Environmental life cycle assessment of cereal and bread production in Norway

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We assessed the environmental cost of producing bread, as delivered to the consumer, assuming the use of Norwegian ingredients only. Ten impact categories, including global warming potential (GWP), were quantified by mixed modelling and life cycle assessment (LCA). Firstly, we quantified the impacts of growing barley, oats, winter and spring wheat on 93 farms that were representative of the main cereal production regions in Norway. We used wide system boundaries, which included all relevant processes occurring both pre-farm and on-farm. Secondly, we assessed a representative production chain for bread, including transport, milling, baking and packing processes. On-farm processes accounted for most of the environmental impact attributable to the production of bread (e.g. 66 % for GWP). There is thus considerable potential for environmental improvements through changes in farm management. In total, the GWP per kg of bread (freshweight) was 0.95 kg CO_2-equivalent. The environmental footprint of transport was small.

Keywords: acidification; carbon stock change; eutrophication; global warming potential; regional variation

1. Introduction

Understanding the environmental impacts associated with our food production and consumption is a prerequisite for identifying pathways towards a sustainable future. The development of sound and efficient future policies for both greenhouse gas (GHG) mitigation and other environmental issues, such as eutrophication, acidification and toxic emissions, requires a solid understanding of the impacts associated with our current activities. Evaluating the environmental footprint of agriculture is, however, a challenge since production is performed under very diverging conditions. Soil type, climate and topography may vary greatly both between regions and between farms within the same region and differences in management and choice of crops and rotations add to the variation.

The traditional way to address environmental challenges in agriculture has been to focus on a single process, nutrient or pollutant. However, this approach often results in the alleviation of one environmental problem whilst creating another. In order to consider the overall environmental impacts of a certain food production system, it is recommended to include the whole production chain and quantify the various environmental impacts per unit produced. Life cycle analysis (LCA) is so far the most developed/well adapted product-oriented assessment method for this purpose (Halberg et al. 2005).

Some LCA studies have been published on the environmental impact of grain production, particularly on that of wheat for bread production (e.g. Brentrup et al. 2004; Charles et al. 2006; Berry et al. 2008; Pelletier et al. 2008; Berry et al. 2010; Williams et al. 2010; Tuomisto et al. 2012), and somewhat fewer on that of cereals produced mainly for feed concentrates (e.g. Flysjö et al. 2008; Usva et al. 2009). Comparing results obtained in different studies is, however, not easy. In a recent case study on cereal production in Eastern Norway, we found that differences in system boundaries explained a large part of the observed differences between LCA studies in terms of environmental impacts (Roer et al. 2012). One conclusion of our work (ibid), was that many studies exclude such impacts as the manufacturing of machinery, buildings, net changes in soil organic matter, production and use of pesticides and NO_X loss due to the use of mineral fertilizer. However, all of these activities make significant environmental impacts and should thus be included in the analyses.

Bread has an important position in our diet, but the environmental impact of its production has been little focussed upon, particularly under Nordic conditions. The studies of Andersson & Ohlsson (1999) and Grönroos et al. (2006) represent two exceptions. Considering the continuous changes that occur within the agricultural sector, resulting from farmers striving to increase their production efficiency and thereby their income, a LCA, or any environmental study for that matter, should only be considered valid for a period of just a few years.

The objective of this study was two-fold: The first objective was to assess the environmental impacts from the production of barley, oats, winter and spring wheat on 93 farms (from cradle to farm gate) that represented the main regions for cereal production in Norway. This assessment should include all pre-farm processes and farm activities related to conventional grain cultivation, including those that have rarely been considered previously (as mentioned above). The second objective was to perform an environmental assessment of the production chain for a loaf of bread, from whole grain at the farm gate to its point of sale to the consumer. This assessment included transport, milling, baking and packing processes.

2. Material and methods

2.1 Studied objects

In the first part of this study we assessed the environmental impact associated with the production of cereals in the main cereal production areas in Norway, using a selection of the farms presented by Bonesmo et al. (2012). Focussing on GHG emissions intensities and gross margins at the farm level, the latter authors used data from the Norwegian Farm Accountancy Survey (NILF, 2009) and, further, they had access to farm-specific soil and weather data. From this data set, which included agronomic and economic data collected annually from about 1000 farms, Bonesmo et al. (2012) selected 95 farms from the 2008 survey, all of them without livestock. These 95 farms formed our starting point. Since our focus was on conventional cereal production, we disregarded two organic farms (without use of inorganic fertilizer). Assessing all the cereal crops (barley, oats, winter wheat and spring wheat) on the remaining 93 farms, gave us a total of 215 inventories to compile.

From the original data, we used the given farm sizes, crop distribution and tillage strategies. In the present study we wished to reflect the situation with greater agronomic precision than that obtained by using the mainly economic-based data, and with a longer perspective than one year only. Hence, data on fertilizer and pesticide inputs were exchanged with data obtained through detailed interviews with local advisory services (Norwegian Agricultural Extension Service), and supplemented with information on buildings, machinery and equipment, as presented in Korsaeth et al. (2013). The original yield data were exchanged with six-year yield averages (2005-2010) at the respective municipality level, obtained from Statistics Norway, for each crop and farm. The assessment covers all processes involved in cereal production and in the production of relevant inputs (from cradle to farm gate), including more underlying/background processes than those commonly reported in previous studies, such as production of machinery and buildings, use of pesticides, changes in the SOC pool (i.e. net humus mineralization) and NO_X loss from use of mineral fertilizer. The functional unit (FU) in this part of the assessment was one kg grain (with 15% water) delivered at the farm gate.

The second part of this study assesses the environmental impact associated with the production chain from farm gate to the consumer for one kg bread (fresh weight), including transport, milling, baking and packing processes. The bread type studied is a typical industrially produced bread sold in Norway.

2.2 Methodology and assumptions

All calculations were performed using Matlab (version R2009b).

