophication, and acidification for four crops produced in different production systems when N emissions from fertilizers were remodeled on the inventory level using a $N_2O$ model that also accounts for N turnover from the soil C-N-pool and adjusting $NO_3$ emissions to the actual N surplus.

## 2. Methods

In our analysis we considered the following ecoinvent v2.2 inventories for four different field crops representing Swiss organic and integrated (representing a less intensive conventional production) agricultural practices (Nemecek *et al.*, 2007):

- Wheat grains organic, at farm/CH
- Wheat grains IP, at farm/CH
- Barley grains organic, at farm/CH
- Barley grains IP, at farm/CH
- Soy beans organic, at farm/CH
- Soy beans IP, at farm/CH
- Potatoes organic, at farm/CH
- Potatoes IP, at farm/CH

Fertilizer based nitrogen inputs and fertilizer based N emissions as $N_2O$, $NH_3$, and $NO_3$ within the product related ecoinvent inventories were transformed to amounts of N input and N emissions per ha using the data in Nemecek *et al.* (2005) and Nemecek *et al.* (2007). In organic, slurry and solid manure were assumed to consist of cattle and pig slurry / solid manure respectively and in IP, of cattle slurry / solid manure respectively as described in Nemecek *et al.* (2005). Dilution of slurry was assumed as 1:1.5 (slurry : water) as in the ecoinvent processes. Based on the same fertilizer N inputs and crop yields assumed in the ecoinvent inventories we recalculated $N_2O$, $NH_3$, and $NO_3$ emissions from nitrogen fertilization as described in the following sections and changed these values in the original inventories.

### 2.1. $NH_3$ emissions

$NH_3$ emissions from slurry and solid manure were determined using an emission factor of 20% of applied total ammoniacal nitrogen (TAN). This factor was determined as the upper bound in a recent Swiss study measuring $NH_3$ emissions from slurry over cropland using improved analytical techniques (Sintermann *et al.*, 2011). TAN values assumed were 0.96 kg/$m^3$ of slurry and 1.32 kg/t of solid manure in the organic inventories and 0.92 kg/$m^3$ of slurry and 0.80 kg/t of solid manure in the IP inventories (Nemecek *et al.*, 2005). $NH_3$ emissions from urea were estimated as 15% and for all other mineral N-fertilizers as 2% from total applied nitrogen (Asman, 1992; ECETOC, 1994). These are the same emission factors as used in the original ecoinvent inventories (Nemecek *et al.*, 2007).

2.2. $N_2O$ emissions

Soil borne $N_2O$ emissions from nitrogen fertilization were estimated using the model by Meier *et al.* (2012), which was developed to better represent the mode of action of organic fertilizers by considering the N-flows in and out of the soil N pool using the IPCC emission factors (IPCC, 2006a). By considering the management induced N turnover processes in the soil the model also integrates nitrogen mineralized from or immobilized in the soil. Deviating from Meier *et al.* (2012) here the amount of management induced nitrogen mineralization from the soil organic N pool was determined by using the SOM-model by Brock *et al.* (2012), in particular by using Equation 1. In combination these models allow for a detailed calculation of the internal N flow based on the N inputs from fertilizers, N fixation, N deposition, and the management induced N turnover in the soil and on the N output by the yield.

$$SON_{MIM} = \frac{N_{PB} - N_{Fix} - N_{Dep} \times NUR_{NDep} - \sum_{i=1}^{n} N_{tot_{Fertilizer\ i}} \times NUR_{NFertilizer\ i}}{NUR_{SONMIM}} + \Delta N_{min} \qquad \text{Eq. 1}$$

