Life Cycle Assessment to Evaluate the Environmental Impact of Arable Crop Production

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Kurzfassung

Ökobilanzen zur Bewertung von Umweltbelastungen im Pflanzenbau

Stichworte: Ackerbau, Pflanzenernährung, Ökobilanzierung

Einleitung

Eine nachhaltige und somit langfristig tragfähige Landwirtschaft muss gleichermaßen ökologische, ökonomische und soziale Aspekte berücksichtigen (Commission of the European Communities, 1999; UN-DSD, 2000). Diese Forderungen sind im wesentlichen aus den Erkenntnissen des Brundtland-Reports (WEC, 1987) abgeleitet, in dem Ökonomie, Ökologie und Soziales als gleichrangige Kernelemente der Nachhaltigkeit definiert sind. Übertragen auf die ackerbauliche Pflanzenproduktion bedeutet dies die Produktion qualitativ hochwertiger Grundnahrungsmittel, die einerseits das Einkommen der Landwirte und andererseits die Versorgung der Bevölkerung sicherstellen. Gleichzeitig soll eine möglichst umweltverträgliche Produktionsweise die Schädigung der natürlichen Ressourcen und der menschlichen Gesundheit minimieren.

Um dem Ziel einer nachhaltigen Landwirtschaft näher zu kommen, ist es daher wichtig, unterschiedliche Produktionsweisen vergleichen und beurteilen zu können. Zu diesem Zweck sind aussagekräftige Indikatoren für die verschiedenen ökonomischen, sozialen und ökologischen Wirkungen notwendig. Das Ziel dieser Arbeit ist es, zusammenfassende Indikatoren für den Teilaspekt der *ökologischen Wirkungen* ackerbaulicher Pflanzenproduktion unter besonderer Berücksichtigung der Pflanzenernährung zu entwickeln.

Die ganzheitliche Analyse der Umweltwirkungen ackerbaulicher Produktionssysteme ermöglicht beispielsweise, unter verschiedenen Anbauformen die aus ökologischer Sicht günstigste zu bestimmen oder Schwachstellenanalysen durchzuführen.

Um die Gesamt-Umweltwirkung eines Produktionssystems beurteilen zu können, müssen die unterschiedlichen Umweltwirkungen des Ackerbaus, wie zum Beispiel Nitratauswaschung, Ammoniakverflüchtigung oder Energieverbrauch, gemeinsam berücksichtigt werden. Es ist weiterhin wichtig zusätzlich zu den Umweltwirkungen, die durch den eigentlichen Ackerbau auf dem Feld verursacht werden, auch die Umweltwirkungen vorgelagerter Prozesse, wie zum Beispiel Produktion und Transport landwirtschaftlicher Betriebsmittel (Dünger, Pflanzenschutzmittel, Saatgut, Maschinen), mit in die Analyse einzubeziehen.

Die *Ökobilanzierung* bietet den Rahmen für genau diese Art der Untersuchung der Umweltwirkungen von Produkten unter Berücksichtigung des gesamten Produktionssystems

einschließlich vorgelagerter Prozesse. Das generelle Konzept der Ökobilanzierung wurde in verschiedene Ökobilanzmodelle umgesetzt (z.B. Goedkoop, 1995 [Eco-indicator’95]; Goedkoop & Spriensma, 1999 [Eco-indicator’99]; Steen & Ryding, 1993 [EPS]; BUWAL, 1998 [Schweizer Ökopunkte-Modell]). Diese Modelle wurden allerdings für industrielle Produkte und Produktionsverfahren entwickelt und weisen daher einige Schwächen auf, wenn sie zur Analyse landwirtschaftlicher Systeme angewendet werden.

Um eine umfassende Ökobilanzierung auch im Bereich des Ackerbaus zu ermöglichen, wurde die Ökobilanzmethodik in dieser Arbeit an die spezifischen Anforderungen der ackerbaulichen Pflanzenproduktion angepasst. Dabei liegt besonderes Augenmerk auf der Rolle des Düngemitteleinsatzes und seiner Umweltwirkungen.

Material und Methoden

Generell werden Ökobilanzen in vier Teilschritte untergliedert: (1) Festlegung des Ziels und des Untersuchungsrahmens, (2) Sachbilanz, (3) Wirkungsabschätzung und (4) Auswertung (SETAC, 1993; ISO, 1997).

Während der Phase der *Festlegung des Ziels und des Untersuchungsrahmens* wird das zu untersuchende System (z.B. Pflanzenproduktion), seine Funktion und Systemgrenzen beschrieben. In der darauf folgenden *Sachbilanz* werden der Ressourcenverbrauch und die auftretenden Emissionen ermittelt und in ihrer Menge erfasst. Die Sachbilanzdaten allein ermöglichen in der Regel noch keine abschließenden Schlussfolgerungen über die Vor- und Nachteile verschiedener Systeme (z.B. unterschiedlicher Düngungsniveaus in der Pflanzenproduktion). Zudem werden die Wirkungen der unterschiedlichen Emissionen und des Verbrauchs verschiedener Ressourcen in der Sachbilanz noch nicht berücksichtigt. Daher müssen die einzelnen Sachbilanzdaten im Teilschritt der *Wirkungsabschätzung* entsprechend ihrer Wirkungen auf natürliche Ökosysteme, menschliche Gesundheit und Verfügbarkeit von Ressourcen bewertet werden. In der anschließenden *Auswertung*sphase werden die Ergebnisse der Sachbilanz und Wirkungsabschätzung diskutiert und Schlussfolgerungen und Empfehlungen gegeben werden.

Zur Entwicklung einer Ökobilanzmethode speziell für die Analyse ackerbaulicher Produktionssysteme wurde in einem ersten Schritt eine in der Industrie etablierte Ökobilanzmethode (Eco-indicator’95; Goedkoop, 1995) auf ein Ackerbausystem angewendet, um die Besonderheiten der Ökobilanzierung von Ackerbausystemen darzustellen und die Schwachstellen verfügbarer Modelle zu ermitteln.

Aus dieser Studie wurde der Bedarf zur Entwicklung von Methoden für den Pflanzenbau in einzelnen Teilbereichen der Ökobilanzierung abgeleitet. Dies betrifft die Abschätzung diffuser, feldbürtiger Stickstoffemissionen (Ammoniak, Lachgas, Nitrat) in der Sachbilanz, sowie die Berechnung von Indikatoren für die Wirkungskategorien „Verbrauch abiotischer Ressourcen" (z.B. Rohphosphat, fossile Brennstoffe) und „Naturraumverknappung" durch Flächeninanspruchnahme innerhalb der Wirkungsabschätzung.

Um einen Vergleich der Umweltverträglichkeit unterschiedlicher Produktionsverfahren im Pflanzenbau zu ermöglichen, wurde zudem ein Wichtungsverfahren entwickelt, das die Berechnung von zwei zusammenfassenden Indikatoren (a) für den Verbrauch unterschiedlicher abiotischer Ressourcen und (b) für die übrigen Umweltwirkungen (Naturraumverknappung, Gewächshauseffekt, Versauerung, Eutrophierung) ermöglicht.

In einem letzten Schritt wurde die neu entwickelte Ökobilanzmethode an einem Fallbeispiel getestet. Bei dem untersuchten System handelt es sich um ein Weizenproduktionssystem mit unterschiedlicher Stickstoffdüngung.

Ergebnisse

(1) Die Anwendung des verfügbaren Eco-indicator'95-Modells (Goedkoop, 1995) auf ein Ackerbausystem hat gezeigt, dass in dieser Methode wichtige Umweltwirkungen (Ressourcenverbrauch, Flächennutzung) nicht berücksichtigt werden. Auch andere für die Industrie entwickelte Ökobilanzmodelle zeigen Schwächen bei der Anwendung auf landwirtschaftliche Systeme (z.B. fehlende Berücksichtigung von Nährstoffemissionen im Eco-indicator'99, Goedkoop & Spriensma, 1999).

(2) Auf der Basis einer Literaturstudie wurden Methoden zur Abschätzung der äußerst variablen Ammoniak- (NH_3), Lachgas- (N_2O) und Nitrat-(NO_3)-Verluste ausgewählt und zu einem Schätzrahmen zusammengefasst. In diesen Schätzrahmen wurden Methoden von Horlacher & Marschner (1990) und ECETOC (1994) für NH_3, von Bouwman (1995) für N_2O und der DBG (Deutsche Bodenkundliche Gesellschaft, 1992) für NO_3 integriert. Dadurch werden bei der Abschätzung der Nährstoffverluste wichtige Einflussfaktoren auf die Emissionen, wie zum Beispiel Boden-, Klima- und Bearbeitungsparameter (z.B. Düngung), berücksichtigt.

(3) Für die Umweltwirkung „Verbrauch abiotischer Ressourcen" wurde im Unterschied zu anderen Ansätzen in dieser Arbeit der Verbrauch solcher Ressourcen, die unterschiedliche Funktionen haben und daher nicht durch einander substituierbar sind (z.B. Rohphosphat, Kalisalz, Öl), jeweils als separates Umweltproblem behandelt. Eine abschließende

Zusammenfassung nicht-äquivalenter Ressourcen zu einem aggregierten Indikator für den Gesamt-Ressourcenverbrauch erfolgte nach einem zusätzlichen Wichtungsschritt, der in dieser Arbeit entwickelt wurde.

(4) Im Ackerbau werden große Flächen für die Pflanzenproduktion in Anspruch genommen. Diese Flächennutzung führt zu einer Verknappung des Naturraums und beeinflusst zum Beispiel die Biodiversität in erheblichem Ausmaß. In der in dieser Arbeit entwickelten Ökobilanzmethode umfasst die Abschätzung der Umweltwirkungen durch die Flächeninanspruchnahme zwei Aspekte: (a) die Größe der Fläche, die für eine bestimmte Zeit genutzt wird und (b) die Intensität dieser Nutzung, das heißt wie sehr eine bestimmte Nutzungsart den Naturraum beeinflusst. Während der erste Aspekt direkt als physikalische Größe ermittelt werden kann, wird für den zweiten Punkt ein geeigneter Indikator benötigt. Das Hemerobie-Konzept liefert genau einen solchen Indikator, da es zu dem Zweck entwickelt wurde, den Natürlichkeitsgrad einer Fläche zu beschreiben (Kowarik, 1999). Im Hemerobie-Konzept werden genutzte Flächen (z.B. bebautes Gebiet oder extensive Weide) entsprechend ihrer Abweichung von einem unbeeinflussten Naturzustand in Klassen unterschiedlicher Natürlichkeit eingeteilt. In dieser Arbeit wurde das Hemerobie-Konzept genutzt, um eine neue Methode zur Wirkungsabschätzung der Naturraumbeanspruchung in Ökobilanzen einzubinden.

(5) Eine Bewertung des Gefährdungspotentials der unterschiedlichen Umwelteffekte wurde vorgenommen, um eine Zusammenfassung der Einzelindikatoren je Effekt zu einem aggregierten Umweltindikator zu ermöglichen. Diese Bewertung, in Ökobilanzen als Wichtung bezeichnet, wurde durch einen Vergleich zwischen dem Ist-Zustand eines jeden Umwelteffektes mit einem definierten Soll-Zustand für den entsprechenden Effekt durchgeführt („Distance-to-Target“ Prinzip). In dieser Studie wurden international akzeptierte Umweltziele für die Definition des Soll-Zustands genutzt, da diese einen Konsens zwischen Wissenschaft, Wirtschaft und Gesellschaft reflektieren.

(6) Während bis hierher die Ergebnisse der Methodenentwicklung im Vordergrund standen, werden im Folgenden die Ergebnisse der Anwendung dieser Methoden auf ein Fallbeispiel dargestellt. Dabei wurden die Umweltwirkungen unterschiedlicher N-Düngungsraten in der Winterweizenproduktion untersucht. Die Anwendung zeigte, dass die neu entwickelte Ökobilanzmethode die Erfassung und Bewertung der Umwelteffekte ermöglicht, die für Ackerbausysteme und insbesondere die Pflanzenernährung relevant sind. Die Berücksichtigung des gesamten Produktionssystems inklusive der Bereitstellung der Betriebsmittel erlaubt, die bilanzierten Umweltwirkungen ihren Ursprüngen zuzuordnen

und auf dieser Basis Verbesserungspotentiale aufzuzeigen. Die abschließende Aggregierung der Einzelwirkungen zu einem Umweltindikator macht einen Vergleich der Vorzüglichkeit unterschiedlicher Pflanzenbausysteme aus Umweltsicht möglich. Das Fallbeispiel zeigte weiterhin, dass die Gesamt-Umweltwirkung bezogen auf eine Tonne Weizenkorn sowohl bei N-Gaben, die den Pflanzenbedarf überschreiten, als auch bei unterlassener N-Düngung stark ansteigt. Im ersten Fall bildet der Beitrag zur Eutrophierung von Gewässern das größte Umweltproblem, im zweiten Fall die Naturraumbeanspruchung. Von reduzierter zu ökonomisch optimaler N-Düngung steigt die Gesamt-Umweltbelastung nur langsam an. Bei ökonomisch optimaler N-Düngung (192 kg N/ha) trägt die Eutrophierung von Gewässern am stärksten zur Gesamtbelastung bei, während die übrigen Umwelteffekte etwa gleiche Beiträge zur Gesamtbelastung aufweisen.

Abstract

Life Cycle Assessment to evaluate the Environmental Impact of Arable Crop Production

Keywords: arable farming, plant nutrition, life cycle assessment

Introduction

Agriculture is expected to comply with sustainability principles, i.e. to be economically competitive, to produce high quality food in sufficient quantities at affordable prices, and to be environmentally benign (Commission of the European Communities, 1999; UN-DSD, 2000). This understanding is mainly based on the results of the Brundtland-Commission (WEC, 1987), which defines economy, ecology and society as equal core-elements of sustainability. In order to evaluate the sustainability of agricultural production, it is necessary to have appropriate economic, social and environmental indicators in place.

The main objective of this study is to develop indicators for the *environmental impacts* of arable crop production. A comprehensive environmental analysis of arable products and production systems is for example important to find the most environmental friendly production alternatives and to detect the environmental hotspots in the systems.

However, in order to evaluate the entire environmental burden associated with arable production it is necessary to analyze all its various environmental impacts like nitrate leaching, ammonia volatilization or energy consumption. Furthermore, it is important to consider the impacts occurring due to the agricultural activities in the field together with those impacts, which are connected to the production and transportation of required farm inputs like mineral fertilizers, seeds or machines.

The *Life Cycle Assessment* (LCA) methodology provides the framework for such an environmental analysis of products that takes into account the entire production system. Different operational LCA approaches have been developed based on this framework (e.g. Goedkoop, 1995 [Eco-indicator'95]; Goedkoop & Spriensma, 1999 [Eco-indicator'99]; Steen & Ryding, 1993 [EPS]; BUWAL, 1998 [Swiss Eco-points]). However, all of these models were primarily designed for industrial applications and thus, show some difficulties when applied to agricultural systems.

In order to enable a comprehensive Life Cycle Assessment of arable farming products, this study suggests adjustments of the LCA methodology to the specifics of arable crop production with a special focus on plant nutrition and in particular on fertilizer use.

Material and methods

According to SETAC (Society for Environmental Toxicology and Chemistry, 1993) and ISO (International Organization for Standardization, 1997), LCA is divided into four steps, which are (1) goal and scope definition, (2) inventory analysis, (3) impact assessment and (4) interpretation.

During *goal and scope definition* the system under investigation (e.g. plant production), its function, and boundaries are described. In the subsequent *Life Cycle Inventory* the resource consumption and emissions associated with the system are compiled. The data of the inventory as such do not allow comparisons to be made between different systems (e.g. different fertilizer regimes in plant production). The potential environmental impact of the various emissions and resource consumption is not considered in this phase. Therefore, during the following *Life Cycle Impact Assessment* the inventory data are evaluated with regard to their potential to harm natural ecosystems, human health, and resources. Finally, in the *interpretation* phase, the inventory and impact assessment results are discussed and conclusions are drawn in order to define options to improve the environmental performance of the product under investigation.

The first step to convert this general LCA framework into an operational LCA method particularly suitable for arable crop production was to apply an established LCA tool (Eco-indicator’95, Goedkoop, 1995) to an arable farming system, in order to analyze the specifics of such systems and to determine the shortcomings of available LCA tools. In particular the estimation of diffuse, on-field nitrogen emissions like ammonia, nitrous oxide and nitrate during the Life Cycle Inventory phase and the impact assessment within the impact categories ‘consumption of abiotic resources’ and ‘land use’ were found to be inappropriately considered in recent LCA studies.

Furthermore, an aggregation (in LCA = weighting) method was developed, which enables the calculation of two summarizing indicators (a) for the depletion of abiotic resources and (b) for impacts on natural eco-systems and human health (land use, climate change, acidification, eutrophication).

As the final step the new LCA method was tested in a case study. The system studied is a wheat production system with different N fertilizer rates.

Results

(1) The application of an existing LCA tool (Eco-indicator’95), which is established in the industry, has shown some problems, when applied on arable farming systems. Particular

shortcomings are the missing consideration of the environmental impacts ‘consumption of abiotic resources’ and ‘land use’, which are important for arable production systems.

(2) Based on a literature study, structured methods for the estimation of diffuse, on-field emissions of ammonia (NH_3), nitrous oxide (N_2O) and nitrate (NO_3) were selected. Methods developed by Horlacher & Marschner (1990) and ECETOC (1994) for NH_3 losses, Bouwman (1995) for N_2O emissions and DBG (1992) for NO_3 leaching were found to be appropriate in order to derive reasonable estimates of these highly variable emissions as an input to LCA studies. These estimation methods consider important soil, climate and management (e.g. plant nutrition) parameters.

(3) In contrast to other approaches, this study suggests to treat the consumption of resources, which are not substitutable by each other, as separate environmental problems. A final aggregation of non-equivalent resources like phosphate rock and fossil fuels into a summarizing resource depletion indicator is found to be only possible after an explicit weighting procedure, which has been developed in this study.

(4) Arable farming uses huge quantities of land for crop production. An assessment of the environmental impacts of land use in LCA has to include two dimensions: (a) the size of an area used for a certain period of time and (b) the potential of a specific land use type to degrade the naturalness of the area under use. Whereas the first aspect can be directly expressed as a physical quantity, the latter aspect needs an appropriate indicator. The Hemeroby concept provides such an indicator, since this concept was specifically developed in order to evaluate the level of naturalness of land area. Hemeroby is a measure for the human influence on ecosystems, which defines the level of naturalness of different land use types (e.g. urban area or extensive pasture) according to their deviation from a natural reference situation. This study employs the Hemeroby concept in order to assess the impacts of different land use types within LCA.

(5) An evaluation of the different environmental effects that are relevant to arable production regarding their potential to harm the environment is performed in order to enable an aggregation of the separate indicators per effect into two summarizing environmental indicators: (a) for abiotic resources and (b) for impacts on ecosystems and human health. This weighting was realized by a comparison of the current status of each effect with defined target values for the respective effects (“distance-to-target principle”). This study suggests internationally agreed environmental targets to be employed in this procedure because they represent a consensus of science, economy and society.

(6) After these methodological developments the method was tested in a case study. In this case study the environmental impact of different N fertilizer rates in winter wheat production was analyzed. The new LCA method has shown to be suitable to evaluate environmental impacts, which are relevant to arable crop production with a special focus on plant nutrition aspects. The consideration of the entire production system enables to trace back the various environmental impacts to their sources. On that basis it is possible to suggest options for environmental improvements. The inclusion of an aggregation procedure makes it possible to compare the environmental preference of alternative arable crop production systems.

The case study revealed that the aggregated environmental impact per tonne of wheat grain increases dramatically at N rates exceeding the crop demand and at zero N fertilization. In the first case aquatic eutrophication was the major problem, whereas in the latter case this is land use. From reduced to economic optimum N rates the environmental indicator values increased only slightly. At economic optimum N fertilization (192 kg N/ha) aquatic eutrophication contributed most to the aggregated indicator; terrestrial eutrophication, acidification, climate change and land use show similar contributions to the aggregated value.

Contents

I. General introduction

1. Problem setting

1.1 Background

Agriculture is expected to comply with the principles of sustainability (Commission of the European Communities, 1999; UN-DSD, 2000). According to the Brundtland report (WCED, 1987), sustainability can be defined as “any development, which meets the needs of the present without compromising the ability of future generations to meet their needs”. Sustainable development has to balance economic, social and environmental aspects. For agriculture this means to ensure a sufficient income for the farmers, to produce high quality food in sufficient quantities at affordable prices, and at the same time to be environmentally benign. However, if the internationally accepted concept of sustainability should be implemented into agricultural practice, it is necessary to have appropriate economic, social and environmental indicators in place (Christen & O’Halloran-Wietholtz, 2001). Particularly if there are choices between alternatives in agriculture like different farming concepts (organic, integrated, conventional), production intensities (intensive, reduced) or options in the use of farming inputs (fertilizers, plant protection substances, application techniques), indicators are needed to evaluate the sustainability of the alternatives on a scientific basis.

This study focuses on the **environmental aspects of sustainability**. Target was to develop an indicator system, which enables the evaluation of **arable farming products** (i.e. crops) or **production systems** (e.g. wheat production at different production intensities) from an environmental point of view with a specific focus on plant nutrition aspects. An economic or social assessment of arable production systems is not aim of this study.

1.2 Environmental assessment of arable crop production

The environmental impacts of arable farming have often been investigated focused on specific single impacts like nitrate leaching (Engels, 1993; Goulding et al., 2001), ammonia volatilization (Bussink, 1996; ECETOC, 1994) or energy consumption (Küsters & Lammel, 1999). However, these single impacts are only part of the entire environmental burden connected to arable production. Furthermore, the different impacts may contribute to different environmental problems, which itself may have a different importance for the environment. If for example one of two alternative crop production systems shows higher nitrate leaching rates, whereas the other system leads to higher ammonia volatilization, it would be necessary to evaluate the potential of these emissions to contribute to the environmental problem of eutrophication. If after such an evaluation procedure one system turns out to be preferential with regard to eutrophication, the importance of this contribution to the eutrophication problem in comparison to other environmental impacts like the release of greenhouse gases would still remain unclear. Depending on the production system under investigation, arable crop production may contribute to various environmental effects including the depletion of non-renewable resources, the physical degradation of natural areas, climate change, acidification, eutrophication or contamination of soil and water with toxic substances (Audsley et al., 1997; Brentrup et al., 2001). Therefore, a comprehensive environmental assessment of arable production systems needs to consider all relevant environmental impacts and their importance simultaneously.

Furthermore, it is important to analyze the environmental impacts connected to agricultural crop production from a holistic perspective, “wherever and whenever these impacts have occurred, or will occur” (Guinèe et al., 2001). That means, the environmental analysis of crop production systems has to go far beyond the farm gates and has to consider more than only the environmental impacts associated to the on-field activities like tillage or fertilizer application. For instance for the analysis of arable products or production systems this means to take into account the production and transportation of required farming inputs (e.g. fertilizers, plant protection substances, seeds and machines) in addition to the on-farm activities (e.g. Anderson & Ohlsson, 1999; Brentrup et al., 2001; Eide, 2002). The environmental impacts connected to these processes as well as to the extraction and supply of necessary raw materials (e.g. fossil fuels, minerals) should be accounted for in a comprehensive environmental assessment of arable products.

The **Life Cycle Assessment (LCA)** methodology is especially designed for the environmental analysis of products (Heijungs et al., 1992a; Consoli et al., 1993; ISO, 1997). The most specific characteristic of the LCA methodology is the “life-cycle thinking” (Finnveden, 1998), i.e. putting the product into focus of the investigation and considering the entire network of main and sub-processes relevant to the production, use and disposal of the product. Subsequently, this network of product-related main and sub-processes is called the “product system”.

In the following a short description of the general LCA concept is given.

1.3 The Life Cycle Assessment (LCA) concept

LCA is defined as an inventory and valuation of all potential environmental impacts related to a production system. In LCA, impacts on natural eco-systems, human health and natural resources are considered.

First attempts to analyze entire product systems were conducted in the 1960s in the United States with a focus on energy consumption (Curran, 1996). The development of the recent LCA methodology started in the 1990s. Milestones of this development are the SETAC (Society of Environmental Toxicology and Chemistry) “Code of Practice” (Consoli et al., 1993) and the publication of ISO standards on LCA (ISO, 1997, 1998a, 1998b, 2000). According to the general concept and principles of the LCA methodology provided by these publications, LCA is divided into four phases, which are (1) goal and scope definition, (2) inventory analysis, (3) impact assessment, and (4) interpretation.

After goal and scope definition, the *inventory analysis* (= Life Cycle Inventory, LCI) compiles all environmental data relevant to the production system under investigation and relates them to a common unit, which is the functional unit of the study (e.g. production of 1 ton of wheat grain). Environmental data means data on resource consumption, land use and emissions to air, to water and to soil (e.g. kg CO_2, CH_4 or N_2O per ton of grain). This list of inventory data is the most objective result of an LCA study. However, a list of single substances is difficult to interpret and especially in comparative LCA studies the LCI data usually do not allow to draw conclusions on the relative environmental preference of one or another alternative.

The *impact assessment* (= Life Cycle Impact Assessment, LCIA) aims at further evaluation of the LCI data. Within the LCIA the various inventory data are summarized into indicators for environmental effects (e.g. CO_2-equivalents for climate change or SO_2-equivalents for acidification). The result of this first step in LCIA (= characterization) is a list of indicator values for environmental effects, which gives the environmental profile of the product under investigation.

During the subsequent normalization step each of these indicator values is divided by a reference value, as for instance the respective indicator values for Europe (e.g. kg CO_2-equiv. per ton of grain / CO_2-equiv. per year in Europe). This normalization is performed in order to get information on the relevance of an impact due to the analyzed product in comparison to a reference value. Furthermore, by normalization the indicator values are getting dimensionless, which is a prerequisite for the following weighting step. The weighting step aims at a final aggregation across all impact categories to an overall environmental indicator. Therefore, each normalized indicator value is multiplied by a weighting factor, which represents the potential of the respective impact category to damage the environment. Figure 1 gives an overview of the general life cycle impact assessment methodology and its different elements.

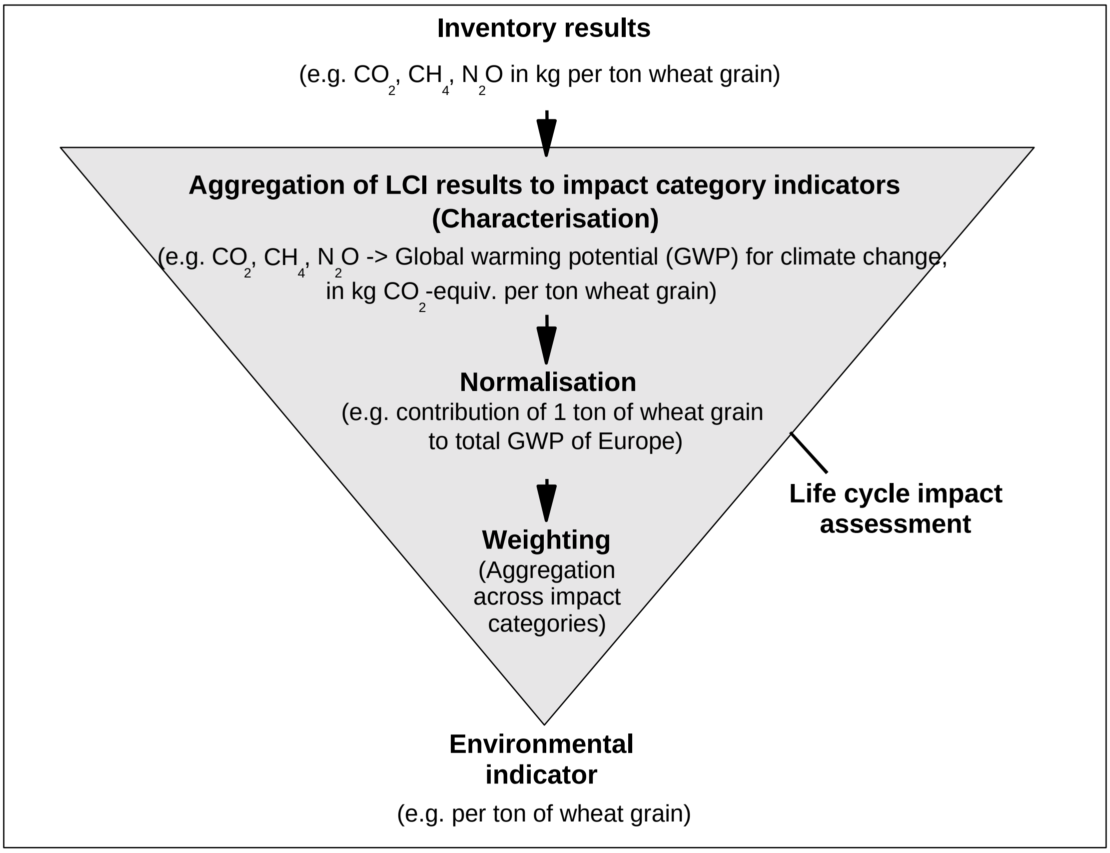

Figure 1: The general Life Cycle Impact Assessment procedure

Many parts of the impact assessment methodology are far from settled from a scientific point of view, but in particular the weighting step is still under an intensive debate. According to ISO (2000) weighting "shall not be used for comparative assertions disclosed to public". Especially this clause of the ISO normation has been controversially discussed (Hertwich & Pease, 1998; Marsmann et al., 1999). However, in LCA studies comparing alternative products or systems the weighting of different environmental impacts is indispensable to finally conclude on the environmental preference of one alternative. If weighting is not performed within LCA, users of LCA studies will tend to weigh the system's contribution to different environmental effects on their own. Instead of this very subjective individual way of weighting, a set of generic, study-independent weighting factors, as they are for instance proposed in this study helps the user of an LCA study to interpret complex environmental data sets on a more transparent and documented basis. However, an evaluation and aggregation of different environmental effects will never be possible on a pure scientific basis and always includes more values and societal preferences than the other parts of LCA (Bischoff, 1994). Therefore, it is particularly important to present LCA results at different levels of detail, i.e. as inventory data, as indicators for impact categories ("environmental profile") and as normalization and fully aggregated weighting results.

Finally, in the *interpretation* phase, the inventory and impact assessment results are discussed and conclusions should be drawn in order to describe options to improve the environmental performance of the product under investigation.

1.4 Problems with the application of the LCA concept on arable crop production

SETAC (Consoli, 1993) and ISO (1997) rather describe a methodological framework and the general concept of LCA and not an operational method for the environmental assessment of products. Based on the basic methodology, different ready-to-use LCA methods have been developed (BUWAL, 1998; Goedkoop, 1995; Goedkoop & Spriensma, 1999; Guineé et al, 2001; Heijungs et al., 1992a; Steen, 1999). These LCA methods are mainly tailored to industrial applications, because the majority of LCA studies were conducted on industrial products and processes (Grotz & Scholl, 1996). Therefore, their application on arable farming systems reveals some difficulties (Brentrup et al., 2001) like the missing consideration of environmental impacts, which are important for agricultural systems (e.g. land use, resource depletion in Goedkoop, 1995; nutrient emissions in Goedkoop & Spriensma, 1999). In particular with regard to LCA studies comparing alternative plant nutrition strategies (e.g. fertilizer types or rates) these impacts are of special importance (e.g. Anderson & Ohlsson, 1999; Cederberg & Mattson, 1999; Küsters & Jenssen, 1998). More basic problems of these LCA methods concern the general impact assessment procedures for important environmental effects (e.g. acidification and eutrophication; Goedkoop, 1995) or a lack of transparency in the weighting procedure (Steen, 1999). Furthermore, some of the models do not enable to calculate an overall environmental indicator (Guineé et al, 2001; Heijungs et al., 1992a) or are restricted in their validity to only one country (BUWAL, 1998, for Switzerland).

In contrast to industrial production systems, agricultural production and in particular arable crop production show important differences. Most of the environmental impacts associated to industrial processes are directly measurable as material-/energy-inputs and as point emissions released via chimneys or effluents (Bischoff, 1994). On the contrary, in arable farming systems many relevant emissions (e.g. NH_3, N_2O, NO_3) are released diffusely on the field and are highly variable depending on site-specific conditions (soil, climate) and plant nutrition. Since these diffuse emissions have a strong impact on the outcomes of an LCA study (Audsley et al., 1997; Brentrup et al., 2001), reliable measurements or estimates are of particular importance (Brentrup et al., 2000). A further difference of arable farming systems to most industrial production systems is the high demand on specific resources, of which phosphate rock, potash and land are particularly important.

Taking into account the general problems with the recently available approaches to convert the basic LCA concept into an operational LCA method together with the specific environmental impacts associated to arable farming and particularly to plant nutrition, an updated LCA method specifically adjusted to arable crop production needs to be developed.

1.5 The potential of LCA to analyze the environmental impacts of arable crop production

Taking into account the increasing demand of public and policy on improved transparency and traceability in the food chain, LCA will play an important role. If an scientifically approved LCA method would exist, it would be for instance possible to compare different plant nutrition strategies in arable crop production (e.g. fertilizer types, fertilizer rates, application techniques) from an environmental point of view. Such an analysis should consider all relevant environmental impacts due to the entire production system needed to produce the crop, i.e. including resource extraction, production and transportation of farming inputs (fertilizers, plant protection substances, seeds, machines) and the on-field operations. The study would enable to detect the most relevant environmental problems (“hot-spots”) and to trace back the impacts to their sources. On that basis it would be possible to search for efficient measures in order to improve the overall environmental performance of the crop production system under investigation. Furthermore, comparisons of alternatives in crop production could be objectified. However, no LCA method capable to analyze the environmental impacts of arable crop production appropriately is available. Today, LCA studies including arable crop production analyze different environmental impacts separately from each other (Audsley et al., 1997; Andersson & Ohlsson, 1999; Cederberg & Mattson, 1999; Eide, 2002). Such a procedure leads to separate sub-results for different environmental problems (e.g. land use, climate change, toxicity etc.). Since in comparative LCA studies usually none of the alternatives under investigation shows lowest impacts in all environmental effects, this procedure does not allow conclusions upon the relative environmental preference of one or another production alternative under comparison. Furthermore, it is not possible to make suggestions upon the most efficient measures for improving the overall environmental performance of a product, because no information about the significance and the importance of the product's contribution to different environmental effects is included in the results.

1.6 Objective of the study

Objective of this study was to develop a LCA method, which considers all environmental impacts relevant to arable crop production with specific focus on plant nutrition and which allows aggregating the environmental impacts into a summarizing environmental indicator in order to enable conclusions upon the environmental preference of alternative crop production systems.

To make this final aggregation across the different impacts possible, the developed LCA method includes normalization and weighting. However, this LCA method not only provides the user with a fully aggregated environmental indicator, but also enables the calculation of indicators separately for each single environmental effect relevant to agricultural crop production. The explicit documentation of these separate indicator results for environmental effects like climate change or acidification is important, because it increases the transparency of the method and makes conclusions on the level of separate environmental themes possible. A direct conversion of inventory data (e.g. CO_2 or NOx emissions) into environmental indices (e.g. Swiss Eco-points; BUWAL, 1998) lacks transparency and mixes up environmental principles (characterization step) with the value-based weighting step.

In the following the separate steps taken to achieve the previously outlined objective are described.

2. Structure of the study and objectives of the separate steps

2.1 Application of an existing LCA approach to analyze the environmental impacts of arable crop production

In a first step the general suitability of the existing LCA methodology to analyze and evaluate the environmental impacts of arable farming systems is investigated (**Chapter II**). As a representative for frequently used LCA tools, the Dutch LCA method "Eco-indicator'95" (Goedkoop, 1995) is applied in order to compare the environmental impact of different forms of nitrogen fertilizers in sugar beet production. This study introduces the general LCA methodology and shows its potential and limitations to analyze and evaluate the environmental impacts of arable farming systems.

The objectives of Chapter II are:

- to examine the potential of the general LCA concept to analyze and evaluate the environmental impacts of arable farming systems
- to investigate the applicability and suitability of a ready-to-use and frequently used LCA method (Eco-indicator'95) to arable farming systems
- to define needs for improving the conversion of the general LCA concept into an operational LCA method focused on the specifics of arable farming and in particular of plant nutrition

2.2 Improving the Life Cycle Inventory: Estimation of diffuse, on-field nitrogen emissions in LCA studies on arable crop production

Furthermore this case study shows in accordance with other LCA studies on agricultural production systems (e.g. Audsley et al., 1997; Küsters & Jenssen, 1998; Cederberg, 1998) that particularly the diffuse, on-field emissions of ammonia (NH_3), nitrous oxide (N_2O) and nitrate (NO_3) often contribute considerably to the LCA results. For most LCA studies it is not possible to carry out reliable measurements of these emissions. Therefore, methods to estimate emission rates as input to the Life Cycle Inventory are requested. **Chapter III** suggests methods, which enable the calculation of study-specific estimates of NH_3, N_2O and NO_3 emission rates under consideration of important management (e.g. plant nutrition), soil and climate conditions.

The objectives of Chapter III are:

- to enable LCA practitioners to calculate study-specific estimates of NH_3, N_2O and NO_3 emissions under consideration of important soil, climate and management parameters
- to illustrate the suggested estimation procedures at an example
- to compare the suggested estimation methods with other estimation procedures applied in recent LCA studies on arable farming systems

2.3 Improving the Life Cycle Impact Assessment: Impact assessment of resource consumption and land use

In addition to specific difficulties in compiling Life Cycle Inventory data for arable farming systems, a need to improve the LCA methodology has been also identified for the impact assessment phase. In particular for the impact categories "depletion of abiotic resources" and "land use" improved impact assessment procedures need to be developed.

The problem related to the *depletion of abiotic resources* (e.g. minerals or fossil fuels) is their decreasing availability for future generations (**Chapter IV**). For arable farming systems this issue is of special importance for two reasons. First of all, arable farming is the most important consumer of phosphate rock and potash salts, which are used as raw materials to produce phosphorus (P) and potassium (K) fertilizers (EFMA, 1999; USGS, 2002). P and K are essential plant nutrients in crop production. Since the reserves of sufficiently concentrated phosphate rock and potash salts are limited (USGS, 2002), the consumption of these minerals is a sustainability issue and should be accounted for in LCA (Consoli et al., 1993). Secondly, the production of mineral nitrogen fertilizers consumes considerable amounts of fossil fuels, which is also a limited resource.

In recent impact assessment methods for resource consumption (Goedkoop & Spriensma, 1999; Guinée, 2001) all inventory data on the consumption of resources are directly aggregated into one resource depletion indicator neglecting the different functions of the resources (e.g. oil and P). However, according to the general LCA methodology, the aggregation to impact categories (characterization) "should be based on scientific knowledge about environmental processes" (Consoli et al., 1993). Transferred to resource consumption this means an aggregation of different resources should consider their function and would be only possible, if these resources can be replaced by each other (e.g. oil, gas, and coal as fossil fuels).

A further aggregation across different impact categories is only possible after normalization and weighting of the indicator values. In order to ensure a consistent LCA methodology, this general procedure should be also applied on resources.

<u>The objectives of Chapter IV are:</u>

- to develop a new impact assessment method for the consumption of abiotic resources
- to assign different resources to separate impact categories according to their function and to calculate normalization values and weighting factors in order to enable a final aggregation into one summarizing resource depletion indicator

Land use describes in LCA the environmental impacts of occupying, reshaping and managing land for human purposes (e.g. arable farming, housing, and traffic). A major environmental consequence of this anthropogenic land use is a decreasing availability of habitats and thus, a decreasing diversity of wildlife species (**Chapter V**). Two aspects of land use determine the extent of the environmental impact, which are (1) the size of the area used for human purposes and (2) the type or intensity of land use. The size of an area under use for a certain time can be directly measured as a physical quantity (e.g. in m^2*year). However, to evaluate the impact of different types of land use (e.g. sealed urban area vs. extensive meadow) is much more complicated and controversial. In current impact assessment methods for land use this evaluation is mainly based on the number of species determined for a specific land use type compared to the number of species in a natural reference situation (Goedkoop & Spriensma, 1999; Köllner, 2000). This procedure encounters two main problems. First, it is difficult to determine a common natural reference situation, because the diversity of species per area varies already naturally by a factor of 10-15 within Europe (BfN, 1999). Secondly, not only the number of species, but also the structure of the species community (e.g. share of indigenous species and neophytes) is decisive for the assessment (Kretschmer et al., 1997).

<u>Therefore, the objectives of Chapter V are:</u>

- to develop an impact assessment method for the impact category "land use", which takes into account (1) the size of the area used for a certain period of time and (2) the intensity of different land use types
- to treat "natural land" like a resource and to develop intensity factors (= characterization factors) for different land use types, which represent the potential of each land use type to reduce the availability of this resource
- to propose a set of characterization factors, normalization values and weighting factors for land use in order to enable the consistent consideration of land use impacts within LCA

2.4 Development of an operational LCA method including normalization and weighting

In order to enable conclusions upon the overall environmental preference of alternative products or processes, an LCA method should include methods, which allow comparing and evaluating a product's contribution to different environmental effects like the consumption of phosphate rock, land use or climate change. In LCA, normalization and weighting are those steps, which enable this comparison and evaluation of different environmental impacts. Whereas normalization is relatively straightforward and depends mainly on the availability of recent data of good quality, the weighting step is always controversially discussed (Hertwich & Pease, 1998; Marsmann et al., 1999).

The main challenge of weighting the different impacts on different parts of the environment (e.g. plants, animals, humans and resources) is to integrate natural science and subjective values, and therefore needs social consensus. Different approaches to weigh different environmental problems have been applied (e.g. expert panels, monetary evaluation, analysis of environmental targets). However, weighting based on the analysis of widely agreed environmental targets like the Kyoto protocol for climate change (UN-FCCC, 1998) represents best a broad consensus, because it considers at the same time scientific knowledge together with economic and social aspects of the respective impacts.

Since the development of an agreed LCA approach would be important for instance in order to contribute to more transparency and traceability in arable production,

<u>the objectives of Chapter VI are:</u>

- to develop a complete and ready-to-use LCA approach, which is adjusted to the specifics of arable farming systems with focus on plant nutrition
- to combine the new characterization approaches for abiotic resource consumption and land use with selected characterization methods for other environmental effects relevant to arable crop production (climate change, toxicity, acidification and eutrophication)
- to develop a consistent set of normalization values and weighting factors, which enable the calculation of an aggregated environmental indicator and an aggregated resource depletion indicator
- to calculate weighting factors based on the analysis of widely accepted environmental agreements in order to integrate scientific knowledge about the environmental effects and social perceptions of the damage potential of these effects

2.5 Application of the developed LCA method in a case study

As an example, the developed LCA method is applied to investigate the environmental impact of different N fertilizer application rates in winter wheat production (**Chapter VII**). Arable crop production traditionally targets the economic optimum production intensity, which usually only partly considers environmental impacts. At the economic optimum production intensity farming inputs like fertilizers are used most efficient from an economic point of view. However, the economic optimum intensity may not equate with the most environmental friendly production intensity. This study tested the suitability of the developed LCA method to investigate and to compare different production intensities from an environmental viewpoint.

<u>The objectives of Chapter VII are:</u>

- to examine the environmental impact of different nitrogen fertilizer application rates in cereals production under Western European conditions
- to compare the eco-efficiency of an economic optimum wheat production intensity with sub-optimum and super-optimum production intensities
- to find environmental hot-spots in the wheat production systems under investigation and to suggest options to improve their environmental performance

- to test the suitability of the developed LCA method to investigate and evaluate the environmental impacts of arable crop production systems in a case study

II. Application of the Life Cycle Assessment methodology to agricultural production: an example of sugar beet production with different forms of nitrogen fertilizers

(European Journal of Agronomy, 14 (2001) 221-233)

Abstract

The suitability of the Life Cycle Assessment (LCA) methodology to analyze the environmental impact of agricultural production is investigated.

The first part of an LCA is an inventory of all resources used and emissions released due to the system under investigation. During the following step, the Life Cycle Impact Assessment the inventory data were analyzed and aggregated in order to finally get one index representing the total environmental burden.

For the Life Cycle Impact Assessment the Eco-indicator 95 method has been chosen, because this is a well documented and regularly applied impact assessment method. The resulting index is called Eco-indictor value. The higher the Eco-indicator value is the stronger is the total environmental impact of an analyzed system.

A sugar beet field experiment conducted in northeastern Germany was chosen as an example for the analysis. In this experiment three different nitrogen fertilizers (calcium ammonium nitrate = CAN, urea ammonium nitrate solution = UAN, urea) were used at optimum N rates.

The obtained Eco-indicator values were clearly different for the N fertilizers used in the sugar beet trial. The highest value was observed for the system where urea was used as N source. The lowest Eco-indicator value has been calculated for the CAN system. The differences are mainly due to different ammonia volatilization after application of the N fertilizers. For all systems the environmental effects of acidification and eutrophication contributed most to the total Eco-indicator value.

The results show that the LCA methodology is basically suitable to assess the environmental impact associated with agricultural production. A comparative analysis of the system's contribution to global warming, acidification, eutrophication and summer smog is possible. However, some important environmental issues are missing in the Eco-indicator 95 method (e.g. the use of resources and land).

1. Introduction

The intensity of arable farming and in particular the use of mineral nitrogen fertilizers are reasons for a continuous debate on the environmental impacts of agriculture. The majority of the investigations focus on specific environmental aspects associated with agriculture, such as ammonia volatilization, nitrous oxide emissions or nitrate leaching (ECETOC, 1988; Sommer, 1992; Engels, 1993; ECETOC, 1994; Granli & Bockman, 1994; Kroeze, 1994; Bach & Becker, 1995; Bouwman, 1995; Bussink, 1996; Kaiser et al., 1996). However, in order to examine and compare the entire environmental burden connected with agricultural production systems it is necessary to consider all environmental impacts at the same time.

Life Cycle Assessment (LCA) is a methodology to assess all environmental impacts associated with a product, process or activity by identifying, quantifying and evaluating all resources consumed, and all emissions and wastes released into the environment. Today LCA is predominantly applied on industrial products or processes. In this study the suitability of the LCA methodology to analyze the environmental impact of agricultural production is investigated. This analysis was performed on data from sugar beet field experiments.

The sugar beet system with different forms of nitrogen fertilizers has been chosen as a case study in order to test whether the LCA method is capable to find out differences in the environmental performance even of very similar systems. Furthermore the sugar beet system was chosen as an example because all relevant information was available to derive a complete data inventory for the LCA. This example was chosen because all relevant information was available to draw up a complete Life Cycle Inventory (e.g. data on fertilizer production, means and routes of transportation, on-field machinery use, soil and climate data, yields etc.).

2. Materials and methods

The LCA concept consists of four major steps: (1) Goal and scope definition, (2) Life Cycle Inventory, (3) Life Cycle Impact Assessment and (4) Interpretation.

2.1 Goal and scope definition

The first component of an LCA is the definition of the *goal and scope* of the analysis. This includes the description of the analyzed system and the definition of system boundaries. Furthermore, a reference unit, to which all environmental impacts are related, has to be defined. According to the LCA terminology this reference unit is called a functional unit.

In this paper the *goal* of the analysis is to quantify and to evaluate the impact of the choice of different N fertilizers on the entire environmental burden associated with a sugar beet production system. The analysis is based on a field trial, which has been conducted in the northeastern part of Germany in 1998. More detailed information on the field trial is given in Table 1.

Table 1: General field trial information

location	Groß Lüsewitz (near Rostock), Germany
year	1998
crop, variety	sugar beet, Penta
rotation	sugar beet - winter wheat - winter barley
soil type	loamy sand
precipitation	561 mm (30-years-average)
evapotranspiration	437 mm (30-years-average)
NO_3 content in spring	35 kg NO_3-N/ha (in 0 - 90 cm depth)
sowing	1998-04-24
harvest	1998-10-12
organic fertilizer	none
sugar beet leaves	incorporated after harvest
crop protection	common practice

The sugar beet production *system* analyzed differed in the form of mineral nitrogen (N) fertilizer applied after sowing. Calcium ammonium nitrate (CAN) and urea were used as solid fertilizers in systems A and B, urea ammonium nitrate solution (UAN) was used as liquid fertilizer in system C (Table 2). All N fertilizers were applied at recommended rates and were applied broadcast on the field. P and K fertilizers have not been applied due to high contents in the soil in spring (48 mg P_2O_5 per 100 g soil and 30 mg K_2O per 100 g soil respectively, measured in 0 - 30 cm depth). Further information on fertilizers, fertilization rates and yields is given in Table 2. All other factors such as plant protection or irrigation were the same for the three N regimes.

Table 2: N fertilization, yields and N contents of the sugar beet production systems

System	A	B	C
N form	calcium ammonium nitrate (CAN)	urea	urea ammonium nitrate (UAN)
N content	27% (50% ammonium-N, 50% nitrate-N)	46%	28% (50% urea-N, 25% ammonium-N, 25% nitrate-N)
applied N rate (kg N/ha)	115	115	115
yield (FM in t/ha)	47,7	44,2	43
GD (P<0.05)	2,57	2,57	2,57
yield (extractable sugar in t/ha)	8,49	7,31	7,82
GD (P<0.05)	0,60	0,60	0,60
N content of leaves (kg N/ha)	133,6	123,8	120,4
N content of beets (kg N/ha)	85,9	79,6	77,4

In addition to the activities on farmer's field, all relevant up-stream activities such as the production, packaging and transport of fertilizers, plant protection substances, seeds and machinery were taken into account in this LCA (Figure 1).

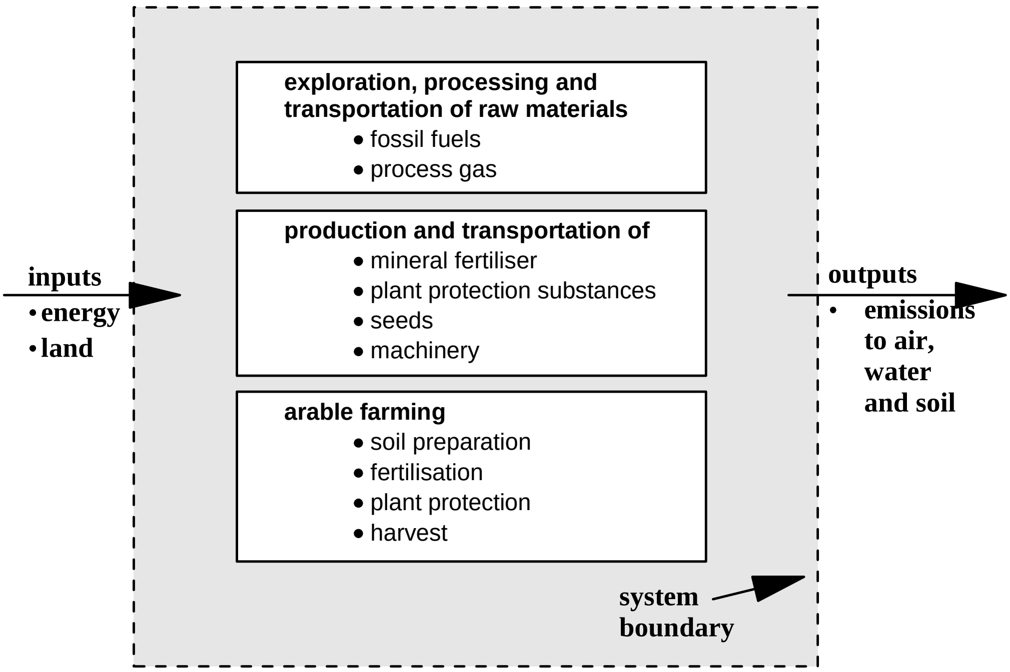

Figure 1: Description of the sugar beet system and its sub-systems

The analysis includes the harvest of the sugar beets, however, the transportation and the processing of the beets in the sugar factory are not included in this study (Figure 1).

