Agricultural data for Life Cycle Assessments

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Financially supported by: LCA Net Food
Concerted Action PL-97-3079 of the EU Food and Agricultural programme (FAIR)

Volume 1

February 2000

Report 2.00.01

Agricultural Economics Research Institute (LEI), The Hague

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Agricultural data for life cycle assessments
Weidema, B.P. and M.J.G. Meeusen
The Hague, Agricultural Economics Research Institute (LEI), 1999
Report 2.00.01; ISBN 90-5242-563-9; Price NLG.120.- (including 6% VAT)
195 p., fig., tab., app.

This book deals with the problem of selection, exchange, and interpretation of agricultural data for use in Life Cycle Assessments. It contains the proceedings of the 2nd European Invitational Expert Seminar on Life Cycle Assessment of Food Products, which was held on 5 and 26 January 1999 at the Agricultural Economics Research Institute (LEI) in The Hague. The papers cover the following topics: energy consumption, substance balances (especially for nitrogen and phosphorous), and the use of farm typologies and farm accountancy systems for LCA data acquisition.

The discussions and conclusions of the seminar, which are also reported in this book, were moderated by experts on LCA on agricultural products. To complement the topics covered by the seminar, this book contains some invited papers on data for other environmental aspects, such as pesticide use, biodiversity, soil quality, and occupational health. All contributions have been peer re-viewed for acceptance by two or more anonymous reviewers. The first volume of the book consists of sections A, B, C, (topics), while the second volume of the book deals with chapter D, E (topics).

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Preface

Life Cycle Assessment (LCA) is a technique for assessing the potential environmental impacts associated with a product, by compiling an inventory of relevant environmental exchanges of the product throughout its life cycle ('cradle to grave') and evaluating the potential environmental impacts associated with those exchanges.

Our motivation for publishing this book on 'Agricultural data for LCA' has been twofold:

- first, data collection is the most time consuming and costly part of an LCA study. This is especially true for the agricultural part of the life cycle, which at the same time determines a major part of the environmental impacts of most food products. Agricultural processes are characterised by a large variation due to the way agricultural produce is structured: in many, small farms. Variation in climate, soil and management systems cause variation in agricultural inputs, yields, and emissions to water, soil, and air. Consequently, one has to be careful in selecting and interpreting the economic and environmental data used in Life Cycle Assessments;
- secondly, LCA data from agriculture are not at all easy to obtain. At first sight, this may seem like a paradox, because out of the entire food chain, agriculture is that part of the life cycle for which the largest amount of data is in the public domain, e.g. from the Farm Accountancy Data Network and from the agricultural on farm research. However, the available environmental data are seldom in a form that is related directly to the amount of specific products produced, and typically they are not collected in any standardised form across national borders.

In summary, there is a need for facilitating the selection, exchange, and interpretation of data for Life Cycle Assessments, especially for agricultural processes.

This book contains the proceedings of the 2nd European Invitational Expert Seminar on Life Cycle Assessment of Food Products, which was held on the 25 and 26 January 1999 at the Agricultural Economics Research Institute (LEI) in The Hague. The invited papers for the seminar cover the topics:

- energy consumption;
- substance balances (especially for nitrogen and phosphorous); and
- the use of farm typologies and farm accountancy systems for LCA data acquisition.

From all over Europe, 32 experts participated with their state of the art knowledge in these areas. The discussions and conclusions, which are also reported in this book, were moderated by experts on LCA on agricultural products.

To complement the topics covered by the seminar, this book contains some invited papers on data for other environmental aspects, such as pesticide use, biodiversity, soil quality, and occupational health.

To ensure a high quality of the papers in this book, all contributions have been peer reviewed for acceptance by two or more anonymous reviewers following the procedures for international, scientific journals. We thank both the authors and the reviewers for the time they have spent in bringing their papers up to these high standards.

We express our thanks to all participants and especially to the chairs of the working groups: Patricia Cortijo, Dirck Ceuterick, Sarah Cowell, and Erwin Lindeijer.

Finally, we thank the European Commission DG XII for financial support through the concerted action 'European Network for Life Cycle Assessment Research and Development within the food chain (LCA Net Food)'; PL-97-3079 of the Food and Agriculture programme (FAIR). The seminar and this book forms part of the activities of one of the working groups of LCA Net Food focusing on 'Data, database and software'.

The managing director,

Prof. Dr. L.C. Zachariasse

Introduction

M.J.G. Meeusen [1] and B.P. Weidema [2]

Background

Every human activity and consequently its environmental impacts can be related to a certain need and the fulfilment of this need by material or non material products. Therefore, products play an important role in a regulation aimed at reducing the total environmental impact. Life Cycle Assessment (LCA) is a technique for assessing the potential environmental impacts associated with a product, by compiling an inventory of relevant environmental exchanges of the product throughout its life cycle ('cradle to grave') and evaluating the potential environmental impacts associated with those exchanges. Thus, this technique considers all processes, which contribute to the environmental impact of the final product. Life Cycle Assessment results can play a role in the decision making processes of governments, non governmental organisations and companies.

LCA is subject of national and international research programs in which the methodology is being further developed and standardised. Two main organisations can be mentioned: the Society for Environmental Toxicology and Chemistry (SETAC) and the International Organisation for Standardisation (ISO). These activities deal with LCA in general. They do not address specifically the problems and issues that are related to Life Cycle Assessments in the food chain. Therefore, other fora have been established in which the focus is on the agricultural sector.

The 1st European Invitational Expert Seminar on Life Cycle Assessment of Food Products was held in Denmark in 1993. The proceedings (Weidema, 1993a) are still available. At that time, Life Cycle Assessment was in its infancy. Approximately half of all published life cycle studies at that date were on packaging (Pedersen and Christiansen, 1992, Rubik and Baumgartner, 1992) and only 11 studies had been made on food (Weidema, 1993b). The 1st seminar was therefore dedicated to presentations on methodology and ongoing research.

Now, the number of life cycle studies in the food sector has grown to an extent that it is difficult to count their numbers precisely. A bibliography is included in Ceuterick et al. (1998). The problems that we face now are not so much on methodology, as facilitating its practical application, so that life cycle thinking can become a part of the every day routines of the food sector. This forms the background of the aims of the EU Concerted Action LCA Net Food (FAIR-97-3079). The aim of LCA Net Food is to develop and support an increased use of LCA results as a basis for strategic, tactical and operational decisions; the aim can be split up in four sub aims:

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- to build a European network for Life Cycle Assessment within the food chain;
- to evaluate and report the state of the art of the present LCA methodology with special emphasis on the applications and knowledge gaps within LCA studies dealing with the entire food chain;
- to develop a Strategic LCA Research Program focused on the food chain; and
- to initiate and promote the formation of a pan European data base for LCA within the food chain.

The focus on data is a consequence of data collection being the most time consuming and costly part of LCA studies. Those life cycle studies, which have so far been made on food products, have shown that choices made in the agricultural part of the life cycle determine a major part of the environmental impacts of the chain.

Therefore, we decided that the 2nd European Invitational Expert Seminar on Life Cycle Assessment of Food Products should focus on the issue of 'Agricultural data', also because the data from agriculture are not at all easy to obtain. At first sight, this may seem like a paradox, because out of the entire food chain, agriculture is that part of the life cycle for which the largest amount of data is in the public domain, e.g. from the Farm Accountancy Data Network and from the agricultural on farm research. However, the available environmental data are:

- seldom in a form that is related directly to the amount of specific products produced, thus requiring additional modelling;
- having a large variation, partly due to the existence of many different production units, each with different products and production methods, partly due to the natural variation in local circumstances, e.g. soil type and climate;
- typically not collected in any standardised form across national borders.

In Summary, there is a need for facilitating the selection, exchange and interpretation of data for Life Cycle Assessments, especially for agricultural processes.

In the Netherlands, the Foundation for a Sustainable Food Chain (Stichting Duurzame Voedingsmiddelenketen, DuVo) has already made a first start. This Foundation has listed the requirements and possibilities for developing a Data Conversion Tool (Meeusen- van Onna, 1997). This is more than just a database, since it involves procedures and facilities for data collection, data treatment, data exchange, and data interpretation. Other sectors (such as the packaging and the automobile industries) have already taken joint action in the area of generation and management of product related environmental data.

The objective of this book

The objective of this book is to lay the foundations for a harmonisation of the techniques used for collection and modelling of agricultural data for use in Life Cycle Assessments.

To make data meaningful for Life Cycle Assessment, means that they must always be related to the products produced. Therefore, the first question that we have set out to answer is:

- How can the environmental data best be modelled to the outputs of individual crops and animals?

Implicitly this question also focuses on whether general, harmonised models can be found and how such models relate to different farm types. A farm typology may also be part of the answer to the next two questions, namely:

- What are the most important parameters determining differences in product related environmental data?
- How can such data be aggregated at different levels and calibrated against regional statistics?

Besides the models and the farm typologies, we have focused on the very practical problems that life cycle practitioners face right now, namely:

- What data are available today? More specifically: How are they actually collected on farm level and regional level and in what form and quality are they available? And to the extent that they are not available (both within Europe and for imported products), how should we that need data now and not tomorrow best approximate the desired data?

The tentative answers that this book gives to these questions reveal an (expected) discrepancy between the available data and the ideal data being data that 1) cover the most important parameters, 2) allow modelling for different farm types and circumstances, 3) relate to the choices that the farmers (and his customers) can make and 4) are calibrated and validated. This discrepancy should lead us to the answer to the ultimate question, namely:

- What mechanisms are necessary to ensure future availability of updated environmental data of this kind to meet the requirements of LCA?

So, to summarise, the objective of this book is to answer the above questions, so that a clear picture can be obtained of:

- what the ideal data, models and procedures are?
- what is available today by default? and
- what should be done in the future?

The 2nd European Invitational Expert Seminar

This book contains the proceedings of the 2nd European Invitational Expert Seminar on Life Cycle Assessment of Food Products, as well as some additional invited papers. The seminar was held on 25 and 26 January 1999 at the Agricultural Economics Research Institute (LEI) in The Hague.

The seminar was structured to obtain the best possible answers to the questions outlined in the preceding section. All papers were available in a preliminary form before the seminar, thus forming a common basis for the discussions in a number of parallel workshops on:

- data on energy use and fuel emissions in stables, field machinery, irrigation and crop drying;
- data on the nitrogen cycle, including emissions from animals, stables, manure and fields;
- data on other substance cycles, notably the phosphorous cycle;
- farm typologies for structuring data collection and data management.

For each of these topics, a number of experts from across Europe had been invited. Furthermore experts of LCA were present as workshop chairs all experienced Life Cycle Assessment practitioners with an agricultural background.

Other environmental aspects (use of pesticides, land use etc.) were not discussed at the seminar. The reason for this was that our preliminary investigation showed either:

- that there were not enough points of discussion (e.g. for pesticides, a European consensus was already present among the experts that we consulted); or
- the field was not yet mature for European harmonisation (e.g. for occupational health, the number of experts involved in the issue was too small to constitute a European workshop).

Instead, we invited papers for this book from among the most renowned experts in each of these fields.

At the seminar, the workshops took place in two sessions. The first session focused on defining the ideal data, the models, and possible harmonisation and validation. After a short plenary co ordination, the second workshop session focused on practical possibilities, both here and now (availability etc.) and for the future.

Outline of this book

The book follows the structure of the seminar. Chapter 2 focuses on general issues of data management, data formats, and the linking of data sources and models. The following chapters deal with the topics of the first three working groups: on energy use (chapter 3), the nitrogen cycle (chapter 4), and other substance cycles (chapter 5). Chapter 6 deals with the remaining environmental aspects not covered by the seminar. Chapter 7 deals with the topics of farm typologies and the use of farm accountancy data for LCA. Each chapter is composed of the papers of the expert participants and a summary of the conclusions of the working groups. Chapter 8 contains the overall conclusions from the seminar.

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A. Datamanagement for Life Cycle Assessments

1. Linking data sources and models at the levels of processes, farm types, and regions

N. Halberg [1], I.S. Kristensen and T. Dalgaard

Abstract

An inventory for Life Cycle Analysis of agricultural products must address questions of representatively and coherence of models. This paper discusses the possibilities of modelling the input and production of typical farms to establish inventories that are consistent with higher level statistical information. Energy use has been modelled for typical Danish farms and compared with the national agricultural statistics. Expected differences in the use of fertiliser and pesticides and in the yields of 12 different types of crop rotations on two groups of soil types were modelled on the basis of 13,000 detailed farm accounts. The overall representatively and consistency were checked against regional and national statistical information based on other sources and in case of larger differences than 2-5%, the models were adjusted. Partial models of emissions in single enterprises should be adjusted or calibrated against consistent data or models of farm types on the one side and against aggregated statistics at a higher level on the other side. This might be done for the losses of N and P using a balance approach. For emissions like greenhouse gasses or heavy metals, where no sector specific aggregated measurements exist, the emissions might be estimated directly from the input inventory. Following the proposed procedure might secure consistent models of input and emissions for a representative set of farm types and none of the tasks should be omitted without clear indication and explanation.

1.1 Background

For most agricultural products the primary production is an important determinant of the total resource use and environmental impact, which is why Life Cycle Assessments (LCA) of food products must carefully address the question of data quality for agricultural production. Weidema (1998) finds that while some retrospective applications of LCA might be based on statistically representative historical averages, tactical and strategic applications (i.e. any application with the aim of changing present production forms) needs to be based on a more thorough knowledge of differences in production systems and causal relationships between inputs and outputs. Therefore, many LCA applications need to build on representative data for specific types of farms and models of the corresponding production systems and potential environmental impacts.

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On this background the aim of this paper is to:

- discuss problems in the establishment of consistent data sets reflecting differences in the production and externalities of agricultural products;
- give examples of a modelling procedure, that combines farm level models and typologies with higher level statistical information; and
- suggest guidelines for establishing valid databases representing typical production systems and their environmental impact.

This paper is based mainly on Danish experiences and is intended as a stimulus to a debate on methods for the establishment of LC inventories.

1.2 Identifying the key problems for establishing an LC inventory

1.2.1 General problems in LCA

In a case study of three wheat production systems, Audsley et al. (1997) identified the most pertinent problems for the LCA methodology used on agricultural products to be:

- the establishment of consistent descriptions of the production system;
- the definition of the functional unit;
- allocation of environmental effects to the different functions of a multi function system; and
- characterisation of specific impacts such as acidification and eutrophication and impact on soil quality and biodiversity.

The present paper addresses the need for an inventory of inputs and outputs in terms of products and emissions, where the term emission denotes externalities in terms of losses or other types of environmental impact as for instance soil quality. A method to secure consistent or coherent models of the primary production is presented. The paper does not discuss the further quantification of emissions in terms of the impact categories, characterisation and problems of weighting/valuation. Nor does it discuss the questions of functional units and allocation in detail.

There are two basic problems when establishing a life cycle inventory in agriculture. The first problem regards the establishment of valid data sets describing the production in terms of resource use per produced unit in a consistent way and on representative farms. The second is to establish a link between the resource use and the emissions and environmental impact from production of given functional units. In the following, these two topics are discussed separately, and examples of preliminary work to solve the problems are presented together with suggestions for guidelines for future work.

Some principles and a consensus have been established for a cash crop system (three ways of cultivating bread wheat; Audsley et al., 1997). Livestock production systems are more complicated to describe because of interdependencies between the crops and the herd, because there are more different emissions than in cash crop systems (methane loss from animals for instance), and because the number of different systems probably are larger. Existing Life Cycle Assessments on livestock products have focused on other aspects of the methodology than data collection and representatively.

1.2.2 Variation and representatively

If an LCA is meant for tactical or strategic decisions it is necessary to know the variation between farms and its importance in relation to the size of the total environmental impact from the production of a given commodity. The resource use and potential environmental impact connected to the production of basic agricultural commodities may vary considerably between livestock farms (Reinhard and Thijsen, 1996; Mignolet et al., 1997; Refsgaard et al., 1998; Halberg, 1999). Some of this variation is caused by differences in physical conditions (soil types, climate) or other characteristics that are easily quantifiable (stocking rate, organic vs. conventional). Variation between years might also be important for some aspects but is in general probably not so important as the variation between different production systems.

In a study of 20 dairy and pig farms over three years, Halberg (1999) found that the yearly variations in the nutrient surpluses and efficiencies, energy use per kilogram milk or per kilogram meat, pesticide use, and in the biodiversity indicator 'percent weeds in grain', were less important than differences between farms with comparable stocking rates. Of the 10 indicators used in the study there was only a significant yearly variation in 'energy use per kilogram grain', due to differences in the need for irrigation. There is probably a difference in the degree to which the input and emissions on different farm types are dependent on climatic conditions. Industrialised pig production based mainly on imported feeds should be less dependent on yearly variations in growth conditions than cattle farms based on grazing and silage. However, it seems that the most important variation to account for is the variation between different ways of producing a given commodity including national and geographical differences as well as different management strategies within a certain region.

Therefore, representatively becomes a crucial topic for any LCA that aim at more than a case study or a strategic analysis of a specific production system. But it is not always obvious at what level of detail representatively is needed. Thus, retrospective LCAs and LCAs that are meant to represent large groups of farms should start by addressing the question of what types of production are described by which data. Important aspects are farm structure, including degree of specialisation, stocking rate, and size (grain yields of Danish cash crop producers increase with increasing size; Anonymous, 1996), and input levels, soil type, and climate. The needs for detail will vary with the aim of the study and therefore there is a need for basic datasets and generally accepted methods for securing representatively of ad hoc typologies.

The Farm Accountancy Data Network (FADN) is an EU database which includes some 58,000 farm accounts which together are representative of 4.4 million production units in the member states (Colson et al., 1998). Data are collected by the member states and should cover as a minimum 1% of the farms of different types, defined primarily by the economic size and the main activity measured by standard gross margin. Since data are based on farm accounts, mainly economic data are included together with some structural and socio economic information (size of area and number of animals of different types, respectively number of employees and age of farmer). Inputs and outputs are not quantified. Colson et al. (1998) demonstrated that it was possible to discriminate between different cattle production systems (dairy, suckler cow, mixed and beef fattening) and to include all farms with cattle production regardless of their main production. Their analysis showed the great variation in the EU cattle sector within and between member states in terms of size (area, number of cattle, standard gross margin) and in terms of specialisation (percentage of total gross income attributable to cattle production), and intensity (stocking rate on fodder crop area). However, it did not seem possible to describe these different types in more detail regarding the production and emissions based on the FADN statistics. Therefore, there is a need for the development of more detailed models of typical farms that might facilitate both a deeper understanding of production and emissions and a generalisation to large groups of farms.

1.2.3 Typologies

In (Danish) national statistics (Anonymous, 1996) and in the FADN farms are classified according to main production enterprise, size and to some extent region or soil type or socio economic characteristics (age of owner, number of employees). However, if the subject of an LCA is to compare different ways of producing a given product within the same region, this type of classification will probably be insufficient. In this case there is a need for a more detailed typology based on for instance stocking rate, type of stable or manure handling system or the amount of fodder supplied by grazing (Mignolet et al., 1997). Moreover, as mentioned above, national or EU farm statistics usually do not include data on amounts of input and output, which is why most LCAs will need data either from modelling or from more detailed farm studies. Dalgaard (1998) discuss different typologies and how they could be described by a combination of farm studies and simulation models. An advantage of such an approach is that detailed farm studies or models might facilitate a more coherent and consistent description of the production than aggregated statistics. The average farms in accounts statistics are often not representing very realistic or abundant farm types. This is because the statistics average over fundamental differences in e.g. the use of contractors vs. own machines, the percentage of roughage used vs. imported feed, or the use of fertiliser vs. use of manure and nitrogen fixation. However, if specific farm data or models are used to describe these different types, they should represent a larger group, and there is a need for methods to test the representatively against e.g. relevant FADN statistics.

1.3 Modelling production systems

As discussed in section 2, the establishment of LC inventories faces a dilemma between the need for large data sets representing the variation between farms on the one hand and the need for detailed descriptions of the production systems on the other hand. One possible solution to this is the use of farm models of input output relations in different enterprises linked together in a coherent farm level description. In the following, two examples are given of a possible method for securing the representatively of detailed production models and typologies by a combination of farm accounts, modelling and checking against higher level statistical information.

1.3.1 Representative data sets and energy use (example)

Dalgaard et al. (1998) used models of energy use in crop and livestock production on typical Danish farms to estimate the potential reduction of CO_2 emissions from Danish agriculture by a 100% conversion to organic farming. The first stage was to develop a coherent model of diesel use in livestock farms. As explained by Refsgaard et al. (1998), this was done by simulating the diesel use on each of 30 dairy farms using experimental data for expected diesel consumption in each field operation. The total expected diesel use per farm, as calculated from the standard values and the registered (field) operations on the farms, was checked against the actual diesel use on the farms (taken from accounts but correcting for private use etc.). As an average of the 30 farms, the actual diesel use was 40% higher than predicted from the experimental standards with no systematic difference between production systems or soil type. Possible explanations of this difference were that the dairy farmers used more diesel than the experimental values for some field operations due to lower maintenance of machinery or non optimal timing of work or that some operations were properly accounted for by the models (especially the handling of fodder in stables). Since the aim of the study was to evaluate energy use per kilogram milk, the distribution of diesel use between field and indoor operations was less important and the farm models were adjusted to reflect the actual diesel use by including the not accounted for diesel in the crop models. The use of electricity on the farms was partitioned to household, stables, and irrigation, using standard values.

In a second stage, the use of diesel and electricity was modelled for different crops and livestock production based on the principles of the first stage but using an improved diesel model (ØKOBÆR) including more details concerning diesel use inside the stables for handling of silage etc. and extrapolating to other farm types than dairy farms (Dalgaard et al., 1998). Table 1.1 gives an example of estimated energy use in different crops, as weighted averages over Danish soil types.

Table 1.1 Energy use for the production of typical crops modelled with ØKOBÆR

		Clover grass	Grain cereals	Fodder beets	Perm. grass
Oil, grease etc. a)	MJ ha^{-1}	3,134	4,495	13,176	823
Electricity b)	MJ ha^{-1}	792	866	446	0
Fertiliser, lime etc.	MJ ha^{-1}	10,243	5,743	4,003	698
Pesticides	MJ ha^{-1}	46	182	265	0
Machinery	MJ ha^{-1}	952	1,366	4,003	250
Total	MJ ha^{-1}	15,166	12,652	21,894	1,770

a) Including refining, distribution etc.; b) Irrigation and drying.
Source: Dalgaard et al., (1998).

The energy costs at enterprise level were then multiplied with the present agricultural production according to Danish Statistical Office (DSO). The resulting estimates of present energy use in Danish agriculture were then compared with the actual use of diesel, electricity etc. according to DSO. Table 1.2 shows that the simulated diesel use was very close to the actual diesel use and that the use of electricity was 2% higher in the statistics than predicted by the models.

Table 1.2 Comparison of the national Danish energy use in 1996, simulated by the ØKOBÆR model, and calculated according to Denmark's Statistics

		ØKOBÆR	Statistics	Corr. a)
Direct use of energy				
Oil, grease etc.	10^{15} J	19.3	19.3	1.00
Electricity	10^{15} J	12.5	12.7	1.02
Indirect use of energy				
Fertiliser, pesticides etc.	10^{15} J	14.5	13.9	0.96
Machinery	10^{15} J	4.4	4.6	1.06
Buildings	10^{15} J	5.7	6.3	1.09
Fodder import	10^{15} J	16.3	16.3	1.00
Total use of energy	10^{15} J	72.6	73.1	

a) The correction factor (corr.) is the relation between simulated energy use and the energy use according to the statistics.
Source: Anonymous (1998), and Dalgaard et al. (1998).

The calculation of the indirect use of energy is based on the amounts of production factors used according to the production models or the official statistics respectively. These amounts were then multiplied with the energy costs per unit, according to the norm tables in the ØKOBÆR model. The modelled fodder import was set equal to the import according to DSO. The energy costs for machinery and buildings are difficult to check, since the both the calculation method and the depreciation rates partly depend on non objective choices as discussed in Refsgaard et al. (1998), but this is a general problem in LCA.

After correction of the energy use in the form of electricity and indirectly in the form of imported feed, fertiliser etc., the models were used (stage 3) to simulate changes in energy use under different scenarios for organic farming, as shown in Dalgaard (1999). A weak point in these simulations is the assumed indirect energy use per kilogram imported feed. This problem could be solved if similar calculations of direct energy use were carried out for cash crops in the exporting countries. Although the predicted and the actual energy use for Danish agriculture were almost similar in Dalgaard et al. (1998), the existence of systematic errors in the models that compensate for each other cannot be rejected. The estimates of energy use in other farm types than dairy farms were not tested against farm data. This might cause systematic errors in the predicted energy use in the scenarios, why high priority should be given to the establishment of more validated farm models based on the principles described in stage 1. However, this task might depend on the establishment of a representative data set of farm types (Dalgaard, 1999), as proposed in the following.

1.3.2 Representative data sets and crop rotations (example)

A database with 13,000 detailed Danish farm accounts was used to quantify typical crop rotations and their average use of input and expected yields. The aim was to describe the present use and benefit of pesticides as a basis for a zero pesticide scenario for Denmark (Mikkelsen et al., 1998). In table 1.3, the 12 crop rotations are presented, 6 on loamy soils and 6 on sandy soils. In this case the term rotation does not imply that there is a unique order and partition of the crops within each group, since farmers apparently choose a large part of their crops from year to year based on more criteria than soil or crop health and simple continuity.

By dividing the farm accounts according to the typology, it was possible to estimate the total area with each crop rotation and the partitioning between grain, fodder, and other crops. The more detailed partitioning between grain species within the 12 rotations was based on different agronomic assumptions (for instance that all broad leafed crops were followed by winter wheat except on dairy farms on sandy soils). An expected total area with each crop rotation is given in table 1.3, together with the percentage area of each crop. For example, the area with cash crop rotations typical for pig farms on loamy soils is estimated to be 220,000 ha, with 8% rape seed and 16+23% winter wheat. In order to secure that the combined models represent the actual crop production on a national scale as closely as possible, the total area of each crop across the types of rotations was checked against the total area according to the agricultural census by the DSO (table 1.3). In case of larger differences than 2%, the area in the models was adjusted.

Using the farm accounts, relative differences in grain yields between the crop rotation types were established. Regional statistical information (DSO, data by county) on grain yields were then used to estimate the average yields in the different rotations (table 1.4). Other cash crop yields were estimated from the regional statistical information. The production of fodder crops in the rotations was calculated indirectly from the estimated fodder need of the respective animal production in each farm type after deduction of fodder import. Finally, it was checked that the average estimated yields of the individual crops across all rotations were comparable with the average yield of the total Danish harvest according to national statistics (table 1.4).

The total use of fertiliser on the different farming types was estimated by dividing the fertiliser expenses in DKK with an average prize per kilogram fertiliser (table 1.5). The total amount was partitioned to the individual crops using the standard fertilisation norms as a key. The pesticide use was estimated using a combination of the average Treatment Frequency Index (TFI) in different crops according to national statistics and the 13,000 farm accounts. While the average TFIs for the crops in the rotation models were determined by the national statistics, an analysis of the accounts allowed to differentiate between grain on different soil types. Thus, the estimated TFI in wheat and barley on loamy soils is higher than on the sandy soils.

Table 1.3 *Estimated and registered total area and percent area with different crops in 12 typical Danish farm groups according to farm accounts 1995-96 (Danish Advisory Center, Henriksen, pers. comm.) and to national statistical information (Anonymous, 1997)*

Soil type	Loamy						Sandy						Total	DSO
Rotation type a)	1	2	3	4	6	9	1	2	5	7	8	9	1995-1996	1996
Total area, 1,000 ha	218	220	168	162	142	156	279	450	135	333	212	148	2.622	2.742
Percent of area														
Set aside, perm. Grass	17	16	10	13	16	10	17	20	14	20	18	22	17	15
Crop rotation with														
- rape seed	8	8	1	5	2	3	6	7	3	2	1	3	4	5
- peas	3	3	1	2	1	1	6	5	7	3	1	2	3	3
- grass seed	2	1	1	22	1	1	4	2	3	0	0	0	3	3
- beets	2	2	26	7	10	6	3	2	0	7	10	3	6	5
- potatoes	1	1	0	0	0	0	1	1	28	2	0	1	2	2
- grass/clover d)	1	2	0	0	22	58	3	6	2	25	31	11	9	11
- whole crop c) + maize d)	0	1	0	0	20	1	0	3	1	20	33	3	7	6
- w. wheat after non cer.	17	16	9	25	15	11	21	19	20	13	10	12	16	
- w. wheat after cereals	23	23	36	23	10	23	8	8	0	0	0	8	12	29
- rye	5	5	0	0	0	5	10	8	0	0	0		4	
- winter barley	15	15	1	4	1	15	15	15	1	1	0	1	8	3
- spring barley b)	25	24	25	13	20	31	24	27	37	29	15	58	26	9
Cereals, w. average e)	84	83	71	65	46	84	77	76	58	43	25	79	66	25
Cereals registered f)	82	83	74	64	46	83	76	74	57	43	26	77	66	66

a) Arable without animal, 2. Pig production, 3. Arable, specialised in sucker beets, 4. Arable, specialised in grass/clover seeds, 5. Arable, specialised in potatoes, 6. Mixed dairy, 7. Mixed dairy with less than 1,4 LSU ha^{-1}, 8. Mixed dairy with more than 1,4 LSU ha^{-1}, 9. Small farms, less than 20 ha; b) Inclusive minor areas with spring wheat and oats; c) Mainly spring grain/barley/pea silage; d) Hkg DM ha^{-1} of silage, calculated with 1,2 kg DM per 1 Scandinavia Feed Units (=feeding value of 1 kg barley); e) Weighted average of predicted grain area (not silage) across species; f) Average grain areas according to accounts.

Table 1.4 Estimated yields (hkg ha^{-1}) in different crops in 12 typical Danish farm groups and registered average grain yields according to farm accounts 1995-96 (Danish Advisory Center, Henriksen, pers. comm.) and to national statistical information (Anonymous, 1997)

Soil type	Loamy						Sandy						Total	DSO
Rotation type a)	1	2	3	4	6	9	1	2	5	7	8	9	1995-1996	1996
Crop														
- rape seed	26	27	32	29	23	24	21	21	19	19	17	17	23	23
- peas	38	39	43	42	37	36	40	39	35	34	35	32	38	38
- grass seed	10	10	10	10	10	10	8	8	8	8	8	8	9	9
- beets	480	480	480	480	645	480	440	440	440	575	581	440	526	540
- potatoes	340	340	340	340	340	340	367	367	367	367	367	367	365	360
- grass/clover d)	82	82	82	82	82	82	77	77	77	77	77	77	78	80
- whole crop c) + maize d)	79	82	91	88	77	76	83	80	73	71	72	67	73	80
- w. wheat after non cer.	78	81	88	86	75	75	71	68	63	61	62	58	72	
- w. wheat after cereals	68	71	78	76	66	65	56	53	48	46	47	43	68	70
- rye	53	56	61	59	52	51	54	52	48	46	47	44	53	48
- winter barley	58	61	67	65	56	56	54	52	48	46	47	44	56	56
- spring barley b)	51	54	59	57	50	49	49	47	43	41	42	39	47	49
Cereals, w. average e)	62	65	73	75	61	58	57	54	50	48	51	43	58	59
Cereals registered f)	65	67	74	72	63	62	57	55	51	49	50	46		

a) Arable without animal, 2. Pig production, 3. Arable, specialised in sucker beets, 4. Arable, specialised in grass/clover seeds, 5. Arable, specialised in potatoes, 6. Mixed dairy, 7. Mixed dairy with less than 1,4 LSU ha^{-1}, 8. Mixed dairy with more than 1,4 LSU ha^{-1}, 9. Small farms, less than 20 ha; b) Inclusive minor areas with spring wheat and oats; c) Mainly spring grain/barley/pea silage; d) Hkg DM ha^{-1} of silage, calculated with 1,2 kg DM per 1 Scandinavia Feed Units (=feeding value of 1 kg barley); e) Weighted averaged of predited grain yields (not silage) across species; f) Average grain yield according to accounts.

Table 1.5 Estimated and registered use of Nitrogen and pesticides in 12 typical danish farm groups according to farm accounts 1995-96 (Danish Advisory Center, Henriksen, pers. comm.) and to national statistical information (Anonymous, 1997)

Soil type	Loamy						Sandy						Total	DSO
Rotation type a)	1	2	3	4	6	9	1	2	5	7	8	9	1995-1996	1996
							Kg N ha^{-1}							
Crop N-norm	144	145	140	141	166	144	144	146	152	169	175	143	152	
- N from animal manure b)	7	48	13	14	53	15	7	47	15	40	67	30	33	
- N from fertilizer	137	97	127	127	113	129	137	98	137	129	108	114	119	119
							Dkr. ha^{-1}							
Fertilizer cost c)	848	557	738	701	637	822	804	610	978	746	772	905	755	771
N-fertilizer cost d)	588	418	545	545	484	555	590	423	589	556	463	489	511	
P&K-fertilizer cost (diff.)	260	139	194	156	153	267	214	187	389	190	309	416	243	
Pesticides cost e)	553	533	752	626	567	471	473	468	701	449	518	350	544	565
Pesticides, TFI f)	2,6	2,5	3,2	2,7	1,9	2,5	2,4	2,3	3,8	1,5	1,4	1,8	2,5	2,5

a) Arable without animal, 2. Pig production, 3. Arable, specialised in sucker beets, 4. Arable, specialised in grass/clover seeds, 5. Arable, specialised in potatoes, 6. Mixed dairy, 7. Mixed dairy with less than 1,4 LSU ha^{-1}, 8. Mixed dairy with more than 1,4 LSU ha^{-1}, 9. Small farms, less than 20 ha; b) Inclusive minor areas with spring wheat and oats; c) Mainly spring grain/barley/pea silage; d) Hkg DM ha^{-1} of silage, calculated with 1,2 kg DM per 1 Scandinavia Feed Units (=feeding value of 1 kg barley); e) Weighted averaged of predited grain yields (not silage) across species; f) Average grain yield according to accounts.

By the described method, expected differences in the use of fertiliser and pesticides and in the yields of the different types of rotations were modelled on the basis of the farm accounts. The overall consistency was checked against regional and national statistical information based on other sources. The set of consistent and typical crop rotation models presented here was representative for Danish farms. The method might therefore facilitate the calculation of resource use and emissions connected to crop production, and thus be used as a basis for an LC inventory covering crop production. This should make it possible either to detail the farm types further or to use the data together with more detailed models of emissions, while at the same time satisfying the need for representatively.

In this example, the interest was only on crop production, and the consistency with data for livestock production was only partly checked as mentioned above. In the future, it is expected that the method will be improved in order to divide the accounts further by differences in animal production systems. Moreover, it is necessary to develop a procedure for checking the consistency of animal production, fodder production and fodder import within each type and across the typology, against national statistical information. In this way, it might be possible to define representative inventories for resource use and production of livestock farms. The next step would be to calculate the emissions and externalities connected to the different types of farms.

1.4 Modelling the emissions

Since much of the environmental impact from farming has a diffuse character, in the sense that major losses of nutrients, pesticides and green house gasses do not originate form point sources, it is usually not possible to measure the actual emissions from a given farm. Therefore, quantification of the environmental effects from farming is usually based either on models of specific losses (process based models or empirical/statistical models) or on measurements and statistics on a high level of aggregation for instance national or regional N leaching estimates. Examples of empirical models are predictions of the average N leaching from different crops based on plot experiments and the ammonia emission from stables. If the goal of an LCA is to identify hot spots in the production chain, i.e. to focus on the processes with most significant environmental impact, it is of course important to avoid systematic errors that lead to a focus on the wrong issues. Such systematic errors might have several causes, for example the use of untypical or not representative farm data, uncritical scaling up of experimentally derived relations, the use of inconsistent values of partial emissions, or extrapolation of empirical models to conditions they cannot represent. Therefore, it is important that partially calculated emissions and impacts are adjusted or calibrated against consistent data or models of farm types on the one side, and against aggregated statistics at a higher level on the other side. These points are discussed in the following.

1.4.1 Securing consistency of emissions at farm level

The estimates of emissions from the production of a given functional unit should be consistent with the assumed type of production, being it a specific case or a typical farming system. This means that the sum of nutrients in products and emissions should equal the overall farm level balance for a production system in stable state as also proposed by Audsley et al. (1997). Thus, there is a need to establish general guidelines for the calculation of farm gate nutrient balances as a basis for consistency check of detailed estimates of product related losses. Halberg et al. (1995) used farm gate balances to analyse the N surplus and turnover of organic and conventional dairy farms and argued that the N surplus expresses the potential loss from leaching and volatilisation in the long run. Using standards for ammonia volatilisation from stables and during the spreading of manure, crop level balances were established, where the surplus expressed the potential leaching, since the emission of nitrous oxides is considered very small on most Danish soils. Sveinsson et al. (1998) discuss different approaches used in the literature and have proposed a general method for calculating farm gate nutrient balances and corresponding balances at enterprise level, including biological N fixation and ammonia deposition.

Nutrient balances give only an indirect indication of emissions and might not be satisfactory for an LCA. Therefore, there is a need for the development of methods to combine the use of empirical or mechanistic models of emissions of N and P with farm level models. Hansen et al. (1998) analysed the leaching from organic farming, using on the one side and a balance approach, and on the other side an empirical model of the leaching from different crops as a function of soil type, manure application, and fertiliser levels. They did not find identical results and the residual was included as a net soil N accumulation in the farm models. This estimated accumulation was thus not based on a dynamic soil model, and the difference between the accumulation in the compared farming systems could not be explained by different practices. The computer model FASSET was developed as an attempt to integrate a nutrient balance approach for livestock farms with models of partial losses (Jacobsen et al., 1998). The dynamic model simulates the production, pesticide use and nutrient losses over several years for typical stockless and pig farms and might be initialised with data from farm studies. It is intended to further develop the model to handle dairy farms also, but more knowledge of N fixation and N turnover in grazed swards is needed.

1.4.2 Checking farm level emissions against regional data

To secure valid information for LCA it would be an advantage to establish a link between realistic and representative production system models and the aggregated emissions and environmental impacts as calculated or observed at regional or national scale. This might be feasible for some emissions like losses of N and P. In theory, it should be possible to sum the estimated leaching and ammonia volatilisation from a set of representative farms and check it against regional or national calculations of leaching, using the same approach as described for diesel use. There are, of course, several problems with this in reality. First, it is problematic to establish a coherent N balance on national level. In the Danish N balance for 1996/97 still 78,000 t out of the total surplus of 409,000 t was not accounted for as leaching, volatilisation, and denitrification, and it was not likely that all of this unexplained surplus could be a net increase in soil N (Kyllingsbæk, 1999). Another problem is that the present process or enterprise level models of leaching and volatilisation will probably not be able to predict the emissions across all farm types so precisely that it will fit the regional or national statistics. If the difference is more than, say, 25% any simple correction of the models will not be valid. There is, therefore, a need to discuss which level should determine the emission factors in an LC Inventory; either the regional level broken down to farm types and enterprises or the consistent farm level models.

The approach of controlling the sum of farm level losses against higher level statistics will probably not be feasible for the emission of for example CO_2 and methane (CH_4), because of their global nature and the number of potential sources within and outside agriculture. It is therefore suggested that this type of emissions are calculated from a consistent and representative inventory based on a set of farm types that have been checked as described above and using internationally accepted standards for the emission per unit of different energy carriers and livestock (IPCC, 1997). Standard values for the emission of CO_2 are relatively well established and this might be calculated from the farm input list and related to the amount of output (if questions of allocation to different products have been solved). Contrary to this, IPCC's standard values of CH_4 emission for different types of livestock are very general and need to be improved. Dalgaard et al. (1998) used IPCC standards for CO_2, CH_4, and N_2O emission to calculate the emission of greenhouse gasses for Danish agriculture. While the estimated emission of CO_2 was 5.2 Tg per year and the emission of CH_4 and N_2O only 0.3 and 0.01 Tg per year, the contributions to the greenhouse effect were almost identical after transforming into CO_2 equivalents (6.7 and 4.0 Tg CO_2 per year for CH_4 and N_2O respectively).

The above examples point at two distinctions in the emissions from primary agriculture. One is that for some emissions it would be possible to compare the aggregated estimates on enterprise and farm level with higher level statistical information coming from other sources. This is not so for other emissions like greenhouse gasses. The second distinction is between emissions, where the present knowledge permits a relatively secure estimation based on the production inventory, and those where this is not the case. The emissions of CO_2 and heavy metals probably belong to the first group. Other topics, however, are not so simple since the losses from a given amount of input might vary between farms due to different stables, soil types, or the farmers' skills. This is especially so for CH_4 emission and for nutrients, where differences in utilisation efficiency may lead to differences in losses at a given production level.

The input of heavy metals directly attributable to the farming practise can be calculated from the declarations of fertiliser and feed and a farm gate balance is relatively simple to establish as demonstrated for Copper in Halberg (1999). Given their nature, most of these surpluses will end in the soil and the emission is thus comparable to the farm level surplus. Nevertheless, these calculations might in future be validated against the results of a broad survey of the development in heavy metal contamination of agricultural soils. Pesticide use might be estimated using a combination of national statistics and representative and typical crop rotations as shown above. But predicting the emission to (the different recipients in) the environment will need more knowledge and will not be discussed here.