Data for the production of various inputs (such as agricultural implements, tractors, lime, pesticides, transportation and the phosphorus and potassium part of the NPK fertilizer) were taken from the LCA-database Ecoinvent (Nemecek et al. 2004). For the production of buildings and grain dryers, the input output database EXIOPOL (2011) was used.

Environmental impacts from the nitrogen component of fertilizer production were included in the inventory and calculated based on Best Available Technique (EFMA 2000; Yara 2011; Davis & Haglund 1999; Nemecek et al. 2004) depending on the specific fertilizers used. Seeds were accounted for by subtracting the amount of seeds used from the grain yield and adding necessary transport and pesticide use.

Basic information on buildings, machinery and management practices on typical grain-producing farms were obtained through detailed interviews with the local advisory services (Norwegian Agricultural Extension Service) in three of the main producing areas in Norway (Central Norway and northern and southern parts of Eastern Norway). Within these regions, conventional cereal production is performed fairly similarly, in terms of management practices, with only minor differences between regions. As a general management regime, we included the following field work processes in our inventory: ploughing, levelling with simultaneous stone picking, harrowing, combined sowing and initial fertilization, rolling, first spraying (herbicides and insecticides), split fertilization, second spraying (fungicides and growth regulation), combine-harvesting (including chopping of straw), spraying against couch grass in autumn after harvest (every third year), liming (every 8^{th} year), and drying of the grain to a moisture content of 15%.

The annual lime requirement was calculated using general Norwegian recommendations. Only gross data for wheat delivery exist in the databases of Statistics Norway. To split between spring and winter wheat yields, we used a method presented by Korsaeth & Rafoss (2009), which utilizes data from series of long-term Norwegian field trials. General levels of water content in grains at harvest were given by the local advisory services. Some key parameters of the inventories are shown in Table 1.

The CO_2-emissions included in the foreground system (i.e. on-farm) were direct emissions from liming, CO_2-emissions from diesel consumption attributed to field operations, and changes in soil organic C (SOC) as a result of soil management. The average annual CO_2-emissions from lime application were calculated as if the lime was added each year, which is in accordance with guidelines given by the IPCC (2006). The diesel requirement for all field-work processes was calculated through a stepwise procedure as described by Roer et al. (2012), taking into consideration tractor size and horse-power, man-hours needed (based on the Danish “DRIFT” model; Nielsen & Sørensen, s.a.), and work load. The consumption of lubrication oil was set proportional to the diesel consumption, as 0.62% thereof (ibid).

Changes in soil organic C were simulated using the ICBM model (Andrén et al. 2004), where we selected the change in the 30^{th} year as a proxy to reflect the fact that the soil carbon loss gradually declines over time in continuous arable cropping systems on soils with a prehistory of mixed cropping (Riley & Bakkegard 2006). Such a transition in Norwegian cereal production has been ongoing for the last 60 years (Bonesmo et al. 2012).

The model requires data on initial SOC, annual C-input and a daily farm-specific decomposer activity factor (r_e), which adjusts the decay rates of the two soil C compartments considered in the ICBM model. The decomposer activity factor is a multiplicative index describing the relative effects of soil moisture (r_W), soil temperature (r_T) and a cultivation factor (r_C). We ran the ICBM model with the same initial C stocks and r_W x r_T products as those used by Bonesmo et al. (2012). The cultivation factor r_C was set to 1 regardless of tillage, due to the lack of clear evidence for any tillage effect on SOC decay (T. Kätterer, pers. com.), and default values (Andrén et al. 2004) were used for all rate constants. Carbon input through crop residues (straw) and roots was calculated in accordance with Andrén et al. (2004), using municipality-specific crop yields as input. Straw removal reduces C input to soil, and greatly alters soil C stock change. Information about straw removal on the farms was not available, but, in order to highlight the effect of straw treatment on SOC change, we ran the model with two scenarios; either with all straw incorporated into the soil (no removal, case A), or with all straw removed (case B).

Emissions of N_2O and conversion into CO_2-equivalents were estimated by the IPCC (2006) framework, which comprises estimates for both direct emissions and two pathways of indirect emissions. Direct N_2O emissions were calculated as 1 % of the total N additions (mineral N fertilizer, N in crop residues and N mineralization associated with loss of SOC, assuming a C:N ratio of 10), without any correction for soil moisture and temperature conditions. The first indirect pathway for N_2O emissions was the volatilization of N as NH_3 and oxides of N (NO_x), and the deposition of these gases and their products NH_4^+ and NO_3^- onto soils and the surface of lakes and other waters. It was assumed that 10 % of the N applied as mineral fertilizer was volatilized (as NH_3 and NO_x), and that 1 % of the volatilized (and re-deposited) N would be emitted as N_2O-N (IPCC 2006). The second indirect pathway was the leaching of N, as some of this N may be nitrified or denitrified in the groundwater, in riparian zones, in ditches, streams and rivers and in estuaries (and their sediments). In accordance with IPCC (2006), we assumed that 0.75 % of the leached N was lost as N_2O-N.

In the ICCP (2006) framework, N leaching is estimated as a fraction ($Nfrac_{LEACH}$) of the total N input of a system. In this study, we used the method designed by Bechmann et al. (2012) to estimate $Nfrac_{LEACH}$ under specific Norwegian conditions, based on long-term monitoring data from agricultural catchments, combined with farm-specific adjustments for runoff (i.e. the difference between annual precipitation and evapotranspiration). Using this approach, we first selected the most representative catchment available from the Agricultural Environmental monitoring program (JOVA) (ibid) for each farm, considering both the dominant production type and the soil type within the catchment. Next we obtained the catchment-specific data on both $Frac_{LEACH}$ ($Frac_{LEACH\ catchment}$) and runoff ($R_{catchment}$). Farm-specific runoff ($R_{farm-specific}$) was found by taking the closest point in a dataset consisting of 1 x 1 km grid values on long-term (1961-1990) annual average runoff, provided by the Norwegian Water Resources and Energy Directorate (2012). Finally, farm-specific $Frac_{LEACH}$ ($Frac_{LEACH\ farm-specific}$) was calculated as:

$$Frac_{LEACH\ farm-specific} = Frac_{LEACH\ catchment} \times R_{farm-specific} / R_{catchment} \quad (1)$$

N leaching was then calculated as the product of N input via fertilizer and $Frac_{LEACH\ farm-specific}$ (in contrast to the ICPP approach, N from soil mineralization is considered only indirectly in the method of Bechmann et al. 2012).