All environmental impacts are related to the production of one ton of extractable sugar, which is the *functional unit* for this analysis.

2.2 Life Cycle Inventory

The second major step is to draw up an inventory (LCI = *Life Cycle Inventory*) of all resources used and all emissions released into the environment connected with the production of one ton of extractable sugar. That means to list and quantify all inputs entering and all outputs leaving the analyzed system. All inputs and outputs associated with the production of sugar beet are shown in Table 3.

Table 3: Environmental impacts associated to the production of sugar beet

resources / emissions	source	reference
energy use	fertilizer production (feedstock, fuel)	own figures
	transportation (fuel)	ETH Zürich (1994); Reinhardt (1993); Rhenus (1997)
	farm machinery, production (fuel)	Grosse (1984)
	farm machinery, repair (fuel)	Haas and Köpke (1995)
	farm machinery, use (fuel)	KTBL (1994); Hydro Agri (1993); Reinhardt (1993)
	seeds, production (fuel)	Oheimb et al. (1987)
	plant protection substances, production (fuel)	Oheimb et al. (1987)
land use	arable farming	own figures
CH_4	fertilizer production	own figures
CO_2	fertilizer prod. (urea synthesis, CO_2-sink)	own figures
	arable farming (urea hydrolysis, CO_2-source)	own figures
	all sub-systems (combustion)	own figures; ETH Zürich (1994); Reinhardt (1993)
Ntot	fertilizer production (effluents)	own figures
NH_3	fertilizer production	own figures
	arable farming (volatilization)	ECETOC (1994)
N_2O	fertilizer production (nitric acid production)	own figures
	arable farming (denitrification/nitrification)	Bouwman (1995)
NO_3-N	arable farming (leaching)	adopted from DBG (1992)
NOx	fertilizer production (nitric acid production)	own figures
	all sub-systems (combustion)	own figures; ETH Zürich (1994)
particles / dust	all sub-systems (combustion)	own figures; ETH Zürich (1994)
pesticides (act. ingred.)	arable farming	own figures
SO_2	all sub-systems (combustion)	own figures; ETH Zürich (1994)
VOC	all sub-systems (combustion)	own figures; ETH Zürich (1994)

There are, however, different levels of accuracy for the quantification of the in- and outputs. For instance the emissions originating from the exploration, refining and combustion of fossil fuels can be accurately estimated using established factors from literature (e.g. Reinhardt & Patyk, 1997). In contrast to that, diffuse emissions of ammonia (NH_3), nitrate (NO_3) and nitrous oxide (N_2O) from the field are highly variable and therefore more difficult to estimate. As these nitrogen emissions often contribute considerably to the final results of the LCA studies (Audsley et al., 1997; Cederberg, 1998; Küsters & Jenssen, 1998; Andersson & Ohlsson, 1999), it is of particular importance to obtain good estimates of N released to air and water.

To get reasonable results from actual measurements of ammonia, nitrate and nitrous oxide emissions in the field considerable effort in terms of money and time would be required. This is mostly not possible in LCA studies which have to take account of a lot of single emissions occurring in many sub-systems (Audsley et al., 1997; Cederberg, 1998; Küsters & Jenssen, 1998; Andersson & Ohlsson, 1999). For LCA purposes it is therefore most often inevitable to use average emission rates adjusted according to the specific conditions of the system under investigation. In this study structured methods were used in order to calculate reasonable estimates under consideration of important parameters influencing the emission rates (Brentrup et al., 2000). A simplification of the complex interactions between soil, climate and management factors had to be accepted. To estimate NH_3 volatilization from mineral fertilizers emission factors developed by ECETOC (1994) were chosen (Table 4).

Table 4: Emission factors for ammonia volatilization (% NH3-N loss of total applied mineral N) for different mineral fertilizers in European countries (adopted from ECETOC, 1994)

Fertilizer type	Greece, Spain	Belgium, France, Italy, Ireland, Luxembourg, Netherlands, Portugal, United Kingdom	Austria, Denmark, Finland, Germany, Norway, Sweden, Switzerland
Urea	20	15	15
CAN	3	2	1
UAN	8	8	8

To calculate the N_2O emissions, a function derived by Bouwman (1995) was selected (Eq. (1)):

N_2O emission [kg N_2O-N*ha^{-1}] = 0,0125 * N application[a] [kg N*ha^{-1}] (1)

[a] the applied N rate should be corrected for NH_3 emissions as these predominantly occur earlier than the N_2O emissions (Kroeze, 1994).

To determine the potential nitrate leaching rate associated with the cultivation of sugar beet, a method developed by the German Soil Science Society (DBG, 1992) was adopted. According to this method a nitrogen balance, which reflects the potential N loss via leaching during autumn and winter is calculated as a first step. Nitrogen from fertilizers, seeds and biological fixation are considered as inputs. On the output side the removal of N with the harvested beets is taken into account. In this paper the calculation procedure has been modified further by integrating additional factors, which influence the content of mineral N in the soil susceptible for leaching after harvest (e.g. net N mineralization, gaseous N losses, atmospheric N deposition). All nitrogen inputs and outputs considered in the N balance of the sugar beet production are shown in Figure 2.

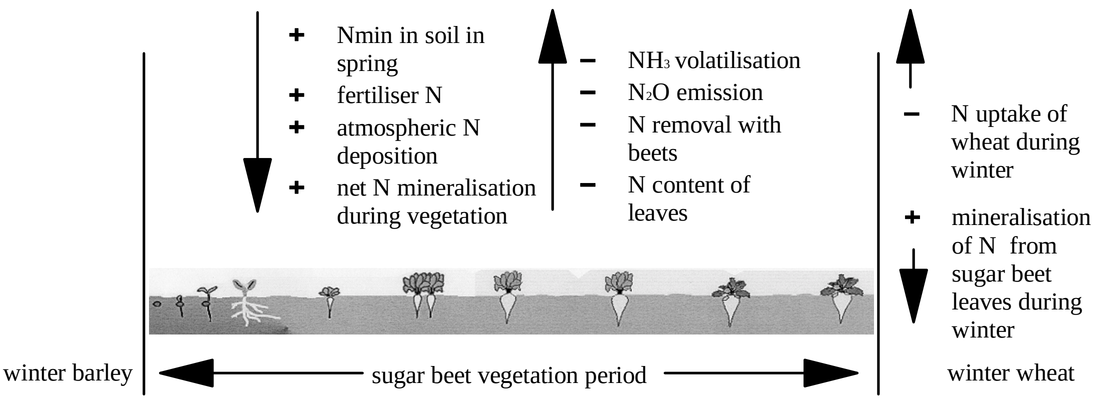

Figure 2: Nitrogen inputs (+) and outputs (-) considered in the N balance for sugar beet production

Table 5 gives background information on the calculation of these N in- and outputs.

Table 5: Background data for the calculation of the nitrogen inputs (+) and outputs (-) in the N balance

Nitrogen inputs / outputs	Background
into the system (+)	
Nmin in soil in spring	considered in the calculation of the N fertilizer rate: N-Sollwert - Nmin in spring = N fertilizer rate
mineral N fertilizer rate	application according to recommendation ("N-Sollwert" method)
atmospheric N deposition	15 kg/ha for Rostock (UBA, 1997)
net N mineralization during vegetation	0,5 kg/ha*day for sugar beet during vegetation period (Engels, 1993)
mineralization of N from sugar beet leaves (easily degradable part)	15% of total leaf-N is mineralized in the following autumn and winter (Engels, 1993)
mineralization of N from sugar beet leaves (slowly degradable part)	50% of the total leaf-N is slowly degradable (Nordmeyer, 1985) of which 30% is supposed to be mineralized in future leaching periods (adopted from Engels, 1993)
out of the system (-)	
NH_3 volatilization	see Table 4 in this paper (ECETOC, 1994)
N_2O emission	see Eq. (1) in this paper (Bouwman, 1995)
N removal with beets	N content = 1,8 kg/t sugar beet (LWK Westfalen-Lippe, 1996)
N content of leaves	N content = 4 kg/t leaves (LWK Westfalen-Lippe, 1996)
N uptake of winter wheat in autumn	5-10% of total N uptake of winter wheat (210 kg/ha) (Reiner et al., 1992; Hydro Agri, 1993)

2.3 Life Cycle Impact Assessment

To further interpret the data of the Life Cycle Inventory it is necessary to evaluate the environmental impact associated with emissions and resource uses. This is done in the third LCA step, the *Life Cycle Impact Assessment* (LCIA).

Several Life Cycle Impact Assessment methods have been developed and published (e.g. Heijungs, 1992a, 1992b; Steen & Ryding, 1993; Goedkoop, 1995; BUWAL, 1998). For this study the Eco-indicator 95 method (Goedkoop, 1995) has been chosen to analyze the environmental impacts of sugar beet production. This method is well documented and regularly used for LCA studies.

The Life Cycle Impact Assessment consists of different sub-steps. First the inventory data are aggregated to effect scores using the equivalence factors shown in Figure 3.

In the LCA terminology this step is called *classification/characterization*. The higher the equivalence factor, the higher is the contribution of an emission to the respective effect. This is a mandatory step for an LCA according to the ISO norm (ISO 14042), whereas the following steps of the Life Cycle Impact Assessment are seen as options to further interpret the LCA results (ISO, 1998c).

During the next step, the *normalization*, the contribution of the analyzed system to the total extent of the environmental effects in Europe is examined. Taking global warming as an example, normalization is done by dividing the global warming potential of the system under investigation by the total global warming potential in Europe. In order to keep the figures manageable the total extent of the different environmental problems in Europe is expressed as environmental effects caused by one person per year (Table 6).

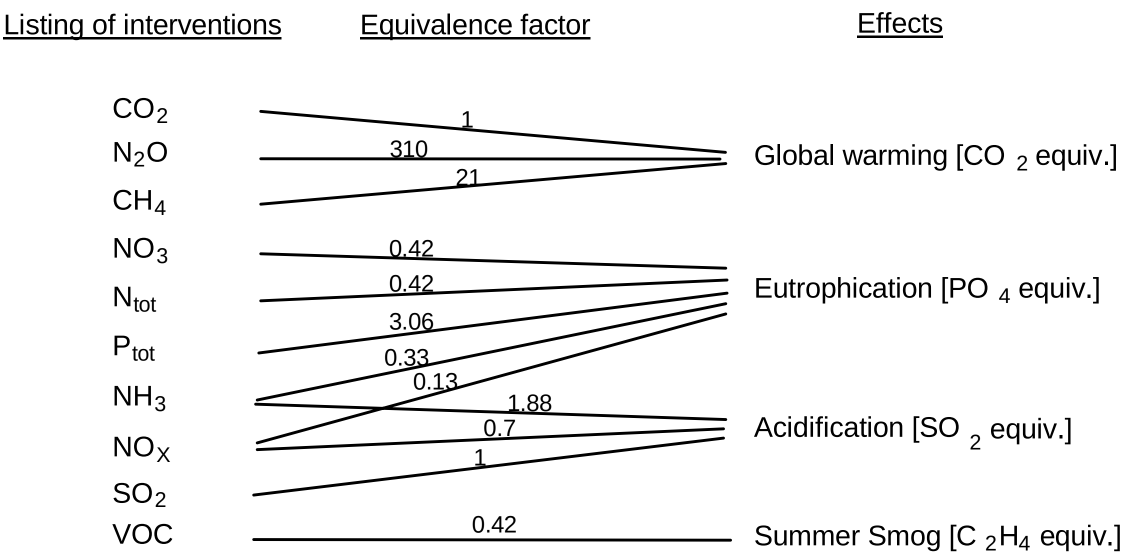

Figure 3: Aggregation (classification and characterization) of emissions in the Eco-indicator 95 method

Table 6: Total emission rates for environmental effects in Europe (without former USSR, per person and year; adopted from Goedkoop, 1995)

Environmental effect	Unit	Normalization value	Uncertainty
Global warming	kg CO_2 equiv.	13100	small
Acidification	kg SO_2 equiv.	113	small
Eutrophication	kg PO_4 equiv.	38,2	moderate
Summer smog	kg C_2H_4 equiv.	17,9	large

However, the normalized and dimensionless data do not allow any conclusion about the potential of the different effects to harm the environment. Therefore, an additional *weighting* step is required to consider the different level of severity of the environmental effects. According to the Eco-indicator 95 procedure this is done by multiplying each normalized effect value by a weighting factor developed for the respective environmental effect. The Eco-indicator 95 method uses the "distance-to-target principle" to establish weighting factors for the different environmental effects. Distance-to-target means the ratio between the current level and a target level of an effect. The target levels of the Eco-indicator 95 method represent a "low damage level". It is assumed that one extra death per million inhabitants per year, no health complaints due to smog periods, or five percent ecosystem impairment are equivalent "low damage levels" (Goedkoop, 1995). These assumptions are the basis for the definition of a target level for each environmental effect. Table 7 gives the resulting weighting factors for the effects covered by the Eco-indicator 95. The result of this weighting procedure is one Eco-indicator value for each effect category. As these scores are dimensionless they can be summed up and then represent the total environmental burden in terms of one Eco-indicator value for each system under investigation.

Table 7: Weighting factors according to the Eco-indicator 95 method (after Goedkoop, 1995)

Environmental effect	Weighting factor	Criterion
Global warming	2,5	0,1° C per decade (5% ecosystem impairment)
Ozone layer depletion	100	Probability of 1 death per year per mio people
Acidification	10	5% ecosystem impairment
Eutrophication	5	Rivers and lakes, impairment of aquatic ecosystems (5% ecosystem impairment)
Heavy metals	5	Cadmium content in rivers (probability of 1 death per year per mio people)
Carcinogens	10	Probability of 1 death per year per mio people
Winter smog	5	Occurrence of smog periods (health complaints)
Summer smog	2,5	Occurrence of smog periods (health complaints) Occurrence of agricultural damage
Pesticides	25	5% ecosystem impairment

2.1.4 Interpretation

In the fourth phase of an LCA the results of the Life Cycle Impact Assessment are used to identify hotspots and possibilities to reduce the negative environmental effects of the systems under analysis.

3. Results and discussion

Important single emissions and fossil fuel consumption for the fertilizing systems related to one ton of extractable sugar are shown in Figure 4. For the UAN as well as for the urea system the CO_2 emissions were higher compared to the CAN system. The differences in the CO_2 emissions are mainly due to differences in the energy consumption during the fertilizer production. The CAN system shows the highest value for N_2O due to higher N_2O emissions during the production of nitric acid which is one step in the CAN fertilizer production. Highest

NH_3 volatilization is observed for systems, in which urea containing fertilizers (UAN and urea) were applied.

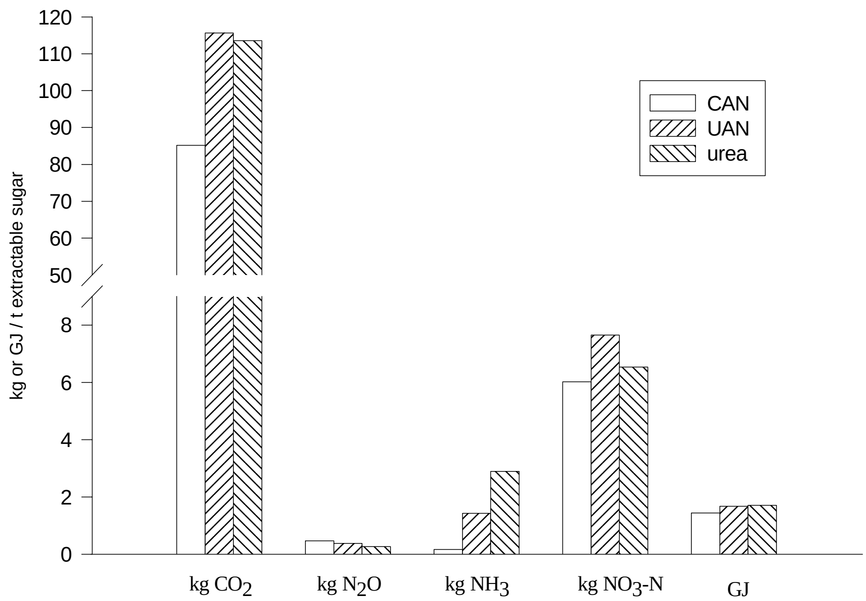

Figure 4: Important emissions [kg] and fossil fuel consumption [GJ] for the fertilizing systems related to one ton of extractable sugar

The systems differ in their NO_3 leaching rate due to differences in the N removal with the beets, N content of the leaves and NH_3 volatilization rates. These factors lead to different amounts of NO_3-N in the soil susceptible for leaching after harvest. Differences in fossil fuel consumption, shown as the related energy consumption in GJ, occur mainly due to differences in energy consumption during N fertilizer production.

To further interpret these Life Cycle Inventory results it is necessary to consider the potential of each emission to contribute to environmental effects. For instance, 1 kg of N_2O has a higher global warming potential than 1 kg of CO_2. Therefore in the Life Cycle Impact Assessment equivalency factors (Figure 3) are used to aggregate the emission scores to scores for effects like global warming or acidification. The Eco-indicator 95 method does not consider the use of energy and land in its impact assessment whereas winter smog is included in the Eco-indicator 95. However, in this study winter smog is excluded from the analysis as the related emissions (SO_2, particles) are predominantly caused by agricultural activities during spring, summer and autumn, and not during winter. The release of pesticides into the environment is excluded from this analysis, although it is also part of the Eco-indicator 95 method. According to

Braunschweig et al. (1996) the assessment of the environmental impacts of pesticide use as proposed in the Eco-indicator 95 appears to be inconsistent and uncertain. The use of the Eco-indicator 95 normalization value and weighting factor for pesticides would result in a contribution of 95% of the pesticides to the total Eco-indicator value. This appears not to be a realistic result. Furthermore the plant protection was equal for the analyzed fertilizing systems and therefore not relevant for this comparison. For these reasons the impact assessment of pesticides has been excluded from this analysis. In this paper the contribution of the sugar beet production systems to the following environmental effects is examined: global warming, acidification, eutrophication and summer smog.

The values per effect of the fertilizing systems are shown in Figure 5. The global warming score is highest for all systems. The values for summer smog are very low compared to the other effects. However, these data do still not provide a clear picture about the total environmental impact of the fertilizing systems. For example, the urea system shows the lowest score for the greenhouse effect, but the highest value for acidification.

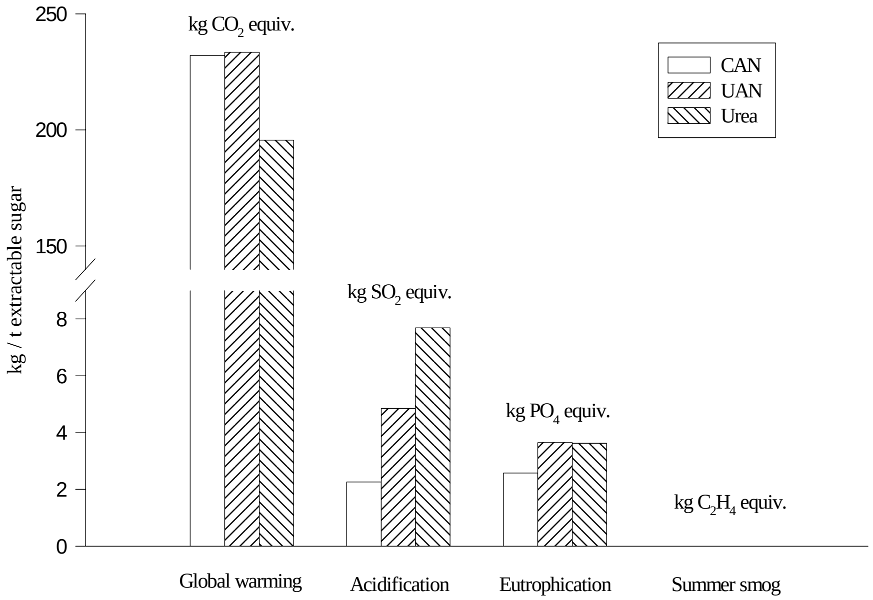

Figure 5: Effect values for the fertilizing systems related to one ton of extractable sugar

Therefore there is a need to further evaluate these results, which is done by relating the share of the analyzed systems to the total extent of the environmental effects in Europe. Therefore each effect value of a system is divided by the respective effect value per person and year in Europe

(Table 6). Figure 6 shows the normalized effect values per ton of extractable sugar for the different fertilizing systems.

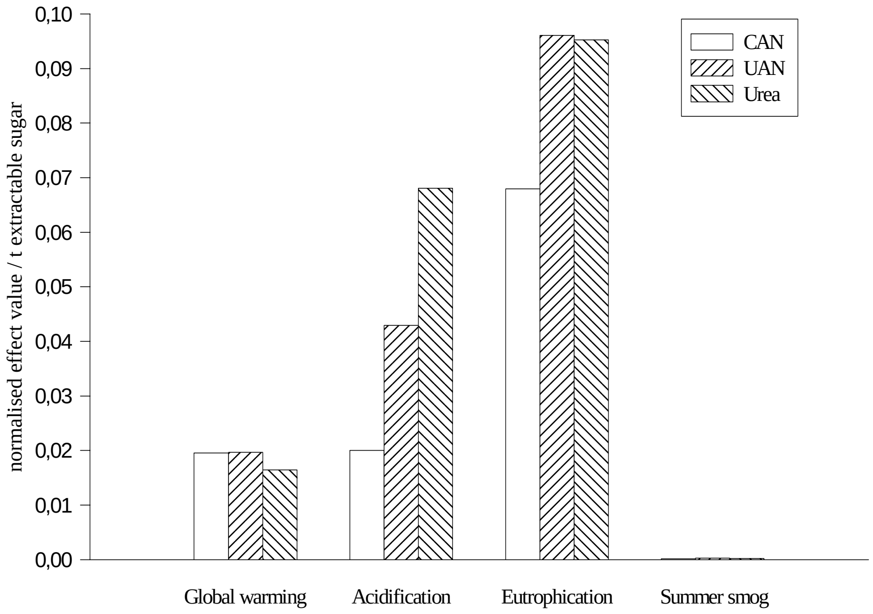

Figure 6: Contribution of the fertilizing systems to environmental effects in Europe

The figure indicates that the contribution of the analyzed systems to acidification and eutrophication is much higher than their contribution to the other effects under investigation. The lower contribution of the fertilizing system with CAN can be explained by the lower NH_3 emission (Figure 5) which result in lower values for acidification and eutrophication. The share of the sugar beet production systems on the formation of tropospheric ozone (summer smog) shows by far the lowest value among all effects. Especially concerning acidification and eutrophication it is important to consider, that the impact assessment according to the Eco-indicator 95 is not site-specific, i.e. it is not considered where for instance the emission and deposition of potentially acidifying substances takes place. To solve this problem, so-called fate models needs to be integrated in the Life Cycle Impact Assessment. For some emissions such fate models are currently under development (Potting et al., 1998).

The normalized effect values still do not allow to conclude on the potential of the different environmental effects to harm the environment. Therefore, in the next step, called weighting, the normalized values are multiplied by weighting factors (Table 7).

The weighting step is still under an intensive scientific debate. According to ISO (1998c) weighting "shall not be used for comparative assertions disclosed to public". Especially this

clause of the ISO normation has been controversially discussed (Hertwich & Pease, 1998; Marsmann et al., 1999). However, in LCA studies comparing alternative products or systems a weighting of the different environmental impacts is indispensable to finally conclude on the environmental preference of one or the other alternative. If the weighting is not performed within the Life Cycle Impact Assessment, the user of the LCA study will weight the system's contribution to different environmental effects on his own. Instead of that a set of generic weighting factors, as they are used in the Eco-indicator 95 method, makes an unbiased aggregation of the different environmental effects possible. Therefore, although "weighting is not allowed under ISO umbrella", the results in this study are weighted "outside this umbrella" (Udo de Haes & Jolliet, 1999a), in order to approach an objective evaluation of the environmental preference of different fertilizers in sugar beet production.
Figure 7 shows the results after weighting. The figure indicates highest Eco-indicator values for acidification and eutrophication for all fertilizing systems. The scores for summer smog are very low in all systems under analysis. The values for acidification and eutrophication are lowest for the CAN system, mainly due to lower NH_3 emission rates. The differences between the systems in the Eco-indicator values for global warming are small.

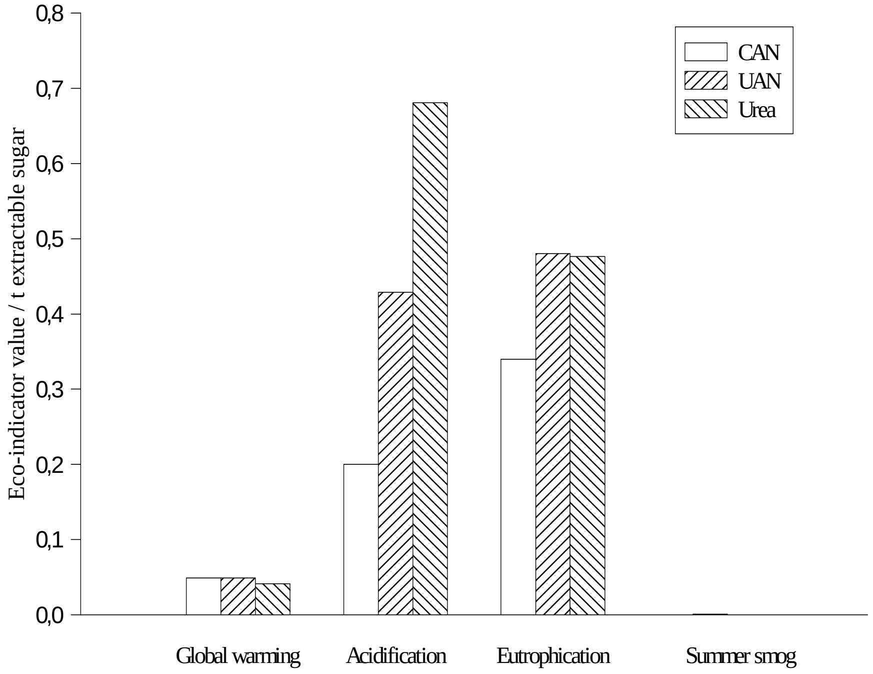

Figure 7: Eco-indicator values per environmental effect for the fertilizing systems related to one ton of extractable sugar

As the resulting Eco-indicator scores for the effect categories are dimensionless, they can be summed up to present the total environmental burden of a system. The higher the Eco-indicator value, the greater the potential to harm the environment. Figure 8 shows the lowest total value for the CAN system, whereas the values for the UAN and urea systems are 63% and 104% higher, respectively.

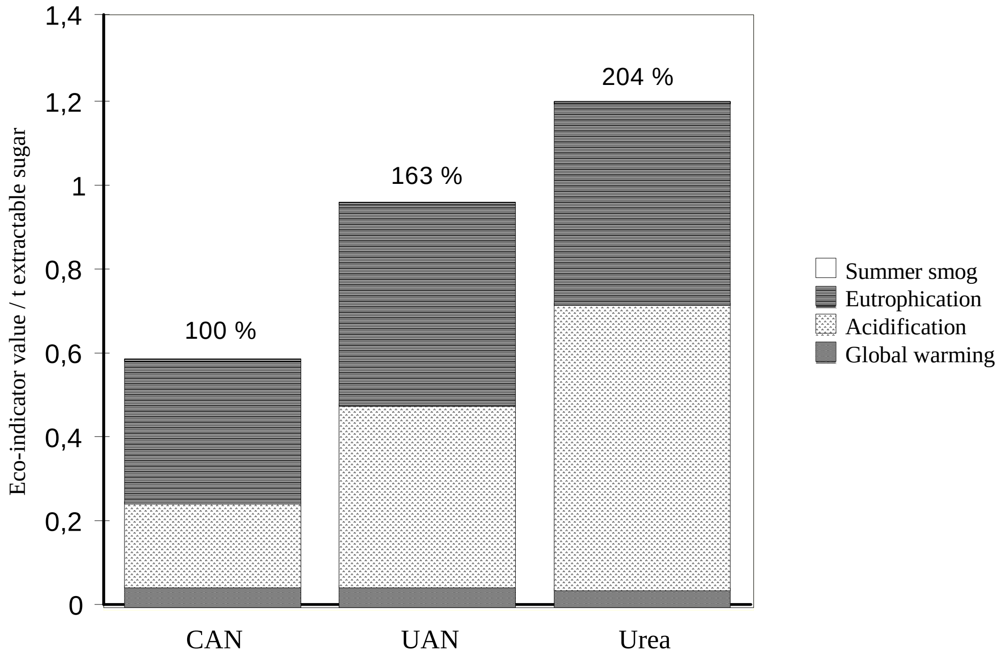

Figure 8: Total Eco-indicator values for the fertilizing systems related to one ton of extractable sugar

This is mainly due to the differences in the acidification and eutrophication potential between the fertilizing systems. The other environmental effects are of minor importance for this ranking. Also the differences in the yield between the fertilizing systems have no influence on this ranking. However, a sensitivity analysis using the same yield for all systems revealed that the score per ton of extractable sugar for the UAN and urea system is still 31% and 61% higher compared to the CAN system.

4. Conclusions

Following the general definition and concept of LCA as described in Consoli et al. (1993), LCA should be a suitable tool to assess the environmental impact associated with agricultural production. The LCA method "Eco-indicator 95" (Goedkoop, 1995) specifically has proven to be applicable to analyze the environmental impact of agricultural systems.

The Eco-indicator 95 method gives a comparative analysis of the systems under investigation related to global warming, acidification, eutrophication and summer smog. However, the investigation shows that the Eco-indicator 95 method has some constraints when applied on an agricultural production system, because not all relevant information listed in the Life Cycle Inventory is considered in the impact assessment. Some important environmental issues are not covered by the Eco-indicator 95 (e.g. use of land and resources) others are included in an inconsistent way (pesticides, winter smog). Another constraint of this method is that the impact assessment cannot be performed site-specific.

However, the obtained Eco-indicator values were sensitive enough to reveal differences between the compared N fertilizing alternatives in sugar beet production. The highest Eco-indicator value, i.e. the strongest environmental impact was observed for the system in which urea was used as N source. The lowest Eco-indicator value has been calculated for the CAN system. The differences are mainly due to different ammonia volatilization after application of the N fertilizers. The results show that all analyzed fertilizing systems particularly contribute to the environmental problems of acidification and eutrophication. This is mainly due to emissions of ammonia and nitrate on the field. Therefore, besides the applied N fertilizer rate and application technique, the choice of a mineral N fertilizer can clearly influence the environmental impact associated with sugar beet production.

In the previous chapter, the LCA methodology was applied in a case study on arable crop production. The case study revealed that the LCA methodology is principally suitable to investigate and evaluate the environmental impacts associated to the production of arable crops. However, the application also revealed that currently available LCA tools like the Eco-indicator'95 method (Goedkoop, 1995) need some adjustments to the specifics of arable farming.

Furthermore, the case study indicated that it is particularly important for LCA studies on arable production to derive reliable inventory data on diffuse, on-field emissions of ammonia (NH_3), nitrous oxide (N_2O) and nitrate (NO_3). Therefore, the following chapter provides LCA practitioners with an appropriate methodology to calculate realistic, study-specific estimates of NH_3, N_2O and NO_3 emissions under consideration of important soil, climate and management parameters.

III. Methods to estimate on-field nitrogen emissions from crop production as input to LCA studies in the agricultural sector

(International Journal of Life Cycle Assessment, 5 (2000) 349-357)

Abstract

Nitrogen compounds emitted from the field are usually considered in Life Cycle Assessments (LCA) of agricultural products or processes. The environmentally most important of these N emissions are ammonia (NH_3), nitrous oxide (N_2O) and nitrate (NO_3). The emission rates are variable due to the influence of soil type, climatic conditions and agricultural management practice. Due to considerable money and time efforts and great variations in the results, actual measurements of emissions are neither practical nor appropriate for LCA purposes. Instead of measurements structured methods can be used to estimate average emission rates. Another possibility is the use of values derived from the literature, which would, however, require considerable effort compared to estimation methods, especially because the values might only be valid for the particular system under investigation.

In this paper, methods to determine estimates for NH_3, N_2O and NO_3 emissions were selected from a literature review. Different procedures were chosen to estimate NH_3 emissions from organic (Horlacher & Marschner, 1990) and mineral fertilizers (ECETOC, 1994). To calculate the N_2O emissions, a function derived by Bouwman (1995) was selected. A method developed by the German Soil Science Association (DBG, 1992) was adopted to determine potential NO_3 emissions. All methods are not computer-based and require only a minimum set of input data. This makes them on the one hand transparent and easy to perform, on the other hand they certainly simplify the complex processes.

1. Introduction

On-field nitrogen (N) emissions are usually considered in LCA studies where agricultural production is part of the investigated system (e.g. production of food). Nitrogen emissions often contribute considerably to the final results of the LCA studies (Audsley et al., 1997; Küsters & Jenssen, 1998; Cederberg, 1998; Andersson & Ohlsson, 1999). However, it is often difficult to derive exact rates of N released to air and water, because emission rates can greatly vary depending on soil type, climatic conditions and agricultural management practices. Measurements of these emissions require considerable investment in terms of money and time and in any case they show great variations (e.g. Isermann, 1990, for NH_3) because they can only reflect a snapshot of the specific conditions at time of measurement. For LCA purposes average potential emission rates adjusted to the conditions typical for the system under investigation would be more appropriate. Methods are, however, required to enable the LCA practitioner to easily calculate potential nitrogen emission rates taking into account important site-specific parameters. In this study, easy to perform methods or factors are proposed to estimate the most important nitrogen emissions (NH_3, N_2O, NO_3) related to crop production.

Figure 1 shows a simplified nitrogen cycle focusing on the most important nitrogen in- and outputs.

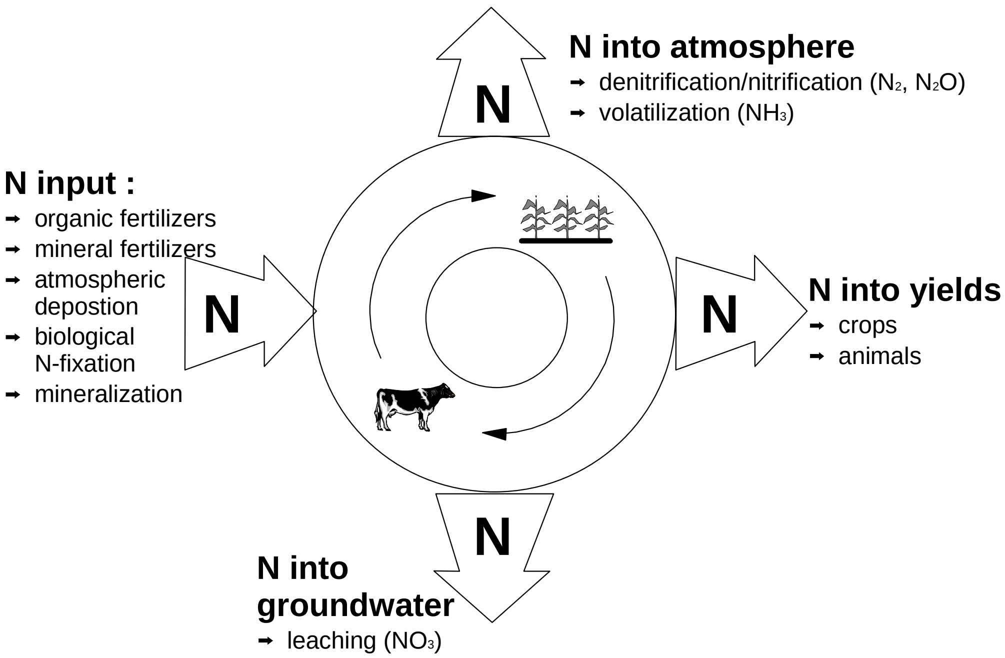

Figure 1: The nitrogen cycle on a farm (adopted from ECETOC, 1988)

Agriculture, including both crop and animal production contributes considerably to total NH_3, NO_3 and N_2O emissions. Especially for ammonia, agriculture is by far the main source of emissions. Table 1 gives information about the contribution of agricultural production on the different total nitrogen emissions at different spatial scales.

Table 1: The share of agriculture on total global, European and German N emissions

	Globe	Europe	Germany
NO_3	[a]	[a]	50 % [b]
NH_3	87 % [c]	97 % [d]	96 % [e]
N_2O	47 % [f]	48 % [d]	33 % [e]

[a] no information

[b] Stanners, 1995

[c] Isermann, 1990

[d] Jol & Kielland, 1997

[e] Enquete-Kommission "Schutz der Erdatmosphäre", 1994

[f] Kroeze, 1994

2. Ammonia volatilization

Nearly 90 % of the global emissions of the volatile gas ammonia (NH_3) are related to agriculture (see Table 1). Within agriculture animal husbandry has by far the greatest share on the ammonia released to the environment (Isermann, 1990, ECETOC, 1994). Ammonia volatilization occurs during and after production, storage and application of organic fertilizers (see chapter 2.1). Mineral fertilizers contribute to a lower extent to the total NH_3 emissions, but show differences e.g. dependent on the N-form used (see chapter 2.2). Especially the use of NH_4 and urea containing mineral fertilizers can result in high NH_3 emissions. Unfortunately no estimation method is available that covers both, the NH_3 losses due to organic and to mineral fertilization. Therefore two different estimation methods were selected to assess the ammonia emissions caused by fertilizer use.

Ammonia losses due to production and storage of organic fertilizers, such as manure and slurry are not in scope of this article as the focus is only on crop production.

2.1 Ammonia volatilization due to organic fertilizer application

According to Isermann (1990) the ammonia losses during and after application of organic fertilizers ranges from 1 to 100 % of the applied NH_4-N. This clearly indicates the need to estimate the NH_3 emissions site specific and dependent on agricultural practices.

In the following an easy to perform procedure proposed by Horlacher & Marschner (1990) to assess the ammonia emissions due to organic fertilizer application is presented. In this method four important and easy to get parameters are chosen to assess the ammonia losses:

- average air temperature
- infiltration rate

- time between application and incorporation or rainfall
- precipitation or incorporation after application

Increasing **air temperature** results in increasing ammonia volatilization rates (ECETOC, 1994; Horlacher & Marschner, 1990). The **infiltration rate** describes the capability of the soil to take up the NH_3/NH_4^+. The infiltration of NH_3/NH_4^+ into the soil reduces the volatilization rate. The amount of volatilized ammonia depends of course on the **time** the NH_3 is present at soil surface. Thus the time between the application and the disappearance of the NH_3/NH_4^+ deeper into the soil profile has to be considered in the estimation (Horlacher & Marschner, 1990). **Rainfall** reduces the volatilization of NH_3 considerably due to increased solution of NH_3/NH_4^+ and increased infiltration into the soil. The extent of this reduction depends on the amount of rainfall (Horlacher & Marschner, 1990). **Incorporation** of the organic fertilizers also reduces the NH_3 losses, as the NH_3/NH_4^+ gets deeper into the soil (Sommer, 1992).

In the estimation method the NH_3 losses are calculated in percentage of the total NH_4-N applied in form of organic fertilizers. Thus the NH_4-N content of the applied organic fertilizer should be known. Some average figures are given in Table 2. The original method of Horlacher & Marschner (1990) is calibrated only for the application of cattle slurry and was transferred to other forms and origins of organic fertilizers (see Table 2). However, it should be noted here that this extension of the original method has not been tested or validated.

Table 2: Dry matter, N and NH4-N content of different organic fertilizers

Fertilizer typ	Dry matter (%)	N (kg/t)	NH_4-N (kg/t)	NH_4-N (% of N, rounded)
Cattle manure [a]	25	5.0	0.5	10
Cattle slurry [b]	8	4.0	2.2	55
Cattle liquid manure [b]	2	4.0	3.5	85
Calf slurry [b]	3	3.6	2.0	55
Pig manure [a]	23	6.0	0.6	10
Pig slurry [b]	6	5.1	3.6	70
Pig liquid manure [b]	2	5.0	4.5	90
Sow slurry [b]	5	4.1	2.9	70
Chicken slurry [b]	14	8.7	6.0	70

[a] Enquete-Kommission "Schutz der Erdatmosphäre", 1994

[b] Hydro Agri, 1993

2.1.1 Temperature

The air temperature is a key parameter for the NH_3 volatilization rate. Therefore the influence of infiltration rate, time period and rainfall on the NH_3 volatilization rate is assessed at different temperature levels. In the following four classes of temperature are distinguished: 0-5, 5-10, 10-15 and 15-20 °C.

2.1.2 Infiltration rate

The infiltration rate can be evaluated according to Table 3. If two evaluation criteria were met, which lead to different infiltration rates, the lower infiltration rate should be chosen, i.e. if for instance liquid manure was applied on a heavily compacted soil, the infiltration rate should be regarded as low.

Table 3: Evaluation of the infiltration rate (Horlacher & Marschner, 1990, modified)

Infiltration rate	Application circumstances
low	• on cereal or corn stubble • on heavily compacted, water saturated soil • slurry with high dry matter content • solid manure
medium	• on non compacted soil • slurry with medium dry matter content
high	• on prepared soil with a lot of macropores (e.g. ploughed soil) • on loose soil • slurry with low dry matter content • liquid manure

The maximum potential ammonia loss in percentage of the applied NH_4-N is shown for different infiltration rates and temperatures in Table 4. This maximum potential ammonia loss has to be taken as an input parameter for an LCA, if no incorporation or rainfall after application took place.

Table 4: Maximum potential ammonia loss in % of the applied NH_4-N dependent on temperature and infiltration rate into the soil (Horlacher & Marschner, 1990, modified)

Temperature (°C)	NH_3 losses (%)		
	low infiltration	medium infiltration	high infiltration
0 - 5	30	22	15
5 - 10	45	35	25
10 - 15	70	55	40
15 - 20	90	75	55

2.1.3 Time

Incorporation of the organic fertilizer into the soil or rainfall after application lead to a reduction of the maximum potential ammonia loss from Table 4. The longer the time period between the application of an organic fertilizer and its incorporation or rainfall the higher is the ammonia loss. This is considered by multiplying the maximum potential NH_3 loss (see Table 4) by a time factor (Table 5), derived from field experiments (Horlacher & Marschner, 1990). The resulting score represents the actual NH_3 loss between application of the organic fertilizer and its incorporation or rainfall.

Table 5: Time factors for different temperature classes (Horlacher & Marschner 1990, modified)

Temperature	Time between application and precipitation / incorporation											
(°C)	1h	2h	4h	8h	12h	1d	2d	3d	4d	6d	8d	12d
0 - 5	0.04	0.07	0.10	0.15	0.19	0.25	0.35	0.45	0.54	0.60	0.80	1.00
5 - 10	0.06	0.10	0.14	0.20	0.25	0.35	0.50	0.65	0.73	0.85	1.00	
10 - 15	0.15	0.25	0.35	0.50	0.60	0.73	0.83	0.92	1.00			
15 – 20	0.20	0.30	0.45	0.65	0.75	0.85	0.95	1.00				

2.1.4 Precipitation

Further NH_3 loss depends on the amount of rainfall. This is taken into account by introducing a rain factor (Table 6), which is again based on field experiments (Horlacher & Marschner, 1990). The remaining potential ammonia loss, i.e. the maximum potential loss minus the loss between application of organic fertilizers and rainfall (see chapter 2.1.3), has to be multiplied by this rain factor. The resulting figure gives the NH_3 loss after rainfall.

Table 6: Rain factors for different temperature classes (precipitation after application and before total potential volatilization, Horlacher & Marschner, 1990, modified)

Temperature	Precipitation			
(°C)	0 - 2mm	2 - 5mm	5 - 10mm	> 10mm
0 - 5	0.30	0.15	0.05	0
5 - 10	0.40	0.20	0.10	0
10 - 15	0.60	0.40	0.20	0
15 - 20	0.80	0.50	0.30	0

2.1.5 Incorporation

Incorporation of slurry or manure into the soil reduces the ammonia losses to very low rates dependent on the depth of incorporation (Sommer, 1992, Horlacher & Marschner, 1990). Therefore, if the organic fertilizer was incorporated, 2% of the remaining potential NH_3 loss at the time of incorporation should be considered as ammonia volatilization (Sommer, 1992). The calculation is similar to the calculation for precipitation.

2.1.6 Other factors

Other climatic factors influencing the NH_3 volatilization rate are **radiation** and **wind speed**. High radiation as well as high wind speed lead to increased ammonia losses. These factors are either well enough reflected by already integrated parameters (radiation by temperature) or very difficult to derive (wind speed) (Horlacher & Marschner, 1990). Nevertheless, especially wind speed may have a great influence on the volatilization rate and therefore it would be desirable to take account of this factor (Erisman, 1999).

Soil related parameters such as **buffer capacity**, **pH** and **cation exchange capacity** have an effect on ammonia volatilization (ECETOC, 1994):

- high pH (>8) -> high NH_3 volatilization rate
- high buffer capacity -> high NH_3 volatilization rate
- low cation exchange capacity -> high NH_3 volatilization rate

However, as there is no estimation framework available considering these factors, they are not integrated. This is supported by Horlacher & Marschner (1990). According to their findings infiltration is the main soil related factor.

2.2 Ammonia volatilization due to mineral fertilizer application

The ammonia emissions due to the application of mineral fertilizers are usually lower compared to slurry and manure (Isermann, 1990). However, dependent on the ammonium and urea content of a mineral fertilizer, the climatic conditions and soil properties, considerable ammonia volatilization can also take place when applying mineral fertilizers. The ECETOC (1994) proposed a method to estimate these emissions taking into account the different soil properties throughout Europe and the different NH_3 volatilization risk dependent on the fertilizer type.

They defined three classes of countries with different regional sensitivity to NH_3 volatilization (Table 7).

Table 7: European countries grouped according to their NH_3 volatilization sensitivity

Group	Countries	Calcareous soil	pH (usually)	Sensitivity
I	GR, E	common	> 7	high
II	I, F, UK, IRL, P, B, NL, L	partly existent	7	medium
III	N, S, FIN, DK, D, CH, A	rare	< 7	low

Based on a literature review ECETOC (1994) developed NH_3 emission factors for six groups of mineral fertilizers taking into account the regional differences in NH_3 volatilization sensitivity. The resulting emission factors are shown in Table 8. These emission factors are also supported by many field trials, as for instance reviewed by Wiesler (1999).

Table 8: Emission factors (% NH_3-N loss of total applied mineral N) for different mineral fertilizers in Europe (ECETOC 1994, modified)

	Groups of European countries (according to Table 10)		
Fertilizer type	Group I	Group II	Group III
Urea	20	15	15
Ammonium Nitrate, Calcium Ammonium Nitrate, NP, NK, NPK	3	2	1
Ammonium Phosphate	5	5	5
Ammonium Sulphate	15	10	5
Anhydrous Ammonia	[a]	[a]	4
Nitrogen solution	8	8	8

[a] fertilizer not common in this group of countries

An incorporation of mineral fertilizer into the soil should be considered. In this case it is proposed to take the ammonia loss related to the application of ammonium nitrate, i.e. 1 - 3 % of the total amount of nitrogen applied.

3. Nitrous oxide emissions

Agriculture has a considerable share on the anthropogenic N_2O emissions (33 - 48%, see Table 1), whereas N_2O itself contributes only to 5% to the total global warming potential.

Nearly 80% of the N_2O emissions due to agriculture are related to the use of mineral and organic fertilizers. Biomass burning (e.g. shifting cultivation, deforestation) is responsible for about 20% (Kroeze, 1994). Two microbial processes in soil are responsible for most of the N_2O emissions in agriculture: denitrification (NO_3 -> NO_2 -> NO -> $N_2O\uparrow$ -> $N_2\uparrow$) and nitrification (NH_4 -> [$N_2O\uparrow$]-> NO_2 -> NO_3).

Anaerobic conditions are a prerequisite for N_2O emissions due to denitrification. Furthermore the available **amount of nitrogen** in the soil is a decisive factor for the rate of N_2O released. As denitrifying microorganisms need organic carbon as an energy source the availability of degradable **organic matter** is a further limiting factor for N_2O formation.

A lot of complex interactions between soil and climate related factors on the one hand and parameters determined by agricultural management on the other hand influence the N_2O

emissions. Table 9 summarizes the findings of Granli & Bøckman (1994) concerning these factors.

Table 9: Key parameters influencing N_2O emissions from agricultural soils

Parameter	Effect on N_2O emissions
Soil aeration	• intermediate aeration -> highest N_2O production • low aeration -> high denitrification rate, but mainly N_2 production
Soil water content	• increasing soil water content -> increasing N_2O emissions, but • under very wet conditions -> decline • changing conditions (dry/wet) -> highest N_2O production
Nitrogen availability	• increasing NO_3/NH_4 concentrations -> increasing N_2O emissions
Soil texture	• from sand to clay -> increasing N_2O emissions
Tillage practice	• ploughing -> lower N_2O emissions • no/low-tillage -> higher N_2O emissions
Compaction	• increasing compaction -> increasing N_2O emissions
Soil pH	• where denitrification is main source of N_2O emission: increasing pH results in decreasing N_2O emissions • where nitrification is main source of N_2O emission: increasing pH results in increasing N_2O emissions
Organic material	• increasing organic carbon content -> increasing N_2O emission
Crops and vegetation	• plants, but especially their residues and remaining roots after harvest increase N_2O emission
Temperature	• increasing temperature -> increasing N_2O emission
Season	• wet summer -> highest N_2O production • spring thaw -> high N_2O production • winter -> lowest N_2O emission

Dependent on these parameters and their interactions, measurements of N_2O emission from different types of agricultural land show great variations (Granli & Bøckman, 1994).

This clearly indicates a need for taking this variability of N_2O fluxes into account, when estimating N_2O emissions in agricultural Life Cycle Assessment. Unfortunately the complexity of the interactions between the various parameters is up to now not well enough understood to propose an estimation or even calculation method for N_2O emissions (Enquete-Kommission "Schutz der Erdatmosphäre", 1994). Despite this, Bouwman (1995) proposed an emission factor for N_2O emissions from mineral and organic fertilizers. From field experiments he derived the following formula:

(1) N_2O emission [kg N_2O-N*ha^{-1}] = 0.0125 * N application[a] [kg N*ha^{-1}]

[a] the applied N rate should be corrected for NH_3 emissions, as these predominantly occur earlier than the N_2O emissions (Kroeze, 1994).