1.5 Conclusions and recommendations

Given the large variation in primary agricultural production, even within a small country like Denmark, care should be taken to check the quality of farm data used for an LC Inventory. In order to avoid misinterpretations and unrealistic extrapolations, it is necessary to base estimates of emissions from the production of a given functional unit on consistent and realistic farm models that have a clearly defined degree of representatively at regional, national, or EU level. Therefore, it is recommended to establish data bases with verified information concerning input and production on typical and representative farms using a combination of detailed farm data, models and comprehensive accounts statistics. Based on the above discussion and examples, the following recommendations for a procedure for establishing LCA Inventories concerning agricultural production and emissions could be given:

- identify typical farms and establish consistent farm level models based on realistic input output relations in the different enterprises (crops, livestock) using detailed farm data from case studies, surveys, or detailed accounts statistics;
- check the representatively of the farms in terms of the soil types, size, stocking rate, production levels in main enterprises, economic performance and possibly socio economic characteristics compared with regional/national or EU statistics;
- if important characteristics of the model farms do not correspond with statistical information (e.g. more than 5% deviation from relevant averages), the models should be adjusted accordingly;
- calculate emissions based on the farm models and best knowledge of emission processes;
- check and adjust partial emissions of nutrients with balances at farm and enterprise level;
- check modelled sum of input use, production, and emissions across farm types against aggregated statistical data for relevant region. Adjust models where deviation is larger than 5-10%.

The proposed approach could give high quality data for an LC inventory and should be seen as a contribution to a necessary debate on the common demands to the methods used for data base establishment. The different tasks might be performed in a different order according to the type of data used but none of the tasks should be omitted without clear indication and explanation. The methodology is realistic but might not always be possible to perform for all emissions and does require some resources in terms of man months and access to databases.

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2. The SPOLD data exchange format

B.P. Weidema [1]

The basic data used for environmental assessments of products (product life cycle inventory data) can be found in many different databases and software. However, practically every database and software use their own format for storing and presenting the data, making data difficult to exchange and compare. To overcome these problems, the SPOLD format was developed in the years 1995 to 1997 as a joint effort by SPOLD (the Society for Promotion of Life-cycle Assessment Development) and the different LCA database and software developers.

Thus, the SPOLD format is a common format for exchange of life-cycle inventory data, allowing data to be understood, compared and exchanged, disregarding how they are stored in their original database. The SPOLD format is *not* intended as a questionnaire for data collection, nor for reporting final life cycle inventories. It is first of all an electronic file format (SPOLD 1997). This file format is now being implemented into many of the commercial software for Life Cycle Assessments (LCA).

The file format is also supported by a freeware (SPOLD Format Software; Weidema and Grisel, 1997) which can be used to create, edit, view, import and export life cycle inventory data in the SPOLD format. The software does not contain any calculation facilities and cannot be used to combine individual data sets into a product system or life cycle inventory. For this, you need to combine the SPOLD Format Software with a spreadsheet or a dedicated LCA software. Software producers may obtain a licence from SPOLD to modify the SPOLD Format Software or to integrate it into their proprietary programs.

The SPOLD format is designed as a generic data format, not for any specific sector. Thus, it is not specifically designed for agricultural processes. Nevertheless, it should not cause any problems to enter agricultural data into the SPOLD format. The LCA Net Food has adopted the SPOLD format as the format to be used for the database structure, which will be part of the results from this EU concerted action. Also, several agricultural research groups are now using the SPOLD format for their internal data storage and exchange.

The data are organised in datasets, which each contain the environmental data relating to a specific human activity, e.g. a production process, related to its reference function, typically its main product(s). Each dataset consists of a number of fields giving information on:

- the activity that the data relates to (*data identification* and *system model*);
- the *environmental inputs and outputs and other exchanges* from this activity;
- the *data source* and *validation*.

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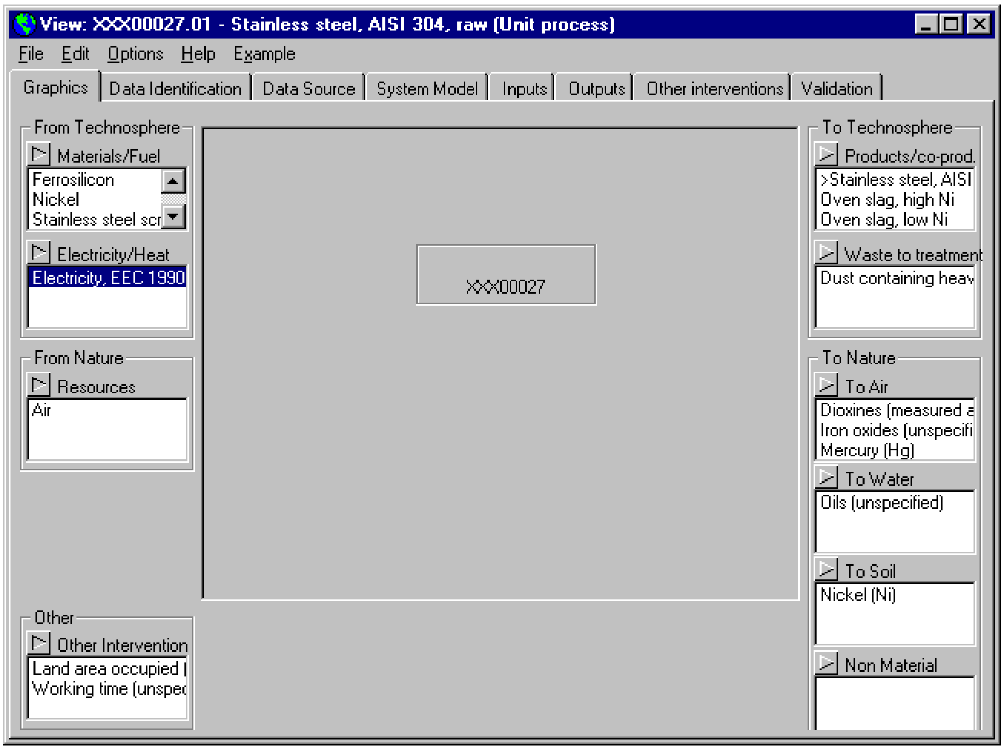

Figure 2.1 The 'graphical page' of the SPOLD Format Software
Source: Weidema and Grisel (1997).

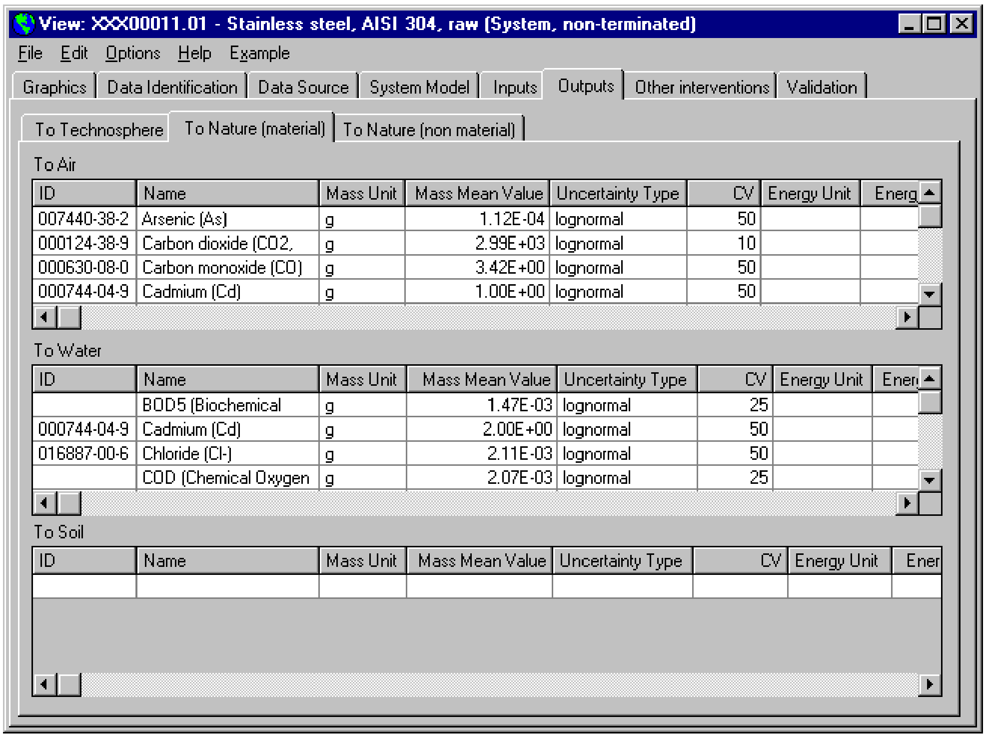

Figure 2.2 'Outputs to nature (material)' in the SPOLD Format Software

Source: Weidema and Grisel (1997).

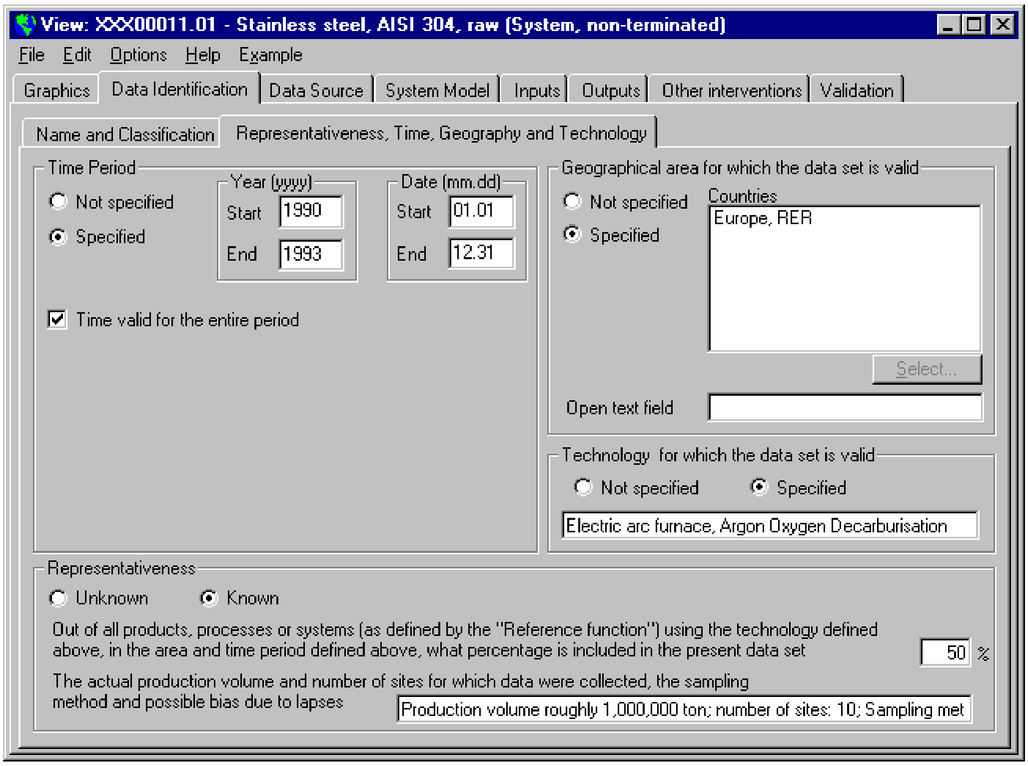

Figure 2.3 The sub tab 'Geography, time, technology and representativeness' in the SPOLD Format Software
Source: Weidema and Grisel (1997).

An electronic file format is difficult to describe in words. Instead, a few screen dumps from the SPOLD format software will demonstrate some of its features. Figure 2.1 shows the so called 'graphical page' of the SPOLD format, where the grey area in the middle signifies the system, which is described (in this case a single unit process). To the left, you can see the inputs, divided on inputs from technosphere and inputs from nature, and to the right you see the outputs, also divided in the receiving media. By clicking any of these inputs and outputs you have access to a dialogue box where you can enter details about these flows. The same data can also be seen in a more conventional table layout (see figure 2.2).

As you can see from both these figures, there are other tabs covering *data identification*, *data source*, *system model* and *validation*. These are each divided in several fields allowing very detailed reporting that is the same time easy to retrieve and compare for the data user. Figure 2.3 gives an example for the sub tab 'Geography, time, technology and representativeness'.

If you want to know more about the SPOLD format, I advise you to download the freeware from the homepage of SPOLD (http://www.spold.org/), where you can also find additional information.

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3. Systematic procedures for calculating agricultural performance data for comparing systems

E. Audsley [1]

Abstract

Comparisons of systems based simply on field measurements may give erroneous answers by confusing experimental error with genuine differences or experimental differences with genuine similarities. It would be better to have a systematic procedure to calculate values for different systems, to which the experimental studies provide the data to fit parameters. This paper describes systematic procedures to determine fuel use by field operations and grain drying, herbicide input, and nitrate leaching. For draught operations, multiply the tractor power, kW and work rate, h/ha to determine the kWh/ha required for this operation, which is largely only a function of soil type, and determine the fuel use by 175 g/kWh. A soil type index is defined to describe the differences that range between sandy, sandy loam, clay loam, and clay soils. For nitrate leaching from each crop, a formula is derived, which requires the amount of nitrogen applied to the crop, the primary and secondary yield and the soil type index.

3.1 Introduction

In carrying out an inventory for a Life Cycle Assessment, it is necessary to determine the inputs required by, and hence emissions from, the two or more competing systems under study. It is possible to carry out a detailed study of a number of individual farms and determine the energy inputs to their different operations, their chemical inputs and the corresponding outputs and emissions. However, comparing these data to ones from another farm or transferring the methods to another farm is very difficult because no two farms are the same. The difference can be physical most notably soil type and climate, historical the existence of a difficult weed problem or the soil organic matter content, or even personal the preferences of the farmer for the level of inputs. Comparisons of systems based simply on these field measurements may therefore give erroneous answers by confusing experimental error with genuine differences or experimental differences with genuine similarities. It would be better to have a systematic procedure to calculate values for different systems, to which the experimental studies provide the data, for example to fit parameters. This paper describes systematic procedures to use for several major parts of agriculture. The sections consider fuel use by field operations, grain drying, herbicide input and nitrate leaching.

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3.2 Two examples on Systematic Procedures for calculating inputs

3.2.1 Fuel Use by Field Operations

The fuel use of a tractor in identical situations is dependent firstly on its make (table 3.1), thereafter on its state of repair and the operator. Fuel use data from an OECD test of a John Deere 2850 engine (John Deere, 1987) shows that, if the operator works at 70% of its rated speed and 70% of its maximum power, it gives optimum specific fuel use of 214 g/kWh, whereas at maximum power and rated speed, the fuel use is 225 g/kWh. Similarly, if the operator sets the plough at 19.5 cm depth instead of 20.5 cm depth he will consume 5% less fuel. However although these factors will have some influence on measured fuel use, I will show that these differences are not major compared to the effects of soil condition, ballast, tyres and implement matching.

Table 3.1 Tractor fuel consumption of selected tractors

Make and Model	Max Power, kW	Specific Fuel Consumptionat maximum power, g/kWh
John Deere 2850	61.5	241
Fiat 90-90	62.0	257
Renault 103-12 RS	65.0	248
Ford 7610	66.0	267
MF 3080	68.0	255
John Deere 3350	70.1	240
Ford 8210	74.7	283
John Deere 4450	106.2	250

Source: from OECD tests, AFRC 1987.

Table 3.2 lists some fuel use figures for tractors carrying out major agricultural operations from three sources. The first column is calculated from first principles by taking the net energy required at the drawbar or power take off, which is independent of the mechanism for delivering it, and then accounting for the efficiency of the power transmission system, the traction efficiency, and the loading on the engine. The other columns are calculated by taking l/h measurements of fuel consumption and, multiplying them by the work rate. Clearly, there is a need for more information.

As part of its tractor research, Silsoe Research Institute developed a model to calculate the work rate of a tractor when ploughing in given conditions. The parameters in the model are detailed and include, for example, the engine's power curve, the gear ratios, the weight and ballast, its tyre sizes and the soil cone index to represent the trafficability of the soil, (incidentally the move to continuously variable gears reduces the drive chain efficiency the energy transmitted to the wheels from the engine).

Table 3.2 Fuel use for some farm operations, l/ha

Operation	Witney a)	FAT b)	Cope c)
Subsoiling	15		
Ploughing	21	34-38	33-38
Heavy cultivation's	13		
Light cultivation's	8	5	
Rotary cultivation's	13	20	
Fertiliser distribution	3	1-2	1
Grain drilling	4	3-4	5
Rolling	4		
Potato planting	8		
Mowing, tedding, baling	3	7-9	4-7
Forage harvesting	15		
Spraying	1	1-3	1-2
Combine harvesting	11	28-37	
Potato harvesting	21		
Bale carting		8-10	
Grain transport		3-5	

a) Witney, (1988): Choosing and using farm machines, Longman Group U.K.; b) From Swiss FAT, In: Audsley, et al. (1997) Harmonisation of environmental Life Cycle Assessment for agriculture, Silsoe Research Institute, U.K.; c) Cope, Silsoe Research Institute, private communication.

Table 3.3 shows some results from the analysis. These consider two and four wheel drive tractors of three different powers on two soil types. The two wheel drive tractors are optimised for ballast to a slip of 15%. The four wheel drive tractors use constant plough sizes. Fuel use is calculated on the assumption that the throttle is fully open but that poor matching of speed and gear means that the power delivered is not the maximum possible.

Two results are shown for the 100 kW five furrow plough. Incorrect ballast gives higher slip, reduced work rate and increased fuel use. But given correct ballast, fuel use is constant at 32 l/ha for all powers and the work rate is proportional to tractor power. On the lighter soil, fuel use is reduced to 26 l/ha with a higher work rate. Thus fuel use for ploughing and any other draught operation is a function of soil type but not a function of tractor size.

The results with the four wheel drive tractors show firstly, the much reduced slip with these tractors which enables a 60 kW tractor to pull a five furrow plough, slowly, resulting in much reduced fuel use. However, for more conventional matches of tractor and plough, the fuel use is still about 32 l/ha on the heavier soil. Similarly on the lighter soil, fuel use is little changed. There is some reduction because the higher powered two wheel drive tractors' tyres are too small to deliver sufficient traction, whereas the extra tyres of the four wheel drive equivalent can do so.

One therefore concludes that fuel use will be greater than the standard figures where equipment is inappropriately matched to the task but that this may be due to poor soil conditions, incorrect ballast or tyre size as much as the wrongly matched size of tractor and plough.

Table 3.3 Calculated tractor plough work rates and performance

Driven wheels	Power kW	Soil a)	Furrows	Slip % b)	Work rate ha/h	Power used kW	Useful power kW	kWh /ha	l/ha c)
2	60	H	3	13.2	0.564	55.8	41.1	99	32
2	100	H	5	15.8	0.946	99.4	69.5	105	32
			5	17.9	0.872	84.4	58.6	97	34
2	120	H	6	15.5	1.119	118.7	80.8	106	32
2	60	L	5	14.3	0.793	57.5	41.1	73	23
2	100	L	6	15.0	1.124	98.9	69.1	88	27
2	120	L	8	16.1	1.357	110.7	75.0	82	26
4	60	H	5	10.2	0.764	59.5	45.3	78	24
4	100	H	5	9.4	0.969	97.9	73.2	101	31
4	120	H	5	9.3	1.054	119.9	88.6	114	34
4	60	L	9	14.4	1.045	58.4	43.5	56	17
4	100	L	9	12.5	1.412	97.6	72.5	69	21
4	120	L	9	12.2	1.561	119.9	88.2	77	23

a) L = lighter soil, H = heavier soil (soil specific weight 14.1kN/m^3 and 16.7kN/m^3 respectively); b) WD tractors are optimised for ballast and 15% slip, 4WD tractors use a constant plough size for all tractor powers; c) Assuming fuel use is the maximum power 250 g/kWh for all the time.

The calculations in table 3.3 use a 70% efficiency factor in converting from a spot work rate to an overall work rate. This allows for turning at headlands, etc. However, comparing the times so calculated with the typical times used for work planning for ploughing, suggests a further 70% factor is needed to allow for sundry delays (clearing blockages, repairs, travel to/from field). During both these times, fuel use will be much lower than when ploughing. Technically, the fuel used for actually ploughing the soil will be 70% of 32 l/ha, close to that calculated by Witney (1988) for an unspecified soil type, and some allowance then needs to be made for fuel use when not actually ploughing for example, travel, turning. If one assumes 50% fuel use rate for turning type delays and 30% fuel use rate for repairs and travel type delays, this gives an overall fuel use of about 32 l/ha for that particular soil type. This provides a good explanation for the difference in fuel use in table 3.2 and suggests that the Witney data may make insufficient allowance for non working fuel use.

3.2.1.1 Procedure for estimating fuel use in general

More usefully, one can thus derive an effective rate of use of fuel per elapsed hour of the task ploughing. A reasonable estimate of the overall rate of fuel use is 70% of 250 g/kWh, i.e. 175 g/kWh. Since in the majority of cases, the data available will be the work rate and the tractor power, for draught operations this suggests that the procedure for calculating the fuel use in general should be as follows:

1. Determine from work planning data, the tractor power, kW and work rate, h/ha for this operation.

2. Determine the average kWh/ha required for this operation from a number of these estimates. Note that it is different for different soil types but not for different tractor powers.
3. Determine the fuel use by 175 g/kWh for this required energy input (diesel is 0.835 kg/l).

Examples

Ploughing		*Power Harrowing*	
Data (ABC, heavy land)	kWh/ha	kWh/ha	
56 kW @ 0.38 ha/h =	147	54-75 kW @ 0.88 ha/h =	57
75 kW @ 0.50 ha/h =	150	80-112 kW @ 1.63 ha/h =	60
'Energy' required =	149	'Energy' required =	59
Fuel used =	31 l/ha	Fuel used =	13 l/ha

A similar principle can be applied to non draught operations such as fertilising and spraying. Doubling the size of the machine is most likely to double the power needed to drive it and therefore the fuel per hectare is constant. In this case, the type of soil surface (hard, soft) as much as soil type will determine the level of fuel use.

Similar considerations apply to harvesting operations. However, broad calculations (200 kW engine @ 0.5 h/ha @ 250 g/kWh = 30 l/ha) suggests that the Witney (1988) figure is too low, and therefore a constant 30 l/ha in wheat is probably a reasonable size independent figure. In this case, condition of the crop is most likely to determine differences. It is known that 70% of the power of the combine is used in idle mode, that is when no crop is passing through the threshing and operating system. A reasonable estimate for other crops is therefore to multiply the fuel use by the increase in time needed to harvest another crop. Thus when winter wheat is 0.7 h/ha, field beans is 1.2 h/ha, oilseed rape is 0.6 h/ha, herbage seed is 3 h/ha (depends on type). Potatoes and sugar beet harvesting are largely power limited operations and fuel use is probably best estimated from the power of the tractor, the work rate and the effective fuel use as for ploughing.

3.2.1.2 Fuel use in grain drying

A major determinant of fuel use in grain drying is the amount of water to be removed from the crop. A high temperature grain drier typically uses 6 MJ/kg water removed from the grain when operated at 90°C. When operated at lower temperatures such as 40°C to prevent cracking (such as in drying beans), this rises to 10MJ/kg water removed. Thus, to dry grain by 5 from say 20% mcwb requires 62.9 kg water to be removed per tonne of 15% mcwb grain. This needs, for example, 10.6 l/t of diesel to deliver the required energy or corresponding amounts of other energy sources such as gas or straw.

A near ambient temperature grain drier uses mainly fan power to dry grain. The energy required depends on the dryness of the air. In wet regions, supplementary heat would be required for the method to succeed at all. A typical UK value is 40 kWh/t grain on the same 20 to 15% basis as above or 2.3MJ/kg water removed.

3.2.2 Herbicide use

A number of pesticide leaching models have been examined but in most cases they register zero leaching of herbicides even though occurrences of chemicals such as IPU being found at high levels in water courses are known to be causing problems (Little, 1998). It is generally concluded that the source of the chemical in surface waters are factors other than leaching through the soil such as soil erosion, preferential flow, spray drift, over spraying ditches and accidents. Therefore, we concluded that the best measure to correlate with the pesticide reaching the environment was the amount of pesticide applied.

We also need to know the amount of pesticide needed to maintain the soil in a steady state over the chosen crop rotation, in terms of weed seedbank. Squires (1998) showed that weed seedbanks increase by an order of magnitude under alternate crop rotations. The amount of pesticide required is dependent on the crop rotation. Thus continuous winter cereals will have serious problems with difficult cereal weeds, but spring sown crops or only 50% cereals in the rotation will produce much less problems and thus require less cereal weed herbicide. Type of cultivation and time of sowing also affects the amount of weeds. Field trials follow specific protocols and are a specific crop rotation. Their herbicide use will thus either over or under control the target weeds. Farmers adjust their herbicide use over years to maintain a satisfactory level of control. Thus although different systems may all contain a herbicide application operation, the dose applied in practice will be different. We have used a weed development model, which systematically determines the amount of herbicide needed for any situation (Sells, 1995).

Currently two types of herbicide are modelled: wild oat and blackgrass herbicides. Use is related to the effective control required, which is a function of crop rotation, timing of planting and cultivation techniques used within the cereal cropping. Also, spring and winter cereals will have different requirements for weed control.

Over a crop rotation, we require that the level of weeds remains constant, assuming that the farm is working at some steady state in terms of control and profitability. Assuming the weed levels are sustainable we can use the simplified equation of weed population (equation?) from Sells (1995)

$$w_{n+1} = w_n [(1-l)sg(1-k) + (1-g-m)] \tag{1}$$

where w_n are weed seed levels in year n, l is the proportion of new weed seeds removed by the combine and natural pests, s is the number of weed seeds produced per plant, g and m are the germination and mortality rates respectively and k is the kill rate of the herbicide. The germination and mortality rates depend upon the timing of planting of the crop, whether the crop is winter or spring sown and the cultivation rate.

Thus, if we define a 3 year cereal rotation with a break crop where 95% kill is achieved, then for a stable weed population using equation? over the 3 years we would need a kill rate k for the cereal crops, calculated from equation?, knowing the cultivation and timing of planting.

$$1 = [(1-l)sg(1-k) + (1-g-m)]^2 [(1-l)sg_b(1-0.95) + (1-g_b - m_{b)}] \quad (2)$$

Equation 2 can be expressed in general terms for any length of rotation, by substituting the squared term to the power n. Thus, it is possible to calculate the kill rate required for any length of rotation, cultivation and planting timings.

If we assume that one dose of herbicide on average produces an 85% kill, we can use equation 2 to calculate the number of applications or doses, x required to achieve the kill rate k.

$$(1-0.85)^x = (1-k) \quad (3)$$

Rearranging gives

$$x = \frac{\ln(1\text{-}k)}{\ln(1\text{-}0.85)} \quad (4)$$

Thus a kill rate, k of 0.67 will require a dose x of 0.58.

Using this weed model with appropriate parameters for different rotation lengths and cereal planting times in blackgrass and wild oat control gives the amount required in different circumstances. By considering many different situations, we have derived linear factors to apply to cereal planting and its timing, and adjustments depending upon the rotation.

For blackgrass, control can be achieved by planting spring sown cereals, for which no herbicide is required. For a first winter cereal crop ploughed and planted in September, 0.15 of a dose is required (assuming one dose gives 85% kill). Planting later requires no herbicide. For more winter crops in the rotation this is increased by additional 0.1, 0.2, 0.3 and 0.5 dose for 2nd, 3rd, 4th and 5th + winter cereals respectively. If barley follows wheat, or wheat follows barley then there is an additional 0.3 dose required. A winter cereal following setaside requires an additional 0.2 dose. For a shallow cultivated rather than ploughed crop, the corresponding rotational doses are 0.3, 0.65, 0.78, 0.85 and 1.07 for the 1st to the 5th year respectively.

For wild oats, the control regime includes spring crops as well. For planting in September 0.99 dose is required. This decreases over time as shown in figure 3.1.

The wild oat herbicide use for longer cereal rotations is an additional 0.2, 0.2, 0.3 and 0.4 dose for 2nd, 3rd, 4th and 5th + cereals (winter and spring cereals) respectively. If barley follows wheat, or wheat follows barley then there is an additional 0.3 dose required. A winter cereal following set a side requires an additional 0.2 of dose.

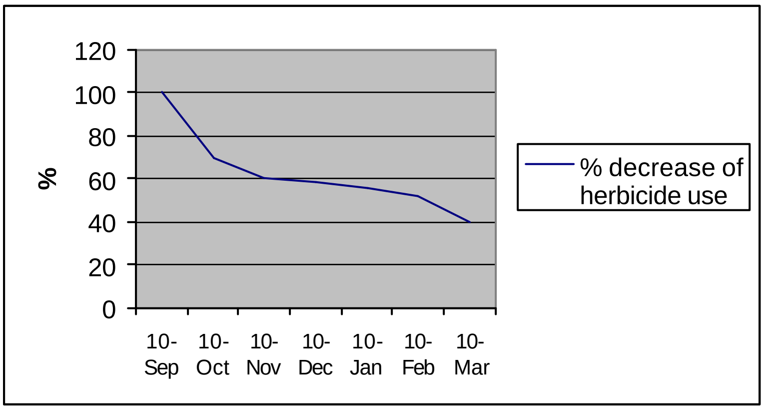

Figure 3.1   Percentage decrease of herbicide required with planting time

3.3 Nitrate leaching equation

Leaching losses of nitrogen occur via two routes:

- overwinter leaching of accumulated soil mineral nitrogen;
- wash out of fertiliser nitrogen in the spring due to an exceptional rainfall event.

We will assume in this analysis that fertiliser applied in the spring is not lost in the spring. A conceptual equation (equation 5) of nitrogen cycle is

$$N(input) = \Delta N(soil) + N(losses) + N(Harvest) \tag{5}$$

where N(input) is inorganic fertiliser, animal manure, atmospheric deposition, N fixation, N(harvest) is N removed in primary and secondary yields, and N(losses) are nitrate leaching, denitrification and volatilisation. ΔN is the increase in soil N, which in steady state is zero.

Considering nitrate leaching we can devise equation 6 to describe the nitrate leaching (B) for any crop.

$$B = (OSMN + FertN + FixN - Offtake) * \frac{D}{D + AWC} \tag{6}$$

where D is net drainage (mm/yr), OSMN is organic soil mineral nitrogen (kg NO_3/ha) and AWC is available water content of the soil. Autumn organic soil mineral nitrogen (OSMN) from stable organic matter is a function of the carbon content and this is typically a linear function of soil type. We define an index of soil type by defining a sandy loam as soil type index 1 and a clay loam as soil type index 2. All other soils can then be expressed in terms of this index. A sandy soil is index 0.5, a very heavy clay is 2.5 and a sandy clay loam is 1.5. Then soil type 0.5 has OSMN=20 kg N/ha and soil type 2.5 has OSMN=100 kg N/ha (adapted from Davies and Sylvester Bradley, 1995).

Table 3.4 lists the typical N composition of selected arable crops (data from feed analysis in McDonald et al., 1981), from which we can calculate crop offtake as a function of primary and secondary yields, in equations 7-9.

Net annual drainage (D) can be taken as the annual rainfall less annual evapo transpiration (about 440 mm for most crops in Eastern Counties of the UK). Most arable regions in the UK are in the drier areas and have a mean annual rainfall of 500-700 mm. Drainage can range from 50 in the driest areas to 400+ in the non arable wetter areas (Hughes, 1988). As an example, we take an average value of 120 mm for the Eastern Counties of the UK.

In order to determine the proportion of the nitrate leached, available water content (AWC) of soil can be approximated as 33+67*soil type index (adapted from data in Hughes, 1988), though it also depends on other factors such as the depth of soil and stoniness.

Table 3.4 Nitrogen in crop offtake

Crop	Primary yield (n_1) KgN/t (fresh weight)	Secondary yield (n_2) KgN/t (fresh weight) a)
Winter milling wheat	19	5
Spring milling wheat	20	5
Winter feed wheat	17	5
Malting barley	14	5.5
Feed barley	17	5.5
Oats	17	5
Winter rape	30	5.7
Spring rape	33	5.7
Field beans	42	
Field peas	35	
Potatoes	3-3.5	3-3.5 (chats)
Sugar beet	1.7	
Linseed/flax	38	5.7 (based on rape)
Maize silage	15 (dry matter)	
Grass silage	29 (dry matter)	
Grass grazed	38.6 (dry matter)	

a) Where the crop is harvested in two parts such as grass and straw, the grain is called the primary yield and the straw the secondary yield.

The following gives the derived base nitrate leaching formula for each crop, where N is the required amount of nitrogen for the crop, Y_1 is the primary yield and Y_2 is the secondary yield and s is the soil type index.

Non nitrogen fixing:

$$B_{ww} = \frac{(N - n_1 Y_1 - n_2 Y_2 + 25)120}{120 + 33 + 67s} \tag{7}$$

Peas and beans:

$$B_{P+B} = \frac{(52+75+25)120}{120+33+76s} \qquad (8)$$

Setaside:

$$B_s = \frac{(0.79(-0.5+50s)+25)120}{120+33+67s} \qquad (9)$$

For peas and beans it is assumed that 40 kgN/ha is made available to the next crop in addition to the average loss from all the crops (thus FixN - Offtake = 52 kgN/ha).

Operations such as ploughing and planting also have effects upon the amount of nitrate leached. The release of organic nitrogen (OSMN) assumes that the land has been ploughed for a wheat crop established early. The effect of ploughing is to approximately double the rate of mineralisation of the biomass and humus in the disturbed layers, in a few hours. After 16 weeks, the effect has gone (Dexter, 1996, pers. comm.). So the later the ploughing the less the OSMN available for nitrate leaching. Thus for ploughing there is the additional leaching shown in equation 10.

$$\frac{11.8s * 120}{120+33+67s} \qquad (10)$$

which decreases with time from July to December, so by December there is no additional leaching.

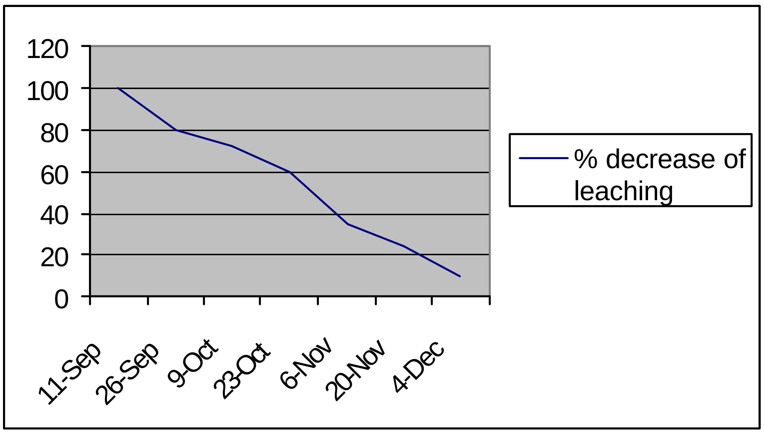

Figure 3.2 Percentage decrease in nitrate leaching with time of planting

Early planting reduces the nitrate leaching, since nitrogen will be taken up by the plant and therefore will not be available for leaching. A crop/soil model (England 1987) was used to predict how planting timing affects the nitrate uptake, resulting in equation 11 which describes the additional nitrate leaching due to planting at the end of December and figure 3.2 shows the percent decrease of nitrate leaching over time.

$$\frac{30.0 * 120}{120 + 33 + 67s} \tag{11}$$

3.4 Discussion and conclusions

The above sections are merely a selection of the many systematic relationships that can be derived between parameters in an agricultural system. There are also, for example, relationships for yield versus nitrogen applied. There is a negative correlation between a variety's protein content and yield. There are also proposed correlation's between nitrogen applied and protein content. Thus, if a new system is proposed with lower N application the yield and protein content can be calculated. If the variety is changed in order to maintain the protein content, then the yield should be adjusted downwards.

Tractor fuel use can be estimated by using energy required per hectare for specific operations. Thus, ploughing on heavy land requires 149kWh/ha and tractor fuel use for power limited operations is 175g/kWh. Process simulations show that this energy use is independent of the size of machine. This even carries over to simple things such as travelling to the field. A machine twice the size, will be twice the weight and therefore have twice the rolling resistance to overcome. Similar constant energy per hectare figures can thus be derived from work study data for any operation.

A common change when comparing alternative systems is to reduce the herbicide or cultivation input. The systematic calculation of the weed population shows whether this maintains the same level of control over the long term. As presented, it calculates the herbicide that will be required for given mechanisation inputs. It could be used in reverse to calculate the mechanisation inputs and crop rotation needed for a given level of herbicide input. It should be noted that it calculates the values for a specific timing of operations. In practice on a farm, operations are carried out over an extended period on different parts of the farm due to the level of mechanisation input in other words it is not possible to plant all crops late to reduce herbicide use without a large amount of, then under utilised, machinery. We, therefore use these relationships within a whole farm labour and machinery planning model to get the overall requirement for herbicide.

Nitrate leaching is very variable from year to year. Two assumptions are essential for LCAs:

- a mass balance must be maintained;
- a steady state system must be assumed.

The nitrate leaching formulae are based on these assumptions as a function of the three external factors: N applied, N removed and soil water holding capacity. However, they should be extended to incorporate emissions to all of the various N components within the mass balance.

The overall message is that agriculture represents a very complex system with many interactions. An existing system can be measured, but it is necessary to be careful when proposing an improvement or when comparing existing systems, that the effects are properly systematically analysed.

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B. Data on energy use and fuel emissions

4. Energy consumption: overview of data foundation and extract of results [1]

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Abstract

A number of investigations into the energy consumption within agriculture were made in the seventies and the eighties in Denmark with particular emphasis on the following topics: soil preparation, seeding, and harvesting of grass. A few studies were moreover made into houses for pigs and cattle. Since 1990, hardly any investigations were made into agricultural energy consumption, but today there is a growing interest in the subject. The main part of the available data was retrieved by use of tractor mounted measuring equipment. In addition to that, a few indirect calculations were made on the basis of labour demand, tractor power, specific fuel consumption and engine load. However, as engine loads are difficult to determine, this method is considered uncertain. For each machine, the fuel consumption is indicated as l/h or l/ha, and as such, the indications may be used in connection with model calculations for the individual work processes. For plant production, many model calculations have been made to point out which tillage methods require lower energy consumption than the conventional treatment. However, great differences are found, depending on local conditions and the technology and the methodology used. Three international working groups were formed at the end of the Eighties and at the beginning of the Nineties under the auspices of FAO and CIGR V to elucidate of the energy and labour requirement of machinery use in plant and milk production. In some cases, the variations between the countries are very great, but so far, the working groups have not had sufficient resources to investigate the causes of the variations. To explain the differences, thereby making it possible to make general use of the results, international co operation is needed.

4.1 Introduction

Since the energy crisis in 1973 and until the beginning of the Nineties, a number of studies were made in Denmark regarding agricultural energy consumption to obtain an optimal reduction of the consumption (see literature list). The two primary aims of the studies were to introduce revised/alternative production methods, e.g. reduced soil treatment and direct drilling, and to obtain behavioural changes, e.g. in tractor driving. During the past 8-10 years, the Danish authorities have shown very little attention to a continued research concerning energy demand in agriculture.

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[1] The paragraph on 'International Comparison of Labour and Energy Consumption' was written by Tarmo Luoma, and the remaining paragraphs, which only refer to Danish results, were written by Villy Nielsen.

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Apparently, the general interest in energy consumption is increasing, especially as regards reduction of the CO_2 discharge into the atmosphere. A minor research programme on cultivation of energy crops, including handling and energy consumption involved with harvesting and transport, is presently in progress.

An increasing number of inquiries are received from consulting engineers, organisations, technical advisers and students on the subject of energy consumption within agriculture.

In 1989, a FAO report was issued on energy consumption within plant production. The report comprises data from 15 European countries and New Zealand, and it is the only attempt made to collect existing knowledge on energy consumption within plant production. No similar attempts have been made since then. The reason for pointing out this specific report is that it has been highly requested and frequently used.

Due to the age of the available data on direct energy consumption, an update is needed to estimate the energy demand in new, modern and sustainable production systems on process or farm level.

In the early Eighties, a detailed analysis of the energy consumption within agriculture was made (Parsby and Fog, 1984). At the end of the Eighties and at the beginning of the Nineties, three working groups were formed to collect data and information about the energy consumption and labour consumption of machinery use. Two of the working groups have been appointed by FAO, Rome, and the third group has been appointed by CIGR V (international organisation for labour research and technology). The investigations covered both plant production and milk production. Among the countries involved, a comparative study was made on the consumption of energy and labour. However, it is difficult to make a sufficiently unambiguous description of a production process, so that it is interpreted the same way by all the countries. Furthermore, the definitions and methods used for estimation vary. In some cases, the results obtained are very different, and it is difficult to explain immediately the reason why.

