

Life cycle assessment of fertilizers: a review

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Abstract. Life cycle assessment has become an increasingly common approach for identifying, quantifying, and evaluating the total potential environmental impact of production processes or products, from the procurement of raw materials (the ‘cradle’), to production and utilization (the ‘gates’) and their final storage (the ‘grave’), as well as for determining ways to repair damage to the environment. The paper describes life cycle assessment of mineral fertilizers. On the basis of results provided by life cycle assessment, it can be concluded that an effective strategy for protecting the environment against the harmful effects of fertilizers is to attempt to ‘seal’ the nutrient cycle on a global, regional, and local scale. Pro-environmental measures aim on the one hand to reduce resource utilization, and on the other hand to limit losses of nutrients, during both production and use of fertilizers. An undoubted challenge for life cycle assessment when used in agricultural production is the need for relevance at each scale.

Keywords: life cycle assessment, fertilizers, environment

INTRODUCTION

Crop production, as a source of plant raw materials and energy, is a key element of the development of civilization. The significant growth in the world population over the last century is integrally linked to the advancement of agricultural technology, and since the process for industrial synthesis of ammonia was developed in Germany by Haber and Bosch, to the wide-scale popularization of the use of mineral fertilizers as well (Cichy, 2012; Fertilizers, Climate Change and Enhancing Agricultural Productivity Sustainably, 2009). Currently nearly half of the world population is supplied with food produced using artificial fertilizers. In 2011/2012, fertilizer consumption in EU-27 countries was 10.5 mln t of N, 2.4 mln t of P₂O₅, and 2.7 mln t of K₂O. The anticipated increase in the population to 2050 will increase agricultural production by another 50-80% and the dependency on ferti-

lizer inputs (Brentrup and Palliere, 2008; Dawson and Hilton, 2011; Fertilizers, Climate Change and Enhancing Agricultural Productivity Sustainably, 2009).

In Poland, a long-term (1985-2008) analysis of the trend in production intensity, measured as nitrogen fertilizer consumption, indicates an average annual increase of about 1.7 kg N ha⁻¹ of agricultural land. Although per unit consumption of nitrogen fertilizer in cereal production, which is the ‘reverse’ of fertilization efficiency estimated from the production function, is similar in Poland to the mean for EU-15 countries, yields of grain crops are lower. This is in part due to the need in Poland to ensure an adequate level of agricultural production in inferior natural conditions (soil, climate) by increasing consumption of mineral fertilizers, which are less efficient in these conditions (Fotyma *et al.*, 2010; Kopiński, 2012).

Environmental threats associated with the intensification of agricultural production are currently becoming a significant factor determining directions for the development of production technology and infrastructure in the mineral fertilizer industry, and agriculture faces new tasks associated with protecting the natural environment. The use of existing products and/or implementation of new ones, including fertilizers, require appropriate tools for evaluating how they interact with their environment. One of these tools is a life cycle assessment (LCA), which can be used to identify, quantify and, evaluate the total potential environmental impact of production processes or products, from the procurement of raw materials (the ‘cradle’), to production and utilization (the ‘gates’) and their final storage (the ‘grave’), as well as to determine ways to repair damage to the environment (Brentrup and Palliere, 2008; Kopiński, 2012).

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The LCA has been defined in ISO norms and on official websites of the European Commission as a process of gathering and evaluating input and output data and assessing a product potential effects on the environment during its life cycle (Fallahpour *et al.*, 2012; Kowalski *et al.*, 2007).

According to the guidelines laid out in ISO 14040, LCA is carried out in four phases:

1. determining the goal and scope of the study (choosing the functional unit and system boundaries);
2. analysis of an inventory of inputs and outputs (analysis of the technological process, balance of flows of raw materials, energy, and auxiliary materials, waste balance, and identification of their potential sources);
3. assessment of the environmental impact of the life cycle (transforming the data collected into impact category or damage category indicators);
4. interpretation (conclusions and verification of results) (Fig. 1).

This paper reviews recent LCA studies in the context of the production and use of mineral fertilizers.

DETERMINATION OF GOAL AND SCOPE IN LCA OF MINERAL FERTILIZERS

This stage involves specification of the end-user of the study, the end-use of the results, and the goal, which determines the scope of the study. Most important in defining the scope of the study is to specify and define the product system (a set of unit processes linked by materials and energy which perform one or more defined functions); its boundaries (the areas of contact between the product system and the environment or other product systems); and functional units (the quantitative effect of the product system used as a plane of reference in LCA, mainly to normalize the system input and output data) (Brentrup *et al.*, 2004a; Kowalski *et al.*, 2007).