This emission factor of 0.0125 kg N_2O-N*ha^{-1} per kg N input is also taken as default value for estimating direct nitrous oxide emissions from arable land by the IPCC (Houghton et al., 1997). The Bouwman formula is commonly used, because it is not yet possible to consider the other key parameters (see Table 9) appropriately. It is therefore suggested to take this approach for estimating the nitrous oxide emissions caused by agricultural practice.

Although the N_2O emissions are in focus of most research activities dealing with denitrification, N_2 is the main product of denitrification and usually released in much higher rates (Wiesler, 1999). N_2 is not of environmental relevance, but, however, N_2 rates emitted to air should be included in the nitrogen balance, which is a prerequisite for the calculation of the nitrate leaching rate (see chapter 4, Table 10). Von Rheinbaben (1990) reviewed and evaluated 38 field experiments and concluded that on average up to 10% of the fertilizer input is lost as N_2O and N_2 on arable and grassland. On the other hand, agricultural practices, soil and climate parameters may greatly influence the N_2 emissions as well as that of N_2O. For practical reasons the N_2-N emissions related to fertilizer application (corrected for NH_3-N volatilization) may be regarded as 9%, taking into account the IPCC emission factor of 1.25% for N_2O-N.

4. Nitrate leaching

The mineral nitrogen in the soil is mainly nitrate (NO_3^-) and to a lower extent ammonium (NH_4^+). As nitrate is hardly adsorbed by soil particles, it can be easily leached into the groundwater. During the vegetation period the risk of NO_3 leaching is low because large amounts of nitrate are taken up by the plants. Furthermore almost no downward water movement occurs during the vegetation period mainly due to high evapotranspiration rates. During the vegetation-free period from late autumn to early spring precipitation often exceeds evapotranspiration so that the mobile NO_3 anion can be leached downwards in the soil.

For LCA purposes it is important to be able to predict the potential NO_3 leaching rate related to an agricultural product or production process. The level of nitrate leaching depends strongly on different parameters. The most important parameters determining the nitrate leaching rate are:

- agriculture related: nitrogen balance [kg N*ha^{-1}*a^{-1}]
- soil related: field capacity in the effective rooting zone (FC_{RZe}) [mm]
- climate related: drainage water rate (W_{drain}) [mm*a^{-1}]

4.1 Agriculture related parameters

The **nitrogen balance** can be used as a measure for the amount of nitrate-N in the soil susceptible for leaching after the vegetation period in autumn,. The nitrogen balance can be calculated as described in Table 10.

Table 10: Calculation of the nitrogen balance in autumn

N input [kg N*ha^{-1}]	N output [kg N*ha^{-1}]
+ Mineral N fertilizer	- N removal with harvested crops
+ Organic N fertilizer	- NH_3-N emissions
+ Biological N fixation	- N_2O-N / N_2 emissions
+ Atmospheric N deposition	- N immobilization
+ N mineralization	
$\sum$ input	$\sum$ output
N balance = $\sum$ input - $\sum$ output	

Some of the nitrogen inputs and outputs are already known, as they are either part of the system under investigation (e.g. fertilizer rate, crop removal) or have been already estimated (e.g. NH_3-N, N_2/N_2O-N). If fertilizer rates or crop removals are unknown, typical figures for the different crops and agricultural production systems should be available at least for European countries (for Germany: e.g. Hydro Agri, 1993). Regarding the biological N fixation for instance Loges et al. (2000) presented a model for the quantification of N_2 fixation of legumes.

The deposition of nitrogen from the atmosphere should also be accounted for when estimating the amount of nitrate in the soil susceptible to leaching during autumn and winter. The N input due to wet and dry deposition of oxidized and reduced nitrogen compounds is not directly influenced by the product or process under investigation. However, the deposited nitrogen may enter the system, as it can be taken up by the plants, similar to mineral fertilizers. Figure 2 gives information about the N deposition rate in Europe.

Based on the assumption that an agricultural production system is relatively constant on a long term, i.e. for more than one crop rotation, and the N fertilizer input is adjusted to the requirements of the plants, it can be assumed that the nitrogen mineralization and immobilization rates more or less equal each other (Engels, 1993). Some other agricultural aspects can influence the nitrogen balance considerably. For example **intercropping** as well as **underseeding** may reduce the nitrogen amount in autumn by more than 40% (Scheffer & Ortseifen, 1996).

That part of the nitrate-N present in the soil in autumn that is actually lost via leaching depends on soil and climate parameters. The influence of these parameters is described and quantified in the following sections.

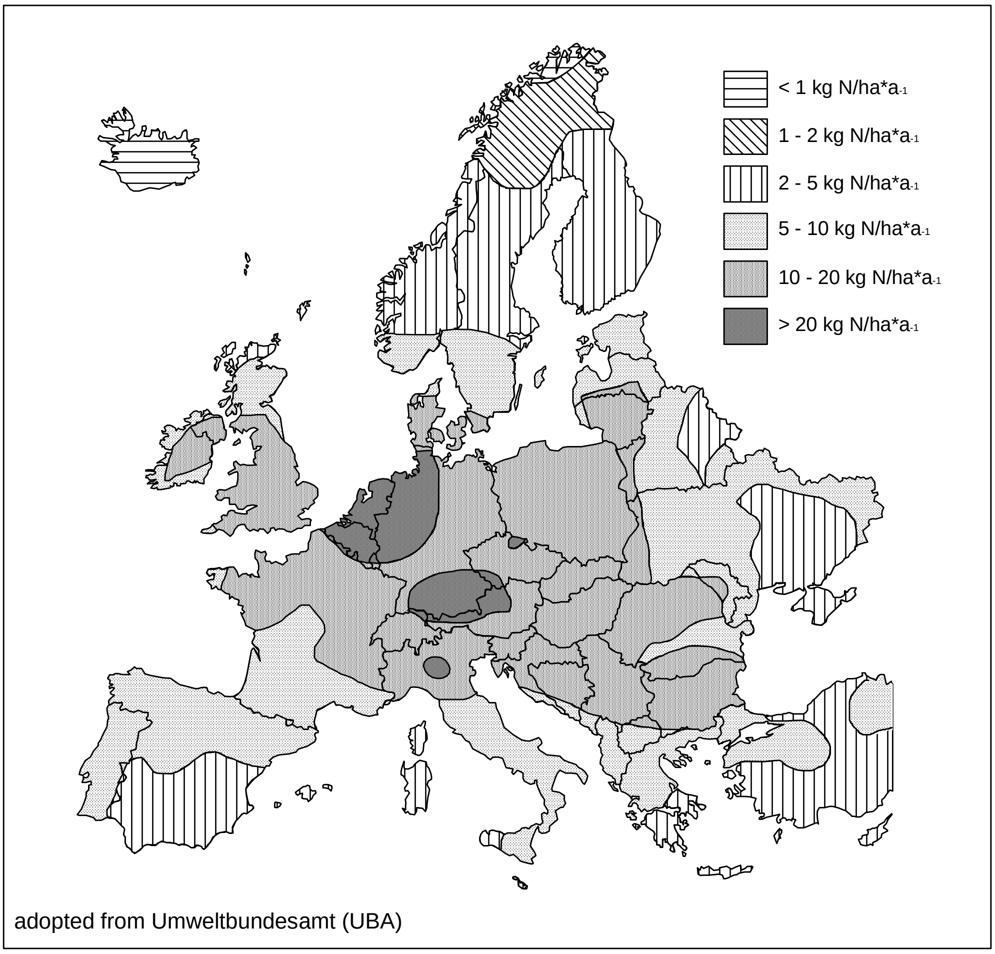

Figure 2: Total nitrogen deposition in Europe in 1993 (adopted from Umweltbundesamt, 1997)

4.2 Soil related parameters

The **field capacity in the effective rooting zone** (FC_{RZe}) describes the capacity of the soil to adsorb water within that part of the soil in which the roots are able to take up water. The FC_{RZe} can be calculated by multiplying the available field capacity (FCa) by the effective rooting zone (RZe).

(2) FC_{RZe} [mm] = FCa [mm*dm^{-1}] * RZe [dm]

The available field capacity as well as the effective rooting zone strongly depend on the soil texture. The German Soil Science Association (1992) proposed six classes of available field capacity (Table 11) and five classes of effective rooting zone (Table 12).

Table 11: Assignment of soil textures to 6 classes of available field capacity (FCa), medium soil density (DBG 1992)

Class (evaluation)	Soil texture [a]	FCa ($mm*dm^{-1}$) range	average
1 (very low)	S	< 10	8
2 (low)	lT	10 – 14	12
3 (medium)	lS, tS, sL, tL, uT, T	14 – 18	16
4 (high)	uS, sU, uL	18 – 22	20
5 (very high)	lU, tU, U	> 22	24
6 (swamp)	Hh, Hn		60

[a] S = sand, s = sandy, U = silt, u = silty, T = clay, t = clayey, L = loam, l = loamy, H = swamp, h = swampy, n = half-swampy

Table 12: Assignment of soil textures to 5 classes of effective rooting zone (RZe), medium soil density (DBG 1992)

Class (evaluation)	Soil texture [a]	RZe (dm) range	average
1 (very low)	Hn	< 3	2
2 (low)	S, Hn	3 – 5	4
3 (medium)	lS, uS	5 – 7	6
4 (high)	tS, lS	7 – 9	8
5 (very high)	U, sU, lU, tU, sL, uL, tL, lT, T	> 9	10

[a] S = sand, s = sandy, U = silt, u = silty, T = clay, t = clayey, L = loam, l = loamy, H = swamp, h = swampy, n = half-swampy

Hence, to calculate the field capacity in the effective rooting zone (FC_{RZe}) only information about the soil texture is needed.

4.3 Climate related parameters

The **rate of drainage water** (W_{drain}) is mainly determined by the precipitation rate (W_{precip}), its distribution through the year and the evapotranspiration rate. The drainage water rate can be either measured or estimated according to formula (3), developed by Liebscher & Keller (1979, in DBG, 1992). This approach is based on regression analysis and is suitable for flat lands. Bach (1987) found a good correlation between values calculated according to formula (3) and own measurements.

$$W_{drain}\ [mm] = 0.86*W_{precip\_year}\ [mm] - 11.6*(W_{precip\_summer} / W_{precip\_winter})\ [mm] - 241.4 \qquad (3)$$

The precipitation rate for the hydrologic summer (04-01 to 09-30) and the hydrologic winter (10-01 to 03-31) should be easily available (e.g. for Germany: Deutscher Wetterdienst).

The nitrate leaching rate is mainly dependent on the quantity of water that percolates through the soil profile into the groundwater. A measure for this quantity is the **exchange frequency of the drainage water per year**. This can be calculated using FC_{eRZ} (2) and W_{drain} (3) as input parameters.

(4) exchange frequency * a^{-1} = W_{drain} [mm*a^{-1}] * $FC_{RZe}{}^{-1}$ [mm]

Due to the fact that almost all NO_3 in the soil is dissolved in water, the whole amount of NO_3-N present in the soil at the beginning of the leaching period in autumn is supposed to be available for leaching. The exchange frequency of the drainage water directly reflects the share of nitrate lost via leaching. If the exchange frequency per year is equal or higher than 1, the whole amount of nitrate is supposed to be leached. Therefore the maximum value for the exchange frequency per year used in (5) is 1.

(5) leached NO_3-N [kg N*ha^{-1}*a^{-1}] = NO_3-$N_{in\ soil\ in\ autumn}$ [kg N*ha^{-1}]*exchange frequency*a^{-1}

5. Example

In the following, an LCA case study on winter wheat production (Küsters & Jenssen, 1998) is chosen to illustrate the calculation procedures given in the previous chapters. The winter wheat system is located on a farm in northern Germany and the yield is 8.5 tons of grain per ha. The straw (8 tons/ha) is baled and removed from the field. The N fertilization was 80 kg N/ha as cattle slurry for the first dressing (containing 44 kg NH_4-N/ha) and 130 kg N/ha as ammonium nitrate (AN) for topdressing. The field has been fertilized with slurry over long-term.

5.1 Ammonia volatilization

Parameters to calculate the NH_3–N volatilization from cattle slurry:

- Temperature during and after application: 10-15°C
- Infiltration rate: medium (medium dry matter content of the slurry, non compacted soil, see Table 3)
- Precipitation after application: no
- Incorporation of the slurry: yes

- Time between application and incorporation: 4 h

Calculation:

- Maximum potential ammonia loss [% of applied NH_4-N] (see Table 4): 55%
- Multiplication with time factor (see Table 5): 55% * 0.35 = 19.25%, i.e. 19.25% of the applied NH_4-N is lost between application and incorporation (= 8.5 kg NH_3-N/ha)
- 44 – 8.5 = 35.5 kg NH_4-N/ha remains on the field after incorporation
- 2% of this 35.5 kg NH_4-N/ha were lost after incorporation (= 0.7 kg NH_3-N/ha)
- total NH_3-N volatilization due to application of cattle slurry: 8.5 + 0.7 = **9.2 kg NH_3-N/ha**

Parameters to calculate the NH_3–N volatilization from mineral fertilizer:

- type of mineral fertilizer: ammonium nitrate (AN)
- location of the crop production: Germany

Calculation:

- 130 kg AN-N * 1% = **1.3 kg NH_3-N/ha** (see Tables 7 and 8)

5.2 Nitrous oxide emissions

Parameters to calculate the N_2O-N emissions from fertilizer use:

- total N rate applied per ha: 130 kg N/ha (AN), 80 kg N/ha (slurry)
- NH_3-N losses per ha via volatilization: 10.5 kg NH_4-N/ha

Calculation:

- 130 kg AN-N/ha + 80 kg slurry-N/ha – 10.5 kg NH_4-N/ha = 199.5 kg N/ha
- 199.5 kg N/ha * 0.0125 = 2.5 kg N_2O-N/ha
- total N_2O-N emission due to fertilizer application: **2.5 kg N_2O-N/ha**
- (N_2-N emission: 199.5 kg N/ha * 0.09 = 18 kg N_2-N/ha)

5.3 Nitrate leaching

Parameters to calculate the NO_3 leaching due to fertilizer use:

- Nitrogen inputs [kg N/ha]: mineral and organic fertilizers: 210, biological N fixation: none, atmospheric N deposition: 25, N net-mineralization: 0
- Nitrogen outputs [kg N/ha]: N removal with harvested crops: 153 (grain) + 40 (straw), NH_3-N, N_2O-N and N_2-N emissions: 10.5 + 2.5 + 18
- Soil texture: loamy silt (lU)
- Average precipitation per year: 738 mm, summer: 387 mm, winter: 351 mm

Calculation:

- Nitrogen balance [kg N/ha]: 210 + 25 – 153 – 40 – 10.5 – 2.5 – 18 = 11
- Field capacity in effective rooting zone: 240 mm (see Tables 11 and 12)
- Rate of drainage water [mm per year]: 0.86 * 738 – 11.6 * (387 / 351) – 241.4 = 380

- Exchange frequency of drainage water per year: 380 mm/year * 240 mm = 1.58/year
- NO_3 leaching rate: 11 kg NO_3-N/ha * 1/year = **11 kg NO_3-N/ha/year**

6. Comparison of the methods to others used in recent LCA studies

In order to compare these models to those used in other LCA studies, the proposed methods have been applied to a wheat production system described by Audsley et al. (1997). In this study, each of four teams of LCA experts from different countries (Denmark, the Netherlands, United Kingdom, Switzerland) examined the environmental impacts of an intensive winter wheat production system located in the UK. In this system three different mineral fertilizers at a total rate of 240 kg N/ha and no manure were applied. For further details see Audsley et al. (1997). The research teams used different methods to estimate the on-field emissions of ammonia, nitrous oxide and nitrate. Table 13 gives the estimated emission rates of the different teams in comparison to own estimates.

Table 13: Estimates of on-field emissions of NH_3, N_2O and NO_3 due to an intensive wheat production system (Audsley et al., 1997) calculated with different models

	NH_3 [kg N/ha]	N_2O [kg N/ha]	NO_3 [kg N/ha]
DK models	- [a]	- [a]	44
NL models	4.8	4.0	21
UK models	- [a]	3.1	31
CH models	12.4	7.2	108
Own models	13.2	2.8	32

[a] no estimation carried out

In the following the main differences between the models described in Audsley et al. (1997) and those proposed in this paper are discussed.

6.1 Ammonia volatilization

- DK and UK team

NH_3 losses were not estimated.

- NL team

One uniform emission factor for all types of mineral N fertilizers was used independent of the site of application (2% of the fertilizer-N as NH_3-N).

- CH team

Emission factors developed by Asman (1992, in Audsley et al., 1997) were used. These factors differ between fertilizer types, but not between sites of application.

- Own model

Emission factors developed by ECETOC (1994) were applied, which are different dependent on the fertilizer type and the site of application. The factors are based on a broad literature review.

6.2 Nitrous oxide emissions

- DK team

N_2O emissions were not estimated.

- NL team

N_2O emissions were estimated according to Bouwman (1995). The background emission of 1 kg N_2O-N/ha is included, although this is not due to the fertilizer application in the analyzed wheat production system. Furthermore, the NH_3-N losses were not substracted from the N fertilizer rate, which is the basis for the calculation of the N_2O emission rate according to Bouwman (1995) (see formula (1)).

- UK team

Emission factors from Amstrong-Brown (in Audsley et al., 1997) were used, which are different dependent on the N form and time of application. The reference given is unpublished and therefore the basis for the emission factors is unknown (field experiments, pot trials, literature study?).

- CH team

A uniform emission factor of 3% of total applied fertilizer N (BUWAL, 1994, in Audsley et al., 1997) was used, which appears to be relatively high compared to values recommended in literature (e.g. Kaiser et al., 1996; Bouwman, 1995).

- Own model

N_2O emissions have been estimated according to Bouwman (1995). The background emission (1 kg N_2O-N/ha) is excluded and the NH_3 losses were substracted from the rate of N fertilizer applied, as these predominantly occur before the N_2O emissions (Kroeze, 1994).

6.2 Nitrate leaching

- DK team

The basis for the estimation is an average NO_3 leaching rate on sandy and loamy soils determined for fertilizer rates according to official recommendation in Denmark (Simmelsgaard, 1991, in Audsley et al., 1997). This relationship was used to calculate the NO_3

leaching at any given fertilizer rate. As the reference leaching has been determined under Danish conditions, this may not be representative for conditions in other European regions. Furthermore, the yield level and specific soil and climatic conditions of the analyzed system were not accounted for.

- NL team

Leaching factors for sandy soils (40.5% of the mineral N remaining in the soil after harvest; Goossensen & Meeuwissen, 1990, in Audsley et al., 1997) and clay soils (20% of the mineral N remaining in the soil after harvest; Breeuwsma et al., 1987, in Audsley et al., 1997) were used. No further soil and climatic parameters were considered. In the calculation of the mineral N remaining in the soil after harvest the atmospheric N deposition for NL was used, which may be different for the UK.

- UK team

A so-called „crop/soil/fungicide simulation model“, which uses e.g. daily weather records, inputs of mineral and organic N, and soil parameter as input data, was used to determine the leaching rate. As no reference is given, it is not known which other input parameter are necessary to run this model, but computer based simulation models most often need a lot of very specific input data, that are not always readily available (Engel et al., 1993).

- CH team

According to a method developed by Walther (1995, in Audsley et al., 1997) the NO_3 leaching rate is supposed to be the sum of (a) the difference between N mineralization and N uptake by the crop and (b) the N rate applied multiplied by crop specific leaching factors. Both figures are calculated on a monthly basis. As N immobilization processes are not considered the nitrate content in the soil may be overestimated. Furthermore, the crop specific leaching factors were estimated for fertilizer rates recommended in Switzerland and, therefore, may be not valid for other fertilizer application rates.

- Own model

The NO_3 leaching rate is calculated from the NO_3-N remaining in the soil after harvest taking into account specific soil and climate parameters.

7. Conclusions

The first step in a Life Cycle Assessment is to make an inventory of all relevant environmental interventions caused by the system under investigation. For agricultural LCA studies usually the emissions of ammonia, nitrous oxide and nitrate are important and need to be considered. Three ways to take these nitrogen emissions into account are possible:

- to measure actual emission rates caused by the system under consideration
- to use values derived from literature in a case-by-case procedure
- to estimate potential emission rates using structured estimation methods

To **measure actual N emission rates** is money and time consuming and therefore often not operational in Life Cycle Assessments. Furthermore, actual measurements of N emissions often show great variations (e.g. Isermann, 1990, for NH_3) and may reflect only a snapshot of the specific conditions at the time of measurement. For LCA purposes average emissions adjusted to the conditions typical for the system under examination are therefore more appropriate than actual emission rates.

Values derived from the literature often reflect an average emission, which is assumed to be representative for the system examined in the LCA. A disadvantage of this procedure is that for each new study a new literature review might be necessary to obtain new appropriate values. Furthermore it is difficult to evaluate the quality of the derived figures as they strongly depend on the quality of the literature source.

An alternative procedure is to use **structured methods** for the estimation of average emission rates. Conditions, which influence the nitrogen emissions, are reflected by certain parameters (soil, climate, and agricultural practice). These parameters should be available and used as input for the estimation methods. Advantages of such procedures are their easy performance, less effort compared to measurements or values derived from the literature and the comparability of the results. The quality of the estimated emission rates might be improvable, because estimation methods simplify the complex conditions leading to the release of emissions into the environment. Only a limited number of well know factors are taken into account, assuming that these are the most important ones. However, the presented estimation methods could provide useful tools to obtain reasonable nitrogen emission data for a Life Cycle Inventory in the agricultural sector.

In the previous chapter, methods were suggested to estimate diffuse, on-field emissions of ammonia (NH_3), nitrous oxide (N_2O) and nitrate (NO_3) as an input to LCA studies on arable crop production.

However, there are further specific environmental impacts associated to arable farming. Since investigations have shown that there is a need to improve the existing LCA methodology for the impact assessment of abiotic resource consumption, in the following chapter a new impact assessment method for this impact category is proposed.

IV. Impact Assessment of Abiotic Resource Consumption - Conceptual Considerations -

(International Journal of Life Cycle Assessment, 7 (2002) 301-307)

Abstract

The impact assessment of the consumption of abiotic resources, such as fossil fuels or minerals, is usually part of the Life Cycle Impact Assessment (LCIA) in LCA studies. The problem with the consumption of such resources is their decreasing availability for future generations. In currently available LCA methods (e.g. Eco-indicator'99/Goedkoop & Spriensma, 1999; CML/Guinée, 2001) the consumption of various abiotic resources is aggregated into one summarizing indicator within the characterization phase of the LCIA. This neglects that many resources are used for different purposes and are not equivalent to each other. Therefore, the depletion of reserves of functionally non-equivalent resources should be treated as separate environmental problems, i.e. as separate impact sub-categories. Consequently, this study proposes assigning the consumption of abiotic resources to separate impact sub-categories and, if possible, integrating them into indicators only according to their primary function (e.g. coal, natural gas, oil -> consumption of fossil fuels; phosphate rock -> consumption of phosphate). Since this approach has been developed in the context of LCA studies on agricultural production systems, the impact assessment of the consumption of fossil fuels, phosphate rock, potash salt and lime is of particular interest and serves as an example. Following the general LCA framework (Consoli et al., 1993; ISO, 1998), a normalization step is proposed separately for each of the sub-categories. Finally, specific weighting factors have been calculated for the sub-categories based on the 'distance-to-target' principle. The weighting step allows for further interpretation and enables the aggregation of the consumption of different abiotic resources to one summarizing indicator, called the Resource Depletion Index (RDI). The proposed method has been applied to a wheat production system in order to illustrate the conceptual considerations and to compare the approach to an established impact assessment method for abiotic resources (CML method, Guinée, 2001).

1. Introduction

The present study deals with the impact assessment of abiotic resource consumption. The main objective of this paper is to critically scrutinize the current impact assessment approaches and to contribute with some new conceptual ideas and a modified impact assessment procedure to the discussion.

According to the U.S. Geological Survey (USGS, 2001) an abiotic resource is defined as "a concentration of naturally occurring ... material ... in such form and amount that economic extraction ... is currently or potentially feasible". This definition implies that there is a need to use these resources as raw materials. The National Environmental Policy Plan 3 (NEPP 3) of the Netherlands, for example, states that, "all societal activities make demands on these resources" and "we all have an interest in their availability, quality and accessibility" (VROM, 1998). Assuming that the principles of sustainability (WCED, 1987) are internationally accepted, this means that future generations will have the same interest in extracting and consuming resources as today's generation for the use of abiotic resources. Therefore, the reserves of abiotic resources are worth being protected and are regarded as one of the safeguard subjects dealt with in LCA (Consoli et al., 1993; ISO, 1998).

2. Impact assessment of the consumption of abiotic resources

2.1 Characterization

Different methods have been proposed to aggregate the consumption of various resources into one indicator, which describes the total resource consumption associated with a product or production system (for reviews see Finnveden, 1996; Heijungs et al., 1997; Müller-Wenk, 1998). Common to all these methods is that the different resources (e.g. gold, phosphate rock, and crude oil) are aggregated into one resource depletion indicator within the first step of the impact assessment (classification/characterization).

According to ISO 14042 (2000), the inventory results are assigned to defined impact categories during classification and within these, as far as possible, aggregated into impact category indicators (= characterization). An impact category is defined as a "class representing environmental issues of concern" (ISO, 2000). The calculation of impact category indicators should be based on distinct environmental processes or mechanisms (ISO, 2000; SETAC, 1993). Subjective evaluations and assumptions should be avoided as far as possible (ISO, 2000).

Transferred to the impact assessment of abiotic resource consumption, this means that an aggregation of resources, which are used for totally different purposes and therefore have different functions (e.g. phosphate rock and fossil fuels), should not be carried out within the characterization phase mainly because of two aspects:

1. The consumption and thus the depletion of functionally different resources contributes to different problems, i.e. different impact categories. For example the depletion of fossil fuel reserves leads to totally different consequences (problems with fuel supply,

electricity production etc.) compared to the depletion of phosphate rock reserves (plant nutrition problems in crop production). Thus, the protection of one resource does not necessarily compensate for the depletion of another resource. A simple example: It does not help to have plentiful coal reserves if no raw phosphate is left in order to maintain sustainable arable farming.

2. Characterization factors for the aggregation of functionally different resources cannot be based on distinct environmental processes only, because there is also a dependency on subjective evaluation and assumptions. If the production-to-reserve ratio is used to derive characterization factors (e.g. Guinée, 2001), for example, various resources are weighted only according to the scarcity of their reserves. Although this is certainly an important weighting criterion, it could also be argued that specific resources should be weighted higher than other resources independently from their scarcity. For instance, raw materials for the production of plant nutrients (e.g. phosphate rock) could be of higher value compared to gold reserves because of their essential role in producing food for humans. This would certainly be a clear subjective evaluation, however, exactly like assessing all resources as being equally valuable.

The function concept

Since the 'function of resources' is of special importance in this study, the background of this idea shall be explained in a bit more detail. In the present study, the function of a resource always means the main use of the resource like the use of phosphate rock as a phosphorus source in the production of mineral fertilizers, or the use of oil, natural gas and coal as energy sources. Principally, the substitutability of resources is the basis to assign different resources into a group of functionally equivalent resources. Taking the example of coal, oil and natural gas, the aggregation of these resources into the group of fossil fuels implies that these abiotic resources are basically substitutable by each other. Of course, in practice this may not always be realized. For example, the production of liquid fuel from coal is technically possible, but as long as enough oil is available or better alternatives are being explored, this possibility is not put into practice. Principally, coal, oil and natural gas fulfill the same function, which is to supply energy, and thus, they have a common denominator (energy content in MJ), which can be used for their aggregation into the same impact category of 'fossil fuels' within this characterization. However, since phosphate rock is not substitutable by any other abiotic resource as a raw material for fertilizer production, for example, the consumption of the resource 'phosphate rock' makes up its own impact sub-category. In theory it would be possible to assign resources with similar functions like the different raw materials used as mineral plant nutrients to one group, but an aggregation into a summarizing indicator for this group (e.g. as "nutrient-equivalents") would make no sense, since phosphorus and potassium, for example, are both essential plant nutrients and can therefore not replace each other. As a consequence of these considerations, many single resources will make up their own impact sub-category. Perhaps the term "impact sub-category" is therefore a bit misleading, because it may suggest an aggregation of resources into groups, which in practice will often not be possible. Therefore, in the following "impact sub-category" always means the consumption of a group of functionally equivalent resources (in this study only fossil fuels) or of a single, unique resource (e.g. phosphate rock, potash salt).

2.2 Normalization

During normalization the indicator values calculated per impact sub-category (e.g. consumption of fossil fuels in MJ) are related to a reference value for the respective impact sub-category (e.g. total annual consumption of fossil fuels in Europe in MJ). The main purposes of normalization are: a) to provide information about the significance of the calculated indicator values and b) to prepare the characterization results for the weighting step by eliminating the units.

In contrast to other impact assessment methods for resource consumption (e.g. CML; Guinée, 2001), the normalization in this paper is performed at the level of separate groups of functionally equivalent resources or unique, single resources and not at the level of an already fully aggregated resource depletion indicator. This procedure has the advantage that the contribution of any product or process to each separate resource-related problem is clearly visible in the normalization result.

European[1] normalization values (NV) for some resource-related impact sub-categories are given in the Annex.

2.3 Weighting

Weighting generally means to evaluate different environmental effects according to their severity and to aggregate the weighted impact indicator values across all impact categories to one overall environmental indicator. In this study, the weighting of the normalized impact

[1] Europe (n=37) = EU15 + Albania, Belarus, Bosnia, Bulgaria, Croatia, Czech Rep., Estonia, FYROM, Hungary, Iceland, Latvia, Lithuania, Moldova, Norway, Poland, Romania, Russia, Slovak Rep., Slovenia, Switzerland, Ukraine, Yugoslavia

indicator values has been performed according to the “distance-to-target” principle, which is also used for instance in the Eco-indicator ’95 (Goedkoop, 1995) and Eco-scarcity method (BUWAL, 1998). The ratio of the actual level of an environmental impact to a target level for the same impact gives the weighting factor.

A crucial point of this procedure is the definition of appropriate target values for the environmental effects. This study suggests defining targets for resource consumption based on the idea of sustainability. In order to give future generations sufficient time to develop alternative sources, materials or recycling techniques for currently used resources; these resources should be available for an appropriate period of time. The present study deliberately suggests different *target time periods* for the availability of resources, because, for the time being, no agreed national or international targets for the protection of reserves of specific resources exist.

Based on data on the estimated global recoverable reserves of a resource (Annex 1), it has been calculated, which theoretical annual extraction would be tolerable in order to ensure an availability of the respective resources for 100, 300, or 1000 years (Equation 1).

$$tolerable\,annual\;production_{res\,i,\,time\;period\;i} = \frac{global\;recoverable\,reserve_{res\,i}}{target\,time\;period_i} \qquad (1)$$

where:

tolerable annual production $_{\text{res i, time period i}}$ = annual production rate that ensures availability of resource i for time period i [in kg or MJ * year^{-1}]

global recoverable reserve $_{\text{res i}}$ = proved recoverable reserve of resource i [in kg or MJ]

target time period $_{\text{i}}$ = time period for which resource i should be available [in years]

The quotient of the current annual production and the tolerable annual production rate for the defined target time periods gives the weighting factors for the depletion of fossil fuels, phosphate rock, potash, and lime (Equation 2). The weighting factors are given in Table 1.

$$WF_{res\,i} = \frac{current\;annual\;production_{res\,i}}{tolerable\,annual\;production_{res\,i,\,time\;period\;i}} \qquad (2)$$

where:

WF $_{\text{res i}}$ = weighting factor for resource i

current annual production $_{\text{res i}}$ = current annual production of resource i [in kg or MJ * year^{-1}]

Table 1: Weighting factors for resource-related impact sub-categories

Impact sub-category	Weighting factor_100 target: reserves should last for 100 years	Weighting factor_300 target: reserves should last for 300 years	Weighting factor_1000 target: reserves should last for 1000 years
Depletion of fossil fuels	1.05	3.16	10.54
Depletion of phosphate rock	1.20	3.60	12.00
Depletion of potash	0.30	0.91	3.05
Depletion of lime	[A]	[A]	[A]

[A] no data on recoverable reserves available, but according to USGS (2001) lime reserves are very large and thus, the weighting factors will presumably be for any target time period near 0

When calculating the weighting factors based on the "target-time-period" concept and the "distance-to-target" principle, the question comes up concerning how to deal with weighting factors between 0 and 1. The "distance-to-target" principle implies that a target value exists, which describes the tolerable extent of the environmental effect to be evaluated. A weighting factor of 1 means that the current situation meets the target value exactly; weighting factors below 1 imply that the defined environmental target is more than met. Thus, the target value could be interpreted as that point, which is equivalent to the solution of the respective environmental problem and consequently, weighting factors below 1 would lead to an exclusion of the respective environmental effect from the weighting. However, on the other hand, it should be considered that weighting factors of 1 or slightly below 1 describe a situation in which the reserve of a resource is not completely, but almost completely depleted within the target time period. This situation should be evaluated more severely than a reserve, which is far from depletion within the target time period, i.e. which has a weighting factor near 0. To take into account these considerations, weighting factors between 0 and 1 are included in the suggested weighting approach.

Another point that should be considered when applying the target time period idea is that if the same target time period is assumed for all resources, this does not influence the relative differences between the weighting factors anymore. In that case, solely the scarcity of the resources, i.e. the production-to-reserve ratio, determines the differences between the resources, whereas the chosen target time period only influences the absolute size of the values. This implies that, except from the scarcity aspect, all resources are valued equally, which itself is of course a kind of weighting too. However, for the moment, it does not seem to be justified to define different target time periods for different resources, but the suggested weighting procedure could be easily adjusted in order to consider differentiated, resource-specific target time periods, if these would be available.

After multiplying the normalized impact sub-category indicator values of a system by the respective weighting factors, the resulting values are equivalent and can be summed up. The sum of the weighted indicator values gives the total resource depletion indicator for the system under analysis. The application of the characterization, normalization and weighting factors for any system under investigation is given in Equation (3).

$$RDI\ sys = \sum_i \left(\frac{\sum (res_{i,cat\,i} \times CF_{res\,i})}{NF_{cat\,i}} \right) \times WF_{cat\,i} \qquad (3)$$

where:

RDI $_{sys}$	= Resource Depletion Index for the system under investigation
res $_{i,\ cat\ i}$	= consumption of resource i belonging to impact sub-category i in the analyzed system [e.g. coal in kg]
CF $_{res\ i}$	= characterization factor for resource i [e.g. for coal in MJ]
NV $_{cat\ i}$	= normalization value for impact sub-category i [e.g. for fossil fuels in MJ]
WF $_{cat\ i}$	= weighting factor for impact sub-category i [e.g. for fossil fuels depletion]

Figure 1 gives an overview of the impact assessment procedure suggested in the present paper.

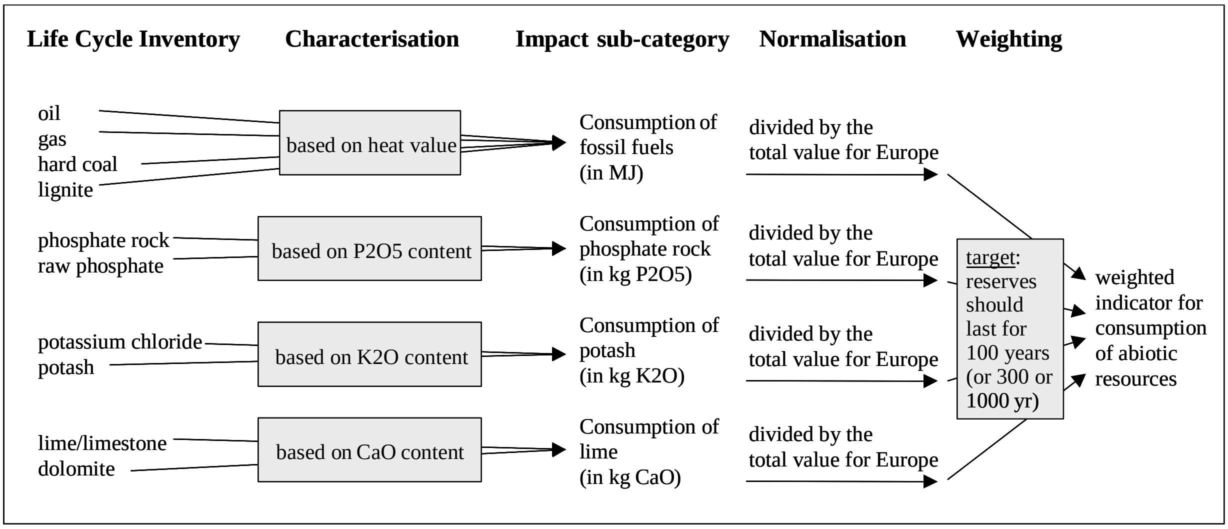

Figure 1: Impact assessment of abiotic resources consumption; Characterization according to the primary function, separate normalization for functionally different resources, and final weighting according to the 'target time period' idea

3. Example of use: Application of the proposed approach to a wheat production system and comparison to the CML method

In order to demonstrate the consequences of applying the suggested modified impact assessment approach, instead of using the common 'traditional' procedure, the same example (the production of 1 ton of wheat grain) is analyzed with both approaches. As a representative for the traditional procedure, the recently updated CML method (Guinée, 2001) is used. In the CML method, the consumption of different resources is aggregated into one indicator (ADP, Abiotic Resource Depletion Potential) within the characterization step based on the ratio between annual extraction rates and ultimate reserves. Since the semi-metal antimony is used as a reference substance, the ADP is expressed in kg antimony-equivalents. The characterization result is subsequently normalized by dividing it by the aggregated annual extraction rate of all resources in Europe, which is also expressed in antimony-equivalents (Van Oers, 2001).

Table 2 gives the consumption of abiotic resources, which were necessary to produce one ton of winter wheat grain at nitrogen (N) fertilizer rates of 100 and 200 kg N/ha in a long-term field experiment (Broadbalk Experiment, Rothamsted, UK). The resource consumption data refer to all agricultural on-field activities (e.g. ploughing, harvest), the production, packaging and transport of farming inputs (e.g. fertilizers, plant protection agents), as well as the exploration and processing of necessary raw materials (e.g. fossil fuels) (Brentrup et al., 2002).

Table 2: Resource consumption associated with the production of 1 ton of winter wheat grain at different N fertiliser rates in a field experiment (Broadbalk, Rothamsted, UK)

Resource	N fertilizer rate (kg N/ha)	
(per t wheat grain)	100	200
Phosphate rock (kg)	36.33	27.92
Potash (kg)	159.49	122.59
Limestone (kg)	77.64	59.68
∑ Fossil fuels (MJ)	1664.5	1677.4
Coal (kg)	8.49	6.85
Lignite (kg)	9.71	7.46
Oil (kg OE) [A]	15.65	12.78
Natural gas (m^3)	20.76	27.17

[A] OE = crude oil equivalents

With regard to the impact assessment procedure proposed in the present paper, Table 2 already shows the characterization results for the example. In comparison to the "N200-system", the "N100-system" contributes less to the impact sub-category 'depletion of fossil fuels', but more to the problems of 'depletion of phosphate rock', 'depletion of potash salt' and 'depletion of lime'. However, from these results, it is not possible to conclude on the relevance of the system's contributions to the different resource-related problems of consumption compared to the total consumption rates of the respective resources in Europe.

Therefore, the values derived after characterization are normalized by dividing them by the respective total resource consumption figures for Europe, which are given in Annex 1. The normalization result for both fertilizer rates is given in Figure 2. As no data for lime (CaO) consumption in Europe were available, the normalization has only been performed for the impact sub-categories 'depletion of phosphate rock', 'depletion of potash salt' and 'depletion of fossil fuels'.

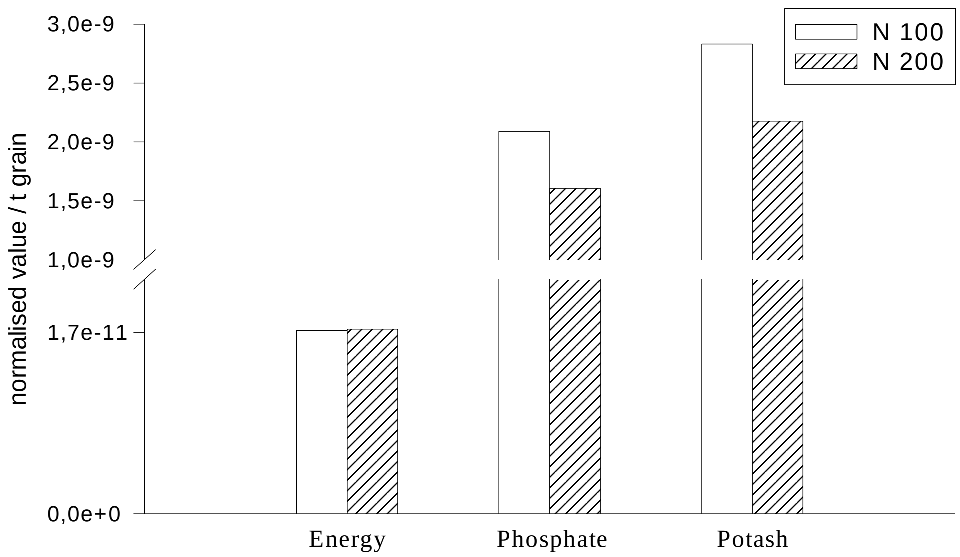

Figure 2: Share of the wheat production systems in the total consumption of fossil fuels, phosphate and potash in Europe (per ton of wheat grain)

Figure 2 shows that the share in the total European consumption of fossil fuels is nearly equal for both systems and generally far lower compared to the normalized indicator values for phosphate and potash consumption. The share of the "N100-system" in the consumption of phosphate and potash is higher compared to the "N200-system".

In the new method suggested in this study, the normalized indicator values for each group of functionally equivalent resources or unique, single resources are multiplied by weighting factors. The background of the calculation of the weighting factors is described in section 2.3. For this example, the "100-year target time period" has been chosen for all resources. The weighting results have been calculated according to Equation (3). The results after weighting can be aggregated to a summarizing Resource Depletion Index (RDI). The higher the RDI for a system, the higher is its damage to the availability of abiotic resources.

The aggregated results for the two fertilizer regimes are given in Figure 3, which also shows the aggregated result calculated with the CML method (characterization + normalization).

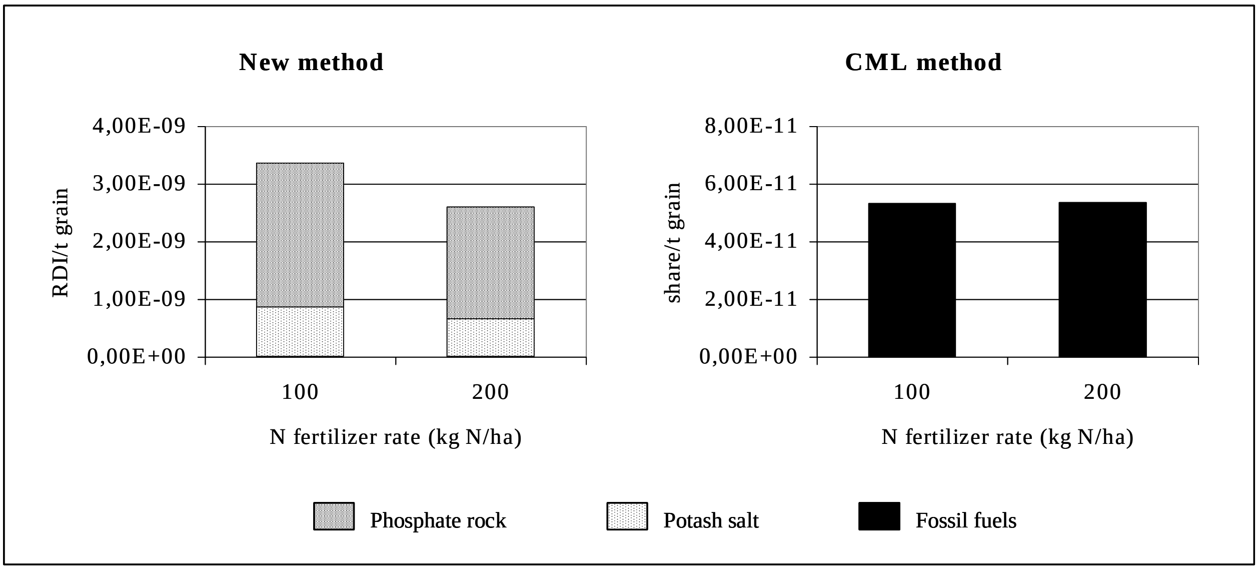

Figure 3: Impact assessment result for the example according to the 'new' method in comparison to the result after characterization and normalization according to the CML method (Guinée, 2001; Van Oers, 2001)

Figure 3 reveals that, in the 'new' method, the RDI value per ton of grain is dominated by the consumption of phosphate rock followed by potash. The contribution of fossil fuel consumption to the normalized and weighted impact assessment result is negligible. The RDI value for the "N200-system" is lowest because of the lower phosphate and potash consumption per ton of grain in this system.

Using the CML method, the consumption of fossil fuels clearly dominates the impact assessment result (Fig. 3). This, on the one hand, is due to the higher characterization factor for the consumption of fossil fuels (4.81E-04 kg antimony-equiv./MJ) compared to those for phosphorus (8.44E-05 kg antimony-equiv./kg P) and potassium (3.13E-08 kg antimony-equiv./kg K). On the other hand, the different results from both impact assessment methods are due to the substantially different normalization procedures. Whereas the normalization step is not specific for different resources in the CML method, each group of functionally equivalent resources and each unique single resource is normalized separately in the new approach. As agriculture is the main user of phosphates (EFMA, 2000), the share of the analyzed wheat production systems in the total P consumption in Europe is relatively high (see Figure 2).

4. Discussion and conclusions

The aim of the characterization step in LCA is to aggregate the inventory data into indicators for environmental effects (impact categories). Traditionally, the consumption rates for different abiotic resources are already aggregated during characterization into one summarizing indicator (e.g. CML method). This procedure neglects that many abiotic resources are being used for completely different purposes and, thus, the depletion of their reserves represents separate environmental problems. For example, the main function of coal is energy supply, whereas phosphate rock is mainly used for the production of mineral phosphate fertilizers. Therefore, this study assigns the consumption of abiotic resources to separate impact sub-categories according to their main function. An aggregation of functionally different resources into one impact category by means of equivalency values (e.g. kg antimony-equivalents in the CML method) is not consistent with the general LCA methodology (ISO, 2000; Consoli, 1993), because neither phosphorus nor coal, for example, are functionally equivalent to the semi-metal antimony.

Following the characterization step, normalization and weighting should be applied in order to aggregate the different resource-related impact sub-categories into one indicator for the depletion of abiotic resources. In this study, target time periods have been used to determine tolerable annual production rates, which in turn can be used to calculate weighting factors according to the distance-to-target principle. Of course the choice of the target time periods influences the weighting factors. However, as long as no clear, internationally agreed upon targets on the protection of reserves of abiotic resources are set, any definition of a target time period means a subjective choice. Therefore, different time scales (100, 300, 1000 years) have been used in this study to calculate weighting factors. The 100-year target could be regarded as default, because it may represent a realistic scenario for the substitution or recycling of abiotic resources. Taking the example of mobility (cars, airplanes, and trains), the dramatic progress over the last 100 years may serve as an indication for what is technically possible within this period of time. On the other hand, it could be also conceivable to define different target time periods for different abiotic resources. It could, for instance, be argued that an availability of 100 years is sufficient for fossil fuels, but that 300 years is necessary for other resources, because a substitution of fossil fuels is more likely to happen in the nearer future than for other resources. However, a major advantage of the suggested weighting step is its transparency and flexibility. Any subjective assumption upon employed target time periods is clearly visible and differentiated target time periods for specific resources can easily be included.

Another point that needs to be discussed is the use of reserve data for the calculation of weighting factors. On the one hand, it is important to consider the reserve of a resource, because the scarcity and thus the future availability of a resource is an important weighting criterion. On the other hand, figures on reserves of minerals and in particular of fossil fuels are often a point of criticism, because reserves can be defined in different ways and reserve data are often supposed to be biased by interested parties, such as mining industries (Guinée & Heijungs, 1995). For this study, the "proved reserve" as it is defined by WEC (1998) for fossil fuels and the very similarly defined "reserve" for minerals (USGS, 2001) have been chosen. Both reserves include that part of the materials, "that geological and engineering data demonstrate with reasonable certainty to be recoverable in future years from known reservoirs under existing economic and operating conditions" (EIA, 2000). Therefore, this definition of the reserves of a material is very much in line with the general definition of a resource given by USGS (2001; see introduction).

Another definition of reserves is the 'reserve base'. The reserve base is defined as that part of a resource that meets specific minimum physical and chemical criteria and includes also those resources, which are only marginally economical or even currently sub-economically exploitable (USGS, 2001). In LCA, potential future developments, such as improved medical treatment of human health problems or improved extraction techniques for low-quality resources are usually not considered. As the use of the reserve base is dependent on such further technical development, these data do not seem to be appropriate for use in LCA. This applies even more to the ultimate reserve, which is used in the CML method (Guinée, 2001) and "estimated by multiplying the average concentrations of chemical elements in the earth's crust by the mass of the crust" (Guinée & Heijungs, 1995). This reserve definition comprises the total deposits of an element in the earth's crust independently from its concentration and thus, is not at all equivalent to what is commonly meant by a resource.

Although having the uncertainty of data in mind, the proven reserve appears to be the most appropriate reserve definition to be used for the weighting of abiotic resources. Other reserve definitions do not really correspond to the actual safeguard subject ‘resources’. In order to address the problem of data variability, most recent data on proven reserves published by independent and reliable organizations like USGS (2001) or EIA (2000) have been used as much as possible in the present approach.

The aim of the present study is to contribute with some conceptual considerations and new ideas to the discussion about the life cycle impact assessment of resource consumption. Since these considerations were made in the context of LCA studies on agricultural production systems, they focus very much on resources, which are particularly important for such systems (phosphate rock, potash salt, lime, fossil fuels). However, it should be possible to also transfer the proposed method to other resources.