| | |
|---|---|
| $SON_{MIM}$ | management induced nitrogen mineralization from the soil organic N pool [kg N]; |
| $N_{PB}$ | nitrogen in plant biomass [kg N]; |
| $N_{Fix}$ | nitrogen derived from the atmosphere by legumes via symbiotic fixation [kg N] (in case of the four crops analyzed only relevant for soy beans); |
| $N_{Dep}$ | nitrogen from atmospheric deposition [kg N]; |
| $NUR_{NDep}$ | nitrogen utilization rate [%] of nitrogen from atmospheric deposition; |
| $NUR_{NFertilizer\ i}$ | nitrogen utilization rate [%] of nitrogen from fertilizer i; |
| $NUR_{SONMIM}$ | nitrogen utilization rate [%] of mineralized nitrogen from the soil; |
| $\Delta N_{min}$ | excessive nitrogen mineralization due to mechanical impact [kg N] (in case of the four crops analyzed only relevant for potatoes). |

Equation 1 estimates the management induced N input from the C-N-pool by calculating the N balance between the amount of N built into plant biomass and the N input from fertilizers, atmospheric deposition, symbiotic N fixation, and excessive N mineralization due to intensive mechanical impact as it is the case in potato cultivation. The fact that only a fraction of the N coming from the various pools can be built into plant biomass is considered by pool specific nitrogen utilization rates (Equation 1).

Nitrogen in wheat biomass (straw and grains) was determined using data from a long term field trial comparing organic with conventional production (Gunst *et al.*, 2013). From this trial different N contents for organic wheat and wheat from integrated production were available. Nitrogen contents for the remaining three crops were taken from Flisch *et al.* (2009) not further differentiating between organic and conventional production.

For soy beans $N_{Fix}$ was assumed to be 150 kg N/ha for integrated production and yield adjusted for organic resulting in 144 kg N/ha.

Atmospheric deposition was assumed to be 25 kg N/ha per year for all crops.

Values for the different nitrogen utilization rates and for the excessive nitrogen mineralization in the case of potatoes (50 kg N/ha for organic and IP) were taken from Brock *et al.* (2012). Since in the case of N from symbiotic fixation 100% of the nitrogen is built into plant the nitrogen utilization rate is 1.

Above and below ground residues were estimated on the basis of the formulas given in the IPCC Guidelines on $N_2O$ emissions from managed soils (IPCC, 2006b).

2.3. $NO_3$ emissions

Emissions of $NO_3$ may occur from the nitrogen surplus during the cultivation period of a crop plant ($NO_3$-$N_{short\ term}$, see Equation 3) and after the cultivation period from the plant residues and excessive nitrogen from mineralization due to mechanical impact ($NO_3$-$N_{long\ term}$, see Equation 4). Total $NO_3$ emissions from the cultivation of a crop plant are obtained by Equation 2.

$$NO_3 - N_{tot} = NO_3 - N_{short\ term} + NO_3 - N_{long\ term} \qquad \text{Eq. 2}$$

$$NO_3 - N_{short\ term} = (\sum_{i=1}^{n}(N_{tot_{Fertilizer\ i}} \times (1 - NUR_{NFertilizer\ i})) + (SON_{MIM} - \Delta N_{min}) \times (1 - NUR_{SONMIM}) + N_{Dep} \times (1 - NUR_{NDep})) - (NH_3 - N) - (N_2O_{direct} - N) \qquad \text{Eq. 3}$$

$$NO_3 - N_{long\ term} = (N_{PR_{ag}} + N_{PR_{bg}} + \Delta N_{min}) \times Frac_{leach} \qquad \text{Eq. 4}$$

| | |
|---|---|
| $NO_3$-$N_{short\ term}$ | amount of N [kg N] potentially being leached from the production of the crop plant; |
| $NO_3$-$N_{short\ term}$ | amount of N [kg N] potentially being leached during the cultivation period of the crop plant; |
| $NO_3$-$N_{long\ term}$ | amount of N [kg N] potentially being leached after the cultivation period of the crop plant by plant; |
| $NH_3$-N | ammonia N [kg N] determined as described in section 2.1; |
| $N_2O_{direct}$-N | N [kg N] from direct $N_2O$ emissions determined as described in section 2.2. |
| $N_{PRag}$ | amount of N [kg N] in above ground plant residues |
| $N_{PRbg}$ | amount of N [kg N] in below ground plant residues |
| $Frac_{leach}$ | fraction of N lost by leaching |