In analysing the environmental consequences of mineral fertilizers, we must distinguish between the impact at the level of industrial production technologies and during their application in agroecosystems. This will be reflected in

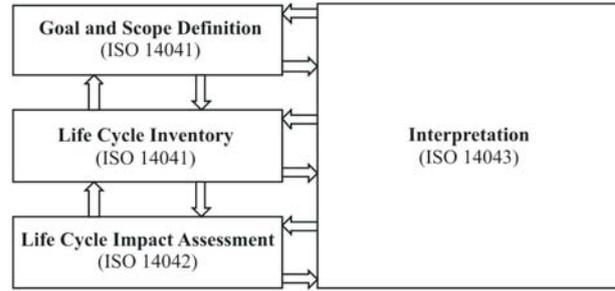


Fig. 1. Life cycle assessment framework (Fallahpour *et al.*, 2012; Kowalski *et al.*, 2007).

work on the LCA of fertilizers by differentiating the product system and the system boundaries (Fig. 2, Table 1). An approach termed ‘from the cradle to the gate’ is often used, which does not take into account the environmental effects of using the fertilizer in the field (Table 1). Some authors emphasize that LCA of fertilizers in crop production should take into account the larger scale of the system, including such factors as the quality of the yield, biodiversity, and the multifunctionality of agroecosystems (Charles *et al.*, 2006).

When LCA is used in agriculture, the functional unit most often chosen is the weight of the raw material or product (eg 1 kg, 1 t) or surface area (eg 1 ha) (Brentrup *et al.*, 2004a; Charles *et al.*, 2006; Hayashi, 2013). Due to their asymmetry, however, some authors recommend using these units simultaneously (Hayashi, 2013; Nemecek *et al.*, 2011). LCIA (life cycle impact analysis) of fertilizers has been shown to depend to a substantial degree on the functional unit chosen and on the goal of the analysis (Charles *et al.*, 2006; Nemecek *et al.*, 2011). According to Charles *et al.* (2006), in assessing the efficiency of a production system for a particular crop (eg wheat), the functional unit should be a tonne of grain, whereas the hectare should be used in analysing production intensity.

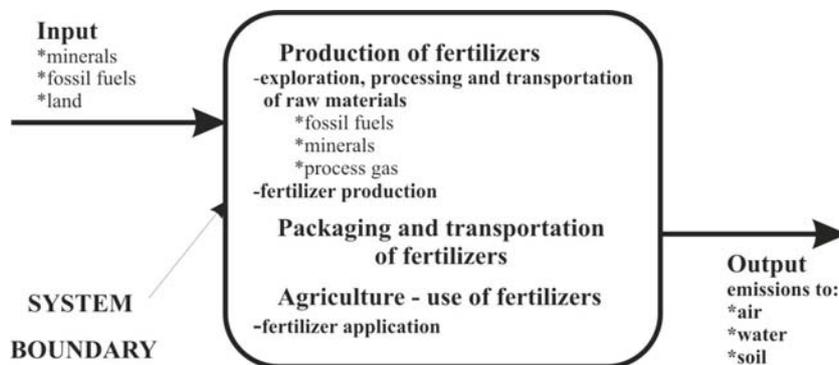


Fig. 2. System boundary to analyse environmental aspects of fertilizer production and use (Brentrup, 2012; Brentrup *et al.*, 2004a; Lammel, 2000).

Table 1. Examples of goals, functional units and system boundaries in LCA studies taking into account the production and use of fertilizers under European conditions

Goal	Functional units	System boundary	Source
The environmental impacts of urea and ammonium nitrate fertilizers from production to their use on the field	1 t of nitrogen	Fertilizer production and use on field	Lammel (2000)
Evaluation and comparison of the environmental impact of fertilizer production under conditions of biogas and natural gas use	1 kg of nitrogen, as ammonium nitrate	From cradle to gate – ammonium nitrate production using natural gas originated from the North Sea and biogas produced from anaerobic digestion of ley grass and maize	Ahlgren <i>et al.</i> (2010)
Assessment of the environmental impact of different ways to supply Swedish agriculture with phosphorus fertilizers	11 kg P ha ⁻¹ (the average phosphorus output -removal with harvest, per hectare from Swedish farmland)	Production and application of phosphorus fertilizers, certified sewage sludge; struvite precipitated from wastewater; and phosphorus recovered from sludge incineration	Linderholm <i>et al.</i> (2012)
Quantification and evaluation of different N fertilizer impact on the entire environmental burden associated with a sugar beet production system	1 t of extractable sugar	The sugar beet production system considering production, transportation and application of fertilizers	Brentrup <i>et al.</i> (2001)
The environmental impact of winter wheat production at different production intensities, which are represented by increasing N application rates	1 t of grain	An arable farming system with the main function to produce winter wheat	Brentrup <i>et al.</i> (2004b)
Optimization of N, P and K fertilization intensity	1 ha 1 t of grain produced 1 t of grain with 13% protein	The wheat production system for breadmaking, considering field emissions, production and transportation of fertilizers	Charles <i>et al.</i> (2006)
Assessment of environmental impacts of extensive farming	per hectare and year kg dry matter yield	All inputs and processes for the plant production including fertilizer manufacturing and application	Nemecek <i>et al.</i> (2011)
Comparison of energy use, land use and GWP of organic, conventional and integrated farming systems	1 t of winter wheat with 86% dry matter content	The production chain, including production of farming inputs, machinery, farming operations and crop cooling and drying	Tuomisto <i>et al.</i> (2012)