Estimates of soil and phosphorus losses through drainage and surface water were based on data from the JOVA monitoring programme (Bioforsk 2010). For farms located in the southern part of Eastern Norway, we used data from the Skuterud catchment directly (annual mean for the period 1993-2009). Data from the Hotran catchment (annual mean for the period 1992-2009) was used for farms located in Central Norway, but the P-losses were set to 30% of those measured, in order to account for unusually high values in the catchment, probably caused by gully erosion observed along the river channel. For farms in the northern part of Eastern Norway, we calculated mean values from two data sources on P-losses: the Bye catchment (JOVA) and a long-term field experiment at Apelsvoll research centre near Kapp (Korsaeth 2012), using the annual average for the period 2000-2009 at both locations.

The acidifying compounds included (on farm) in this work were NO_x from diesel consumption and volatilized NH_3 and NO_x from fertilizer. Emissions of NO_x from diesel consumption were estimated on the basis of Li et al. (2006). The sum of volatilized NH_3-N and NO_x-N from fertilizer application was calculated following the IPCC framework described above, and to separate between the two, the proportion of NH_3 volatilizing from fertilizer was set to 2 % (Bouwman et al., 1997), the rest being NO_x.

Data on milling were based on Cederberg et al. (2008), whereas baking and packing data were based on actual industry data from a Norwegian bakery (withheld from public access). The bread consisted of 35 % water, 50 % wheat, 9 % rye, 4 % oats and 2 % other ingredients. All cereals were assumed to be produced in Norway. For wheat, we assumed a 50/50 mixture of winter and spring wheat. In our calculation, we substituted rye with wheat, since rye was not included in the farm inventories. The post-farm transport was estimated using the assumption that the cereals were produced in

Eastern Norway and that milling, baking and consumption occurred in Western Norway. The distances used were 80 km by truck and 690 km by boat from farm to mill, 45 km by truck from mill to bakery, and 50 km from bakery to shops.

For life cycle impact assessment, the ReCiPe method was used (Goedkoop 2011), and 10 categories were selected based on their relevance: Global warming potential (GWP), agricultural land use (ALU), freshwater eutrophication (FE), marine eutrophication (ME), freshwater ecotoxicity (FET), terrestrial acidification (TA), fossil fuel depletion (FD), human toxicity (HT), marine ecotoxicity (MET) and terrestrial ecotoxicity (TET). For pesticides not included in ReCiPe, the USES-LCA model (van Zelm et al. 2009) was used to develop characterization factors.

When the straw was not incorporated, it was regarded as a product, and the environmental impacts were allocated between grain and straw using their monetary value (2010 prices). The price ratios (grain 85% DM:straw DM) used were thus 4.3, 3.9, 5.0, 5.0 for barley, oats, spring wheat and winter wheat, respectively.

3. Results

3.1 Cradle to farm-gate

The environmental impacts related to cereal production up to the farm-gate are shown for all selected impact categories and for each crop in Table 2. The impacts are expressed either per tonne of grain, with the straw incorporated (Case A), or per tonne of grain and straw, with the straw baled and removed (Case B), using economic allocations to distribute the impact between the two products.

There were clear differences between the crops in all impact categories. These were largest for HT and the eco-toxicity categories (FET, MET and TET), and least for ME and TA. Barley was the crop with the highest impact in six of the ten categories

(Table 2, case A). The average GWP's for the four cereal crops were in the range of 879-997 kg CO_2-equivalent (CO_2-eq) per tonne grain, and there was a slight increase when the straw was assumed removed. Spring wheat had the largest GWP of the four crops, barley and oats had on average about 3 % less, whereas winter wheat showed a GWP of about 12 % below that of spring wheat. Winter wheat also showed a different pattern than the other cereal crops, with respect to their cumulative distribution functions of GHG emissions (Fig. 1). The variation in GWP was smaller for winter wheat, illustrated by a higher minimum and a lower maximum value, and thus a steeper form of the cumulative distribution curve.

When the straw was assumed to have been removed, all impacts were reduced for the cereals, except for GWP which increased slightly (Table 2, case B). The relative reductions were almost the same for all impact categories (GWP excluded), reflecting the allocation of impact between grain and straw based on their price ratio.

Each of the impact categories were grouped into pre-farm processes related to the manufacturing of machines and buildings (Machinery and buildings), fertilizer, pesticides and other inputs needed for cereal production (Inputs), along with on-farm emissions related to driving (On-farm driving), field emissions (Field emissions) and emissions related to drying the grain after harvest (Drying) (Fig. 2). Field emissions accounted for more than 50 % of the total impact for GWP, ALU, FE, ME, TA and TET. The other dominant process-group was machinery and buildings, which accounted for the largest parts of FET, HT and MET.

Changes in the SOC pool had a great impact on the field emissions, as the resulting CO_2-eq losses amounted to 46 % of the total field emissions (Fig. 3). The emissions of CO_2-eq originating from other sources than SOC, were mainly in the form of N_2O. Emissions of CH_4 were negligible.

3.2 Farm-gate to point of sale

The environmental burdens of the post-farm processes milling, baking, packing and transport were calculated for each of the ten selected impact categories (Fig. 4). Packing was the major source of emission for half of the impact categories (ALU, FET, FE, HT and ME), particularly for ALU and ME, where it accounted for 93 and 67 %, respectively. The baking process caused the largest emissions for GWP, FD and TET, whereas transport was the most important source for TA, as milling was for MET.