4.2 Materials and methods

This paper will mainly comprise basic data and excerpts of findings and model calculations deriving from studies carried out from the end of the Seventies and until the beginning of the Nineties at Research Centre Bygholm (the former National Institute of Agricultural Engineering, SjF). To deduce further evidence from the findings, a major elucidation job would be needed. Besides, the model calculations are only to be regarded as examples, as several other model calculations with different prerequisites can likewise be made based on the given data.

This article will primarily be dealing with plant production. However, a few studies in houses for pigs and cattle have also been made, see the enclosed literature list.

Special attention has been given to the following topics: soil preparation, plant care, seeding, and harvesting and handling of grass. Hardly any studies were made on harvesting of grain crops, beets, potatoes, etc.

Data were collected from practical farming regarding the energy consumption involved with the individual processes of treatment. Moreover, additional registrations were made, e.g. on yield, dosage, load weight, transport distance, travelling speed, working width and depth, etc.

Measurements of energy consumption were made either by means of tractor mounted measuring equipment or by control of the fuel level in the tanks before and after the tests. The last mentioned method is, however, uncertain in the case of a low total consumption, owing to the fact that for some tractors it may be difficult to fill the tanks completely. More accurate and correct measurements can be obtained if the tractors are mounted with fuel flow meters. However, fuel flow meters may be somewhat difficult to mount and the system must be completely tight to avoid problems with fuel admission and measurement uncertainty (Nielsen 87, Reports No. 17 and 48). For ambulatory practical measurements, the most frequently used method is to fill up the tank with fuel before and after the test, and in case of long term tests, fuel flow meters will be mounted on the tractors. In a few cases, tractors have been tested with a view to making indirect calculations of the recorded PTO effect (Nielsen, 1987, Report No. 72).

If no data is available, indirect calculations of the energy demand can be made, by calculating the hourly energy consumption of the tractor based on the maximum PTO effect, the specific fuel consumption, and the engine load. Generally, information on the PTO effect (kW) and the specific fuel consumption (g/kWh) of tractors can be found in tractor reports from the OECD. The engine load should be estimated in the most objective way possible, which is the weakest point of the method, because great experience with the matter is needed. After calculating the hourly energy consumption of the tractor, the result is multiplied with the labour requirement, which gives the fuel consumption per hectare in litres, according to DRIFT (Nielsen and Sørensen, 1993).

Simplified model for indirect calculation of the fuel consumption:

$$F = \frac{P \times s \times m}{1000} \tag{1}$$

F = fuel consumption, kg/h;
P = PTO power, kW;
s = specific fuel consumption, g/kWh;
m = engine load, %.

Model for calculation of labour requirement (tractor hours):

$$A = \left(\frac{h \times 600}{v \times e} + \frac{p \times b \times n}{e \times (1+a)} + k + s \times h \right) \times (1+q) \tag{2}$$

A	=	labour requirement, minutes;
h	=	field size, ha;
v	=	effective travelling speed, km/h;
e	=	effective working width, m;
p	=	turnings, min/turning;
b	=	width of field, m;
n	=	number of turnings per round;
a	=	parameter, depending on shape of field and travelling pattern;
k	=	turnings on treatment of headland, min/field;
s	=	crop and soil stops, adjustment, control, tending of machine, etc., min/ha;
q	=	personal breaks, normally 5% additional time.

Thus, the are two mutually supplementary measuring methods, namely direct and indirect measuring. Both methods can be used, and providing the data basis and the models are sufficiently accurate, the methods can be considered equally exact.

Fuel consumption is influenced by many factors, e.g. type and structure of the soil, weather conditions, earth moisture, landscape, crops, tractor type (2WD/4WD), tractor size, relation between tractor and implement, driving technique, tractor driver, etc. Thus, fuel consumption does not remain a constant figure from one measurement to the other, but satisfactory results, taking into account the variations which might occur as a result of the above-mentioned factors, can be obtained by carrying out measurements over a number of years.

In the below paragraphs, results and model calculations from the three below mentioned reports will especially be dealt with:

Energy Consumption and Input Output Relations of Field Operations, FAO Regional Office for Europe, REUR Technical Series 10, Food and Agriculture Organization of the United Nations, Rome, 1989 (Pick et al., 1989)

The data presented in the report have been collected and worked up by Evzen Pick, Czechoslovakia, Olle Norén, Sweden and Villy Nielsen, Denmark. Part I was edited by Villy Nielsen, and Part II by Olle Norén. The report comprises data from 15 European countries and New Zealand. Part I deals with 'Fuel Consumption of Field Operations', and part II deals with 'Energy Input Output and Losses in Plant Production'.

Green Fields Operational Analyses and Model. Danish Institute of Agricultural Engineering: Report No. 59 (Nielsen and Sørensen, 1994)

The above report deals with the operational analyses and model simulations made in the attempt to harmonise with the legal demands for establishment of second crops (green fields) in different crop rotations. The studies were accomplished partly at the experimental fields of Research Centre Bygholm and partly at the fields of different Danish test farms. The studies furthermore included measurements of energy consumption in connection with the tests made at Research Centre Bygholm. In this case, the tractor had been equipped with a fuel flow meter, and the registrations included many different parameters such as working width, travelling speed, turning times, stops, capacity, motor load, weather conditions, yields, dosages, etc. Villy Nielsen carried out the studies at Research Centre Bygholm, and Claus G. Sørensen carried out the studies at the test farms.

Energy Consumption on Handling of Grass. Danish Institute of Agricultural Engineering: Report No. 47 (Nielsen 1991)

The above report deals with the energy consumption involved with harvesting, handling, and feeding of grass (grass silage). The report gives very detailed information on the subjects of harvesting and feeding. Thereby it becomes possible to carry out model calculations in connection with the selection of different types of machine and under different conditions. Villy Nielsen carried out the studies.

4.4 Results

Danish results

This paragraph presents extracts of the data collected on the individual machine operations and model calculations on the total fuel consumption for one single crop and for all the crops in one crop rotation.

Table 4.1 shows the fuel consumption of different types of machine. Besides information about fuel consumption, the table includes information about working width, travelling speed, working depth, and labour capacity. For some machine types, too few observations have been made to calculate the standard deviation, and for other machines, the fuel consumption has been calculated indirectly. Table 4.1 only shows the Danish data from the FAO report (REUR Technical Series 10), whereas the report includes data from 15 other countries, as well.

In the report 'Green Fields' from 1994 there is a paragraph which deals with fuel consumption on plant production in a crop rotation system related to pig and dairy production (Nielsen and Sørensen 1994). The collected basic data are shown in table 4.2. The study included three different types of stubble cultivators, reversible ploughs and land packers, weed harrows, inter row cultivators, slurry wagons, combine harvesters, etc. In some cases, a comparison of tables 4.1 and 4.2 will reveal appreciable differences in the measurement results, some of which are caused by the differences between the implements. The stubble harrows referred to in table 4.2, for instance, are different in construction, and they have been equipped with rear mounted harrows. On the other hand, there are no appreciable differences between the two types of field sprayers. However, in table 4.1 the fuel consumption has been estimated to be 1.5 l/ha, whereas in table 4.2 it was estimated to be 0.93 l/ha. In both cases, however, the fuel consumption values of the field sprayers remain within the confidence interval.

Table 4.3 shows some of the basic data obtained on picking up and chopping of grass where different types of precision chop forage harvesters and self-loading wagons equipped with different types of choppers have been used. The grass is mowed and conditioned. The fuel consumption involved was recorded in l/t contrary to in l/ha, because the capacity of precision chop forage harvesters is restricted by their chopping capacity rather than by their travelling speed or their working width. Naturally, there are limits to the travelling speed but unless the chopping capacity of the machine can be fully utilised by increasing the travelling speed, it will be necessary to join several swaths. It will thus be possible by means of model calculations to make corrections for differences in yield, which would not be possible if the fuel consumption were measured in l/ha.

As shown in table 4.3, substantial differences in fuel consumption can be seen when different types of choppers and different chopping principles are used. For instance, on picking up and chopping with a trailed precision chop forage harvester equipped with knife cylinder, the fuel consumption was 3.86 l/t of dry matter. If the knife cylinder is replaced by a cutter wheel, a reduction in fuel consumption to 2.75 l/t of dry matter can be achieved. This corresponds to a saving of 29%, but an even greater saving of 37% can be obtained by using a self propelled precision chop forage harvester instead of a trailed one.

Table 4.1 Specific fuel consumption involved with field operations

Machine types	Nominal working width m	Effective forward speed km/h	Working depth cm	Working capacity (gross) ha/h	Fuel consumption l/h		Fuel consumption L/ha		Total area ha
					aver.	Std b)	aver.	Std b)	
Direct Drill	2.6	14.6	4-6	2.18	12.73	2.93	6.44	1.31	64.9
Direct Drill	4.0	14.6	4-6	4.27	17.87	3.40	4.63	0.62	102.1
Ordinary seed drill	4.0	9.3	2-4	2.10	4.87	1.38	2.59	0.73	75.5
Combined drill	3.0	11.0	2-4	1.36	8.70	1.20	6.40	0.70	71.5
Harrow drill	4.0	9.3	2-4	2.10	6.40	0.90	3.60	0.60	13.0
Harrow drill	3.0	9.0	2-4	1.80	6.40	0.90	3.60	0.60	8.6
Rotary harrow + mounted drill a)	3.0	8.0	2-4	1.46	12.13	-	8.30	-	-
Fertiliser distributor a)	6.0	10.0	-	5.29	4.30	-	1.70	-	-
Beet drill	4.0	4.5	2-3	2.22	2.58	-	1.29	-	6.0
Cambridge roller	6.0	7.5	-	3.83	6.18	2.77	1.80	1.15	41.3
Extra light harrow	10.0	9.6	2-3	6.17	10.89	-	1.96	-	76.0
Skim plough, 3-furrow	1.1	8.3	10-12	0.58	8.14	-	15.54	-	3.0
Ordinary plough, 3 furrow	1.1	6.2	20-22	0.52	9.19	2.02	19.63	2.02	25.1
Reversible plough, 3 furrow	1.1	6.2	20-22	0.51	9.02	1.08	19.50	2.23	23.1
Seed bed harrow	5.6	8.0	6-8	3.00	10.32	1.28	3.83	1.07	26.7
Rotary cultivator + drill	3.0	6.8	5-7	1.48	9.62	2.73	7.24	0.81	17.5
Rotary cultivator + drill	2.3	7.1	5-7	1.08	9.09	1.54	9.36	2.86	37.4
Finish rotary harrow	2.5	11.0	5-8	2.31	9.87	2.75	4.74	0.91	56.2
Disc harrow	2.3	7.8	6-8	1.85	12.16	2.24	7.27	1.22	15.9
Disc harrow	2.6	7.8	5-8	1.74	9.86	1.79	6.33	0.86	38.0
Stubble cultivator	4.3	9.1	6-8	2.78	10.01	0.72	3.98	0.79	20.0
Stubble cultivator	3.3	9.1	6-10	2.36	10.51	1.77	4.93	0.85	111.6
Springtime harrow	5.6	9.3	4-7	3.83	9.58	2.10	2.81	1.02	150.3

Field sprayer a)	12.0	9.2	-	4.12	5.56	-	1.50	-	-

a) Indirectly calculated fuel consumption; b) Standard deviation.
Source: Pick et al. (1989).

Table 4.2 Specific fuel consumption involved with field operation

Machine types	Nominal working width m	Effective forward speed km/h	Fuel consumption l/h aver	Fuel consumption l/ha aver.	No. of observations	std a)	confidence interval
Stubble cultivator mounted with single Finnish rotary harrow	2.9	9.0	12.3	6.63	39	1.55	6.1-7.1
Stubble cultivator mounted with 4 axis Finnish rotary harrow	2.5	9.0	12.2	7.66	34	1.69	7.1-8.3
Stubble cultivator mounted with finger weeder	3.5	8.9	12.1	6.27	14	1.02	5.7-6.9
Field sprayer, 600 l	12.0	5.9	4.0	1.13	23	0.93	0.7-1.5
Band sprayer, 600 l	4.0	6.4	-	1.40	2	-	-
Reversible plough with land packer	3-f. 14'	5.8	12.6	23.99	36	3.35	22.9-25.1
Rotary harrow + seed drill	3.0	7.5	14.3	12.45	12	1.44	11.5-13.4
Compact harrrow + seed drill	3.0	6.8	8.8	7.77	8	0.82	7.1-8.4
Seed bed harrow	5.6	9.7	10.4	3.35	4	1.37	1.4-5.3
Mounted fertiliser spreader	12.0	8.0	8.0	1.81	5	0.36	1.4-2.3
Finger weeder	6.0	8.4	5.9	1.92	16	0.40	1.7-2.1
Finger weeder	9.0	9.0	9.5	1.75	10	0.33	1.5-2.0
Light spiked chain weed harrow	6.0	8.9	6.3	1.95	5	0.43	1.5-2.4
Combine harvester b)	4.5	-	13.6	16.48	17	6.15	11.1-17.4
Row crop cultivator (beets)	4.0	4.0	-	3.02	2	-	-
Slurry tanker, 10 t, spreading boom c)	12.0	3.89.9	6.56	16	1.42	5.8-7.3	

a) Standard Deviation; b) All crops; c) 35 t/ha.
Source: Nielsen and Sørensen (1994).

Table 4.3 Fuel consumption on picking up and chopping of grass with precision chop forage harvester and self loading forage wagon (all passes)

Machine types	Effective speed km/h	Capacity t/h	Effective working width m	Yield t/ha	Dry matter content %	Dry matter yield t/ha	Fuel consumption l/t	Fuel consumption l/t dry matter	Fuel consumption confid. interval l/t dry matter
Trailed									
Knife cylinder	7.2	22.6	2.5	20.1	33.7	5.79	1.30	3.86	3.20-4.52
Knife wheel	10.1	23.5	2.5	13.8	44.5	5.77	1.24	2.75	0.75-4.75
Multi knife cylinder	8.8	27.1	2.5	19.3	27.3	5.12	0.86	3.23	2.68-3.78
Self propelled									
Knife cylinder	9.5	58.9	5.4	16.6	30.2	4.96	0.74	2.44	1.78-3.10
Self loading wagon									
Multi knife cylinder	6.9	26.3	2.6	22.6	34.2	7.58	0.65	1.92	1.13-2.71
Fixed cutter	9.5	22.1	3.3	14.2	36.4	4.80	0.71	2.07	1.19-2.96
Average	8.7	30.1	3.1	17.8	34.4	5.67	0.92	2.71	

Source: Nielsen (1991).

Table 4.4 Fuel consumption involved with transport and unloading of grass. Transport distance, 1,000 m.

Trailer type	Load weight kg	No. of loads	Fuel consumption Transport and unloading	
			l/load	l/t
High tipping trailer, 7 m^3	2,177	548	0.64	0.29
Forage trailer with side unloader	5,098	60	1.42	0.28
Low tipping trailer	2,086	61	0.79	0.38
High tipping trailer, 12 m^3	3,427	95	1.45	0.42
Farmyard manure spreader	3,293	66	2.21	0.67
Combi trailer	4,900	42	1.46	0.30
Self loading wagon a)	6,396	220	0.85	0.13
Average	3,911	1.26	0.35	

a) Only transport.
Source: Nielsen (1991).

Figure 4.1 Fuel consumption involved with transport (Tractor, trailer, material). Travelling speed: 25 km/h. Tractor, about 65 kW on the PTO shaft. $Y = 7.64 + 0.29x$ $R^2 = 0.86$

Table 4.5 Grass harvesting. Fuel consumption on mowing, conditioning, picking up, transport, unloading and depositing into silo. Yield from 3 passes (10 t of dry matter per hectare)

Mowing	Conditioning	Picking up	Transport	Type of silo	Fuel consumption		
					l/ha	l/t dry matter	l/t 30% dry matter
Rotary without crimper	Rotary rake	Multi knife cylinder	Self loading wagon	clamp	60	6.00	1.80
Disc, crimper, belt		Multi knife cylinder	Self loading wagon	clamp	61	6.10	1.83
Disc, crimper, belt		Knife wheel, trailed	Silage trailer	clamp a)	62	6.20	1.86
Disc, crimper, belt		Fixed cutter	Self loading wagon	clamp	63	6.30	1.89
Disc + crimper	Rotary rake	Multi knife cylinder	Self loading wagon	clamp	63	6.30	1.89
Disc, crimper, belt		Knife wheel, trailed	Combi trailer	clamp	75	7.50	2.25
Disc, crimper, belt	Rotary rake	Knife wheel, Self propelled	Combi trailer	clamp	78	7.80	2.34
Disc, crimper, belt		Multi knife Cylinder, trailed	Combi trailer	clamp	86	8.60	2.58
Disc, crimper, belt		Knife wheel, trailed	Silage trailer	tower	90	9.00	2.70
Disc, crimper, belt		Knife cylinder, trailed	Tip up trailer	clamp	90	9.00	2.70
Disc, crimper	Rotary rake	Knife cylinder, trailed	Tip up trailer	clamp	95	9.50	2.85
Disc, crimper		Knife cylinder, trailed	Farmyard manure spreader	clamp	104	10.40	3.12
Disc, crimper		Knife cylinder, trailed	Tip up trailer	clamp	106	10.60	3.18
Disc, crimper b)	Rotary tedder	Knife cylinder, Trailed	Farmyard manure spreader	clamp	117	11.70	3.51

a) Deposit into silo with conveyor/elevator no consolidation; b) Adverse weather conditions.
Source: Nielsen (1991).

Table 4.4 shows the fuel consumption involved with transport to and from field at a distance of 1,000 m and at unloading. Because it has not been possible in this study to separate the figures from transport and unloading, a supplementary study concerning transport was made, the results from which is shown in figure 4.1.

Table 4.5 shows the total fuel consumption involved with harvesting of grass. The fuel consumption varies from 6.0 to 11.7 l/t of dry matter, depending on the technique and methodology used. The treatment methodology is both dependent on the techniques chosen and on the weather conditions, on which the farmers have no influence.

Table 4.6 shows an example of the total fuel consumption involved with cultivation of winter wheat. The individual operations and the number of treatments made are listed in the table. The total fuel consumption has been calculated to 80 l/ha. It will be seen that ploughing is the most energy demanding individual operation, and for that reason many studies have been made with reference to finding soil treatment methods where ploughing is not needed, e.g. reduced soil treatment or direct drilling. However, those methods have not been used very much.

Table 4.6 Fuel consumption involved with growing and harvesting of winter wheat. Conventional soil preparation

Operations	No. of treatments	Machines	Consumption l/ha
Cultivation	2	Stubble cultivator	6.6
Ploughing	1	3 furrow plough with land packer	21.4
Sowing	1	Rotary harrow + seed drill	8.3
Fertilisation	1	Mounted broadcaster	2.0
Rolling	1	Cambridge roller	1.6
Spraying	5	Field sprayer	7.5
Harvesting a)	1	Combine harvester	17.4
Transport of grain a)	1	Trailer, 5 t	3.2
Pressing of straw a)	1	Pick up baler	7.3
Transport of straw 1)	1	Trailer 1.5 t	5.3
		Total	80.6

a) Indirectly calculated. All other data were recorded in practice.

Crop yield:	6.5 t/h
Soil type:	10-15% of clay
Field slope:	0-5%
Size of field:	4 ha
Transport distance:	500 m
Doses of fertiliser:	800 kg/ha

Source: Pick et al. (1989).

Table 4.7 shows the fuel consumption involved with direct drilling of winter wheat. It appears that the fuel consumption can be reduced to about 49 l/ha, corresponding to savings of about 39%.

Tables 4.8 and 4.9 show the total fuel consumption involved with field work on two different crop rotations. In both cases, the total cultivated areas are 72 ha, divided into six fields of 12 ha. Naturally, the conditions will differ from one farm to the other in practise, but the possibilities to model the fuel consumption will still remain, only it will be a little more complicated and time consuming.

Table 4.7 Fuel consumption involved with growing and harvesting of winter wheat. Direct drilling

Operations	No. of Treatments	Engineering	Consumption l/ha
Sowing	1	Direct drill, 4 m	4.6
Fertilisation	1	Mounted broadcaster	2.0
Rolling	1	Cambridge roller	1.6
Spraying	5	Field sprayer	7.5
Harvesting a)	1	Combine harvester	17.4
Transport of grain a)	1	Trailer, 5 t	3.2
Pressing of straw a)	1	Pick –up baler	7.3
Transport of straw a)	1	Trailer, 1.5 t	5.3
		Total	48.9

a) Indirectly calculated. All other data are measured in practice

Crop yield:	6.5 t/h
Soil type:	10-15% of clay
Field slope:	0-5%
Size of field:	4 ha
Transport distance:	500 m
Doses of fertiliser:	800 kg/ha

Source: Pick et al (1989).

Table 4.8 Total fuel consumption involved in a crop rotation system related to milk production. Field size, 12 ha

Field No.	Crops	Fuel consumption	
		l/ha	l total
1	Fodder beets	169	2,028
2	Spring barley	73	876
3	Total crop + underseed	111	1,332
4	Grass clover	97	1,164
5	Spring barley	67	804
6	Winter wheat	87	1,044
Total fuel consumption for 72 ha:			7,248
Average per ha:			101

Source: Nielsen and Sørensen (1994).

The total fuel consumption involved with field work in the systems related to milk production amounts to 7,248 l, whereas it only amounts to 5,964 l in the systems related to pig production. The difference is mainly owing to the fact that in production of roughage (beets and total crop) the fuel consumption will be higher than on production of seed crops. Therefore, by choosing pig production rather than milk production, savings of about 18% can be obtained. However, there will be great variations, because the fuel consumption will depend on which crops are chosen and on the composition of crops.

Table 4.9 Total fuel consumption involved in a crop rotation system related to pig production. Field size, 12 ha

Field No.	Crops	Fuel consumption	
		l/ha	l total
1	Winter rape	102	1,224
2	Spring barley	97	1,164
3	Barley + underseed	92	1,104
4	Grass seed	30	360
5	Spring barley	96	1,152
6	Winter barley	80	960
Total fuel consumption for 72 ha:		5,964	
Average per ha:		83	

Source: Nielsen and Sørensen (1994).

Basis of data in Denmark

In some areas the available basis of data is very comprehensive, and in other areas it is insufficient, particularly in the areas of farmyard manure, commercial fertilisers, plant care and harvesting. The number of studies made in houses for pigs and cattle is very limited, and moreover, the basis of data regarding recent engineering is insufficient, as since 1993 hardly any studies were made on energy consumption.

The available data have been sufficiently documented. However, the studies were mainly carried out on nearly flat soil containing 10-15% of clay. A higher degree of specification would be desirable, especially with reference to modelling, but then a considerably higher contribution of resources would be needed. A suitable rate of amplification would be one fairly corresponding to that found in connection with labour investigations, see equation 2. This would permit a higher degree of modelling instead of carrying out measurements in practice.

The reason why a higher degree of specification is desired is that the load on the tractor will vary considerably depending on whether the implement is activated (main work), or if other part operations are being performed, e.g. turning, reloading, filling, crop or soil stops, control and tending of machines, etc. (ancillary and disturbance work). The fuel consumption involved with ploughing will e.g. be 25 l/h when the plough is activated, but it will only be 5 l/h for ancillary and disturbance work. If the distribution of work is 70% for main work and 30% for ancillary and disturbance work, the mean hourly fuel consumption will amount to 19 l/h. Ancillary and disturbance work, however, will make up between 10 and 50% of the total work, depending of the nature of work.

Other factors important for model calculations are: size and shape of fields, transport distance, travelling speed, load size, yield, dosage, material capacity, etc.

International comparison of labour and energy consumption

Within the FAO European Co operative Networks on Rural Energy, which operated in 1982-1989, a seminar was held in April 1986 in Belgium on the theme 'Energy Conservation with Tractors and Agricultural Machines'. A small working group was appointed to carry out comparative studies on specific topics of energy conservation with field machinery. The report of the working group was presented at the Third Consultation on Energy Conservation in 1988 in Helsinki, Finland. It was suggested to continue the study and widen the scope to include labour usage and costs of labour, energy, and machinery. The European Commission on Agriculture decided in its meeting in May 1989 to establish a Working Group on Labour, Machinery and Energy Data Bases in Plant Production. This Working Group consisted partly of the following members of the previous group: E. Pick, Czechoslovakia, V. Nielsen, Denmark, and O. Norèn, Sweden, and partly of the following members of the international organisations CIGR and CIOSTA: L. Weiershäuser, Germany and R.K. Oving, the Netherlands. Hungary was represented in 1990 by K. Kocsis, and later by D. Faust. The chairman of the group was E.H. Oksanen, Finland. The data was gathered in 1990-1992 from the seven above mentioned European countries. The secretary of the working group, J. Palonen, assisted by A. Laine and other researchers at the TTS Institute Finland compiled the data and unified it as much as possible.

The working group proposed that also the energy use in animal production should continue and enlarge its theme. As FAO no longer was able to support the new group, it was set up as a CIGR Section V Working Group named 'Labour and Technology in Milk Production' (1993-1995). The members of the working group were B. Sonck, Belgium, P. Keller and C.G. Sørensen, Denmark, G. Szeles and J. Fejes, Hungary, H.W.J. Donkers, A. Migchels and G.H. Kroeze, The Netherlands, J. Palonen and E.H. Oksanen Finland.

The national data from each country consists of a written description of plant production technologies, a written description of work study methods, a list of commonly used field operations and tables of work phases used on cultivation of different crops. Each country provided data of the eight following crops: spring barley, winter wheat, sugar beets, potato for human consumption and four other plants typical for each country.

Comparisons were made of plant production technologies and of selected field operations. The parameters for both comparisons were labour requirement (h/ha), fuel consumption (l/ha), fuel costs (ECU/ha), machine costs (ECU/ha) and total operating costs (ECU/ha). The compared field operations were ploughing, combine harvesting and sugar beet harvesting.

All the values presented in the study were based on the data provided by the representatives of each country, and they represent a typical way of doing certain field work in the country. Production costs were originally given in the local currencies and later changed to ECU.

The average labour requirement in study countries for spring barley and winter wheat were quite close to each other (10.7-11.9 h/ha). The corresponding figures for sugar beets and potato were 27.1 h/ha and 38.1 h/ha. The variation between countries was great. For example, the lowest and highest figures for spring barley were 5.3 h/ha (in Hungary) and 16.6 h/ha (in Germany).

Data from fuel consumption was in most cases not available. That is why it has to be calculated. Average fuel consumptions for spring barley and winter wheat were quite close to each other (97 l/ha and 101 l/ha). The figures for potato and sugar beets were also close to each other (144 l/ha and 149 l/ha). The variation was smaller than that found for labour requirement, but still large. The smallest and largest figures for spring barley were 73 l/ha in Czechoslovakia and 125 l/ha in Finland. Only Finland has included the fuel used for grain drying into these figures (55 l/ha). If this were left out, the highest fuel consumption would be 91 l/ha in Germany.

A comparison of electricity consumption was difficult because not all the countries were able to provide reliable data on that issue. Also, the variation in electricity consumption was very great from farm to farm within each country.

When evaluating the results of the study, one will be surprised to notice the vast differences between the countries. Some of these differences are due to the differences between circumstances, such as climate, farm and machinery sizes, prices of labour, fuel and machinery, etc. But these differences do not explain it all; there are some errors or at least some bias in the results.

The main reason for the errors are the different work study and calculation methods used in each country. This creates variation to the labour usage comparisons, which in turn affects all the other comparison parameters. The fuel consumption was in many cases not measured, but calculated by multiplying the time during which the tractor is used with the tractors' nominal consumption at a certain load.

Another factor to bear in mind when evaluating the results is that the figures presented from each country are only examples of a typical or traditional way of doing certain field work. There is a great variation in methods within each country. For example, new cultivation methods may change the whole picture.

Before exact comparisons in this area can be made, a common standard for work studies and calculation will be needed.

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5. Energy consumption in agricultural mechanisation

O.W.C. Vitlox and B. Michot [1]

Abstract

Air pollution is closely linked to fuel consumption. Therefore, it is interesting to evaluate the energy employed in agricultural engineering, especially with a stress on soil cultivation techniques, soil compaction, and tractor/implement combinations. A great part of the total energy consumed concerns soil cultivation. This paper gives an overview of the parameters to be included in calculation models and of opportunities to save energy by a better knowledge of soil and implements. The great variability of available data for estimating the energetic needs of different cultural operations requires a standardisation of measurement methods. Moreover, the great number of parameters that influence these measurements makes a global modelling difficult. Thus, it is necessary to isolate the main parameters that influence the energy requirements to highlight the correlation's between forces and powers needed by the implements and the required soil parameters. In this way, data given by penetrometer and profilemeter may be of great interest.

5.1 Generalities

Diesel oil, because of its energy density and relatively easy and safe handling, is of outstanding importance as an energy source for automotive machines and tractors. Despite the cost, another important aspect is the change of the climate by the consumption of fossil energy and the consequence on the environment.

In the EU, the average energy amount consumed for one hectare is over 4.2 kW. Less than 50% of installed power is employed. Furthermore, machines are utilised only a few hundred hours or even less per year, for example in Belgium, there is a combine harvester for every 33-35 ha of cereal, that is to say the overall coefficient of utilisation for all self propelled machines and tractors is under 35%.

Soil cultivation operations account for about 38% of the total direct energy demand and 27-30% of the total mechanisation costs, and harvesting represents 32% of the total energy and 25-30% of the mechanisation costs. The energy requirement variations for field operation are thus higher for soil cultivation tools than for harvesting operations.

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In agriculture, the energy requirements are closely linked to the crop type. Average values for field operations, from soil preparation to harvesting and transportation of the material, are for example of about 65 to 120 l/ha of fuel for cereal cultivation, instead of about 165 l/ha for both sugar beet and potatoes. Values exist, see table 5.1, for different operations, but are applicable with difficulty in specific situations due to the great number of parameters influencing the fuel consumption. Regarding working conditions, the energy requirements could vary from 1 to 3 times. If the conditions are optimal, a combine harvester should require an average fuel consumption of about 17 to 20 l/ha. If the conditions are bad, the fuel consumption could increase up to 25 l/ha. With the data of fuel consumption, we must specify, as much as possible, the information on the way the data have been taken and in which conditions. Furthermore, if tractor and implement are not correctly matched during the test, the values of fuel consumption may vary in a large way.

Table 5.1 Average performances of different implements

Machines and implements	Nominal working with (m)	Forward speed (m)	Working depth (cm)	Working capacity (ha/hour)	Average fuel consumption (litres/h)	Average fuel consumption (litres/ha)
Stubble cultivator	2.85	6.0	6-8	1.28	9.9	7.7
Plough 4 furrow	1.70	5.8	26-27	0.79	16.0	20.3
Plough 4 furrow	1.67	7.1	21-22	0.95	13.5	14.2
Vibro tiller + Land packer	3.25	6.0	10	1.47	13.5	9.2
Rotary harrow	3.00	5.6	5	1.26	10.5	8.3
Seed drill	3.00	8.0	2-4	1.43	7.1	4.9
Rotary harrow + Seed drill	3.00	7.7	10	1.73	19.0	11.0
6 Rows drill, sugar beets	2.70	5.0	2-4	0.95	7.1	7.5
4 Rows drill, maize	3.20	5.5	2-4	1.14	8.3	7.3
Fertiliser distributor	12.00	7.0		3.70	7.1	1.9
Field sprayer	16.00	6.0		4.03	11.4	2.8
Combine harvester	4.3	4.5		1.25	31.5	25.2
Combine harvester	4.0	4.0		1.04	23.00	22.12
Sugar beets harvester						50
Forage harvester, maize						73.0
Straw baling						6.0

Looking at the first graph (figure 5.1) which represent fuel consumption curves, one can see that:

- for a power requirement which remains constant, if the engine speed varies, the fuel consumption is modified ; for example going from point A to point B, the power output is kept constant despite the engine speed reduction from 2,360 rpm to 1,360 rpm, the main consequence is that the fuel consumption decrease from 17.2 l/h to 13.8 l/h, that is to say a reduction of 20%;

- with the fuel consumption, it is also possible to determine the power output which is required by the field operation, but in this specific case the engine speed and the fuel temperature must be noticed, otherwise it is impossible to evaluate the power which is demanded; for example going from point C to D, the fuel consumption is always 18 l/h, but the power output decreases from 65 kW to 54 kW, while the engine speed increases, that is to say a power reduction of 17%.

Those examples illustrate the danger of evaluating the required power output for a specific field task, only by measuring the fuel consumption. The engine speed, the fuel consumption and the fuel temperature must be measured, so as to compare with fuel consumption curves calculated from bench testing results. Ideally, the power required should be evaluated by sensors able to measure all the constrains at implements level.

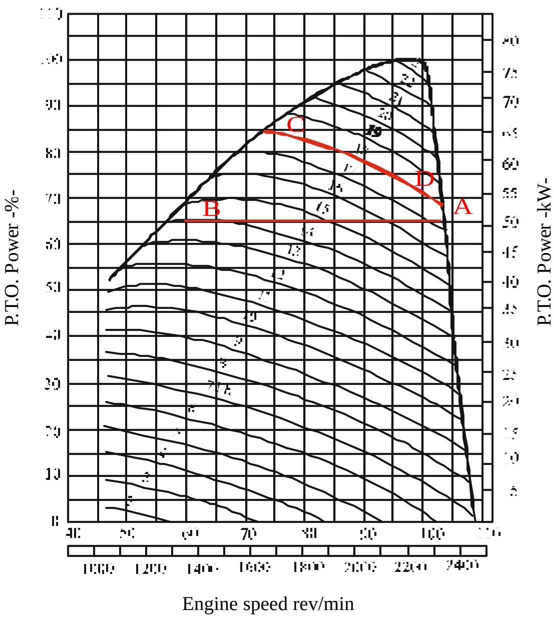

Figure 5.1 Power and fuel consumption at partia load

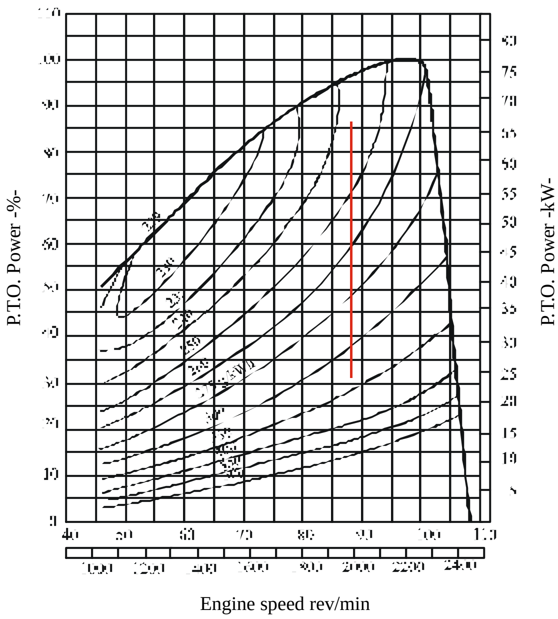

Figure 5.2 Power and specific fuel consumption at partial load

The second graph (figure 5.2), giving the specific fuel consumption, illustrates the fact that if the tractor and the implement are not matched, the energy delivered by the tractor may be a lot higher than that required by a machine or implement. The ideal situation should be to have a tractor for each implement, so as to make the engine rotate at the better speed and to have the right weight, or to have implements which are the most appropriate to the task and to the tractor performances. For example, if one have to use a 3m width rotary harrow at 4 km/h the required power is 24 kW; the specific fuel consumption is of about 330 g/kWh at 2,000 rpm. If the working speed is increased at 7 km/h and the working width at 4.5 m, the required power for the implements as well as for traction reaches 66.5 kW and the fuel consumption falls down to 242.5 g/kWh, all the other conditions being the same. The reduction is 16.5%.

Soil cultivation

The energy requirements could vary regarding agricultural practices: direct drilling, soil working depth and number of cultivation passes. These are also linked to crop type because the working depth depends of the plant.

Soil cultivation implies innovative, low energy technologies and agronomic techniques appropriated for specific soil conditions. In addition, the improvement of tractor implement combinations through the installation of electronic control systems on tractors must also be taken into account. Preliminary analyses have indicated that average current energy could be cut by 35-40%.

Soil compaction

The weight of tractors, implements, and self propelled machines is applied to the ground through their tyres, which has several consequences:

- soil deformation (mainly in soil with low bearing capacity), which causes the increase of rolling resistance, this strongly affects direct energy consumption, all other conditions being the same;
- changes in the soil structure due to compaction and damage to vegetation, which decrease yield and often requires additional tillage;
- saturated soil with low bearing capacity cannot be trafficked, which reduce the period of time during which field operations may be performed and requires the use of large machines to carry out the work in time;
- compacted soil strongly decreases hydraulic conductivity which leads to water run off and erosion with as main consequence pollution due to an increasing amount of pesticides at the outlet of watersheds;
- higher use of fertiliser to compensate the lack of disposability of nitrogen due to its mineralisation.

Tractor implement combinations

When selecting a tractor, the power required for a task to be achieved is often confused with the required drawbar pull; sometimes a higher powerlift is necessary to use implement combination. As a consequence, oversized tractors are purchased and underused.

In addition, implements are mostly selected based on their performances without considering the requirements of rationalisation, inherent to their combination. Furthermore, implements and tractor could not be replaced in the same time. Consequently, the investment is not optimally used, which leads to direct energy consumption far more important than necessary.

5.2 Soil cultivation

The extreme form of reduced cultivation is direct drilling. This technique could lead to a reduction of fuel consumption of about 70-75%. Seeds are sown directly into unmoved ground after all weeds being entirely controlled by spray applications. Such techniques are mostly used for cereal crops, however for sugar beet cultivation, modified equipment have been designed for deeper strip tillage.

Reduced depth of cultivation is widely used to reduce energy consumption. Sub soiling every three or four years may be necessary if compaction becomes a problem. Lower depth of cultivation is a suitable technique for cereals. However, very shallow cultivation is more dependent on the amount of crop residues than on sprays to control weeds. Reducing cultivation depth has lead to substantial reductions in energy requirements for cultivation. The energy reduction rate obviously depends on the amount of reduction in depth of work.

Reduced numbers of cultivation operations may be achieved by using combinations of implements. Reduced number of passes has application to all types of crops. The technique offers potential for saving energy, reducing labour needs and reducing soil compaction. Furthermore correct machine setting can also lead to energy savings, for example with a rotary harrow excessive rotor speed requires a high power input.

The common method for reducing the number of passes onto the ground is to combine secondary cultivation mechanisms with a seed drill (table 5.2). The single pass system, which both cultivates and drills directly on cereal stubble, requires less energy and labour. The drilling rate is also too low because of the other implements in the combination. Both systems are used satisfactorily for cereal cultivation and give savings in time of 20-30%.

Combining cultivation operations with a drill not only gives energy savings and reductions in tractor tracks, but traffic on just tilled soil can be avoided eliminating therefore, further soil compaction.

Table 5.2 Performances of several tillage techniques

	Time/ha	Fuel/ha	Area/day	Mini. pow.
Ploughing + seedbed preparation and seeding	3 h	41.5 l	2.6 ha	100 HP
Seedbed preparation and seeding with pto driven implement (3 m)	45 min	18 l	9.6 ha	130 HP
Seedbed preparation with a disk implement (5 m), seeder (6 m)	30 min	18 l	14 ha	120 HP
	30 min	18 l	14 ha	120 HP
Decompaction + seedbed preparation + seeding				
1 pass	1 h	24 l	8.5 ha	150 HP
2 passes	2 h	40.5 l	3.8 ha	150 HP

The best way to fit the power and energy required by implements related to working depth and travel speed, is to measure strengths, torques and speeds with sensors between tractor and implement (figure 5.3). So, it is possible to know their net power or energy needs, with a minimum of interfering parameters. The fuel consumption must be calculated from these measurements, taking into account of the tractor implement combination.

Figure 5.3 Measurements of draught, torque and pto speed for a rotary cultivator

To avoid the need of repeating the tests for each condition, models have been developed to link the required power or draught to soil conditions. To be practical, it is necessary to limit the number of parameters to be measured on the field, particularly soil parameters, which require long and costly tests in laboratory. Most of the models developed for the determination of traction or power needs for tillage are based on a few parameters like cone index, humidity, and soil bulk density. The Cone Index is internationally recognised and used in numerous formulae. It is determined by introducing a cone shape probe (penetrometer) into the ground at constant speed. A load cell measures the strength required to sink in the probe. The data processing determines the soil consistency, called Cone Index, dividing the strength by the base area of the cone. So, it is possible to determine directly the level of soil compaction.

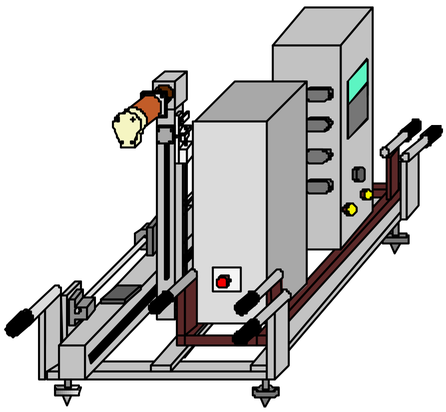

Figure 5.4 Drawing of the penetrometer

The measurements with cone penetrometer are strongly influenced by soil humidity and a little bit less by bulk density. The accuracy depends on the kind of penetrometer and on soil variability. With a hand instrument, more than fifty measurements are sometimes necessary on a parcel to have a sufficient accuracy, while with an instrument driven by a stepper motor, only ten measurements are sufficient to reach the same precision.