LIFE CYCLE INVENTORY (LCI) – ANALYSIS OF INPUTS AND OUTPUTS (INVENTORY OF DATA)

The second phase of the LCA method involves identifying, gathering, and analysing the inventory of inputs (consumption of natural resources, use of land, materials, energy carriers) and outputs (*eg* emissions of substances into the atmosphere, water, and land, waste, by-products) in the product system, which are usually assigned to each unit

process. Data on the processes are procured by means of measurements and calculations, from available databases, publications, and reports, and on the basis of unpublished information from companies, research institutions, *etc.* The difficulties arising at this stage of the study involve obtaining data protected as trade secrets, and precisely assigning input and output flows of the process to the defined functional unit (Fallahpour *et al.*, 2012; Kowalski *et al.*, 2007).

In the case of fertilizers, emissions mainly involve production of carbon dioxide (about 1.6 t CO₂ per 1 t NH₃) and nitrous oxide (about 2-2.5 kg N₂O per 1 t HNO₃), as well as dispersion in the environment of 30-40% of the nitrogen used on crop fields, in the gaseous form as NH₃, nitrogen oxides (N₂O, NO, NO₂), or molecular nitrogen (N₂), or by leaching in the form of NO₃⁻ or NH₄⁺ (Fig. 3). Nitrous oxides are mainly produced during denitrification (3-10% of the main product of the process – N₂) and nitrification (0.3-3% of oxidized NH₄⁺). The latter is of greater significance where soil moisture is low or average, and the higher the percentage of NO₃-N in the fertilizer, the lower the N₂O emission (Brentrup and Palliere, 2008; Lammel, 2000).

In the case of phosphorous fertilizers, the main environmental problems are associated with the following:

- exploitation of non-renewable phosphate rock (PR), which unlike fossil fuels has no substitutes (global PR resources are estimated at about 290 billion t, and usable or marketable reserves at 60 billion t),
- hydrated calcium sulphate, known as phosphogypsum, which is a troublesome waste of phosphoric acid production (production of one tonne of phosphoric acid generates about 4-5 t of phosphogypsum, which in Poland is about 2.5 mln t per year),
- input of heavy metals (mainly cadmium) into agroecosystems,
- release of phosphates from phosphogypsum stacks and from soil-applied fertilizer into surface waters (mainly – 75-90% – during water erosion, and to a lesser degree by leaching) (Cichy, 2012; Dawson and Hilton, 2011).

It should also be kept in mind that the energy consumption of fertilizer installations is considerable – on average production of 1 kg NPK (15-15-15) requires 9.81 MJ and is mainly associated with production of nitrogen fertilizers, which account for 90% of the global energy input into fertilizer production (Brentrup and Palliere, 2008; Dawson and Hilton, 2011).

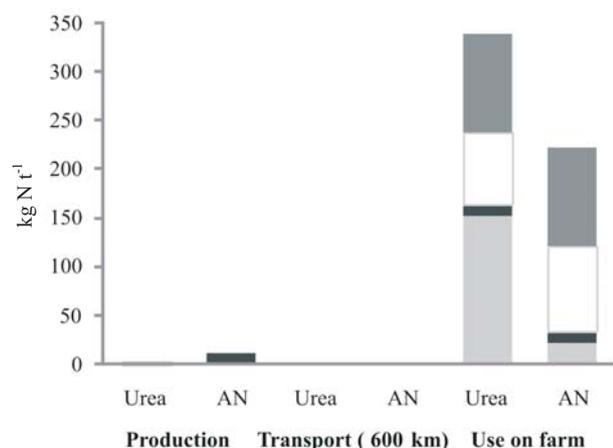


Fig. 3. Average nitrogen emissions caused by fertilization with urea and ammonium nitrate (AN) (Lammel, 2000).