We determined the amount of $NO_3$ leached during the cultivation period of the crop plant from the nitrogen surplus within the respective crop plant system as a proxy for potentially leached N using Equation 3. The nitrogen surplus is determined by the fractions of nitrogen not used by the crop plant from fertilizers, management induced soil-N mineralization (minus the amount of N from excessive mineralization due to mechanical impact ($\Delta N_{min}$)), and deposition. If the amount of nitrogen lost as $NH_3$ and $N_2O$ is subtracted from the surplus, the nitrogen left is basically what can potentially be lost as $NO_3$. For the calculation of the amount of N potentially leaching from above and below ground plant residues and from N from excessive mineralization due to mechanical impact we used the IPCC (2006b) emission factor of 0.3 ($Frac_{leach}$).

2.4 Impact assessment

The remodeled N emissions per hectare from fertilizer applications were transformed to emission values per kg product again using the crop yields and allocation factors assumed in the ecoinvent inventories. Impacts of global warming, eutrophication and acidification were assessed for the former ecoinvent inventories as well as for the inventories with the remodeled N emissions from fertilizer applications. Impact assessment methods used were IPCC GWP 100a (IPCC, 2007) for global warming and EDIP2003 (Hauschild and Potting, 2003) for eutrophication (N only) and acidification. Relative differences between impacts of crops from organic and integrated production were calculated setting IP as the 100% basis.

## 3. Results

For all crops, irrespective of organic or IP production, the amount of N supplied by fertilizers was not sufficient to match the required N content in the plant biomass (Table 1). Thus, in most cases a substantial amount of N had to be supplied by the soil C-N-pool besides the inputs from fertilization, symbiotic fixation, and atmospheric deposition. Especially in the case of IP crops the amount of N mineralized from the soil C-N-pool seems fairly high. However, except for the harvested products all N in above and below ground biomass will be returned to the C-N-pool after harvest. The sum of the N inputs from fertilizers, fixation, deposition and soil C-N-pool always exceeded the N content in total plant biomass (Table 1) because – except for N from symbiotic fixation – never a 100% of the N from the inputs is built into plant biomass. Between 7% (soy beans) and 27% (barley) of the N from the inputs remained as surplus depending on the share of the different N input sources and their nitrogen utilization rates (see Eq. 1). Due to yield differences between the crops from organic production and IP the amount of N in plant biomass was 1.6 to 1.7 times lower in organic crops expect for soy beans where the N content in plant biomass was only 4% lower in organic soy bean which corresponded well with the yield gap of also 4%. For wheat, barley, and potatoes the amount of N mineralized from the soil C-N-pool to meet the plants N demand was approximately 4 to 12 times higher in IP production compared to organic. In contrast for the same crops amounts of N supplied by fertilizers were only 1.2 to 1.5 times higher in IP compared to organic (Table 1).

Table 1. N content in plant biomass vs. N inputs of the four crops analyzed.