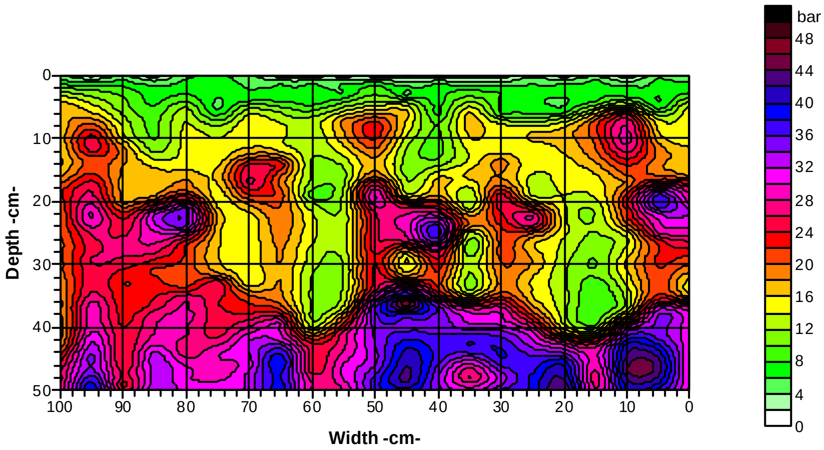

Figure 5.5 Action of a teeth implement on the soil

For the determination of the Cone Index, there may be a great interest to replace punctual measurements by a soil profile on which the action zone of the implement is shown. To display the resistance profile of the soil, we use a penetrometer driven by two stepper motors, one for each axe (figure 5.4). The measurements are regularly spaced (5 cm). With this instrument, a profile one meter wide and 50 to 60 cm deep is taken. The points of same pressure value are linked by a curve (figure 5.5). It is also easier to see the action area of the implement to calculate the cone index of the concerned depth, to put it into the model.

Using dimensional analysis and data obtained in field tests, Upadhyaya et al. (1984) have determined, by regression, a simple equation for draught of tillage implements (Ft) related to depth and width cut, working speed, bulk density and static cone index. The equation is of the form:

$$F_t = B_0 \cdot CI \cdot l \cdot z + B_1 \cdot \rho_h \cdot z \cdot l \cdot v^2$$

where l = width of implement
CI = static cone index
ρh = wet bulk density
z = working depth
v = working speed
B0 = 0,05
B1 = 0,001

5.3 Soil compaction

There is a direct connection between crop/soil management and soil compaction as well as between compaction, tillage, and traffic operations.

Forces applied by tyres, tracks, tillage tools, and vibratory devices, cause soil compaction, modifying the pore volume and pore structure of the soil by reducing mainly the size of the macropores. Furthermore, soil behaviour varies widely from one soil type to another as well as with moisture content.

Especially in climatically deviating years, soil compaction creates a higher energy demand and a greater number of tillage operations. At the same time, yields are decreasing as a result of over compaction.

Ten years ago, Lyne et al. (1989) reached the conclusion that the dynamic load and inflation pressure of pneumatic tractor tyres greatly influence traction efficiency. Results of experiments have shown that the difference between maximum and minimum traction efficiency can be up to 30%.

Implements draught also increases with soil compaction. Traction resistance for ploughing operations (ASAE Standards, 1990), expressed for a single plough section, is given for plough bodies. It is expressed in N/m^2 and the speed in km/h

$$f_t = a + b\,v^2$$

Silty clay (South Texas)	$70.000 + 490\,v^2$
Decatur clay loam	$60.000 + 530\,v^2$
Silty clay (N Illinois)	$48.000 + 240\,v^2$
Davidson loam	$30.000 + 200\,v^2$
Sandy silt	$30.000 + 320\,v^2$
Sandy loam	$28.000 + 130\,v^2$
Sand	$20.000 + 130\,v^2$

For an increase of 0.1 g/cm^3 of the soil bulk density or 1% of the soil moisture content, the traction resistance has to be respectively increased and decreased by 10%. Our services have measured a draught resistance difference up to 40%, with various soil conditions when soil density varies from 1.30 g/cm^3 to 1.70 g/cm^3.

Therefore, keeping these aspects in mind, there is a need to reduce soil pressure to improve soil conditions and to reduce energy consumption. In earlier times, most of the soil compaction occurred during soil cultivation and could be removed by normal tillage practices. Equipment increased in size and weight, and traffic now extends to many other operations. This dramatic increase in power, weight and numbers of operations not only affects the cultivated layer, but may also cause gradual deterioration of the subsoil structure. There is a need to develop systems that avoid subsoil compaction. Research is therefore particularly required to determine the relationship between wheel and traffic tracks, the action of tillage tools on the one hand, and soil compaction on the other hand.

The pressure at one spot below the surface of the soil is a function of the contact pressure and the area over which the pressure is applied (total load). It was commonly believed that an increase of dimensions of the tyres, section width or diameter, would not result in compaction increase as long as inflation pressure could be kept constant. The view that constant inflation pressure gives no increase in compaction has been discredited, at least for the situation when the total load has not been taken into account. Currently, it could be considered that the zone of critical compaction occurs deep below the wheel rut, increasing with the tyre width.

In the past, frost and thaw were believed to relieve the negative symptoms of compaction which may be true as long as the total weight of the implement only influences the topsoil or the ploughed layer. Trials in Belgium with shallower tillage showed that the previous sole plough remains unchanged after 15 years of non ploughing.

When compaction below the ploughed layer has been recognised, subsoiling was tried by many farmers to relieve these problems. Beside the high level consumption for this operation (more than 40 l/ha), results have been variable. The reason may be differences in weather and soil conditions.

The weather interaction in the relationship between soil compaction and plant growth is evident. A plant is a living system and its development is influenced by the ability to obtain sufficient requirements from its environment at the correct time. It is quite possible to produce good yields in very poor physical soil conditions, when for example rainfall occurs daily during critical periods.

Moreover, soils that have been subsoiled are very vulnerable to further compaction. The cost of subsoil amelioration can therefore only be justified if no significant recompaction is anticipated.

The use of the rut depth to characterise the soil compaction has the advantage of simplicity but does not provide data related to changes within the soil. There is no single approach on theoretical or practical grounds that will provide information about the nature or the distribution of these changes. Many relevant properties should be measured, as circumstances permit.

These are:

- for the soil: dry bulk density, porosity, permeability and diffusivity, strength, cone resistance, shear strength, surface bearing strength, soil surface and subsurface deformation, stress distribution, clod and aggregate characteristics, textural analysis;
- for the tyre/track: load, slip, contact areas and contact pressures on hard and soft surfaces, tyre deflection, impact loading, static loading, soil water status and compactability of field soils.

Regarding to rolling resistance, it has been pointed that two main parameters intervene in this phenomenon. They are linked to tyre characteristics, mostly inflation pressure and to soil bearing capacity, as shown in table 5.3 (Dwyer et al., 1987).

Using a profilemeter (figure 5.6), with two screw jack driven by stepper motors for the shifting along axes and with a laser cell to measure the distance from the soil to the frame, one can analyse the soil deformation for different types of tyres, different pressures and loads.

In figure 5.7, we can see the negative footprint of a tyre. Fixing a reference level, it is possible to determine the deformed soil volume and the link with the tyre rolling resistance. The projection on a plane gives the representation of the level curves and an idea of the pressure distribution on the contact area, combined with the profile resistance (figure 5.8).

Table 5.3 Rolling resistance for different driving tyres, inflation pressure, and load

	Front wheels		Rear wheels		Rear wheels	
Tyre type	16.9/14-30		18.4/15 – 38		18.4/15-38	
Load	2280 kg		3260 kg		2860 kg	
Inflation Pressure	1.3 bar		1.4 bar		1.1 bar	
Field conditions	kN	%	kN	%	kN	%
Dry grassland	1.8	8.4	2.7	8.4	2.3	8.2
Dry stubble	2.0	8.9	2.9	9.0	2.4	8.6
Wet stubble	2.4	10.7	3.5	11.0	2.9	10.3
Dry loose soil	2.6	11.6	3.8	11.8	3.2	11.4
Wet loose soil	3.6	16.1	5.4	16.8	4.4	15.7

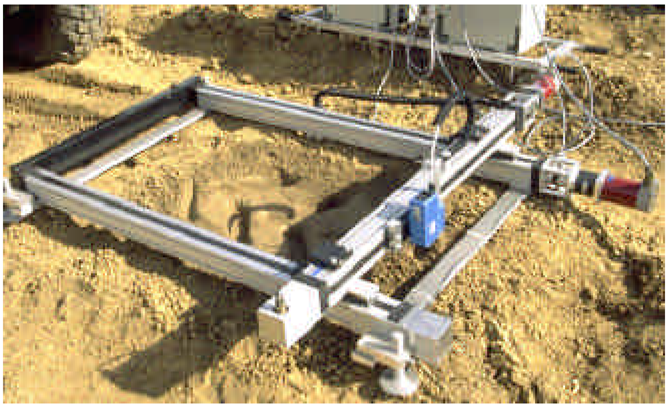

Figure 5.6 View of the profilemeter

This presentation allows the comparison of several types of tyres related to load and pressure. In figure 5.9, two tyres of different sizes were used to support equal loads with the same inflation pressure. We can see that the deformations are different and that the pressure is more uniform with wider tyres despite the fact that the inflation pressure could be lowered for the wider tyre and therefore would give an even better distribution of contact pressure.

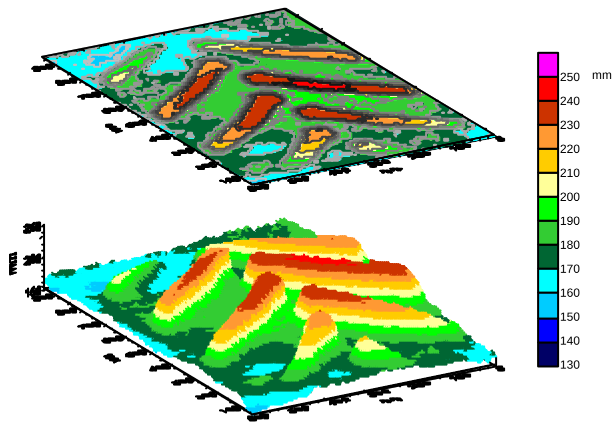

Figure 5.7 Footprint of a tyre with projection of level curves

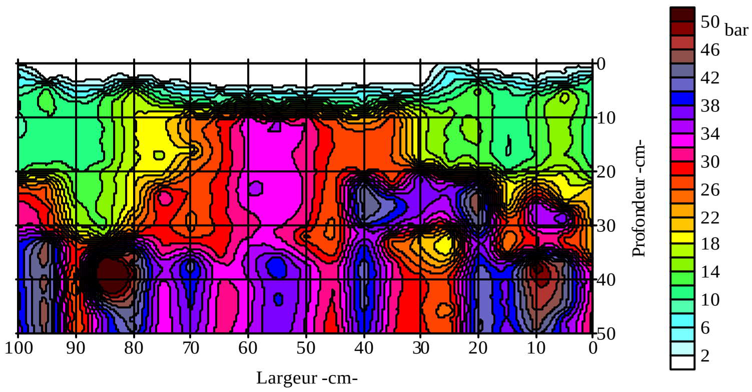

Figure 5.8 Compaction profile after travel of tyres inflated at 3 bars

Research results should again call attention to the practice of using tyres with a low ground pressure to support heavily loaded vehicles on cultural soils. If the total load on the tyres is above some critical level, then the subsoil may be severely compacted regardless of how the weight is spread over the soil surface by the tyres. Axle loads above 6 tons may result in compaction at a depth below 40 cm.

Combined with the profile resistance, these results suggest that there is an optimum level of soil compaction in relation to crop performance and economic returns.

It is evident from the foregoing that the reduction of compaction is essential for a major reduction in tillage energy inputs and to the maintenance and improvement of crop yields. Modern cropping systems are based on agricultural machinery and this equipment is responsible for most of the soil compaction.

If the rut is reduced by two (14 to 7 cm), the working depth is reduced in the proportion and the energy consumption is decreased by 40 to 50% for seedbed preparation.

As a result of a lower rolling resistance, an increase in traction efficiency is also obtained. To calculate this improvement, the formulae presented by Gee Clough (1980) can be used. Reduction of rolling resistance (and slip) results in an increase of speed, reducing the costs of more than 10%.

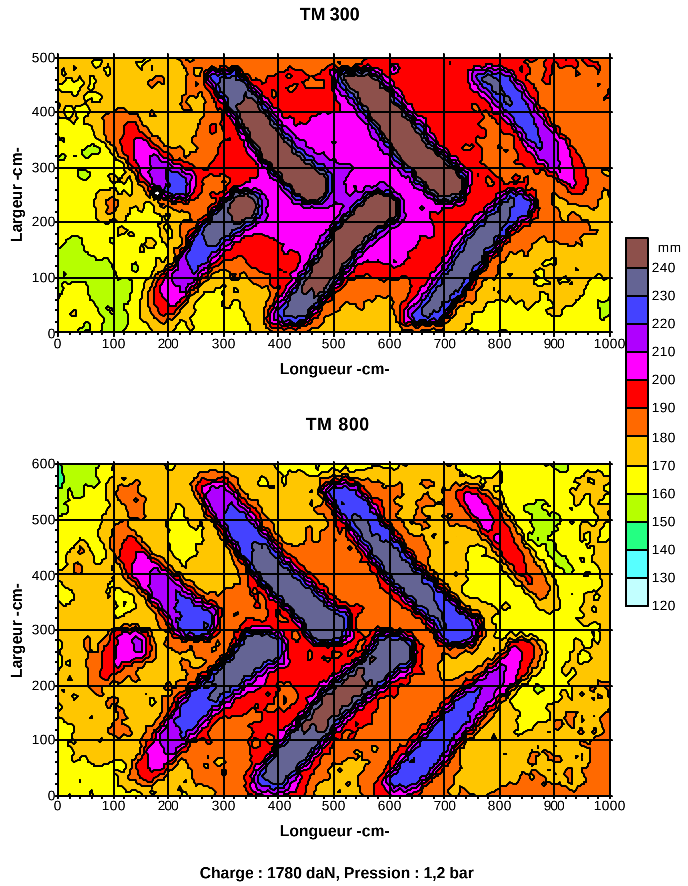

Figure 5.9 Comparison of two types of tyre, for the same load and pressure

5.4 Tractor/implement combinations

For soil cultivation, in most cases, less than 60% of the total power available is utilised. For cultivation, an improvement in the tractor/implement combination will produce a large return as an improvement in efficiency, with particular reference to the medium to large tractor coupled with a plough.

All the other mechanised field practices represent approx. 60% of the total tractor consumption of diesel oil per year. For these operations there is a need to optimise the tractor/ implement combination and to give farmers simple procedures to choose the best implement dimension for a specific tractor, taking into account the three main types of power transmission from tractor to the implements: by traction, by PTO and by hydraulics.

Because of the high technical standard of tractors and implements, energy savings by improving tractor and machine design are expected to be minor compared to cost reduction by reduced cultivation. But there is no doubt that it is precisely this high standard that makes it difficult for the farmer to utilise this sophisticated technology efficiently. Improvement of the efficiency of tractors and implements and reduction of costs require both information technologies to enable the farmer to make optimum and timely use of the technology and a simple procedure to be used by farmers for their choices, to select and match tractors and implements most efficiently.

Electronics are the hardware of the information technologies, the knowledge for the models are the basis for the software; data from field tests provide the real link between the software and the physical reality. Currently, technology is ahead of available data.

Theoretical considerations, computer simulation, and field tests indicate that energy savings seem to be likely at different levels:

- engine: the minimum specific fuel consumption of a good tractor is at present less than 220 g/kWh. It can be assumed that further improvements in tractor diesel engines are still possible;
- power train: it is well known that the increasing number of gears and more shift comfort increases labour efficiency but decreases the power train efficiency. Taking these tendencies into account especially for a bigger shift comfort it is doubtful whether the power train efficiency will be significantly improved (shift comfort is not a luxury, it is necessary to enhance farm work efficiency and relieve the operator);
- maintenance: several field tests show that tractors and implements in working condition have significantly higher fuel consumption. They also show that by adequate maintenance this fuel consumption can be reduced. Fuel savings of 5%, in some extreme cases even 25%, are possible. For example, when the air flow is reduced about 7 to 22% because of a restricted air filter, the fuel consumption may increase up to 10 to 20%. Tests on tractors in use in Belgium show that more than 40% have specific consumption higher than 5% above the references (figure 5.10);
- power/mass ratio: computer models show that for draught work an increase of the tractor mass by 10% by ballasting can result in fuel savings of about 3%, which every farmer is aware of, but this is not without problems in respect to soil compaction. Consequently, inflation pressure of the tyres must be adapted;
- 4 wheel drive: using the results of tyre tests prove that even on dry soil, fuel savings up to 5% are possible by using 4 wheel drive; by blocking the transmission each time it is possible save 2 or 3% slip of the wheels;
- engine loading: whenever total engine power is not necessary to perform a particular task, fuel can be saved by using lower engine speed, well known as 'gear up; throttle down'. Energy savings up to 25% can be achieved, but it is necessary that the right gears are available to use low engine speed.

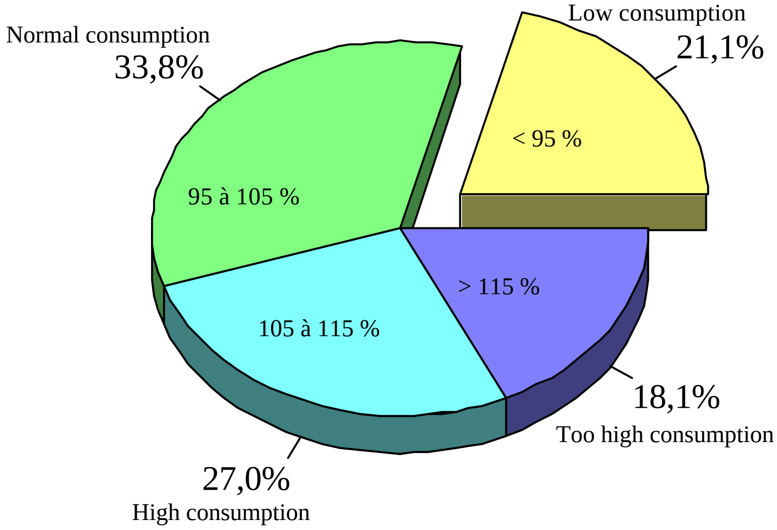

Figure 5.10 Specific energy consumption of Belgian tractors related to reference

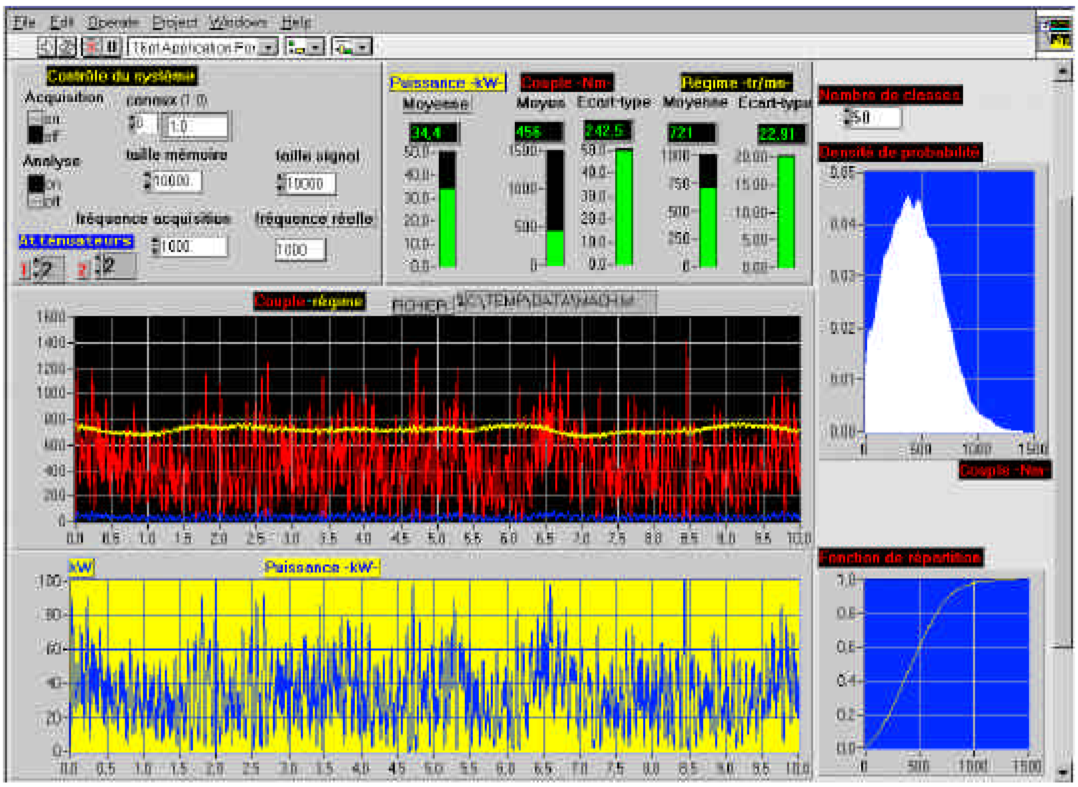

Figure 5.11 Determination of torque, speed, power and probability density of a rotary harrow

Unfortunately, it is not permissible simply to add the percentages of the different savings, but it may help to identify the most promising topics.

To have a good traction/implement combination, it is necessary to know not only the mean of the constraints, but also the probability density, to determine the load rate of the tractor (figure 5.11).

5.5 Proposals for estimating fuel consumptions

Numerous data exist on energy consumption in agriculture. Those data depend on the manner they have been measured, on the matching of tractor and implement, on the load rate of the tractor and on several parameters which are difficult to evaluate. These parameters are related to the soil (type, composition, moisture content, etc.), to the machine employed (type, weight, tyres, etc.) and finally to the crop itself.

It would be reasonable to cover different crops and cultivation treatments but, at the same time, to include both compaction and tractor/implement aspects.

Models for calculating fuel consumption exist. Generally, they are used to determine interactions between the soil and the machine. The evaluation parameters are mostly based on field experimentation with the help of specially designed testing benches to determine the influence of each parameter, one by one, to be included into a general formula.

In 1974, R.D. Wismer and H.J. Luth introduced a model about off road traction prediction for wheeled vehicles, involving the load (W), the towed force (TF), the pull (P), the torque (Q), the slip (S), and a wheel parameter (Cn), which depends on the unloaded tyre section width (b), the unloaded overall tyre diameter (d), the load (W) and the cone index (CI), which expressed the soil properties. Equations have been written for towed and driving wheel considering three soil classes: purely cohesive, purely frictional and cohesive frictional soils. The general equation of R.D. Wismer and H.J. Luth expressed the traction efficiency:

$$TE = \left\{ 1 - \left[\frac{\frac{1.2}{C} + 0.04}{0.75\left(1 - e^{-0.3 C_n S}\right)} \right] \right\} (1 - S)$$

The main interest into this relation is the use of the Cone Index (CI) which corresponds to what could be called the soil strength (consistency) or soil supporting stress. Cone Index is the force per unit base area required to force a cone shaped probe into the soil at a steady rate.

In 1975, M.J. Dwyer presented a relation in which the geometric characteristics of tyres are included in the ratio of the tyre deflection and of the tyre section height.

$$TE = \frac{CI \,.\, b \,.\, f \left(\frac{d}{h} \right)^{1/2}}{W \left(1 + \frac{b}{2f} \right)}$$

With : ϕ = tyre diameter
δ = tyre deflection
h = tyre section height

In 1980, D. Gee Clough gave equations for the calculation of the maximum coefficient of traction and the coefficient of rolling resistance. In both equations, the cone index is also employed, as well as the characteristics of the tyre (dimensions and inflation pressure). Recommendations were given about the relations between the drive tyre load, the speed and the power available, between the tyre size, the load and tyre inflation pressure and between the draught load and the wheel slip. In the future, new researches have to be managed to adapt the old existing models to new tyres type.

To determine the traction efficiency of tyres, it is recommended to check on graphs into which curves represent the traction efficiency as a function of wheel slip. Obviously, the larger the traction efficiency, the more the energy consumption is optimised.

Figure 5.12 Single wheel tester

Looking at figure 5.13, one can see that to calculate the fuel consumption for a specific task, one could involve about 40 different parameters. However, to ease calculation, the most important parameters must be identified instead of the ideal parameters.

The first step is to establish an internationally recognised method for the fuel consumption measurements, for field measurement as well as for laboratory or bench tests. The purpose is to build a model which can be used abroad, thanks to its base or its structure which must be standardised. The tractor that will serve to the field measurements has to be tested before. The curves have to be established at different partial loads (figures 5.1 and 5.2). On the field, the fuel consumption has to be measured at the same time as motor revolutions and fuel temperature.

In a second step, the most important parameters have to be isolated from the large non exhaustive list given above. An important parameter is one, which have a major influence on energy consumption and does not depend on a specific situation. Then, one has to determine the ability of those data to be measured easily.

The final step is to establish a model, which includes those important data and leads to an evaluation of the energy consumption corresponding to a particular situation. Comparing the values of the model with those of the farm, we may calculate a correction factor.

Parameters	Evaluation	Parameters	Evaluation
Soil type	On the field and in laboratory	Tyres type	In technical specifications
Soil cone index	On the field	Tyres size	In technical specifications
Soil water content	In laboratory	Tyres pressure	On the field
Soil composition	In laboratory	Traction efficiency	On the field
Working season		Weight and ballast	On the field
Landscape relief	On the field	Tool type	In technical specifications
Field size	On the field	Tool size	In technical specifications
Country	On the field	Tool working depth and width	On the field
Tractor brands	In technical specifications	Fitting equipment to the task	On the field
Tractor type 2 or 4 WD	In technical specifications	Transporting time, speed and distance	On the field
Slip of the tyres onto the soil surface	On the field	Turning time	On the field
Gear ratio	On the field	Repairs, clearing blockage time	On the field
Engine output curves	In laboratory	Farming system	
Power transmission efficiency	On the field and in laboratory	Farming practices	
Transmission type	In technical specifications	Yield	On the field
Working speed / rated speed	On the field	Dosage (fertiliser)	On the field
Loading on the engine	On the field and in laboratory	Fuel temperature	On the field and in laboratory
Energy required at the drawbar or p.t.o.	On the field and in laboratory	Human Factor	

Figure 5.13 Parameters involved in the estimation of fuel consumption

5.6 Conclusions

Energy or fuel consumption calculation in the agriculture is not really simple. The number of parameters that have to be included into formulae is large. They are related to soil properties, to crop type, to power providing machines, to soil working machines, and to interrelation between all of them.

To estimate the amount of energy employed during field operations for example, one can use existing models leading to a gross value. Generally, those models involve several steps:

- mechanical soil properties measurements for having the Cone Index value (penetrometer);
- identification of the technical specifications of the machine, or tractor and implements, including weight, tyres type and geometry, engine curves (torque, power output at the pto, engine speed), specific consumption curves, etc.;
- implements power requirements determination (sensors and specific curves);
- wheel slip evaluation (sensors);
- traction efficiency calculation;
- rolling resistance evaluation (profilometer and penetrometer);
- implements efficiency measurements;
- total energy demand calculation (traction and field operation), in terms of power (kW/ engine speed);
- fuel consumption determination (l/h or l/ha) regarding to the total power requirement.

Machine parameters can be evaluated thanks to charts or graphs determined in laboratory and measured on the field with the help of sensors. A global model, in which parameters correspond to a specific field operation, do not yet exist. Therefore, the use of sensors is still necessary, until enough values are provided for a database.

Regarding soil properties, authors have determined categories of soil types with parameters to be included in models, but it could be more precise to measure them directly with penetrometer.

The following topics have to be considered with priority:

- energy requirements for implements and machines, in different soil conditions (region or country);
- determination of the relationship between soil conditions, ground pressure, axle load and the depth and severity of soil compaction;
- building of a model including interactions between soil, implement, tractor and crop;
- application of the model in a few well-known conditions, where practical data are available, in order to compare and correct the model.

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6. Direct and indirect energy use in arable farming - an example on winter wheat in Northern Germany

J. Moerschner and B. Gerowitt [1]

Abstract

Energy use in agriculture receives increasing interest. There still is a lack of accessible, complete life cycle inventory data sets on farm level. Data should be documented based on substance and energy flows within the whole considered farming system. A short description of the energy coefficients selected for calculation is given, since the way of generating the energy coefficients for each agricultural supply has an important influence on the calculation results. In this study, all energy input finally is considered as primary energy use and it is budgeted at the farm gate. As a case study on energy budgeting in agriculture, production data of winter wheat, obtained from the INTEX project at the University of Göttingen, is used for the calculations. N fertiliser and diesel fuel use are identified as the main energy input factors (about 65% of total together), tillage, harvesting and storage processes as the operation categories with main influence on energy use for mechanisation. The discussion focuses possibilities to generalise and to improve the assumptions in the case study as well as occurring problems.

6.1 Introduction

Initialised by the world energy crisis in the seventies, figures on energy use and energy efficiency of agriculture appeared in scientific contributions in Germany (Lünzer, 1979; Weber, 1979). Rough assumptions or averages were often used in order to show major potentials for saving energy and to increase energy efficiency in agriculture (BML 1979, KTBL 1987).

Often fossil and renewable energy sources (agricultural ones like biomass or non agricultural ones like wind or direct solar energy) were compared (e.g. Hartmann and Strehler 1995). Recently energy budgets in agriculture are calculated in order to identify and promote sustainable farming practices, which should help to save limited resources (e.g. Moerschner et al., 1997a, b, Geier et al., 1998).

6.2 Calculating direct and indirect energy use data required

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A main problem of various presented agricultural energy budgets is that some of the used energy coefficients are not yet directly connected to the underlying substance flows, which are suggested as basic references in point 5.1.2.2 of the ISO standard 14040. Therefore, such documentation can hardly be used unprocessed within LCA according to the ISO standards 14040-14043. The link between substance and energy flows is definitely required when energy consumption as well as connected emissions and depletion of natural resources are to be studied in combination.

As another main task, energy input for LCA purposes should be initially expressed in percentages on end energy level; primary energy use may be derived from this figures afterwards. Details which have been included into the underlying process chains must become clear by the documentation (system borders, e.g. transports taken in account or not), in order to have an equal level for further budgeting. In this study, all energy input finally is considered as primary energy use and it is budgeted at the farm gate. For most industrial inputs, the primary energy coefficients were gathered by transforming end energy use figures along the process chains reported in literature into primary energy use.

6.3 Energy coefficients and agricultural energy use data

Any calculation of energy use requires energy coefficients for all relevant variables in the substance flows. Aggregated primary energy coefficients of inputs and outputs required for calculating agricultural energy use are given in table 6.1.

Table 6.1 Energy coefficients used in the case study for supplies, grain yield and storage processes (inputs: MJ primary energy per functional unit)

Main category	Sub category	Energy coefficient	Functional unit	Source, comments
Diesel fuel, oil, lubricants		47.3	MJ/kg	Kaltschmitt and Reinhardt 1997 a)
Electricity		11.4	MJ/kWh	Kaltschmitt and Reinhardt 1997 a)
Fertiliser	N	47.1	MJ/kg	Kaltschmitt and Reinhardt 1997 a)
	P_2O_5	15.7	MJ/kg	Kaltschmitt and Reinhardt 1997 a)
	K_2O	9.3	MJ/kg	Kaltschmitt and Reinhardt 1997 a)
	CaO	2.1	MJ/kg	Kaltschmitt and Reinhardt 1997 a)
	MgO	0.00	MJ/kg	contained in K_2O fertilizer, own assumption
Pesticides	active ingred.	274.1	MJ/kg	Kaltschmitt and Reinhardt 1997 a)
Seeds	winter wheat	2.5	MJ/kg	Kaltschmitt and Reinhardt 1997 a)
Machinery		70.5-92.5	MJ/kg	Scholz and Kaulfuss 1995
Storage processes		0.17	MJ/kg	Diepenbrock et al., 1995; 15 kWh/t grain yield
Grain yield, wheat		14.5	MJ/kg	Brenndörfer et al., 1995; 85% dry matter

a) Slightly adapted by own calculations.

Figures on *diesel fuel use* for specific field operations differ seriously, because of:

- differences in tractor make (brands);
- differences in demand for mechanical energy for specific cropping activities (e.g. one hectare ploughing versus spreading of pesticides).

In addition, working depth, soil types, slope, and field size cause variation in diesel fuel use per hectare at the same work category. Since these variations have not been quantified in detail in Germany yet, actual calculations normally are done with average figures for diesel fuel use per hectare or per working hour. Either the average mechanical energy needed for a specific field operation is taken as reference for the diesel fuel use (e.g. 220 g/kWh of power, Hartmann and Strehler 1995), or the average tractor power category used on the farm for a specific field operation (e.g for a 80 kW tractor about 9.2 kg/h, Scholz and Kaulfuss, 1995; KTBL, 1994; Moerschner et al., 1997a b; Moerschner and Gerowitt, 1998).

The working hours for specific field operations are taken from standardised data bases, which provide figures of a large variety of different cropping practices and machinery sizes/ types and also of variations in field size (KTBL, 1994).

In this study calculation of direct energy use per hectare for each cropping activity follows equation (1):

$$ED = h * AFU * PEU * RU \qquad (1)$$

ED = Direct energy use (diesel fuel and motor oil) for a specific cropping activity (primary energy, MJ/ha).

h = Specific working hours per run (h/ha, KTBL, 1994).

AFU = Average fuel use per working hour and used tractor category (kg/h, 2% for motor oil added, KTBL, 1994; Scholz and Kaulfuss 1995).

PEU = Specific primary energy use per kilogram diesel fuel and motor oil, calculated from the use of end energy sources, including the process chain for production (MJ/kg, Kaltschmitt and Reinhardt, 1997).

RU = Runs, number of applications in the considered cropping period (1 to n apps., from cropping data).

Lubrication of motors is assumed 2% of the total diesel fuel use and the amount of motor oil is added to the direct energy use (KTBL 1994).

Fertilisers: for calculating the energy use for fertilisers (nutrients N, P, K, Mg, Ca) under German conditions data from Kaltschmitt and Reinhardt (1997) are considered. They represent German averages of origin and nutrient content, assuming furthermore typical production processes and efficiencies per kilogram of each nutrient. Original data provide figures on specific end energy use, which were then converted into primary energy use. For Mg it is assumed, that the nutrient is contained in the spread amount of potassium fertiliser (40% K_2O, 6% MgO) with no need for extra energy input. The amount of basic fertilisers spread per hectares also adapted to average conditions, though detailed figures are documented in the INTEX project. Hereby, only the average nutrient export per t dry matter grain yield in the referred year is taken in account (8 kg P_2O_5; 6 kg K_2O; 2 kg MgO; HYDRO agri 1993). Resulting figures are documented in table 6.2. The amount of Ca fertilisation is taken from LK Hannover (1997). Possible soil melioration aspects have not been considered. Spreading of basic fertilisers is assumed as one time in the crop rotation for each nutrient (all three years, K and Mg together). Therefore, only a percentage (33%) for those activities is taken in account in the referred year. Manure was not applied.

Pesticides: an average of 274.08 MJ/kg active ingredient for the production of pesticides is used, relaying on data for about 40 substances (Green, 1987, adapted by Kaltschmitt and Reinhardt, 1997). The energy use is considered to be primary energy use, the average is derived from end energy use data.

Seeds: because seeds are primary agricultural products, energy consumption for the production of seeds for wheat can be approximated by an iteration of known or virtual standard production processes. Energy use for a special treatment of seeds (more plant protection, sharper cleaning, dressing, and packaging) and for transportation to the farm is included (Kaltschmitt and Reinhardt, 1997).

Machinery: the contribution of machinery to the total energy demand on the farm varies, depending on farm size and type, the number of different crops within the rotations, the cropping intensity and the level of mechanisation. In the presented study, average figures are used, based on Scholz and Kaulfuss (1995). In detail, they are calculated for each cropping activity following equation (2).

$$EID = ((TW * CED)/UL) * h * RU \qquad (2)$$

EID = Indirect energy use for a specific cropping activity (MJ).
TW = Total weight of the specific machine (kg, Scholz and Kaulfuss 1995 and other sources).
CED = Cumulative energy demand, accounted according to VDI 1997, energy for space requirements for machinery housing as part of maintenance included (MJ/kg, Scholz and Kaulfuss 1995).
UL = Assumed total use in lifetime (self propelled machinery: total h, other machinery: total ha, KTBL, 1994 and Scholz and Kaulfuss, 1995).
h = Specific working hours per run (h/ha, KTBL 1994).
RU = Runs, number of applications in the cropping period (1 to n apps., from cropping data).

Storage processes: a demand of 170 MJ/t net grain yield (at 85% standardised dry matter content), mainly for cold ventilation was found in literature for all mechanical processes of storing (storing, cold ventilation and cleaning; Diepenbrock et al., 1995). Assuming *electricity use* the figure is transformed into a coefficient for end energy use dividing the 170 MJ/t grain yield by a factor of 11.4 MJ/kWh. Indirect energy input in this part of the process chain was not considered since no figures were available. Grain drying with heated air is not considered, since it can be avoided in the region of Göttingen under normal harvest conditions.

Grain yield: grain yield is transformed into energy terms by its specific lower heating value (H_u at 85% dry matter content; Brenndörfer et al., 1994).

6.4 Production inventory of winter wheat, a case study

Cropping data are collected in the INTEX project at the University of Göttingen on single field level. Detailed information's about the conceptual background and on recent results of the whole project are given by Gerowitt and Wildenhayn (1997). A typical Northern German three year rotation consists of rape seed, winter wheat, and winter barley. The example origins from the arable farming system called 'Good Farming Practice', representing a reference system, grown at the location Reinshof, nearby Göttingen.

Generally all reported substance and energy flows refer to the farm gate as system boarder. For the energy input evaluation, the whole production chains for supplies are included, as far as reported by the corresponding authors (see section 6.3). As functional unit, one hectare of arable land is used, cultivated with winter wheat. All energy budgets are calculated on field level (figure 6.1). The field size is 3.2 ha, field to farm distance is assumed to be one km. The reference cropping period is 1996/97, beginning with stubble cultivation and ending with the storage, ventilation and cleaning of the grain yield. The straw was not used, but chopped and left on the field.

Table 6.2 Aggregated production inventory per hectare of winter wheat production

Main category	Sub category	Quantity/ha	Comments	Data quality a)
Seeds	winter wheat	220 kg		C
Mineral fertilizers	N	170 kg	CAN	C
	P_2O_5	61.68 kg	TSP, as nutrient export	C/L
	K_2O	46.26 kg	MOP, 40%, as nutrient export	C/L
	MgO	15.42 kg	in TSP, as nutrient export	C/L
	CaO	300 kg	according to reg. extension advice	L
Pesticides	active ingredients	1.46 kg	total amount, incl. seed dressing	C
	whole pesticides	4.04 l	total amount, incl. seed dressing	C
End energy use	diesel fuel	83.90 l	= 100,48 l (densitiy: 0,835 kg/l)	O/L
	lubricant motors	1.68 kg	2% of diesel fuel	L
	electricity	135 kWh	storage, ventilation, cleaning	C/L
Human labour	working hours	4.73 h	hours of machinery use counted only	O/L
Capitals	machinery	8.36 kg	material depreciation as in economy	O/L
Yield	grain yield	9071 kg	average of yield reference plots	C

a) C = Cropping data; L = Average figures from literature; O = Own calculations, usually combined from more than one source.