3.3 Cradle to point of sale

When considering the entire production chain from cradle to consumer, the processes occurring on-farm appeared to be the largest source of emissions for all impact categories (Fig. 5). This was most pronounced for ALU, FE, ME and TET, and least for FD. On-farm processes accounted for 66 % of the GWP attributed to the production of bread based on grains produced in Norway. The impact from pre-farm processes did not exceed 17 % of any of the totals, whereas the proportions of post-farm impacts fluctuated more. Post-farm processes were the second most important source for half of the impact categories (GWP, FET, FD, HT and MET).

4. Discussion

In this study we have assessed the environmental impacts from producing bread based on cereals cropped in Norway. To do so, we analyzed data from 93 conventional farms that represented the main regions for cereal production in Norway, and data from the production chain of industrially produced bread. The first part of the study focuses on the cradle to farm-gate perspective, i.e. the assessment of all pre-farm and on-farm processes related to the production of whole grains. The second part covers the farm-gate to consumer perspective, i.e. all post-farm processes attributed to the production chain starting with whole grain at the farm-gate and leading to consumer ready bread on the shop shelf.

4.1 Cradle to farm-gate

Firstly, it was of interest to assess the overall level of our calculations (Table 2). In general, the calculated impacts were larger than values commonly reported in the literature, particularly for GWP (e.g. Brentrup et al. 2004; Flysjö et al. 2008; Tuomisto et al. 2012). In a previous study (Roer et al., 2012), we showed that this can in part be explained by differences in the choice of system boundaries. When we excluded processes which have rarely been included in previous studies, such as the production of machinery and buildings, use of pesticides, changes in the soil organic carbon (SOC) stock, and NO_X loss from use of mineral fertilizer, our results were more comparable with other studies (ibid).

Besides system boundaries, yield levels should also be considered when comparing results, as this has a strong effect on the calculated impacts. As an example, Williams et al. (2010) used almost the same system boundaries as in our study when analyzing impacts of bread wheat production in England and Wales, but they reported a markedly lower GWP (700 kg CO_2-eq Mg^{-1}) than that which we found (938 kg CO_2-eq Mg^{-1} on average for winter- and spring wheat). The yield level in the study from England and Wales was, however, much higher, with 7.7 Mg grain ha^{-1} compared with our average of 4.3 Mg ha^{-1}. The same effect of yield level may, of course, be seen for other impact categories. Acidification (TA) is frequently reported for wheat, and is typically 1.5-3.3 kg SO_2-eq Mg^{-1} in studies with relatively high yields (>7.0 Mg ha^{-1}, e.g. Brentrup et al. 2004; Williams et al. 2010). In a study with low yields (<2.7 Mg ha^{-1}), Pelletier et al. (2008) reported TA of 9.7-10.2 kg SO_2-eq Mg^{-1}, which was somewhat larger than in the present study (7.1-7.6 kg SO_2-eq Mg^{-1}, Table 2).

Raising yields without increasing inputs proportionally would appear to be an efficient way of reducing the environmental impact, and should be a goal regardless of the natural conditions setting the yield limits. This is in line with Burney et al. (2010), who concluded that yield improvement compares favourably with other commonly proposed strategies for mitigation of GHG emissions.

Since the ReCiPe method (Goedkoop 2011) used in the present study is quite new, literature containing comparable results for all the impact categories is relatively scarce. We did, however, use the same method in a recent study of a case farm in Eastern Norway (Roer et al. 2012), including almost the same impact categories (except ALU) calculated for barley, oats and spring wheat. The impacts were slightly smaller in the case study, but the yields were higher than in the current study.

Removing the straw (case A) instead of incorporating it into the soil (case B) resulted in a reduction of all impact categories but GWP (Table 2). Since economic allocation was used to divide the environmental costs between grain and straw, these results are highly dependent on the price ratios used. Lower cereal prices and/or higher straw prices would increase the effect of straw incorporation on the environmental impact of cereal cropping, and *vice versa*.

The larger GWP of grain for case B (Table 2) is basically due to the reduction in annual C-input to the soil resulting from the C-export via straw removal. Reduced annual C-input to soil increases the modelled net release of C. If one considers only the grain GWP, one may get the impression that case A is environmentally superior to case B (lower C-footprint). This depends, however, on the fate of the C removed with the straw. Energy production by burning straw, and the resulting potential for substitution of e.g. fossil fuel, is a highly complex field which is beyond the scope of this study. Nevertheless, the theme is of great interest when assessing the total impacts of grain production and alternative farm management regimes, and should be focussed upon in future research.

The SOC factor affected particularly the field emissions related to GWP (Fig. 3), as almost half the emissions (on average 46 %) originated from changes in the SOC stock. This relatively large share emphasizes the importance of including such changes when assessing the environmental impact of agricultural activities. SOC dynamics are, however, rarely included in LCA studies of food production, with some exceptions (Meisterling et al. 2009; Röös et al. 2011). The dynamics of SOC in soil are a result of complex biological processes which are greatly affected by small-scale variations in soil and climatic conditions. Whether a system will have a net release or sequestration of C depends also on the annual input of C to the system and the initial level of SOC in the soil. These issues are addressed in more detail in a study (Korsaeth et al. 2013). The results showed further (Fig. 3) that CO_2 and N_2O contributed with about 50 % each (when expressed as CO_2-eq) to the field emissions related to GWP, whereas the contribution from CH_4 was negligible (Fig. 3). Small CH_4 emissions are commonly reported from cropping systems without ruminants (e.g. Brentrup et al. 2004).

Winter wheat (WW) appeared to have a lower environmental impact than the other crops (Table 2), as illustrated for GWP (Fig. 1). The main reason for this was that the highest yields were measured in WW (Table 1). Also the cumulative distribution curve of GHG emissions shows differences between crops. The steeper slope for WW (Fig. 1) indicates little variation between farms. This reflects the fact that the geographical spread of farms producing WW in our selection was less than that for the other cereals. The lion's share of WW is produced in the southern part of Eastern Norway (Statistics Norway 2012). Winter wheat is usually cropped on the best soils, and its high yield potential compensates for the higher inputs of fertilizer that are often used.