LCIA (LIFE CYCLE IMPACT ANALYSIS/LIFE CYCLE IMPACT ASSESSMENT)

This phase assesses the impact of the product or process on the environment. In order to transform the LCI data into impact category indicators and to obtain values (indicator results for each impact category), three mandatory tasks have been introduced: selection of impact categories, category indicators, and characterization models; classification (assigning LCI results to particular impact categories); and characterization (calculating the value for the impact category indicator using characterization factors). The LCIA phase includes optional procedures as well – normalization (calculating the normalized indicator values with respect to reference values, without assessing the importance of the environmental impact), grouping (assigning the impact category to one or more sets according to the goal and scope of the study), weighting (assigning importance to each impact category and damage category), or data quality analysis (Kowalski *et al.*, 2007). LCIA is the most important and most controversial phase of LCA. Transforming LCI results into indicators is made more difficult by the fact that there are many models for replacing a given type of emission with units of indicator result and assigning them to an impact category. Moreover, the relationships between a given emission and its impact are usually determined with respect to averaged European data sets and do not take local conditions into account. Thus, the LCIA results often only identify and define potential environmental impacts (Brock *et al.*, 2012; Jensen *et al.*, 1997).

When Brentrup *et al.* (2004a) applied the LCA method to crop production, they distinguished input-related impact categories - abiotic resource depletion and land use - and output-related categories *eg* global warming, acidification, and eutrophication (Fig. 4). Not all researchers, however, take all these categories into account in LCA, often limiting their analysis to carbon footprint or energy balance (Brentrup and Lammel, 2011). On the other hand, in some publications the list of categories is expanded to include soil quality, or biodiversity (Mourad *et al.*, 2007; Nemecek *et al.*, 2011).

Consumption of abiotic resources during fertilizer production and application mainly involves consumption of fuels and of phosphorus- and potassium-rich rocks, which will no longer be available for future generations (Table 2).

Resources should be differentiated, however, based on their functions *ie* those used as energy sources (coal, natural gas, oil) and those procured to obtain substrates for the fertilizer industry (phosphorites, apatites, potassium salt deposits) (Brentrup *et al.*, 2004a). During ammonia synthesis, only 30% of the natural gas is used as fuel for maintaining the proper temperature for the process, and the rest (70%) of the methane provides the substrate (60% of the hydrogen) required for production of NH₃; 40% of the H₂ is obtained by steam reforming (Brentrup and Palliere, 2008). The most energy-consuming element of ammonia production is water

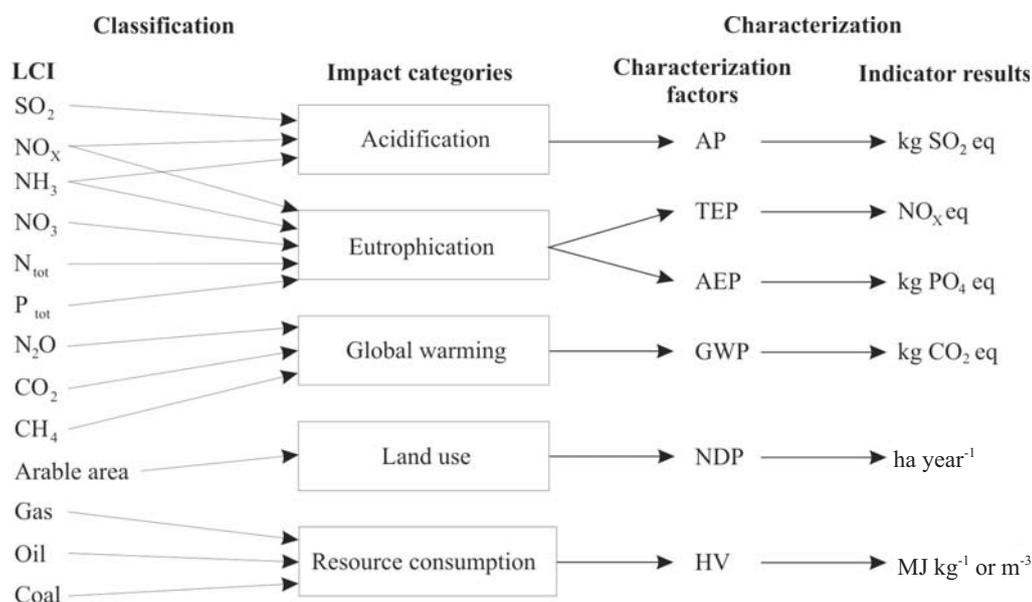


Fig. 4. Aggregation (classification and characterization) of emissions (Brentrup and Lammel, 2011), LCI – life cycle inventory, AP – acidification potential, TEP – terrestrial eutrophication potential, AEP – aquatic eutrophication potential, GWP – global- warming potential, NDP – naturalness degradation potentials, HV – heat values.