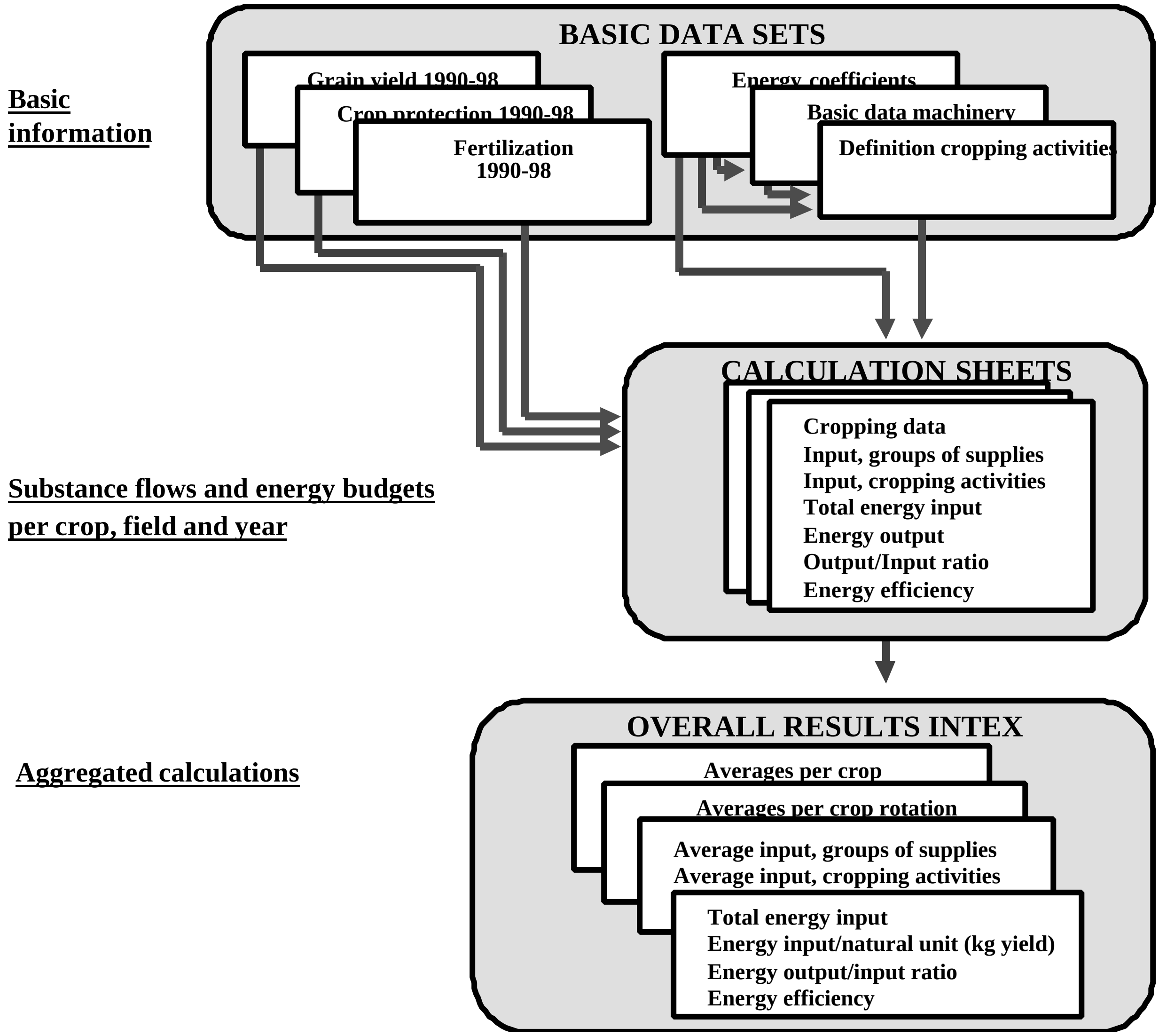

Figure 6.1 Calculation matrix of the INTEX energy budgets

The following field operations were undertaken during the cropping period 1996/97:

- Basic fertilisation Mechanisation: from the whole rotation calculated (1/3)
- Rotary spade harrowing 1 run
- Ploughing 1 run
- Rotary harrowing + mounted drilling machine 1 run
- N fertilisation 3 runs
- Plant protection 5 runs
- Harvest (threshing, transportation to farm)
- Storage on farm (storing process, two times cold ventilation for drying, cleaning)

Table 6.2 gives an aggregated extract of the reported information on substance flows.

The figures in table 6.2 can be processed for aggregation in energy terms (MJ/ha) with the corresponding energy coefficients as suggested in table 6.1.

6.5 Results

Two ways of presentation of the results have been selected to illustrate the main influences within the total energy budget (figure 6.2 and figure 6.3).

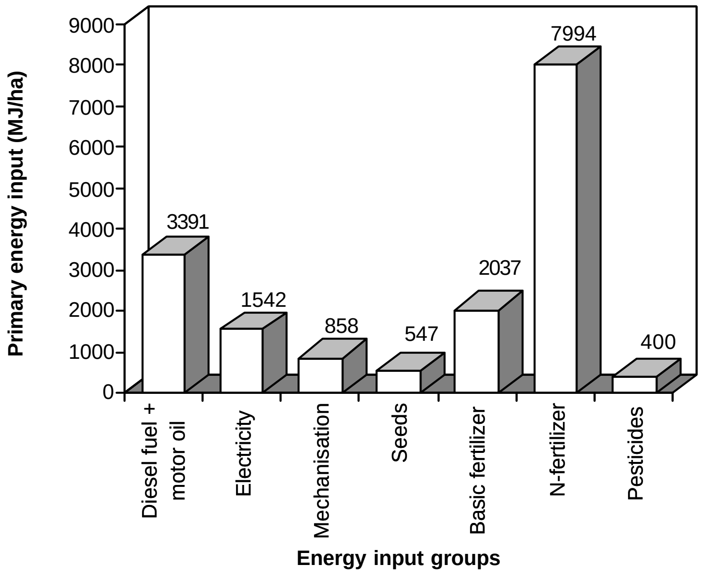

Figure 6.2 Total primary energy input (MJ/ha) split into different groups of supplies in the selected wheat cropping system

Source: Reinshof, 'Good Farming Practice', (1996/97).

Total primary energy input adds to about 16,780 MJ/ha. As main consumers of energy the N fertiliser (7,994 MJ/ha; 47.7%) and the diesel fuel use (3,390 MJ/ha; 20.2%, motor oil included) are identified (figure 6.2). Additional energy input is observed for basic fertilisers (2,040 MJ/ha; 12.1%) and for electricity (about 1,540 MJ/ha; 9.2%), whereas pesticides, seeds and indirect energy for machinery use have only marginal importance (figure 6.2, bars 3, 4 and 7).

Focusing on the part of energy use for mechanisation (direct and indirect energy for machinery use), the main energy is used in field cultivation activities (1,590 MJ/ha; 27.4%, seeding included), harvest (1,500 MJ/ha; 25.9%), and storage processes (1,540 MJ/ha; 26.6%, most of this for ventilation, only direct energy use included), while spreading of fertilisers and pesticides has only little importance within total mechanisation (figure 6.3, bars 2 and 3). Indirect energy use for mechanisation takes 14.8% from total energy use for mechanisation, including diesel fuel, motor oil and electricity use (figure 6.3). Compared to total energy demand indirect energy use for mechanisation is even less important (5.1%, figure 6.2, bar 3).

Grain yield was 9,071 kg/ha representing an energy output of 129,715 MJ/ha, that is 112,946 MJ net energy yield or 7.75 MJ output per MJ input. Energy input per kilogram grain yield is 1.85 MJ/kg.

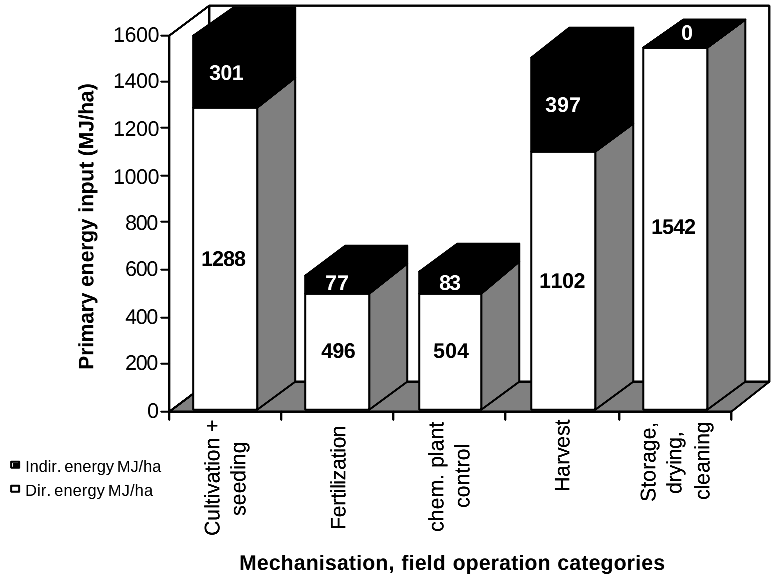

Figure 6.3 Energy input (MJ/ha) for mechanisation (direct and indirect energy) split into different groups of cropping activities in the selected wheat cropping system

Source: Reinshof, 'Good Farming Practice', (1996/97).

6.6 Discussion

Aspects of energy use in agriculture became most obvious to farmers as direct energy use on the farm (see figure 6.2, bars 1 and 2). In contrast, in intensive cropping systems a much higher amount of total energy use (in the example about 65% of total energy input) is caused indirectly (see figure 6.2, bars 3-7). Therefore, calculations including direct as well as indirect energy use, like the one presented here, are basically required for any discussion on energy use and energy efficiency in arable production. Separating these two ways of energy use will consequently lead to wrong conclusions about the most efficient way of saving energy within farming activities.

Considering data quality, three characteristics with individual influences on the results are most important: Variation and representativeness of the used cropping data, variation and quality of the energy coefficients and the way of connecting the two in algorithms for individual calculations. Since it is impossible strictly to separate these three aspects, the following discussion is structured according to the different operational supplies.

Current approaches to assess *diesel fuel use* should be improved. A combination of different models suggested in recent literature is likely to give a more detailed way of gaining relevant data for specific cropping activities (Maeyer et al., 1995; Audsley, 1999; Nielsen, 1999; Vitlox, 1999) on different levels of aggregation. However, such more theoretical calculations should be always practically verified by direct investigations of fuel use on working tractors.

For modelling *indirect energy use for machinery*, the effective overall service time of a specific machine on a farm during its lifetime is very important. KTBL (1994) and Kalk and Hülsbergen (1996) give some rough orientation, but in reality a discrepancy will remain between the given economic depreciation time and 'real' times of use, depending on farm size and the level of mechanisation and cropping intensity. However, on a more generalized level of energy budgeting for arable farming this part will usually be of minor importance (see figure 6.2) and therefore might be taken in account with average assumptions per hectare. Nevertheless, differences in energy use between farming systems will appear also due to mechanisation intensity (see figure 6.3).

Hot air drying of grain was excluded in the example, partly for methodical reasons. For drying processes, standardised average figures are available for end energy use per amount or percentage of withdrawn water. Different drying options appear for cereals, oil seed rape, maize, or other arable products (e.g. Hydro Agri 1993). However, on a larger scale it is difficult to decide, which part of the whole harvest should be assumed each year to be dried in different regions. Furthermore, the grain moisture at harvest, which requires drying for storage, must be defined. This depends on contracts between farmers and the grain using sector as well as on fundamental quality criteria. Finally the demand of energy per difference percent of moisture content depends on starting grain moisture and is not linear. If hot drying becomes necessary, a considerable percentage of total energy input can be expected for this process, e.g. Audsley et al. (1997) calculate 10% of total energy input.

Considering the energy use in *fertiliser production*, recent figures from Kaltschmitt and Reinhardt (1997), Patyk and Reinhardt (1997) and Kongshaug (1998) should be improved by more specific data from the fertiliser industry. More details about the process chain definitions are required. Average values for energy consumption of N fertilisers per kilogram nutrient content cannot reflect the on farm situation. The individual energy input by N fertilisers strongly depends on the specific formulation of the used fertiliser, considering especially the individual production processes and on the origin of the fertiliser. Therefore the specific N fertilisers used and their amounts should be reported. Own sensitivity analysis showed a reduction in total energy input of about 2,000 MJ/ha using figures for individual N fertilisers from Kongshaug (1998) instead of the averages from Kaltschmitt and Reinhardt (1997). Relying on a mean energy coefficient for total N fertilisation will make the calculations easier to handle on a higher level of aggregation (e.g. EU). On the other hand, it can be expected that this give only a very rough reflection of the real conditions.

Calculating energy consumption for *plant protection*, the average energy use per kilogram active ingredient as used in this case study will be sufficient for most applications of energy budgets, because pesticides are of little importance for the total energy demand in arable farming systems. In fact, a great variation in energy consumption for the production and formulation of different active ingredients in pesticides was observed, without correlation to their use in agriculture (e.g. herbicide, insecticide; Green 1987). Therefore, for more detailed investigations on energy budgets the energy use for each active ingredient applied during the cropping period is required. Here, actually only little information is available.

The amount of *seeds* applied per hectare is rather constant for each crop. KTBL (1994) makes differences between best, medium and bad locations, so the amount might be standardised for those three groups for modelling German relations on a European level without loosing too much of important information.

The underlying data for the calculations here belong to a large scaled farming systems experiment running since 1989 all management practices are therefore precisely documented. However, the absolute figures presented here also agree with more generalised calculations on use of supplies and energy in German wheat production systems (e.g. Scholz and Hahn 1998). The example is given to illustrate, which information generally is required and how the data can be generated by models in some parts. Of course, a case study cannot be representative, but it can show basic principles and problems which will occur everywhere. The derived energy coefficients used for calculations are selected from literature, reflecting German average conditions and are usually based on reported end energy use.

In future, a widely accepted method for energy budgeting in agriculture, including system borders, energy coefficients and the referred level of energy, should be applied to situa tions representing different farm sizes, farming intensities, husbandry practices and natural site potentials. Therefore, an important task is to convince farmers or farmers organisations about the benefits of providing data on farming activities for LCA purposes.

Since the complexity of farming increase with the number of products, focussing on arable farming is not sufficient for final applications. Especially the exchanges between arable farming and animal husbandry (e.g. animal feed, manure, straw) actually cause some unsolved questions, e.g. how to quantify and to allocate substance flows and the energy consumption connected with organic fertilisers spread on arable land.

Appropriate production inventories (substance flows and end energy use) of different farm types are still required to provide reliable and representative data on agricultural energy use, direct as well as indirect, for further application within LCA's in the food sector. The development of a clear and concise framework for energy budgeting in agriculture, which is neither over simplified nor over complicated but able to meet the specific demands of agriculture as an 'outdoor business', built by numerous individual and independent units with an immense diversity of production methods is a challenge within the sector for the next years. It should be at same time transparent and suitable for planning, for comparing and for marketing the agricultural production of food.

6.7 Acknowledgement

Investigating energy aspects in the INTEX project and the developement of more general rules of energy budgeting in agriculture is funded by the Deutsche Bundesstiftung Umwelt (Federal German Environmental Foundation).

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7. Conclusions: data on energy use and fuel emissions in agriculture

P. Cortijo [1]

7.1 Introduction

This paper sums up the conclusions of the group working on 'Data on energy use and fuel emissions in stables, field machinery, irrigation and crop drying'.

The tasks of the working group were:

- to *exhaustively list the parameters* influencing energy consumption ('what are the ideal data');
- to *select the most important ones*, i.e. those whose variations imply major modifications in fuel consumption, as relevant for a European database; and
- to figure out whether the values of these parameters are *available for Europe* ('what is available today?'/'what has to be done in the future?').

The energy consumption dealt with in the working group were:

- field energy consumption;
- drying and storing energy consumption (grain, potatoes);
- energy consumption in stables.

To grasp a complete view of the LCA data linked to energy consumption, the agricultural stages leading to fuel consumption should be listed and, for each stage, the whole life cycle of the fuels consumed should be taken into account (see figure 7.1).

The amounts of energy consumed at these different agricultural stages may be classically designated as the foreground data (e.g. fuel consumption for ploughing) whereas the data related to the upstream and downstream stages (e.g. fuel production and combustion) correspond to background data. Foreground and background data enable to calculate all the consumed resources (petroleum, natural gas, water, etc.), the emitted pollutants and the generated waste linked to energy consumption.

The working group concentrated on the foreground data. European models could first be used for fuel production and current data on emissions from combustion could be completed by data referring to all the normalised European cycles existing for the machinery used in the agricultural sector.

Problems of allocation were not considered as it is assumed that no pre defined allocation will be chosen in the database. Such methodological choices will remain the decision of the LCA practitioner.

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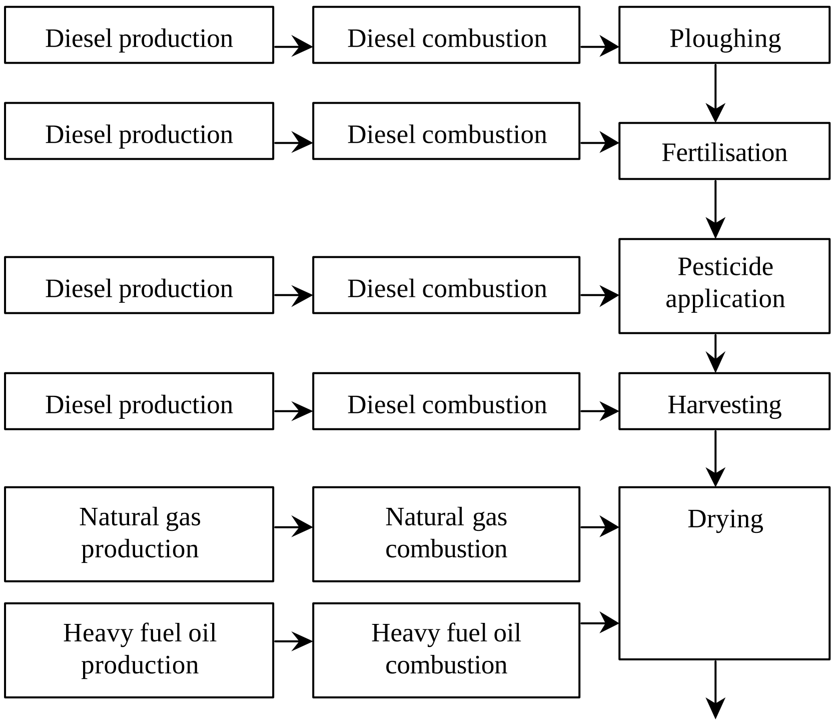

Figure 7.1 Example of stages that should be considered in the database for energy consumption and fuel emissions

7.2 Energy consumption for field operations

Table 7.1 presents the parameters identified by the experts as influencing the amount of energy required for a given yield of a given crop and those which where selected as the most important. This selection was based on an approximation of the maximum variation in energy consumption when the value of the parameter varies.

The main types of field operations are the following:

- preliminary soil cultivation (e.g. stubble cultivation, succeeding harvest);
- primary soil cultivation (e.g. ploughing, heavy field cultivation);
- seedbed preparation;
- planting/seeding;
- fertilising and in cultivation operations (spraying, mechanical weed control, etc.);
- harvesting;
- secondary harvesting.

Table 7.1 Selection of the parameters for field energy consumption

Parameters	Max. var. %	Selection
Number and type of field operations	30	X
Fit of the equipment to the operation (including machinery make)	10	
Behavioural factor (operator factor)	15	
Soil texture type	25	X
Slope	2	
Field size	5	

The working depth of soil cultivation was identified as a key parameter distinguishing between the different cultivation operations.

Figure 7.2 gives, for the selected parameters, the feasibility of the modelling of energy consumption and the availability of values for the selected parameters at a European level.

Selected parameters	Modelling	Parameters value availability
Number and type of field operations	yes : the energy consumption for a given operation may be either measured or calculated according to the energy requirement.	best data source: to refer to farm enquiry default data source: to refer to agricultural expert advice on regional level for building 'typical farming practices'.
Soil texture type	yes	European mapping of the soil texture type
Working depth	yes	best data source: to refer to farm enquiry default data source: experts typology

Figure 7.2 Modelling and availability of values for the parameters selected for field energy consumption

As pointed out in J. Moerschner's contribution (this volume), the setting of the value of energy consumption for each type of field operation is a multi stage procedure: a) modelling of the energy consumption for a given type of operation, b) definition of the average conditions for the referred level of aggregation (regional, national, EU) for a given type of operation, c) identification of the average cropping process for a given crop over the considered region (e.g. EU).

7.3 Energy consumption for crop drying

Figure 7.3 presents the parameters influencing the type and the amount of energy consumed for drying. All of them were identified as key parameters for a European database but, as shown in the table, doubts remain on the availability of some data.

Selected parameters	Modelling	Parameters value availability
Average moisture content at harvest	yes	cooperative: it is assumed that the data are collected but that they are not gathered at a more centralised level
Type of fuel	yes	as above

Figure 7.3 Modelling and availability of values for the parameters selected for drying energy consumption

Energy is also used for conservation of potatoes and other crops. The sole important parameter identified for this was the number of weeks of storage.

7.4 Energy consumption in stables

Three types of animal husbandry have been identified to analyse the parameters influencing energy consumption in stables:

- dairy farms;
- meat production farms (beef, pork, poultry);
- breeding farms.

7.4.1 Dairy cows

Energy consumption in stables has been separated between the main tasks. Figure 7.4 lists the parameters determining energy consumption and indicates which ones were selected for a European database.

It was stressed during the working session that whether the concentrate is produced in the farm or imported would not cause a major change in the overall quantity of energy required to produce 1 kg of concentrate. However, when using the FADN data to check the models, it may be a problem that the location (inside or outside the farm) of the production of concentrate is not indicated. This could lead to an important uncertainty as the total energy consumption for feeding (direct as well as indirect) amounts up to even more than 80% of total energy consumption in some animal production systems. A solution could be to add to the national energy consumption on farms the energy consumed for feeding production outside farms.

Activity	Parameters	Selection
Milking (including cooling)	type of milking parlour	
	milk yield (kg/cow and year)	X
	milking frequency (times per day)	
Feeding (milling, mixing and distribution of the food)	types of feed (concentrate a)/roughage)	X
	amount of each type of feed per animal	X
Cleaning (including manure handling) at farm	type of storage (solid/slurry)	
Bedding		

Figure 7.4 Selection of the parameters for energy consumption in stables (case: dairy cows)

a) The concentrate is a mixture of various components like soy, wheat, barley, oats, vitamins, etc., which changes with the type of animals but also with production intensities, local preferences, world market prices, etc. As a first approximation, a 'standard concentrate' for each type of animal (cattle, pigs, poultry at least) is suggested.

Selected parameters	Modelling	Parameters value availability
Milk yield	Previous studies have shown that the energy consumption per kg of milk was declining with the farm yield (milk production on the farm). Hence, the following relation was proposed : E (milking ; farm) = a + b * yield, or E (milking ; 1 kg of milk) = a/yield + b (a and b are constant value which would have to be defined)	FADN
Type of feed	The above mentioned relation seems also valid for feeding, cleaning and bedding. The farm milk yield is replaced by the number ('n') of animals (including calves, bull) in the cattle a) Hence, E (feeding, cleaning, bedding ; farm) = c + d * n, with c = c1 + c2 * %concentrate + c3*%roughage d = d1 + d2 * %concentrate + d3*%roughage (c, c1, c2, c3, d, d1, d2, d3 are constant value which would	The quantity of concentrate used for have to be defined) cattle feeding is dealt with in the FADN enquiry b)

Figure 7.5 Modelling and availability of values for the parameters selected for energy consumption in stables (case: dairy cows)

a) This number is roughly equal to 1.3 times the number of dairy cows (when the dairy cows are bred in the farm) or to the number of dairy cows when this breeding is done outside the farm. However, this total number should be obtained directly from FADN; b) Amount of fed roughage might be also modelled by regional expert advice.

Figure 7.5 is only dedicated to the selected parameters and gives some first indications on how to model energy consumption according to a parameter and if the values of the parameters would be available at a European level.

The calibration of the model with FADN data may require a more precise model, differentiating cows (for which feeding depends on the milk yield) from the other animals (calves, heifers, fattening bull in mixed farms, etc.).

7.4.2 Meat production (beef, pork, poultry)

The selected parameters are similar to those chosen for the dairy cows. The milk yield is replaced by the meat production in the considered time period, which is equal to the difference between the total (live) weight of animals sold and the total weight of all animals which have been purchased in the considered time period (δ).

Selected parameters	Modelling	Parameters value availability
Meat production (δ)	E (feeding, cleaning, bedding; farm) = c + d * δ a)	FADN
Type of feed	c = c1 + c2 * %concentrate +c3*%roughage d = d1 + d2 * %concentrate + d3*%roughage	FADN

Figure 7.6 Modelling and availability of values for the parameters selected for energy consumption in stables (case: meat production)

a) The values of 'c' and 'd' for 'meat' cattle are different from those defined for dairy cows.

7.4.3 Livestock breeding

The selected parameters are the number of breeding cows (or sows) and the type of breeding (outdoor/indoor).

Selected Parameters	Modelling	Parameters value availability
Number of breeding cows (sows)	E (feeding, cleaning, bedding; farm) = c + d * δ	FADN
Type of breeding	c = c1 + c2 * outdoor (0/1) + c3*% indoor (0/1) d = d1 + d2 * outdoor (0/1) + d3*% indoor (0/1)	FADN

Figure 7.7 Modelling and value availability for the parameters selected for energy consumption in stables (case: breeding cattle)

7.5 Conclusion: needs for further research

It appeared that there are many models about energy consumption; however there is no single model linking all necessary parameters to the level of energy consumption. To answer the question 'What has to be done in the future?' some directions were outlined towards *satisfying models* in the close future:

- it was stressed that a prerequisite for these models is that they should fit regional or national values obtained from statistical approaches, e.g. the Farm Accountancy Data Networks (FADN). Two approaches were considered. The first one can be qualified as an 'engineering bottom up' approach, where the models are based on a deterministic approach. The second one is a statistical top down approach, where the model coefficients are directly calculated to fit the values of energy consumption for the farms presented in the statistics samples. This top down approach requires that the values of the model parameters are available for these farms. An intermediate way between both methods might be most practicable. Models could be first built at the European scale and then refined for each agricultural region;
- the main parameters that should be taken into account in a European database for Life Cycle Assessment are known and have been identified;
- the main tasks in future research is to harmonise the existing models, enabling the calculation of energy consumption according to the selected parameters, and to check that the values of these parameters are really available for all the European countries. These models should be then extensively checked at lower levels of aggregation, e.g. on site specific 'typical' farms.

C Data on the nitrogen cycle

8. Key factors necessary to determine the impact on the nitrogen cycle and the resulting environmental effects as part of LCA for agricultural products

J.-W. Erisman [1]

Abstract

Nitrogen pollution is one of the most important environmental issues in Europe at this moment. One of the major contributors to nitrogen pollution is the agricultural sector. The amount of nitrogen produced and used in intensive agricultural practice to ensure economical optimal production far exceeds the level where high losses to the environment are observed. Overloads of nitrogen can lead to a cascade of environmental problems reaching from a threat to human health and biodiversity to the influence on the climatic system. Nitrogen pollution occurs when an optimum level (or critical limit) is exceeded. Below this level, nitrogen has a positive effect on growth rates and vitality and no (air, soil and groundwater) pollution is expected. The nitrogen status of a region is therefore an important factor to determine if the nitrogen waste from a product can be considered as a useful or as a negative factor. It is recommended to use the combination of the N status of a region with the loss of N of a product as a way to compare the environmental risks from different agricultural products.

8.1 Introduction

In order to perform a life cycle analysis on agricultural products, it is necessary to determine the nitrogen cycle associated with the production, fixation, and waste. Nitrogen in its various chemical forms plays a major role in a great number of environmental issues. It contributes to acidification and eutrophication of soil, groundwater and surface waters, decreasing ecosystem vitality and biodiversity and effecting groundwater pollution through nitrate and aluminium leaching. Nitrogen compounds play an important role in the formation of ozone, oxidants, and aerosols, potentially posing a threat to human health and affecting visibility. One reactive nitrogen molecule can have a cascade of effects: for example, first it contributes to urban smog or direct effects on vegetation, then it contributes to acidification/eutrophication and/or pollution of surface water, groundwater and/or coastal water, and finally it contributes to the greenhouse effect through emission of nitrous oxide (N_2O).

The primary problem related to these issues is the production and associated accumulation of reactive nitrogen (Erisman et al., 1998; Erisman and Monteny, 1998). The natural nitrogen cycle can be disturbed by addition of nitrogen through three processes resulting from human activities: 1) transport of nitrogen from countries in transition having low nitrogen availability to countries that already have an excess of nitrogen for human food and animal feed; 2) fixation of atmospheric nitrogen

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through NH_3 and fertiliser production; and 3) fixation of atmospheric nitrogen by legumes and by oxidation during combustion processes. The first process relates to shifting reactive nitrogen from one place to the other, whereas the other two add new reactive nitrogen to the nitrogen cycle of the earth.

Three forms of nitrogen of special importance in the biosphere are: 1) oxidised nitrogen (NO_x), mainly emitted as an unnecessary waste product of combustion processes (e.g., traffic and industries); 2) reduced nitrogen (NH_3), mainly formed and emitted by agricultural practices; and 3) N_2O formed by nitrification and de-nitrification processes in the soil.

In this paper, the data necessary to determine the (changes in) key factors in the nitrogen cycle are discussed. First, the effects of changes in the nitrogen cycle are given, followed by a description of the key factors in the nitrogen cycle necessary to quantify the effects. The next section describes the current understanding and major gaps in knowledge for developing a modelling system that can be used to determine the effects of addition of N to the cycle. Then follows an alternative approach to compare the N pollution resulting from different products.

8.2 Effects of nitrogen

Nitrogen is an essential nutrient for all plants, humans, animals, and micro-organisms. Because of this, nitrogen emissions are not harmful to the environment until a certain level has been reached. For each system, there is an optimum nitrogen level related to the optimum production of the system. Ecosystems show an optimum curve. Figure 7.1 shows an example in the

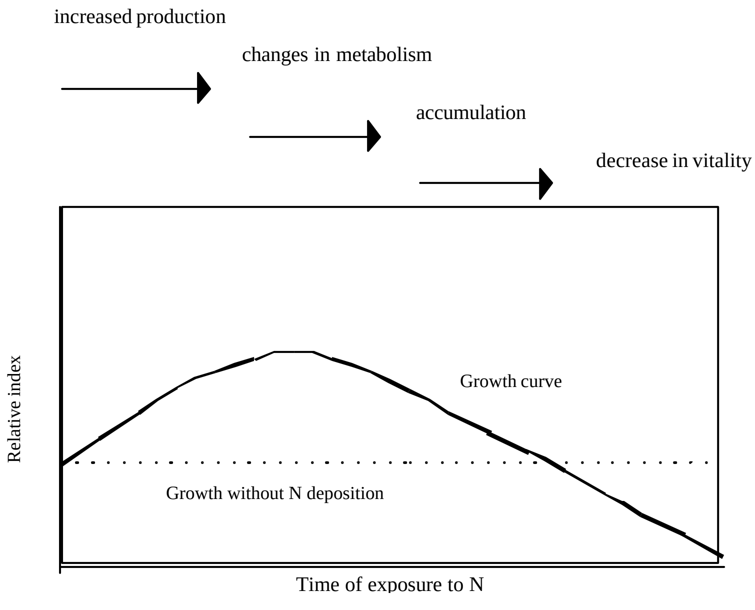

Figure 8.1 Hypothetical growth curve as a function of the time a certain N deposition level lasts
Source: Gundersen (1992).

form of a temporal form of the optimum nitrogen curve for forests, as suggested by Gundersen (1992). It indicates that production increases until a certain optimum level, and above that level production decreases. These optimum curves exist for all kinds of systems (see figure 7.2), also for agricultural crop production systems.

Increased amounts of all oxidised forms of nitrogen (NO, NO_2, N_2O_5, NO_3, HNO_2, HNO_3 and proxy acetyl nitrate, PAN) play a role in atmospheric pollution, deposition and soil and water pollution. Reduced forms of nitrogen, such as ammonia, ammonium, and amines also play an important role in atmospheric pollution, deposition, and soil and water pollution. N_2O is a greenhouse gas and contributes to global warming. The major sources, sinks, and transport mechanisms for exchange of nitrogen between the atmosphere and the biosphere of the earth are reasonably well understood. The nitrogen cycle of the whole earth is reasonably well quantified. On smaller scales, however, uncertainty increases rapidly. Cowling et al. (1998) compiled a long list of negative impacts of excess nitrogen. These include:

- respiratory disease in humans caused by exposure to high concentrations of:
  - ozone;
  - other photochemical oxidants;
  - fine particulate aerosol; and
  - (on rare occasions) direct toxicity of NO_2;
- nitrate contamination of drinking water inducing illness in infants;
- ozone damage to crops, forests, and natural ecosystems;
- acidification of soils, lakes, streams, and ground waters;
- eutrophication of freshwater lakes and ecosystems;
- blooms of toxic algae and decreases in swimability of water bodies;
- nitrogen saturation of forest soils;
- odour problems associated with animal agriculture;
- biodiversity impacts on ecosystems;
- global climate change induced by emissions of N_2O;
- ozone layer destruction by aircraft NO_x emission at high altitude;
- acidification effects on monuments and engineering materials;
- reduced visibility at scenic vistas and airports;
- arctic hazes.

One molecule of reactive nitrogen does not necessary result in negative effects. Only in situations where there is accumulation of nitrogen, effects can be expected. Furthermore, one single source, such as a farm, does not lead to effects, again only in high emission/pollution areas. In those areas the chance that one molecule can lead to a cascade of effects is highest. In low nitrogen areas the molecule is recycled through products (grass, crops, meat, milk, etc.) or fixed in the system. Effects only take place after a certain accumulation, when an optimum level is exceeded. The effect indices will be higher in areas exceeding the optimum, compared to areas where the optimum is not reached. The optimum levels are different for different systems or effects as illustrated in figure 8.2

(Cowling et al., 1998). The optimum can represent different measures, such as highest yields, maximum biodiversity, growth, etc.

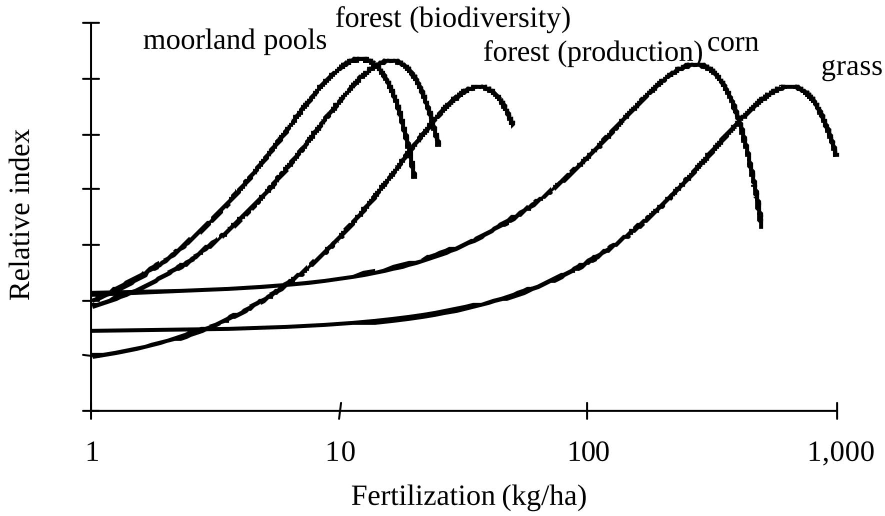

Figure 8.2 Optimum curves (fictive) for different systems in relation to N-fertilisation (fictive curves)

8.3 Key factors in the nitrogen cycle

An overview of the present state of knowledge and the main gaps in this knowledge have been presented in the report *Nitrogen pollution on the local and regional scale; The present state of knowledge and research needs* (Erisman et al., 1996) with respect to the processes involved in the cause effect relationship between nitrogen emissions and its effects. The report, which also list the state of knowledge on modelling of the key factors and the main uncertainties and gaps in knowledge, is summarised here. The key factors of the causal relationship of atmospheric nitrogen inputs and its effects can be summarised as follows:
effect parameters: excess nitrogen deposition and/or fixation and/or fertilisation leads to:
- ecosystem changes and reduction in biodiversity;
- reduction in vitality and changes in trees growth;
- groundwater pollution;
- N_2O emission;

key factors which determine the (risk of such) effects resulting from N excess:
- reactive N production;

and in ecosystems:
- N input to the soil;
- NH_x and NO_y deposition (or net input);
- NO_3^- and/or NH_4^+ leaching from the soil;
- land use;
- nutrient availability (N, base cations, P);

- soil acidity;
- plagues and diseases;

and in agricultural systems:

- fertilisation;
- NH_x and NO_y deposition (or net input);
- NO_3^- and/or NH_4^+ leaching from the soil;
- soil characteristics;
- water availability and groundwater level.

Key factors and effect parameters (or changes in these) can be *quantified* when information on the following topics is available:

- reactive N production and NH_3 and NO emission (diurnal/monthly);
- ambient concentrations of NH_3, NH_4+, NO, NO_2, HNO_2, HNO_3, NO_3^- (hourly);
- N fertilisation (events) and/or N fixation;
- (eco)system structure and species composition;
- meteorological parameters precipitation, temperature, wind speed, radiation (hourly);
- nutrient content in leaves and soil (seasonally);
- chemical and physical soil characteristics (annually);
- N transformation parameters;
- groundwater level and concentrations (monthly).

When these key factors can be assessed, the production and effects of N can be estimated. Several models exist to quantify parts of the processes involved. No models are available which describe all the key and effect parameters on different scales. Currently the STOP (Dutch Nitrogen Research Plan) is executed in the Netherlands. This programme ad-dresses the major uncertainties and most important gaps of knowledge, which were taken from Erisman et al. (1996). The following ratings were used to set priorities:

1 = very important to causal chain, should definitely be improved;
2 = important, but not within the scope;
3 = important, but it is questionable whether it should be taken into consideration (e.g. because research is too expensive in relation to the increase in knowledge or because knowledge is available from elsewhere);
4 = not important.

The ratings (bold print) were derived with the ecosystem approach in mind. The main gaps in knowledge and their ratings may be summarised as follows.

1. Emission and deposition

2 Natural emissions (soil, vegetation, wildlife, etc.).
1 Emission factors in different stages of the N cycle: NO and NH_3.
3 Influence of responses on local dispersion/deposition.
1 Dynamic processes in emission/deposition (compensation points, saturation, chemical and biological interactions).

3 Roughness transition zones.
1 Simple emission -dispersion- deposition model.

2. Effects on ecosystems (terrestrial, including forests, and aquatic)

1 Quantification of ecological effects and the associated role of N on different species (species response to N levels).
3 The role of the understorey and mycorrhizal fungi.
3 N cycle in forests (changes in nitrogen allocation and assimilation, interaction between above ground and soil uptake of nitrogen).
1 Recovery speed of N cycling and ecosystem status.
1 Sensitivity of ecosystem relevant species to the NH_4^+/NO_3^- ratio in soils and lakes.
1 Inter species competition.
3 Plant animal relations.

3. Effects on forest trees

1 Risk of storms, plagues, diseases, etc.
1 Growth rate.
1 Drought- N interaction.

4. Modelling: ecosystems (terrestrial, including forests, and aquatic)

1 Improvement of N balance (N_2, N_2O emission, mineralisation, denitrification, immobilisation, N uptake, litter production, sediment processes, etc.).
3 Inclusion of 'catastrophic events', i.e. for whole ecosystems.
1 Interaction of moisture content and N cycling.
1 Vegetation- soil linkage, i.e. effects of drought stress and nutrient stress on uptake.
1 Management of ecosystems.
1 Relationships between the nutrient status of the site and tree growth.

5. Agricultural systems

1 Improvement of N balance (N_2, N_2O emission, mineralisation, denitrification, immobilisation, N uptake, litter production, sediment processes, etc.).
1 Estimation of maximum N fertilisation.
1 Input output balances.
1 Groundwater pollution.

6. Further gaps in knowledge

4 Reference situations: what is natural development of ecosystems (succession) and what has been the influence of anthropogenic activities (including management, recreation, etc.)?
1 Risk assessment: how, what, where and when?
1 Groundwater pollution under nature areas (modelling).
2 Eutrophication of surface waters and seas.
4 Food chains.
2 Effects due to increased N concentrations resulting from drying of vegetation after, for example, fog exposure or dew accumulation.

Based on Erisman et al. (1996), in which the above research gaps were listed, the Dutch Ministries of Agriculture and of the Environment financed a (limited) Nitrogen research programme. The programme addresses the most important issues at this moment. The programme is split into two parts:

1. derivation of ecosystem specific critical loads for oxidised and reduced nitrogen;
2. improvement of the emission deposition relations.

The first topic is focus on deriving the doses effect relationships for reduced and oxidised nitrogen for different ecosystems. For this, first a literature review is made and secondly experimental work is conducted. The research is aimed at improving and validating the SMART-MOVE model (Latour and Reiling, 1991; Latour et al., 1994; Wiertz et al., 1992). This model estimates the soil quality, in terms of nitrogen and water availability, and acidification, given inputs of different components. Furthermore, it estimates the occurrence of different specie given the soil quality and ecosystem nature. This model can be used to derive critical deposition levels or different protection levels given certain inputs.

The second topic is aimed at improving and validating models describing the relation between nitrogen emissions (ammonia and nitrogen oxides), ambient concentrations, and deposition. An area with intensive livestock breeding and nature areas will be selected. In this area, the emissions will be estimated using statistical data. Furthermore, measurements will be made in- and outside the housing to determine the emission. This will be complemented with plume measurements. The ambient concentration will be determined at several locations using passive samplers. Finally, deposition will be measured using the gradient technique, simple deposition methods, and throughfall measurements. The measurements will take place in two years. The first year is the reference year, while during the second year measures will be implemented aiming at strong emission reductions. In this way, a wide range in concentrations and depositions will be obtained, for model validation. Furthermore, the effectiveness of the reduction measures can be evaluated.

The results are expected to be reported in September 1999. The model system resulting from both parts might form a good basis for LCA studies.