| | **Wheat organic** | **Wheat IP** | **Barley organic** | **Barley IP** | **Soy beans organic** | **Soy beans IP** | **Potatoes organic** | **Potatoes IP** |
|---|---|---|---|---|---|---|---|---|
| N content in plant biomass[1] [kg N/ha] | 136 | 230 | 99 | 159 | 220 | 230 | 89 | 137 |
| N input from fertilizers [kg $N_{tot}$/ha] | 122 | 143 | 98 | 121 | 20 | 27 | 84 | 127 |
| N input from fixation [kg N/ha] | 0 | 0 | 0 | 0 | 144 | 150 | 0 | 0 |
| N input from deposition [kg N/ha] | 25 | 25 | 25 | 25 | 25 | 25 | 25 | 25 |
| N mineralized from soil C-N-pool [kg N/ha] (without $\Delta N_{min}$) | 35 | 134 | 13 | 69 | 49 | 48 | 2 | 23 |
| N surplus [kg N/ha][2] | 46 | 72 | 37 | 56 | 18 | 20 | 22 | 38 |
| | | | | | | | | |
| $NH_3$-N remodeled [kg/ha] | 11.0 | 6.4 | 8.9 | 5.8 | 1.2 | 2.1 | 4.1 | 6.4 |
| $N_2O$-$N_{direct}$ remodeled [kg/ha] | 1.4 | 2.2 | 1.1 | 1.7 | 0.7 | 0.7 | 1.0 | 1.5 |
| $N_2O$-$N_{indirect}$ remodeled [kg/ha] | 0.4 | 0.7 | 0.3 | 0.5 | 0.2 | 0.3 | 0.3 | 0.5 |
| $NO_3$-$N_{short\ term}$ remodeled [kg/ha] | 33.5 | 63.4 | 27.2 | 48.3 | 15.5 | 17.4 | 16.7 | 29.8 |
| $NO_3$-$N_{long\ term}$ remodeled [kg/ha] | 9.5 | 18.2 | 6.5 | 11.6 | 13.2 | 13.6 | 21.0 | 22.2 |
| Delta N [kg/ha][3] | 1 | -0.1 | -0.2 | -0.1 | 0 | 0 | 1 | -0.1 |

[1] Total plant biomass including harvested products and above and below ground plant residues.
[2] N in plant biomass minus fertilizer N, minus N from fixation, minus N from deposition, minus N mineralized.
[3] N surplus minus $NH_3$-N, minus $N_2O$-$N_{direct}$, minus $NO_3$-$N_{short\ term}$.

The remodeled N emissions from fertilizer applications on a per ha basis showed the least differences for ‘Wheat IP’ (Table 2). In the case of ‘Barley IP’ $NH_3$ and $N_2O$ emissions also differed only slightly whereas $NO_3$ emission were 1.6 times lower after remodeling. For ‘Potatoes IP’ $NH_3$ emissions were 2.3 times lower for the remodeled emissions whereas $N_2O$ and $NO_3$ emissions changed only little after remodeling. Changes in the inventories of organic crops were generally higher than for the respective IP crops (Table 2).

Table 2. Fertilizer based N emissions per ha from the ecoinvent inventories and the remodeled values for the four crops analyzed.

| | **Wheat organic** | **Wheat IP** | **Barley organic** | **Barley IP** | **Soy beans organic** | **Soy beans IP** | **Potatoes organic** | **Potatoes IP** |
|---|---|---|---|---|---|---|---|---|
| **N emissions in ecoinvent inventories** | | | | | | | | |
| $NH_3$ [kg/ha] | 33.93 | 9.06 | 29.22 | 9.63 | 3.45 | 5.70 | 15.71 | 18.31 |
| $N_2O$ [kg/ha] | 4.93 | 6.07 | 4.69 | 4.29 | 6.96 | 7.34 | 3.40 | 4.07 |
| $NO_3$ [kg/ha] | 400.27 | 331.57 | 406.00 | 425.72 | 194.34 | 200.26 | 292.21 | 261.85 |
| | | | | | | | | |
| **remodeled N emissions** | | | | | | | | |
| $NH_3$ [kg/ha] | 13.41 | 7.79 | 10.78 | 6.99 | 1.41 | 2.55 | 5.03 | 7.80 |
| $N_2O$ [kg/ha] | 2.93 | 4.47 | 2.24 | 3.41 | 1.40 | 1.55 | 2.10 | 3.12 |
| $NO_3$ [kg/ha] | 190.15 | 361.37 | 149.44 | 265.00 | 127.23 | 137.45 | 167.03 | 230.54 |