Table 2. Main burdens for producing, packing, and delivering the main types of fertilizers (Brentrup and Palliere, 2008; Williams *et al.*, 2010)

Fertilizer product	Unit (kg)	Primary energy consumption (MJ)	Global warming potential (kg CO ₂ eq)			Eutrophication potential (g PO ₄ ³⁻ eq)	Acidification potential (g SO ₂ eq)	Abiotic resource use (g Sb eq)
			CO ₂	N ₂ O	Total			
AN	N	40a /29.8	2.34a /1.77b	3.69a /0.83b	6.2a /2.74b	0.5	4.7	23
Urea		51.6a /44.1b	1.39a/0.98b	0	1.59a /1.13b	0.54	5.3	23
CAN		42.6a /31.4b	2.49a /1.89b	3.66a /0.83b	6.3a /2.83b	0.55	5.3	21
AS		42	ld	ld	3	0.52	5.3	20
TSP	P	30.25	1.6a	0	1.66a	0.74	8.1	15
SSP		13	ld	ld	0.6	0.57	6.6	16
MOP	K	10.06	0.58a	0	0.60a	0.30	7.2	3.9
L	Ca	2.3	ld	ld	0.15	0.26	1.6	2.4

*AN – Ammonium nitrate, CAN – Calcium ammonium nitrate, AS – Ammonium sulphate, TSP – triple super phosphate, SSP – single super phosphate, MOP – muriate of potash, L – limestone, ld – lack of data, a – production (European average) at plant gate, b – production (BAT) at plant gate.

hydrolysis, followed by coal gasification. Modern natural gas reforming is the most energy-efficient (Table 3); since 1903 energy consumption in this process has decreased sixfold to 34 GJ/t N, and currently documented natural gas reserves will enable it to continue for the next 1 000 years.

Some authors emphasize that the energy used in fertilizer production accounts for a small percentage of its total use, both globally (1.1%) and in Europe (0.6%) (Dawson and Hilton, 2011). According to an analysis by Brentrup *et al.* (2004b), nitrogen application rates under 96 kg N ha⁻¹ and

Table 3. Various processes for the manufacture of ammonia (Dawson and Hilton, 2011)

Process	Reaction	Approx relative energy consumption when making ammonia (%)
Water electrolysis	$2\text{H}_2\text{O} \rightarrow 2\text{H}_2 + \text{O}_2$	300
Coal gasification	$\text{C} + 2\text{H}_2\text{O} \rightarrow 2\text{H}_2 + \text{CO}_2$	170
Naphtha reforming	$\text{CH}_2 + 2\text{H}_2\text{O} \rightarrow 3\text{H}_2 + \text{CO}_2$	104
Natural gas reforming	$\text{CH}_4 + 2\text{H}_2\text{O} \rightarrow 4\text{H}_2 + \text{CO}_2$	100

over 144 kg N ha^{-1} cause the greatest rises in energy consumption. These mainly involve increased consumption of gas, used both as fuel and as a source of hydrogen in ammonia synthesis. Energy balance analyses using LCA have shown that energy accumulation in crops is as much as 15 times greater than the energy consumed to produce them (Brenttrup and Lammel, 2011).

Land use

This category is distinctive for LCA applied to agricultural production and is characterized by a complex network of interrelationships (Fig. 5). Agricultural land use involves the occupation of land to be cultivated (expressed in ha year^{-1} or $\text{m}^2 \text{ year}^{-1}$) and the change in its quality while it is being used, degradation of natural land, reduction in biodiversity, and reduction of its ecosystem functions, such as carbon sequestration, productivity, or regulation of erosion (Charles *et al.*, 2006; Cowell and Lindeijer, 2000; Koellner *et al.*, 2013; Mourad *et al.*, 2007).