8.4 Using the N status as an indicator in LCA

The main problem of nitrogen in intensive agricultural production is the excess of nitrogen used for optimal production: the use of nitrogen is inefficient. For meat production, e.g. concentrate is used as animal feed with a high nutrient and heavy metal content to optimally stimulate animal growth and increase the meat quality. Only 30% of the input nitrogen is fixed in meat of pigs, the remaining 70% is wasted and removed via manure (Aarnink 1997). Manure, however, is a valuable product because it is used as fertiliser. For grassland, however, the utilisation of N by the grass from manure is only about 50% at optimal fertilisation rates of 400 kg/ha (CBS, 1998). The remaining N is partly retained in the soil, or emitted as NH_3, N_2O or N_2 to the atmosphere or as NO_3 to the surface water or groundwater. By optimising the N chain (input output balance), the losses of N to the environment can be minimised. The consequence of this, however, would be a decrease in production efficiency, which might have economical consequences. This situation only holds for intensive areas where the accumulation on N in the system has reached such levels that the increase in growth or production (N limiting) has changed into effects of overloads (N excess). In order to apply LCA, it is therefore very important to know the N status of the areas for which the product LCA is valid. This would mean that e.g. in the Netherlands (where there is an excess of reactive N), a different judgement will be made than e.g. in parts of Spain with shortage of N. It is necessary to determine criteria for establishing the N status of the area.

The N output for different products can be determined using a farm level mass balance system, such as that currently used in the Netherlands: MINAS (Nutrient Registration System) or proposed by Jarvis 1993 (see also Peel et al., 1997) for dairy farms in the UK. The outline of such systems is given in figure 8.3. The administrative data and/or model results provide information about the net output of nitrogen. In the Dutch MINAS system no distinction is made between the output as ammonia emission and/or nitrate leaching to the groundwater. However, by using data on the manure application system and the soil type and climate conditions, a first order approximation on this distinction can be made, as shown by Peel et al. (1997).

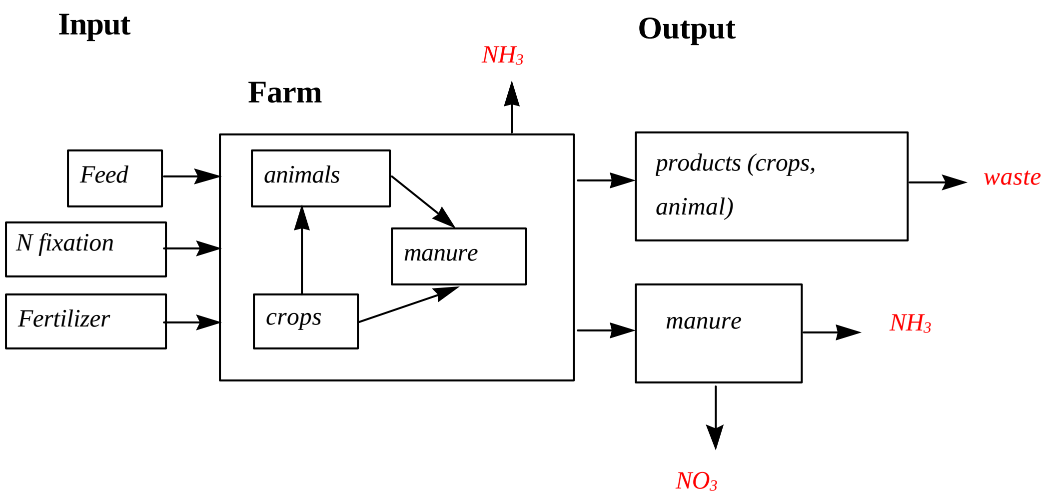

Figure 8.3 MINAS in the Netherlands

The difficulty of LCA for a certain product is that the contribution of nitrogen to the degree of pollution depends on the total N production in the area, which in turn determines the N status of the region. The N status in a region can be determined by the extent to which critical levels are exceeded, such as those for soil leaching of NO_3, deposition of N on nature areas, and the emission of N_2O. Especially the first two indicators are important, because if the critical limits for these indicators are not exceeded, the other indicators are also in a safe range. When the N leaching from the soil in most areas in the Netherlands is below 50 kg N/ha, and the N deposition is below 600 mol/ha/y, the N status is in an 'optimal' stage (Erisman et al., 1996). Above these thresholds, the region can be determined as an excess region and every 'extra' reactive N production may be considered as negative for the environment. Below these levels, the extra reactive nitrogen is not expected to cause environmental problems. A pollution factor for nitrogen per product can be determined by using the MINAS data, supplemented with a way to divide the total N loss between NH_3 emission and NO_3^- leaching, and together with an estimate of the N status of the region.

8.5 Conclusions

The increased reactive nitrogen production and use in some regions has led to an increased accumulation of N in the system resulting in different effects, such as eutrophication, acidification, groundwater pollution, climatic change, human health, etc. One molecule of reactive nitrogen can cause a cascade of effects. In agricultural food production, it is therefore necessary to determine the emission of different reactive N compounds to the environment. The difficulty of LCA for a certain product is that the contribution of nitrogen to the degree of pollution depends on the total N production in the area, which in turn determines the N status of the region. The N status is defined as the amount of excess nitrogen in the system (or region) causing effects at different levels in the cascade of effects. The N status is an important parameter to consider in LCA: Production of food in areas with a high N status has a higher risk to contribute to effects than in areas with low N status.

There are many uncertainties associated with the N cycle and with the effects resulting from accumulation of N in the system. The uncertainties are not associated with the basic knowledge of the different processes, but rather with the different chains connecting the different process descriptions to describe and quantify the whole N cycle.

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9. Nitrogen in arable farming

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Abstract

This paper summarises recent UK field and desk studies quantifying N fluxes in arable farming. Fertiliser-N is the largest N input (average c. 60%) to the majority of non-leguminous crops in England and Wales. Average fertiliser-N inputs to arable crops increased from 84 to 162 kg per hectare from 1969 to 1984, but subsequently decreased to 149 kg per hectare in 1994. However, total N inputs continued to increase to 239 kg per hectare in 1994 (from 155 kg per hectare in 1969) due mainly to increased application of animal manures and an increase in the area of grain legumes.

Crop offtakes at 55-95% are the greatest N outputs for major arable crops, the proportion varying according to the fate of unharvested residues. This offtake increased from 70 to 123 kg per hectare from 1969 to 1994. Over the same period estimated losses to the environment increased from 39 to 62 kg per hectare. Nitrate leaching was the greatest loss where animal manures were not applied. Application of manures may lead to large losses of ammonia and also greatly increased nitrate leaching if applied in late summer or autumn.

Fertiliser-N applications are based on the requirement for optimum economic return (N_{opt}). These N_{opt} values are adjusted to take account of N residues left by the previous crop, soil type and organic manure use. The N supply from crop residues has been found to be related to N applications to that crop. Differences in N_{opt} according to soil type are in consequence of overwinter leaching loss, fertiliser-N recovery and mineralisation, and soil moisture supply during the growing season. Little justification has been found for adjusting N_{opt} according to anticipated yield. Doing so is likely to over-fertilise large crops.

Current estimates of fertiliser-N application are accurate to 1 kg per hectare (± < 1%) for major crops. Estimates of manure-N applications are no better than ± 50%. Crop N offtake is known to ± 10% and ammonia loss ± 30%.

9.1 Introduction

This paper summarizes recent UK work quantifying N fluxes within arable farming. Both field and desk studies are used to identify the major N inputs and outputs, and factors affecting them. In section 1 we identify the major N inputs and N outputs in arable production, and suggest how they may differ between farm types, with particular reference to the type of crops grown and output levels in

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England and Wales (Eand W). In section 2, we discuss how factors such as soil type and climate affect N inputs, N outputs and N pollution. Sections 3-7 deal with uncertainties, data collection, aggregation, and updating, and approximations for missing data.

9.1.1 Nitrogen Inputs

Fertiliser nitrogen

Fertiliser-N is by far the largest N input to the majority of non-leguminous arable crops in E and W. In a current field study fertiliser-N was c. 80-95% of total N inputs for cereals, sugarbeets, and potatoes in arable systems without animal manure application (Webb et al., 1998b). Similar observations have been made elsewhere (figure 9.1).

Author	Data	Remarks
Webb et al., 1998b	Fertilizer-N 80-95% of N inputs	Arable systems without livestock
Mitchell and Shepherd pers comm	Fertilizer-N 87% of N inputs	Arable systems without livestock
Van Faassen and Lebbink 1990.	Fertilizer-N 91-95% of N inputs	
Loveland et al., 1998	Fertilizer-N 62% Manure-N 18% Atmos. dep. 8% N fixation 7%	National data

Figure 9.1 Relativen nitrogen inputs by source

A recent desk study quantified nutrient balances for arable farming in E and W from 1969-1994 (Loveland 1998). These data include estimates of all N inputs, animal manures, N deposition and fixation, and current minor sources such as sewage sludge, industrial by products, and irrigation water. From 1969 to 1994 average fertiliser-N applications to arable crops increased from c. 84 to c. 162 kg per hectare, but subsequently decreased to c. 149 kg per hectare by 1994. However, total N inputs increased from c. 155 kg per hectare in 1969 to c. 239 kg per hectare in 1994. This was due to greater manure-N application (+ c. 8 kg per hectare), N deposition (+ c. 3 kg per hectare) and N fixation by legumes (+ c. 8 kg per hectare) following an increase in the agricultural area under peas and beans. Over a period in which total N inputs have increased by c. 54%, the proportion as fertiliser-N has increased from c. 54 to c. 62% of the total, with a peak in 1984 when it was 68%. The other major sources (1994) were manure-N (c. 18%), N deposition (c. 8%) and N fixation by legumes (c. 7%).

Animal manures

Animal manures may be applied to arable crops, depending upon their availability, cropping, and soil type. Rates of application are extremely variable, and are summarised in table 9.1.

Table 9.1 *Typical manure application rates in UK arable farming*

Manure type	Application rate	
	manure t per ha	total ammoniacal N (TAN) kg per ha
Straw-based cattle and pig manure	10 – 40	15 – 100
Poultry manure	10 – 25	150 – 250

Nitrogen fixation

The greatest N fixation in E and W is from grain legumes (pulses), and has been estimated at c. 265 and c. 285 kg N per hectare for peas and beans respectively (Sylvester-Bradley, 1993). Fixation by free living micro-organisms has been estimated at no more than 5 kg per hectare under UK conditions (Witty et al., 1977).

Nitrogen deposition from the atmosphere

Published data is summarised in figure 9.2.

Author	Data	Comments
UKRGIAN 1994	Wet N deposition 5-13 kg per ha	Lowland E and W, where most arable farming takes place
Anonymous 1996, 1997, 1998	Wet N deposition 5-9 kg per ha	Nitrate Sensitive Areas (NSAs) in Central and Eastern England
Webb et al., 1998b	Wet N deposition 20 kg per ha	Western England, in an area with a large livestock population.
Webb et al., 1998b	Wet N deposition 5 -9 kg per ha	Central and Eastern England
Goulding et al., 1998	Dry N deposition 24 - 40 kg per ha	South Eastern England. This estimate may be greater than in arable areas to the North and West.

Figure 9.2 Deposition of nitrogen from the atmosphere

Seed

The N content of seed is usually included in estimates of N balance, e.g. Van Faassen and Lebbink (1990). However, such inputs represent, at most, only c. 4% of the total.

9.1.2 Nitrogen outputs

Crop nitrogen offtake

The greatest outputs of N are usually as crop N offtake. See figure 9.3.

Author	Data	Remarks
Webb et al., 1998b	55-95%	Greatest proportion from cereals when straw was removed, and sugarbeets.
Mitchell and Shepherd (pers. comm.)	47-96%	
Rahn et al., 1992	Up to 240 kg per ha N remain as crop residues at harvest	Field vegetable crops

Figure 9.3 Crop N offtake as a % of total N offtake

Nitrogen offtake in crops increased from c. 79 to c. 123 kg per hectare from 1969 to 1994. Offtake in straw increased from c. 11 kg per hectare in 1969, to c. 21 kg per hectare in 1984. This offtake included N lost as NO_x during straw burning, and since the ban on burning in the UK, N removal in straw has reduced to c. 11 kg per hectare. Overall crop N offtake is c. 67% of total outputs. Losses to the environment were estimated to have increased from c. 39 to c. 62 kg N per hectare, broadly reflecting increased N inputs. However, some of these estimates of N emissions are subject to considerable uncertainty.

Nitrate leaching

Nitrate leaching was the second largest loss measured by Webb et al. (1998b). The use of animal manures, which are often applied several weeks before planting arable crops, will increase the risk of NO_3- leaching, particularly if liquid slurries or poultry manure are applied before late November (Smith and Chambers, 1997).

Ammonia volatilization

Losses of ammonia (NH_3) may occur following application of N fertilisers, and from growing crops, especially during senescence. In practice, it is difficult to distinguish between these sources, and in effect, many field experiments measure both, simultaneously. Sutton et al., (1995) suggested that on an annual basis, plant emissions of NH_3 be approximately in balance with dry NH_3 deposition. Webb et al. (1998b) measured no significant net fluxes, except from a potato crop when c. 13 kg per hectare N were emitted.

Manure applications may greatly increase NH_3 losses (Smith et al., 1994). Potential NH_3 emissions are extremely variable, but from typical surface applications may be c. 10-30 kg per hectare N.

Nitrous oxide and dinitrogen emissions

Several ^{15}N studies have attributed greater losses of fertiliser-N to denitrification than to leaching over the period between fertiliser-N application and harvest of the crop to which the fertiliser was applied (e.g. Bhogal et al., 1997; Addiscott and Powlson, 1992). Since fertiliser-N was applied to growing crops in spring, after the end of the field capacity (FC) period, leaching losses were unlikely to occur. However, no measurements of gaseous losses were made in those studies. Recous et al. (1988) measured losses from labelled fertiliser-N and found only small (c. 1 kg per hectare N) emissions by denitrification. However, they considered that those small measured losses may have been a consequence of the short period over which measurements were made. Moreover, N_2O may be emitted as a consequence of nitrification, as well as of denitrification (Klemendtsson et al., 1988), and so measuring only one pathway is likely to lead to an underestimation of N_2O loss. However, Webb et al. (1998b) measured total N_2O (and N_2) emissions and found them at c. 5 kg per hectare to be much smaller than the c. 30 kg per hectare N estimated by Addiscott and Powlson (1992) lost by denitrification from equivalent amounts of fertiliser-N. Those measurements were not, however, continuous, and estimates of total annual N losses are still being refined. Questions of scale should be asked when comparing ^{15}N and other measurements of N_2O/N_2. Estimates from small discrete areas may inadvertently hit hot spots; whereas more spatially distributed data may smooth these out.

Nitric oxide

Both denitrification and nitrification may also give rise to emissions of nitric oxide (NO), although in agricultural soils, where pH is usually maintained above 5.0, nitrification is considered to be the dominant pathway (Remde and Conrad, 1991). Few measurements have been made of NO emissions from agricultural soils, and losses are currently considered to be small (Skiba et al., 1997), although Jambert et al. (1997) measured much larger losses of NO (c. 40% of total gaseous N losses) than of N_2O (c. 14%) or NH_3 (<1%).

9.2 Factors affecting N flows

9.2.1 N inputs

Fertilizer nitrogen

Current UK fertilizer-N recommendations are for an economic optimum (N_{opt}) (Anonymous 1994). Fertiliser-N is recommended such that any further application will cost more than the value of any extra crop produced. At c. 200 kg per hectare, Nopt have been found to be broadly similar for many crops, although substantially less for crops such as sugarbeets, onions and linseed, and >300 kg per hectare for some leafy brassicae. These N_{opt} are adjusted according to previous crop, soil type, organic manure use and, in the cases of wheat and oilseed rape, expected yield. In recent years, several studies, comprising large numbers of replicated field experiments have systematically examined, on a range of soil types in E and W, the effects of previous crop, fertiliser-N applied to the previous crop, organic manure use, sowing date, cultivar, type of fertiliser-N and soil mineral N (SMN) on N_{opt} for cereals. This work was reviewed by Webb et al. (1997) and the main findings are summarised below:

- the amounts of N recovered by unfertilised cereal crops from previous crop residues (apart from legumes) was found to be well-related to the amount of fertiliser-N applied to the previous crop, rather than an inherent characteristic of that crop.
- the effects of such residues varied greatly with soil type. No residual effects were found following potatoes on a light sandy soil where NO_3 leaching losses were large.
- soil type had a bigger effect on N_{opt} than is currently allowed in recommendations (Anonymous 1994), and was not simply a consequence of differences in NO_3 leaching losses over winter. Mean cereal N_{opt} ranged from c. 145 kg per hectare on sandy soils to c. 240 kg per hectare on shallow soils developed over chalk, despite both soils being prone to large leaching losses, and optimum yield (Y_{opt}) being similar on both. These large differences in N_{opt} between soil types were due to: a) greater apparent fertiliser-N recovery (AFR) and greater apparent mineralisation (AM) during the growing season on sands, reducing crop demand for fertiliser-N, and: b) to utilisation of recovered N being restricted by drought on sandy soils, further reducing N_{opt}.
- no justification was found for adjusting N_{opt} according to anticipated cereal yield. This was because demand for N was related to the increase in yield from fertiliser-N (Δ_y) rather than Y_{opt}. On a clay or silt soil, yield may increase from c. 6 to c. 10 t per hectare (Δ_y = 4 t per hectare), while on a shallow chalk the increase may be from c. 2 to 7 t per hectare (Δ_y = 5t per hectare). Thus despite the much smaller Y_{opt} on the chalk, measured N_{opt} reflects the larger Δ_y. Differences in AFR between soil types also helped account for the lack of correlation between Y_{opt} and N_{opt}. This finding has implications for fertiliser-N recommendations based on the balance sheet approach. By relating N_{opt} to yield, crops grown on productive soils on which fertiliser-N is recovered efficiently and where the ratio of assimilate produced per kilogram recovered N is large, may be consistently over fertilised.

The large residual effects of fertiliser-N applied to previous crops on nitrate-retentive soils appear to be at variance with the results of ^{15}N studies, such as that of Bhogal et al. (1997), who found recovery of ^{15}N by the succeeding unfertilised crop increased negligibly with increasing fertiliser-N. Sylvester-Bradley (1996) noted increases of c. kg/ha by cereals following w wheat on a retentive soil. It remains possible that the lack of residual effect of fertiliser N suggested by ^{15}N studies may be an artefact induced by pool substitution

Animal manures

Although the majority of animal manures are applied to grassland in E and W (c. 44% of grass area), some are applied to arable crops (c.16% of arable area), especially from pigs and poultry, as these livestock are more likely to be found in arable areas. Fertiliser-N applications should be reduced if organic manures have been applied (Anonymous 1994). However, in E andW insufficient allowance is made for the total crop available N (TAN) in manures. Smith and Chambers (1995) found fertiliser-N was reduced by only c. 22 and 4 kg per hectare respectively for crops of winter wheat and potatoes that had been given animal manures. As stated above, it is not possible to accurately determine manure N applications to crops in E and W. Where straw based manures (FYM) and slurry, are applied at rates of c. 25 t per hectare and 50 m^3 per hectare to arable soils in winter, crop 'available' N is likely to be c. 45 and c.75 kg per hectare respectively (Anonymous 1994). Some of the TAN will have been lost by NH_3 volatilisation, or by leaching if applied before the winter period (Smith et al., 1994), the small allowance made in fertiliser-N for potato crops in particular, suggests fertiliser-N applications could be further reduced when animal manures have been applied. These allowances should increase as measures are adopted to reduce nitrate leaching and NH_3 loss, leading to greater conservation of TAN in soils, and hence greater availability for crops.

Nitrogen deposition

Results presented earlier suggest that, in some parts of E and W both wet and dry deposition of N may be greater than currently estimated, and that the variability between arable areas may also be greater than previously thought. Where N deposition is as large as reported by Goulding et al. (1998), then this source of N becomes more significant than is commonly recognised.

9.2.2 Nitrogen outputs

Crop nitrogen offtake

A major factor in accounting for differences in N output is the fate of unharvested crop residues. When removed from the field, e.g. cereal straw for animal bedding, losses may be increased by c. 30% (e.g. Webb et al., 1998b) and N balances are more likely to be negative (e.g. Van Faassen and Lebbink 1990). No adjustment to fertiliser-N recommendations is currently made to allow for this greater offtake. Given the overall national gross surplus of N applied to arable crops (c. 50 kg per hectare), increasing fertiliser-N to compensate for losses in straw appears unwise, and likely to lead to greater N losses to the environment. Nicholson et al. (1997) showed that even after c. 10 years, straw incorporation had no effects on N_{opt}. Moreover, straw is more likely to be removed in areas, or on farms, with livestock, and so animal manures are likely to be applied to at least some crops in the rotation and hence the straw-N will be returned to the soil. Since reductions in fertiliser-N are generally insufficient to take account of manure TAN, it seems unlikely that, in practice, straw removal will lead to major losses of N from the soil.

Nitrate leaching

UK research on nitrate leaching does not indicate that nitrate is lost by leaching in direct proportion to the amount of N applied in the previous cropping season. Among the factors that cause the relationship to be non linear for fertiliser-N are summarised in figure 9.4.

Author	Factor	Remarks
Lord 1992	Cereals. Up to N_{opt} losses (kg N per kg N applied) are c. 0.07. Above N_{opt} losses are c.0.50.	Losses of nitrate under arable cropping may nevertheless be substantial when, because of large residual fertility, no fertiliser-N is applied.
Lord 1992	Some crops (e.g. potatoes are less efficient at utilising fertiliser-N than cereals.	
Addiscot and Whitmore. 1991	The proportion of fertiliser-N lost by leaching differs according to soil water capacity (SWC), increasing as SWC decreases.	
Anthony et al., 1996	Overwinter rainfall (OWR).	The proportion of N lost will tend to increase, although not linearly, with increasing OWR. Effect predicted by the SLIMMER model.
Smith and Chambers 1997	Potential for leaching loss increases as the ratio of TAN to total N increases.	Losses are proportionately greater from slurries and poultry manure than from Straw based manures.
Froment et al., 1992	Greater leaching losses occur from manures applied in late summer and autumn than from those applied in Winter and Spring	Especially to slurries and poultry
Chambers et al., 1998	MANNER model	Predicts losses of nitrate and NH_3 following manure applications.

Figure 9.4 Factors affecting nitrate leaching

Ammonia losses

Emissions of NH_3 from mineral fertilisers depend on the type of N-fertiliser applied, soil type (especially soil pH), meteorological conditions and time of application in relation to crop canopy development. In particular, the type of N-fertiliser applied has a great effect on emissions (Whitehead and Raistrick 1990). Emissions are largest from urea fertiliser because it hydrolyses rapidly in the soil to release NH_3. Emissions from ammonium sulphate (AS) may also be large, but these are very dependent on soil pH, with larger emissions from calcareous soils. Other fertilisers, such as ammonium nitrate (AN), are more neutral in pH and produce much smaller emissions. These are often difficult to distinguish in measurements of plant atmosphere fluxes (e.g. Sommer and Jensen 1994). Fertilisers containing only NO_3^- will not emit NH_3 directly, but may increase NH_3 emissions by fertilised crops.

Following spreading of manures and slurries to land, the amount of NH_3 lost depends upon a number of factors, of which the most important are TAN and dry matter content of manure/slurry. Temperature, windspeed, soil cation exchange capacity, and infiltration rate also influence the amount of NH_3 lost. Emissions decrease with decreasing slurry dry matter (Brunke et al., 1988) since dilute slurries infiltrate the soil more readily. While a greater proportion of NH_3 is lost following incorporation of solid manures (c. 65% for pig manure, Chambers et al., 1998) compared with slurries (c. 25% for pig slurry, Pain et al., 1997), the proportion of N as TAN is much less in solid manure (c.25%) than in slurry (c. 50-60%) (Anonymous 1994) so losses will tend to be smaller from field applications.

Nitrous oxide/dinitrogen

In soil N_2O is produced predominantly by two microbial processes: nitrification (the oxidation of ammonium (NH_4^+) to NO_3^- and denitrification (the reduction of NO_3^- to gaseous forms of N, ultimately N_2O and N_2). The rate of N_2O production is primarily dependent on the availability of mineral N in the soil (e.g. Bouwman, 1996). Maximum N_2O emissions are generally observed within 2 to 3 weeks of N-fertiliser application. The magnitude of the emissions depends on the rate and form of fertiliser applied, the crop type, the soil temperature, and soil moisture content. However, it is not possible to derive emission factors for different fertilisers or soil types from existing data (Bouwman, 1996). Therefore, the IPCC method defines only one emission factor for all types of N input.

Following the IPCC Methodology (IPCC/OECD 1997), N_2O emissions from agricultural soils may be calculated as the sum of :

i direct soil emissions (1.25% of N inputs are emitted as N_2O-N); (where N inputs are from fertilisers, manure applications, biological N fixation and crop residues). See IPCC Worksheet 4-5, sheet 1;

ii direct N_2O emissions from cultivation of organic soils (histosols) IPCC Worksheet 4-5, sheet 2);

iii direct soil emissions (2% of N inputs) from grazing animals (IPCC Worksheet 4-5, sheet 3);

iv indirect emissions following deposition of NH_3 and NO_x (1% of N deposited as NH_3 and NOx is subsequently re-emitted as N_2O), or leaching or run off (2.5% of N leached or run off, IPCC Worksheet 4-5, sheets 4 and 5).

Prior to estimation of direct N_2O emissions, fertiliser-N inputs are reduced by 10%, and excretal- and manure-N returns by 20%, to allow for N lost as NH_3.

Direct emissions include emissions which are induced by N input (fertiliser, manure, excretal-N deposited during grazing, biological N fixation and crop residues). In addition, cultivation of organic soils (histosols) is regarded as a direct source of N_2O. The magnitude of direct N_2O emissions varies with a range of soil and environmental factors. Application of N-fertiliser to, or incorporation of N rich crop residues into, moistureretentive soils produces greater N_2O emissions than application to freedraining soils (Skiba et al., 1992). Application to or incorporation into warm soils is likely to lead to greater emissions than from soils which are cold. However, some recent studies have shown, that the largest N_2O emissions occur during thawing of frozen soils (MÜller et al., 1997), and the total emissions between November and February were 50% of the total annual flux (Kaiser et al., 1997). Rapid crop growth, and demand for NO_3^-N, will reduce N_2O emissions by reducing the pool of mineral N available for denitrification (Yamulki et al., 1995). Increased exudation of C from plants may also increase denitrification.

These soil and environmental factors also influence the magnitude of indirect N_2O emissions following atmospheric deposition of NH_3 and NO_x.

Nitric oxide

Nitric oxide (NO) may be emitted either as a consequence of nitrification, or denitrification. In agricultural land, where pH is likely to be maintained above 5.0, nitrification is considered the dominant pathway of NO emission (Remde and Conrad, 1991; Skiba et al., 1997). The main determinant of NO production in agricultural soils is mineral N concentration (Skiba et al., 1997). This is increased by N-fertiliser application, manure application, excretal N deposited during grazing, crop residue incorporation, and cultivation.

Current data on NO emissions in relation to fertiliser-N use were reviewed by Skiba et al. (1997). Losses ranged from 0.003 to 11% of applied fertiliser-N, with a geometric mean emission of 0.3% applied N. In view of the sparse and skewed nature of the data, this estimate is proposed in preference to that of Yienger and Levy (1995) who used an arithmetic mean of 2.5% loss of fertiliser-N to estimate NO emissions.

Activities such as tillage and incorporation were considered to increase NO emissions by a factor of 4 (Skiba et al., 1997). Thus, knowledge of the N concentration and mineralisation rate of crop residues could provide an estimate of soil NH_4^+ on which to base an emission estimate. Knowledge of soil N content could also allow an estimate to be made of NO emissions following cultivation.

9.3 Uncertainties

9.3.1 N inputs

Fertilizer nitrogen

Because of differences in the sample sizes of different crops (Burnhill et al., 1996), the accuracy with which fertiliser-N applications are estimated by the UK Survey of Fertiliser Practice differs between crops. Standard errors of the estimates range from c. 1 kg per hectare N for winter barley (0.5% of the overall mean), to 12 kg per hectare N for vegetable brassicae. This precision means that for the major crops, changes in fertiliser-N applications between years of only c. 3-5 kg per hectare may be significant.

Animal manures

Current estimates of total animal manure applications are reasonably well known, to ± c. 20%. However, estimates of application to different crops are uncertain by at least ± 100%.

N deposition

Current estimates of N deposition are much less certain, and were estimated at c.20% and c, 50% for wet deposition and dry deposition respectively by UKRGIAN (1994).

9.3.2 N outputs

Crop nitrogen offtake

Annual estimates of crop yields are published for the UK, and these are ± 5%. Using standard values for crop N concentration will give estimates of crop N offtake accurate to ± 10%.

Nitrate leaching

In the E and W the 'NEAP-N' model (Anthony pers. comm.) is being used to estimate annual NO_3-N losses. This model uses a single N loss factor for individual crop and livestock types with land use data available from the MAFF agricultural census. The model is applied at 1 km^2 resolution interpolated from agricultural land use reported at parish level, with estimated NO_3-N leaching losses based on spatially distributed information on soil type, hydrologically effective rainfall and land use. The NEAP-N baseline values for NO_3-N leached under UK arable crops are derived from Lord (1992). Values have been revised based on the results of the most recent research (e.g. Lord et al., 1995; Webb et al., 1998a). Losses modelled using this approach have been found to give good agreement (c. ± 10%) with measurements made in UK Nitrate Sensitive Areas using porous pots.

Ammonia losses

Emissions of NH_3 are based upon estimates of fertiliser- and manure-N applications and of estimates of emissions from those sources. As noted in 3.1. above, fertiliser-N applications are well characterised. However, the amount and N concentration of animal manures to arable land is uncertain, and current estimates may be no better than c. 20%.

Percentage NH_3 losses are estimated to be c. 30% from excreta/manure and c. 50% from fertiliser-N. Given the influence of weather on NH_3 emissions there are likely to be significant differences between years.

Nitrous oxide

As estimates of N_2O emissions require data on fertiliser- and manure-N applications, these will be subject to the same uncertainties as are NH_3 emissions. Current emission factors for N_2O emissions are considered to be uncertain by a factor of 9 (range 0.25-2.25% of N inputs, Bouwman (1996).

Nitric oxide

Much less information is available on factors determining losses of NO from soils. While application of fertiliser-N may be estimated with an accuracy of >10%, other factors such as returns of N in crop residues and soil N contents may be estimated to within ± 25%. However, the greatest uncertainty is over emission factors. Using data from essentially the same body of published work, Yienger and Levy (1995) and Skiba et al. (1997) arrived at mean emission factors almost an order of magnitude different, suggesting an uncertainty factor of 10.

9.4 Aggregation/Calibration

9.4.1 Nitrogen inputs

Fertiliser-N

Application rates reported separately for E and W and for Scotland (Burnhill et al., 1996).

Manure

Areas of crops to which manures are applied, but not amounts (Burnhill et al., 1996).

N deposition

Reported on 20 x 20 km grid square (UKRGIAN 1994).

Crop N offtake

Yields of major crops given separately for each county in E and W. No equivalent data on N concentrations in crops or their residues (Anonymous 1998).

9.4.2 Nitrogen outputs

Nitrate leaching

To validate the model against actual field measurements, NEAP-N runs were undertaken on selected NSAs for which measurements of nitrate leaching losses have been made using porous ceramic cups.

Ammonia losses

The simplest approach to spatially desegregate NH_3 emissions is to scale these by the distribution of total arable land. In a more detailed approach census data on the distribution of different crop types may be combined with characteristic fertiliser inputs to each crop type, together with the overall fertiliser emissions factor.

Nitrous oxide and nitric oxide

Direct emissions may be spatially desegregated using census data on the distribution of different crops together with mean fertiliser-N inputs to those crops. Data on the distribution of histosols may also be included to improve spatial desegregation. Indirect emissions may also be spatially desegregated if spatial data is available for N deposition, and N leaching and runoff.

9.5 Data collection

9.5.1 Nitrogen inputs

Fertiliser nitrogen

Fertiliser-N application to all major crops and grass, and for aggregates of minor crops e.g. soft fruit and vegetable brassicae, are published annually for the UK in the British Survey of Fertiliser Practice (e.g. Burnhill et al., 1996). The data is desegregated for England and Wales and for Scotland, but not further. Desegregated information is available on request, but such data, being of smaller sample size, are therefore less accurate. The distribution of fertiliser-N application rates, as well as means for each crop (or crop group), are given. In addition the percentage of each crop to which animal manures are applied is given.

Animal manures

No information is given on the type or amount of animal manure applied, but this can be supplied on request.

N deposition

Data on wet deposition for the UK are available from the UK Precipitation Composition Network, run by NETCEN at the Culham Laboratory, Oxford (AEA, 1995). Measurements of dry deposition are less comprehensive, but are made at sites in the UK as part of the Environmental Change Network and are published by NETCEN (AEA, 1995).

9.5.2 N outputs

Crop N offtake

Statistics on average crop yields are published annually by the UK Ministry of Agriculture Fisheries and Food (e.g. Anonymous 1998). However, these do not include data on N concentrations which are likely to differ between years.

Nitrate leaching

Estimates of N loss factors for the NEAP-N model are likely to be updated as new information is published.

Ammonia losses

These are estimated nationally as part of the UK Ammonia Emissions Inventory (Pain et al., 1997). The Inventory is updated annually. However, the information is only desegregated between England and Wales and Scotland.

Nitrous oxide

A national Inventory of UK N_2O emissions is currently being compiled (Jarvis pers comm). Estimates may also be made using the default IPPC Methodology (IPPC/OECD, 1997) for which data are available.

Nitric oxide

These is no estimate of NO emissions from UK agriculture, although a simple method has been proposed (Webb et al., 1998), based on the review of NO emissions by Skiba et al. (1997).

9.6 Updating

9.6.1 Nitrogen inputs

The UK MAFF is likely to continue to make available annual statistics on crop areas and crop yields. Data on N deposition, both wet and dry is available on the NETCEN website (http://www.aeat.co.uk/netcen/aqarchive/archome.html).

9.6.2 Nitrogen outputs

Projects are currently funded to update estimates of N losses as nitrate, ammonia and nitrous oxide. The UK Ammonia Emissions Inventory is updated annually. This is also likely to be the case with the N_2O emissions Inventory. A Project has just begun to make preliminary estimates of NO emissions from UK agriculture.

9.7 Approximations for missing data

9.7.1 Nitrogen inputs

Fertiliser-nitrogen

Fertiliser-N inputs may be approximated by analogy with data from countries of similar climate and agriculture where fertiliser use is recorded.

Manurenitrogen

This may also be approximated by analogy with data from countries of similar climate and agriculture. However, given the uncertainties of estimating N inputs from manures, this will give rise to significant errors.

N deposition

This data is published for Europe, (e.g. Van Pul et al., 1995). However, the scale is large, and will only give a crude approximation.

9.7.2 Nitrogen outputs

Crop nitrogen offtake

Yields may be approximated by reference to countries of similar climate and agriculture. Standard N concentrations may be applied to derive N offtake.

Environmental losses

Where N inputs are known, losses may be approximated using standard models or spreadsheets. For gaseous losses the methodology is available in the EMEP/CORINAIR Atmospheric Emissions Inventory Guidebook (EEA, 1996). The methodology used to estimate NH_3 emissions is based on the MARACCAS model (Cowell and ApSimon, 1998).

9.8 Uncertainties

Fertilizer-N	0.5-5.0%	(Burnhill et al., 1996)
Manure applications	c. 100%	
Wet N deposition	c. ± 20%	(UKRGIAN 1994)
Dry N deposition	c. ± 50%	(UKRGIAN 1994)
Nitrate Leaching	± 10%	
Ammonia emissions	± 30%	
Nitrous oxide	x9	(Bouwman 1996)
Nitric oxide	x10	

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10. Methods to estimate potential N emissions related to crop production

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Abstract

Nitrogen emissions mainly arise from the application of N containing organic and mineral fertilisers. The most important of these N emissions are ammonia (NH_3), nitrous oxide (N_2O) and nitrate (NO_3). These emissions are strongly influenced by soil type, climatic conditions, and agricultural practice and therefore they can vary considerably. Since actual measurements of emissions are neither practical nor appropriate for LCA purposes, structured methods are required to derive estimates of average emission rates. An alternative could be the use of values derived from the literature which would, however, require considerable effort compared to structured methods, especially because the values might be only valid for the particular system under investigation.

Methods to determine estimates for NH_3, N_2O and NO_3 emissions were selected from a literature review. Different procedures were chosen for the determination of estimates of NH_3 emissions from organic (Horlacher and Marschner, 1990) and mineral fertilisers (ECETOC 1994). To estimate the N_2O emissions a function derived by Bouwman (1995) was selected. A method developed by the German Soil Science Association (DBG, 1992) was proposed to determine approximate NO_3 emissions. An example is given to illustrate the different procedures.

10.1 Introduction

Three relevant nitrogen emissions are released into the environment due to agricultural production: ammonia (NH_3) and nitrous oxide (N_2O) as gas emissions and nitrate (NO_3) leached into the groundwater. Other N emission pathways such as surface water runoff or soil erosion are comparably of less importance (ECETOC, 1988). Figure 10.1 shows a simplified nitrogen cycle focusing on the most important nitrogen in- and outputs.

Agriculture, including both crop and animal production contributes considerably to the NH_3, NO_3 and N_2O emissions. Especially for ammonia, agriculture is by far the main source of emissions. Table 10.1 gives information about the share of agricultural production on the different nitrogen emissions at different spatial scales.

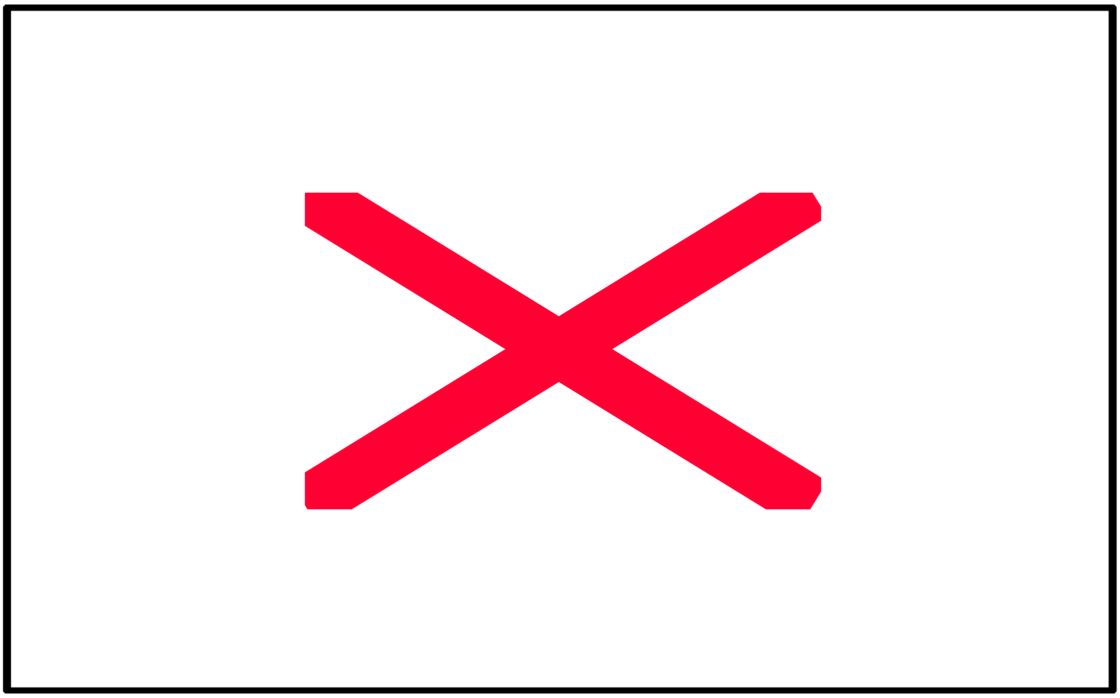

Figure 10.1 The nitrogen cycle on a farm
Source: ECETOC, 1988 (modified).

Table 10.1 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 and Kielland, 1997; e) Enquete-Kommission 'Schutz der Erdatmosphäre', 1994; f) Kroeze, 1994.

Another important background information in this context is the contribution of the N emissions to environmental effects (table 10.2). The share of each emission to a potential environmental effect on a global scale was calculated using LCA normalisation data of Guinée (1993). Ammonia contributes to about 20% to the total global acidification potential. Furthermore, NH_3 is responsible for approximately 14% of the eutrophication potential. Nitrate contributes to about 65% to the eutrophication potential, while its contribution to the total global human toxicity potential is very low. Nitrous oxide belongs to the greenhouse gases and contributes to about 5% to the total global warming potential.

Table 10.2 Share of N emissions in global environmental effect

	NO_3	NH_3	N_2O
Acidification		20%	
Eutrophication	65%	14%	
Global warming			5%
Human toxicity	0.0001%		

Source: Guinée, 1993.