5. Annex

Annex 1: Characterization factors (CF), normalization values (NV), data on annual world production, proven reserves and calculated tolerable annual extraction rates for selected resource-related impact sub-categories

Resource	Unit	CF [B]	Impact sub-category	NV (Europe) [C]	Annual production (World) [D]	Proved reserves (World) [D]	Tolerable annual production [E] (target: reserves should last for 100 years)	Tolerable annual production (target: reserves should last for 300 years)	Tolerable annual production (target: reserves should last for 1000 years)
Oil	kg OE[A]	42.868	Depletion of fossil fuels (in MJ)	9.69E+13	3.25E+14	3.08E+16	3.08E+14	1.03E+14	3.08E+13
Natural gas	m^3	31.736							
Hard coal	kg	29.704							
Lignite	kg	8.506							
Phosphate rock [F]	kg	0.25	Depletion of phosphate rock (in kg P_2O_5)	5.57E+09	4.32E+10	3.60E+12	3.60E+10	1.20E+10	3.60E+09
Raw phosphate [F]	kg	0.32							
Potash, Potassium chloride	kg	0.105	Depletion of potash (in kg K_2O)	5.92E+09	2.56E+10	8.40E+12	8.40E+10	2.80E+10	8.40E+09
Limestone / lime	kg	0.54	Depletion of limestone and dolomite (in kg CaO)	[C]	1.16E+11	very large	no problem to be expected (very large reserves)	no problem to be expected (very large reserves)	no problem to be expected (very large reserves)
Dolomite	kg	0.3							

[A] OE = crude oil equivalents

[B] CF = characterization factor = heat values of fossil fuels (BMWi, 1995); P_2O_5, K_2O and CaO contents (Patyk & Reinhardt, 1997; www.dolomit.de)

[C] NV = normalization value = total consumption in Europe; data for fossil fuels: 1999 (EIA (2000), for other resources: 1998 (FAO (2001), no data are available for lime

[D] Data for fossil fuels: EIA (2000), for other resources: USGS (2001)

[E] Annual production rate that ensures resource availability for 100, 300 or 1000 years based on currently known recoverable reserves; calculated according to Equation (1)

[F] Phosphate rock = unprocessed phosphate ore (P_2O_5 content=25%), raw phosphate = upgraded raw material for P fertilizer production (P_2O_5 content=32%)

In the previous chapter, conceptual considerations on the impact assessment of abiotic resource consumption were discussed and a new impact assessment method was developed.

Further specific environmental impacts of arable farming are due to the use of land for crop production. The environmental impacts of occupying, reshaping and managing land for human purposes could be decreasing diversity of habitats and wildlife species. Similar to abiotic resource consumption, also for the “land use” impact category a need was identified to improve the existing impact assessment methodology. The development of an appropriate impact assessment procedure is described in the following chapter.

V. Life Cycle Impact Assessment of land use based on the Hemeroby concept

(International Journal of Life Cycle Assessment, 7 (2002) 339-348)

Abstract

The impact category 'land use' describes in the Life Cycle Assessment (LCA) methodology the environmental impacts of occupying, reshaping and managing land for human purposes. Land use can either be the long-term use of land (e.g. for arable farming) or changing the type of land use (e.g. from natural to urban area). The impact category 'land use' comprises those environmental consequences, which impact the environment due to the land use itself, for instance through the reduction of landscape elements, the planting of monocultures or artificial vegetation, or the sealing of surfaces. Important environmental consequences of land use are the decreasing availability of habitats and the decreasing diversity of wildlife species. The assessment of the environmental impacts of land use within LCA studies is the objective of this paper. Land use leads to a degradation of the naturalness of the area utilised. In this respect the naturalness of any area can be defined as the sum of land actually not influenced by humans and the remaining naturalness of land under use. To determine the remaining naturalness of land under use, this study suggests applying the Hemeroby concept. "Hemeroby is a measure for the human influence on ecosystems" (Kowarik, 1999). The Hemeroby level of an area describes the intensity of land use and can therefore be used to characterize different types of land use. Characterization factors are proposed, which allow calculating the degradation of the naturalness of an area due to a specific type of land use. Since the resource 'nature/naturalness' is on a larger geographical scale by far not homogeneous, the assessment of land use needs to be regionalized. Therefore, the impact category 'land use' has been subdivided into the impact sub-categories 'land use in European biogeographic regions'. Following the general LCA framework, normalization values for the impact sub-categories are calculated in order to facilitate the evaluation of the characterization results with regard to their share in a reference value. Weighting factors, which enable an aggregation of the results of the different land use sub-categories and make them comparable to other impact categories (e.g. climate change or acidification) are suggested based on the assumption that the current land use pattern in the European biogeographic regions is acceptable.

1. Introduction

In Life Cycle Assessment (LCA) studies the impact category ‘land use’ describes the environmental impacts of occupying, reshaping and managing land for human purposes. Land use can either be the long-term use of land (e.g. for arable farming) or a change in the type of land use (e.g. from natural to urban area) (Heijungs et al., 1997; Lindeijer et al., 1998; Müller-Wenk, 1998b; Köllner, 2000).

It is important to understand that in an LCA study direct impacts, which are related to land use, like nitrate leaching or diffuse emissions from soil to air, are accounted for elsewhere. These emissions are part of the Life Cycle Inventory (LCI) and would be considered in other impact categories than land use during the Life Cycle Impact Assessment (LCIA). The impact category ‘land use’ comprises solely those environmental consequences, that impact the environment due to the land use itself, for instance through the reduction of landscape elements (e.g. by removing forests, hedges, ponds, bushes), the planting of monocultures (e.g. cereals, conifers) or artificial vegetation (e.g. gardens), or the sealing of surfaces (e.g. for buildings or roads). There is a general agreement that such conversion, fragmentation, or degradation of natural and semi-natural ecosystems such as forests, grasslands, and wetlands for human purposes is a major reason for the decreasing diversity of habitats and wildlife species (EEA, 1998a; BfN, 1999; Statistisches Bundesamt, 1999; UNEP, 2000). Natural ecosystems are defined as one of the safeguard subjects in LCA, therefore environmental impacts of land use should be taken into account in LCA studies (Consoli et al., 1993).

Land use can be expressed in terms of the size of an area used for a specific product or process for a certain time, i.e. in m^2*year per product unit (Heijungs et al., 1992a). However, such a procedure neglects the obvious fact that different types of land use (e.g. built-up land or extensive pasture) have different impacts on the environment (Müller-Wenk, 1998b). Therefore, a measure is required to approximate the degree of environmental damages due to different land use types. One possibility to describe the extent of the influence due to human land use activities is to determine the remaining naturalness of an area used for human purposes (Kowarik, 1999). This study suggests treating ‘nature’ or ‘naturalness’ like a resource, necessary to be protected. In analogy to the impact assessment of the extraction of abiotic resources (Brentrup et al., 2002a) this approach is based on the assumption that the more naturalness is preserved the better it is for the environment. This assumption is supported by international initiatives to install a pan-European network of protected natural areas (Natura 2000, EEA, 1998a). Therefore, the basic concept in the proposed impact assessment approach for land use is to assess the naturalness of a region similar to abiotic resources.

This study suggests considering any land use that degrades the naturalness of an area as an environmental problem. The degradation of the naturalness of land not only means the conversion of natural land to land under use, but also the continuous utilisation of land, which prevents the area of getting back to a more natural status.

2. The resource “naturalness”

For the suggested approach the naturalness of an area is defined as the amount of land, which is actually not influenced by humans and the remaining naturalness of land that is currently being used. Purely natural areas, i.e. land without any direct (e.g. built-up area) or indirect (e.g. deposition of emissions) human influence hardly exist in Europe (Stanners & Bourdeau, 1995; EEA, 1998a; Kowarik, 1999). Almost the entire land in Europe is more or less influenced by human activities. Even ecosystems, which are in some regions under protection due to their high environmental value, like heathland or low-productive permanent pastures are a result of, and therefore dependent on specific forms of human land use. Generally the intensity of land use determines the ability of an area to maintain or regain a certain level of naturalness. To determine the naturalness of an area it is necessary to consider that land, which is used differently, may have a different level of naturalness.

It is important to take into account that ecosystems and thus the resource ‘nature/naturalness’ are on a larger geographical scale by far not homogenous and the protection of one ecosystem does not necessarily compensate for the intensive use of an area in another region. Therefore, the impact assessment of land use has to be related specifically to ecologically homogenous land units. Taking Europe as an example, the biogeographic regions of Europe (EEA, 1998a) have been defined based on the map of natural vegetation and thus reflect roughly the pattern of environmental conditions in Europe and can be used for such a regionalized approach. Figure 1 shows the distribution of the 11 biogeographic regions of Europe. This study describes the impact of land use always separately for each of the biogeographic regions in order to take account of the uniqueness of the nature within the different regions. Therefore, the impact category ‘land use’ is always subdivided into impact sub-categories, such as ‘land use in the Atlantic region’ or ‘land use in the Boreal region’. An approach is presented, which enables an analysis of the degree of human influence on an area due to different types of land use.

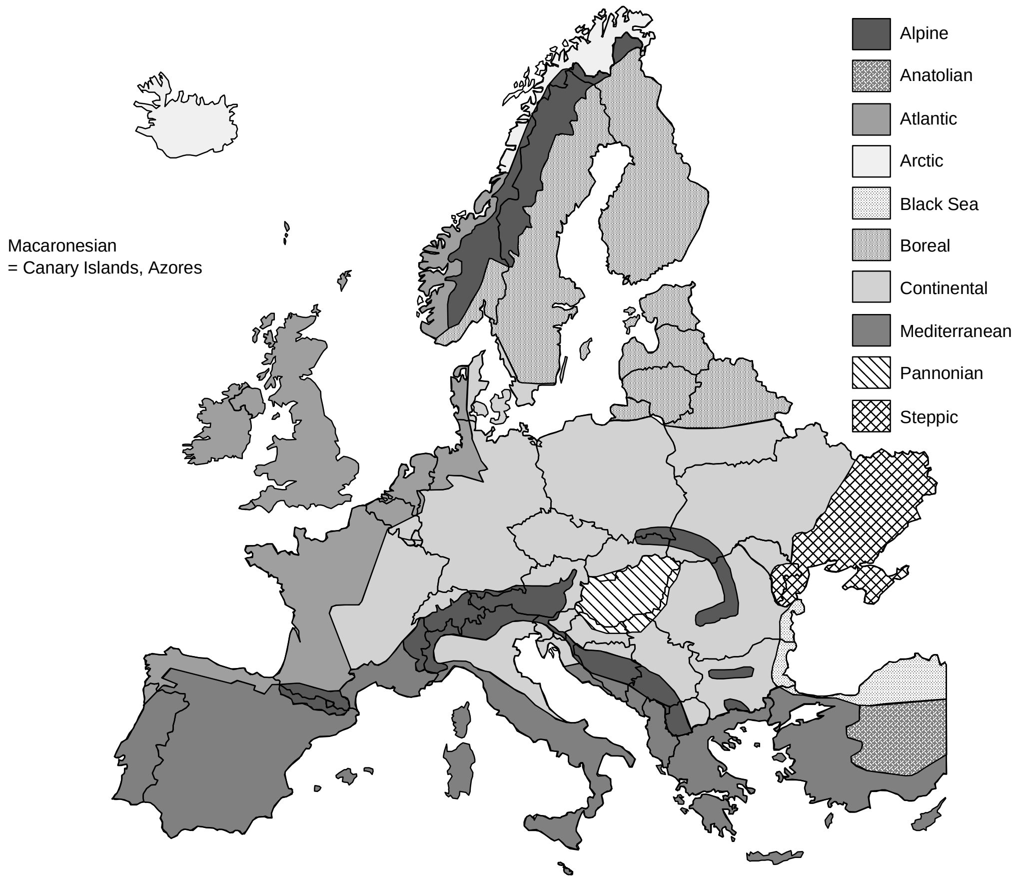

Figure 1: The biogeographic regions of Europe (adopted from EEA, 1998a)

3. The Hemeroby concept

According to Kowarik (1999) “Hemeroby is a measure for the human influence on ecosystems”. The level of Hemeroby depends on the degree of human impacts that prevent the system from developing towards a natural endpoint situation (Kowarik, 1999). This natural endpoint situation describes the reference to which any modified situation is compared.

With the Hemeroby concept it is possible to describe the degree of human influence on an area (Sukopp, 1972, 1976; Kowarik, 1999). Therefore, this concept is used to determine the deviation from naturalness as a result of specific land use types. The Hemeroby concept has been founded by Jalas in 1955 (Kowarik, 1999). Others (Sukopp, 1972; Blume & Sukopp 1976; Kowarik, 1999) have expanded the concept and developed a very differentiated scale of levels of human influence. Grabherr et al. (1998, Hemeroby of Austrian forests), Grunicke et al. (1999, Hemeroby of urban and sub-urban areas) and Rühs (2001, Hemeroby of agricultural areas) used the Hemeroby concept to investigate the level of naturalness of specific landscapes and ecosystems. These studies provide the basis for Hemeroby levels of specific land use types in the present paper.

Klöpffer & Renner (1995) were the first, who suggested the use of Hemeroby for the treatment of land use in LCA. They recommended classifying land use types according to their hemerobic level, but the approach did not include an aggregation into a summarizing land use indicator or even the possibility to further interpret the land use results by normalization or weighting. Baitz et al. (1998) criticized the use of Hemeroby in LCA, because of its focus on nature preservation and the missing integration of human needs like the increasing demand on food and living space. However, at first Hemeroby is only a descriptive indicator for the impact of human land use on the naturalness of an area and does not include any evaluation. The evaluation of the land use impacts can only be performed within the weighting step in LCA, which certainly should take into account more than only environmental considerations. This opinion is also supported by Giegrich & Sturm (1999), who adopted the Hemeroby concept as classes of naturalness for land use in forest ecosystems within a LCA study on paper (Tiedemann, 2000).

In the Hemeroby concept 11 classes of human influence on land use are distinguished in a descriptive, qualitative way (Kowarik, 1999). Table 1 gives the Hemeroby classes and descriptions of typical ecosystems, their vegetation and land use types. In addition, Table 1 contains the relative use intensity (%) and characterization factors, called 'naturalness degradation potentials' (NDP) assigned to the different levels of Hemeroby. Section 4 of this publication describes the use of the Hemeroby concept for the impact assessment of land use.

Table 1: Definition and description of Hemeroby classes and the Naturalness Degradation Potential (NDP) (Sukopp, 1972; Sukopp & Blume, 1976; Grunicke et al., 1999; Kowarik, 1999 and Rühs, 2001)

Hemeroby code (Hx), **use intensity** (%), **NDP** [a]	**Hemeroby class**	**Description** (typical ecosystems and vegetation, types of human influence)
H0 0 % NDP = 0.0	ahemerobic	no human influence, e.g.: - untouched rocky, peatbog and tundra regions in some parts of Europe
H1 10 % NDP = 0.1	oligohemerobic	small human influence, e.g.: - only indirect human influence through deposition of airborne emissions - salt meadows, growing dunes and peatbogs - hardly influenced primary forests and their natural succession levels (i.e. only cut of single trees, "Plenterwald", no introduction of site-atypical species)
H2 20 % NDP = 0.2	oligo- to mesohemerobic	small to moderate human influence, e.g.: - extensively managed forests (i.e. only little removal of timber, trees of different age at the same site, "Altersstufenwald", introduction of site-atypical species possible) - extensively drained wetlands - restored peatbogs - some wet pastures

Table 1 (continued)

Hemeroby code (Hx), **use intensity** (%), **NDP** [a]	**Hemeroby class**	**Description** (typical ecosystems and vegetation, types of human influence)
H3 30 % NDP = 0.3	mesohemerobic	moderate human influence, e.g.: - moors and heathland - managed forests - moderately managed nutrient-poor grassland and extensive meadows - shrubs and herbaceous vegetation along unspoilt lakes and rivers - permanent fallow land, fallow pasture (i.e. rare mulching and mowing (0.2-0.5/year))
H4 40 % NDP = 0.4	meso- to β-euhemerobic	moderate to strong human influence, e.g.: - intensively managed forests and young secondary forests, frequented forests near recreation areas, forest with unnatural high share of conifers - woods and bushes in parks, shrubs and hedges in agricultural areas, shrubs and herbaceous vegetation along rebuilt lakes and rivers - extensive orchard meadows - extensively used permanent grassland (i.e. 0.5 – 1.0 cuts/year, no fertiliser, no pesticides)
H5 50 % NDP = 0.5	β-euhemerobic	strong human influence, e.g.: - site-atypical coniferous forests, younger reforestation - orchard meadows - ruderal vegetation of perennials - permanent grassland (pasture or meadow) managed with medium intensity (i.e. 1.5-3.0 LU/ha (LU = livestock units), no ploughing, 1-2 cuts/year, fertilisation according to nutrient removal)
H6 60 % NDP = 0.6	β-eu- to α-euhemerobic	strong to very strong human influence, e.g.: - plantation of hedges and bushes (e.g. in gardens, along roads etc.) - ruderal meadows, lawns with meadow species - permanent grassland (pasture or meadow) managed with higher intensity (i.e. 1.5-3.0 LU/ha, ploughing max. 0.2/year, 2-3 cuts/year, fertilisation exceeds nutrient removal slightly)
H7 70 % NDP = 0.7	α-euhemerobic	very strong human influence, e.g.: - tree nurseries - intensive gardening and cultivation of special crops (e.g. fruits, vine) - annual ruderal vegetation - pasture under rotation, arable land, gardens, which are managed according to the principles of organic or extensive integrated farming (i.e. >3 LU/ha, ploughing 0.2-3.0/year, >3 cuts/year, fertilisation exceeds nutrient removal slightly, application of pesticides max. 0.3/year)
H8 80 % NDP = 0.8	α-eu- to polyhemerobic	very strong human influence to mainly artificial, e.g.: - larger relicts of vegetation within urban or industrial areas, vegetation of gravelled surfaces - intensively managed arable land and gardens (i.e. ploughing >3/year, fertilisation exceeds nutrient removal significantly, application of pesticides >0.3/year)
H9 90 % NDP = 0.9	polyhemerobic	mainly artificial, e.g.: - landfill and dump sites - partly built-up areas (railways, streets etc) - surfaces covered with new materials - strong and long-term modification of biotopes
H10 100 % NDP = 1.0	metahemerobic	purely artificial, e.g.: - completely sealed, built-up or contaminated surfaces (i.e. no habitat for plants)

[a] NDP = Naturalness degradation potential

4. Impact assessment of different land use types

The impact assessment approach for the impact category ‘land use’ follows the general Life Cycle Impact Assessment (LCIA) methodology as described by ISO (ISO, 2000) and SETAC (e.g. Consoli et al., 1993; Udo de Haes, 1999b, c). It consists of the three steps: characterization, normalization and weighting.

4.1 Characterization

The *characterization* of land use impacts means to calculate to what extent a particular type of land use degrades the naturalness of an area. This is done by multiplying the Life Cycle Inventory data for land use (given in m^2*year of a specific land use type, e.g. arable land) by the respective ‘naturalness degradation potentials’ (NDP), which are the characterization factors for the land use impact category. In order to derive the NDP values the Hemeroby concept is applied. Table 1 gives the Hemeroby classes with the respective land use intensities and NDP values. This procedure is in contrast to Giegrich & Sturm (1999), who did not convert the distinct Hemeroby classes into a cardinal scale of characterization factors. However, in the context of the present study it seems justified to define ‘ahemerobic’ as a situation, in which the naturalness of an area is not influenced by human activities, i.e. the land use intensity is 0% and the resulting characterization factor is 0 (Table 1). On the other hand a ‘metahemerobic’ land use situation can be regarded as being equivalent to 100% use intensity and thus gets a characterization factor of 1. In between these extremes Table 1 gives 9 levels of Hemeroby. In order to enable a comparison and aggregation of the impact of different land use types and thus also of different land use intensities, the Hemeroby classes have been linearly transformed into characterization factors between 0 and 1 (Table 1). However, this linear transformation might be a simplification of the complex parameter ‘Hemeroby’ (Giegrich & Sturm, 1999), but for the time being this procedure appears to be the only operational and obvious approach of modelling characterization factors on the basis of Hemeroby classes.
Table 2 gives characterization factors (= NDP values) for some relevant land use types.

Table 2: NDP values for different land use types

Land use type [a]	**NDP** [b]
Continuous urban fabric	0.95
Industrial or commercial units	0.95
Road and rail networks	0.90

Table 2 (continued)

Land use type [a]	**NDP** [b]
Discontinuous urban fabric	0.85
Intensive arable	0.80
Extensive arable	0.70
Green urban areas	0.70
Intensive permanent pasture	0.60
Extensive permanent pasture	0.50
Intensively managed forests	0.40
Permanent fallow land	0.30
Extensively managed forests	0.20
Salt meadows, growing peatbogs	0.10

[a] Description of the land use types is given in Annex 1

[b] NDP = Naturalness degradation potential; NDPs for other land use types can be derived from Table 1.

The NDP values can be applied according to equation (1).

$$NDI_{bioreg\ i,sys} = area\ under\ use_{type\ i,bioreg\ i} \times NDP_{type\ i} \qquad (1)$$

with: NDI $_{\text{bioreg i}}$ = naturalness degradation indicator for the analysed system in biogeographic region i [in m^2*year]

area under use $_{\text{type i, bioreg i}}$ = area used for land use type i in biogeographic region i [in m^2*year]

NDP $_{\text{typ i}}$ = naturalness degradation potential for land use type i

That means that for instance 1000 m^2 used as intensive permanent pasture for 1 year, degrades the naturalness of that area by 60 %, i.e. the total 1000 m^2 used as pasture are supposed to be equivalent to 400 m^2 of natural land.

4.2 Normalization

In LCA the normalization relates the environmental impacts derived from a specific product or process under analysis to the total environmental impact in a defined reference region (e.g. Europe or world). The normalization is done separately for each environmental impact (e.g. climate change, acidification, land use). “The aim of ... normalization ... is to better understand the magnitude for each indicator result of the product system under study” (ISO, 2000). Therefore, the indicator results per functional unit for each environmental impact (e.g. NDI for land use) are related to the respective indicator results for the defined reference situation (e.g. total NDI for Europe). The normalization step “increases the comparability of the data from different impact categories and thus creates a more sound basis" for the weighting step (Consoli et al., 1993). Furthermore, the normalization eliminates the different dimensions (e.g. m^2*year for land use, CO_2-equivalents for climate change), which is a prerequisite for the subsequent weighting step.

To perform the normalization of land use indicator results (NDI in m^2*year per functional unit) derived after characterization, it is at first necessary to determine the total degradation of naturalness in the defined reference region. For this publication, the biogeographic regions of Europe and total Europe have been chosen as references. Principally, this approach is applicable to any region provided that the necessary land use data are available. For the biogeographic regions of Europe the European Topic Centre on Land Cover (ETC/LC), established by the European Environment Agency, provides a comprehensive inventory of land cover data (Satellus, 2000). To estimate the total degradation rate of naturalness for each biogeographic region, the NDP values (Tables 1, 2) have been assigned to the ETC/LC land use classes. The ETC/LC land use classes, their main characteristics, and the assigned Hemeroby classes and NDP values are given in Annex 1. By multiplication of the area containing a specific type of land use with the respective NDP, the NDI value can be determined. After doing so for all land use types occurring in one biogeographic region, the results can be aggregated in order to get the total NDI value for that region (equation (2)).

$$NDI_{Region,\ bioreg\ i} = \sum_i \left(area_{type\ i,\ bioreg\ i} \times NDP_{type\ i} \right) \qquad (2)$$

with: $NDI_{Region,\ bioreg\ i}$ = total naturalness degradation indicator for biogeographic region i = Regional normalization value for biogeographic region i [in m^2*year]

$area_{type\ i,\ bioreg\ i}$ = land area of land use type i in biogeographic region i [in m^2*year]

$NDP_{type\ i}$ = naturalness degradation potential for land use type i in biogeographic region i

However, a comparison of land use impacts occurring in different biogeographic regions or even to other environmental impacts like acidification or climate change is difficult using these normalization values because they are based on different reference regions (i.e. the biogeographic zones). An example: To make normalized NDI values for land use in the Atlantic region comparable to indicator values for acidification, it would be necessary to know about the total acidification potential in the Atlantic region, since this is the reference region for the normalization for the land use impacts. Usually no data for environmental impacts other than land use are available on the level of biogeographic regions. Therefore, to make the normalization results of land use comparable to the results of other impacts a more common reference region like total Europe has to be selected. To calculate normalization values for land use impacts on a total European scale the normalization values for each biogeographic region had to be extrapolated according to (3). This extrapolation means to artificially project the land use intensity and therefore the degradation rate of naturalness within one biogeographic region to the total area of Europe (EEA, 1998b).

$$NDI_{Europe,\ bioreg\ i} = \frac{area_{Europe}}{area_{bioreg\ i}} \times NDI_{Region,\ bioreg\ i} \qquad (3)$$

with: NDI $_{\text{Europe, bioreg i}}$ = total naturalness degradation indicator for biogeographic region i = European normalization value for biogeographic region i [in m^2*year]
area $_{\text{Europe}}$ = total area of Europe (2.298*10^{13} m^2)
area $_{\text{bioreg i}}$ = total area of biogeographic region i (in m^2)
NDI $_{\text{Region, bioreg i}}$ = total naturalness degradation indicator for biogeographic region i = Regional normalization value for biogeographic region i [in m^2*year]

The normalization factors for the regional and the European level for land use are given in Table 3. For the Anatolian and Arctic region no land cover data were available.
The normalization values can be applied according to equation (4).

$$normalised\ NDI_{bioreg\ i,sys} = \frac{NDI_{bioreg\ i,sys}}{NDI_{Europe,\ bioreg\ i}} \qquad (4)$$

with: normalized NDI $_{\text{bioreg i, sys}}$= normalized naturalness degradation indicator for the analysed system in biogeographic region i
NDI $_{\text{bioreg i,sys}}$ = naturalness degradation indicator for the analysed system in biogeographic region i [in m^2*year]
NDI $_{\text{Europe, bioreg i}}$ = total European naturalness degradation indicator for biogeographic region i [in m^2*year]

The result of the normalization step is the contribution of a particular land use to the degradation of the naturalness in a specific biogeographic region of Europe. For the example of using 1000 m^2 as intensive permanent pasture for 1 year in the Atlantic region this means, that the resulting NDI value of 600 m^2*year is divided by 1.3*10^{13} m^2*year. The result of 4.6*10^{-11}

represents the contribution of using 1000 m^2 as pasture to the total degradation of naturalness in the Atlantic type of land in Europe.

Table 3: Normalization values for land use in European biogeographic regions

Biogeographic region	**Normalization value** (regional level, NDI $_{Region,\ bioreg}$, m^2*year) [a]	**Normalization value** (European level, NDI $_{Europe,\ bioreg}$, m^2*year) [a]
Alpine region	8.84E+06	8,87E+12
Atlantic region	4.34E+07	1,30E+13
Black sea region	5.62E+05	1,14E+13
Boreal region	8.77E+06	1,15E+13
Continental region	7.71E+07	1,35E+13
Macaronesian region	2.30E+05	7,27E+12
Mediterranean region	4.40E+07	1,17E+13
Pannonian region	7.22E+06	1,52E+13
Steppic region	2.00E+06	1,53E+13

[a] NDI $_{bioreg\ i,\ total}$ = Total naturalness degradation indicator for biogeographic region i

4.3 Weighting

The weighting step is the final part of the life cycle impact assessment. During weighting the different effects such as land use or acidification are evaluated according to their potential to harm the environment. Weighting provides a valuable tool to interpret the normalized indicator values for the different impacts further in order to support users of LCA studies with clear and aggregated results. During the weighting step the normalized indicator values for each environmental impact are multiplied by so-called weighting factors. These weighting factors represent the potential of the different impacts to harm the LCA safeguard subjects ‘ecosystems, human health and resources’. Weighting factors can be calculated on the basis of expert panels (Landbank, 1994; Goedkoop & Spriensma, 1999), monetarisation approaches (Steen, 1999) or by the application of the so-called ‘distance-to-target’ principle (BUWAL, 1998; Goedkoop, 1995). After weighting the indicator values for each environmental impact can be summed up to one overall environmental indicator. For this study the ‘distance-to-target’ principle has been used to calculate weighting factors. Following this principle, weighting factors are derived from the ratio between the current level of an environmental impact (e.g. climate change in CO_2-equivalents per year in Europe) and a target level defined for that impact (e.g. goals of Kyoto protocol for climate change in Europe). This very transparent calculation of weighting factors makes it possible to consider that the evaluation of environmental problems may differ substantially between regions and within time. Through the selection of appropriate environmental targets and the use of recent data it is possible to derive weighting factors valid for a defined region and time. The application of widely accepted environmental targets set by international organisations allows an integration of scientific knowledge about environmental effects, possible damages to the safeguard subjects, social values and economic pressures.

Weighting of the impact category ‘land use’ and the proposed indicator ‘degradation of naturalness’ requires to find targets on a tolerable anthropogenic utilisation of land in Europe. Today such targets, based on scientific evaluation or political decisions, are not available. Therefore, this study suggests weighting factors, which are based on the assumption that the current land use pattern and the resulting naturalness in the different European biogeographic regions is tolerable. By applying the distance-to-target principle to land use, this assumption implies that the current situation is seen as equivalent to the target situation and the weighting factor for land use is 1, independent of the biogeographic region the land use takes place.

The proposed target is not based on a wider scientific or political agreement. However, if such agreements will be reached in future, it is no problem to integrate other targets into this approach in order to calculate new weighting factors. The weighting factors can be applied according to equation (5).

$$NDI_{sys} = \sum_i normalised\ NDI_{bioreg\ i,sys} \times WF_{bioreg\ i} \quad (5)$$

with: NDI $_{sys}$ = naturalness degradation indicator for the analysed system
normalized NDI $_{bioreg\ i,sys}$ = normalized naturalness degradation indicator for the analysed system located in biogeographic region i
WF $_{bioreg\ i}$ = weighting factor for land use in biogeographic region i = 1

5. Discussion and conclusions

Current approaches to assess the impact of human land use are often based on empirical investigations of species diversity (e.g. of vascular plants) as a result of different types of land use (Goedkoop & Spriensma, 1999; Köllner, 2000). These approaches encounter two main problems. First, it is difficult to determine a reference situation on the basis of a single indicator such as species diversity. For instance Goedkoop & Spriensma (1999) and Köllner (2000) take the species richness of Swiss Lowlands as a reference. Species numbers of vascular plants found on areas used for different human purposes have been compared to this reference in Swiss lowlands to derive characterization factors valid for the whole of Europe. However, the number of plant species per area varies already naturally by factor 10-15 within Europe (200 – 3000 species per 10,000 km^2, BfN, 1999), so that the data for the Swiss Lowlands (270 species) cannot be representative for Europe. Furthermore, not only the number, but also the species composition (e.g. share of neophytes) is important for the assessment of the influence of human land use (Kretschmer et al., 1997). Consequently, “the number of species is an indicator of limited value for the ecological integrity of landscapes” (Kretschmer et al., 1997).

The proposed impact assessment method for land use based on the Hemeroby concept enables to estimate the impact of a specific land use type on the degradation of the naturalness of an area (Figure 2).

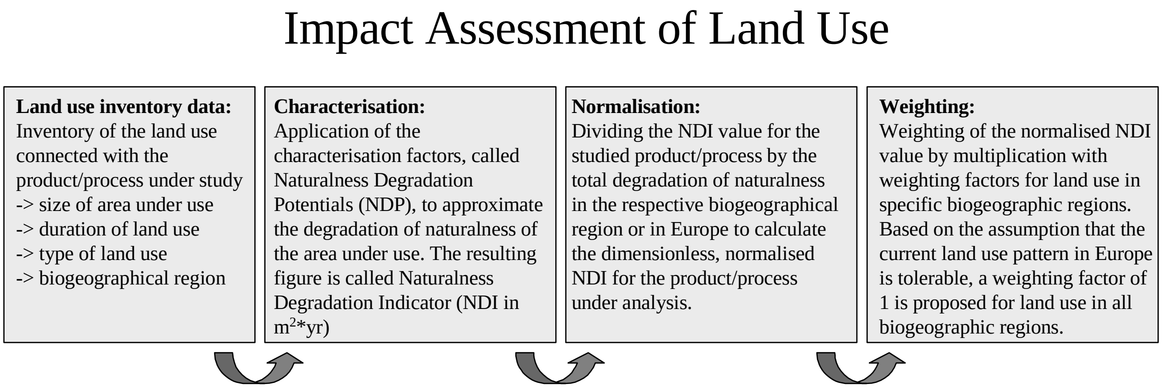

Figure 2: Proposed impact assessment procedure for land use

The Hemeroby concept has been used to assess the intensity of different land use types and their potential to degrade the naturalness of land. This part of the impact assessment is called characterization. A main advantage of the Hemeroby concept is that the description of the intensity of land use is not based on a single, eventually misleading indicator like species variety. Hemeroby is rather an integrated, descriptive measure of different human influences, which prevent a system from developing towards a situation without any anthropogenic influence (Rühs, 2001). The description of Hemeroby levels given in the scientific literature (Sukopp, 1972; Sukopp & Blume, 1976; Grunicke et al., 1999; Kowarik, 1999 and Rühs, 2001) provides an independent frame, which makes it possible to assign Hemeroby levels to specific areas based on the analysis and description of the land use types (see Table 1).

The proposed approach enables an assessment of land use for ecologically homogenous regions like the biogeographic regions of Europe separately. This separation into regions is important, because of the great spatial diversity of the resource ‘nature’ for instance throughout Europe.

Preserving a high level of naturalness in the Alps would not compensate for degrading Mediterranean forests, as both regions show very different environmental conditions (climate, soil, water) leading to different types of ecosystems. The proposed method could even be improved, if land use data for smaller and therefore ecologically more homogeneous units (e.g. biotope types) could be used. Unfortunately sufficient land use data for Europe are currently not available on the level of biotope types.

To evaluate the relevance of land use impacts, which are calculated for any system under investigation with the help of the proposed characterization method, it is necessary to relate these characterization results to an independent reference value. This is the aim of the normalization step. In this respect it is important that the Hemeroby concept is not only applicable on small areas connected to the specific system under study, but also on a larger reference region. Based on land cover data published by the European Topic Centre on Land Cover (Satellus, 2000), first the entire land use impacts have been calculated for each biogeographic region separately. By extrapolation of the land use impacts within one biogeographic region to the total European area, European normalization values have been calculated, which enable the comparison of normalized indicators for land use to indicators of other environmental impacts, provided that these indicators are normalized using the same reference region (i.e. Europe).

To make the normalized indicator values for land use (NDI) equivalent to normalized indicator values of other environmental impact categories (e.g. acidification, climate change) an evaluation of the potential of the different environmental problems to harm the safeguard subjects ecosystems, human health, and resources is necessary. In this approach the weighting step is based on the distance-to-target principle. As no generally agreed target on a tolerable land use intensity in Europe exists the current land use situation has been regarded as acceptable and thus a weighting factor of 1 is proposed for land use independent of the biogeographic region.

From an environmental point of view only, this assumption may not be justified, because the intensification of land use has led to a serious decrease in the diversity of habitats and species (EEA, 1998a; UNEP, 2000). However, weighting factors based on environmental targets set by international conventions (e.g. UN-FCCC, 1998; UN-ECE/CLRTAP, 1999) usually comprise more than only the environmental dimension of the impacts. These conventions are a result of long discussion processes between science, economy, and policy and can therefore be regarded as a compromise considering all elements of sustainability, i.e. environmental, economic, and social aspects. For land use it has to be considered that a certain level of land utilisation and consequently a reduced naturalness must be accepted because of important human requirements. The need for land to produce food and to provide living space for humans is an important additional dimension of land use. Given the trends of a growing population, mobility, and urbanisation, the competition between different types of land use (e.g. nature reserves vs. agricultural land vs. urban area) will increase in future (FAO, 2000). Since land is a strictly limited resource, an as efficient as possible use of land for whatever purpose is beneficial and should be considered in LCA. In particular improving the land use efficiency without changing the Hemeroby level of that area (e.g. by increased yields in arable farming) could help to maintain the current average land use intensity in spite of a higher demand on food products. For LCA studies in the agricultural sector it is therefore sensible to choose rather a product related functional unit (e.g. 1 ton of cereal grain) instead of an area related functional unit (1 ha under cultivation) in order to consider possible differences in the land use efficiency. Taking into account the increasing competition between nature preservation and land utilisation the maintenance of the current land use situation in Europe may be already a quite ambitious target.

The proposed approach to impact assessment of land use based on the Hemeroby concept could also be used for other purposes. It would for instance be possible to use it in environmental management systems like EMAS (Spindler, 1998), which aim at the investigation, monitoring and improvement of the overall environmental performance of entire enterprises like farms. Part of such systems is an inquiry of the initial status and the definition of environmental goals for the particular enterprise. In this context also land use impacts determined with the suggested land use impact assessment method could be included into such an environmental management system.

6. Annex

Annex 1: Description of the ETC/LC Land Cover categories and assignment of Hemeroby classes and characterization factors to these categories [a]

ETC/LC land cover category	**Main characteristics** (min. area 25 ha)	**Hemeroby class** (Hx) and **characterization factor** (NDP) [b] (see Tab. 1)
1. artificial surfaces		
1.1.1. continuous urban fabric	- mainly covered by buildings, roads, 80-100% sealed surface - non-linear vegetation and bare soil exceptional	H9-H10 NDP=0.95
1.1.2. discontinuous urban fabric	- buildings, roads, sealed surface dominates (50-80%), but is associated with vegetated areas and bare soil - e.g. suburbs and urban districts in rural areas	H8-H9 NDP =0.85
1.2.1. industrial or commercial units	- mainly artificially surfaced area without vegetation - e.g. hospitals, commercial centres, university sites, major livestock facilities etc.	H9-H10 NDP =0.95
1.2.2. road and rail networks	- motorways, railways plus associated structures - min. width 100 m	H9 NDP =0.90
1.2.3. port areas	- infrastructure of port areas - incl. quays, dockyards etc. - excl. water basins	H9-H10 NDP=0.95
1.2.4. airports	- runways, buildings, associated grassed area	H9 NDP=0.90
1.3.1. mineral extraction sites	- sand and gravel pits, quarries, open-cast mines, incl. associated infrastructure - disused sites with vegetation are excluded	H9 NDP=0.90
1.3.2. dump sites	- public, industrial or mine dump sites - partly vegetated	H9 NDP=0.90
1.3.3. constructions sites	- spaces under construction, soil or bedrock excavation, earthworks - agricultural interventions (e.g. drainage) are excluded	H9-H10 NDP=0.95
1.4.1. green urban areas	- vegetated areas within urban fabric - e.g. parks, cemeteries	H7 NDP=0.70
1.4.2. sport and leisure facilities	- e.g. camping parks, sport grounds, golf courses etc. - incl. formal parks outside urban areas	H7 NDP=0.70
2. agricultural areas		
2.1.1. non-irrigated arable land	- all arable crops, fallow land, vegetables, flower and tree nurseries, pasture under rotation - permanent pasture is excluded	H8 NDP=0.80
2.1.2. permanently irrigated land	- permanent or periodical irrigation with necessary infrastructure - excl. sporadically irrigation	H8 NDP=0.80
2.1.3. rice fields	- flat surfaces with irrigation channels	H8 NDP=0.80
2.2.1. vineyards	- areas planted with vines	H7 NDP=0.70
2.2.2. fruit trees and berry plantations	- parcels planted with fruit/nut trees or shrubs	H7 NDP=0.70
2.2.3. olive groves	- areas planted with olive trees - incl. mixed olive/vine cultivation	H7 NDP=0.70
2.3.1. pastures	- not under rotation - mainly for grazing, sometimes for fodder - incl. hedges - close to inhabited/cultivated areas	H5-H6 NDP=0.55
2.4.1. annual crops associated with permanent crops	- mixture of annual and perennial crops on the same parcel	H6 NDP=0.60
2.4.2. complex cultivation patterns	- composition of small units of diverse annual and perennial crops	H6 NDP=0.60

Annex 1 (continued)

ETC/LC land cover category	**Main characteristics** (min. area 25 ha)	**Hemeroby class** (Hx) and **characterization factor** (NDP) [b] (see Tab. 1)
2.4.3. land principally occupied by agriculture, with significant areas of natural vegetation	- agricultural land interspersed with significant natural areas - 25-75 % agricultural land	H5 NDP=0.50
2.4.4. agro-forestry areas	- annual crops or pasture under wooded cover - frequently in Southern Europe	H4 NDP=0.40
3. forests and semi-natural areas		
3.1.1. broad-leaved forest	- broad-leaved species predominate (> 75 %)	H3 NDP=0.30
3.1.2. coniferous forest	- coniferous species predominate (> 75 %)	H4 NDP=0.40
3.1.3. mixed forest	- neither broad-leaved nor coniferous species predominate	H3-H4 NDP=0.35
3.2.1. natural grassland	- low productivity grassland - frequently includes rocks, briars, heathland - extensive agricultural use - normally no parcel boundaries - often far from inhabited areas	H3-H4 NDP=0.35
3.2.2. moors and heathland	- low and close vegetation cover (bushes, shrubs, herbaceous plants) - atlantic and subalpine moors	H3 NDP=0.30
3.2.3. sclerophyllous vegetation	- bushy vegetation, e.g. garrigue - typical for the Mediterranean	H2-H3 NDP=0.25
3.2.4. transitional woodland/shrub	- bushy or herbaceous vegetation with scattered trees - due to degradation or regeneration processes	H3 NDP=0.30
3.3.1. beaches, dunes, sands	- excl. artificial surfaces - often used as recreation area	H2-H3 NDP=0.25
3.3.2. bare rock	- bare rock	H1 NDP=0.10
3.3.3. sparsely vegetated areas	- includes steppes, tundra, badlands, areas of high altitude	H1-H2 NDP=0.15
3.3.4. burnt areas	- areas affected by recent fires	H5 NDP=0.50
3.3.5. glaciers and perpetual snow	- glaciers and permanent snow	H1 NDP=0.10
4. wetlands		
4.1.1. inland marshes	- lowlands, usually flooded in winter and water-saturated all year round	H1-H2 NDP=0.15
4.1.2. peatbog	- mainly consisting of decomposed sphagnum mosses - partly exploited	H1-H2 NDP=0.15
4.2.1. salt marshes	- low-lying areas vegetated with halophytes - above high-tide line, but sometimes flooded by sea water	H1-H2 NDP=0.15
4.2.2. salines	- sections of salt marshes under use for salt extraction, oyster or fish farming	H7 NDP=0.70
4.2.3. intertidal flats	- unvegetated areas of mud, sand or rock lying between high- and low-tide line	H1-H2 NDP=0.15

[a] ETC/LC = European Topic Centre on Land Cover

[b] NDI = Naturalness degradation potential

In the two previous chapters, impact assessment methods for resource consumption and land use were proposed. Environmental indicators for both impact categories were developed.

In the following chapter, indicators for the other environmental impacts relevant to arable crop production (climate change, toxicity, acidification, and eutrophication) are suggested. Finally, an aggregation procedure is developed, which enables the calculation of two summarizing indicators for (a) resource depletion and (b) impacts on natural eco-systems and human health.

VI. Investigation of the Environmental Impact of Agricultural Crop Production using the Life Cycle Assessment (LCA) Methodology

I. Development of a LCA method tailored to agricultural crop production

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Abstract

A new Life Cycle Assessment (LCA) method is presented, which is specifically tailored to plant nutrition in arable crop production. Generally, LCA is a methodology to assess all environmental impacts associated with a product or a process by accounting and evaluating its resource consumption and emissions. In LCA studies always the entire production system is considered, i.e. for crop production systems the analysis includes not only the on-field activities, but also all impacts related to the production of raw materials (minerals, fossil fuels) and farm inputs like fertilizers, plant protection substances, machinery or seeds.

The LCA method developed in this study evaluates the impact of emissions and resource consumption associated with crop production on the following environmental effects: *depletion of abiotic resources*, *land use*, *climate change*, *toxicity*, *acidification*, and *eutrophication*. In order to enable conclusions on the overall environmental preference of alternative crop nutrition systems under comparison, an aggregation procedure for the calculation of summarizing indicators for resource depletion (RDI) and environmental impacts (EcoX) has been developed. The higher the EcoX value, the higher is the overall environmental burden associated with the product under investigation. An environmental analysis of agricultural crop production systems based on this LCA method is especially appropriate in order to:

(1) detect environmental hot spots in the system,

(2) trace back environmental impacts of arable farming products to their sources and on that basis to suggest options for improvement and

(3) contribute to the discussion on the environmental preference of alternative crop nutrition systems in an informed way.

1. Introduction

Agriculture is expected to be competitive, to produce high quality food in sufficient quantities and to be environmentally benign (Commission of the European Communities, 1999; UN-

DSD, 2000). To evaluate the sustainability of agricultural production systems and to define the appropriate production intensity, it is necessary to have appropriate indicators in place.

The environmental impacts of agriculture have been analyzed in numerous investigations. These focus only on individual effects such as nitrate leaching or ammonia volatilization (e.g. Bach & Becker, 1995; ECETOC, 1988, 1994; Engels, 1993; Sommer, 1992). However, agricultural production systems contribute to a wide range of environmental impacts (e.g. climate change, acidification, eutrophication etc.). The analysis of individual effects do not permit an overall conclusion from an environmental point of view on the overall preference of one or another production strategy. Different environmental management tools such as EMAS (Eco Management and Audit Scheme; Spindler, 1998) or KUL (Kriterien umweltverträglicher Landbewirtschaftung [Criteria for an Environmentally Compatible Agriculture]; Eckert et al., 1999) have been developed to investigate the overall environmental performance of farms. Such systems are used (1) to detect options for improvement and (2) to compare or to monitor the environmental impact of farms. In order to analyze agricultural products, the product itself and the entire production system to produce it should be investigated. The Life Cycle Assessment (LCA) methodology is especially designed to study all environmental impacts connected to an entire production system. For crop production not only on-field activities but also all impacts related to the production of farm inputs, such as emissions and resource consumption due to the production of fertilizers, are included. All impacts are related to one common unit (e.g. 1 tonne of wheat grain) and summarized into environmental effects (such as climate change or acidification) or even aggregated into a summarizing environmental index. Such an index allows the ranking of different product or production alternatives according to their overall environmental performance.

ISO (International Organization for Standardization) and SETAC (Society for Environmental Toxicology and Chemistry) (ISO, 1997; ISO, 1998a, b; ISO, 2000; Consoli et al., 1993; Udo de Haes et al., 1999b, c) provide a general description of the LCA methodology. However, the impact assessment procedure, the aggregation methods for the different impact categories and the final calculation of a summarizing environmental index are still under discussion. Furthermore, if currently available LCA applications are used to investigate agricultural products or processes, the methods reveal some shortcomings, such as the missing integration of impacts relevant to agriculture (e.g. land use, resource consumption; Brentrup et al., 2001).

This paper describes an LCA method that has been developed to cover the environmental effects, which are relevant to agricultural crop production with a special focus on plant nutrition, and to integrate the best available procedures within the impact assessment phase. In

addition, a normalization and weighting procedure is suggested, which enables the aggregation of the environmental impacts into two summarizing indicators, one for impacts on eco-systems and human health, and the other one for resource depletion.
A subsequent paper will describe the application of this methodology to investigate the environmental impact of different production intensities of winter wheat.

2. General introduction of the Life Cycle Assessment (LCA) methodology

LCA is a methodology to assess all environmental impacts associated with a product, process or activity by accounting and evaluating the resource consumption and the emissions. According to ISO (ISO, 1997) LCA is divided into four steps, which are (1) *goal and scope definition*, (2) *inventory analysis*, (3) *impact assessment*, and (4) *interpretation*.

2.1 Goal and scope definition

The first step in LCA is the definition of the goal and scope of the study. This step defines the reasons for the LCA study and the intended use of the results. For LCA studies in the agricultural sector this could be for instance to investigate the environmental impacts of different intensities in crop production or to analyze the advantages and disadvantages of intensive or extensive arable farming systems.

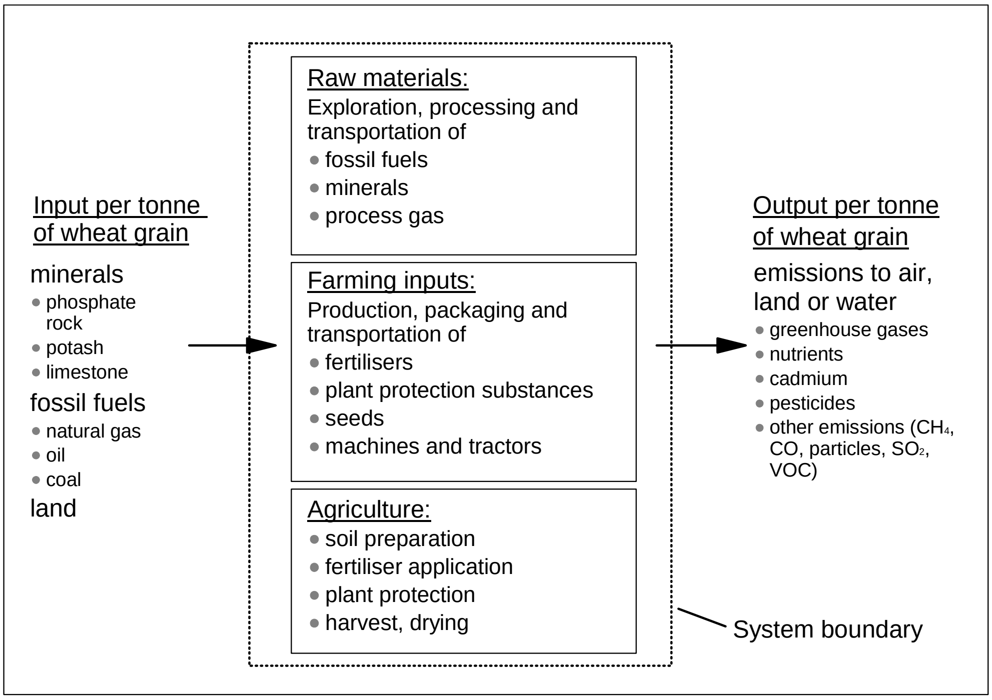

Figure 1: System boundary, relevant in- and outputs, and functional unit of a wheat production system

Furthermore, this step describes the system under investigation, its function, and boundaries. Subsequently, a reference unit (*functional_unit*; ISO, 1998a) is defined, to which all environmental impacts are related to, and which should represent the function of the analyzed system. Figure 1 gives an example for an arable farming system with the primary function to produce winter wheat. The appropriate functional unit (FU) for this system is one tonne of grain.

2.2 Life Cycle Inventory (LCI) analysis

The inventory analysis compiles all resources that are needed for and all emissions that are released by the specific system under investigation and relates them to the defined functional unit (ISO, 1998a).

2.3 Life Cycle Impact Assessment (LCIA)

The *impact assessment* aims at a further interpretation of the LCI data. The inventory data are multiplied by *characterization factors* (CF) to give indicators for the so-called environmental *impact categories* (Equation 1).

$$impact\ category\ \ indicator_i = \sum_j (E_j\ or\ R_j) \times CF_{i,j} \tag{1}$$

where:	impact category indicator $_i$	= indicator value per functional unit for impact category i
	E_j or R_j	= Release of emission j or consumption of resource j per functional unit
	$CF_{i,j}$	= Characterization factor for emission j or resource j contributing to impact category i

The characterization factors represent the potential of a single emission or resource consumption to contribute to the respective impact category (ISO, 2000). An example for such an indicator is the Global Warming Potential (GWP) expressed in CO_2-equivalents, which is derived from the rate of CO_2, CH_4, N_2O and CFC emissions multiplied by their respective characterization factor (e.g. 1 for CO_2, 310 for N_2O). According to ISO the aggregation of inventory results to impact categories is mandatory in LCIA (ISO, 2000). The list of impact category indicator values for a system under investigation is called its *environmental profile*. Table 1 gives a list of the impact categories as proposed by the SETAC-Europe Working Group on LCIA (WIA-2) (Udo de Haes et al., 1999b, c).