Field emissions and the manufacturing of inputs, particularly machines and buildings, appeared to be dominant process groups in the production chain of cereals up to the farm-gate (Fig. 2). The results demonstrate the importance of carefully considering where to draw the system boundaries when analysing the environmental impact associated with food production.

Impact factors with field emissions as the major contributing process group, indicate where the potential for improving farm management is greatest. This was particularly true for ALU, FE, ME, TA, TET (Fig. 2). As already mentioned, all changes that improve yields would reduce the environmental impacts, but this effect would be most pronounced for ALU (as a change in yield would alter both dividend and divisor when calculating ALU). Improving fertilizer utilization would have a direct influence on FE and ME, as excess nutrients (i.e. nutrients not utilized by the crop) increase the risk of P-losses (affecting FE) and N-losses (affecting ME) (Korsaeth & Eltun, 2008). The application of fertilizer has also a direct effect on TA, as the main contributing factors to acidification on the fields were emissions of NH_3 and NO_x. The use of the coarse ICCP framework to calculate these emissions, implies that the only way to achieve any reductions is by reducing the amount of N-fertilizer applied (or by increasing the yields at the same level of input). We hope, however, that more refined methods for estimating such emissions will be available in the near future, so that we may visualize possible positive effects of alternative management methods (e.g.

precision agriculture; Korsaeth & Riley, 2006), which may reduce gaseous N-emissions by increasing the utilization of N, irrespective of fertilizer level.

Reducing TET would require reduced application of herbicides, fungicides and insecticides, as the use of these inputs was the major source in this case. As for TA, the current method of TET impact assessment does not incorporate the effects of new and promising technology for site-specific spraying, which will/may lead to improved utilization by adjusting the doses to the site-specific requirements (e.g. Berge et. al 2012).

Manufacturing of machinery and buildings was overall the second most important process-group (following field emissions), and it dominated the emissions of FET, HT and MET (Fig. 2). For these impact categories, the improvements are thus not to be sought primarily through field management, but on-farm options to reduce these impacts do exist. Increasing the area covered by each tractor, harvester and other equipment would, for example, effectively reduce FET, HT and MET. There is a potential for such a development in Norway, as there has been an on-going decrease in the number of farmers and an increase in the area cropped by each unit over the last decades (Statistics Norway 2009). The average machinery park per hectare still appears to be large compared with most other countries (NationMaster 2003). One reason is that Norwegian farmers are generally reluctant to share machinery/equipment or to hire agricultural services from contractors, due to frequently occurring time/capacity constraints caused by unfavourable weather conditions both in spring and during harvest. The results presented here, show, however, that machinery-sharing solutions would contribute significantly to a reduction of the environmental footprint of cereal production.

4.2 Farm-gate to point of sale

Transport was generally of little importance for the environmental impact, when considering the processes from farm-gate to consumer (Fig. 4), which is in line with the findings of Narayanaswamy et al. (2004). Hence, the results were relatively insensitive to our assumptions regarding the pathway for the grain from farm-gate to consumer.

The rather evenly distributed contributions from the milling, baking and packing processes within most of the impact categories, did not pinpoint any hot-spots for emissions. Considering the small contribution from transport, it would appear, however, to be an advantage to develop production chains with large, efficient processing plants, instead of maintaining the present regionalized system of smaller (and presumably less efficient) mills and bakeries. A study on the comparison of different process chains is in progress.

4.3 Cradle to point of sale

Our results show that the major environmental impact attributable to the production of bread, based on cereals produced in Norway, occurred within the farm. Hence, improved farm management is a main key for reducing the environmental footprint of bread production. Naryanaswamy et al. (2004) found very similar results for eutrophication and terrestrial ecotoxicity impacts, when analyzing the bread supply chain in western Australia, where about 95 % of the impacts occurred up to the farm-gate. In contrast to our study, they reported that storage and processing contributed more to the total GWP and TA than the sum of pre-farm and farming processes. Their emissions levels were, however, at a much lower level than those in our study, presumably due to differences in system boundaries.

Conclusions

Assessment of environmental footprints of food production systems by LCA analysis depends largely on the choice of system boundaries and the actual yield levels used. Increasing yields is therefore an efficient way of reducing the environmental impact, so long as the inputs do not increase correspondingly.

The major environmental impacts attributable to the production of bread take place on the farm. Although there is certainly a potential for improvements of the environmental efficiency of processes occurring both pre-farm and beyond the farm-gate, our main effort should therefore be to improve the management of soil and crops at the farm level.

Straw removal affects the SOC level negatively, but its overall impact on GWP depends on the fate of the C in the removed straw. Currently, there is a lot of debate on related issues, such as the use of natural resources, e.g. straw, for bioenergy, the potential for substituting fossil energy sources in this way, and the production of biochar for long-term C-immobilization. Future solutions for improved synergies in the management of C stocks will most likely affect our future recommendations regarding on-farm straw management.