In the following, the nitrogen emissions are further investigated concerning the main parameters influencing the emission rates. Easy to perform estimation methods will be presented in order to provide tools to calculate potential emissions relevant in a Life Cycle Assessment (LCA) of agricultural products of processes.

An example will follow every step to illustrate the proposed procedures. An LCA case study of a winter wheat production system was chosen as an example (Küsters and Jenssen 1998). Goal of the study was to evaluate the environmental impact associated to the production of one tonne of winter wheat grain. The system is located on a farm in northern Germany and the yield is 8.5 tonnes per hectare. The parameter values used in the calculations are mentioned in the context of these calculations. *All information related to the test case will be written in italic letters.*

10.2 Ammonia volatilisation

Nearly 90% of the global emissions of the volatile gas ammonia (NH_3) are related to agriculture (see table 10.1). Within agriculture, animal husbandry has by far the greatest share on the ammonia released to the environment (Isermann, 1990; ECETOC, 1994). Ammonia volatilisation occurs during and after production, storage and application of organic (see chapter 10.2.1) and to a lower extent of mineral fertilisers (see chapter 10.2.2). The ammonia lost through volatilisation comes from NH_4 and urea containing fertilisers. Urea decomposes in soil according to equation (1):

$$
\begin{aligned}
CO(NH_2)_2 + 2\,H_2O &\xrightarrow{\text{urease}} (NH_4)_2CO_3 \\
(NH_4)_2CO_3 + 2\,H^+ &\longleftrightarrow CO_2 + H_2O + 2\,NH_4^+ \\
2\,NH_4^+ &\longleftrightarrow 2\,NH_3 + 2\,H^+
\end{aligned}
\tag{1}
$$

Unfortunately no estimation method is available that covers the NH_3 losses due to both organic and mineral fertilisation. For this reason, two different estimation methods were selected to assess the ammonia emissions caused by fertiliser use. The parameters considered in the respective method and their relationships are described in the context of the methods.

Ammonia losses due to production and storage of organic fertilisers, such as manure and slurry are out of scope of this paper, because up to now it is not clear within Life Cycle Assessment, whether these emissions are to be allocated to animal husbandry or crop production.

10.2.1 Ammonia volatilisation due to organic fertiliser application

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

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

The parameters considered in the NH_3 volatilisation estimation method for organic fertilisers are:

- average air temperature;
- infiltration rate;
- time between application and incorporation or rainfall;
- precipitation or incorporation after application.

The *air temperature* influences the ammonia volatilisation in different ways. Firstly, the solubility of NH_3 and NH_4 decreases with increasing temperatures (ECETOC, 1994). Secondly, the higher the temperature is, the more the equilibrium of NH_3 and NH_4^+ is moved to NH_3 (ECETOC 1994). Finally, increasing temperatures lead to increasing concentrations of the NH_3/NH_4^+ solution due to drying (Horlacher and Marschner 1990). All these relationships result in increasing NH_3 emissions with increasing temperatures. The *infiltration rate* describes the capability of the soil to take up the NH_3/NH_4^+. Within the soil, the NH_3/NH_4^+ either stays in solution and is plant available, is biological oxidised (nitrification), or is adsorbed by clays and organic matter (ECETOC, 1994). Therefore, the infiltration of NH_3/NH_4^+ into the soil reduces the volatilisation rate. The amount of volatilised ammonia of course depends 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 and Marschner 1990). *Rainfall* reduces the volatilisation of NH_3 considerably due to increased solution of NH_3/NH_4^+ and increased infiltration into the soil. The amount of this reduction depends on the amount of precipitation (Horlacher and Marschner, 1990). *Incorporation* of the organic fertilisers also reduces the NH_3 losses, as the NH_3/NH_4^+ gets deeper into the soil (Sommer, 1992).

In the following estimation method, the NH_3 losses are calculated in percentage of the total NH_4-N applied in form of organic fertilisers. Thus, the NH_4-N content of the applied organic fertiliser should be known. Some figures in this respect are given in table 10.3. The original method of Horlacher and Marschner (1990) is calibrated only for the application of cattle slurry. The transfer of this method to other forms and origins of organic fertilisers (see table 10.3) has not been tested and should be validated.

Example: 80 kg N per hectare in form of cattle slurry were applied. Cattle slurry contains 55% NH_4-N. Thus 44 kg NH_4-N per hectare were applied in the example.

Table 10.3 N and NH_4-N content of different organic fertilisers

Fertiliser type	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.

10.2.1.1 Temperature

Air temperature is a key parameter for the NH_3 volatilisation rate. Therefore, the influence of the other parameters on the NH_3 volatilisation 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.

Example: temperature during and after application of the cattle slurry was 10-15°C.

10.2.1.2 Infiltration rate

The *infiltration rate* can be evaluated according to figure 10.2. 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.

Example: cattle slurry with medium dry matter content was applied on non compacted soil, therefore the infiltration rate was medium.

The maximum potential ammonia loss in percentage of the applied NH_4-N is shown for different infiltration rates and temperatures in table 10.4. The maximum potential ammonia loss is the ammonia loss that occurs, if no incorporation or rainfall after application took place.

Infiltration rate	Example
Low	application on cereal/corn stubble application of slurry with high dry matter content application of solid manure application on heavily compacted, water saturated soil
Medium	application on non compacted soil application of slurry with medium dry matter content
High	application on prepared soil with a lot of macropores, e.g. ploughed soil application on loose soil application of slurry with low dry matter content application of liquid manure

Figure 10.2 Evaluation of the infiltration rate
Source: Horlacher and Marschner, 1990 (modified).

Table 10.4 Maximum potential ammonia loss in % of the applied NH_4-N dependent on temperature and infiltration rate into the soil

°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

Source: Horlacher and Marschner, 1990 (modified).

Example: the maximum potential ammonia loss is 55% of the applied NH_4-N.

10.2.1.3 Time

Incorporation of the organic fertiliser into the soil or rainfall after application lead to a reduction of ammonia losses. The longer the *time period* between the application of an organic fertiliser and its incorporation or rainfall the higher is the ammonia loss. This is considered by multiplying the maximum potential NH_3 loss (see table 10.4) by a time factor (table 10.5), which is derived from field experiments (Horlacher and Marschner, 1990).

Example: precipitation took place one day after application of the organic fertiliser. Thus the maximum potential ammonia loss of 55% is multiplied by a time factor of 0.73, i.e. during the day without rainfall (or without incorporation) 73% of the maximum possible NH_3 emission *was lost. 55% multiplied by 0.73 gives 40%, i.e. 40% of the applied* NH_4*-N (17.6 kg* NH_3*-N per hectare) was lost between application and precipitation.*

Table 10.5 Time factors for different temperature classes

Temp.	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				

Source: Horlacher and Marschner, 1990 (modified).

10.2.1.4 Precipitation

The NH_3 volatilisation rate depends also on the *amount of rainfall* after application of the organic fertiliser. This is taken into account by introducing a rain factor (table 10.6), which is based on field experiments (Horlacher and Marschner 1990).

Table 10.6 Rain factors for different temperature classes (rainfall after application and before complete volatilisation in mm)

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
10-5	0.60	0.40	0.20	0
15-20	0.80	0.50	0.30	0

Source: Horlacher and Marschner, 1990 (modified).

*Example: One day after application of the cattle slurry: 8mm rainfall, temperature 10 to 15°C. Forty percent of the maximum potential ammonia loss (55%) is already lost (see chapter 10.2.1.3), i.e. 15% potential loss remained. Due to rainfall of 8mm and temperature of 10-15°C this remaining 15% potential loss is multiplied by a rain factor of 0.2. This results in 3% ammonia loss (15%*0.2 = 3%). That means 3% (1.3 kg NH_3-N per hectare) of the applied NH_4-N was lost since the beginning of precipitation.*

→ *ammonia emission due to application of organic fertilisers:*
17.6 kg NH_3-N per hectare + 1.3 kg NH_3-N per hectare = 18.9 kg NH_3-N per hectare= 22.95 kg NH_3 per hectare

2.7 kg NH_3 per tonne of wheat grain (yield: 8.5 tonne per hectare)

10.2.1.5 Other factors

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 and Marschner, 1990). Therefore, if the organic fertiliser was incorporated, 2% of the NH_4-N remained in the soil at the time of incorporation should be considered as loss (Sommer, 1992). The calculation is similar to the calculation for precipitation.

Other climatic factors influencing the NH_3 volatilisation rate are radiation and wind speed. High radiation as well as high wind speed lead to increase in ammonia losses. These factors are either well enough reflected by already integrated parameters (radiation by temperature) or very difficult to derive (wind speed) (Horlacher and Marschner, 1990). Nevertheless, especially wind speed may have a great influence on volatilisation 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 volatilisation (ECETOC, 1994):

- high pH (>8) -> high NH_3 volatilisation rate;
- high buffer capacity -> high NH_3 volatilisation rate;
- low cation exchange capacity -> high NH_3 volatilisation rate.

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

10.2.2 Ammonia volatilisation due to mineral fertiliser application

The ammonia emissions due to the application of mineral fertilisers are usually lower than from slurry and manure application (Isermann, 1990). However, considerable ammonia volatilisation can also take place when applying mineral fertilisers, dependent on the ammonium and urea content of the fertiliser, the weather conditions, and soil properties. The ECETOC (1994) proposed an estimation method to evaluate these emissions taking into account the different soil properties throughout Europe and the different NH_3 volatilisation risks dependent on the fertiliser type.

Group	Countries	Calcarous soil ($CaCO_3$)	pH	Sensitivity
I	Greece, Spain	common	mostly > 7	high
II	Italy, France, UK, Eire, Portugal, Belgium, Netherlands, Luxemburg	partly existent	>/< 7	medium
III	Norway, Sweden, Finland, Denmark, Germany, Switzerland, Austria	rare	mostly < 7	low

Figure 10.3 European countries grouped according to their NH_3 volatilisation sensitivity

As mentioned above, soil related parameters influence the risk of ammonia volatilisation. With increasing pH the equilibrium of NH_4^+ and NH_3 moves to ammonia, i.e. the risk of ammonia losses increases. Another soil related parameter influencing the ammonia volatilisation is the *buffer capacity* of a soil. High $CaCO_3$ contents counteract acidification and can therefore result in increased NH_3 losses. These parameters were considered by ECETOC (1994) to define three classes of different regional sensitivity to NH_3 volatilisation (figure 10.3).

Example: the wheat production system is located in Germany and is therefore allocated to group III.

ECETOC (1994) defined NH_3 emission factors for the following mineral fertilisers:

- urea;
- ammonium nitrate (AN), calcium ammonium nitrate (CAN), compound fertiliser (NP-N, NK-N, NPK-N), all with the same emission factor;
- ammonium phosphate;
- ammonium sulphate;
- other N fertilisers, with different emission factors.

Table 10.7 Emission factors (% NH_3-N loss of total applied mineral N) for different mineral fertilisers in Europe

Fertiliser type	European countries grouped according to figure 10.4		
	group I	group II	group III
Urea	20	15	15
Ammonium nitrate	3	2	1
Ammonium phosphate	5	5	5
Ammonium sulphate	15	10	5
Anhydrous ammonia	a)	a)	4
Nitrogen solution	8	8	8

a) Fertiliser not common in this group of countries.
Source: ECETOC, 1994 (modified).

The emission factors were derived by reviewing the literature (Asman, 1992; Buijsman et al., 1986; Whitehead and Raistrich, 1990; Isermann, 1990; SCB, 1991; all in ECETOC 1994) and taking into account the regional differences in NH_3volatilization sensitivity related to the fertiliser type (table 10.7).

Example: in the case study 130 kg N per hectare ammonium nitrate was applied to the winter wheat. According to table 10.12 the NH_3-N loss of total applied mineral N is 1%. One percent

NH_3-N loss of 130 kg N per hectare gives 1.3 kg NH_3-N per hectare, i.e. 1.6 kg NH_3 per hectare.
® *ammonia emission due to application of mineral fertilisers:*
1.6 kg NH_3 per hectare
0.19 kg NH_3 per tonne of wheat grain (yield: 8.5 tonne per hectare)

An incorporation of mineral fertiliser 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.

10.2.3 Result of the example

® *ammonia emissions due to application of organic and mineral fertilisers:*
22.95 kg NH_3 per hectare + 1.6 kg NH_3 per hectare = 24.55 kg NH_3 per hectare
2.7 kg NH_3 per tonne of grain + 0.19 kg NH_3 per tonne of grain = 2.89 kg NH_3 per tonne of grain

10.3 Nitrous oxide emissions

Nitrous oxide (N_2O) is one of the greenhouse gases, similar to CO_2 and water vapour, which are responsible for the absorption of about 95% of the longwave radiant energy in the atmosphere. The benefit of these gases is their 'potential to make our planet habitable', as the temperature on the earth's surface would be more than 30°C colder, if these gases were absent (Bouwman, 1995). However, increasing concentrations of greenhouse gases in the atmosphere are supposed to lead to increasing global temperatures, which for instance may result in rising sea levels.

Agriculture has a considerable share in the anthropogenic N_2O emissions (33-48%, see table 10.1), whereas N_2O itself contributes to a relatively small extent to the total global warming potential (5%, see table 10.2).

Nearly 80% of the N_2O emissions due to agriculture are related to the use of mineral and organic fertilisers. Biomass burning (e.g. shifting cultivation, deforestation) is responsible for about 20% (Kroeze, 1994). Two microbial processes are responsible for the 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).

Denitrification is the microbial reduction of NO_3 to N_2O and N_2. Denitrification occurs under anaerobic conditions, when special microorganisms (e.g. *Pseudomonas denitrificans*, *Thiobacillus denitrificans*) use NO_3^- and NO_2^- as a substitute for the absent oxygen. Under completely anaerobic conditions N_2 is the main product, whereas low oxygen concentrations lead to a higher N_2O/N_2 ratio (Granli and Bøckman, 1994).

Nitrification is the microbial oxidation of ammonium to nitrate. N_2O emissions can occur under aerobic conditions during oxidation of ammonium to nitrite. However, under anaerobic conditions, ammonium oxidising microorganisms, e.g. *Nitrosomonas* are also capable to reduce NO_2^- to N_2O similar to denitrification (Granli and Bøckman 1994).

Hence, *anaerobic conditions* are a prerequisite for N_2O emissions due to denitrification. But also the available *amount of nitrogen* 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.

Many 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. Figure 10.4 summarises the findings of Granli and Bøckman (1994) concerning these factors.

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

Figure 10.4 Key parameters influencing N_2O emissions from agricultural soils

Dependent on these parameters and their interactions, measurements of N_2O emission from different types of agricultural land show great variations. Nearly half of 36 analysed sets of measurements showed emissions rates above 3 kg N_2O-N per ha*year with a variation mainly from 3 to 10 kg N_2O-N per ha*year. But also N_2O emission rates up to 42 kg N per ha*year on an irrigated, heavily fertilised soil and 165 kg N per ha*year on a peat soil were measured. The other half of the measurement sets gave N_2O fluxes at or below 2.5 kg N_2O-N per ha*year (Bouwman 1990, in Granli and Bøckman 1994).

This clearly indicates a need for considering this variability of N_2O fluxes 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 fertilisers. From field experiments, he derived the following emission factor:

N_2O emission (kg N_2O-N per ha) = 0.0125 * N application [1] (kg N per ha) (2)

This emission factor of 0.0125 kg N_2O-N per hectare per kilogram N input is also used as default value for „estimating direct nitrous oxide emissions excluding cultivation of histosols' by the IPCC (1997). The soil order of histosols is characterised by organic matter contents of more than 30% (on sandy soils > 20%; Scheffer, 1989). For these histosols, the IPCC (1997) proposed area based default values of 5 kg N_2O-N per ha*year for temperate and 10 kg N_2O-N per ha*year for tropical climates. However, by far most of the agricultural soils do not belong to the soil order of histosols.

Example: the N application was 130 kg N per hectare mineral fertiliser and 80 kg N per hectare cattle slurry. The ammonia losses were estimated to be 20.2 kg NH_3-N per hectare (see chapter 10.2). This gives a NH_3-corrected N application of 189.8 kg N per hectare. According to equation (2) this means:

® *nitrous oxide emissions due to fertiliser use:*
*0.0125 * 189.8 kg N per hectare = 2.4 kg N_2O-N per hectare = 3.8 kg N_2O per hectare*
For the yield of 8.5 t wheat grain per hectare this results in 0.45 kg N_2O per tonne of wheat.

This emission factor is commonly used, because it is not yet possible to consider the other parameters (see table 10.4) appropriately. It is therefore suggested to take this approach for estimating the nitrous oxide emissions caused by agricultural practice.

10.4 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 the plants take up large amounts of nitrate. Within the plants, nitrate is reduced to ammonia and incorporated into organic structures. Furthermore, almost no downward water movement occurs during the vegetation period mainly due to high evapotranspiration rates.

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

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 as part of the life cycle inventory. As already stated the level of nitrate leaching depends strongly on different parameters.

The most important parameters determining the nitrate leaching rate are:

- *soil related*: field capacity in the effective rooting zone (FC_{RZe}) (mm);
- *climate related*: drainage water rate (W_{drain}) (mm/year);
- *agriculture related*: nitrogen balance (kg N per ha*year).

Nitrate leaching depends on the amount of water that drains through the soil profile. This *drainage water* moves the nitrate from the soil to the groundwater. The quantity of water in the soil is mainly determined by the water input through *precipitation* and the water output through *evapotranspiration* of plants and soil surface. To what extent and how fast this water drains through the profile is dependent on the capacity of the soil to adsorb the water in the soil. The *field capacity* is a measure for this property. The field capacity in the *effective rooting zone* takes additionally into account to what depth plants are able to extract water from the soil. A measure for the rate of drainage water that leaves the effective rooting zone within one year is the *exchange frequency of drainage water*. This measure expresses how often the whole drainage water rate of the effective rooting zone has been exchanged within one year. How much and in what concentration nitrate is emitted into the groundwater certainly depends also on the amount of nitrate in the soil. The *nitrogen balance* can be used to quantify the amount of NO_3 in the soil.

In the following, an estimation method for the prediction of nitrate emissions due to leaching from agricultural soils will be presented (DBG 1992). The parameters needed for this calculation should be either readily available or can be estimated.

10.4.1 Soil related parameters

The *field capacity in the effective rooting zone* (FC_{RZe}) can be calculated by multiplying the available field capacity (FCa) by the effective rooting zone (RZe).

$$FC_{RZe}\ (mm) = FCa\ (mm*dm^{-1}) * RZe\ (dm) \tag{3}$$

The available field capacity as well as the effective rooting zone strongly depends on the soil texture. The German Soil Science Association proposed six classes of available field capacity and five classes of effective rooting zone (DBG 1992) as described in table 10.8 and 10.9.

Table 10.8 Assignment of soil textures to 6 classes of available field capacity (FCa), medium soil density

Class (evaluation)	Soil texture a)	Available field capacity (FCa)(mm*dm^{-1})	
		range	average
1 (very low)	S1	< 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.
Source: DBG, 1992.

Table 10.9 Assignment of soil textures to 5 classes of effective rooting zone (RZe), medium soil density

Class (evaluation)	Soil texture a)	Effective rooting zone (RZe) (dm)	
		range	average
1 (very low)	Hh	< 3	2
2 (low)	S, Hn	3 – 5	4
3 (medium)	l'S, 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.
Source: DBG, 1992.

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

*Example: the soil texture is a loamy silt (lU), i.e. FCa is 24 mm*dm-1 and RZe is 10 dm. This results in a* FC_{RZe} *of 240 mm.*

10.4.2 Climate related parameters

The rate of drainage water (W_{drain}) is mainly determined by the precipitation rate (W_{precip}) and the evapotranspiration rate (W_{et}). The drainage water rate can either be measured or be estimated according to the climatic water balance (CWB), i.e. the calculation of the difference between precipitation (W_{precip}) and potential evapotranspiration (pW_{et}) per year.

W_{drain} (mm/year) = W_{precip} (mm/year)-pW_{et} (mm/year) (4)

The parameters precipitation and potential evapotranspiration should normally be available (e.g. for Germany: Deutscher Wetterdienst) or, in case of pW_{et}, can be calculated, e.g. using a method according to Haude (DVWK 1995).

Example: W_{precip} = 747 mm/year, pW_{et} = 538 mm/year, i.e W_{drain} = 747-538 = 209 mm/ year.

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*. This can be calculated using FC_{RZe} (3) and W_{drain} (4) as input parameters.

$$\text{(5)} \quad \text{exchange frequency } (a^{-1}) = \frac{W_{drain}\ \text{(mm/year)}}{FC_{RZe}\ \text{(mm)}}$$

*Example: exchange frequency = 209 mm*a-1/240 mm = 0.87 *a^{-1}*

The whole amount of NO_3 present in the soil at the beginning of the leaching period in autumn is supposed to be available for leaching due to its high mobility in the soil. The exchange frequency of the drainage water therefore 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 (6) is 1.

(6) leached NO_3-N (kg N per ha*year) = NO_3-N (kg N per ha) * exchange frequency (a^{-1})

Example: This calculation will be performed at the end of section 10.4.3, because the amount of NO_3-N (kg N per hectare) available for leaching is not yet known.

10.4.3 Agriculture related parameters

As a measure for the amount of nitrate in the soil after the vegetation period a nitrogen balance can be used. The N balance can be calculated as described in figure 10.5.

The nitrogen fertiliser input and the nitrogen outputs should be known within an LCA, as they are part of the defined system under investigation (fertiliser rate, crop removal) or are already estimated (NH_3-N, N_2O-N). If fertiliser 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, among others Loges et al. (1999) have presented a model for the quantification of N_2 fixation of legumes.

N input (kg N per ha)	N output (kg N per ha)
+ Mineral fertilizer	- Removal with harvested crops
+ Organic fertilizer	- NH_3-N emissions (volatilization, see section 10.2)
+ Biological N fixation	- N_2O-N/N_2-N a) emissions (denitrification, see section 10.3)
Σ input	Σ output
N-balance = Σ input-Σ output	

Figure 10.5 Calculation of the nitrogen balance (DBG 1992, modified)
a) N_2-N emissions are not considered as no method to estimate N_2 emissions is available.

Some other agricultural aspects can influence the nitrogen balance considerably:

- a reasonable nitrogen balance depends on the assumption that the nitrogen in- and outputs are relatively constant over long term, i.e. more than one crop rotation. However, short term changes may have a strong influence on the nitrogen balance, such as grassland ploughing that usually will lead to high nitrogen mineralisation rates;
- *intercropping* as well as *underseeding* may reduce the nitrogen surplus in autumn by more than 40% (Scheffer and Ortseifen 1996);
- due to the grazing and digesting animals (N out- and input) the nitrogen balance of pastures is very difficult to calculate and therefore highly uncertain.

Example: N inputs:
mineral fertiliser: 130 kg N per hectare
organic fertiliser: 80 kg N per hectare
biological N fixation: none
N outputs:
removal: 153 kg N per hectare
NH_3-N emissions: 20.2 kg N per hectare
N_2O-N emissions: 2.4 kg N per hectare
-> nitrogen balance: 130 + 80-153-20.2-2.4 = 34.4 (kg N per hectare)
no intercropping or underseeding

Nitrate emission into water via leaching, using results of the nitrogen balance and exchange frequency (see section 10.4.2):
*34.4 kg NO_3-N per ha * $0.87*a^{-1}$ = 29.9 kg NO_3-N per ha*year*
For the yield of 8.5 tonne wheat grain per hectare: 3.52 kg NO_3-N per tonne of wheat and year.

10.5 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 LCAs 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 as they are presented in this paper.

To *measure actual N emission rates* is money and time consuming and therefore often not feasible in Life Cycle Assessments. Furthermore, actual measurements of N emissions often show great variations (e.g. Isermann 1990, for NH_3) and reflect a snapshot of the specific conditions of the moment when measured. 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 reflect an average emission, which is assumed representative for the system examined in the LCA. This means to review the literature in order to look for emission rates obtained under conditions similar to those of the system under investigation. 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 this strongly depends 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, 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 estimation methods simplify the complex conditions responsible for the formation and amount of emissions, taking into account only a few well know factors, assuming that these are the most important ones. However, the presented procedures could provide useful tools to obtain reasonable nitrogen emission data for al life cycle inventory. Of course checking the data derived from such estimation methods for instance against official regional statistics (Halberg 1999) may be required.

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11. Scaling up of milk production data from field plot to regional farm level

R. Loges, M. Wachendorf and F. Taube [1]

Abstract

Dairy farming in northern Germany as elsewhere in Western Europe is generally intensive with high production per animal and acreage. Due to high nitrogen inputs through purchased concentrates and nitrogen fertiliser accompanied by low nitrogen outputs through milk and meat production, these farms show on average a high nitrogen surplus, which especially on sandy soils threatens groundwater quality through nitrogen leaching. To reduce this surplus it is important to develop strategies to improve nitrogen utilisation in the barn and on the field. Under northern German conditions a reasonable experimental basis for this is missing. Based on an interdisciplinary research project, the agricultural faculty of the University of Kiel investigates in a system-analytical way nitrogen fluxes on dairy farms. The aim with this project is to develop management strategies to increase the nitrogen use efficiency to reduce nitrogen surpluses on dairy farms. Through for example variations of fertiliser input, botanical composition of the sward and the form of grassland use nitrogen surpluses can be influenced. To show interactions between animal and forage production as well as economical and ecological aspects models will be used after calibration with collected experimental data. A weather based crop growth model already is able to simulate growth and quality of forage for a wide range of different soil and climatic conditions and different management strategies. Since it is always difficult to scale up findings from small experimental plots to farm level, chosen management strategies are analysed parallel on small plots and on large scale fields of the experimental farm, as well as on farms in other regions, to find e.g. factors for forage losses while using farm machinery, with which plot data can be corrected so that it is possible to give advice that is more acceptable to farmers than small plot data.

11.1 Introduction

Milk production in northern Germany usually takes place on specialised dairy farms. According to the 1998 report of the agricultural advisory service (Landwirtschaftskammer Schleswig-Holstein 1998) these specialised farms keep on average 65 dairy cows on 85 ha farm land with a milk yield of 6,820 kg/cow/year. Because of high nitrogen inputs through purchased concentrates and nitrogen fertiliser accompanied by low nitrogen outputs through milk and meat production, these farms show on average a quite high nitrogen surplus of 170 kg N/ha/year. Especially on sandy soils where specialised dairy farms are the main farmtype, such a nitrogen surplus threatens groundwater quality through nitrogen leaching.

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To reduce this surplus it is important to develop strategies to improve nitrogen utilisation in the barn and on the field. Generally, a reasonable experimental basis for this is missing. Under German conditions, single strategies have been investigated, but mostly in different experiments. Systemanalytical studies of the process of milk production, which consider interactions between animal, forage production, as well as economical and ecological aspects, do not exist for northern Germany.

Also for the analysis of environmental effects of milk production, the availability of data for Germany is limited. The common main data source for agriculture in Germany the 'Statistisches Jahrbuch über Ernährung, Landwirtschaft und Forsten' (Bundesministerium für Ernährung, Landwirtschaft und Forsten 1998), an annually published report on German agriculture statistics, allows just indirect calculations of the environmental aspect of dairy farming via average input and output data. More detailed information is sometimes available on a regional level, but not for all regions in Germany. A good example for a detailed analysis of the economic situation and ecological effects of dairy farming is given by the annual report 'Rinder-Report' published by the agricultural advisory service (Landwirtschaftskammer Schleswig-Holstein, 1998), where also information on nutrient surpluses in different dairy farm types are given. Measured data for N losses via leaching, ammonia volatilisation, and denitrification for German dairy farms especially in relation to farm productivity are hardly available.

On this basis, an interdisciplinary research project with the topic 'Nitrogen fluxes on specialised dairy farms' was established at the agricultural faculty of the University of Kiel (figure11.1).

The aims of this on-going project are:

- quantification of the specific nitrogen use efficiency in the different stages in the process of milk production;
- modelling N fluxes with respect to different environmental conditions and management strategies;
- validation of modelled data against farm level data;
- transfer and validation of results to other farm types;
- optimising nitrogen use efficiency in the process of milk production;
- reducing nitrogen losses on dairy farms.

11.2 Material and methods

The investigation includes field plot trails as well as investigations on farm level and regional farm level data. The project period is from 1997 to 2003. Experimental basis is field plot trails on the dairy research farm Karkendam belonging to the agricultural faculty of the University of Kiel. Different management strategies to increase the nitrogen use efficiency are tested in these multifactorial field experiment, through investigations of crop yield, forage quality, soil nitrogen balance, and ground water quality.

For system analysis (figure 11.1) and to scale up the findings, two main farm intensities are chosen and compared with each other on farm level. For this part of the investigation the experimental farm Karkendamm has been divided and two main farming systems which differ in their N input established; each with 55 dairy cows and 70 ha farmland. System 1 represents the average intensity of typical northern German dairy farms. System 2 represents a low-N-input system close to organic farming with nitrogen fixation by clovers as the main nitrogen source. Due to statistics and to management aspects a division into smaller herds and further farm partitions was not possible.

Comparing data from field plots with farm level data from the experimental farm, regional farm level data (which will be gathered on chosen pilot-farms in the second project phase from the year 2000 to 2003), and data published annually by the agricultural advisory service (Landwirtschaftskammer Schleswig-Holstein, 1998), should make it easier to transfer findings from field experiments to advisors and farmers in the northern part of Germany.

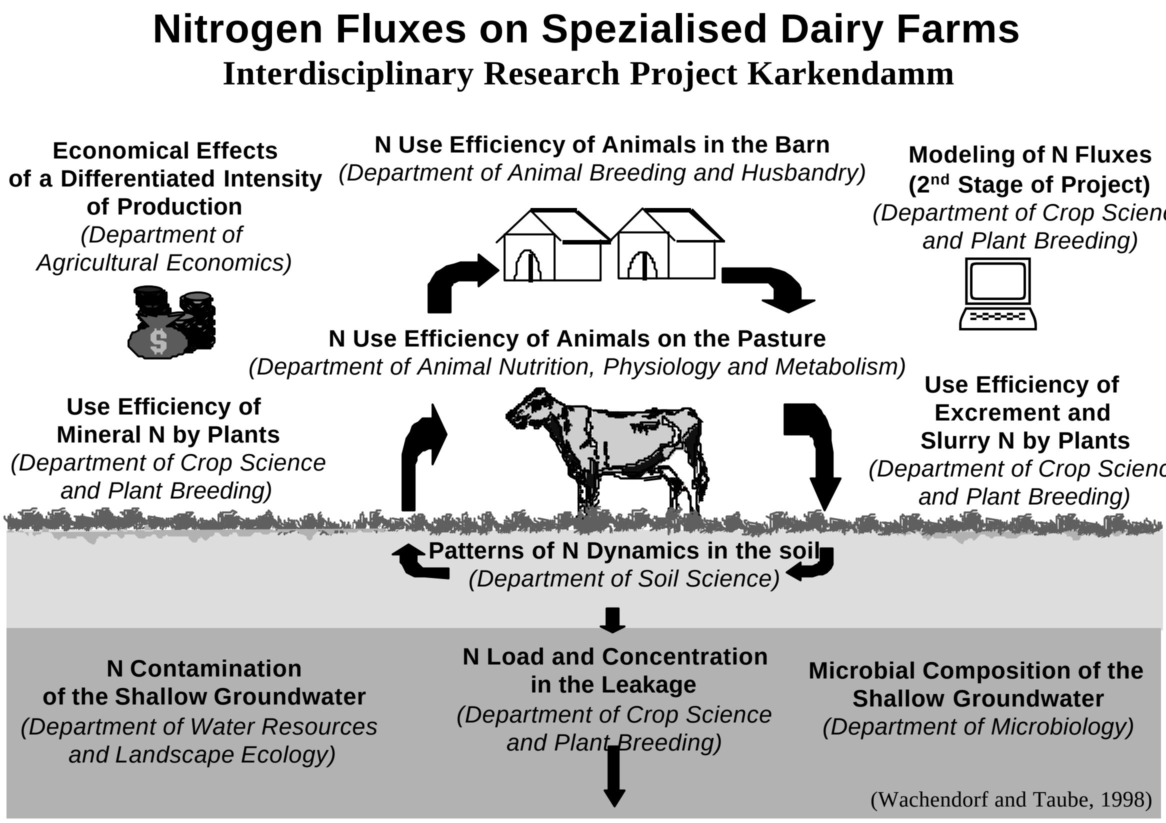

Figure 11.1 Subprojects and participants of the interdisciplinary research project

Due to the fact that a dairy farm consists of different components, and milk production is a result of different processes that interact and depend on each other, it is difficult to experimentally evaluate different management factors on the dairy farm system as a whole. The use of models may overcome some of these problems. Therefore it is the intention to use existing models or to develop own models which are easy to handle and will need only easily available input parameters, like weather data, amount of used fertiliser or clover content.

Until now simple regression models for nitrogen fixation have been developed (Høgh-Jensen et al., 1998; Loges, 1998), which use clover content or clover yield as input variables. The departments own crop growth model FOPROQ (Kornher and Nyman, 1992), which simulates weather based yield and quality changes in the herbage, has been parameterised.

The large and detailed dataset obtained in the presented project consists of:

- yield formation studies based on weekly gathered plant samples;
- examination of N_2-fixation and nitrogen use efficiency by ^{15}N-techniques;
- soil-water samples gathered each week using more than a thousand porous ceramic cups; and
- measured changes in the mineral and organic N pool in the soil.

It is the main intention to use this dataset to calibrate and validate some of the existing models and estimation methods, since there already exist good models for the simulation of nitrate leaching (Addiscott and Whitmore, 1987 and 1991; Scholefield et al., 1991; Hutchings and Kristensen, 1995 and Hansen et al., 1990 and 1991), simulation models of ammonium volatilisation (Hutchings et al., 1996 or Elzing and Monteny 1997) and an estimation method by Horlacher and Marschner (1990), and farm level models like the 'Integrated Economic and Environmental Farm Simulation Model (FASSET) by Jacobsen et al., (1998), which includes besides an economic analysis on farm level also the simulation of nitrate leaching and denitrification.

Also models describing the relationship between the fodder inputs and the nitrogen use efficiency of the animal can be calibrated with the data gained during the experiment.

The data provided by the models will be used in dairy farm simulation models in order to evaluate the overall impact of crop management decision and to compare these with effect of measures in other components of the milk production system.

11.3 Results

In the following, the first results of the ongoing project are presented. Figure 11.2 shows the impact of management on nitrate leaching under grassland. A decrease of mineral nitrogen fertilisation reduces leaching of nitrate. Grazing causes higher nitrate leaching than cutting. At Karkendamm, grazing caused already without nitrogen fertilisation high nitrate concentrations in the leakage water, whilst as a result of cutting, high nitrate concentrations (above the EU-limit for drinking water of 50 mg l^{-1}) were observed only after a nitrogen application of over 300 kg N ha^{-1}. Grazed white clover/grass with a calculated N_2-fixation of 120 kg N ha^{-1} showed higher nitrate leaching than grass swards with a nitrogen fertilisation of 120 kg N ha^{-1}.

To demonstrate that modelling is a useful tool for showing the consequences of management decisions, figure 11.3 shows the first results of the use of the weather based crop growth model FOPROQ (Kornher and Nyman 1992), which is a model for the prediction of growth and quality change of grass swards. Here, the model was calibrated for the yield of different managed swards of red clover, Italian ryegrass, and red clover/grass mixtures from different years and different sites in northern Germany. Linear regressions are shown for the dry matter yield as a measure of the compatibility of the model values with the observed data. As additional F-tests for intercepts and slopes of the regression lines did not show any significant deviation from the line that represents equivalence between observed and calculated data, yields of different swards can be simulated for a wide range of different combinations of soil and climatic data for different management strategies (Wachendorf et al., 1996).

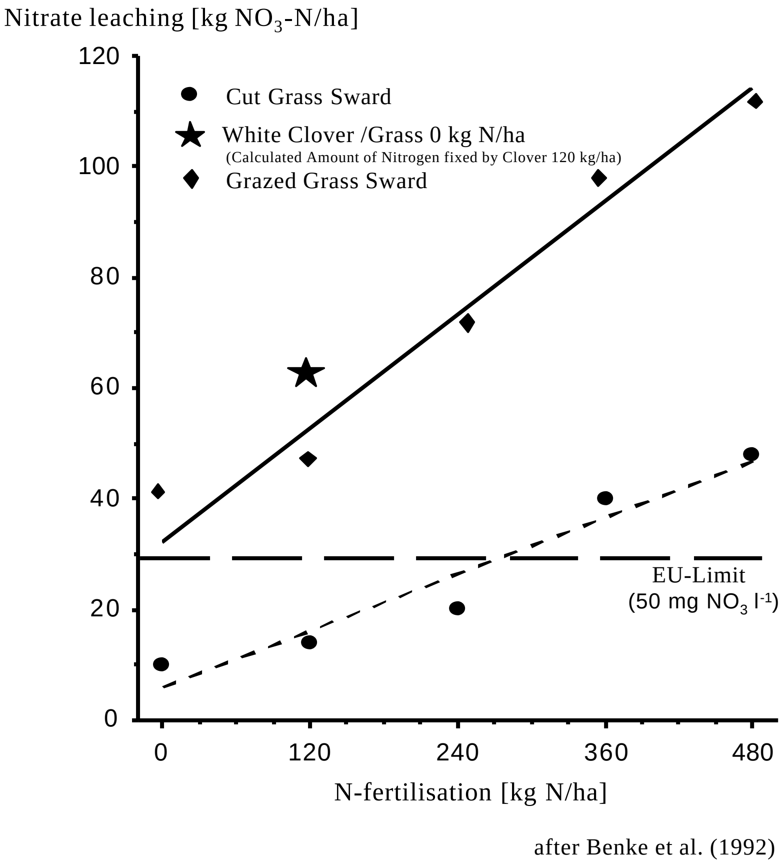

Figure 11.2 Relationship between nitrogen fertilisation and nitrate leaching as affected by the form of grassland use (Karkendamm 1989-91)

The named model is already used by the advisory service to predict the point of time for the first silage cut in North Germany, based on soil conditions, meteorological conditions in spring, and the weather forecast.

For the calculation of N balances in legume based forage production, it is necessary to know the input of nitrogen through N_2-fixation by clovers. As the measurement of N_2-fixation is laborious and expensive, on-farm measurement is not possible. It is therefore necessary to find simple methods based on easily available parameters to predict N_2-fixation. Høgh-Jensen et al. (1998) and Loges (1998) showed strong correlation's between clover yield and N_2-fixation (figure 11.4), which was much more strongly correlated than the correlation between clover content and N_2-fixation that is typically used.

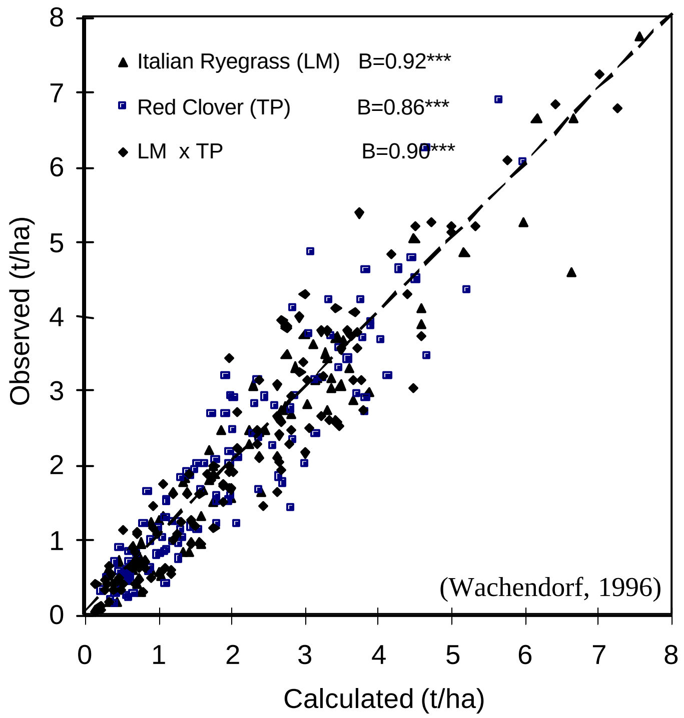

Figure 11.3 Relationship between observed experimental data and calculated results of the prediction model for growth and quality change of grass swards FOPROQ

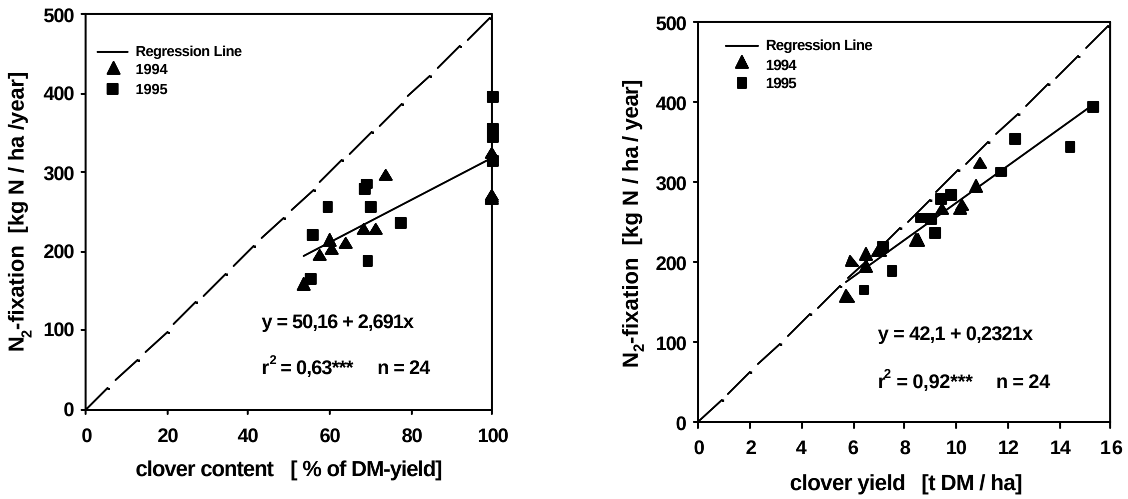

Figure 11.4. Effect of clover content and clover yield on N_2-fixation by red clover/grass in the first production year measured with the ^{15}N-dilution method (Loges 1998)

Based on data from Denmark and Northern Germany, Høgh-Jensen et al. (1998) created the following empirical model for quantification of N_2-fixation of legumes:

$$N_{fix} = DM_{legume} * N\% * P_{fix}$$

$$P_{fix} = P_{shoot} * (1 + P_{root+stubble} + P_{trans\text{-}soil} + P_{trans\text{-}animal} + P_{immobil})$$

with:

DM_{legume}	=	harvested drymatter yield of the legume;
N%	=	N-concentration in legume- drymatter;
P_{fix}	=	percentage of fixed N in the total N of the legume;
P_{shoot}	=	percentage of fixed N in the legume shoot;
$P_{root+stubble}$	=	percentage of fixed N in the legume root and stubble;
$P_{trans\text{-}soil}$	=	percentage of fixed N transferred through soil to the companion grass;
$P_{trans\text{-}animal}$	=	percentage of fixed N transferred through the animal to the companion grass;
$P_{immobil}$	=	percentage of fixed N which is immobilised in the soil.