Table 1: List of environmental effects (= impact categories) treated in LCA

General distinction	**Impact category**
Input related categories	Depletion of abiotic resources
	Land use
Output related categories	Climate change (= Global warming)
	Stratospheric ozone depletion
	Human toxicity, ecotoxicity
	Photo-oxidant formation (=”Summer smog”)
	Acidification
	Nutrification (= Eutrophication)

For further interpretation of the environmental profile, a *normalization* step relates the indicator values to reference values. The resulting normalized indicator values give the share of the analyzed system in the defined reference, e.g. European values for the respective impact categories. For a system under investigation this would mean the division of the Global Warming Potential calculated for this specific system by the total Global Warming Potential for a defined region, e.g. Europe.

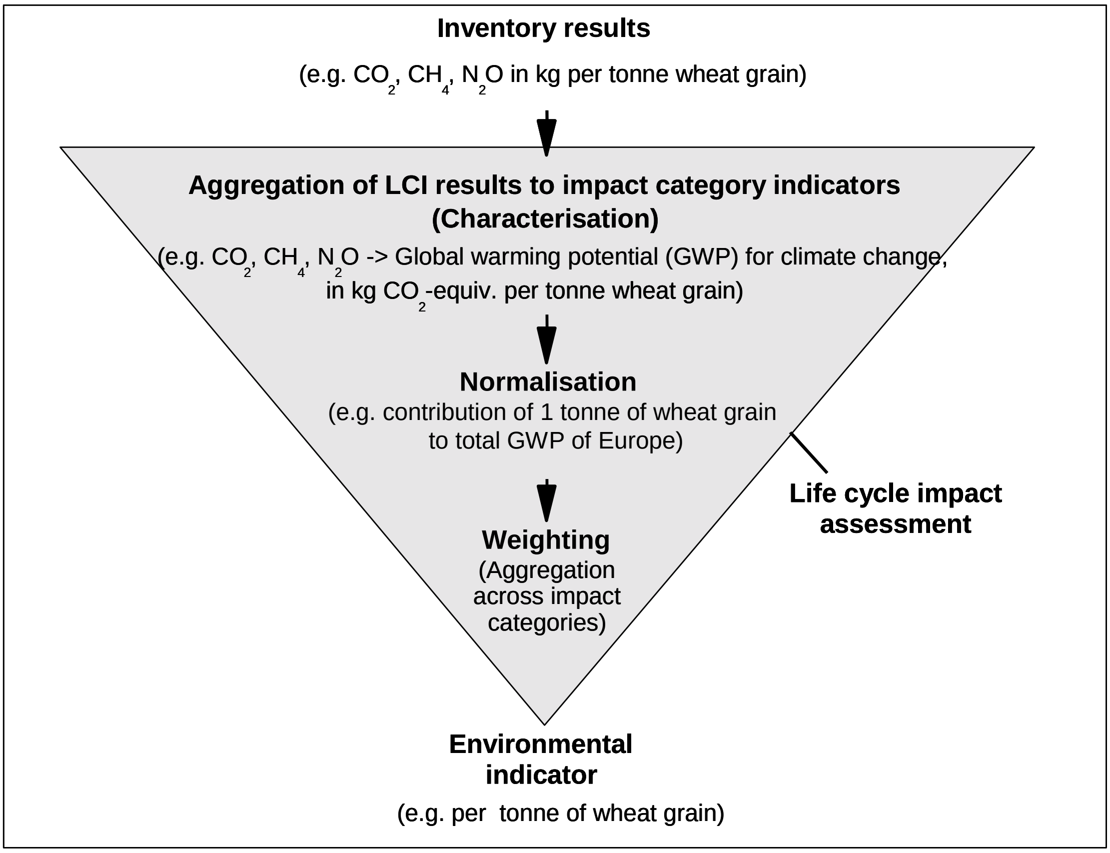

Figure 2: The general Life Cycle Impact Assessment procedure

In the following *weighting* step the normalized indicator values are multiplied by *weighting factors*, which represent the potential of the different environmental impact categories to harm natural ecosystems, human health and resources. For example the normalized indicator value for Global Warming for a product or system under analysis is multiplied by a specific weighting factor for Global Warming. Subsequently, the weighted indicator values can be summed up to one overall environmental indicator. ISO (2000) describes both, normalization and weighting as optional elements of LCIA. Figure 2 gives an overview of the general life cycle impact assessment procedure and its different elements.

3. Life cycle impact assessment tailored to plant nutrition in agricultural crop production

This study describes an LCIA method, which has been refined to evaluate the environmental impact of plant nutrition in arable crop production and which includes a new combination of impact assessment procedures. In addition this LCIA approach includes new normalization values and weighting factors in order to enable a conclusion on the overall environmental preference of different plant nutrition systems.

3.1 Characterization

For most impact categories various methods for the aggregation of LCI data to impact category indicators are described in the literature. These methods have been analyzed in a literature study. For climate change, human toxicity, eco-toxicity and acidification appropriate characterization methods are available. However, for depletion of abiotic resources and land use impact categories, a need for improvement has been identified (Brentrup et al., 2002a, b). Since in agricultural crop production systems there are no emissions, which contribute to the depletion of the stratospheric ozone layer (such as chlorofluorocarbons), this impact category has been excluded from the suggested LCA approach. Recent LCA studies on crop production systems have shown that the contribution of arable farming to the formation of tropospheric photo-oxidants (e.g. ozone, “summer smog”) due to emissions of nitrogen oxides (NOx) and volatile organic compounds (VOC) is negligible compared to the contribution of other human activities like traffic or industrial production (Brentrup et al., 2001; Küsters & Brentrup, 1999; Küsters & Jenssen, 1998). Therefore, the “formation of photo-oxidants” impact category can be regarded as not relevant to agricultural crop production systems. However, if this impact category is to be considered in an LCA study, a characterization method developed by Hauschild et al. (2000a) can be recommended, as this is the only method that considers the impact of both, NOx and VOC emissions.

The chosen aggregation methods are described separately for each impact category in the following sections.

3.1.1 Depletion of abiotic resources

The issue related to the depletion of abiotic resources, such as fossil fuels or minerals is their decreasing availability for future generations. In currently available LCIA methods (e.g. Goedkoop & Spriensma, 1999; Guinée, 2001), the consumption of different abiotic resources is aggregated to one summarizing indicator for resource depletion within the characterization step. However, these methods do not consider that many resources have different functions and are not equivalent to each other. By contrast, emissions are usually aggregated based on their "function", i.e. the effect on the environment (e.g. N_2O, CH_4, CO_2 -> climate change). Therefore, a new aggregation method has been developed to separate the consumption of different abiotic resources into different impact sub-categories and aggregate them to indicators according to the primary function of the resources (e.g. coal, natural gas, oil -> depletion of fossil fuels; Brentrup et al., 2002a). For LCA studies on agricultural crop production the consumption of fossil fuels and minerals such as phosphate, potash and lime are sub-categories of particular importance. Table 2 gives the characterization factors for abiotic resources typically consumed in an agricultural crop production system.

Table 2: "Depletion of abiotic resources" impact category

Characterization factors for the aggregation of single resources to resource depletion indicators for each impact sub-category

Resource	**Unit**	**CF** [A]	**Impact sub-category** (unit of resource depletion indicator, RDI)
Oil	kg OE [B]	42.868	Depletion of fossil fuels (RDI $_{\text{fossil fuels}}$ in MJ)
Natural gas	m^3	31.736	
Hard coal	kg	29.704	
Lignite	kg	8.506	
Phosphate rock	kg	0.25	Depletion of phosphate rock (RDI $_{\text{phosphate rock}}$ in kg P_2O_5)
Raw phosphate	kg	0.32	
Potash, potassium chloride	kg	0.105	Depletion of potash (RDI $_{\text{potash}}$ in kg K_2O)
Limestone/lime	kg	0.54	Depletion of lime (RDI $_{\text{lime}}$ in kg CaO)
Dolomite	kg	0.30	

[A] CF = characterization factor (heat values [in MJ per kg or m^3] for fossil fuels taken from BMWi, 1995; P_2O_5, K_2O, CaO contents [in kg per kg] taken from Patyk & Reinhardt, 1997 and www.dolomit.de)

[B] OE = crude oil equivalents

The characterization result (RDI = resource depletion indicators) for the sub-categories can be calculated according to equation (1) (see 2.3).

3.1.2 Land use

The "land use" impact category describes the environmental impacts of utilizing and reshaping land for human purposes (Heijungs et al., 1997; Lindeijer et al., 1998; Müller-Wenk, 1998b; Köllner, 2000). The environmental consequences of land use such as arable farming or urban settlement are the decreasing availability of habitats and the decreasing diversity of wildlife species (EEA, 1998a; BfN, 1999; Statistisches Bundesamt, 1999). Current approaches to assess the impact of human land use are mainly based on empirical investigations of species diversity (e.g. of vascular plants; Goedkoop & Spriensma, 1999; Köllner, 2000). There are two main problems associated with these approaches. First, it is difficult to determine a natural reference situation, to which the situation of land under use can be compared, because the diversity of species per area within Europe varies naturally by a factor of 10-15 (BfN, 1999). Furthermore, not only the number, but also the composition of species (e.g. share of neophytes) is important for the assessment of the influence of human land use (Kretschmer et al., 1997). Consequently, a new method for the assessment of land use impacts has been developed (Brentrup et al., 2002b).

This new method treats "natural land" like a resource and it is assumed that the utilization of land leads to a reduced availability of this resource. Natural land can be defined as the sum of actually uninfluenced area and the accumulated remaining naturalness of the land under use. To determine the remaining naturalness of land under use, the Hemeroby concept (Kowarik, 1999) can be applied. Hemeroby is a measure for the "human influence on ecosystems" and is therefore used to characterize the environmental impact of different land use types. The characterization factors are described as "naturalness degradation potential" (NDP) and given in Table 3 for selected land use types.

Table 3: *"Land use"* impact category

Characterization factors (NDP = Naturalness Degradation Potentials) for selected land use types

Land use type (in ha*year)	**NDP** [A]
Continuous urban area	0.95
Industrial/commercial units	0.95
Road/rail networks	0.90

Table 3 (continued)

Land use type (in ha*year)	**NDP** [A]
Discontinuous urban area	0.85
Intensive arable land	0.80
Extensive arable land	0.70
Green urban area	0.70
Intensive permanent pasture	0.60
Extensive permanent pasture	0.50

[A] NDP = Naturalness Degradation Potential

NDP values for additional land use types and a detailed description of this aggregation method can be found in Brentrup et al. (2002b).

On a larger geographical scale the "natural area/naturalness" resource is not homogeneous. Therefore the assessment of land use should be regionalized. Consequently, the impact category "land use" is subdivided into different sub-categories.

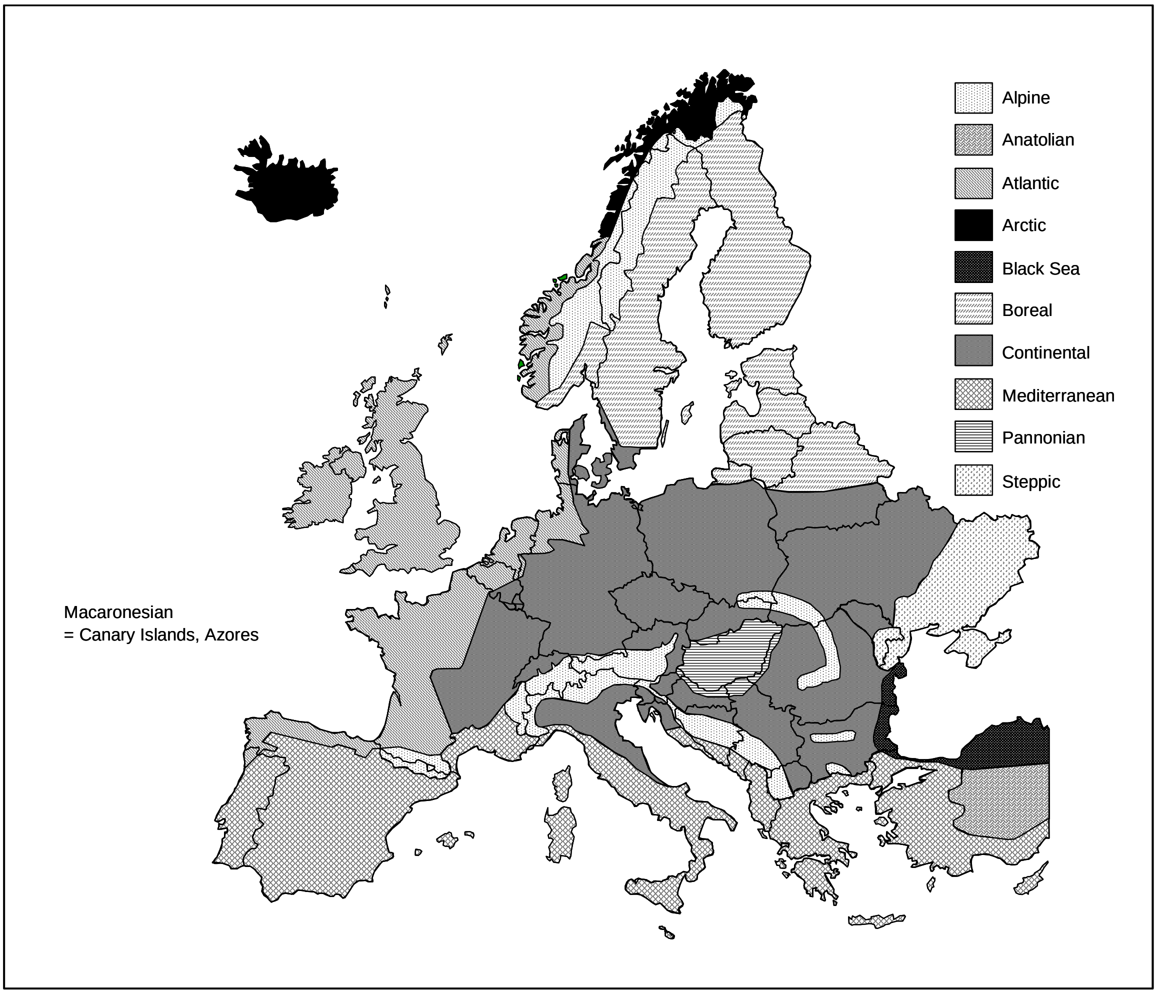

Figure 3: The biogeographic regions of Europe (adopted from EEA, 1998a)

The definition of these sub-categories should be based on ecologically homogenous land units as for example the biogeographic regions for Europe (EEA, 1998a). Figure 3 shows the biogeographic regions of Europe. The land use indicator (NDI = naturalness degradation indicator) for a system under investigation can be calculated according to equation (1) (see 2.3). If land is used in more than one biogeographic region, this calculation should be done separately for each biogeographic region.

3.1.3 Climate change

Emissions of gases with specific radiative characteristics like carbon dioxide (CO_2) and nitrous oxide (N_2O) lead to an unnatural warming of the Earth's surface, which in turn will cause global and regional climatic changes. This environmental impact is commonly described as "global warming". The term "climate change" indicates that the possible consequences of global warming concern more elements of the global climate than only the temperature (e.g. precipitation, wind). The main anthropogenic contributors to the enhanced greenhouse effect are (sorted according to their contribution): CO_2 (65%), methane (CH_4, 20%), halogenated gases (e.g. CFCs, 10%) and N_2O (5%; EEA, 1998a). The different potential of these emissions to contribute to climate change is represented by their Global Warming Potential (GWP, Table 4). The climate change indicator for a system under investigation can be calculated according to equation (1) (see 2.3).

Table 4: *"Climate change"* impact category

Characterization factors (= Global Warming Potentials) for selected greenhouse gases

Substance (in kg)	**Global warming potential** (GWP 100) [A] (in kg CO_2-equivalents per kg)
CO_2	1
CH_4	21
N_2O	310

[A] GWP 100 = Global warming potential for the time horizon of 100 years

3.1.4 Toxicity

This impact category includes all direct toxic effects of emissions on humans (human toxicity) and ecosystems (eco-toxicity). Emissions, which may be potentially toxic and are released by arable farming systems, are (1) *inorganic air pollutants* like NH_3, SO_2 and NOx, (2) *plant protection substances*, and (3) *heavy metals*.

Inorganic air emissions (e.g. SO_2, NOx, CO, NH_3, particles) are potentially toxic to humans due to their contribution to winter smog episodes with high concentrations of air pollutants in urban areas (Stanners & Bourdeau, 1995). The contribution of these emissions to other environmental problems like acidification or eutrophication is accounted for in the respective impact categories. Winter smog is associated with specific weather conditions together with high emission rates particularly of SO_2 and suspended particles, leading to respiratory problems. However, own investigations have shown that in arable farming systems at least 70% of the SO_2, NOx, NH_3, CO and particle emissions are released during on-field activities (e.g. tractor use, fertilizer application) in spring and summer. Also, because of the short atmospheric residence time of these substances, the exclusion of these emissions as far as arable farming systems are concerned from the "toxicity" impact category can be justified, because they are unlikely to contribute to the winter smog problem.

Plant protection substances are applied in order to control certain organisms (e.g. weeds, fungi, and insects) in order to improve the productivity of arable farming. However, via wind drift, evaporation, leaching, and surface run-off, a part of the applied agro-chemicals may impact upon terrestrial and aquatic ecosystems or even humans (Hauschild, 2000b). To estimate the rate of unintended emissions of toxic substances, their fate in the environment and the final effects on ecosystems and humans different models have been developed (Goedkoop & Spriensma, 1999; Guinée et al., 1996; Huijbregts, 2001; Jolliet & Crettaz, 1997). These models concentrate on toxic substances other than pesticides and do not include the currently available plant protection agents. Because of uncertainty of the database and because plant nutrition is the focus of this study, the possible toxic impacts of agro-chemicals have been excluded from this LCIA approach.

Other "non-toxic" environmental impacts, which are due to the production, packaging, transport and application of plant protection agents (e.g. consumption of fossil fuels, emissions related to energy use), are included in the relevant impact categories.

The agricultural use of mineral phosphate fertilizers and organic materials like slurry, sewage sludge or compost may lead to emissions of *heavy metals* to soils. The contamination of these materials with heavy metals varies substantially depending on the origin of the raw material (P rock, industrial and household waste). For heavy metal emissions to soil, models developed by Goedkoop & Spriensma (1999) for human toxicity and Huijbregts (2001) for eco-toxicity are most suitable for the estimation of their toxic potential. Both models take into account information on the environmental fate, the probable exposure of humans or ecosystems, and the potential toxic effects.

For *human toxicity* Goedkoop & Spriensma (1999) use the concept of DALYs (Disability-Adjusted Life Years), which was developed on behalf of WHO and World Bank (Murray, 1994) and adopted for LCA by Hofstetter (1998). In the DALY concept, weights for the different severity of human health effects have been established. These weights allow for comparisons between time lived with a certain limitation and time lost due to premature mortality. The human toxicity potential (HTP) for emissions of toxic substances is therefore expressed in DALYs.

For *eco-toxicity*, the eco-toxicity potential (ETP) has been calculated by Huijbregts (2001) for 5 different types of ecosystems: (1) terrestrial, (2) fresh water, (3) sea water, (4) fresh water sediment, and (5) sea water sediment. The ETP are expressed relative to a reference substance, which is 1,4-dichlorobenzene (1,4DCB) and are therefore called 1,4DCB-equivalents. The human and eco-toxicity potentials for cadmium emissions to soil are given in Table 5. Toxicity potentials for other toxic substances like other heavy metals or persistent organic pollutants can be found in Goedkoop & Spriensma (1999) and Huijbregts (2001). The toxicity potentials for a system under investigation can be calculated according to equation (1) (see p. 6).

Table 5: *"Human and eco-toxicity"* impact category

Characterization factors (Toxicity Potentials) for cadmium (Cd) emissions to soil

Sub-category	**Unit of sub-category indicator**	**Toxicity potential per kg Cd to soil**
Human toxicity	DALY [A]	3.98E-03
Terrestrial eco-toxicity	kg 1,4-DCB-equiv. [B]	1,7E+02
Aquatic eco-toxicity, fresh water	kg 1,4-DCB-equiv.	7,8E+02
Aquatic eco-toxicity, marine	kg 1,4-DCB-equiv.	1,1E+05
Sediment eco-toxicity, fresh water	kg 1,4-DCB-equiv.	2,0E+03
Sediment eco-toxicity, marine	kg 1,4-DCB-equiv.	1,1E+05

[A] DALY = Disability adjusted life-years (Goedkoop & Spriensma, 1999)
[B] 1,4-DCB-equiv. = 1,4-dichlorobenzene-equivalents (Huijbregts, 2001)

3.1.5 Acidification

Acidification is mainly caused by air emissions of sulfur dioxide (SO_2, share: 36% for EU15), nitrogen oxides (NOx, 33%) and ammonia (NH_3, 31%; EEA, 2001a). SO2 primarily originates from combustion of sulfur-containing coal and oil, NOx from combustion processes in motor vehicles, whereas NH_3 predominantly originates from animal husbandry (EEA, 1998a). SO_2, NOx and NH_3 are also released during arable crop production. In particular the use of organic and mineral fertilizers can result in important emissions of NH_3 due to volatilization during and after application of urea and ammonium-containing fertilizer (Brentrup et al., 2000).

Acid deposition has negative effects on terrestrial and aquatic ecosystems.

The effect of potentially acidifying emissions depends on the deposition pattern (fate) and the susceptibility of the receiving area to acidification (e.g. buffer capacity, $CaCO_3$-content). Comparing the acidification potential of NH_3 emissions (expressed in SO_2-equivalents) from Sweden and Greece illustrates this effect. Whereas 1kg NH_3 released in Greece results in only 0.13 kg SO_2-equivalents, the same emission released in Sweden has an acidification potential of 4.4kg SO_2-equivalents (Huijbregts, 2001). This difference is due to the different deposition pattern of the emission and to the different sensitivity of the receiving area (e.g. buffer capacity of soils and surface waters). This illustrates the importance of a site-specific characterization approach.

A method developed by Huijbregts (2001) includes this kind of information and has therefore been selected for this LCIA approach. As a result separate characterization factors for acidifying emissions released in different European countries are proposed. In addition Huijbregts (2001) calculated average characterization factors for Western, Eastern and total Europe, which should be used, if the source region of an emission in not known in more detail. Table 6 gives the CFs for SO_2, NOx and NH_3 emissions released in Western European countries, which can be used to calculate the acidification indicator for a system under analysis according to equation (1) (see 2.3).

Table 6: "*Acidification*" impact category

Regionalized characterization factors (Acidification Potentials) for SO_2, NOx and NH_3 emissions (Huijbregts, 2001; modified)

Emission source region	**Acidification Potential** (in kg SO_2–equivalents per kg emission)		
	SO_2	NOx	NH_3
Switzerland [A]	1.00	0.28	1.30
Austria	1.00	0.27	1.30
Belgium	1.00	0.49	1.00
Denmark	1.80	0.88	1.50
Finland	5.00	1.90	6.40
France	1.10	0.43	2.00
Germany	1.30	0.53	1.50
Greece	0.066	0.037	0.13
Ireland	0.57	0.34	0.79
Italy	0.46	0.13	0.59
Luxembourg	1.30	0.50	1.50
Netherlands	0.92	0.51	1.00

Table 6 (continued)

Emission source region	**Acidification Potential** (in kg SO_2–equivalents per kg emission)		
	SO_2	NOx	NH_3
Norway	3.80	1.20	6.00
Portugal	0.18	0.08	0.28
Spain	0.22	0.10	0.27
Sweden	3.80	1.30	4.40
United Kingdom	0.86	0.43	1.50
Western Europe, average	0.79	0.41	1.30
Eastern Europe, average	1.60	0.70	1.80
Europe, average	1.20	0.50	1.60

[A] All acidification potentials are calculated relative to the acidification potential of 1kg SO_2 released in Switzerland

3.1.6 Eutrophication

Eutrophication can be defined as an undesired increase in biomass production in aquatic and terrestrial ecosystems caused by high nutrient inputs, which result in a shift in species composition. In surface waters eutrophication is particularly serious because it can lead to algal blooms and the subsequent oxygen-consuming degradation processes, which finally may result in the death of the total aquatic biocoenosis (EEA, 1998a; Potting et al., 2000). Terrestrial vegetation (i.e. mainly higher plants) and aquatic plants (i.e. mainly algae) respond differently to an additional supply of nutrients. Therefore, in this LCIA approach the eutrophication impact category is separated into *terrestrial* and *aquatic eutrophication*.

Terrestrial eutrophication

Huijbregts (2001) developed a characterization method for terrestrial eutrophication that considers atmospheric pathways, deposition patterns and eutrophication effects of NOx and NH_3 emissions. Since for terrestrial ecosystems nitrogen is the major limiting nutrient, NOx and NH_3 depositions are the most important contributors to terrestrial eutrophication (Finnveden & Potting, 1999; Potting et al., 2000). Huijbregts (2001) calculated regionalized terrestrial eutrophication potentials (TEP) expressed in NOx-equivalents (Table 7). The terrestrial eutrophication potential for any system under evaluation can be calculated according to equation (1) (see 2.3).

Table 7: “*Terrestrial eutrophication*” impact sub-category

Regionalized characterization factors (Terrestrial Eutrophication Potentials) for NOx and NH_3 emissions (Huijbregts, 2001; modified)

Emission source region	**Terrestrial Eutrophication Potential** (in kg NOx–equivalents per kg emission)	
	NOx	NH_3
Switzerland [A]	1.00	5.00
Austria	0.89	4.20
Belgium	1.20	2.90
Denmark	1.60	2.50
Finland	3.50	11.50
France	1.30	6.40
Germany	1.50	4.60
Greece	0.27	1.50
Ireland	0.52	1.00
Italy	0.60	2.80
Luxembourg	1.40	4.40
Netherlands	1.10	2.30
Norway	1.60	6.20
Portugal	0.49	2.40
Spain	0.52	2.00
Sweden	2.10	5.70
United Kingdom	0.76	1.70
Western Europe, average	0.99	3.70
Eastern Europe, average	1.70	5.00
Europe, average	1.20	4.30

[A] All terrestrial eutrophication potentials are calculated relative to the terrestrial eutrophication potential of 1kg NOx released in Switzerland

Aquatic eutrophication

Important anthropogenic N and P emissions to surface waters are: (1) *deposition of airborne NOx and* NH_3 on surface waters from combustion processes and livestock farming, (2) *direct effluents of N and P* (point sources, e.g. municipalities, industries), and (3) *diffuse losses of N via leaching* (non-point sources, e.g. arable farming) (Klepper et al, 1995). In LCIA it should not be assumed that all of the nutrients initially released to air and soil actually reach surface waters.

Fate factors developed by Huijbregts & Seppälä (2000) enable the approximation of the fraction of *airborne NOx and* NH_3 *emissions* entering surface waters. The factors indicate which fraction of a NOx or NH_3 emission released in different European countries reaches marine ecosystems (Table 8). Freshwater systems are been considered, because they only have a small fraction of the total surface water area and are mainly limited by P and not by N.

Table 8: Regionalized fate factors for airborne NOx and NH_3 emissions to determine the fraction reaching marine surface waters (Huijbregts & Seppälä, 2000; modified) for the *“Aquatic eutrophication”* impact sub-category

Emission source region	**Fate factors**	
	NOx	NH_3
Austria	0.088	0.048
Belgium	0.240	0.230
Denmark	0.290	0.430
Finland	0.200	0.250
France	0.230	0.250
Germany	0.170	0.140
Greece	0.180	0.230
Ireland	0.470	0.460
Italy	0.190	0.210
Luxembourg	0.170	0.110
Netherlands	0.280	0.260
Norway	0.280	0.450
Portugal	0.160	0.230
Spain	0.170	0.160
Sweden	0.240	0.330
Switzerland	0.094	0.051
United Kingdom	0.390	0.430
Western Europe, average	0.250	0.240
Eastern Europe, average	0.089	0.072
Europe, average	0.210	0.160

For *direct effluents of N and P* into surface water it is assumed, that these nutrients are either directly available for eutrophication (P in fresh water systems) or will be transported to places where they potentially contribute to nutrient enrichment (N in sea water systems). Therefore the total N and P emission rates from point sources (e.g. from fertilizer production or wastewater treatment plants) are considered.

The main pathway for *diffuse N emissions* from soil to aquatic ecosystems is via nitrate (NO_3) leaching. Nitrate losses to groundwater via leaching are strongly dependent on agricultural management (e.g. fertilization rates, yields) as well as site-specific soil and climate conditions (e.g. soil texture, precipitation; Brentrup et al., 2000). Therefore, NO_3 leaching losses from soil to groundwater are highly variable and should be carefully estimated considering all relevant parameters determining the NO_3 content in the soil at the beginning of the leaching period in autumn (N inputs and outputs, site-specific soil and climate characteristics). Aggregation methods proposed for aquatic eutrophication by Potting et al. (2000) and Huijbregts (2001) ignore the strong dependency on the conditions of any agricultural production system under investigation. They suggest the application of fixed national factors for nutrient losses due to fertilizer and manure application. Since such a general procedure is inadequate for an LCA method tailored to arable farming systems, a site- and study-specific estimation of N losses is proposed in this LCA method. Methods, which enable an estimation of fertilizer N reaching groundwater as nitrate, are described in detail by Brentrup et al. (2000). According to Potting et al. (2000) the calculated NO_3 leaching rate should be further reduced by 30% assuming denitrification losses on the way from groundwater to the sea.

Subsequent to this fate analysis, the rates of the different N and P emissions (airborne emissions, effluents, diffuse losses) assumed to reach surface waters can be finally aggregated to a total aquatic eutrophication potential using characterization factors based on the typical nutrient ratio of a phytoplankton (Redfield ratio; Heijungs, 1992a). These factors are taken from Van Oers et al. (2001) and are given in Table 9. The characterization factors are calculated relative to the eutrophication potential of phosphate (PO_4).

Table 9: *"Aquatic eutrophication"* impact sub-category

Characterization factors (Aquatic Eutrophication Potentials) for N and P emissions

Substance (in kg)	**Aquatic Eutrophication Potential** (in kg PO_4-equivalents per kg emission)
N	0.42
NH_3	0.35
NH_4	0.33
NOx	0.13
NO_3	0.10
NO_3-N	0.42
P	3.06
P_2O_5	1.34
PO_4	1.00

The AEP for a system under analysis can be calculated according to (2).

$$\mathrm{AEP}_x = \left[\sum_i \mathrm{E}_{\mathrm{air,i,j,x}} \times \mathrm{FF}_{\mathrm{i,j}} \times \mathrm{AEP}_{\mathrm{i}}\right] + \left[\sum_i \mathrm{E}_{\mathrm{water,\ i,x}} \times \mathrm{AEP}_{\mathrm{i}}\right] + \left(\mathrm{E}_{\mathrm{soil->groundwater,\ NO_3},x} \times 0.3 \times \mathrm{AEP}_{\mathrm{NO_3}}\right) \qquad (2)$$

where: AEP_x = Aquatic eutrophication potential for system x [in kg PO_4-equivalents/FU]

$E_{air,\ i,\ j,\ x}$ = Air emission i released in region j due to the analysed system [in kg/FU]

$FF_{i.\ j}$ = Fate factor for air emission i released in region j (Table 10)

AEP_i = Aquatic eutrophication potential for emission i [in kg PO_4-equivalents/kg]
$E_{water,\ i,\ x}$ = Water emission i due to system x [in kg/FU]
$E_{soil \rightarrow groundwater,\ NO3,\ x}$ = Emission of nitrate from soil to groundwater after site- and study-specific fate analysis due to system x [in kg NO_3/FU]
AEP_{NO3} = Aquatic eutrophication potential for nitrate [in kg PO_4-equivalents/kg]

3.2 Normalization

Even after aggregation of the inventory data to impact categories (section 3.1) it is not possible to conclude on the relative importance of these values. A high indicator value may represent only a small contribution to the total environmental effect, whereas a several times smaller indicator value may represent an important contribution to the respective environmental effect. Thus, "the aim of the normalization of indicator results is to better understand the magnitude for each indicator result of the product system under study" (ISO, 2000). During the normalization the indicator results per functional unit (i.e. a tonne of grain) are related to the respective indicator results for a defined reference area according to (3).

$$N_i = \frac{I_i}{NV_i} \qquad (3)$$

where: N_i = Normalization result per functional unit for impact category i
I_i = Indicator value per functional unit for impact category i
NV_i = Indicator value for a reference situation (e.g. total Europe) for impact category i = Normalization value

The decision about which reference situation shall be used depends on the subsequent weighting procedure as well as on the availability of normalization data (Lindeijer, 1996). Table 10 gives the European normalization values suggested in this LCIA method and Table 11 gives the sources of data, information, and models used for their calculation. The European reference situation has been chosen as an example due to data availability and because the calculation of weighting factors, given in the following section, is also mainly based on environmental targets for Europe. The European normalization values suggested in this study have been calculated per person (Table 10), in order to keep the resulting numbers in a manageable range. Normalization values could be calculated for any reference region as long as reliable data for the different impact categories are available. Data for a worldwide normalization are currently only available as crude estimates based on extrapolations (Guinée, 1996; Van Oers et al., 2001).

Table 10: Normalization values (NV) for the different impact categories

(= Indicator values per person in Europe [A]; for data sources see Tab. 11)

Impact category	**Impact sub-category**	**Unit**	**Year** [B]	**NV** [C] **Europe**
Abiotic resources	Lime consumption	kg CaO		- [D]
	Phosphate consumption	kg P_2O_5	1999	7,66E+00
	Potash consumption	kg K_2O	1999	8,14E+00
	Fossil fuel consumption	MJ	1999	1,33E+05
Land use	Land use, Alpine region	ha*year	1998	1,22E+04
	Land use, Anatolian region	ha*year	1998	- D
	Land use, Arctic region	ha*year	1998	- D
	Land use, Atlantic region	ha*year	1998	1,79E+04
	Land use, Black sea region	ha*year	1998	1,57E+04
	Land use, Boreal region	ha*year	1998	1,58E+04
	Land use, Continental region	ha*year	1998	1,86E+04
	Land use, Macaronesian region	ha*year	1998	1,00E+04
	Land use, Mediterranean region	ha*year	1998	1,61E+04
	Land use, Pannonian region	ha*year	1998	2,09E+04
	Land use, Steppic region	ha*year	1998	2,11E+04
Climate change		kg CO_2-equiv.	1999	9,73E+03
Toxicity	Human toxicity	DALY	1995/99	7,50E-03
	Terrestrial eco-toxicity	kg 1,4 DCB-equiv.	1995/99	1,15E+02
	Freshwater aquatic eco-toxicity	kg 1,4 DCB-equiv.	1995/99	1,24E+03
	Marine aquatic eco-toxicity	kg 1,4 DCB-equiv.	1995/99	2,88E+05
	Freshwater sediment eco-tox.	kg 1,4 DCB-equiv.	1995/99	1,28E+03
	Marine sediment eco-toxicity	kg 1,4 DCB-equiv.	1995/99	2,65E+05
Acidification		kg SO_2-equiv.	1999	4,77E+01
Eutrophication	Terrestrial eutrophication	kg NOx-equiv.	1999	6,07E+01
	Aquatic eutrophication	kg PO_4-equiv.	1999	8,56E+00

[A] European indicator values were divided by European population figure for 1994 (727 Mio; EEA, 1998b) in order to keep resulting numbers manageable
[B] Most recent data available were chosen
[C] NV = Normalization value for Europe (Europe = Albania, Austria, Belarus, Belgium, Bosnia and Herzegovina, Bulgaria, Croatia, Czech Republic, Denmark, Estonia, Finland, France, FYR of Macedonia, Germany, Greece, Hungary, Iceland, Ireland, Italy, Latvia, Lithuania, Luxembourg, Moldova, Netherlands, Norway, Poland, Portugal, Romania, Russian Federation, Slovakia, Slovenia, Spain, Sweden, Switzerland, Ukraine, United Kingdom, Yugoslavia)
[D] No data available

Table 11: Sources of data, information, and models used for the calculation of normalization values

Impact category	**Source**
Abiotic resources	Brentrup et al. (2002a), EIA (2001), FAO (2001), EFMA (2000), USGS (2001)
Land use	Brentrup et al. (2002b), Satellus (1999)
Climate change	EEA (2000, 2001b), UN-ECE/EMEP (2001), Houghton et al. (1993), UN-FCCC (2000)
Human toxicity	UN-ECE/EMEP (2001), UN-ECE/EMEP/MSC-E (2001), Goedkoop & Spriensma (1999)

Table 11 (continued)

Impact category	Source
Eco-toxicity	UN-ECE/EMEP (2001), UN-ECE/EMEP/MSC-E (2001), Huijbregts (2001), Van Oers et al. (2001)
Acidification	UN-ECE/EMEP (2001), Huijbregts (2001)
Eutrophication	UN-ECE/EMEP (2001), Eurostat (2000), FAO (2001), Huijbregts (2001), Hydro Agri (1993), IFA (2001), Klepper et al. (1995), LWK Westfalen-Lippe (1996), UBA (1997), Hambüchen (1999a, b)

3.3 Weighting

The weighting step is necessary to conclude on the overall environmental preference of one or the other products or processes under investigation. Weighting means an evaluation of the different effects such as global warming or acidification according to their potential to harm the environment. In LCA the so-called safeguard subjects "human health, natural ecosystems, and resources" represent the environment (Consoli et al., 1993; Lindfors et al., 1995). Weighting allows the further interpretation of complex environmental profiles in order to support users of LCA studies with clear and aggregated results. Weighting factors represents the environmental weight of each impact category. The higher the weighting factor for an impact category, the higher is the potential of that impact category to harm the environment. For this LCA method weighting factors were derived by using authorized environmental goals like the Kyoto protocol for climate change in the so-called "distance-to-target" principle (Müller-Wenk, 1996; Lindeijer, 1996). "Distance-to-target" means a comparison of the current level of an environmental effect in a certain region and time to a target level of the same effect. The ratio between both values gives the weighting factor for the environmental effect (equation 4).

$$WF_{i,j,k} = \frac{CI_{i,j,k}}{TI_{i,j,k}} \qquad (4)$$

where: $WF_{i,j,k}$ = Weighting factor for impact category i, valid for region j and year k
$CI_{i,j,k}$ = Current indicator value for impact category i for region j and year k
$TI_{i,j,k}$ = Target indicator value for impact category i for region j and year k

The selection of accepted environmental targets makes it possible to consider that the evaluation of environmental problems may differ substantially between regions and societies. Accepted environmental goals implicitly include a range of criteria for the evaluation of environmental impacts like the magnitude, the reversibility, and the geographical extent of the ecological damage, the uncertainty of the damage and the substitutability of the damaged item

(Müller-Wenk, 1996). Furthermore, accepted environmental goals consider not only the environmental but also economic and social aspects of the respective impacts (e.g. Kyoto protocol, UN-FCCC, 1998). The application of such authorized targets can be regarded as the most objective and justified way to include value judgments within the weighting of different environmental impacts. Therefore, the distance-to-target principle with accepted environmental goals as targets for the calculation of weighting factors is suggested for this LCA method. Table 12 gives the weighting factors for the environmental effects and their sub-categories.

Table 12: Weighting factors for the impact categories and data used for the calculation (current status, target value)

Impact category Impact sub-category	**Indicator value (for unit see Table 10)**		**Basis for target**	**Weighting factor**
	current status (year)	**target value**		
Abiotic resource depletion				
Lime consumption	1.16E+11 (1999)	[a]		0.00
Phosphate consumption	4.32E+10 (1999)	3.60E+10	100 years availability [b]	1.20
Potash consumption	2.56E+10 (1999)	8.40E+10		0.00
Fossil fuel consumption	3.25E+14 (1999)	3.08E+14		1.05
Land use				
Land use, Alpine region	8,84E+06 (1998)	8,84E+06		
Land use, Atlantic region	4,34E+07 (1998)	4,34E+07		
Land use, Black sea region	5,62E+05 (1998)	5,62E+05		
Land use, Boreal region	8,77E+06 (1998)	8,77E+06		
Land use, Continental region	7,71E+07 (1998)	7,71E+07	maintenance of current land use intensity in Europe [c]	1.00
Land use, Macaronesian region	2,30E+05 (1998)	2,30E+05		
Land use, Mediterranean region	4,40E+07 (1998)	4,40E+07		
Land use, Pannonian region	7,22E+06 (1998)	7,22E+06		
Land use, Steppic region	2,00E+06 (1998)	2,00E+06		
Climate change	3.50E+06 (1998)	3.32E+06	UN-FCCC (1998) [d]	1.06
Toxicity				
Human toxicity, eco-toxicity			no target defined [e]	-
Acidification	1.42E+04 (1999)	1.06E+04	UN-ECE/CLRTP (1999) [d]	1.34
Eutrophication				
Terrestrial eutrophication	2.46E+04 (1999)	1.95E+04	UN-ECE/CLRTP (1999) [d]	1.26
Aquatic eutrophication	1.10E+06 (1995)	8.07E+05	OSPAR (1995), HELCOM (2001) [f]	1.37

[a] no problem expected due to very large lime reserves (USGS, 2001)
[b] target based on assumption that an availability of the resource for at least 100 years is sufficient for the development of substitution or recycling techniques (Brentrup et al., 2002a)
[c] target based on the assumption that the current land use intensity in each biogeographic region of Europe is tolerable and should be maintained (Brentrup et al., 2002b)
[d] values based on emission rates and reduction targets for Western European countries
[e] only separate targets for specific groups of toxic substances (e.g. heavy metals to air), but no overall international target for the reduction of toxic emissions available
[f] values based on emission rates and reduction targets for Western European signatory states of OSPAR and HELCOM conventions

Due to the fact that emissions of numerous substances to air, water and soil contribute to the "human toxicity" and "eco-toxicity" impact categories, no overall international targets for human- or eco-toxicity could be found. Consequently, toxicity is only considered within characterization and normalization, but excluded from the weighting step.

However, in contrast to other LCA methods (e.g. Goedkoop, 1995; Goedkoop & Spriensma, 1999; Steen, 1999), this approach suggests two separate final indicators for (a) resource depletion and (b) impacts on natural ecosystems and human health. This separation is important, because the problems related to the depletion of abiotic resources are substantially different to those related to the other impact categories. The impact categories other than the depletion of abiotic resources have direct effects on either natural ecosystems (land use, acidification, eutrophication, eco-toxicity), or human health (human toxicity), or even on both (climate change). In contrast, the depletion of abiotic resources, i.e. the decreasing availability of raw materials for future generations has no direct impact on human health or the shape of natural ecosystems. The environmental impacts associated with the extraction and processing of resources (e.g. land use, emissions or effluents) are considered in the respective impact categories. Resources such as fossil fuels or phosphate rock rather have an intrinsic value for humans, as they substantially contribute to development and wealth creation (e.g. through mobility and nutrition). Therefore, the availability of abiotic resources for future generations is more an economic and social issue than an environmental problem. This is the reason why this LCA method separates the aggregated resource depletion indicator (RDI) from the aggregated environmental indicator (EcoX). The aggregation of these two indicators would necessitate the calculation of a kind of a sustainability indicator. The development of a sustainability indicator would be an ambitious goal for further research, but should certainly comprise more economic and social aspects besides resource depletion (e.g. income, employment, prices, food security and quality, rural development etc.).

The environmental index “EcoX” can be calculated for a specific product or system under examination by multiplying the normalization result for each impact category by the respective weighting factor and summing up the weighted results (Equation 5).

$$EcoX = \sum_i N_i \times WF_i \qquad (5)$$

where: EcoX = Environmental index per functional unit

N_i = Normalization result per functional unit for impact category i

WF_i = Weighting factor for impact category i

The aggregation of the normalized indicator values for the different resource categories (e.g. P rock, fossil fuels) into the summarizing resource depletion index “RDI” can be performed equivalently (Equation 6).

$$RDI = \sum_i N_i \times WF_i \tag{6}$$

where: RDI = Resource depletion index per functional unit
N_i = Normalization result per functional unit for impact category i
WF_i = Weighting factor for impact category i

4. Discussion and conclusions

The LCA method described in this paper is based on the general LCA methodology given by ISO (1997) and SETAC (Consoli et al., 1993) and adapted to the study of plant nutrition in crop production systems. Different ready-to-use LCA approaches, primarily designed for industrial applications, have been published (BUWAL, 1998; Goedkoop, 1995; Goedkoop & Spriensma, 1999; Guinée et al, 2001; Heijungs, 1992a; Steen, 1999). It is therefore not surprising that there are problems concerning their application to agricultural crop production systems (Brentrup et al., 2001). In particular, some important environmental impacts are not included (e.g. land use, resource depletion in Goedkoop, 1995, nutrient emissions in Goedkoop & Spriensma, 1999). Some methods do not use state-of-the-art aggregation procedures (e.g. for acidification in Goedkoop, 1995). Others are not transparent in their weighting procedure (Steen, 1999), do not calculate an overall environmental indicator (Guinée et al, 2001; Heijungs, 1992a) or may be valid for only one country (BUWAL, 1998, for Switzerland).

Therefore, in this paper a new LCA approach is described that has been developed to study the environmental impact of arable crop production. A major advantage of this approach is the integration of all impact categories relevant to agricultural crop production. New impact assessment procedures, including aggregation, normalization and weighting, have been developed for “land use” and “resource consumption” impact categories (Brentrup et al., 2002a, b). For the “climate change”, ”toxicity”, ”acidification” and ”eutrophication” impact categories, the currently best available aggregation methods have been chosen and refined to include new normalization values and weighting factors. The “depletion of the stratospheric ozone layer” and “formation of tropospheric photo-oxidants” impact categories have been shown to be unimportant for agricultural crop production systems because usually no (stratospheric ozone depletion) or only negligible emissions (photo-oxidants) are released from crop production. With regard to the “toxicity” impact category only heavy metal emissions to soil resulting from the application of contaminated organic and inorganic fertilizers are considered. Possible toxic effects of plant protection substances on natural ecosystems and humans are not considered because plant nutrition is the focus of this study.

However, the proposed LCA method enables a comprehensive analysis of all other environmental impacts related to arable farming products. This is currently of particular interest because of an increasing public awareness and interest in the environmental effects of food production. The life-cycle perspective in LCA studies of arable farming products like wheat grain, i.e. the consideration of sub-processes such as raw material extraction or fertilizer production together with the on-farm processes allows the detection of environmental hot-spots in the total production system. For example, own investigations on the environmental impact of wheat production have shown that the main environmental impact of the production system is related to on-field activities (e.g. fertilizer application), whereas the production and transport of farm inputs has a much smaller effect (Küsters & Brentrup, 1999). Other environmental problems, which depend on nitrogen (N) fertilizer management, can be eutrophication (if N application rates exceed the crop demand), acidification (if urea or ammonium-containing fertilizers are used) or climate change. The application of this LCA method provides an insight into the contribution of different sub-systems, e.g. the transport, production and application of farm inputs to the environmental impact and enables the suggestion of measures to improve the overall environmental performance of arable farming systems. Furthermore, the proposed method can be used to support the choice of alternative products or processes to reduce environmental effects.

To this end, the weighting step should be seen as a valuable and objective interpretation tool, which prevents LCA users from deriving their own subjective conclusions on the overall environmental preferences of different alternatives. However, the main challenge with weighting is that the evaluation of environmental impacts on humans, ecosystems and resources is not only a matter of natural science. Natural science is necessary to describe and to quantify the single effects and their impact on the environment during the impact assessment (e.g. the different potential of nutrients to contribute to the eutrophication of aquatic and terrestrial ecosystems). The challenge of the final evaluation of the impacts on the different environmental compartments (fauna, flora, humans, resources) is the integration of natural science with subjective values and therefore needs consensus of the society. Furthermore, weighting procedures are expected to be transparent (Lindeijer, 1996). A set of generic weighting factors, based on the *distance-to-target principle* with accepted environmental goals as targets, fulfills these requirements. Environmental goals, such as the UNECE emissions reduction targets, to reduce acidification, eutrophication and photo-oxidant formation (UN-ECE/CLRTAP, 1999) are a result of an intensive debate between science, society's economic goals, and policy guidelines and reflect the society's view on these environmental problems.

Other possible methods to derive weighting factors such as *expert panels*, *proxy approaches* or *monetary methods* reveal some specific problems. For example, weighting factors based on *expert panels* (e.g. Landbank, 1994; Goedkoop & Spriensma, 1999) can be influenced by the personal priorities and perceptions of the chosen panel members or by the way they are interviewed (Landbank, 1994). Furthermore, the panel members may be not representative of all social groups (Goedkoop & Spriensma, 1999). Another option to derive weighting factors are *proxy approaches*, which use for instance the accumulated energy consumption or the total material input (Giegrich et al., 1995) as a representative for the total environmental impact of a product or process under investigation. However, inputs (resources) need not to be representative for outputs (emissions), i.e. the output of toxic substances does not necessarily need high inputs. Therefore, this kind of weighting does not comply with the goal of LCA, since only part of the total environmental impact of a product or process is considered. In *monetary methods* (e.g. Steen, 1999) cash values are assigned to environmental impacts by applying for example market prices for resources like energy or willingness-to-pay surveys for externalities like decreasing biodiversity. Such real and virtual market prices together with costs for the technical avoidance or mitigation of environmental impacts (acidification, toxicity) result in a common unit for all impacts (money). Provided that the varying methods to evaluate the different environmental impacts lead to equivalent weighting and that the monetary values represent more than only the economic aspect of the environmental impacts, a monetary weighting approach would be possible.

In a following publication (Brentrup et al., 2002d) the proposed LCA method will be used to assess the environmental effects of different intensities of winter wheat production.

5. Acknowledgements

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In the previous chapter, a new LCA method was developed, which is specifically tailored to the investigation of arable crop production systems. The following chapter describes the application of this LCA method on winter wheat production at different N fertilizer rates.

VII. Investigation of the Environmental Impact of Agricultural Crop Production using the Life Cycle Assessment (LCA) Methodology

II. Application of the LCA methodology to investigate the environmental impact of different N fertilizer rates in cereal production

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Abstract

This study examined the environmental impact of different nitrogen (N) fertilizer rates in winter wheat production by using a new Life Cycle Assessment (LCA) method, which was specifically tailored to agricultural crop production. The wheat production system studied was designed according to “good agricultural practice”. Information on crop yield response to different N rates was taken from a long-term field trial in the UK (Broadbalk Experiment, Rothamsted). The analysis considered the entire system, which was required to produce one tonne of wheat grain. It included the extraction of raw materials (e.g. fossil fuels, minerals), the production and transportation of farming inputs (e.g. fertilizers) and all agricultural operations in the field (e.g. tillage, harvest).

In a first step, all emissions and the consumption of resources connected to the different processes were listed in a Life Cycle Inventory and related to a common unit, which is one tonne of grain. Next a Life Cycle Impact Assessment was done, in which the inventory data are aggregated into indicators for environmental effects, which included *resource depletion, land use*, *climate change*, *toxicity*, *acidification*, and *eutrophication*. After normalization and weighting of the indicator values it was possible to calculate summarizing indicators for resource depletion and environmental impacts (EcoX).