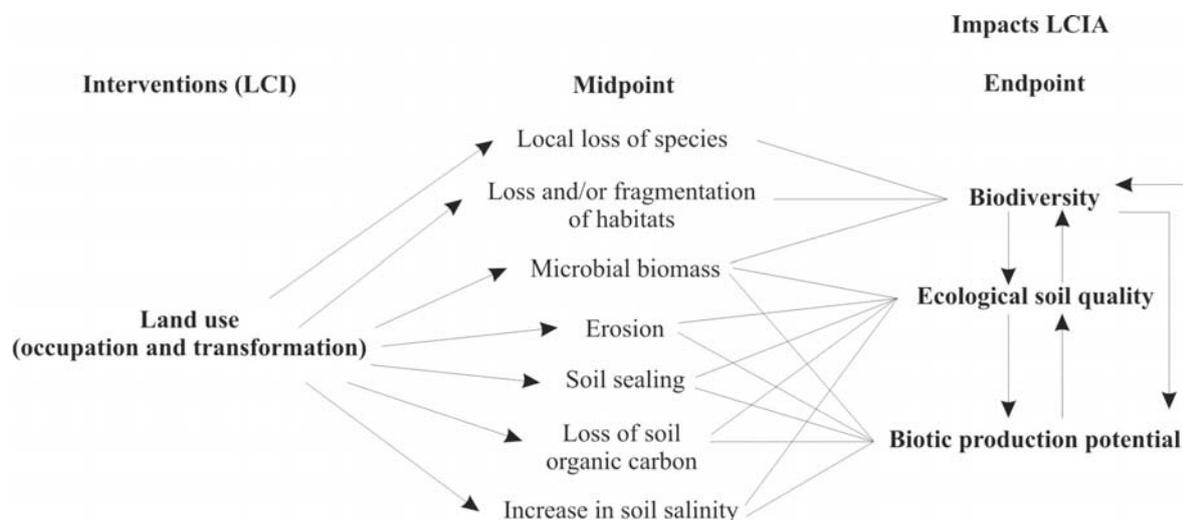
Given the global demand for food production, if fertilizer application rates are reduced and thus crop yield decreases in one location in the world, it will be necessary to increase crop production in another, often by changing the

land use from natural ecosystems to agroecosystems. For this reason, some authors postulate that the 'land use' category should be examined on a global scale, and the point of reference should not be production, but consumption of crops (Kloverpris *et al.*, 2010).

Studies by Brenttrup *et al.* (2004b) and Charles *et al.* (2006) have found that land use efficiency increases with fertilization intensity, where the functional unit is 1 tonne of grain.

Greenhouse effect

Both the production and use of mineral fertilizers contribute to changes in the global-warming potential (GWP), calculated according to the formula used by the Intergovernmental Panel on Climate Change (IPCC) (Table 2). Production of mineral fertilizers increases greenhouse gas emissions (GHG), mainly CO_2 from fossil fuels used in ammonia production, and to a lesser degree CO_2 in the reaction of phosphorites with sulphuric acid or during extraction of phosphorus- or potassium-rich materials, and N_2O , mainly during production of nitric acid. The size of emissions varies depending on the type of fertilizer, the raw materials used, the use of technology for CO_2 recovery

**Fig. 5.** Overview of the cause-effect chain related to land use (Cowell and Lindeijer, 2000).

(eg to produce urea or carbonated beverages, or for natural gas and oil extraction), and reduction of N_2O emissions (eg non-selective catalytic reduction – NSCR). Fertilizer transport also contributes to greenhouse gas emissions (37 Tg CO_2 eq globally) (Brentrup and Palliere, 2008; Czyż *et al.*, 2007; Fertilizers, Climate Change and Enhancing Agricultural Productivity Sustainably, 2009; Szarlip *et al.*, 2010). The results of the analyses indicate that the global warming potential of fertilization in agroecosystems is mainly determined by emissions of nitrous oxide, and to a lesser degree carbon dioxide.

With greater application rates for nitrogen fertilizers, which are mainly a source of nitrous oxide (N_2O) with high GWP (298), their contribution to the greenhouse effect increases as well. At the same time, crop plants fix about 1.6 tonnes of CO_2 per 1 tonne of biomass produced. Thus, when fertilization is economically optimal (according to the Code of Good Agricultural Practice), a yield of 18.5 t of wheat (grain + straw) fixes 29.6 t CO_2 . Unfortunately, a substantial net saving of CO_2 emissions can be achieved when the biomass obtained will be used to produce biofuel, thus reducing fossil fuel consumption (Brentrup, 2012). On the other hand, without any nitrogen input increases GWP due to ‘land compensation for crop yield loss’. Hence, intensification of crop production that more efficiently utilizes resources, including arable land, leads to a decrease in GWP by minimizing the transformation of natural ecosystems into agroecosystems (Brentrup and Palliere, 2008).

Acidification

LCA of fertilizers takes into account the acidification process, which occurs as a result of NH_3 , NO_x , and SO_2 emissions (Brentrup *et al.*, 2004a). It does not, however, include the H^+ ion load generated by their chemical conversions in the soil. The main source of sulphur dioxide is fossil fuel combustion, whereas ammonia and nitrogen oxides appear both during production and after the fertilizer is applied. Ammonia losses range from 1 to 15% of the nitrogen mass in the fertilizer and are the greatest following application of urea. Nitrogen oxide emissions result from fossil fuel combustion on the one hand, and on the other hand, from increases in the efficiency of nitrification and denitrification processes following application of mineral fertilizer, as well as ammonia volatilization (about 10% of ammonia from agriculture is oxidized to NO_x in the atmosphere). NO emissions from mineral fertilizers constitute from 0.3 to 3% of the nitrogen applied. Sapek (2008) cites a study that reported the highest emission, expressed as the percentage of nitrogen in the fertilizer, following application of urea (3.2%); in the case of ammonium nitrate, it was only 0.7%.