Comparing the crops from organic with the crops from IP production assessing the original ecoinvent inventories organic wheat and organic soy beans showed lower impacts for global warming (-9%, -12% respectively) than the respective crops from IP production (Figure 1). Impacts for global warming from organic barley and organic potatoes were higher (+17%, +16% respectively) than from IP crops. Impacts from eutrophication were higher for all organic crops (wheat: +108%; barley: +72%, potatoes: +39%) except for soy beans showing 26% lower impacts for eutrophication than IP soy beans. Impacts for acidification were manifold higher in organic cereals than in the respective cereals from IP production (wheat: +271%; barley: +241%). Also organic potatoes had higher impacts for acidification (+32%), whereas organic soy beans showed also lower impacts for acidification (-38%) than IP soy beans (Figure 1).

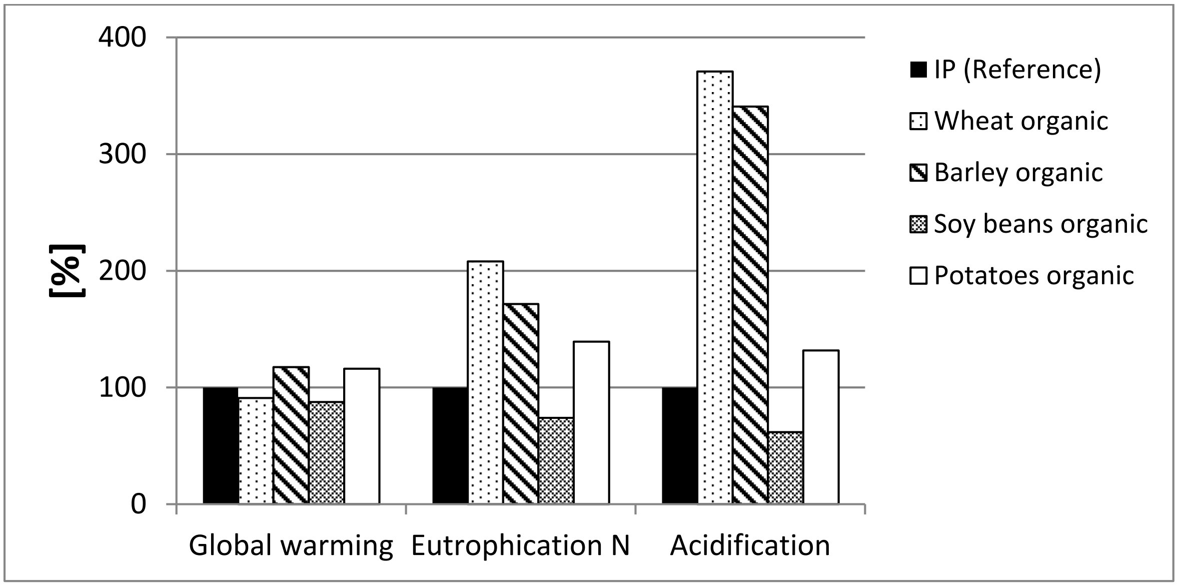


Figure 1. Relative differences between organic crops and the respective crops from IP production for global warming, eutrophication, and acidification per kg of grains, pulses and tubers respectively assessing the original ecoinvent inventories.

Comparing the four organic crops with their counterparts from IP production assessing the inventories with the remodeled N emissions from fertilizer applications all organic crops but potatoes showed lower impacts for global warming (wheat: -24%; barley: -17%; soy beans: -23%) (Figure 2). Impacts for global warming of organic potatoes were 8% higher than for the potatoes from IP production. Again impacts for eutrophication were lower (-40%) for organic soy beans. However, after remodeling N emissions, impacts for eutrophication of the other three organic crops were comparable to the impacts of the respective IP crops (wheat: -4%; barley: +6%; potatoes: +1%). Impacts for acidification of the organic cereals were still clearly higher than for the IP cereals (wheat: +89%; barley: +81%; Figure 2) but the difference became much smaller compared to the original ecoinvent inventories (Figure 1). Also the difference in the impacts for acidification of organic potatoes compared to IP potatoes became smaller in the remodeled inventory (+13%). Finally, impacts for acidification of organic soy beans were also lower (-40%) compared to IP soy beans in the inventory with the remodeled N emissions (Figure 2).