At N rates of 48, 96, 144 or 192 kg N/ha the environmental indicator “EcoX” showed similar values per tonne of grain (0.16-0.22 EcoX/tonne of grain). At N rates of 0, 240 or 288 kg N/ha the EcoX values were 100% to 232% higher compared to the lowest figure at an N rate of 96 kg N/ha. At very low N rates, land use was the major environmental problem, whereas at high N rates eutrophication was the major problem. The results revealed that economic optimal arable farming does not necessarily come into conflict with economic and environmental considerations.

1. Introduction

Farming is expected to comply with the principles of sustainability (Commission of the European Communities, 1999; UN-Division for Sustainable Development, 2000), which include providing sufficient food of high quality at affordable prices produced with minimum environmental impact. However, arable farming traditionally targets the economic optimum production intensity, which usually only partly considers environmental aspects.

This study examines the eco-efficiency of cereal production at economic optimum nitrogen (N) fertilization in comparison to other N fertilizer application rates. *Eco-efficiency* can be defined as “creating more value with less [environmental] impact” (WBCSD, 2000). Thus, to measure the eco-efficiency of crop production, it is necessary to consider both environmental impacts connected to arable farming and at the same time the value created, i.e. the crop yield achieved or other production parameters like crop quality aspects.

Life Cycle Assessment (LCA) is an appropriate methodology to investigate the eco-efficiency of products and production systems, because it relates the different environmental impacts associated to a production system to the value produced. The main value produced by arable farming is the crop yield. Therefore, in this case the environmental impact has to be related to the production of a unit of yield (e.g. one tonne of grain).

However, most applications of LCA are to be found in the manufacturing industry, see for example studies on alternative packaging materials (BUWAL, 1991; UBA, 1995) or recycling options (IFEU, 1999), whereas LCA studies on arable farming systems are rare (Audsley et al., 1997; Küsters & Jenssen, 1998). Therefore, the existing LCA methodology needed adjustments to agriculture. Consequently, a new LCA method tailored to the specifics of crop production has been developed with a special focus on plant nutrition (Brentrup et al., 2002c).

The objective of this paper is the application of the new LCA method to the environmental impacts of different N fertilizer rates in cereal production.

The basis of this LCA study is information on the impact of different N fertilization rates on the productivity of arable farming. The effect of N application rate on cereal yields has been investigated in numerous ad hoc field trials. However, reliable conclusions on the impact of different N rates on yield can only be drawn from long-term experiments (Steiner, 1995). Therefore, the yield response to different N rates determined in the well documented Broadbalk Wheat Experiment (Rothamsted, UK) has been chosen in order to represent the productivity of different N application rates in wheat production under Western European conditions. A detailed description of the Broadbalk field trial and the soil and climatic conditions at Rothamsted Experimental Station is given in Johnston (1994). All other aspects of the wheat production system investigated in this study (e.g. P, K, Mg fertilization, plant protection, use of agricultural machinery etc.) have been defined according to good agricultural practice in Western Europe (MAFF, 1998; BMU, 1998, 2002).

2. Materials and methods

2.1 Description of the wheat production system analyzed in this LCA study

The wheat production system analyzed in this study is a theoretical system based around the long established Broadbalk field experiment at Rothamsted (England) and complying with the codes of good agricultural practice (e.g. MAFF, 1998; BBodSchG, 1998; BNatSchG, 2002). Only N rates, yields and nutrient content of grain and straw were directly taken from the Broadbalk field experiment. The annual application of phosphate (P), potash (K) and magnesium (Mg) fertilizer is assumed, for the purposes of this study, to be according to crop removal (i.e. grain and straw) (Table 2). Plant protection agents were applied as necessary. With regard to the use of agricultural machines (e.g. tractors, combine harvester) average technical equipment and times of use needed for the cultivation of a 5ha field are assumed (e.g. KTBL, 1998). Table 1 shows all agricultural operations considered.

Table 1: Agricultural operations included in the analyzed wheat production system

Agricultural operation	**Timing, Machinery**	**Additional information**
Fertilizer application: P as Triple Super Phosphate (TSP, 46% P_2O_5), K as Potassium Sulfate (50% K_2O), Mg as Kieserite (26% MgO)	October, Fertilizer spreader, Tractor (60 kW)	TSP from El Jorf-Lasfar, Morocco; Potassium Sulphate and Kieserite from Hattorf, Germany; Spread as bulk blend
Soil preparation: Ploughing	October, Plow (5 shares), Tractor (83 kW)	
Sowing: Seedbed preparation, drilling	October, Seedbed combination, drill, Tractor (105 kW)	Variety Hereward at 380 seeds/m^2
Fertilizer application: N as Ammonium Nitrate (AN, 33.5% N)	March/April, Fertilizer spreader, Tractor (60 kW)	Applied in one dressing, broadcast; AN from Sluiskil, Netherlands
Plant protection: 1st herbicide application	May, Sprayer, tractor (60 kW)	Topik at 250 ml/ha, Starane at 1 l/ha, Ally at 40 g/ha
Plant protection: 1st fungicide application	May/June, Sprayer, tractor (60 kW)	Opus Top at 0.5 l/ha (N0-N2), at 0.7 l/ha (N3-N6)

Table 1 (continued)

Agricultural operation	**Timing, Machinery**	**Additional information**
Plant protection: 2nd herbicide application	June/July, Sprayer, tractor (60 kW)	Mutiny at 2.4 l/ha
Plant protection: 2nd fungicide application (only in N3-N6)	June/July, Together with 2nd herbicide application	Folicur at 0.5 l/ha (N3-N6)
Harvest: Combine harvesting, bale pressing	August, Combine (95 kW), baling press, tractor (60 kW)	No drying included; straw baled and removed

2.2 The Broadbalk Wheat Experiment

Broadbalk is the oldest continuously running field experiment in the world having been set up by John Bennet Lawes in 1843 (Goulding et al., 2000). The experiment compares the effects of different fertilizer treatments on the yield of winter wheat. In 1978, a 5-course rotation was introduced (fallow, potatoes, 3 x winter wheat), which was modified in 1997 (winter oats, forage maize, 3 x winter wheat). In this LCA study, the average yield response of the 1st wheat in a rotation to increasing N fertilizer rates in the years 1996 to 2000 has been chosen to determine the productivity of the different N application rates. Table 1 shows the N fertilizer rates, average yields and nutrient removals for the wheat plots selected for the study.

Table 2: Fertilizer rates, yields and nutrient removal in the Broadbalk treatments

Plot no.	**N rate (kg/ha)**	**Yield (t/ha, 85% DM)** [a]		**Nutrient removal** [b] **(grain + straw, kg/ha)**			
		Grain	Straw	N	P_2O_5	K_2O	MgO
N0	0	2.07	0.94	30	15	19	4
N1	48	4.81	2.55	64	36	48	9
N2	96	7.11	3.66	107	53	70	13
N3	144	8.53	4.35	147	63	83	15
N4	192	9.25	4.72	177	69	90	16
N5	240	9.27	4.96	196	69	93	17
N6	288	9.11	5.28	212	69	95	17

[a] Mean yield (1996-2000) for 1st wheat in the rotation (after potato or maize)

[b] nutrient contents of grain and straw according to Poulton (2002, pers. comm.)

Figure 1 shows the average yield response to fertilizer N. Up to N4 (192 kg N/ha), a strong increase in grain yield from 2.1 to 9.3 t/ha can be observed, whereas at N5 and N6 (240 and 288 kg N/ha, respectively) no further yield increase occurred. The economic optimum N fertilizer rate (Nopt) is 210 kg N/ha (Figure 1). Nopt was calculated from a fitted quadratic response function and current prices for N (0.57 Euro/kg N) and wheat grain (115 Euro/tonne). N4 (192 kg N/ha) is closest to Nopt and can therefore be regarded as the optimum treatment in this trial for the years 1996-2000. In this paper, the production intensity of N4 will be used as the reference treatment, to which the environmental impacts of the other treatments are compared.

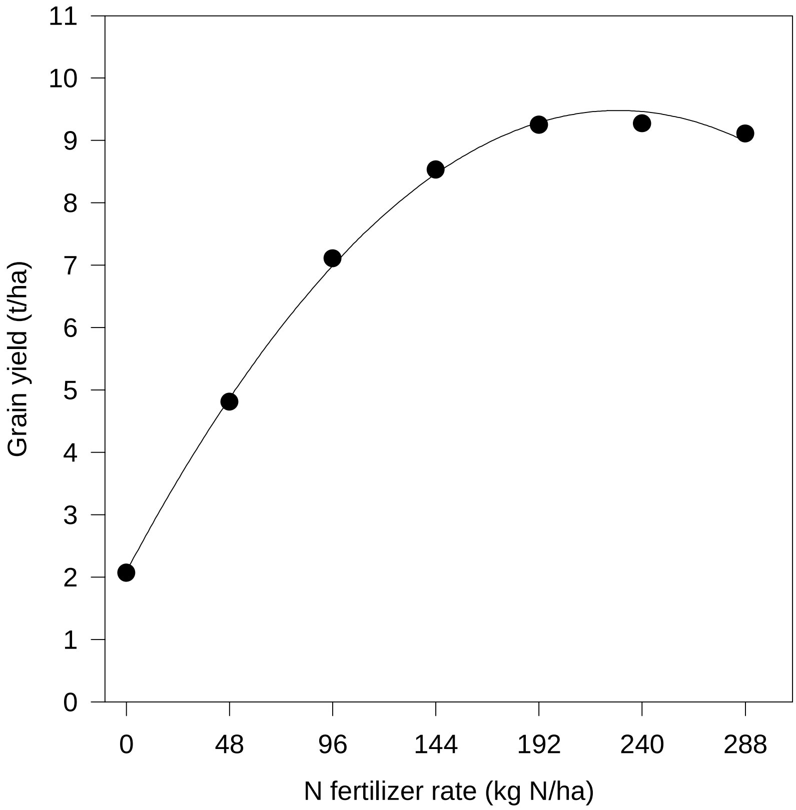

Figure 1: Yield response of winter wheat to increasing N fertilizer rates (average for 1st wheat in rotation in 1996-2000)

2.3 Life Cycle Assessment

According to ISO (ISO, 1997) LCA is divided into four steps, which are *goal and scope definition*, *inventory analysis*, *impact assessment* and *interpretation*.

The first step in LCA is the *goal and scope definition*. Within this phase the system under investigation, its function, and boundaries are described. The system investigated in this study is an arable farming system with the main function to produce winter wheat. Figure 2 shows the system needed to produce winter wheat as considered in this LCA study.

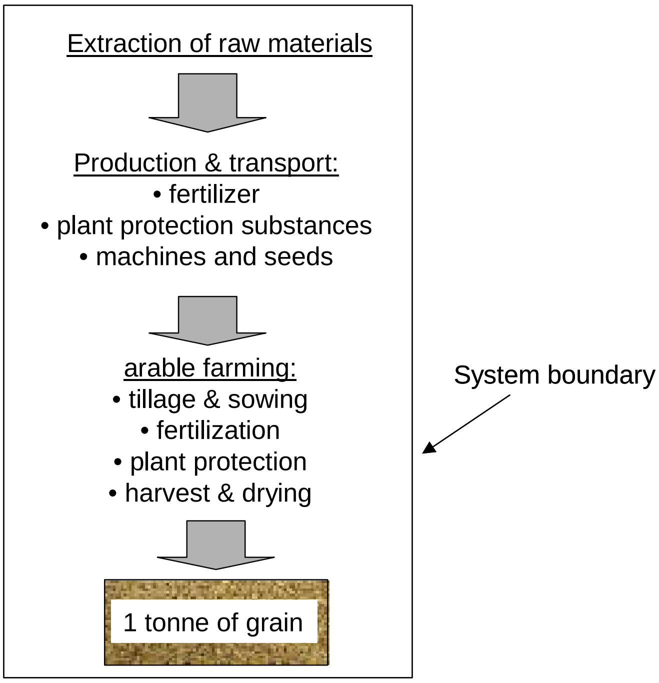

Figure 2: The wheat production system

In the subsequent *Life Cycle Inventory* (LCI) the resource consumption (inputs) and emissions (outputs) connected to the system are compiled (ISO, 1998a). To make the various inputs and outputs comparable, it is necessary to relate these data to a common functional unit, which shall represent the main function of the system (ISO, 1998a). Therefore, this study relates all resource consumption and emissions to one tonne of grain. Table 3 depicts all resources and emissions considered for the system and the data sources.

Table 3: Data sources for resource consumption and emissions related to the different sub-systems

Resources / emissions	**Sub-system**	**Data source**
Fossil fuels (oil, natural gas, hard coal, lignite)	Fertilizer production (process gas and fuel)	Hydro Agri (2002, pers. comm.), Davis & Haglund (1999), Kongshaug (1998), Patyk & Reinhardt (1997)
	Transportation	ETH Zürich (1994)
	Farm machinery, production	Grosse (1984)
	Farm machinery, repair	Haas & Köpke (1995)
	Farm machinery, use	KTBL (1998)
	Seeds, production	Oheimb et al. (1987)
	Plant protection agents, production	Oheimb et al. (1987), Gaillard et al. (1998)
Minerals (phosphate rock, potash)	P and K fertilizer production	Patyk & Reinhardt (1997)

Table 3 (continued)

Resources / emissions	**Sub-system**	**Data source**
Land	Arable farming	Rothamsted data
Cd	P fertilizer application	Roberts & Stauffer (1996)
CH_4, CO_2, CO, NOx, particles, SO_2, NMVOC	all sub-systems (emissions due to energy consumption in all sub-systems)	Hydro Agri (2002, pers. comm.), ETH Zürich (1994), Kongshaug (1998), Patyk & Reinhardt (1997)
Ntot	Fertilizer production (effluents)	Hydro Agri (2002, pers. comm.)
NH_3	Fertilizer production	Hydro Agri (2002, pers. comm.)
	Arable farming (volatilization)	ECETOC (1994)
N_2O	Fertilizer production (nitric acid production)	Hydro Agri (2002, pers. comm.)
	Arable farming (denitrification/nitrification)	Bouwman (1995)
NO_3-N	Arable farming (leaching)	DBG (1992), Brentrup et al. (2000)
Ptot	P fertilizer production (effluents)	Kongshaug (1998)

The data of the Life Cycle Inventory *per se* do not allow comparisons to be made between different systems. Furthermore, the potential environmental impact of the various emissions and resource consumption is not considered in this phase. Therefore, in the third step, a *Life Cycle Impact Assessment* (LCIA) must be made (ISO, 2000) in order to evaluate the inventory data. Within the LCIA, the different inputs and outputs are summarized into environmental effects, the so-called *impact categories*. Table 4 gives the list of impact categories relevant to the wheat production system.

Table 4: Environmental impact categories considered in the LCA on wheat production at different N application rates

Impact category
Depletion of abiotic resources (fossil fuels, phosphate rock, potash)
Land use
Climate change (= Global warming)
Toxicity (human toxicity and eco-toxicity)
Acidification
Eutrophication (terrestrial and aquatic)

The first step in LCIA is the *characterization* step. During characterization the inventory data are aggregated into indicators for each impact category (Table 4). For instance for the impact category “climate change”, CO_2, N_2O and CH_4 emissions are aggregated into the impact category indicator "CO_2-equivalents". Characterization is achieved through the use of *characterization factors*, which represent the potential of each emission to contribute to a specific environmental effect.

During the second step in LCIA, which is *normalization*, each of the indicator values is divided by a reference value, as for instance the respective indicator values per person in Europe (e.g. kg CO_2-equiv. per tonne of grain / CO_2-equiv. per capita in Europe). This normalization is performed in order to get information on the relevance of a product's impact in comparison to a reference value. Furthermore, with normalization, the indicator values become dimensionless, which is a prerequisite for the final weighting step. The third step or *weighting* step aims at a final aggregation across all impact categories to one overall environmental indicator. Therefore, each normalized indicator value is multiplied by a weighting factor, which represents the potential of the respective impact category to harm natural ecosystems, human health, and resources in Europe. Figure 3 shows the general LCIA procedure.

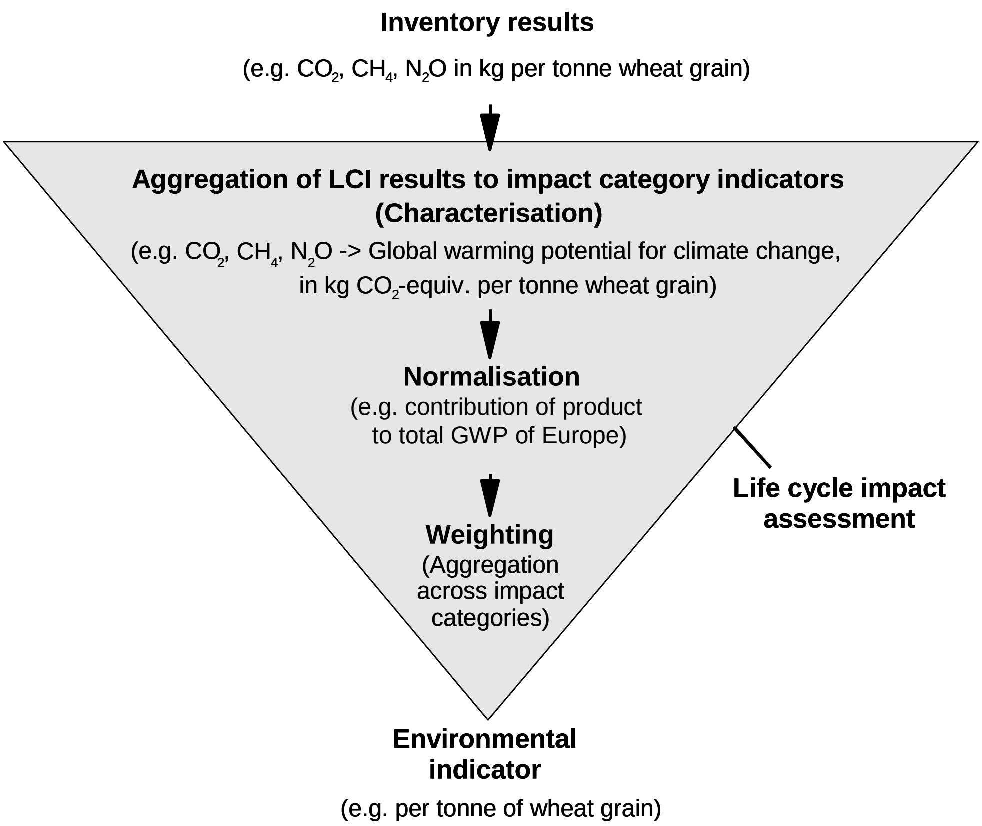

Figure 3: The general Life Cycle Impact Assessment procedure

3. Results and discussion

3.1 Life Cycle Inventory

3.1.1 Selected resource inputs

Figure 4 gives the consumption rates for selected resources per tonne of wheat grain for the different N fertilizer rates.

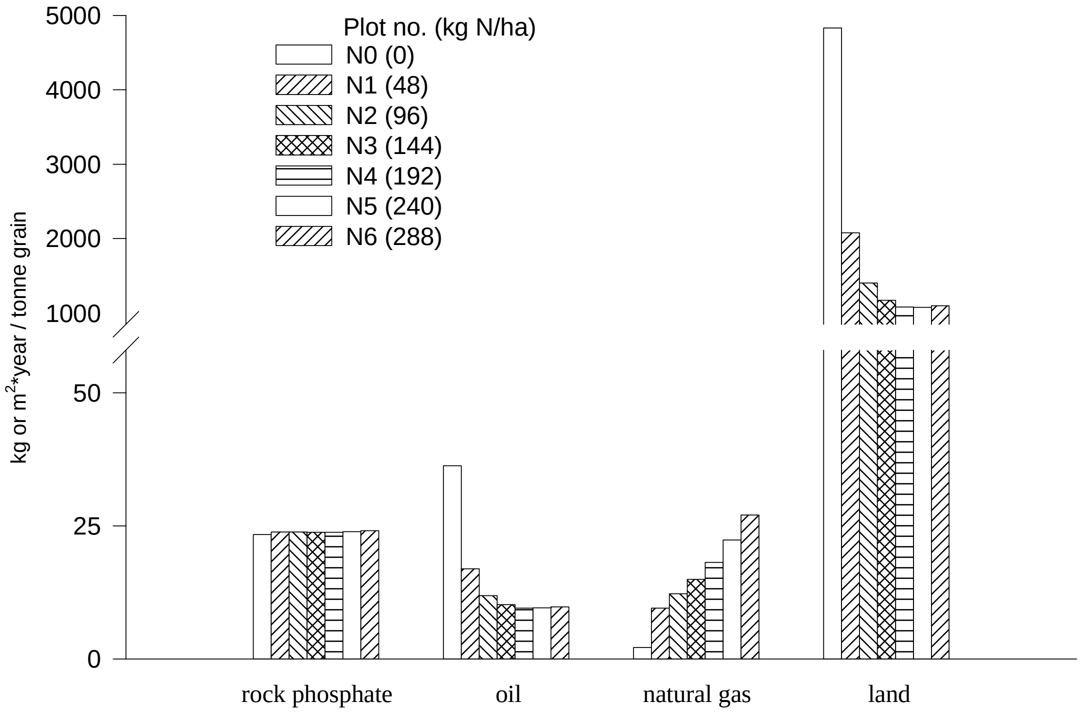

Figure 4: Consumption of selected abiotic resources (kg) and use of land (m^2*year) per tonne of grain at increasing N fertilizer rates

The application of P fertilizers per ha is for all treatments according to the P removal with the crops (see Tab. 1). Therefore, the consumption of rock phosphate per tonne of grain is almost constant for all plots (~ 24 kg/t grain).

Oil, which includes heavy and light oil as well as diesel, is primarily consumed due to on-field machinery use (67% of total oil consumption for N4). As this is partly constant for all production intensities (e.g. application of seeds, plant protection agents and base fertilizers, tillage, and harvest), the consumption of oil per tonne of grain strongly depends on the productivity of the analyzed system, i.e. on the grain yield per ha. Thus, oil consumption per tonne of grain is lowest for the most productive systems, which are N4 and N5 (about 9.6 kg/t grain).

Natural gas is predominantly consumed within N fertilizer production (ammonia synthesis), where it is used as process gas and energy source. Therefore, gas consumption per tonne of grain increases with increasing N fertilizer rates and is highest in N6 (27.1 kg/t grain). In contrast, the use of the resource “land” solely depends on the grain yield. Consequently, the higher the grain yield is per ha, the lower is the land use per tonne of grain. N4, N5 and N6 show the most efficient use of land (about 1100 m^2*year/t grain). The inventory data for all resources are given in Annex 1.

3.1.2 Selected emissions

Important emissions (CO_2, N_2O, NH_3 and NO_3-N) are shown for the different N fertilizer rates in Figure 5. These emissions were selected because earlier LCA studies have shown them to be of particular relevance for arable farming systems (e.g. Brentrup et al., 2001; Kuesters & Jenssen, 1998). Detailed inventory data for all emissions can be found in Annex 1.

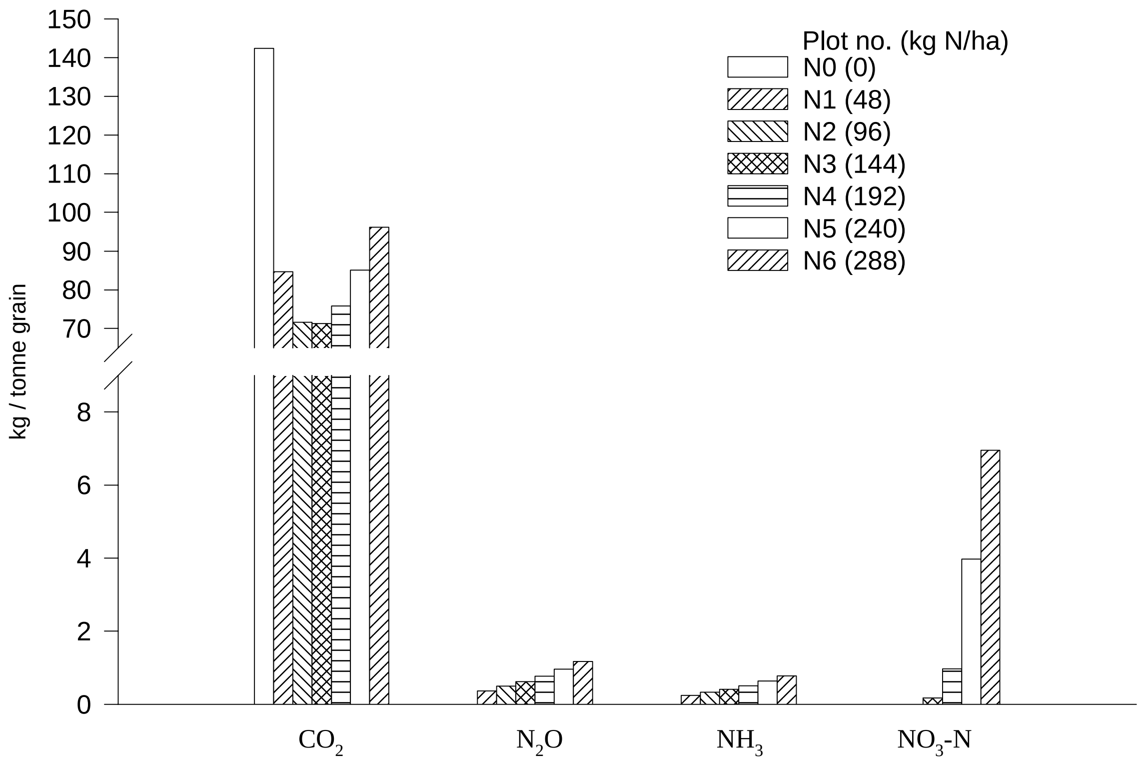

Figure 5: Release of selected emissions per tonne of grain at increasing N fertilizer rates

CO_2 emissions are released due to energy utilization during production and transportation of farming inputs (74% of total CO_2 emissions for N4) and on-farm machinery use (26%). As some of these activities (e.g. tillage, sowing, and harvest) are more or less similar for all treatments, the production efficiency per ha influences the CO_2 emission rates per tonne of grain. This leads to higher CO_2 emissions for N1 (84.7 kg/t grain) compared to N3, which shows the lowest value (71.3 kg/t grain). However, depending on the treatment, 0-59% of the total CO_2 emissions are directly connected to the production of N fertilizers. Therefore, increasing N fertilizer rates, which do not result in an equivalent high yield increase per ha, lead to increasing CO_2 rates per tonne of grain (highest rate in N6: 96.1 kg/t grain).

The N_2O emissions are very much dependent on the production intensity. This is due to the fact that the total N_2O emissions are related to the production and application of N fertilizers. For instance, 48% of the N_2O emissions for N4 are attributable to nitric acid production, which is part of ammonium nitrate production, and 52% is released via denitrification and nitrification of fertilizer N after application. Therefore, the highest N_2O emissions can be found in the treatment with the highest N application rate (N6: 1.17 kg/t grain).

Similar to N_2O, NH_3 emissions are strongly dependent on the N fertilizer rate. Since most of the ammonia emissions in the analyzed system occur after application of ammonium nitrate fertilizer, the NH_3 emission rates increase with increasing N fertilizer rates.

At Broadbalk NO_3 leaching has been measured only in one section of the field trial, which is cropped continuously with winter wheat (Goulding et al., 2000). Since yields per ha and therefore also N uptake and removal are clearly higher in the "rotation-plots" compared to the "continuous-wheat-plots" (e.g. average of 9.25 t/ha for N6 in the rotation vs. 7.20 t/ha for the continuous wheat), leaching rates will be different for these sections. Therefore, the potential NO_3 leaching caused by the wheat production systems under investigation has been estimated according to a method described by Brentrup et al. (2000). It considers all relevant N inputs and outputs (mineral fertilizers, atmospheric deposition, gaseous losses, removal with grain and straw), as well as important soil and climatic parameters like soil texture, precipitation and evapotranspiration. The NO_3 leaching estimated by this method represents only that fraction of the total NO_3 leaching, which can be directly assigned to the wheat production under investigation. The potential NO_3 emissions show a strong dependence on the N fertilizer rate applied. Since in N0, N1 and N2 the N removal by the crops exceeds the N inputs, no leaching of nitrate attributable to the N fertilizer use has been calculated. In contrast, the potential NO_3-N loss for N6 amounts to nearly 7 kg per tonne of grain (= 63 kg NO_3-N/ha).

3.1.3 LCI conclusions

Life Cycle Inventory data give a comprehensive insight into the environmental implications of the different wheat production intensities. The LCI provides product-related data on single emissions or resource consumption. This kind of data is important for exploring possibilities for the improvement of the environmental performance of one or other production intensities. If, for example, the contribution to climate change would be the major problem related to wheat production, the analysis of the LCI data enables the most relevant sources to be identified and to check whether and where reductions are possible. However, the LCI data do not allow conclusions to be drawn on the overall environmental preference of one or the other production intensity. Furthermore, the importance of a single emission or resource consumption to the overall environmental problem is not yet considered. Therefore, the impact assessment step has to follow the inventory in order to aggregate the data into indicator values for impact categories like climate change or eutrophication.

3.2 Life Cycle Impact Assessment

3.2.1 Characterization

3.2.1.1 Depletion of abiotic resources

Within the impact category "depletion of abiotic resources", those resources, which are functionally equivalent to each other, are aggregated into sub-categories (Brentrup et al., 2002a). The abiotic resources consumed in wheat production systems can be aggregated into the sub-categories *depletion of fossil fuels* (expressed in MJ), *phosphate rock* (in kg P_2O_5), and *potash* (in kg K_2O). The results for the sub-categories "depletion of phosphate rock" and "depletion of potash" per tonne of grain are equal to the respective LCI results, because the only contribution of these categories is consumption of phosphate rock and potash.

Figure 6 gives the combined consumption of fossil fuels in MJ per tonne of grain for the 7 N fertilizer rates (bars). In addition, the yields are shown in order to illustrate the different productivity of the treatments, which strongly influences the RDI values for fossil fuel consumption per tonne of grain.

For N2 and N3 the energy consumption is lowest per tonne of grain (~1060 MJ). That means it is possible to increase the yield from 7.11 (N2) to 8.53 t/ha (N3) by an additional application of 48 kg N/ha without increasing the energy use per tonne of grain. The application of more than 144 kg N/ha or less than 96 kg N/ha results in an increasing energy use per tonne of grain (N0, N1, N4, N5, N6). For N4 (192 kg N/ha) the consumption of natural gas and oil contribute 51 and 35% respectively in the total fossil fuel consumption per tonne of grain (coal = 14%).

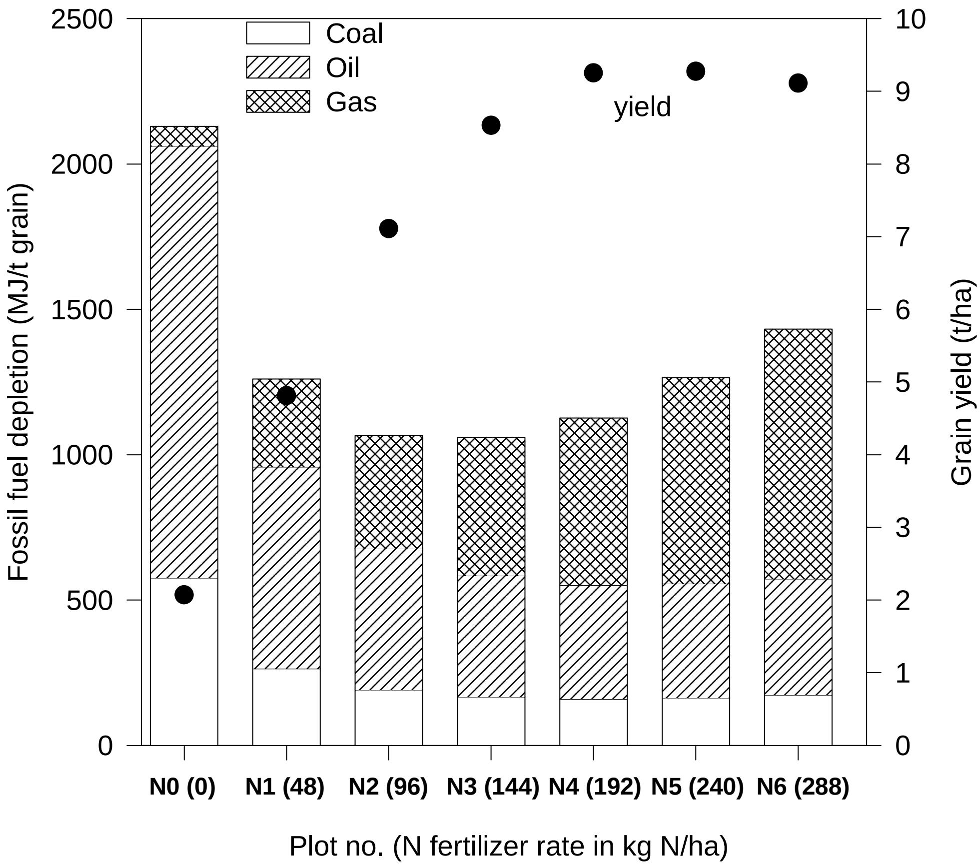

Figure 6: Combined consumption of fossil fuels (MJ/t grain, bars) and yields (t/ha, dots) at increasing N fertilizer rates

3.2.1.2 Land use

The impact category "land use" deals with the degradation of natural land due to human utilization for agriculture, housing, roads, industry etc. The calculation of a Naturalness Degradation Potential (NDP) not only includes the area used for a certain period of time but also the intensity of land use, e.g. built-up area vs. extensive pasture (Brentrup et al., 2002b). However, for the compared wheat production systems the intensity of land use is uniform, since the wheat production in any treatment includes intensive soil preparation, plant protection measures, base fertilization etc. Thus, for each N application rate the land area used for a certain period of time (in m^2*year/t grain) is multiplied by the characterization factor for intensive arable land use (0.8; Brentrup et al., 2002b) to give the NDP values per tonne of grain for each treatment. Figure 7 reveals that the land use per tonne of grain is lower, the higher the yield. Therefore, the plots with the highest yields (N4, N5 and N6) show the lowest NDP values (<900 m^2*year/t grain). In N0, the land is used most inefficiently (3865 m^2*year/t grain).

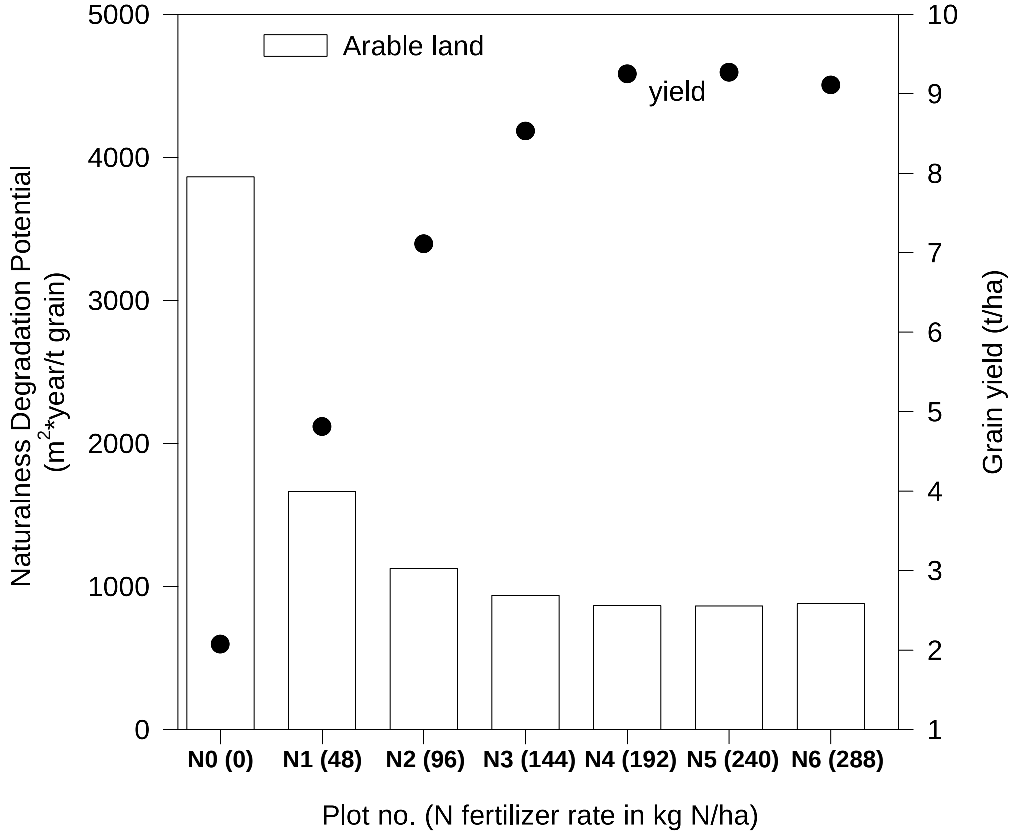

Figure 7: Naturalness Degradation Potentials (m^2*year/t grain, bars) and yields (t/ha, dots) at increasing N fertilizer rates

3.2.1.3 Climate change

The Global Warming Potential (GWP) is used to express the contribution that gaseous emissions from arable production systems make to the environmental problem of climate change. Figure 8 reveals that the GWP per tonne of grain increases almost linearly with increasing inputs. The N_2O emissions, which are closely related to the N input (see Fig. 5), are responsible for this close relationship. Although the absolute emission rates of N_2O are much lower compared to those of CO_2 (see Fig. 5), N_2O dominates the total GWP per tonne of grain in all production intensities, except for N0. This is due to the fact that 1 kg of N_2O has a GWP 310 times higher than that of 1 kg of CO_2. For N4, N_2O contributes 76% of the total GWP per tonne of grain (CO_2 = 24%). Methane (CH_4) emissions are negligibly low for all N rates.

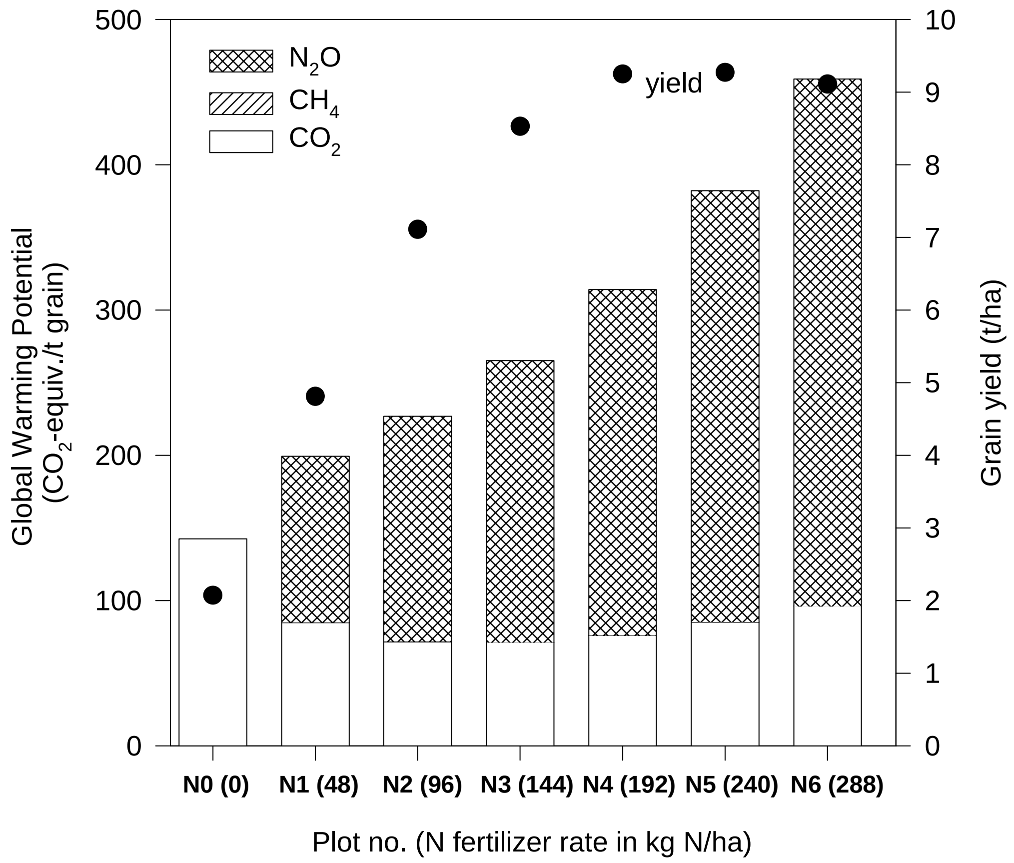

Figure 8: Global Warming Potentials (kg CO_2-equivalents/t grain, bars) and yields (t/ha, dots) at increasing N fertilizer rates

3.2.1.4 Toxicity

The impact category "toxicity" comprises the effects of toxic substances on humans and ecosystems. In arable farming, use of pesticides and heavy metals may contribute to this environmental problem. However, since no appropriate aggregation method is currently available for plant protection agents, the potential toxic effects of pesticide emissions are not included in the present study (Brentrup et al., 2002c). The only substance that is considered in the toxicity sub-categories human toxicity, fresh water and marine eco-toxicity, and fresh water and marine sediment eco-toxicity (Goedkoop & Spriensma, 1999; Huijbregts, 2001) is cadmium. Cadmium is added to soil as an impurity in phosphate fertilizers. In the present wheat production system, P fertilizer is applied to all treatments according to the P removed in the crops. Therefore, for all toxicity sub-categories almost no differences per tonne of grain between the treatments can be observed. As an example, Figure 9 shows the Human Toxicity Potentials expressed in Disability Adjusted Life-Years (DALY) per tonne of grain for the increasing N fertilizer rates. According to WHO (Murrey, 1994) the unit of DALY describes the time a person loses due to premature death or lives with a certain limitation as a result of a specific toxic emission.

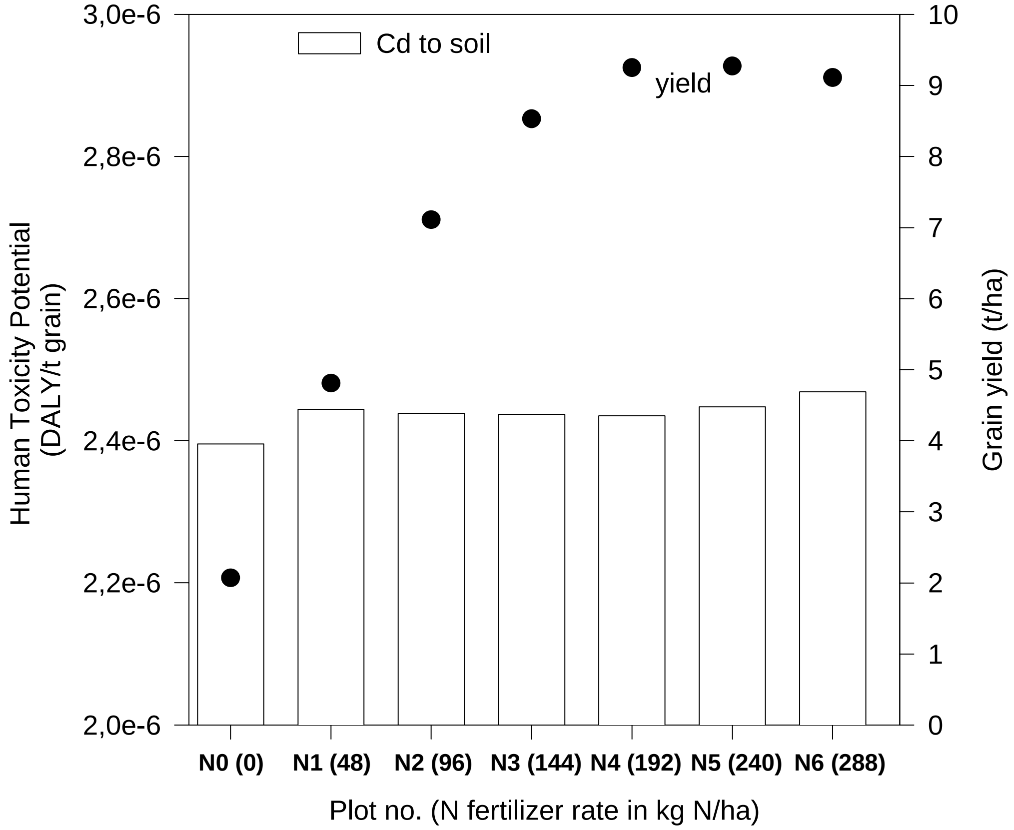

Figure 9: Human Toxicity Potentials (DALY/t grain, bars) and yields (t/ha, dots) at increasing N fertilizer rates

3.2.1.5 Acidification

The Acidification Potential (AP) of a system, expressed in the present system as kg SO_2-equivalents/t grain, represents its contribution to the acidification of natural ecosystems like forests or lakes. Figure 10 shows the lowest AP for N2 (1.11 kg SO_2-equiv./t grain), and the highest for N6 (1.75 kg). The more intensive the wheat production, the higher the contribution of NH_3 emissions to the total AP and the lower is the relevance of SO_2 and NOx. NH_3 is emitted due to volatilization after application of ammonium nitrate fertilizer. For N4, NH_3 contributes 58%, SO_2 24% and NOx 18% of the total AP per tonne of grain (calculated from absolute numbers given in Annex 1).

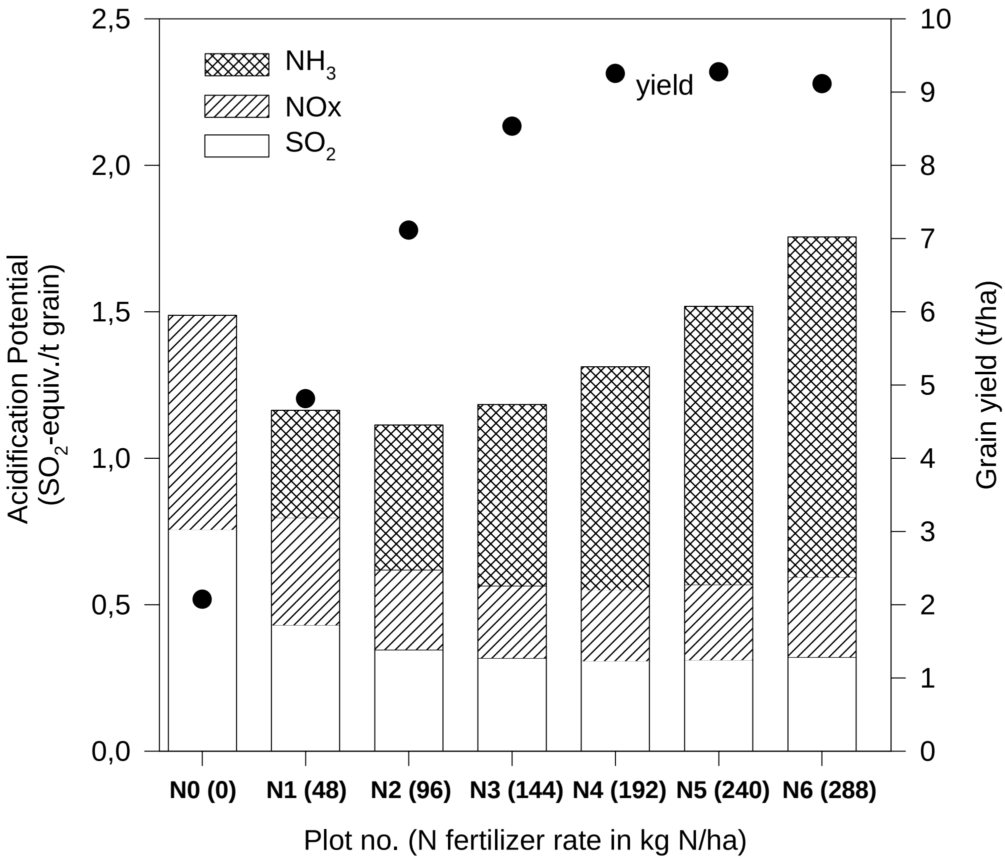

Figure 10: Acidification Potentials (kg SO_2-equivalents/t grain, bars) and yields (t/ha, dots) at increasing N fertilizer rates

3.2.1.6 Eutrophication

The impact category “eutrophication” is divided into two sub-categories (terrestrial and aquatic eutrophication), because terrestrial (mainly higher plants) and aquatic plants (mainly algae) respond differently to an additional supply of nutrients (Brentrup et al., 2002c).

Terrestrial eutrophication is caused by atmospheric deposition of nutrients on natural land ecosystems. Figure 11 reveals for the Terrestrial Eutrophication Potential (TEP in kg NOx-equivalents per tonne of grain), a very similar picture compared to the Acidification Potential (see Fig. 10). This is due to the fact that NH_3 and NOx contribute to both environmental effects. For the less intensive wheat production systems (N0, N1, N2), NOx (mainly from transport and tractor use) is more important for terrestrial eutrophication, but for the higher production intensities NH_3 is more important. As for acidification, N2 shows the lowest (1.31 kg NOx-equiv./t grain), and N6 the highest TEP (2.09 kg).

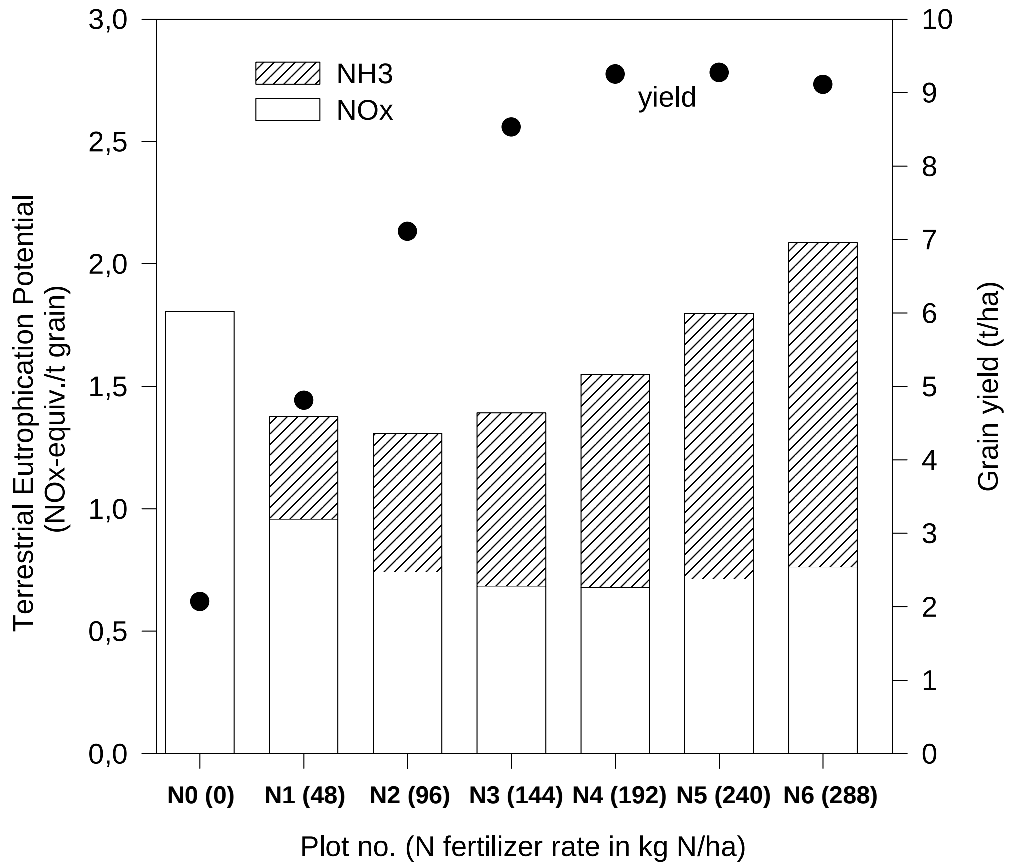

Figure 11: Terrestrial Eutrophication Potentials (kg NOx-equivalents/t grain, bars) and yields (t/ha, dots) at increasing N fertilizer rates

Figure 12 shows the Aquatic Eutrophication Potential (AEP) for the different treatments. The figure clearly indicates that at N rates higher than N3 (144 kg N/ha), the AEP is dominated by NO_3 leaching (e.g. 92% of the total AEP for N6; calculated from absolute numbers given in Annex 1). In particular at production intensities above the optimum N rate (N4, 192 kg N/ha, see Fig. 1), nitrate losses via leaching result in high AEP values (2.22 kg PO_4-equiv./t grain for N6 vs. 0.42 kg for N4). At N rates higher than 144 kg N/ha, airborne nutrient emissions, which deposit on surface waters (NOx, NH_3) and direct effluents of P (from P fertilizer production) contribute less to the AEP compared to nitrate (e.g. for N4: NOx and NH_3 = 24% of total AEP, P_{tot} = 9% and NO_3-N = 67%; calculated from absolute numbers given in Annex 1).