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Figure legends

Figure1. Cumulative distribution functions of GWP as kg CO_2 equivalent kg grain^{-1} for cereal crops produced on 93 farms located in the main cereal production regions in Norway

Figure 2. Relative contribution of each category of processes/inputs of spring wheat production (assuming straw incorporation)

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Figure 4. The relative, environmental burdens of post-farm processes of bread production (farm-gate to consumer) for the selected impact categories. Total impact in absolute values are indicated alongside each bar (for units, see Tab. 2)

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Table 1. Inventory data used for the cradle to farm assessment, mean values with standard deviations in parentheses

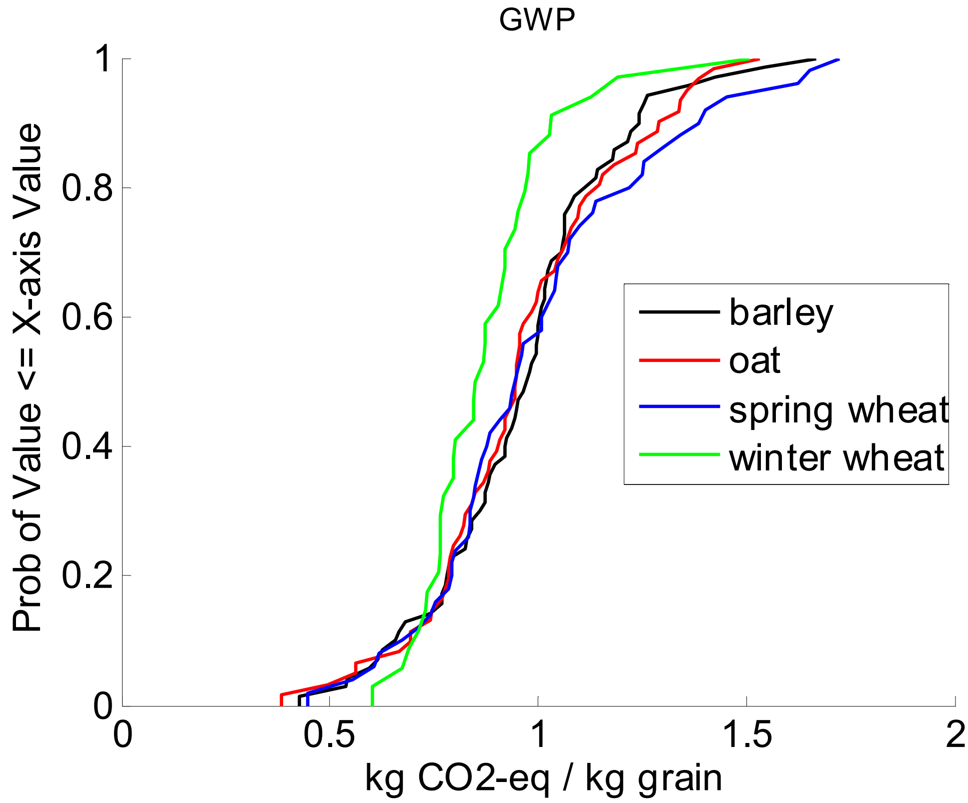
	Barley	Oat	Spring wheat	Winter wheat
Number of fields	70	61	50	34
Yield, t ha^{-1} (0.85% DM)	3.75 (0.36)	3.86 (0.47)	4.01 (0.47)	4.59 (0.59)
Straw to grain ratio (t DM t^{-1} DM)[a]	0.52	0.64	0.74	0.39
N-fertilizer[b], kg ha^{-1}	111 (8.35)	109 (7.33)	92.6 (0.76)	101 (1.35)
N-fertilizer[c], kg ha^{-1}	0	0	31.2 (3.79)	44.1 (3.75)
Lime, kg ha^{-1}	431 (16.5)	423 (13.1)	421 (10.2)	419 (6.93)
Chemical fallow[d], kg ha^{-1}	0.93	0.93	0.93	0.93
Spraying (herbicide)[d], kg ha^{-1}	0.07	0.08	0.07	0.01
Spraying (fungicide)[d], kg ha^{-1}	0.17	0	0.25	0.24
Spraying (insectcide)[d], kg ha^{-1}	<0.01	<0.01	0	0
Spraying (growth regulator) [d], kg ha^{-1}	0.02	0.38	0	0
Diesel, l ha^{-1}	74.4 (5.40)	76.7 (3.20)	77.2 (3.23)	83.9 (2.91)
Initial SOC-stock, t C ha^{-1}	67.9 (13.8)	71.3 (12.4)	71.5 (12.7)	74.3 (0.88)
N-leaching, kg N ha^{-1}	30.1 (7.67)	30.3 (8.81)	33.6 (8.19)	39.0 (11.1)
P-loss, kg P ha^{-1}	1.47 (0.78)	1.81 (0.62)	1.84 (0.65)	1.99 (0.44)
Buildings (M€ yr^{-1}farm^{-1})[e]			0.01	
Machinery (t yr^{-1} farm^{-1})[f]			1.9	

[a] From Riley et al. (2012).
[b] Compound fertilizer with 21.6 % N, 2.6 % P and 9.6 % K.
[c] Containing 27 % N.
[d] Active ingredience.
[e] Assuming a lifetime of 30 yrs.
[f] Assuming lifetimes of 10-20 yrs (based on Roer et al. 2012). When the straw was removed (case B), the total, annual machinery weight was increased by 0.49 t yr^{-1} to account for the baler.

Table 2. Environmental impacts from producing 1 tonne of barley (B), oat (O), spring wheat (SW) and winter wheat (WW) on 93 cereal farms calculated for case A: All straw was incorporated, with grain as the only product, and case B: All straw was removed and the impacts were allocated between the products grain and straw based on their economic value. Standard deviations are shown in parentheses