The acidification potential (AP) differs for NH_3 , NO_x , and SO_2 . In the case of the first two compounds, one mole generates 1 mole of H^+ , while sulphur dioxide becomes the

source of 2 moles of H^+ . Thus, in LCA the acidification potential of these pollutants is expressed as the SO_2 equivalent – SO_2 eq t^{-1} of grain or other functional unit (Brentrup *et al.*, 2004a).

The main source of acidifiers on unfertilized fields is fuel (SO_2 and NO_x), but due to low yields, the acidification potential is relatively high (Brentrup and Lammel, 2011). AP grows with increased nitrogen application in the form of NH_4NO_3 , mainly due to ammonia volatilization.

Eutrophication

In this category, some researchers distinguish two sub-categories – eutrophication of terrestrial ecosystems caused by nutrient deposition (Terrestrial Eutrophication Potential - TEP, expressed in $kg NO_x$ eq/tonne of grain or other functional unit) and eutrophication of aquatic ecosystems, involving gaseous losses and leaching of nitrogen and phosphorus compounds (Aquatic Eutrophication Potential - AEP, expressed in $kg PO_4$ eq/t of grain or other functional unit). As application rates of nitrogen fertilizers increase, changes in the terrestrial eutrophication potential will have a similar pattern as in the case of acidification (Brentrup, 2012; Brentrup *et al.*, 2004b).

Analysis of the aquatic eutrophication potential indicates that when nitrogen fertilizer application rates exceed $144 kg N ha^{-1}$, changes in AEP are mainly determined by the amount of nitrate leaching; phosphorus, which mainly comes from phosphorus fertilizer production, accounts for only 9% of AEP even when the fertilizer is applied at the highest rates (Brentrup *et al.*, 2004b). Normalization and weighting conducted by Brentrup and Lammel (2011) under European conditions indicate that mineral fertilizer application rates that are either too high or too low reduce the eco-efficiency of crop production, mainly due to eutrophication in the former case and to inefficient land use in the latter (Fig. 6).

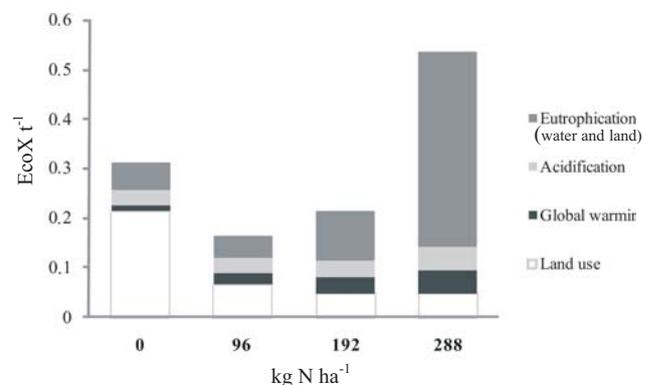


Fig. 6. Aggregated environmental index (EcoX) per tonne of wheat grain at increasing N fertilizer rates (Brentrup and Lammel, 2011).

LIFE CYCLE INTERPRETATION

The aim of the final phase of LCA is to analyze the results, specify the limitations on the accuracy of the analysis, and formulate conclusions and recommendations in accordance with the goal defined in the first stage (Fallahpour *et al.*, 2012; Kowalski *et al.*, 2007).

Life cycle assessment of fertilizers indicates that despite the technological improvements in its manufacture and use during the last 100 years, greater production intensity increases emissions of pollutants (N_2O , NO_x , NH_3 , PO_4-P) contributing to the greenhouse effect, acidification, and eutrophication. Fertilizers containing heavy metals (Cd, Zn, Co, Se, Hg) also have a toxic effect on water, land, and human beings. The greatest impact on the environment in crop production relying on mineral nitrogen fertilizers is usually associated with changes in land use and eutrophication of aquatic ecosystems (Brentrup and Lammel, 2011; Charles *et al.*, 2006) and, in warmer climate conditions, with increased GWP (Fallahpour *et al.*, 2012). The negative environmental impact of the production and use of phosphorus fertilizers is mainly due to the greenhouse effect (transport of raw materials and products) and eutrophication (dispersion of phosphates during fertilizer production and accumulation of phosphogypsum) (Silva and Kulay, 2003, 2005).