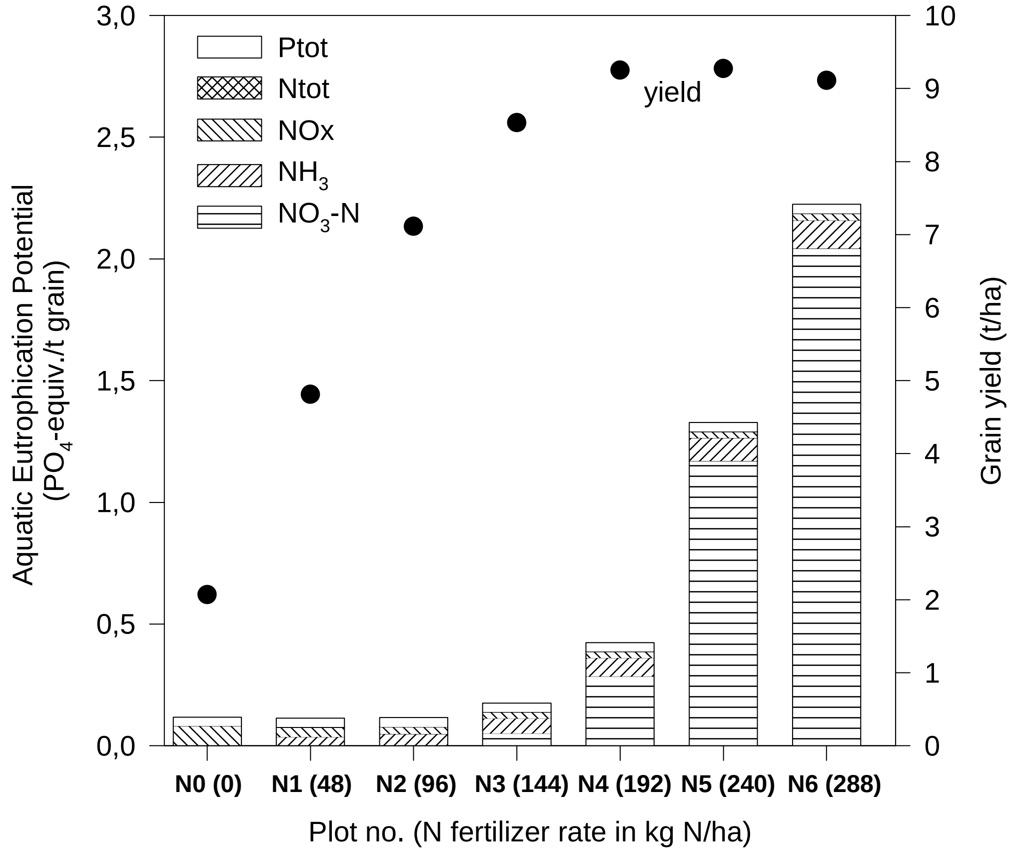

Figure 12: Aquatic Eutrophication Potentials (kg PO_4-equivalents/t grain, bars) and yields (t/ha, dots) at increasing N fertilizer rates

3.2.2 Normalization

Figure 13 shows the contribution of the different wheat production intensities to the total consumption of phosphate rock, potash and fossil fuels per person in Europe. These normalized indicator results are calculated by dividing the indicator values per tonne of grain by reference values, which are the respective indicator values per person in Europe (Brentrup et al., 2002c).

Figure 13 reveals that arable farming systems contribute far more to the depletion of potash and phosphate rock than to the depletion of fossil fuels. This is due to the fact that agriculture is the dominant consumer of potash (90% of total consumption in US; USGS, 2002) and rock phosphate (79% in W-Europe; EFMA, 1999). In contrast, fossil fuels are predominantly consumed by industrial production (33% in Europe), households (27%) and transportation (23%), whereas agriculture accounts for only 4% of the total energy consumption in Europe (WRI, 2000). Therefore, it is not surprising that arable farming systems, in which P and K fertilizers are applied, show a much greater contribution to the depletion of P and K resources than to the depletion of fossil fuels. Since P and K fertilizers are applied in each treatment according to the P and K removal in the crops, almost no differences can be observed between the treatments (see 3.2.1.1).

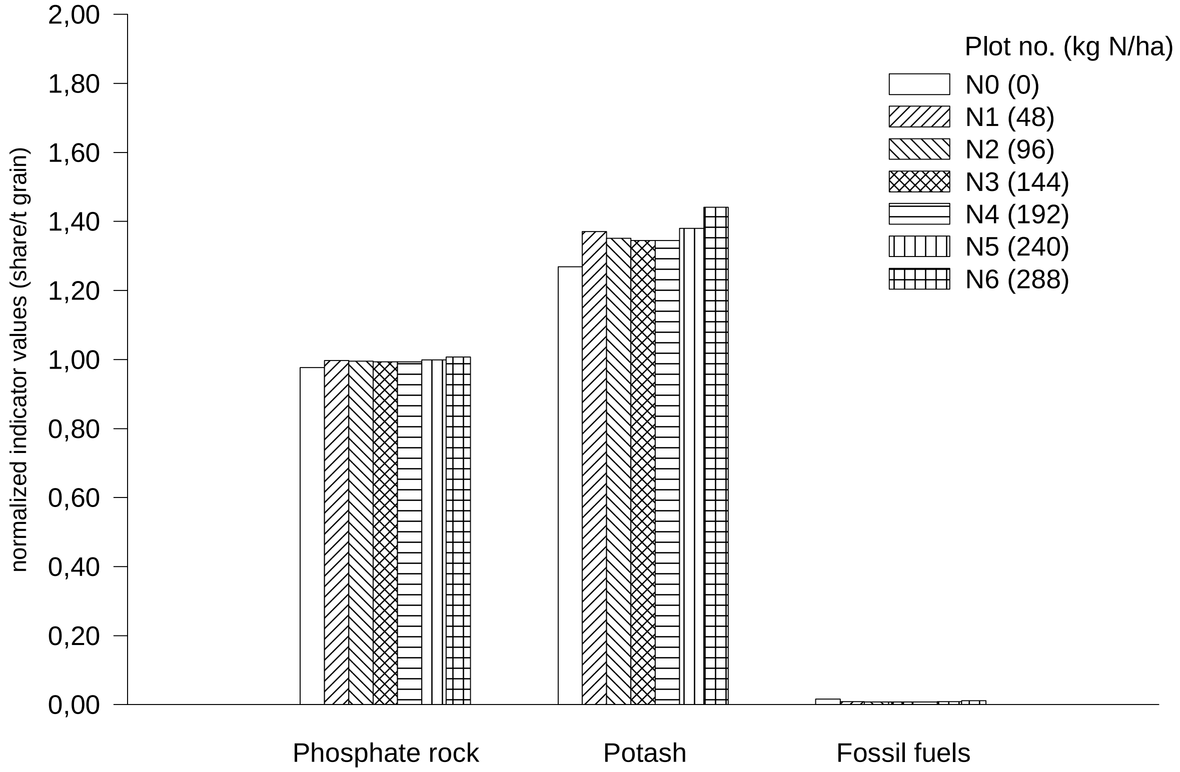

Figure 13: Contribution of wheat production at increasing N fertilizer rates to the depletion of phosphate rock, potash and fossil fuels per person in Europe

Figure 14 gives the normalization results for the impact categories, which deal with effects on natural ecosystems and human health. The results indicate that at production intensities up to 96 kg N/ha (N0, N1 and N2) the contribution to the degradation of naturalness due to land use in Europe is the most relevant environmental impact connected to the production of one tonne of wheat grain. At N rates of 192 kg N/ha and above the contribution of wheat production to the total aquatic eutrophication in Europe becomes more relevant than the contribution to other impact categories. All treatments show a comparable share in the total European indicator values for climate change, acidification and terrestrial eutrophication. In comparison, the share in the European toxicity indicators is in any case very low.

However, from the normalization results no decision about the overall environmental preference of one or other production alternatives is possible. Although theoretically possible, the normalized values should not be summed up because information on the potential of the different effects to harm resource availability, natural ecosystems and human health (weighting) is still not included. Therefore, the different impact categories are weighted based on the so-called distance-to-target principle (Brentrup et al., 2002c).

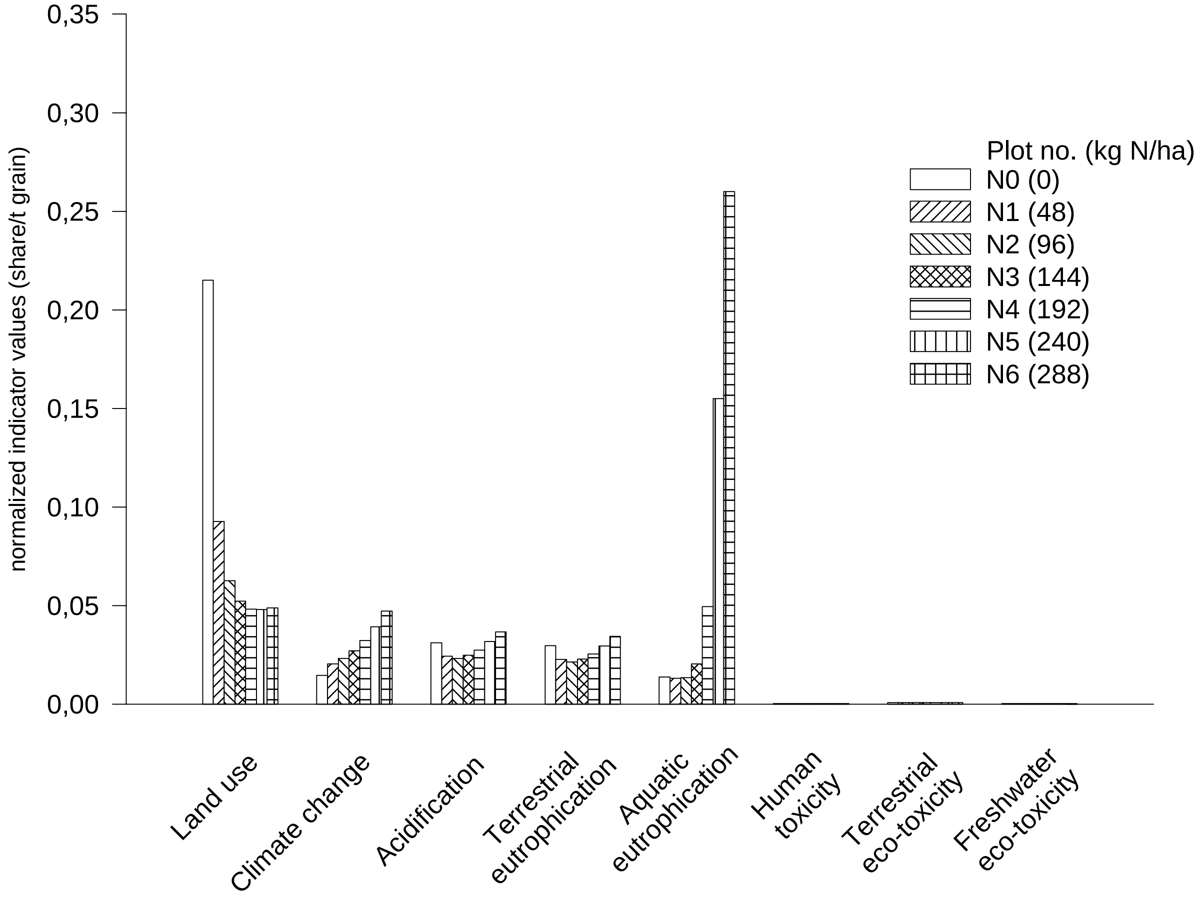

Figure 14: Contribution of wheat production at increasing N fertilizer rates to environmental and human health effects per person in Europe

3.2.3 Weighting

During the weighting step, each normalized indicator value is multiplied by a weighting factor, which represents the potential of the respective impact category to harm resources, natural ecosystems and human health. Due to substantial differences between impacts on ecosystems and human health, and impacts on abiotic resources, in the present LCA method two separate indicators result from the weighting step. The reasons for this differentiation are discussed in Brentrup et al (2002a).

The weighting factors for the different impact categories have been developed independently from any LCA case study and are based for each impact category separately on the ratio between a defined target indicator value and the current status of the impact (“distance-to-target”; Brentrup et al., 2002c). For the consumption of resources, the target value is a yearly consumption rate, which ensures the availability of the respective resource for another 100 years (Brentrup et al., 2002a). At current consumption rates the proved reserves of potash salts will last for more than 300 years (USGS, 2002). Therefore, potash consumption is not part of the aggregated resource depletion indicator (RDI), which is shown in Figure 15.

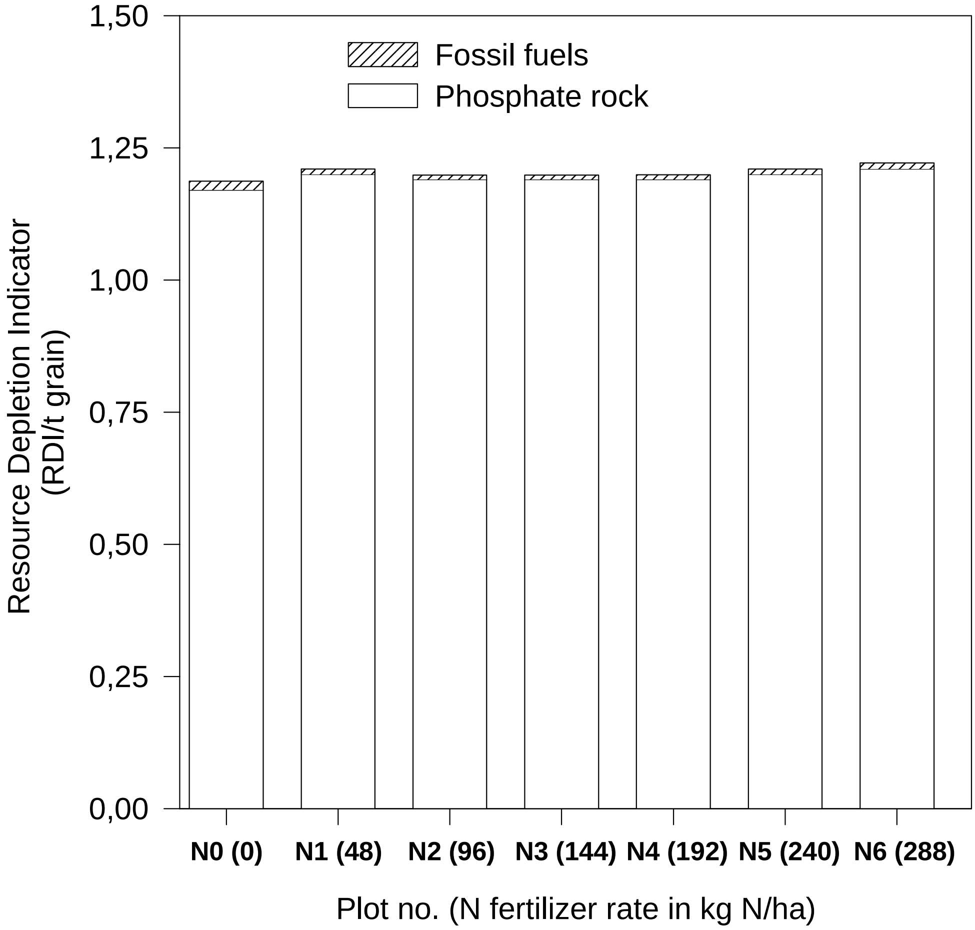

Figure 15: Aggregated resource depletion indicator values (RDI) per tonne of grain at increasing N fertilizer rates

The normalized indicator values for the consumption of phosphate rock and fossil fuels are multiplied by the respective weighting factors, which are 1.05 for fossil fuels and 1.20 for phosphate rock. Since the normalized indicator value for P rock is far higher than that for fossil fuels, P rock dominates the aggregated RDI value. Since P fertilizers are applied in each treatment according to the P removed in grain and straw (see Table 1), the RDI values per tonne of grain show almost no differences. Differences in the use of fossil fuels are hardly perceptible, as fossil fuel consumption accounts for only 0.7-1.4% of the total RDI values.

Figure 16 shows the aggregated environmental indicators (EcoX) per tonne of grain for the increasing N application rates. The lowest environmental impact is calculated for N2 (96 kg N/ha, 0.16 EcoX/t grain). N3 shows almost the same environmental impact per tonne of grain (144 kg N/ha, 0.17). The EcoX value for the economic optimum treatment (N4, plot 9, 192 kg N/ha, 0.22) is 28% higher compared to N3. The highest indicator values were for N6 (288 kg N/ha, 0.55), N5 (240 kg N/ha, 0.38) and N0 (zero N, 0.33). Whereas for N0 the aggregated environmental indicator is dominated by land use impacts (66% of the total value), the increasing EcoX values for N5 and N6 can be mainly attributed to aquatic eutrophication (55% for N5, 65% for N6). The highest share in the environmental impact calculated for N4 (192 kg N/ha) shows aquatic eutrophication (31%), followed by land use (22%), acidification (17%), climate change (16%), and terrestrial eutrophication (15%).

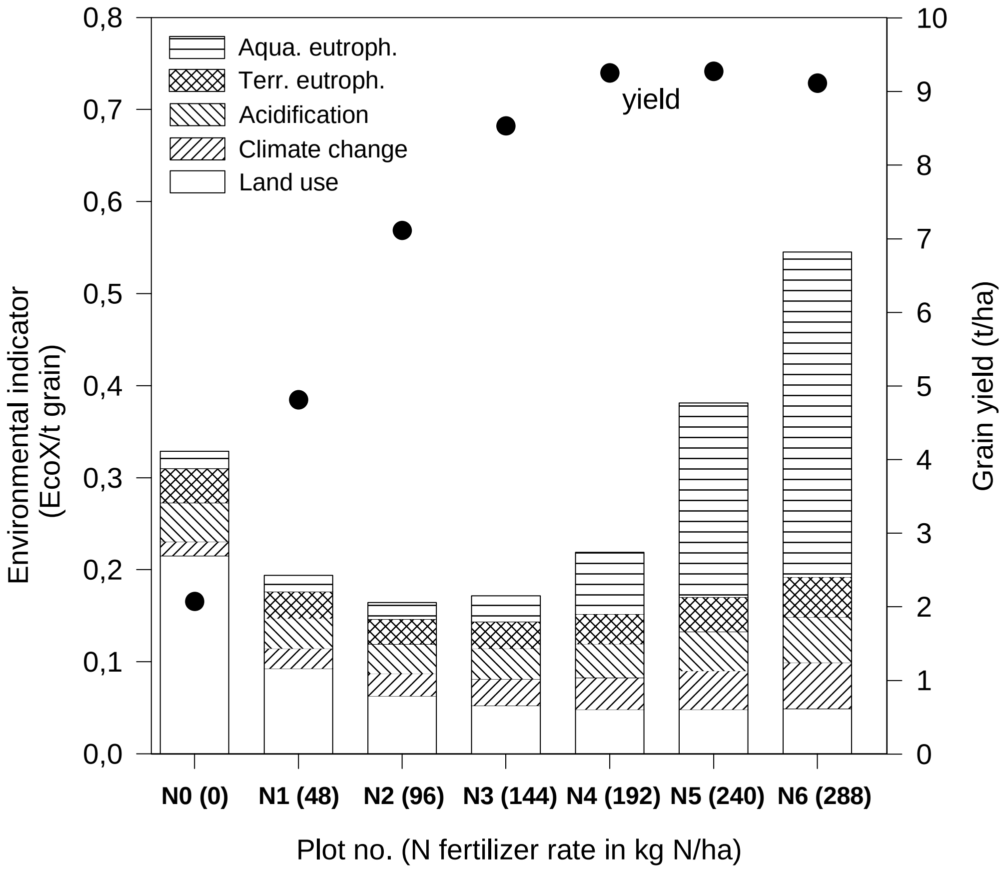

Figure 16: Aggregated environmental indicator values (EcoX) per tonne of grain (stacked bars) and yields (t/ha, dots) at increasing N fertilizer rates

4. Discussion and conclusions

In this study a new LCA method specifically tailored to crop production systems (Brentrup et al., 2002c) is used to investigate the environmental impact of different N fertilizer rates in winter wheat production. The wheat production system studied was part practical and part theoretical involving elements of "best farming practice". The yield response to the different N rates is taken from a long-term field trial (Broadbalk Experiment, Rothamsted, UK). The LCA method takes into account the contribution of the wheat production to the effects *resource depletion*, *land use*, *climate change*, *toxicity*, *acidification*, and *eutrophication*.

Firstly, an inventory of all single emissions and consumption of resources is compiled, and indicator values for the environmental effects are calculated (characterization). Next, the contribution of the wheat production to the total environmental effects in Europe is determined (normalization). Finally, the indicator values are evaluated and aggregated further by multiplication with weighting factors, which represent the potential of each impact to harm the environment.

This study clearly illustrates the advantages of the LCA methodology when it comes to an evaluation of environmental preferences of different production intensities in arable farming. If, for instance, just the release of greenhouse gases had been chosen for the evaluation, the most extensive production system (least inputs) would perform best (see Fig. 8). If fossil fuel consumption were the only indicator, then medium N rates would be most favorable (see Fig. 6). However, both approaches would totally ignore the fact that land is used most efficiently in intensive treatments with high yields per unit area (see Fig. 7). Given sufficient grain supply for a defined region, the most efficient use of the most productive land would release less productive land for other purposes in that area (e.g. as nature reserves). In this respect, the advantage of LCA is that all relevant impacts are considered and evaluated simultaneously.

Another advantage of LCA is that it shows the relevance of the impacts associated with the system under investigation in relation to the respective total impacts in a given region, e.g. Europe. This normalization procedure shows how the system under analysis contributes to different impact categories.

In the present case study, the aggregated **environmental impact** calculated for the N treatments of 48, 96, 144 or 192 kg N/ha were within a range of EcoX values of 0.16 to 0.22 per tonne of grain. The treatments receiving 0, 240 or 288 kg N/ha show 100% to 232% higher EcoX values compared to the lowest figure (N2, 96 kg N/ha). This result indicates that in high-yielding crop production systems (e.g. N4, 192 kg N/ha) economic and environmental considerations are not necessarily in conflict, whereas a significant under- or oversupply with nitrogen fertilizers (e.g. N0, N5, N6) leads to decreasing eco-efficiency in crop production.

Wheat production involves two environmental hotspots - *land use* and *aquatic eutrophication*. Thus, the greatest potential to minimize the environmental impact per tonne of grain is to achieve high yields per unit of land (i.e. a high land use efficiency) and at the same time low NO_3 leaching rates, which are most responsible for aquatic eutrophication (see Fig. 12). At N3 (144 kg N/ha), the aggregated EcoX value for these two competing aspects is lowest. Consequently, other impacts than land use and aquatic eutrophication get a higher relative importance in this treatment compared to the other N rates (see Fig. 16).

The differences in the *acidification* potential are mainly determined by NH_3 emissions (see Fig. 10). Since in the system under consideration, ammonium nitrate (AN, Tab. 2.) is used as N fertilizer, the NH_3 emissions are low compared to the use of other mineral and organic fertilizers (ECETOC, 1994). Other LCA studies have shown that, for example, the use of urea or organic fertilizers (e.g. slurry) as N sources results in much higher acidification potentials (Kuesters & Jenssen, 1998; Brentrup et al., 2001). Therefore, in the wheat production system under investigation the release of acidifying emissions will be difficult to reduce. The same holds true for the contribution to *terrestrial eutrophication* since again the NH_3 emissions are a decisive factor for this indicator value (see Fig. 11).

For *greenhouse gases* the picture is different. Part of the global warming potential could be avoided by choosing an ammonium or urea based N fertilizer. Significant amounts of N_2O are emitted during the production of nitric acid, which is part of ammonium nitrate production. However, as already mentioned a switch to urea or another non-nitrate fertilizer would lead to higher contributions to acidification and eutrophication. Ideally, the N_2O emissions related to nitric acid production should be reduced, which is technically possible (Laegreid et al., 1999).

From this LCA case study, it can be concluded that a good environmental performance in wheat production can be achieved by:

- maintaining high yields, in order to use land most efficiently.
- applying nitrogen according to crop demand, in order to minimize NO_3 leaching.
- using nitrogen fertilizers with low NH_3 volatilization rates (e.g. AN), in order to keep the acidification and terrestrial eutrophication potentials low.
- reducing N_2O emissions during nitrate fertilizer production (scrubbing techniques), in order to reduce the global warming potential.

For the problem of **resource depletion**, the present case study reveals clearly that the impact of arable farming on decreasing availability of exploitable phosphate rock resources is by far greater than that on decreasing availability of fossil fuel resources. Since phosphate rock reserves are scarce (about 85 years availability of proved reserves at current extraction rates; USGS, 2002) and phosphates are essential nutrients in crop production, a responsible use of P resources is important. This could be achieved by considering the P status of the soil and P removal by crops in P fertilization, and the recycling of phosphates contained in animal manures). Fossil fuel consumption in agriculture is relatively low (4% in Europe; WRI, 2000), so that efforts to reduce the consumption may be more efficient in other sectors like industry (33%), domestic (27%) or transport (23%). However, the most energy efficient production intensities (96 and 144 kg N/ha) are also favorable from an environmental point of view (see Fig. 16).

VIII. Discussion and conclusions

In the following the general suitability of the LCA methodology and in particular the suitability of currently available LCA tools to investigate and evaluate the environmental impacts of arable farming systems will be discussed. New methodological developments have been proposed, which will be described and discussed in a subsequent section. This methodological work is part of a new LCA method, which is specifically tailored to evaluate the environmental impact of plant nutrition in arable crop production. In the final section of this chapter the main results of the application of this new LCA method to investigate the environmental impacts of wheat production at different fertilizer rates will be described.

1. Application of the LCA methodology to investigate the environmental impacts of arable crop production

LCA is a methodology, which is designed to analyze the environmental impact of products (Heijungs et al., 1992; Consoli et al., 1993; ISO, 1997). LCA is defined as an inventory and valuation of all potential environmental impacts related to a product. In LCA, environmental impacts are impacts on natural eco-systems, human health and natural resources. The most specific characteristic of LCA is the “life-cycle thinking” (Finnveden, 1998), i.e. life-cycle thinking is to focus on the product under investigation and to include the entire system of main and sub-processes necessary to the produce, use and dispose the product.

LCA is divided into four steps, which are *goal and scope definition*, *inventory analysis*, *impact assessment* and *interpretation* (SETAC, 1993; ISO, 1997).

During *goal and scope definition* the product under investigation, its function, and boundaries are described. In the subsequent *Life Cycle Inventory* the resource consumption and emissions associated with the product are compiled. The inventory data as such do not allow comparisons to be made between different systems. Furthermore, the potential environmental impact of the various emissions and resource consumption is not considered in this phase. During the *Life Cycle Impact Assessment* the inventory data are therefore evaluated with regard to their potential to harm natural ecosystems, human health, and resources. Finally, in the *interpretation* phase, the inventory and impact assessment results are analyzed and conclusions are drawn in order to define options to improve the environmental performance of the product under investigation.

Different approaches to convert this general LCA concept into an operational LCA method have been published (e.g. BUWAL, 1998; Goedkoop, 1995; Goedkoop & Spriensma, 1999; Guinée et al, 2001; Heijungs et al., 1992a; Steen, 1999). All these methods have been primarily designed for industrial applications.

The Dutch LCA method "Eco-indicator'95" (Goedkoop, 1995) has been applied in a case study on sugar beet production in order to test its suitability to investigate the environmental impacts of arable farming systems (Chapter II). The Eco-indicator'95 was chosen as a representative for currently available LCA methods, because it is a well recognized, frequently used and in detail documented method.

The Eco-indicator'95 method enables a comparative analysis of a product's contribution to the environmental problems of global warming, acidification, eutrophication, summer smog, winter smog, depletion of the ozone layer, and emissions of pesticides and heavy metals. The method includes the calculation of indicators for each of these environmental issues. After normalization and weighting it is possible to calculate a fully aggregated environmental indicator, the Eco-indicator'95. However, the study revealed that the Eco-indicator'95 method, as well as the other currently available LCA tools, has some specific constraints, when applied to arable farming systems and also shows some general methodological problems.

<u>The following conclusions can be drawn (Chapter II):</u>

(1) The LCA methodology is principally suitable to investigate and evaluate the environmental impacts associated to the production of arable crops.

(2) The application of the LCA methodology revealed that diffuse, on-field nitrogen emissions (NO_3, NH_3, and N_2O) are of particular importance for LCA studies of arable production systems. Inventory data of good quality for these emission types are therefore important for reliable LCA results.

(3) The environmental consequences of resource consumption (e.g. phosphate rock, fossil fuels) and land use are missing in the Eco-indicator'95 method; those of nutrient emissions are neglected in the Eco-indicator'99 approach. Since arable farming considerably contributes to these impacts, they should be included in a LCA method for arable crop production systems.

(4) In current LCA methods the impact assessment of acidifying or eutrophying emissions does not consider the distribution and deposition pattern (fate) of the emissions and the sensitivity of the receiving region. Since arable farming considerably contributes to acidification and eutrophication, a more accurate and regional assessment of acidification

and eutrophication impacts should be included in a LCA method for arable crop production.

(5) The procedure used in the Eco-indicator '95 method to derive weighting factors by application of the "distance-to-target" principle is basically convincing. However, the weighting step is often controversially discussed, because it cannot be based solely on scientific knowledge but always involves subjective assumptions and values. In order to gain broader acceptance for this important step in LCA, it is therefore necessary to employ environmental targets, which are based on widely agreed international conventions like the "Kyoto protocol" for climate change (UN-FCC, 1998) rather than on subjective assumptions.

Based on the conclusions drawn from this study, methodological adjustments and improvements of the LCA methodology have been developed in order to make LCA more suitable for the environmental analysis of arable crop production systems.

2. Methodological developments to adjust LCA to the requirements of arable crop production

As shown in Chapter II it is particularly important for LCA studies that include arable production to use reliable estimates of diffuse, on-field nitrogen emissions as inventory data. These emissions usually play an important role within the subsequent impact assessment. It was one aim of this study to provide LCA practitioners with a methodology to calculate estimates of NH_3, N_2O and NO_3 emissions under consideration of important soil, climate and management parameters (Chapter III). Furthermore, new proposals for the impact assessment of the consumption of abiotic resources (Chapter IV) and land use (Chapter V) have been developed. These new methodological developments are part of a comprehensive LCA method, which is tailored to the environmental analysis of arable crop production systems (Chapter VI).

2.1 Estimation of diffuse, on-field nitrogen emissions as an input to LCA studies including arable crop production

As shown in Chapter II and other studies (Audsley et al., 1997; Küsters & Jenssen, 1998; Cederberg, 1998) diffuse, on-field emissions of ammonia (NH_3), nitrous oxide (N_2O) and nitrate (NO_3) often contribute considerably to the LCA results.

Inventory data on diffuse, on-field nitrogen emissions can be obtained by measurements. However, to actually measure ammonia, nitrous oxide and nitrate emissions is money and time consuming and therefore often not operational in LCA studies. Furthermore, measurements of N emissions often show great variations in time (e.g. Isermann, 1990, for NH_3) and thus, especially short-term measurements may only reflect a snapshot of the specific conditions at the time of measurement. For LCA purposes average emissions adjusted to the conditions typical for the system under examination would be more appropriate than short-term measurements.

It is possible to derive representative emission rates from a literature study. Such data are assumed to reflect an average emission rate representative for the system examined in the LCA. A disadvantage of this procedure is that for each new study a new literature review might be necessary to obtain appropriate values. Furthermore it is difficult to evaluate the quality of the derived figures as they strongly depend on the quality of the literature source.

A third way would be to employ structured estimation methods to calculate average, study-specific emission rates for arable crop production. Conditions, which influence the nitrogen emissions, are considered by appropriate parameters (soil, climate, and agricultural practice). Most of the required parameters are usually available in LCA studies and can therefore be used as input for the estimation methods. This study (Chapter III) suggests different approaches to estimate NH_3 emissions from organic (Horlacher & Marschner, 1990) and from mineral fertilizers (ECETOC, 1994). A function derived by Bouwman (1995) is selected to calculate the N_2O emissions. A method developed by the German Soil Science Association (DBG, 1992) is adopted to determine potential NO_3 emissions. A comparison of the suggested methods with other estimation procedures revealed big differences in the calculated emission rates, even applied on the same wheat production system (Chapter III, Table 13). This result clearly confirms the need for consistent methods to estimate diffuse on-field nitrogen emissions. In contrast to the other approaches employed in the case study, the methods suggested in this study include important specific soil, climate and management parameters, which should be available in any LCA study on arable production, and thus provide realistic study-specific estimates of diffuse on-field nitrogen losses.

An inevitable disadvantage of estimation methods is the need to simplify the complex conditions that lead to the release of emissions into the environment. However, if the most important conditions are considered like in the methods suggested by this study, the quality of the Life Cycle Inventory data can be improved.

2.2 Impact assessment of the consumption of abiotic resources

In addition to the difficulties to compile inventory data for LCA studies on arable systems, also a need to improve the impact assessment procedure for the consumption of abiotic resources like phosphate rock or fossil fuels has been identified. The issue related to the consumption of abiotic resources is their decreasing availability for future generations rather than the environmental impacts related to their consumption, which is considered in other impact categories in LCA (Chapter IV).

In recent impact assessment methods (e.g. Goedkoop & Spriensma, 1999; Guinée, 2001) the inventory data on the consumption of various resources are directly aggregated into one resource depletion indicator neglecting for what purpose the resources are used like for supplying energy or plant nutrients for the production of mineral fertilizers (e.g. oil and phosphate rock). The aggregation of all resources independent of their functions to equivalency values (like antimony-equivalents in Guinée, 2001) is questionable, because for example neither phosphorus nor coal is functionally equivalent to the semi-metal antimony, i.e. phosphorus and coal cannot replace each other. However, according to the general LCA methodology, the aggregation to impact categories (= characterization) "should be based on scientific knowledge about environmental processes" (Consoli et al., 1993). Transferred to resource consumption that means the characterization of different resources should consider their function. To aggregate them would only be sensible, if these resources are actually equivalent to each other (e.g. oil, gas, and coal as fossil fuels).

This study suggests to assign and to aggregate abiotic resources into separate impact sub-categories according to their main function (e.g. oil, coal and gas to fossil fuels expressed in MJ). If a resource is functionally unique (like phosphate rock or potash), its consumption and the resulting scarcity should be treated as a separate environmental problem and thus makes up its own sub-category. This approach is consistent with the problem-oriented aggregation of emissions into different impact categories (e.g. CO_2, CH_4 and N_2O to climate change expressed in CO_2-equivalents or SO_2, NOx and NH_3 to acidification expressed in SO_2-equivalents).

Following the characterization step, normalization and weighting are steps, which can be applied to aggregate the different resource-related impact sub-categories into one summarizing indicator for the depletion of abiotic resources. Normalization means to divide the resource consumption of the specific product under investigation by the total yearly consumption of the respective resource in a defined reference region, as for instance in Europe. The result shows to what extent the specific product contributes to the total European consumption of a specific resource. Whereas normalization is relatively straightforward, the weighting step is always controversially discussed, since it cannot be based solely on scientific facts but always involves subjective assumptions and values (Hertwich & Pease, 1998; Marsmann et al., 1999).

Time periods, for which a resource should at least be available, have been used in this study to determine tolerable annual extraction rates. These tolerable annual extraction rates employ the “distance-to-target” principle for the calculation of weighting factors (Chapter VI). The choice of the time periods influences the weighting factors. However, as long as no internationally agreed targets on the protection of reserves of abiotic resources have been defined, any definition of a time period, for which a resource should last, is arbitrary. In this study different time scales (100, 300, 1000 years) have therefore been used to calculate weighting factors (Chapter IV). The 100 years target is suggested as default, because it may represent a realistic scenario for the substitution or recycling of abiotic resources.

Another point that needs to be discussed is the use of data on reserves of the resource for the calculation of weighting factors. On the one hand it is important to consider the reserve of a resource, because it determines its scarcity and thus its future availability. On the other hand, concrete data on reserves of minerals and in particular of fossil fuels are often a point of criticism, because reserves can be defined in different ways and data are often supposed to be uncertain because of the continuous discovery of new reserves (Guinée & Heijungs, 1995). This study suggests using the “proven reserve” for fossil fuels, as it is defined by WEC (1998), and the very similarly defined “reserve” for minerals (USGS, 2001) have been chosen. Both reserves include that part of the materials, “that geological and engineering data demonstrate with reasonable certainty to be recoverable in future years from known reservoirs under existing economic and operating conditions” (EIA, 2000). The main reason for selecting this kind of reserves instead of other reserve data is that their definition corresponds best with the common description of a resource, which is defined as “a concentration of naturally occurring ... material ... in such form and amount that economic extraction ... is currently or potentially feasible” (USGS, 2001).

Another definition of reserves is the “reserve base”. The reserve base is defined as that part of a resource that meets specific minimum physical and chemical criteria and includes also those resources, which are only marginal economically or even currently sub-economically exploitable (USGS, 2001). In LCA potential future developments, such as improved extraction techniques for low-quality resources are usually not considered. As the use of the reserve base resources is dependent on such further technical development, these reserves seem to be not appropriate for the use in LCA. The same applies to the use of the “ultimate reserve”, which is used in the CML method (Guinée, 2001) and “estimated by multiplying the average concentrations of chemical elements in the earth's crust by the mass of the crust" (Guinée & Heijungs, 1995). This reserve definition comprises the total deposits of an element in the earth's crust independently from its concentration and thus, is not at all equivalent to what is commonly meant by a resource.

Although having the uncertainty of data in mind, the proven reserve appears to be the most appropriate reserve definition to be used for the weighting of abiotic resources.

2.3 Assessment of the environmental impacts of land use

A need has been identified to improve the existing impact assessment procedures for the environmental impacts of land use (Chapter V). Land use describes in LCA the environmental impacts of occupying, reshaping and managing land for human purposes (e.g. arable farming, housing, and traffic). A major environmental consequence of this anthropogenic land use is a decreasing availability of habitats and thus a decreasing diversity of wildlife. Basically two aspects of land use determine the environmental impact, which are (1) the size of an area used for a certain time and (2) the type or intensity of land use. The size of an area under use for a certain time can be directly measured as a physical quantity (e.g. in m^2*year). However, to evaluate the impact of different types of land use (e.g. sealed urban area vs. extensive meadow) is much more controversial. Current methods mainly base the impact assessment of land use on the number of species determined for a specific land use type compared to the number of species in a natural reference situation (Goedkoop & Spriensma, 1999; Köllner, 2000). These approaches encounter two main problems. First, it is problematic to determine a reference situation on the basis of a single indicator such as species diversity. For instance Goedkoop & Spriensma (1999) and Köllner (2000) take the species richness of Swiss Lowlands as a reference. Species numbers of vascular plants found on areas used for different human purposes have been compared to this reference situation in the Swiss lowlands to derive characterisation factors, which are then assumed to be valid for the whole of Europe. However, within Europe the number of plant species per area already naturally varies by factor 10-15 (200 – 3000 species per 10,000 km^2, BfN, 1999). Therefore, the data for Swiss Lowlands (270 species) cannot be representative for Europe. Furthermore, not only the number, but also the structure of the species community (e.g. share of indigenous and introduced species, or "red-list" and ubiquitous species) is decisive for the assessment of the impact of human land use (Kretschmer et al., 1997). Consequently, "the number of species is an indicator of limited value for the ecological integrity of landscapes" (Kretschmer et al., 1997).

The impact assessment method for land use, which is proposed in this study, treats "naturalness" as a resource. It is assumed that the utilization of land for human purposes leads to a reduced availability of this resource. Besides the size of an area under use for a certain time, it is important to determine the potential of different types of land use to reduce the resource "naturalness". In other words, it is important to determine how much naturalness is left on an area if used for different human purposes.

In order to determine the share of naturalness remaining as a result of different land use types the Hemeroby concept has been applied. According to Kowarik (1999), "Hemeroby is a measure for the human influence on ecosystems". The level of Hemeroby depends on the degree of human impacts that prevent an area from developing towards a natural endpoint situation (Kowarik, 1999). This natural endpoint situation describes the reference to which any modified situation is compared.

In this study the Hemeroby concept has been applied to assess the intensity of different land use types and their potential to degrade the naturalness of land under use. A main advantage of the Hemeroby concept is that the description of the intensity of land use is not based on a single, eventually misleading indicator like species variety. Hemeroby is rather an integrated, descriptive measure of different human influences, which prevent a system from developing towards a situation without any anthropogenic influence (Rühs, 2001). The description of Hemeroby levels as given in the scientific literature (Sukopp, 1972; Sukopp & Blume, 1976; Grunicke et al., 1999; Kowarik, 1999 and Rühs, 2001) provides an independent frame, which makes it possible to assign Hemeroby levels to specific land use types (Chapter V).

Furthermore, the new impact assessment approach suggests assessing land use for the different biogeographic regions of Europe separately. This separation into ecologically homogenous regions is important, because of the great spatial diversity of the resource "nature" throughout Europe. Preserving a high level of naturalness in the Alps does not compensate for degrading Mediterranean forests, as both regions show very different environmental conditions (climate, soil, water) leading to different types of ecosystems. The proposed method could even be improved, if land use data for smaller and therefore ecologically more homogeneous units (e.g. biotope types) could be identified. Unfortunately sufficient land use data are currently not available on the level of biotope types.

Up to now the new method enables to estimate to what extent a specific land use type reduces the naturalness of an area. The result of this characterization step is a quantification of the reduction in natural land due to a specific anthropogenic land use type expressed in m^2*year (Chapter V).

Similar to the impact assessment of resource consumption (Chapter IV), a normalization step follows the characterization in order to evaluate the relevance of the land use impacts in comparison to a reference value, e.g. the total land use impacts in European biogeographic regions. Separate land use normalization values for each biogeographic region have been calculated. This calculation is based on land cover data published by the European Topic Centre on Land Cover (Satellus, 2000).

In order to enable the calculation of an aggregated environmental indicator (Chapter VI), which includes the different environmental effects like climate change, acidification and also land use, it is necessary to weight these effects with regard to their potential to harm the environment. This study deals with the weighting of land use impacts. For this weighting the distance-to-target principle is chosen (Chapter VI). As no generally agreed target on a tolerable land use intensity in Europe exists, the current land use situation has been assumed to be acceptable and thus a weighting factor of 1 is proposed for land use.

Just from an environmental point of view, this assumption may not be justified, because the intensification of land use has already led to a decrease in the diversity of habitats and species (EEA, 1998a; UNEP, 2000). However, weighting factors based on environmental targets set by international conventions (e.g. “Kyoto protocol”, UN-FCCC, 1998; “Convention on Long-Range Transboundary Air Pollution”, UN-ECE/CLRTAP, 1999) usually comprise more than only the environmental dimension of the impacts. These conventions are a result of long discussion processes between science, economy, and policy and can therefore be regarded as a compromise considering all elements of sustainability, i.e. environmental, economic, and social aspects. For land use it has to be considered that a certain level of land utilisation and consequently a reduced naturalness must be accepted because of important human requirements. The needs to produce food and to provide living space for humans are most important additional dimensions of land use.

Given the trends of a growing population, mobility, and urbanisation, the competition between different types of land use (e.g. nature reserves vs. agricultural land vs. urban area) will certainly increase in future (FAO, 2000). Since land is a strictly limited resource, a most efficient use of land for whatever purpose is beneficial and should be given highest priority in LCA. LCA studies in the agricultural sector shall therefore choose a product related functional unit (e.g. 1 ton of cereal grain) instead of an area related functional unit (1 ha under cultivation) in order to consider possible differences in the land use efficiency.

2.4 Development of a new LCA method specifically tailored to arable crop production

Currently available LCA approaches have been primarily designed for industrial applications (BUWAL, 1998; Goedkoop, 1995; Goedkoop & Spriensma, 1999; Guinée et al, 2001; Heijungs et al., 1992a; Steen, 1999). As shown in Chapter II the application on arable crop production systems reveals some problems (missing integration of important environmental impacts, inconsistent impact assessment of some impacts or questionable value choices; see Chapter II). Other LCA tools like the Eco-indicator'99 (Goedkoop & Spriensma, 1999), the EPS method (Steen, 1999), the CML approaches (Heijungs et al., 1992; Guinée et al., 2001) or the Swiss Eco-point model (BUWAL, 1998) also show methodological problems, some of which are specific to the application on arable systems others are more general (see Chapter VI).

Therefore, it was the main objective of this study to develop a LCA approach, which is specifically suitable for the environmental analysis of arable crop production systems. This new LCA method is based on the general LCA methodology given by ISO (1997) and SETAC (Consoli et al., 1993). A major advantage of this approach is the integration of all impact categories relevant to agricultural crop production. For the impact categories "land use" and "resource consumption" new impact assessment procedures described in Chapters IV and V are integrated. After analysis of all available methods, the currently best available aggregation methods have been chosen for climate change, aquatic and terrestrial eutrophication, toxicity and acidification. The impact categories "depletion of the stratospheric ozone layer" and "formation of tropospheric photo-oxidants" have proven not to be relevant to agricultural crop production systems (Chapter II). This is because of the fact that usually no (stratospheric ozone depletion) or only negligible low emissions (photo-oxidants) are released from crop production. With regard to the impact category "toxicity" only heavy metal emissions to soil due to the application of organic and inorganic fertilizers are taken into account in the suggested approach. Potential emissions of plant protection substances and their possible toxic effects on natural ecosystems and humans are not considered. Available characterization methods for human and eco-toxicity concentrate on toxic substances other than pesticides and do not include the currently available plant protection agents. Because plant nutrition is the focus of this study, the possible toxic impacts of agro-chemicals have been excluded from this LCA approach.

However, the proposed LCA method enables a comprehensive analysis of all other environmental impacts connected to arable farming products. This is of particular interest with regard to an increasing demand on indicator systems with the aim to measure and compare the environmental impact of agricultural production systems.

The LCA-specific holistic view on entire production systems allows detecting environmental “hot-spots” connected to the analyzed product. Investigations on the environmental impact of wheat production for instance revealed that the main contribution to the total environmental burden of the production system is due to on-field activities (e.g. fertilizer application), whereas the production and transportation of farming inputs shows a much smaller contribution (Küsters & Jenssen, 1998). Depending on the nitrogen (N) management the environmental “hot-spots” may be eutrophication (e.g. if N application rates are exceeding the crop demand), acidification (e.g. if urea or ammonium-containing N fertilizers are used) or climate change (e.g. at reduced N rates using nitrate-based N fertilizers, Chapters II and VII). From this interpretation of the LCA results efficient measures to improve the overall environmental performance of arable crop production can be suggested. The proposed method can be further used to support decisions upon the choice of alternative products or processes from an environmental point of view.

For this latter aspect, especially the weighting step should be seen as a valuable and important interpretation tool. If weighting is not performed within LCA, users of LCA studies will tend to weigh the system's contribution to different environmental effects on their own. Instead of this individual subjective way of weighting, a set of generic, study-independent weighting factors, as they are for instance proposed in this study, helps the user of an LCA study to interpret complex environmental data sets on a more transparent and documented basis.

However, the main challenge with weighting is that a comparative evaluation of environmental impacts on humans, ecosystems and resources can hardly be based solely on natural science. Natural science is necessary to describe and to quantify the single effects and their impact on the environment during the impact assessment (e.g. the different potential of nutrients to contribute to the eutrophication of aquatic and terrestrial ecosystems). The challenge of the final evaluation of various damages to the environment (e.g. decreasing species diversity due to land use vs. rising sea levels due to climate change) is to integrate natural science with society values and therefore needs social consensus. Different weighting approaches have been developed, which can be basically assigned to four groups: *proxy*, *panel*, *monetary*, and *distance-to-target* approaches. In the following the advantages and disadvantages of these weighting methods will be discussed.

One option to derive weighting factors is to employ *proxy approaches*, which use for instance the accumulated energy consumption (Giegrich et al., 1995) or the total material input (Schmidt-Bleek, 1993) as a representative (proxy) for the total environmental impact. This procedure is straightforward, transparent and easy to operate. However, inputs are often not representative for outputs, i.e. the output of toxic substances does not necessarily need high inputs. Therefore, this kind of weighting does not comply with the goal of LCA, since only part of the total environmental impact of a product or process is considered.

Weighting factors based on *panels* (e.g. Landbank, 1994; Goedkoop & Spriensma, 1999) are a result of questioning a selected group of people (e.g. experts, representatives) about their evaluation of different environmental impacts. An advantage of this way of deriving weighting factors is that an explicit integration of different societal groups and opinions is possible and an open discussion may lead to good transparency. On the other hand, such a survey can be influenced by personal priorities and perceptions of the chosen panel members or by the way they are interviewed (Landbank, 1994). Furthermore, the panel members may be not representative for all social groups (Goedkoop & Spriensma, 1999).

In *monetary methods* (e.g. Steen, 1999) cash values are assigned to environmental impacts by using for example market prices for resources (e.g. fossil fuels) or willingness-to-pay surveys for externalities like decreasing biodiversity. The main problem with monetary methods is the mixture of varying weighting approaches. Real and virtual market prices together with costs for the technical avoidance or mitigation of environmental impacts (acidification, toxicity) are employed to calculate a common monetary unit for all impacts. However, these varying methods include different problems. For instance real market prices for resources may be not only determined by the scarcity of the resource, but additionally by various economical or political considerations (e.g. by monopoly situations, price agreements between competitors or governmental subsidies). Willingness-to-pay surveys are basically similar to panel approaches and thus show the same difficulties; because again a selected group of people is asked about their attitude towards specific environmental problems like decreasing biodiversity and the price they would be willing to pay in order to reduce the problem.