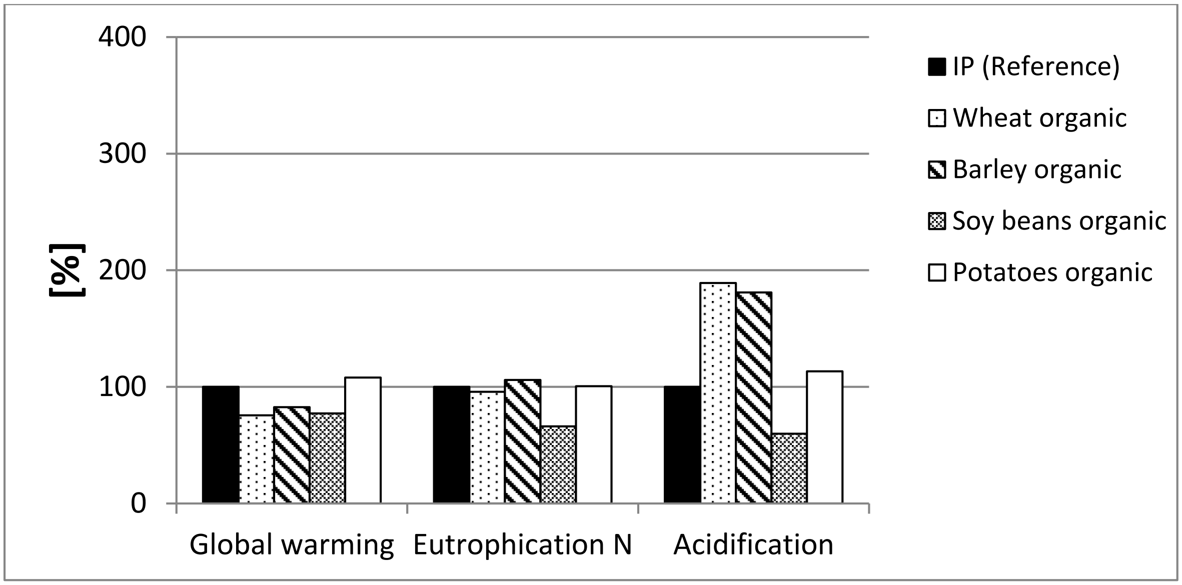


Figure 2. Relative differences between organic crops and the respective crops from IP production for global warming, eutrophication, and acidification per kg of grains, pulses and tubers respectively assessing the inventories with the remodeled N emissions from fertilizer applications.

## 4. Discussion

4.1. N emission modeling

The remodeled $NH_3$ emissions from fertilizer applications on a per ha basis were lower for all crops than the emissions in the original ecoinvent inventories. This can be explained by the different ammonia emission rates from slurry that were used in the estimations of the remodeled values and which are based on recent ammonia emission measurements from slurry application in Switzerland (Sintermann *et al.*, 2011). However, reductions in $NH_3$ emissions were always higher in organic crops after remodeling. This is because in organic crop production nitrogen is solely applied as organic fertilizers (slurry and solid manure for the crops analyzed in this study) having higher $NH_3$ emissions than mineral fertilizers whereas the crops in integrated production are mainly fertilized by mineral fertilizers.