Fertilization which enables optimal yields, in accordance with the nutrient requirements of crops, ensures the most efficient land use and reduces leaching of nitrates (Brentrup, 2012; Brentrup and Lammel, 2011; Charles *et al.*, 2006). The best results in terms of eco-efficiency are obtained in crop systems of medium production intensity (Brentrup and Lammel, 2011, Nemecek *et al.*, 2011). Nemecek *et al.* (2011) emphasize that a 50% reduction in high application rates for mineral fertilizer decreases its negative impact to the greatest degree (per 1 ha) with respect to consumption of natural reserves of phosphorus-rich rock (by 48%) and potassium-rich rock (by 49%), acidification potential (by 36%), GWP (34%), eutrophication potential (by 31%), energy consumption (19%), and ozone formation potential (15%). When this impact is calculated per unit of weight (kg d.w. of yield), due to the decrease in yield where fertilizer application rates are reduced, changes in particular categories will have a completely different pattern.

Attempts to reduce consumption of abiotic resources do not always lead to the expected results. Replacement of mineral fertilizers with organic ones or with waste from biomass fermentation in biogas plants has reduced resource consumption, but also increased nutrient losses, and thus eutrophication and acidification (Nemecek *et al.*, 2011; Tuomisto *et al.*, 2012). The use of biogas from maize or hay instead of natural gas to produce ammonium nitrate reduced primary energy use from fossil fuels (from 35 MJ kg^{-1} N to 2-4 MJ kg^{-1} N) and GWP (from 2.410 g CO_2 eq kg^{-1} N to 1.121-1.450 g CO_2 eq kg^{-1} N). At the same time, however, the eutrophication potential increased more than tenfold,

mainly due to leaching of nitrates during production of fermentation substrates. There was a rise in the acidification potential as well (from 2 g SO_2 eq kg^{-1} N to 5-8 g SO_2 eq kg^{-1} N) caused by emissions during spreading of digestate and nitrogen production for fertilizing maize and grasses (Ahlgren *et al.*, 2010). In a study on the use of traditional fertilizers (triple superphosphate) and fertilizers from recycled phosphorus sources (sewage sludge, struvite ($MgNH_4PO_4 \cdot 6H_2O$) precipitated from wastewater, phosphorus recovered from incinerated sewage sludge), Linderholm *et al.* (2012) determined that on the one hand soil-applied certified sewage sludge was the most efficient option in terms of energy consumption and greenhouse gas emissions, but on the other hand substantial amounts of cadmium entered agroecosystems with it, increasing ecotoxicity.

Sometimes the main determinant of emissions during field fertilization is the type of the fertilizer used; when chosen properly, the negative impact on the environment can be significantly reduced. In a study by Charles *et al.* (2006), substituting triple superphosphate for Thomas slag decreased toxicity associated with the presence of heavy metals, while the use of ammonium nitrate instead of urea reduced the impact of fertilization on eutrophication and acidification caused by ammonia volatilization (Brentrup *et al.*, 2004b).

According to an analysis by Hillier *et al.* (2012), in temperate climate conditions, the most effective mitigating measures reducing GHG emissions from fertilization are as follows:

- where carbon content in the soil is low and nitrogen is applied at 150-200 kg, use of lower emission fertilizers;
- for N application rates up to 100 kg, increasing carbon content in the soil;
- for N at 200-300 kg, reducing N application rates.

According to Brock *et al.* (2012), GHG emissions can be reduced by decreasing levels of mineral fertilizers, provided that yields remain unchanged.

An effective strategy for protecting the environment against the harmful effects of fertilizers is thus to attempt to 'seal' the nutrient cycle on a global, regional, and local scale. Pro-environmental measures aim on the one hand to reduce resource utilization (*eg* to decrease the energy consumption of processes), and on other hand to limit losses of nutrients, during both their production (*eg* by using catalytic decomposition of N_2O to N_2 and O_2 in HNO_3 production) and their use (*eg* point, local, and foliar application of fertilizer, or the use of slow release or controlled release fertilizers).

CONCLUSIONS

1. The LCA can be helpful in fertilizer production and use in the following areas:
 - selecting environmentally-friendly technologies that optimally utilize resources for fertilizer production and use, *eg* by comparing alternative products and/or technologies;

- identifying ‘hot spots’ that generate the greatest impact on the environment in the entire life cycle of the product and preventing pollution;
- selecting significant indicators for assessing the effects of environmental activity;
- working out market strategies based on the ecological competitiveness of a given product;
- decision-making in industry and in governmental and non-governmental organizations;
- evaluating undertakings aimed at protecting the environment (life cycle assessment can be integrated with other tools that assist in the decision-making process).

2. An undoubted challenge for this tool when used in agricultural production is the need for relevance on both a global and local scale. Although the general principles of the life cycle assessment method have been included in ISO norms, detailed procedures for each of its phases are still under discussion, and the uncertainty arising from variability of measurements or from a lack of data or model assumptions remains one of the main problems significantly affecting the decision-making process, particularly with respect to input data.

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