In *distance-to-target approaches* (Goedkoop, 1995; BUWAL, 1998) weighting factors are calculated for each impact category by comparison of the current extent of an environmental impact with a defined target value for the same impact. The quotient of both values gives the weighting factor (Chapter II and VII). The higher this factor the worse is the respective environmental impact. However, a sound definition of target values for the different environmental effects is an important prerequisite for the calculation of acceptable weighting factors. The major advantage of this weighting procedure is the possibility to integrate scientific knowledge about environmental effects and damages together with social values and priorities by the choice of appropriate target values. International agreements like the UN-ECE emissions reduction targets to abate acidification, eutrophication and photo-oxidant formation (UN-ECE/CLRTAP, 1999) or the Kyoto protocol (UN-FCC, 1998) provide such environmental targets. These conventions are a result of intensive discussion processes between science, economy, and policy and can therefore be seen as a good representation of the society's view on these environmental problems. Consequently, in this study new weighting factors have been calculated based on the distance-to-target principle using widely agreed international targets for the impact categories as far as available and most recent data representing the current status of the respective environmental effects.

However, in contrast to other LCA methods (e.g. Goedkoop, 1995; Goedkoop & Spriensma, 1999; Steen, 1999), this study suggests two separate indicators for (a) resource depletion (Resource Depletion Index = RDI) and (b) impacts on natural ecosystems and human health (Environmental Index = EcoX). This separation is important because the problems related to the depletion of abiotic resources are substantially different to those related to the other impact categories. The impact categories other than the depletion of abiotic resources have direct effects on either natural ecosystems (land use, acidification, eutrophication, eco-toxicity), or human health (human toxicity), or even on both (climate change). In contrast, the depletion of abiotic resources, i.e. the decreasing availability of raw materials for future generations has no direct impacts on human health or the shape of natural ecosystems. The environmental impacts associated to the extraction and processing of resources (e.g. land use, emissions or effluents) are considered in the respective impact categories. Resources itself like fossil fuels or phosphate rock rather have an intrinsic value for humans, as they substantially contribute to development and wealth creation (e.g. through mobility and nutrition). Therefore, the availability of abiotic resources for coming generations is more an economic and social issue than an environmental problem. This is the reason why this LCA method separates the aggregated resource depletion indicator (RDI) from the aggregated environmental indicator (EcoX). To aggregate these two indicators would mean to calculate a kind of a sustainability indicator. The development of a sustainability indicator would be an ambitious goal and should certainly comprise more economic and social aspects besides resource depletion (e.g. income, employment, prices, food security and quality, rural development etc.).

2.5 Summary of the methodological contributions

The contributions to the methodological development of LCA proposed in this study concern the inventory phase (Chapter III) as well as the impact assessment step (Chapters IV, V, VI). Figure 1 gives an overview about the single methodological contributions and how they fit into the general LCA concept.

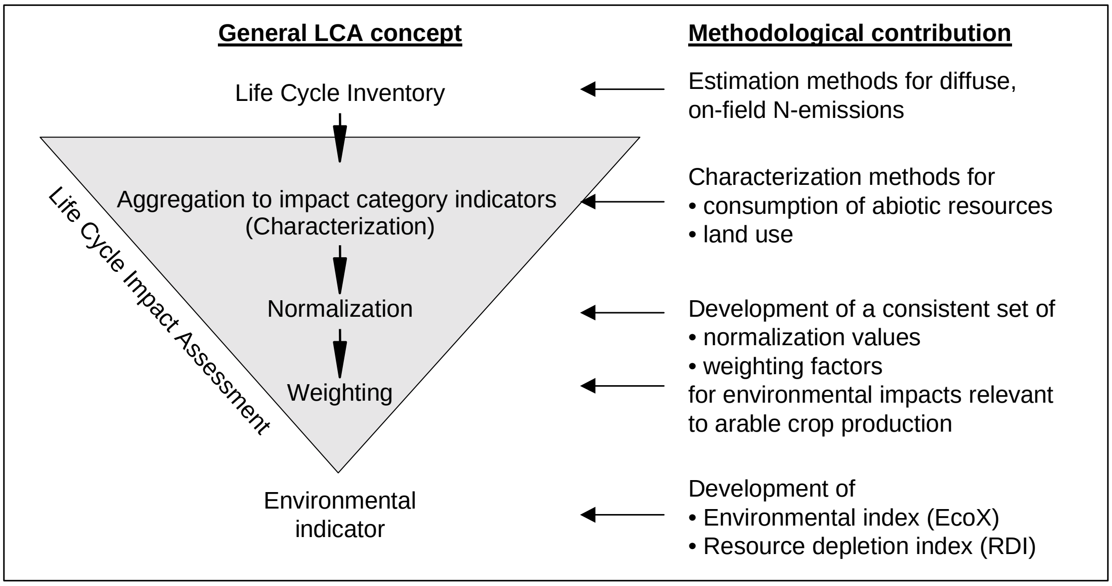

Figure 1: General LCA concept and methodological contribution proposed in this study

<u>Summarizing conclusions:</u>

(1) Diffuse, on-field emissions of nitrate (via leaching), ammonia (via volatilization) and nitrous oxide (via denitrification) often are particularly important in LCA studies dealing with arable crop production. Structured estimation procedures considering some decisive parameters are an appropriate measure in order to derive sound, study-specific estimates of these highly variable emissions as an input to LCA studies.

(2) Arable farming contributes to the depletion of specific abiotic resources like phosphate rock, potash or fossil fuels. The impact assessment of the consumption of these resources should consider their different function. An aggregation to one summarizing resource depletion indicator is therefore only possible after an explicit normalization and weighting step.

(3) Arable farming utilizes huge quantities of land for crop production. An assessment of the environmental impacts of land use in LCA has to include two pieces of information: (a) the size of an area used for a certain period of time and (b) the potential of a specific land use type to degrade the naturalness of the area under use.

(4) The Hemeroby concept has been developed in order to evaluate the degree of naturalness of land area. The Hemeroby concept is therefore suitable to assess the potential of different land use types to degrade the naturalness of an area under use within LCA.
(5) A comprehensive LCA method capable to analyze the environmental impacts of arable crop production should include the following impact categories: consumption of abiotic resources, land use, climate change, toxicity, acidification, and eutrophication.
(6) The weighting of the different impact categories can be realized by applying the distance-to-target principle. Internationally agreed environmental targets should be used as far as possible in order to represent a consensus of science, economy and society.
(7) Even after weighting, the indicator values calculated for resource depletion on the one hand and those for the actual environmental effects should not be aggregated because of the substantial differences between these two groups of impacts. The reduced availability of abiotic resources for coming generations is an economic and social problem. All other impact categories are actual environmental issues, because they directly affect the quality of natural eco-systems and human health.

3. Application of the new LCA method to investigate the environmental impacts of different nitrogen fertilizer rates

As the final step the new LCA method has been applied to investigate the environmental impact of winter wheat production at different N fertilizer rates. The yield response to the different N rates is taken from a long-term field trial (Broadbalk Experiment, Rothamsted, UK). The LCA study takes into account the contribution of the wheat production to the effects *resource depletion*, *land use*, *climate change*, *toxicity*, *acidification*, and *eutrophication*.

After an inventory of all single emissions and consumption of resources, indicator values for the environmental effects are calculated (characterization). Furthermore, the share of the wheat production in the total environmental effects in Europe is determined (normalization). Finally, the indicator values are evaluated and aggregated further by multiplication with weighting factors, which represent the potential of each impact to harm the environment.

This study clearly illustrates the advantages of the LCA methodology when evaluating the environmental preferences of different production intensities in arable farming. If for instance just the release of greenhouse gases would have been chosen for this evaluation, the most extensive production system performs best (Chapter VII, Fig. 8). When taking fossil fuel consumption as the only indicator, medium N rates would be favorable (Chapter VII, Fig. 6). However, both approaches would totally neglect, that the use of land is most efficient in intensive treatments with high yields per area (Chapter VII, Fig. 7). In this respect the advantage of LCA is that all relevant impacts are considered and evaluated simultaneously.

As the final result of this LCA study two separate indicators were calculated, one of which represents the contribution to resource depletion (RDI) and the other comprises the impacts on natural eco-systems and human health (EcoX).

The EcoX values calculated for the treatments receiving 48, 96, 144 or 192 kg N/ha remain between 0.16 and 0.22 per ton of grain (Chapter VII, Fig. 16). The treatments receiving 0, 240 or 288 kg N/ha show 50 to 150% (0 kg N/ha = 0.33, 288 kg N/ha = 0.55) higher EcoX values compared to the economic optimum treatment (192 kg N/ha). This result indicates that in efficient crop production systems economic and environmental aspects are not necessarily in conflict.

The wheat production shows two environmental hotspots, which are *land use* and *aquatic eutrophication*. Thus, under conditions as described in the study the greatest potential to minimize the environmental impact per ton of grain is to achieve high yields (i.e. a high land use efficiency) and at the same time low NO_3 leaching rates, which are most responsible for aquatic eutrophication (Chapter VII, Fig. 12). In plot 8 (144 kg N/ha) the aggregated EcoX value for these two competing aspects is lowest. Consequently, other impacts than land use and aquatic eutrophication get a higher relative importance in this treatment compared to the other plots (Chapter VII, Fig. 16).

The differences in the *acidification* potential are mainly determined by the NH_3 emissions (Chapter VII, Fig. 10). Since in the analyzed system ammonium nitrate is used as N fertilizer, the NH_3 emissions are low compared to the use of other mineral and organic fertilizers (ECETOC, 1994). Other calculations have shown that for instance the use of urea or organic fertilizers (e.g. slurry) as N sources results in clearly higher acidification potentials (Chapters II and III; Kuesters & Jenssen, 1998).

For *greenhouse gases* the picture is different. Part of the global warming potential could be avoided by choosing an ammonium or urea based N fertilizer. Relevant amounts of N_2O are emitted during the production of nitric acid, which is part of the ammonium nitrate production. However, as already mentioned a switch to urea or another non-nitrate fertilizer would lead to higher contributions to acidification and eutrophication. Ideally, the N_2O emissions during nitric acid production could be mitigated, what is technically possible (Laegreid et al., 1999).

From this LCA case study it can be concluded that a good environmental performance in wheat production can be achieved

- by maintaining yields close to the optimum, i.e. using land most efficiently,

- by applying nitrogen according to crop demand, in order to minimize NO_3 leaching,
- by using nitrogen fertilizers with low NH_3 volatilization rates (e.g. ammonium nitrate), in order to keep the acidification and terrestrial eutrophication potentials low, and
- by reducing N_2O emissions from nitrate fertilizer production (filter techniques), in order to reduce the global warming potential.

For the problem of resource depletion, this case study clearly reveals that the impact of arable farming on decreasing availability of exploitable phosphate (P) rock resources is by far greater than that on decreasing availability of fossil fuel resources (Chapter VII, Fig. 13). Since P reserves are scarce and P is essential in crop production, a responsible use of P resources (e.g. by considering the P status of the soil in P fertilization and P fertilization according to P removal by crops) is important. Furthermore, the recycling of phosphates (e.g. contained in sewage sludge and slurry) should be realized as far as possible. The share of agriculture in total energy consumption is comparably low (4% in Europe; WRI, 2000), so that efforts to save energy may be more efficient in other sectors like industry (33%), households (27%) or transportation (23%). However, the most energy efficient production intensities (96 and 144 kg N/ha) are also favorable from an environmental point of view (Chapter VII, Fig. 16).

4. Concluding remarks

LCA has proven to be an appropriate concept for the evaluation of the environmental performance of arable crop production systems. However, in order to convert this general concept into an operational LCA method, which is suitable for the analysis of arable systems, new methodological developments have been proposed. The resulting LCA method enables to

(1) determine and evaluate the environmental impacts relevant to arable crop production with a special focus on plant nutrition aspects under consideration of the entire production system,
(2) trace back the various environmental impacts to their sources and on that basis to suggest options for environmental improvement, and
(3) compare the environmental performance of alternative arable crop production systems in a transparent way.

The proposed LCA method shall therefore contribute to more traceability and transparency in the food production chain.

IX. Summary

The main objective of this study was to develop a Life Cycle Assessment (LCA) method, which enables the evaluation of arable farming products (i.e. crops) or production systems (e.g. wheat production at different production intensities) from an environmental point of view. A special focus was on the environmental impacts associated with plant nutrition. The main results of this study are:

(1) A LCA case study on the environmental impacts of different mineral nitrogen fertilizers in sugar beet production revealed that the LCA methodology is principally suitable to investigate and evaluate the environmental impacts associated to the production of arable crops. However, the application of currently available LCA tools (e.g. Eco-indicator'95; Goedkoop, 1995) on an entire system of arable farming showed shortcomings. An example is the missing consideration of specific resources, land use or nutrient emissions, which are particularly important for arable production.

(2) For arable production systems it is particularly important for LCA studies to derive reliable inventory data on diffuse, on-field emissions of ammonia (NH_3), nitrous oxide (N_2O) and nitrate (NO_3). Therefore, the present study suggests different estimation methods for NH_3 emissions: from organic (Horlacher & Marschner, 1990) and from mineral fertilizers (ECETOC, 1994). A function derived by Bouwman (1995) is selected to calculate the N_2O emissions. A method developed by the German Soil Science Association (DBG, 1992) is adopted to determine potential NO_3 emissions. These estimation procedures consider decisive soil, climate and management parameters, which are appropriate to derive sound, study-specific estimates of the highly variable on-field nitrogen emissions.

(3) Arable farming consumes considerable amounts of mineral (plant nutrition) and fossil fuel resources. The consumption of such abiotic resources should therefore be addressed in LCA studies on arable production. In this study a new impact assessment approach to abiotic resource consumption was developed that treats the consumption of resources, which are used for different purposes, as separate environmental problems. Normalization and weighting procedures enable the aggregation of those functionally different resources into one resource depletion indicator.

(4) Similar to the consumption of abiotic resources also for the environmental impacts of land use a new impact assessment method was developed. To assess the environmental impacts of land use the developed LCA method includes two pieces of information: (a) the size of an area used for a certain period of time and (b) the intensity of different land use types. Whereas the first aspect can be directly expressed as a physical quantity, the latter aspect needs an appropriate indicator. The Hemeroby concept provides such an indicator, since this concept was specifically developed in order to evaluate the level of naturalness of land area. Hemeroby is a measure for the human influence on ecosystems, which defines the level of naturalness of different land use types (e.g. urban area or extensive pasture) according to their deviation from a natural reference situation. Therefore, in this study the Hemeroby concept was integrated into a new impact assessment method for land use impacts.

(5) The developed LCA method is specifically suitable to investigate arable crop production systems and considers the following environmental effects: consumption of abiotic resources, land use, climate change, toxicity, acidification, and eutrophication. In addition to the calculation of separate indicators for each environmental effect, an aggregation procedure was developed, which enables the calculation of two summarizing indicators for (a) resource depletion and (b) impacts on natural eco-systems and human health. The weighting of the different impact categories was realized by a comparison of the current status of each effect with defined target values for the respective effect ("distance-to-target principle"). Internationally agreed environmental targets were employed in order to represent a consensus of science, economy and society as much as possible.

(6) The developed LCA method was tested in a case study in order to investigate the environmental impact of different N fertilizer rates in winter wheat production (Broadbalk Experiment, Rothamsted, UK). This method proved to be capable to determine and evaluate those environmental impacts, which are relevant to arable crop production, and in particular to plant nutrition. The consideration of the entire production system enables to trace back the various environmental impacts to their sources and on that basis to suggest options for environmental improvements. The inclusion of a transparent aggregation procedure makes it possible to compare the environmental performance of the alternative arable crop production systems.

(7) This particular case study revealed that the aggregated environmental impact per tonne of wheat grain increases dramatically at zero N fertilization and at N rates exceeding the crop demand. In the first case inefficient land use was the major problem, whereas in the latter case the main problem was a relatively high contribution to aquatic eutrophication. From reduced to economic optimum N rates the environmental indicator values increased only slightly. At optimum N fertilization (192 kg N/ha) aquatic eutrophication contributed most to the aggregated indicator; terrestrial eutrophication, acidification, climate change and land use show similar contributions. For the problem of resource depletion the consumption of phosphate rock turned out to be the major problem in the analyzed wheat production system.

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XI. Annex

Basic data for LCA studies on arable crop production

Table 1: Energy consumption for extraction, transportation and processing/generation of different sources of energy (based on: BMWi (1995), Menard et al. (1995), Patyk & Reinhardt (1997))

Source of energy	Heat value	Unit	Energy consumption (MJ/MJ)						
			Total	Coal	Lignite	Oil	Gas	Nuclear	Others
Coal, Germany	29,3	MJ/kg	0,05941	0,042	0,00009	0,017	0,00003	0,00021	0,00008
Coal, Netherlands	29,3	MJ/kg	0,06143	0,044		0,017	0,00039	0,00004	
Coal, E-Europe	29,3	MJ/kg	0,0373	0,023		0,0143			
Lignite, Germany	8,5	MJ/kg	0,047		0,047				
Heavy oil, Germany	41	MJ/kg	0,09708	0,0026	0,0026	0,08	0,0091	0,0025	0,00028
Heavy oil, Netherlands	41	MJ/kg	0,07877	0,0034		0,062	0,013	0,00037	
Heavy oil, E-Europe	41	MJ/kg	0,1498	0,0024		0,132	0,013	0,0016	0,0008
Diesel, Germany	42,7	MJ/kg	0,11316	0,0033	0,0033	0,093	0,01	0,0032	0,00036
Diesel, Netherlands	42,7	MJ/kg	0,09489	0,0044		0,075	0,015	0,00049	
Diesel, E-Europe	42,7	MJ/kg	0,1693	0,0022		0,148	0,016	0,0021	0,001
Natural gas, Germany	48,3	MJ/kg	0,0653				0,0653		
Natural gas, Netherlands	48,3	MJ/kg	0,0107				0,0107		
Natural gas, E-Europe	48,3	MJ/kg	0,0504				0,0504		
Electricity, Germany	3,6	MJ/kWh	3,0079	1,42		0,1483	0,2593	1,1094	0,0709
Electricity, Sweden	3,6	MJ/kWh	2,2474	0,0214		0,019	0,0815	1,4751	0,6504
Electricity, UK	3,6	MJ/kWh	2,8502	0,8605		0,3536	0,2444	1,1867	0,205
Electricity, Norway	3,6	MJ/kWh	1,3041	0,0086		0,0165	0,0015	0,0027	1,2748
Electricity, France	3,6	MJ/kWh	3,1127	0,3149		0,0907	0,0546	2,4602	0,1923
Electricity, Italy	3,6	MJ/kWh	3,0604	0,7169		1,5537	0,5126	0,0276	0,2496
Electricity, EU15 average	3,6	MJ/kWh	2,9869	0,6687	0,2437	0,222	0,2951	1,039	0,5184
Steam, Germany average			1,1926	0,0003		1,1062	0,0713	0,0003	0,0145
Steam, Europe average			1,1362	0,0568		0,1704	0,909		

Table 2: Emissions due to fuel use (based on: BMWi (1995), Patyk & Reinhardt (1997))

Fuel type	Heating value	Use	Emission (g/MJ)								
	MJ/kg		CH_4	CO	CO_2	NH_3	N_2O	NOx	Particles	SO_2	NMVOC
Coal	29,3	Heating boiler, W-Europe	0,00038	0,095	93,3		0,0500	0,076	0,0095	0,092	0,0019
Lignite	8,5	Heating boiler, W-Europe	0,00044	0,044	112,0		0,0440	0,065	0,0087	0,057	0,0044
Diesel	42,7	Mining engine, W-Europe	0,00321	0,250	74,4	0,0027	0,0034	0,810	0,0700	0,021	0,1300
Diesel	42,7	Truck	0,00280	0,230	74,4	0,0027	0,0034	0,940	0,0520	0,023	0,1100
Diesel	42,7	Train	0,00370	0,420	74,4	0,0027	0,0034	1,290	0,0700	0,023	0,1500
Diesel	42,7	Barge	0,00280	0,280	74,4	0,0027	0,0034	1,410	0,0470	0,023	0,1100
Diesel	42,7	Tractor, average use pattern	0,00480	0,400	74,4	0,0027	0,0034	0,840	0,0890	0,023	0,2000
Natural gas	48,3	Heating boiler, W-Europe	0,00251	0,014	55,2		0,0010	0,028	0,0001	0,000	0,0025
Natural gas	48,3	Gas turbine (e.g. pipeline)	0,00420	0,084	55,2		0,0025	0,290	0,0042	0,000	0,0084
Oil, heavy	41,0	Heating boiler, W-Europe	0,00300	0,043	78,8		0,0020	0,110	0,0110	0,490	0,0030
Oil, heavy	41,0	Ship	0,00150	0,220	77,6	0,0028	0,0035	2,050	0,1500	1,960	0,0600
LPG (light petroleum gas)	46,4	Heating boiler, W-Europe			64,7			0,050	0,0021	0,003	0,0000

Table 3: Energy consumption due to extraction and processing of fertilizer raw materials (based on: Davis & Haglund (1999), Kongshaug (1998), Patyk & Reinhardt (1997))

Raw material		Production line	Nutrient content	Energy consumption									
				Total	Coal		Diesel		Gas		Electricity		Steam
				GJ/t	kg/t	MJ/t	kg/t	MJ/t	kg/t	MJ/t	kWh/t	MJ/t	MJ/t
Dolomite		Europe, average	15% MgO	0,8			9,4	400			111,1	400	
Kieserite	$MgSO_4$	Germany, average	26% MgO, 21% S	1,1					22,8	1100			
Limestone	$CaCO_3$	Germany, average	55% CaO	0,2			0,7	30			50,6	182,2	
Phosphate rock	P rock	Dry sedimentary rock, e.g. North Africa	32% P_2O_5	0,1			0,6	25			6,1	22	53
Phosphate rock	P rock	Apatite rock, e.g. Russia	32% P_2O_5	0,9			5,3	225			55	198	477
Potassium chloride	MOP	BAT, based on high quality sylvinite salt	60% K_2O	1,5	4,6	135	1,8	75	25,5	1230	16,7	60	
Potassium chloride	MOP	Europe, average	60% K_2O	3,0	9,2	270	3,5	150	50,9	2460	33,3	120	
Potassium chloride	MOP	Old technique (30 years ago)	60% K_2O	4,0	12,3	360	4,7	200	67,9	3280	44,4	160	
Sulphur	S	Mining, Europe, average	100% S	1,3			4,0	170			35	126	1000
Sulphur	S	Frasch process, Europe, average	100% S	5,1							35	126	5000
Sulphur	S	Claus process, Europe, average	100% S	0,0									

Table 4: Emissions and waste generation due to extraction and processing of fertilizer raw materials (<u>only</u> process-specific, <u>not</u> energy-related emissions) (based on: Kongshaug (1998), Patyk & Reinhardt (1997))

Raw material		Production line	Emission (kg/t) Particles	Waste (t/t) Waste rock
Dolomite		Europe, average	0,5	
Kieserite	$MgSO_4$	Germany, average	0,5	
Limestone	$CaCO_3$	Germany, average	0,5	
Phosphate rock	P rock	Dry sedimentary rock, e.g. North Africa	0,072	4
Phosphate rock	P rock	Apatite rock, e.g. Russia	0,072	4
Potassium chloride	MOP	BAT, based on high quality sylvinite salt	0,5	
Potassium chloride	MOP	Europe, average	0,5	
Potassium chloride	MOP	Old technique (30 years ago)	0,5	
Sulphur	S	Mining, Europe, average		
Sulphur	S	Frasch process, Europe, average		
Sulphur	S	Claus process, Europe, average		

Table 5: Input of fertilizer raw materials during the production of fertilizer intermediates (based on: Davis & Haglund (1999), Kongshaug (1998), Patyk & Reinhardt (1997))

Fertilizer intermediate		Production line	Nutrient content	Input of raw materials and intermediates (t/t)					
				P rock	S	NH_3	HNO_3	48% H_3PO_4	H_2SO_4
Ammonia	NH_3	BAT, gas-based	82% N						
Ammonia	NH_3	Europe, average, gas-based	82% N						
Ammonia	NH_3	Old technique (30 years ago)	82% N						
Ammonia	NH_3	Germany, average	82% N						
Ammonia	NH_3	E-Europe, gas-based	82% N						
Ammonia	NH_3	E-Europe, oil-based	82% N						
Ammonia	NH_3	E-Europe, coal-based	82% N						
Nitric acid	HNO_3	BAT, dual pressure technique	60% HNO_3			0,28			
Nitric acid	HNO_3	Europe, average	60% HNO_3			0,28			
Nitric acid	HNO_3	Old technique (30 years ago)	60% HNO_3			0,28			
Phosphoric acid, 48%	H_3PO_4	Europe, average	48% P_2O_5	1,77					1,46
Phosphoric acid, 54%	H_3PO_4	BAT, hemihydrate process	54% P_2O_5	1,77					1,46
Phosphoric acid, 54%	H_3PO_4	Hemihydrate process, avg.	54% P_2O_5	1,77					1,46
Phosphoric acid, 54%	H_3PO_4	Dihydrate process, avg.	54% P_2O_5	1,77					1,46
Phosphoric acid, 54%	H_3PO_4	Europe, average	54% P_2O_5	1,77					1,46
Phosphoric acid, 54%	H_3PO_4	Old technique (30 years ago)	54% P_2O_5	1,77					1,46
Sulphuric acid	H_2SO_4	BAT (S -> H_2SO_4)	98% H_2SO_4		0,33				
Sulphuric acid	H_2SO_4	Europe, average (S -> H_2SO_4)	98% H_2SO_4		0,33				
Sulphuric acid	H_2SO_4	Old technique (S -> H_2SO_4, 30 years ago)	98% H_2SO_4		0,33				
Sulphuric acid	H_2SO_4	Germany, avg. (S -> H_2SO_4)	98% H_2SO_4		0,33				
Sulphuric acid	H_2SO_4	Recycling of waste acid	98% H_2SO_4						
Ammonium nitrate, liquid (for UAN)	UAN-AN	BAT	33,5% N			0,21	0,78		
Ammonium nitrate, liquid (for UAN)	UAN-AN	Europe, average	33,5% N			0,21	0,78		
Ammonium nitrate, liquid (for UAN)	UAN-AN	Old technique (30 years ago)	33,5% N			0,21	0,78		
Urea, liquid (for UAN)	UAN-Urea	BAT	45% N			0,567			
Urea, liquid (for UAN)	UAN-Urea	Europe, average	45% N			0,567			
Urea, liquid (for UAN)	UAN-Urea	Old technique (30 years ago)	45% N			0,567			
Ammonium phosphate (for NPK)	AP	W-Europe, average	49% P_2O_5, 11% N			0,134		1,02	
Nitro ammonium phosphate (for NPK)	Nitro-AP	W-Europe, average	52% P_2O_5, 8,4% N	1,625		0,102			

Table 6: Energy consumption due to the production of fertilizer intermediates (based on: Davis & Haglund (1999), Kongshaug (1998), Patyk & Reinhardt (1997))

Fertilizer intermediate		Production line	Energy consumption Total	Coal		Oil		Gas		Electricity		Steam
			GJ/t	kg/t	MJ/t	kg/t	MJ/t	kg/t	MJ/t	kWh/t	MJ/t	MJ/t
Ammonia	NH_3	BAT, gas-based	28,3					633,5	30600	55,6	200,0	-2500
Ammonia	NH_3	Europe, average, (EFMA, pers.comm.)	36,0					745,3	36000			
Ammonia	NH_3	Old technique (30 years ago)	38,7					801,2	38700			
Ammonia	NH_3	Germany, average	35,6	61,4	1800	122,0	5000	596,3	28800			
Ammonia	NH_3	E-Europe, gas-based	38,0					786,7	38000			
Ammonia	NH_3	E-Europe, oil-based	41,7			1017,1	41700					
Ammonia	NH_3	E-Europe, coal-based	53,1	1812,3	53100							
Nitric acid	HNO_3	BAT, dual pressure technique	-2,4							9,0	32,4	-2442
Nitric acid	HNO_3	Europe, average	-1,5							9,0	32,4	-1554
Nitric acid	HNO_3	Old technique (30 years ago)	-1,1							9,0	32,4	-1111
Phosphoric acid, 48%	H_3PO_4	Europe, average	2,9							23,9	86,0	2794
Phosphoric acid, 54%	H_3PO_4	BAT, hemihydrate process	1,4							11,1	40,0	1310
Phosphoric acid, 54%	H_3PO_4	Hemihydrate process, avg.	1,9							16,7	60,0	1830
Phosphoric acid, 54%	H_3PO_4	Dihydrate process, avg.	4,1							33,3	120,0	3930
Phosphoric acid, 54%	H_3PO_4	Europe, average	3,8							31,4	113,0	3667
Phosphoric acid, 54%	H_3PO_4	Old technique (30 years ago)	4,9							41,7	150,0	4710
Sulphuric acid	H_2SO_4	BAT (S -> H_2SO_4)	-6,0									-6000
Sulphuric acid	H_2SO_4	Europe, average (S -> H_2SO_4)	-3,0									-3000
Sulphuric acid	H_2SO_4	Old technique (S -> H_2SO_4, 30 years ago)	-1,0									-1000
Sulphuric acid	H_2SO_4	Germany, avg. (S -> H_2SO_4)	-3,6									-3600
Sulphuric acid	H_2SO_4	Recycling of waste acid	5,0									5000
Ammonium nitrate, liquid (for UAN)	UAN-AN	BAT	0,0									
Ammonium nitrate, liquid (for UAN)	UAN-AN	Europe, average	0,3									300
Ammonium nitrate, liquid (for UAN)	UAN-AN	Old technique (30 years ago)	1,0									1000
Urea, liquid (for UAN)	UAN-Urea	BAT	3,1							119,2	429,0	2671
Urea, liquid (for UAN)	UAN-Urea	Europe, average	3,9							148,1	533,0	3367
Urea, liquid (for UAN)	UAN-Urea	Old technique (30 years ago)	4,4							166,1	598,0	3802
Ammonium phosphate (for NPK)	AP	W-Europe, average	0,0									
Nitro ammonium phosphate (for NPK)	Nitro-AP	W-Europe, average	3,3					37,3	1802	273,1	983,0	491

Table 7: Emissions and waste generation due to the production of fertilizer intermediates (only process-specific, not energy-related emissions)
(based on: Davis & Haglund (1999), Kongshaug (1998), Patyk & Reinhardt (1997))

Fertilizer intermediate		Production line	Emission (kg/t)										Waste (t/t)
			CH_4	CO	CO_2	NH_3	N_2O	NOx	Ntot	Particles	Ptot	SO_2	Gypsum
Ammonia	NH_3	BAT, gas-based				0,8		0,9					
Ammonia	NH_3	Europe, average, gas-based				0,8		0,9					
Ammonia	NH_3	Old technique (30 years ago)				0,8		0,9					
Ammonia	NH_3	Germany, average				0,8		0,9					
Ammonia	NH_3	E-Europe, gas-based				0,8		0,9					
Ammonia	NH_3	E-Europe, oil-based				0,8		0,9					
Ammonia	NH_3	E-Europe, coal-based				0,8		0,9					
Nitric acid	HNO_3	BAT, dual pressure technique					6,67	4,2					
Nitric acid	HNO_3	Europe, average					6,67	4,2					
Nitric acid	HNO_3	Old technique (30 years ago)					5,93	4,2					
Phosphoric acid, 48%	H_3PO_4	Europe, average								0,65			2,75
Phosphoric acid, 54%	H_3PO_4	BAT, hemihydrate process								0,65			2,75
Phosphoric acid, 54%	H_3PO_4	Hemihydrate process, avg.								0,65			2,75
Phosphoric acid, 54%	H_3PO_4	Dihydrate process, avg.								0,65			2,75
Phosphoric acid, 54%	H_3PO_4	Europe, average								0,65			2,75
Phosphoric acid, 54%	H_3PO_4	Old technique (30 years ago)								0,65			2,75
Sulphuric acid	H_2SO_4	BAT (S -> H_2SO_4)										6,75	
Sulphuric acid	H_2SO_4	Europe, average (S->H_2SO_4)										6,75	
Sulphuric acid	H_2SO_4	Old tech. (S->H_2SO_4, 30 years ago)										6,75	
Sulphuric acid	H_2SO_4	Germany, avg. (S->H_2SO_4)										6,75	
Sulphuric acid	H_2SO_4	Recycling of waste acid										6,75	
Ammonium nitrate, liquid (for UAN)	UAN-AN	BAT				0			0				
Ammonium nitrate, liquid (for UAN)	UAN-AN	Europe, average				0,092			0,072				
Ammonium nitrate, liquid (for UAN)	UAN-AN	Old technique (30 years ago)				0,53			0,43				
Urea, liquid (for UAN)	UAN-Urea	BAT	0,36	1,32	-733	0,8			0,13				
Urea, liquid (for UAN)	UAN-Urea	Europe, average	0,36	1,32	-733	0,8			0,13				
Urea, liquid (for UAN)	UAN-Urea	Old technique (30 years ago)	0,36	1,32	-733	0,8			0,13				
Ammonium phosphate (for NPK)	AP	W-Europe, average				0,132				0,5	0,009		
Nitro ammonium phosphate (for NPK)	Nitro-AP	W-Europe, average				0,102				0,5	0,166		

Table 8: Input of fertilizer raw materials and intermediates during the production of fertilizers (based on: Davis & Haglund (1999), Kongshaug (1998), Patyk & Reinhardt (1997))

Fertilizer product		Production line	Nutrient content	Input of raw materials and intermediates (t/t)												
				Dolom.	Lime	P rock	MOP	NH_3	HNO_3	48% H_3PO_4	54% H_3PO_4	H_2SO_4	AN, liqu.	Urea, liqu.	AP	Nitro-AP
Ammonium nitrate	AN	BAT	33,5% N					0,21	0,78							
Ammonium nitrate	AN	Europe, average	33,5% N					0,21	0,78							
Ammonium nitrate	AN	Old technique (30 years ago)	33,5% N					0,21	0,78							
Calcium ammonium nitrate	CAN	W-Europe, average	26% N	0,243				0,161	0,596							
Calcium nitrate	CN	BAT, nitro-phosphate process	15,5% N					0,013	0,65							
Calcium nitrate	CN	Europe, average	15,5% N					0,013	0,65							
Calcium nitrate	CN	Old technique (30 years ago)	15,5% N		0,51				0,74							
Diammonium phosphate	DAP	BAT, granulation/pipe reactor	18% N, 46% P_2O_5					0,22		0,96						
Diammonium phosphate	DAP	Europe, average	18% N, 46% P_2O_5					0,22			0,88					
Diammonium phosphate	DAP	Old technique (30 years ago)	18% N, 46% P_2O_5					0,22			0,88					
Monoammonium phosphate	MAP	BAT, granulation/pipe reactor	11% N, 54% P_2O_5					0,13		1,08						
Monoammonium phosphate	MAP	Europe, average	11% N, 54% P_2O_5					0,137			0,97					
Monoammonium phosphate	MAP	Old technique (30 years ago)	11% N, 54% P_2O_5					0,137			0,97					
Triplesuperphosphate	TSP	Europe, average	46% P_2O_5			0,45				0,7						
Singlesuperphosphate	SSP	Europe, average	20% P_2O_5, 12% S			0,626						0,384				
Urea	Urea	BAT	45% N					0,567								
Urea	Urea	Europe, average	45% N					0,567								
Urea	Urea	Old technique (30 years ago)	45% N					0,567								
Urea ammonium nitrate	UAN	Europe, average	32% N										0,457	0,348		
Potassium sulphate	SOP	Mining and beneficiation	50% K_2O, 18% S													
Potassium sulphate	SOP	BAT, mod. Mannheim process	50% K_2O, 18% S				0,856					0,563				
Potassium sulphate	SOP	Europe, average	50% K_2O, 18% S				0,856					0,563				
Potassium sulphate	SOP	Old technique (30 years ago)	50% K_2O, 18% S				0,856					0,563				
NPK compound, 15-15-15	NPK 15-15-15	Europe, average, mixed acid route	15% N, 15% P_2O_5, 15% K_2O	0,112			0,25						0,332		0,306	
NPK compound, 15-15-15	NPK 15-15-15	Europe, average, nitrophosphate route	15% N, 15% P_2O_5, 15% K_2O	0,103			0,25						0,359			0,288

Table 9: Energy consumption due to the production of fertilizers (based on: Davis & Haglund (1999), Kongshaug (1998), Patyk & Reinhardt (1997))

Fertilizer product		Production line	Energy consumption					
			Total	Gas		Electricity		Steam
			GJ/t	kg/t	MJ/t	kWh/t	MJ/t	MJ/t
Ammonium nitrate	AN	BAT	0,2					150,0
Ammonium nitrate	AN	Europe, average	0,7					700,0
Ammonium nitrate	AN	Old technique (30 years ago)	1,4					1400,0
Calcium ammonium nitrate	CAN	W-Europe, average	0,6					636,0
Calcium nitrate	CN	BAT, nitro-phosphate process	0,8			68,9	248,0	552,0
Calcium nitrate	CN	Europe, average	1,0			86,1	310,0	690,0
Calcium nitrate	CN	Old technique (30 years ago)	3,9			119,2	429,0	3471,0
Diammonium phosphate	DAP	BAT, granulation with pipe reactor	0,4	4,0	192,5	29,2	105,0	52,5
Diammonium phosphate	DAP	Europe, average	0,9	10,2	495,0	75,0	270,0	135,0
Diammonium phosphate	DAP	Old technique (30 years ago)	0,9	10,2	495,0	75,0	270,0	135,0
Monoammonium phosphate	MAP	BAT, granulation with pipe reactor	0,4	4,0	192,5	29,2	105,0	52,5
Monoammonium phosphate	MAP	Europe, average	0,9	10,2	495,0	75,0	270,0	135,0
Monoammonium phosphate	MAP	Old technique (30 years ago)	0,9	10,2	495,0	75,0	270,0	135,0
Triplesuperphosphate	TSP	Europe, average	2,0			61,1	220,0	1780,0
Singlesuperphosphate	SSP	Europe, average	1,4			42,8	154,0	1246,0
Urea	Urea	BAT	3,3			119,2	429,0	2871,0
Urea	Urea	Europe, average	4,1			148,1	533,0	3567,0
Urea	Urea	Old technique (30 years ago)	4,6			166,1	598,0	4002,0
Urea ammonium nitrate	UAN	Europe, average	0,2			61,1	220,0	
Potassium sulphate	SOP	Mining and beneficiation	2,0	41,4	2000,0			
Potassium sulphate	SOP	BAT, modern Mannheim process	2,7	52,0	2511,0	52,5	189,0	
Potassium sulphate	SOP	Europe, average	3,1	59,7	2883,0	60,3	217,0	
Potassium sulphate	SOP	Old technique (30 years ago)	3,4	65,5	3162,0	66,1	238,0	
NPK compound, 15-15-15	NPK 15-15-15	Europe, average, mixed acid route	0,0					
NPK compound, 15-15-15	NPK 15-15-15	Europe, average, nitrophosphate route	0,0					

Table 10: Emissions due to the production of fertilizers (only process-specific, not energy-related emissions)
(based on: Davis & Haglund (1999), Kongshaug (1998), Patyk & Reinhardt (1997))

Fertilizer product		Production line	Emission (kg/t)						
			CH_4	CO	CO_2	NH_3	Ntot	Particles	Ptot
Ammonium nitrate	AN	BAT				0	0	0,07	
Ammonium nitrate	AN	Europe, average				0,092	0,072	0,273	
Ammonium nitrate	AN	Old technique (30 years ago)				0,53	0,43	1,18	
Calcium ammonium nitrate	CAN	W-Europe, average				0,2		0,5	
Calcium nitrate	CN	BAT, nitro-phosphate process				0,0012	0,315	0,164	
Calcium nitrate	CN	Europe, average				0,0012	0,315	0,164	
Calcium nitrate	CN	Old technique (30 years ago)					2,3	0,26	
Diammonium phosphate	DAP	BAT, granulation with pipe reactor				0,216		0,5	0,008
Diammonium phosphate	DAP	Europe, average				0,216		0,5	0,008
Diammonium phosphate	DAP	Old technique (30 years ago)				0,216		0,5	0,008
Monoammonium phosphate	MAP	BAT, granulation with pipe reactor				0,132		0,5	0,009
Monoammonium phosphate	MAP	Europe, average				0,132		0,5	0,009
Monoammonium phosphate	MAP	Old technique (30 years ago)				0,132		0,5	0,009
Triplesuperphosphate	TSP	Europe, average						0,072	0,692
Singlesuperphosphate	SSP	Europe, average						0,072	0,303
Urea[1]	Urea	BAT	0,36	1,32	-733	0,8	0,13	1,5	
Urea	Urea	Europe, average	0,36	1,32	-733	0,8	0,13	1,5	
Urea	Urea	Old technique (30 years ago)	0,36	1,32	-733	0,8	0,13	1,5	
Urea ammonium nitrate	UAN	Europe, average							
Potassium sulphate	SOP	Mining and beneficiation						0,5	
Potassium sulphate	SOP	BAT, modern Mannheim process							
Potassium sulphate	SOP	Europe, average							
Potassium sulphate	SOP	Old technique (30 years ago)							
NPK compound, 15-15-15	NPK 15-15-15	Europe, average, mixed acid route							
NPK compound, 15-15-15	NPK 15-15-15	Europe, average, nitrophosphate route							

[1] The CO_2-credit considered during urea production will be balanced by the same amount of CO_2 emitted after application and hydrolysis of urea in the field.

Table 11: Energy consumption per kg active ingredient due to production of plant protection substances (based on: Gaillard et al. (1997))

Group of substances	Additional info	Total MJ/kg	Coal MJ/kg	Oil MJ/kg	Gas MJ/kg	Diesel MJ/kg	Electricity MJ/kg
Fungicide, average	Incl. feed stock and fuel	104	3,5	22,7	34,8	30	13
Growth regulator, average	Incl. feed stock and fuel	140	1,3	35,7	39,5	45,2	18,3
Herbicide, average	Incl. feed stock and fuel	150	1,3	37,7	41	51,5	18,5
Insecticide, average	Incl. feed stock and fuel	313		134	66,7	66	46,3

Table 12: Energy consumption due to production, processing and storage of seeds (based on: Oheimb (1987), Gaillard et al. (1997))

Crop	Total MJ/kg	Coal MJ/kg	Lignite MJ/kg	Oil MJ/kg	Gas MJ/kg	Electricity MJ/kg
Cereals, average	3,5	0,63	0,1	1,57	0,67	0,52
Maize	3,4	0,61	0,1	1,52	0,64	0,51
Oats	3,2	0,58	0,1	1,44	0,61	0,48
Winter barley	3,3	0,6	0,1	1,49	0,63	0,5
Winter oil seed rape	5,7	1,02	0,17	2,56	1,08	0,85
Winter rye	4,0	0,72	0,12	1,8	0,76	0,6
Winter wheat	3,5	0,63	0,1	1,57	0,67	0,52

Table 13: Means of transport and related energy consumption (based on: Patyk & Reinhardt (1997), IEA (1992, in Patyk & Reinhardt))

Means of transport	Additional info	Load capacity	Total		Diesel		Oil		Gas	Electricity
		t	MJ/t*km	MJ/km	l/km	l/t*km	kg/km	kg/t*km	kg/t*km	kWh/t*km
Barge	For transportation on inland waters	900	0,43	387,11	9,85	0,011				
Cargo ship	9000 - 23000 t load capacity	10000	0,37	3690,00			90	0,009		
Ocean ship	Transportation of bulk goods	40000	0,13	5084,00			124	0,003		
Train	Transportation of bulk goods ("Ganzzug")	1000	0,38	381,21	9,70	0,010				
Truck	Total weight: 7,5 t; fuel use at full capacity	3,75	1,89	7,09	0,18	0,048				
Truck	Total weight: 10-20 t; fuel use at full capacity	10,5	0,87	9,17	0,23	0,022				
Truck	Total weight: 25 t; fuel use at full capacity	15,3	0,70	10,64	0,27	0,018				
Truck	Total weight: 30 t; fuel use at full capacity	20,5	0,60	12,32	0,31	0,015				
Truck	Total weight: 40 t; fuel use at full capacity	28	0,53	14,70	0,37	0,013				
Pipeline	Transport of gas; driven by gas turbine		0,75						0,02	
Pipeline	Transport of oil; driven by electric motor		0,07							0,02

Table 14: Energy use during production, maintenance and repair of agricultural machines (based on: KTBL (1999), Gaillard et al. (1997), manufacturer websites (weight))

Machine	Description and assumptions	Energy consumption per kg machine due to production/maintenance/repair				
		Total MJ/kg	Oil MJ/kg	Diesel MJ/kg	Gas MJ/kg	Electricity MJ/kg
Tractor (34-40 kW)	Rear-wheel drive, 37 kW (Fendt)	68,32	23,40	2,52	7,13	35,27
Tractor (60-74 kW)	All-wheel drive, 74 kW (Fendt)	59,42	21,05	2,25	6,38	29,75
Tractor (75-92 kW)	All-wheel drive, 81 kW (Fendt)	59,42	21,05	2,25	6,38	29,75
Tractor (93-111 kW)	All-wheel drive, 103 kW (Fendt)	59,42	21,05	2,25	6,38	29,75
Tractor (130-147 kW)	All-wheel drive, 132 kW (Fendt)	59,42	21,05	2,25	6,38	29,75
Combine harvester (80-110 kW)	92 kW, 3,00 m cutter bar width (Claas)	55,82	20,54	2,19	6,21	26,87
Combine harvester (110-130 kW)	110 kW, 4,35 m cutter bar width (Fendt)	55,82	20,54	2,19	6,21	26,87
Combine harvester (130-160 kW)	132 kW, 4,90 m cutter bar width (Fendt)	55,82	20,54	2,19	6,21	26,87
Combine harvester (160-200 kW)	184 kW, 5,55 m cutter bar width (Fendt)	55,82	20,54	2,19	6,21	26,87
Fertilizer spreader	1000 l / 1,5 t capacity, 24 m working width, 6000 ha life time	52,50	20,54	2,33	6,59	23,05
Plow	5-furrow reversible plow, 2500 ha life time	52,50	20,54	2,33	6,59	23,05
Seedbed combination	5 m working width, 2500 ha life time	52,50	20,54	2,33	6,59	23,05
Drill	Pneumatic, 4 m width, 3000 ha life time	52,50	20,54	2,33	6,59	23,05
Sprayer	Mounted, 1500 l capacity, 15 m working width, 6000 ha life time	52,50	20,54	2,33	6,59	23,05
Baling press	Round bales, 1,5 m diameter, 30000 bales life time	52,50	20,54	2,33	6,59	23,05

Table 14 (continued)

Machine	Weight	Lifetime	Energy consumption per hour of use due to production/maintenance/repair				
			Total	Oil	Diesel	Gas	Electricity
	kg	h	MJ/h	MJ/h	MJ/h	MJ/h	MJ/h
Tractor (34-40 kW)	2450	10000	16,74	5,73	0,62	1,75	8,64
Tractor (60-74 kW)	4220	10000	25,08	8,88	0,95	2,69	12,55
Tractor (75-92 kW)	5240	10000	31,14	11,03	1,18	3,34	15,59
Tractor (93-111 kW)	6180	10000	36,72	13,01	1,39	3,94	18,39
Tractor (130-147 kW)	6800	10000	40,41	14,31	1,53	4,34	20,23
Combine harvester (80-110 kW)	8000	3000	148,85	54,78	5,85	16,57	71,66
Combine harvester (110-130 kW)	8000	3000	148,85	54,78	5,85	16,57	71,66
Combine harvester (130-160 kW)	9000	3000	167,46	61,62	6,58	18,64	80,62
Combine harvester (160-200 kW)	14000	3000	260,49	95,86	10,23	29,00	125,40
Fertilizer spreader	400	1000	21,00	8,22	0,93	2,64	9,22
Plow	1500	3250	24,23	9,48	1,07	3,04	10,64
Seedbed combination	2100	1000	110,25	43,13	4,88	13,83	48,41
Drill	770	2000	20,21	7,91	0,90	2,54	8,87
Sprayer	800	2160	19,44	7,61	0,86	2,44	8,54
Baling press	2100	700	157,50	61,61	6,98	19,76	69,15

Tab. 15: Fuel use of agricultural machines (based on: KTBL (1999))

Machine	Assumption	Diesel consumption during average use		
		MJ/h	kg/h	l/h
Tractor (34-40 kW)	Rear-wheel drive, 37 kW	172,92	4,05	4,4
Tractor (60-74 kW)	All-wheel drive, 74 kW	318,33	7,46	8,1
Tractor (75-92 kW)	All-wheel drive, 81 kW	400,86	9,39	10,2
Tractor (93-111 kW)	All-wheel drive, 103 kW	487,32	11,41	12,4
Tractor (130-147 kW)	All-wheel drive, 132 kW	723,12	16,93	18,4
Combine harvester (80-110 kW)	92 kW, 3,00 m cutter bar width	589,5	13,81	15
Combine harvester (110-130 kW)	110 kW, 4,35 m cutter bar width	707,4	16,57	18
Combine harvester (130-160 kW)	132 kW, 4,90 m cutter bar width	864,6	20,25	22
Combine harvester (160-200 kW)	184 kW, 5,55 m cutter bar width	1021,8	23,93	26

Table 16: Duration of agricultural operations (based on: KTBL (1999))

Type of operation	Machines	Assumption	Duration h/ha
Fertilizer application	Tractor, fertilizer spreader	Bulk fertilizer, 5 ha field, application rate 0,4 t/ha, only on-field activity	0,15
Fertilizer application	Tractor, fertilizer spreader	Bulk fertilizer, 20 ha field, application rate 0,4 t/ha, only on-field activity	0,11
Fertilizer application	Tractor, sprayer	Liquid fertilizer, 5 ha field, application rate 300 l/ha, only on-field activity	0,22
Fertilizer application	Tractor, sprayer	Liquid fertilizer, 20 ha field, application rate 300 l/ha, only on-field activity	0,18
Plant protection	Tractor, sprayer	5 ha field, only on-field activity	0,22
Plant protection	Tractor, sprayer	20 ha field, only on-field activity	0,18
Soil preparation	Tractor, plow	5-furrow, 5 ha field, only on-field activity	1,3
Soil preparation	Tractor, plow	5-furrow, 20 ha field, only on-field activity	1,1
Soil preparation	Tractor, seedbed combination	5 m working width, 5 ha field, only on-field activity	0,39
Soil preparation	Tractor, seedbed combination	5 m working width, 20 ha field, only on-field activity	0,3
Seeding	Tractor, drill	4 m width, 5 ha field	0,65
Seeding	Tractor, drill	4 m width, 20 ha field	0,63
Harvest	Combine	3 m working width, 6 t grain/ha, 5 ha field	1,6
Harvest	Combine	6 m working width, 6 t grain/ha, 20 ha field	0,87
Harvest	Tractor, baling press	Round bales, 1,5 m diameter, 200 kg per bale, 5 ha field	0,58
Harvest	Tractor, baling press	Round bales, 1,5 m diameter, 200 kg per bale, 20 ha field	0,53

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