After remodeling, $N_2O$ emissions became less on a per ha basis for all organic crops except for potatoes. For the IP crops $N_2O$ emissions increased after remodeling except for soy beans where the same decrease could be observed as for organic soy beans. For soy beans similar amounts of nitrogen fertilizers were applied, which explains the similar amounts of $N_2O$ emissions. Further, in soy beans the greatest difference between $N_2O$ emission values in the original ecoinvent inventory and the remodeled values could be observed. This can be explained by the modeling of the $N_2O$ emissions according to IPCC guidelines 1996 in the original ecoinvent inventory (Nemecek *et al.*, 2007) where the amount of N from fixation was added to the amount of N from fertilizers. The higher $N_2O$ emissions for organic potatoes after remodeling is due to the additional N from excessive nitrogen mineralization due to mechanical impact, which the model used here accounts for. The higher $N_2O$ emissions after remodeling in wheat, barley, and potatoes from integrated production can be explained by inclusion of the management induced N mineralization from the soil C-N-pool by which the full N turnover in the modeling of $N_2O$ emissions is considered. Relative to the N input from fertilizers the amount of N from the soil C-N-pool was larger in wheat, barley, and potatoes from integrated production than in the respective organic crops. This also explains why the differences in $N_2O$ emissions between these organic and the respective IP crops became larger after remodeling.

Since $NO_3$ emissions are difficult to model reliably we estimated the total amount of leachable N on the basis of the nitrogen balance which resulted in lower $NO_3$ emissions per ha for all crops except potatoes irrespective of organic or IP. This indicates that in the original ecoinvent inventories $NO_3$ emissions are overestimated. Especially for organic fertilizers the $NO_3$ model used within the original ecoinvent inventories seems to produce values that are too high. In contrast the higher $NO_3$ emissions in potatoes after remodeling can again be explained by the additional N from excessive nitrogen mineralization due to mechanical impact which was considered for the remodeled values.

4.2 Impact assessment

In absolute terms impacts for global warming, eutrophication, and acidification became lower after remodeling for all crops irrespective of organic or IP with the exception of impacts for eutrophication in IP wheat resulting in higher absolute impacts after remodeling. The lower relative impacts for global warming per kg of organic wheat, barley and soy beans compared to the respective IP crops after assessing the inventories with the remodeled N emissions from fertilization can be explained by the considerably lower $N_2O$ emissions. In the case of potatoes the relative difference in impacts for global warming between organic and IP potatoes became smaller after remodeling with still slightly higher impacts for global warming in organic.

Assessing the impacts for eutrophication on the amount of leachable N estimated on the basis of the N balance shows that impacts for eutrophication on a per kg of product basis of organic wheat, barley, and potatoes is comparable to the impacts for eutrophication of the corresponding crops from integrated production. Impacts for eutrophication of 1 kg of soy beans were already lower in the original ecoinvent inventories. However, the relative difference between organic and IP soy beans increased after remodeling because the relative difference of $NO_3$ emissions on the per ha basis increased.

Impacts for acidification on a per kg of product basis of organic crops decreased substantially when assessing the inventories with the remodeled N emissions. However, especially for organic wheat and barley impacts for acidification were still considerably higher than for the respective crops from integrated production. This is due to the sole use of organic fertilizers in organic crop production. Even after remodeling $NH_3$ emissions from organic wheat and barley were higher on a per ha basis than from the IP crops. Though, with improved slurry application techniques and management practices $NH_3$ emissions from organic fertilizers could be further reduced resulting in a further decrease in impacts for acidification from organic crops.

## 5. Conclusion

This analysis shows that N emissions from fertilization in LCA inventories of organic and conventional agricultural products are not well adapted to the respective farming system. In particular the crop specific changes in the soil C-N-pool need to be considered in the emission modeling to better represent $N_2O$ emission from organic fertilizers and to take account of the total nitrogen turnover from crop production.

Calculating N emissions based on the actual N flow in the respective farming system leads to a more precise differentiation of impacts for global warming, eutrophication, and acidification between organic and conventional products. By that higher accuracy in N modeling from fertilization helps to improve the differentiation of environmental impacts of products from different agricultural systems such as organic and conventional agriculture and allows for conclusions on their N efficiency.

## 6. Acknowledgements

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