Impact categories[a]			System		
			Case A	Case B	
	Unit	Crop	Grain (t 85% DM)	Grain (t 85% DM)	Straw (t DM)
GWP	kg CO_2-eq	B	966 (228)	997 (200)	356 (82.1)
		O	963 (234)	963 (194)	342 (76.4)
		SW	997 (279)	1000 (239)	291 (81.5)
		WW	879 (170)	951 (161)	270 (58.9)
ALU	ha	B	2858 (298)	2486 (259)	715 (74.6)
		O	2819 (398)	2368 (335)	705 (99.7)
		SW	2704 (390)	2299 (332)	549 (79.3)
		WW	2349 (356)	2161 (328)	483 (73.3)
FE	kg P-eq	B	0.54 (0.23)	0.47 (0.20)	0.16 (0.06)
		O	0.62 (0.19)	0.52 (0.16)	0.17 (0.05)
		SW	0.61 (0.22)	0.51 (0.18)	0.14 (0.05)
		WW	0.55 (0.14)	0.50 (0.13)	0.13 (0.03)
ME	kg N-eq	B	10.3 (2.39)	8.98 (2.08)	2.61 (0.60)
		O	9.58 (2.49)	8.05 (2.09)	2.42 (0.62)
		SW	10.2 (2.50)	8.70 (2.12)	2.10 (0.51)
		WW	10.2 (2.55)	9.42 (2.35)	2.13 (0.52)
FET	kg 1,4-DCB-eq	B	4.00 (1.50)	3.49 (1.31)	1.93 (1.03)
		O	3.83 (1.37)	3.26 (1.09)	1.64 (0.76)
		SW	2.79 (1.71)	3.24 (1.45)	1.40 (0.88)
		WW	2.92 (0.94)	2.69 (0.86)	1.39 (0.68)
TA	kg SO_2-eq	B	7.36 (0.97)	6.41 (0.84)	2.04 (0.32)
		O	7.09 (1.08)	5.97 (0.89)	1.93 (0.32)
		SW	7.60 (1.19)	6.46 (1.02)	1.68 (0.31)
		WW	7.49 (1.20)	6.89 (1.10)	1.70 (0.31)
FD	kg oil-eq	B	115 (33.4·)	99.9 (29.1)	52.1 (19.6)
		O	108 (32.1)	91.6 (25.5)	46.0 (15.3)
		SW	112 (39.6)	95.6 (33.7)	40.5 (17.3)
		WW	95.9 (24.1)	88.0 (22.2)	39.2 (13.5)
HT	kg 1,4-DCB-eq	B	133 (68.5)	116.7 (60.2)	57.0 (33.1)
		O	120 (56.5)	102.7 (46.7)	47.7 (24.8)
		SW	125 (75.2)	107.1 (64.2)	41.8 (28.7)
		WW	91.6 (39.0)	84.7 (36.0)	39.3 (20.4)
MET	kg 1,4-DCB-eq	B	2.90 (1.56)	2.54 (1.37)	1.70 (1.07)
		O	2.64 (1.31)	2.24 (1.08)	1.37 (0.78)
		SW	2.75 (1.76)	2.35 (1.50)	1.22 (0.91)
		WW	1.97 (0.89)	1.82 (0.82)	1.23 (0.69)
TET	kg 1,4-DCB-eq	B	1.52 (0.15)	1.32 (0.14)	0.39 (0.04)
		O	0.64 (0.09)	0.54 (0.08)	0.17 (0.02)
		SW	1.61 (0.23)	1.37 (0.20)	0.33 (0.05)
		WW	1.53 (0.23)	1.41 (0.22)	0.32 (0.05)

[a] GWP: Global warming potential; ALU: Agricultural land use; FE: Freshwater eutrophication; ME: Marine eutrophication; FET: Freshwater ecotoxicity; TA: Terrestrial acidification; FD: Fossil fuel depletion; HT: Human toxicity; MET: Marine ecotoxicity and TET: Terrestrial ecotoxicity.



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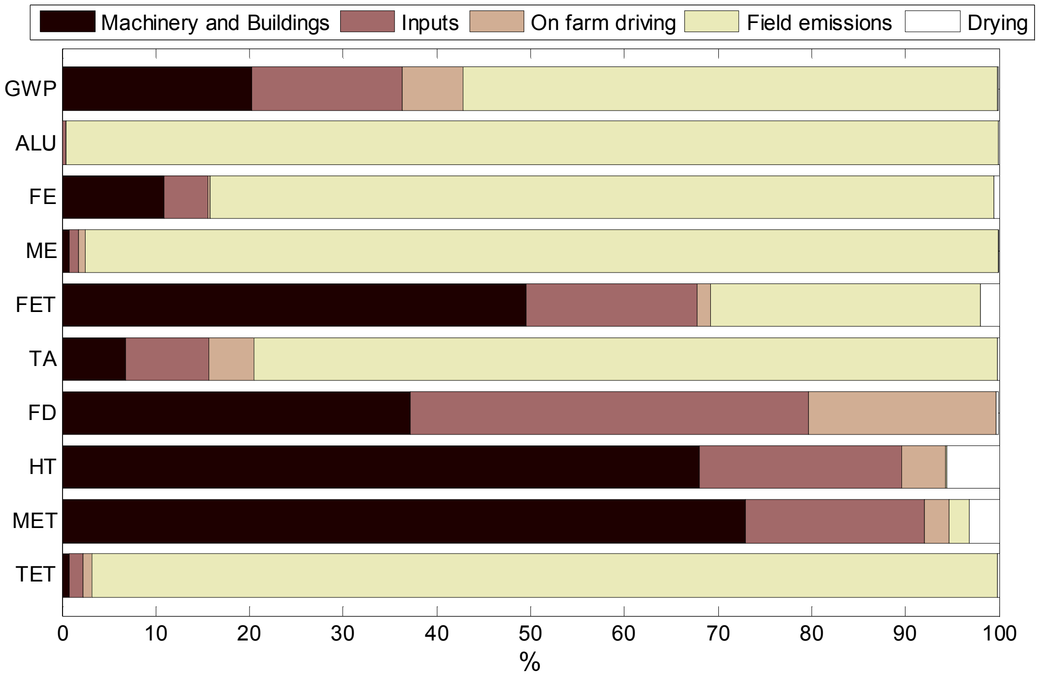

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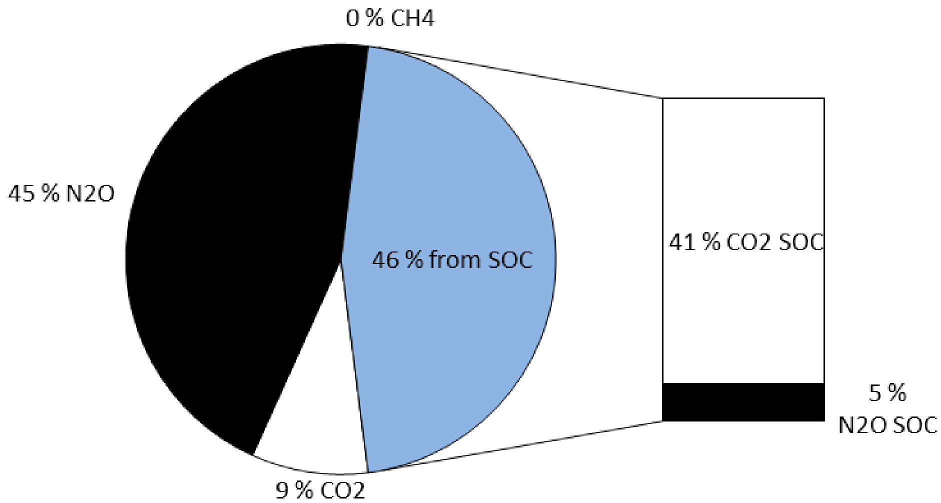

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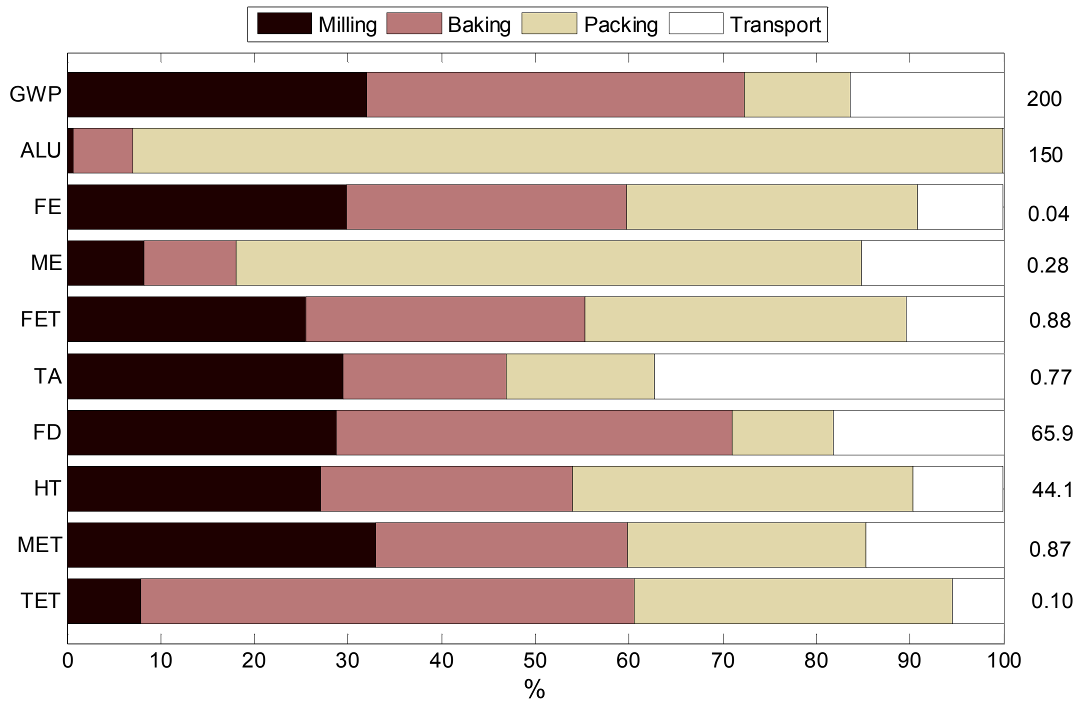

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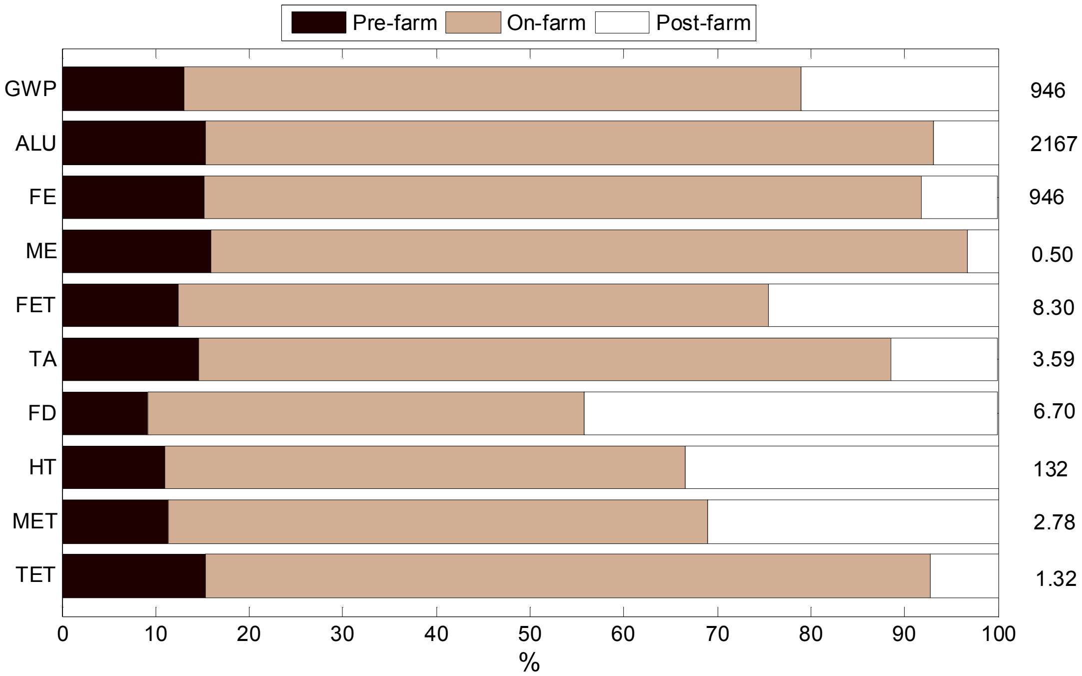

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