After parameterising the model for red clover with the parameters from table 11.1, it is possible for farmers and advisors to attain a quite accurate estimate of the amount of fixed nitrogen based on red clover yield and available data in the literature for Denmark and northern Germany. Preliminary parameters for other legumes like peas, alfalfa and white clover/grass can be found in the paper of Høgh-Jensen et al. (1998).

Table 11.1 Parameters for prediction of N_2-fixation of red clover and red clover/grass

N%	P_{shoot}	$P_{root+stubble}$	$P_{trans\ soil}$	$P_{trans\ animal}$	$P_{immobil}$
3.3	0.75	0.25	0.10	-	0.25

11.4 Validation

A selection of first results from the project was presented in figures 11.2, 11.3 and 11.4. The aim of the project is now to validate the findings of the project on a larger scale so that it is possible to use them, for example by advisors.

As a first step towards a proper scale up, these results and model based predictions derived from small plots have to be compared with measurements from the farm scale at Karkendamm. Each plot experiment integrates plots managed in the same way as whole fields or pastures on the farm.

By comparing the data from these plots with the large scale fields, it is possible to find factors for forage losses, e.g. while using farm machinery, with which plot data can be corrected, so that it is possible to give advice that is more acceptable to farmers than small plot data.

The next validation step has to consider regional variation in soil and climate. In the second project phase from the year 2000 to 2003, chosen appropriate management strategies are to be tested on various farms throughout North Germany and compared with the results from Karkendamm. With this information it is possible to transfer the findings to other regions in North Germany more accurately than if the transfer was carried out directly from the small plot experiments, which is the normal procedure.

The agricultural advisory service (Landwirtschaftskammer Schleswig-Holstein 1998) publishes annually high quality data about productivity and nutrient use efficiency of different farm types on a regional level. Therefore, new management strategies, which are based on parallel plot experiments and on-farm-research, can be better integrated into recommendations to farmers by the advisory service than data from only small plot experiments.

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12. Interrelationships between nitrogen balances and technical and structural characteristics of dairy farms in Northern Italy

Kees de Roest [1]

Abstract

In this paper, the interrelationships are explored between the nitrogen surplus and technical and structural variables measured in dairy farms in the northern Italian region of Emilia-Romagna. Accountancy data and farm survey data have been used to calculate nitrogen balances. Multiple regression analyses applied to factor scores derived from principle component analysis was used to identify independent factors explaining the variance in the farm nitrogen surplus.

The high variability in nitrogen surplus among farms indicates that many farms have large margins for improvement of their nitrogen balance. In particular, dairy farms which rely for their roughage production on alfalfa have significant better environmental performances than farms using maize silage. Moreover, herds with high milk yields are less efficient in their nitrogen use than less productive herds, and nitrogen efficiency is higher in small herds than in large herds.

12.1 General outline of the nitrogen problem in Italy

Italian agriculture is highly differentiated with very large differences in intensity of farming systems, and its development is severely hampered by the natural handicaps. Almost 70% of the Utilised Agricultural Area (UAA) is classified as hillside or mountain farming, where extensive forms of agriculture prevail. The plains have the highest production potential for agriculture and here the far most majority of agricultural production takes place. Because of the natural handicaps and the very rapid increase of food demand in the last decades, Italy is not selfsufficient with many agricultural products. In order to reduce dependency from imports intensity of agriculture in the plains has been raised considerably.

Livestock production finds most favourable climatic and infrastructural conditions in the north of the country where most of animal production is concentrated. The Po valley representing altogether only 18% of the utilised agricultural area of the country accounts for 49% of cattle, 62% of pigs and 63% of poultry population of Italy. The high intensity of land use by animal production farms combined with the high vulnerability for nitrogen leaching has created nitrogen pollution problems in the ground- and surface water.

The livestock sector has undergone an extensive reorganisation process characterised by increasing farm specialisation associated with an increasing concentration of slurry production.

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According to the 'Report on the State of the Environment' (Italian Ministry of the Environment. 1997), nitrate is the most widespread pollutant in groundwater. Values exceeding 50mg nitrate per litre can be found in Campania plain, in the regions of Puglia and Marche.

Groundwater exceeds the level of 50 mg nitrate per litre (up to 150 mg/l) also in some areas of the Po Valley, due to heavy fertilisation with mineral fertilisers, animal manure, but also to leaking from sewers.

The most recent SINA report on surface water conditions shows that 58% of the data on nitrate in surface water are in the range 1 to 10 mg nitrate per litre (water quality class 3, bad, according to Italian Water Research Institute (IRSA) ranking), and 26% is exceeding 10 mg nitrate per litre (water quality class 4, very bad).

Significant instances of eutrophication were recorded in recent years, mainly in the Adriatic sea, due to nutrient transport by the rivers and, possibly, to the unbalanced ratio of nitrogen and phosphorus.

Ammonia emissions are mainly related to livestock production. According to CORINAIR, 1990 inventory for Italy, 91% of total NH_3 emissions are assigned to agriculture. Ammonia pollution originates from animal sheds, manure and slurry storage, and manure and slurry application. As pasturing of sheep and cows in Italy occurs only in mountain areas with low animal densities, this source of ammonia emission is not relevant. Ammonia emissions from storage tanks are to be considered high, as these are seldom covered. A recent national inventory on some atmospheric pollutants (ENEA, 1997, still not published) gives an estimation for 1994 ammonia emissions from animal husbandry equal to 331 10^6 kg NH_3/year, of which 33% attributed to animal housing, 25% to manure stores, 38% to manure application and 4% to pasture. Ammonia emissions come mainly from cattle (62%), followed by pigs (17%), poultry (14%) and ovine (5%). Other animal categories are almost negligible.

In this study, the total nitrogen surplus of dairy farms in Emilia-Romagna has been estimated. The methodology used arrives at an estimate of the total emissions of dairy farms, but is not able to distinguish between the different types of emissions. CRPA model studies based on experimental data are underway to estimate ammonia emissions from stables and manure storage in order to determine emission factors appropriate for the livestock production in the Po Valley (Bonazzi, et al.,1996 and 1997).

12.2 Introduction to the nitrogen balance study

In the region Emilia-Romagna, two dairy farm types dominate the sector. Farms which destine milk to the production of Parmigiano-Reggiano cheese (PR), and farms which destine the milk to industrial processing. The first type of dairy farm has to follow strict production regulations that raise their production costs (De Roest et al., 1994). As the PR cheese is made out of raw milk and knows a maturing period of at least 18 months dairy farms which deliver milk have to respect a code of practice, which defines the way milk has to be produced and processed. On dairy farms, which deliver milk to PR processing plants, it is forbidden to feed silage or industrial by products to the cows. Furthermore, a list of feed components has been set up, which is not to be used in compound feeds destined to cows of the PR system. The production of PR cheese is highly relevant for the Italian dairy system, as it interests about 15% of Italian milk production.

In this study, we will go into the details of the nitrogen balance of PR dairy farms with respect to farms that destine milk to industrial processing. At first, the question posed here is to which extent the PR dairy system is able to produce in a more ecocompatible way than the industrial dairy farmers? Secondly, we will ask ourselves which factors influence the nitrogen balance of these farmers independently of the destination of the milk.

After a brief description of the methodology used in the implementation of environmental auditing techniques, the economic and structural characteristics of the farms under study will be illustrated. The findings of the research into nitrogen balances will then be analysed and the importance of the altitude level, milk destination, milk yield and herd size will be assessed. The multivariate analysis, based on a principal component analysis, followed by a multiple regression analysis, will seek to determine which factor has the greatest bearing on the creation of nitrogen surpluses in dairy farms.

12.3 Methodology

In the adoption of the methodology required to assess a farm mineral balance, account was taken of the techniques already used in previous published research in this area (CLM et al., 1992; Brouwer et al.1994; Schleef and Kleinhanss, 1993). Consistently with principles enunciated in these earlier studies, a methodology was developed which takes account of the specific nature of farming in Emilia Romagna (De Roest and Fornacari, 1995). The data used are drawn from three distinct sources:

- accountancy data drawn from the Regional Accountancy Network of the Region Emilia-Romagna;
- technical data collected through a questionnaire;
- various sources from literature. These sources have been used primarily to assist in the creation of parameters on which the analytical estimates have then been based.

The farm questionnaire provided the source of data identifying the use of chemical and organic fertilisers on the farms.

The group of farms surveyed was made up of 179 specialised dairy farms. Table 12.1 shows the detailed distribution of the farms in accordance with the end use for the milk produced and in relation to the altitude level in which they are located.

The specialised nature of the dairy farms shown in table 12.2 is typical of the area under consideration: mediumsized livestock and forage producing farms. The percentage of cultivated land devoted to the forage crops is never less than 75% and in the most extreme cases it is up to 95%. The herd size of the farms varies between 18 and 26 cows in the Appennine Mountains and between 41 and 72 cows on the plain. The intensity of the land use, measured in heads of cattle per hectare of forage crops, ranges from 1.1 to 2.2 cows per hectare in the Parmigiano Reggiano group and from 1 to 2.4 cows per hectare in the industrial dairy farms. The annual milk yield per cow ranges between

4,800 and 6,100 kg in the Parmigiano Reggiano group and between 3,700 and 6,200 kg for the industrial dairy farms.

Table 12.1 The farm sample

Destination of milk	Mountains		Hills		Plain		Sample	
	farms	%	farms	%	farms	%	farms	%
Parmigiano Reggiano	54	85.7	42	80.8	55	85.9	151	84.4
Industrial milk	9	14.3	10	19.2	9	14.1	28	15.6
Total sample	63	100	52	100	64	100	179	100

Source: Own calculations.

Table 12.2 Characteristics of the farm sample

	Parmigiano Reggiano farms			Industrial dairy farms		
	mount. 54 cases	hills 42 cases	plain 55 cases	mount. 9 cases	hills 10 cases	plain 9 cases
1- Division of crops						
Cultivated land area (in ha)	27.8	30.2	23.7	29.3	30.3	36.1
% forage crops	95.2	87.9	85.7	83.2	76.7	82.3
2-Herd Size and Working Force						
Number of milking cows	26.0	31.6	41.8	18.4	33.17	71.5
Cows per ha of forage area	1.1	1.3	2.2	1.0	1.6	2.4
Hours per cow per year	192	173	145	180	149	92
3-Efficiency and intensity						
Kg milk per forage ha	5,352	6,970	13,373	3,952	7,811	15,168
Forage Maize Yields (tons/ha)	3.74	16.90	28.61	13.60	19.67	55.09
Alfalfa yields (tons/ha)	5.70	7.22	10.86	6.04	7.60	9.33
Milk yields per cow (kg/cow)	4,800	5,130	6,130	3,730	4,210	6,200
kg milk per kg feed	2.25	2.44	2.56	3.83	3.16	3.01
4- milk cost and management results (millions ITL)						
Gross Margin/Annual Work Unit	27.9	34.8	48.0	20.2	39.2	55.0
Net income per family worker	20.5	24.2	38.2	10.2	45.5	63.7

In making the calculations required to assess the nitrogen balance, the flow diagram (figure 12.1) shows 13 input headings and 7 output headings. The difference between input and output represents the nitrogen balance.

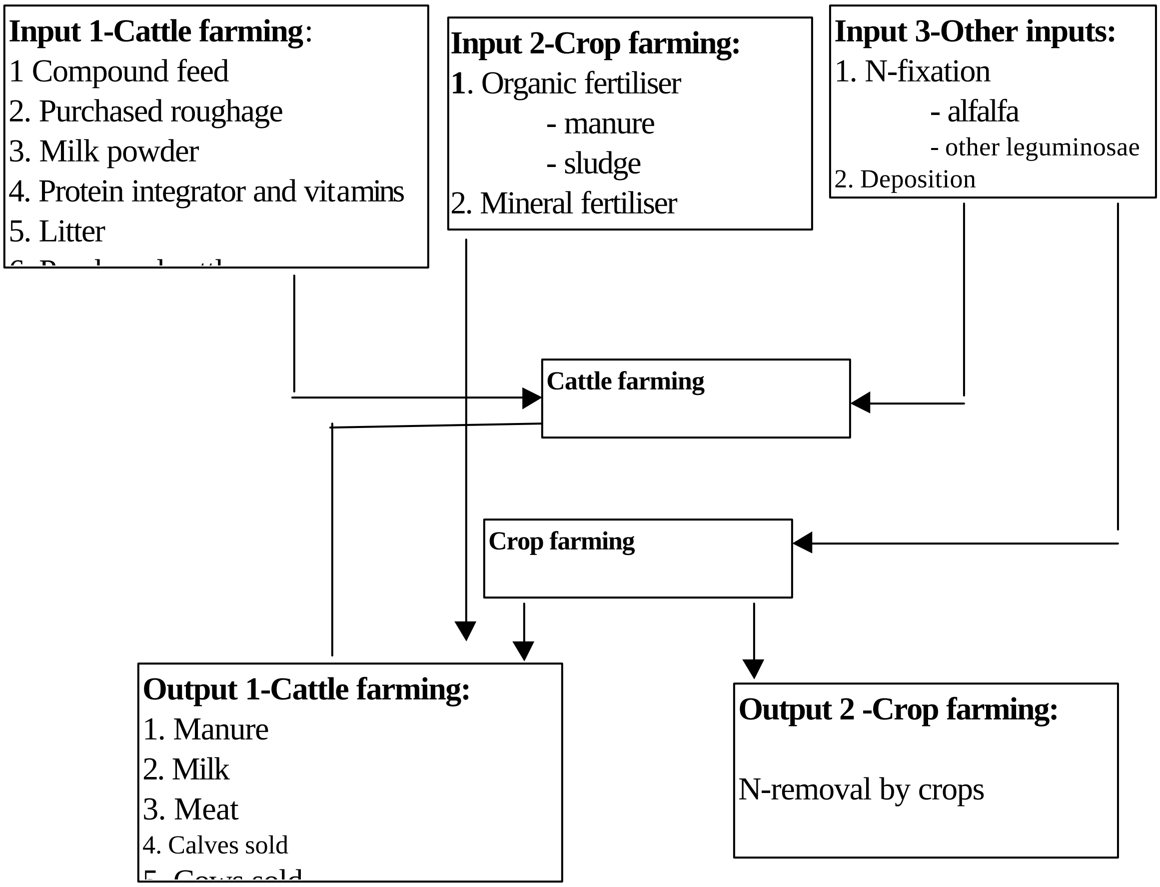

Figure 12.1 Diagram of nitrogen inputs and outputs for dairy farms
Source: CLM et al. (1992).

12.4 Farm nitrogen balance and intensity of land use

The greatest problem arising from intensive farming techniques is the breaking of the mineral cycle on the farm. The continuous increase in the animal to hectare ratio, caused by the increase in the price of land, has created an increase in mineral input flows. The loss of nitrogen in the form of ammonia, N_2O and nitrates are a result of this development in livestock farming. It is thus evident that there is a close relationship between the total nitrogen surplus of the farm and the stocking rate, measured by the number heads of cattle per hectare. The nitrogen surplus derived from the nitrogen balance calculations represents the farm's total nitrogen losses irrespective of the form they may take.

The following graph indicates the relationship between the nitrogen surplus per hectare and the number of cows per hectare of cultivated land. The linear regression coefficients are statistically significant ($P <0.05$)-proof of the close link between the two variables. Interesting is the variance around the regression line, which indicates that the same stocking rate may still present significant differences in the nitrogen balance.

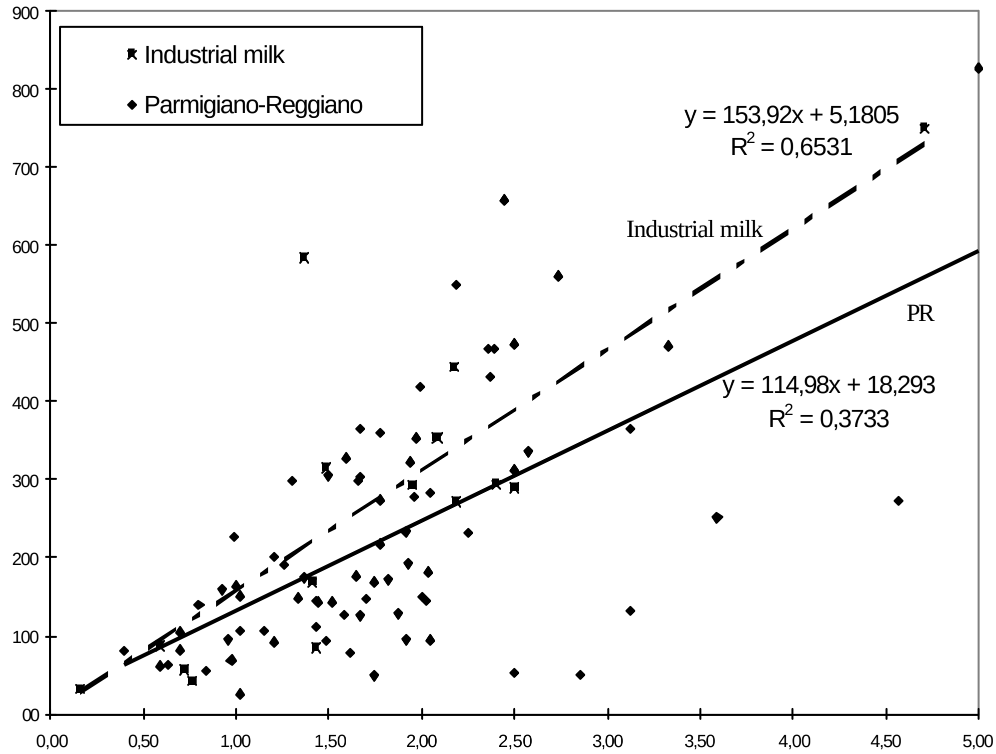

Figure 12.2 Nitrogen surplus per hectare according to cow density and milk destination - plain farms -

12.5 Farm nitrogen balance, altitude, and milk destination

A second analysis of differences in the farm nitrogen balances was designed to identify the importance of the topological location of the farm and the end use of the milk (for Parmigiano Reggiano cheese or industrial milk).

There is a marked difference in the nitrogen surpluses produced by farms on the plain and mountain farms (table 12.3). The nitrogen surpluses produced by mountain farms varies between 48 and 122 kg of nitrogen per hectare. In contrast, on the plain the surplus ranges between about 230 kg and 309 kg per hectare. These differences in nitrogen surpluses can be explained by the higher stocking rate and the increased dairy cow productivity on the plain farms compared with the mountain farms. It is not surprising to find that farms on the plain exercise greater pressure on their environment than those in the mountains.

It is interesting to compare farms working within the Parmigiano Reggiano system and industrial dairy farms (table 12.3 and 12.4). This comparison has been done only for the farms on the plain, to eliminate the effect of the altitude zone. Thus, the Parmigiano Reggiano farms on the plain show a total nitrogen loss of 239 kg of nitrogen per hectare. This figure compares with the 309 kg per hectare for the industrial dairy farms. The difference is in the order of almost 30%. A substantial part of the difference in nitrogen surpluses has to be attributed to the different farming system, considering that the number of cows per hectare for the industrial dairy farms is only 10% greater than the Parmigiano Reggiano farms, and that milk yields per cow are almost identical between the two groups.

If the composition of the nitrogen balance is examined in greater detail, one notices that the industrial farms use more purchased feeds and mineral fertilisers per hectare. This indeed, reflects the crucial difference existing between the two farming types in relation to the different cattle feeding regimes. A further factor marking the differences between the two farming types is in the nutrient ratios for the different crops. The fertilisers coming into the farm (whether organic or chemical in nature) represent 18% of the total nitrogen input in the Parmigiano Reggiano balance, compared with 23% for the other farm group. This can be explained by the fact that the first group's requirement for nitrogenous substances is reduced because of the use of N-fixing alfalfa in the cropping pattern. The second group of farms concentrates more on the growing of graminaceous and maize crops for silage, which require the application of mineral fertilisers.

Table 12.3 Inputs and outputs of nitrogen in dairy. Parmigiano-Reggiano dairy farms

	Mountains 54 cases		Hills 42 cases		Plain 55 cases	
	kg/ha	%	kg/ha	%	kg/ha	%
Purchased feed	66.65	43.6	81.54	40.5	131.60	40.7
Purchased roughage	27.12	17.7	40.86	20.3	85.20	26.3
Purchase of organic fertiliser	6.04	3.9	12.79	6.3	22.17	6.8
Chemical fertiliser	9.12	6.0	21.89	10.9	35.72	11.0
Atmospheric deposition	18.30	12.0	18.30	9.1	17.63	5.4
Purchase of young beef stock	1.26	0.8	0.34	0.2	0.72	0.2
Purchase of milk powder	1.20	0.8	0.77	0.4	1.16	0.4
Litter	2.52	1.6	3.39	1.7	6.99	2.2
N-fixing by leguminous crops a)	20.80	13.6	21.59	10.7	22.48	6.9
Total inputs	*153.02*	*100*	*201.45*	*100*	*323.68*	*100*
Sales of organic manure	1.19	3.9	3.92	7.8	8.57	10.2
Milk sold	23.77	77.1	32.66	64.6	54.33	64.6
Cows sold	3.44	11.2	4.78	9.5	7.22	8.6
Calves sold	0.51	1.7	0.94	1.9	1.20	1.4
Meat Sold	3.95	12.8	5.72	11.3	8.41	10.0
N-removal by non legum. crops	1.91	6.2	8.23	16.3	12.77	15.2
Total outputs	*30.83*	*100*	*50.53*	*100*	*84.08*	*100*
Nitrogen balance	*122.19*		*150.92*		*239.60*	

a) N-Fixation, net of removal.

Thus, the Parmigiano Reggiano dairy farms can, on average, be said to use nitrogen with greater efficiency than those unrestricted by the Parmigiano Reggiano production regulations. figure

12.2 shows also clear evidence of this statement. At increasing stocking rates, the industrial dairy farms face a more rapid deterioration of the nitrogen surplus per hectare than the dairy farms that deliver milk for the production of Parmigiano-Reggiano cheese.

Table 12.4 Inputs and outputs of nitrogen in industrial dairy farms

	Mountains 9 cases		Hills 10 cases		Plains 9 cases	
	kg/ha	%	kg/ha	%	kg/ha	%
Purchased feed	19.56	27.8	70.39	44.1	189.88	47.1
Purchased roughage	6.00	8.5	16.89	10.6	82.45	20.4
Purchase of organic fertiliser	1.65	2.3	3.01	1.9	0.00	0.0
Chemical fertiliser	5.90	8.4	37.50	23.5	93.09	23.1
Atmospheric deposition	18.30	26.0	18.30	11.5	18.30	4.5
Purchase of young beef stock	0.22	0.3	0.04	0.0	0.62	0.2
Purchase of milk powder	0.59	0.8	0.74	0.5	1.78	0.4
Litter	0.75	1.1	1.08	0.7	2.45	0.6
N-fixing by leguminous crops a)	17.32	24.6	11.80	7.4	14.68	3.6
Total inputs	*70.29*	*100*	*159.76*	*100*	*403.25*	*100*
Sales of organic manure	0.00	0.0	0.00	0.0	0.00	0.0
Milk sold	12.89	58.1	27.01	59.6	62.33	66.3
Cows sold	1.84	8.3	3.33	7.3	8.06	8.6
Calves sold	0.44	2.0	0.87	1.9	1.54	1.6
Meat Sold	2.28	10.3	4.20	9.3	9.60	10.2
N-removal by non legum. crops	7.00	31.6	14.10	31.1	22.11	23.5
Total outputs	*22.17*	*100*	*45.31*	*100*	*94.04*	*100*
Nitrogen balance	*48.12*		*114.44*		*309.20*	

a) N-Fixation, net of removal.
Source: Own calculations.

12.6 Nitrogen balance and intensity of milk production

Table 12.5 sets out details of the nitrogen balances for different groups of farms classified in accordance with their productivity levels. It can be seen that the nitrogen surpluses show a progressive increase over the first three yield categories while flattening out in the last. Taking the average productivity figure of up to 4,000 kg per cow as a starting point, nitrogen surplus per hectare is 188 kg, while the maximum surplus of 292 kg of nitrogen per hectare is generated by the group of farms with yields between 5,000 and 6,000 kg per cows. The increase of nitrogen inputs in the form of feed concentrates and forage crops is not correlated to herd productivity. This is probably because the number of cows per hectare for the high production farms is less than for farms whose herds give a smaller yield. The total input of nitrogen per hectare, in the form of feed, is 160 kg for the less productive group, while the equivalent figure for the two central categories is 260-270 kg. The same figure for the farms with the highest unit productivity is 247 kg. The proportion of total inputs represented by feed is hence much the same between the various productivity categories, and is never less than 40% in any case.

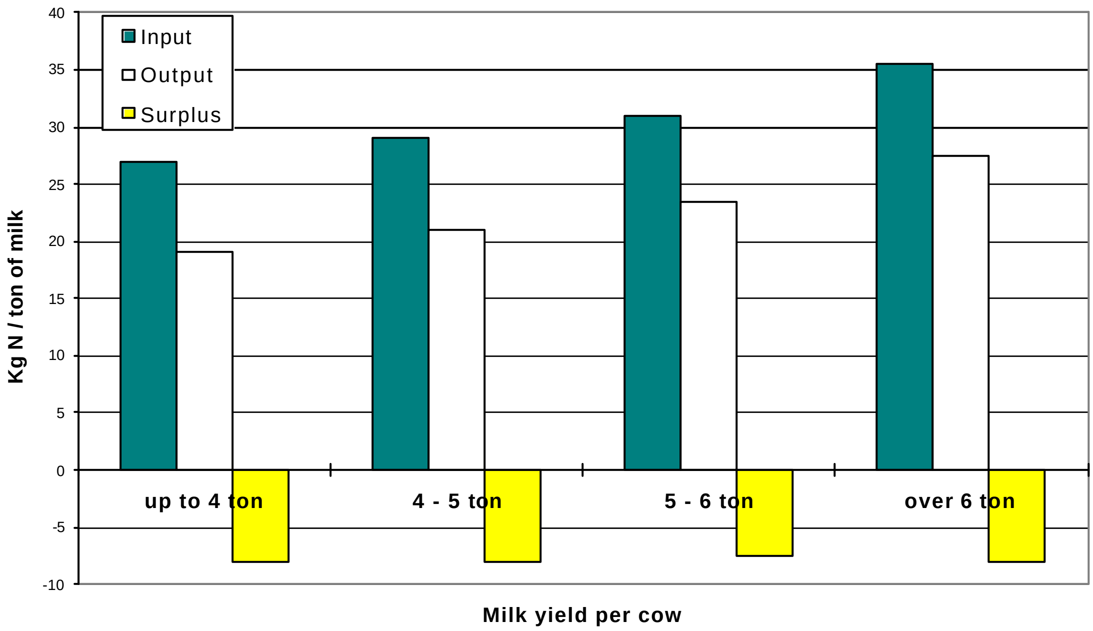

Figure 12.3 Nitrogen balance per ton of milk according to milk intensity levels per cow - Plain -

From the positive correlation between the average cow productivity and the size of the nitrogen surplus, the conclusion can be drawn that those farms with higher milk yields are more likely to experience environmental problems. The farms with higher unit yields are less efficient in their use of nitrogen.

Another indication of this relationship is obtained through the ratio of nitrogen surplus per ton of milk produced. Figure 12.3 shows that the inputs and outputs per ton of milk increase with increased unit yields, but at the balance, no significant differences between the milk yield groups are to be noticed.

In the next section, the impact of the milk yield on the nitrogen surplus, independently of other factors, will be assessed.

Table 12.5 Inputs and outputs of nitrogen by classes of milk production per cow - plain

	Milk production per cow							
	up to 4,000 kg 23 farms		4,000-5,000 kg 13 farms		5,000 - 6,000 kg 7 farms		over 6,000 kg 21 farms	
	kg/ha	%	kg/ha	%	kg/ha	%	kg/ha	%
Purchased feed	117.44	43.1	151.26	41.3	164.73	44.4	148.86	40.0
Purchased roughage	51.88	19.1	117.49	32.1	96.07	25.9	96.90	26.1
Purchase of manure	18.17	6.7	7.98	2.2	18.11	4.9	27.20	7.3
Chemical fertiliser	36.70	13.5	42.58	11.6	40.41	10.9	53.44	14.4
Atmospheric deposition	18.30	6.7	16.89	4.6	15.69	4.2	18.30	4.9
Purchase young beef stock	0.63	0.2	0.87	0.2	1.30	0.4	0.49	0.1
Purchase of milk powder	1.15	0.4	0.88	0.2	1.96	0.5	1.34	0.4
Litter	5.88	2.2	6.24	1.7	9.11	2.5	6.01	1.6
Leguminous N-fixing	22.07	8.1	22.31	6.1	23.52	6.3	19.35	5.2
Total inputs	*272.22*	*100.0*	*366.50*	*100.0*	*370.90*	*100.0*	*371.89*	*100.0*
Sales of organic manure	11.18	13.3	1.89	1.9	0.47	0.6	8.89	11.0
Milk sold	48.77	57.9	71.31	72.0	54.70	69.4	53.20	66.0
Cows sold	6.63	7.9	8.39	8.5	7.65	9.7	7.36	9.1
Calves sold	1.05	1.2	1.63	1.6	1.27	1.6	1.21	1.5
Meat Sold	7.68	9.1	10.01	10.1	8.92	11.3	8.57	10.6
N-removal by crops	16.65	19.8	15.81	16.0	14.77	18.7	9.97	12.4
Total outputs	*84.27*	*100.0*	*99.03*	*100.0*	*78.85*	*100.0*	*80.63*	*100.0*
Nitrogen balance	*187.95*		*267.47*		*292.05*		*291.26*	

Source: Own calculation.

12.7 Interrelations between technical efficiency, farm structure and nitrogen balance

A principal component analysis was carried out on indicators chosen as being the most representative of all those available. This was done in order to highlight relations between nitrogen surplus generation and the structure, technical characteristics and income generating capacity of the farms concerned. In this analysis, it was necessary to neutralise variations due to the altitude of the farms and the end use for the milk which have already be analysed in the previous paragraphs. Consequently, the analysis was carried out using only data from 55 farms based on the plain whose milk goes to Parmigiano Reggiano production.

Of the most representative variables, 13 were chosen. It was then confirmed that the data were suitable for use in the analysis (the 0-hypothesis of even distribution could not be rejected). A factor analysis was then carried out based on a methodology based on the extraction of the principal components.

Four factors were obtained in this way, which together explain 82% of the variance in the variables. All values less than the threshold value of 0.5 were deleted to provide an easier reading of the results obtained in the rotated factorial weight matrix (using Varimax rotation criteria).

The weight of the respective factors, as set out in table 12.7, gives a clear picture of the relative significance of the four factors obtained.

Table 12.7 Matrix of rotated factors-Plain

Description of variables	Variable	Factor 1 STRUTT	Factor 2 INTLAT	Factor 3 SPECIAL	Factor 4 INTFOR
Utilised Agricultural Area (UAA)	SAUC	0,84564			
Number of cows	VACTR	0,88065			
Hours worked per cow	ORESVAC	-0,86396			
Cows per Working Unit	VACULS	0,84011			
Average yields per cow	RESUTR		0,85493		
kg of milk per kg concentrate	LATMANC		-0,84956		
Cost of concentrates per cow	CMANVAC		0,95802		
Forage % of UAA	INCFOR			0,90935	
Alfalfa % of UAA	INCMED			0,91674	
Gross value of beef cattle in %	SPECPLV			0,71548	
Kg. of Milk per forage Hectare	LATFOR				0,80406
Cows per forage hectare	VACFOR				0,90594
Cost of forage per cow	CFORVAC				0,69881
% Variation explained		26,9	24,3	18,1	12,8

Source: Own calculations.

The first factor contains all the main size variables introduced at the beginning. The variation explained by this factor is 27.9% of the total. The main correlated variables are: average herd size (VACTR), labour productivity (ORESVAC, VACLS), and farm size (SAUC). For this reason, the factor can be defined as *Size* (STRUTT).

The second factor, accounting for 24.3% of the variation, is an amalgamation of all indicators used to measure the production intensity of the farm. The following variables are of particular relevance in this regard: average productivity of the cows (RESUTR), the production of milk per unit of feed purchased (LATMANC), and another variable closely linked to herd productivity the cost of feed per cow (CMANVAC). This second factor has thus been called *Milk production Intensity* (INTLAT).

The third factor encompasses all those characteristics, which could be said to be typical of specialised dairy farm production. It accounts for 18.1% of the total variation. Within this factor the following variables are of particular importance: the proportion of cultivated land devoted to forage crops (INCFOR) and alfalfa (INCMED), together with the economic index of milk specialisation

(SPECLPV). It was thus decided that the most appropriate name for this factor was *Milk Specialisation* (SPECIAL).

The fourth factor accounts for 12.8% of the total variation. This includes all three indicators concerned with land use intensity: the ratio of cattle population numbers to land under forage crops (LATFOR), milk production per hectare of forage crops (latfor), and, finally, the cost per cow of forage not produced on farm. The name given to this factor was *Land Use Intensity* (INTFOR).

On the basis of the above calculations, the second stage of this analysis was to use the factors obtained as independent variables to calculate, through multiple regression, the relations with the dependent variable of surplus nitrogen per hectare. The multiple regression calculations were carried out using the Stepwise method.

The main result to come out of this analysis was that all four factors were included in the regression equation. They were ordered within the model in accordance with their degree of correlation, whether simple or partial, with the dependent variable. The resultant ranking of the factors was as follows: Land Use Intensity; Milk Production Intensity; Size, and lastly Milk Specialisation. It can be seen that the value of R^2 changes from 0.42 to 0.51 and 0.57 for the third factor. Its final value is 0.63 when all four factors are included in the model. This means that 63% of the variation in the surplus nitrogen per hectare is thus explained by the model.

The formulation of the surplus nitrogen per hectare of cultivated land leads to the following equation:

$$BILAN\_N = 239{,}6 + \underset{(2{,}830)^*}{\overset{(F3)}{0{,}245STRUTT}} + \underset{(3{,}356)^*}{\overset{(F2)}{0{,}291INTLAT}} + \underset{(2{,}815)^*}{\overset{(F4)}{0{,}244SPECIAL}} + \underset{(7{,}493)^*}{\overset{(F1)}{0{,}649INTFOR}}$$

$R^2 = 0{,}62503^*$

* T values of Student reliable to 99%

42% of the variation in the dependent variables in the model is thus to be accounted for by the fourth factor. Land use intensity represents the crucial variable when seeking to understand the problem of nitrogen within specialist dairy farms.

An increase in soil use intensity may arise from increased pressure by the farm to exploit its land resources. This brings with it increased problems linked to the production of nitrogen surpluses. These problems may become evident both from the growth in nitrogen inputs and in relation to the output levels.

Concerning the inputs, the increase in surpluses due to the increased intensity of the farming techniques may take the form of:

- a greater increase in forage purchased from outside the farm where requirements exceed internal supply;
- a greater input of chemical fertilisers where the farmer decides to force forage crop growth in order to increase self sufficiency in forage production.

Concerning outputs, increases due to greater intensity in farming techniques may be manifested in the reduction of available land per head of cattle for slurry spreading, leading to problems of effluent disposal.

Of greater interest is the influence of the other three factors in the model. This is because they are included in the model, independently of the land use intensity factor. In the first place, those farms putting the greatest emphasis on herd productivity (the INTLAT factor) generate greater nitrogen losses than less intensive farms. While this relationship has already been identified in the findings of the bivariate analysis carried out in the previous paragraph, here it has been 'purified' of the effects exerted by the other factors. As the cow milk yield rises, the uncontrollable nitrogen loss factor increases and hence the efficiency of nitrogen use declines.

The large dairy farms too, are less efficient in their use of the mineral nitrogen. This may be attributed to reduced precision in the on farm utilisation of feed and fertilisers. It is reasonable to assume that as the size of the farm grows there is a corresponding reduction in the attention given to the use of resources.

Finally, the findings show that those farms with greater specialisation in dairy production generate greater nitrogen imbalances as compared with less specialised farms. This may be due to a greater efficiency in nitrogen use for crop production compared to livestock production. There are a large number of opportunities for wasting nitrogen in livestock farms. Examples, which can be cited, include the concentration of ammonia in the cowsheds and during manure spreading activities. Nitrogen losses are less in crop production.

12.8 Conclusions

Estimates of the total nitrogen surplus of farms are obtainable by means of accountancy and farm survey data. Accountancy data alone are insufficient to generate a complete overview of the nitrogen cycle on the farm, since quantitative data are often lacking. Farm surveys have to be carried out to integrate the accountancy datasets.

In this study, both data sources have been used to calculate nitrogen balances of dairy farms in Italy. Nitrogen pollution in Italy is particularly felt in the more intensive livestock producing areas of the country and cattle farms contribute significantly to the total nitrogen emissions of the Italian livestock sector. Over 60% of ammonia emissions of the livestock sector can be attributed to cattle farming.

The large variability in the farm nitrogen surplus can be considered an indicator of the possibility for many dairy farms to reduce their pressure on the environment. High intensity levels of milk production and land use are positively correlated with the nitrogen surplus. Nitrogen efficiency decreases with increasing intensity levels of the farm. Dairy farmers with high yielding herds require higher skills to control the nitrogen management of the farm. Mineral balance bookkeeping systems may induce intensive livestock farms to adjust their farm practices towards a closure of the farm nitrogen cycle and often these adjustments may go to the benefit of the economic balance of the farm.

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13. N losses in Swedish agriculture and examples from milk production

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Abstract

The target of reducing the nitrogen load to coastal waters by 50% in Sweden between 1985 and 1995 has not been reached. Due to the important environmental impacts of N losses from agriculture, the nitrogen problem is under continuous discussion and investigation. Reliable models are available for assessing N leaching from arable land in which climate, crop, production level, and fertilising strategy are considered. There are also good data available for assessing ammonia emissions from farmyard manure. One problem is, however, to ascertain all the variables, e.g. weather conditions, feeding intensity, technique for application and storing, which have crucial impacts on the size of the NH_3-N losses. An LCA study on milk shows that N losses are vital for the important effect categories: eutrophication, acidification, and global warming. It is therefore important to have correct data on emissions, and the nutrient balance seems to be a useful tool when assessing nutrient losses.

13.1 Introduction

Losses of nitrogen compounds to water and air are important environmental problems related to the agriculture sector in Sweden. Eight percent of the land area (i.e. 2.7 10^6 has) is arable land and the production is mainly for domestic use since Sweden is not a major food exporting country. Similar to all developed countries, there has been a sharp increase in fertiliser nitrogen use since the 1950s, culminating in 1985 and slowly declining to the present total use in Sweden of approximately 190 10^6 kg N, corresponding to 75 kg N per hectare. This is, however, an average figure and in southern Sweden where there is an intense production of grain, potatoes, and sugar beets, average fertiliser use per hectare is often considerably higher.

The political target for nitrogen discharges was to reduce the nitrogen load on the coastal waters by 50% between 1985 and 1995. This target was based on international agreements: the North Sea Conference in 1987, and for the Baltic Sea, Helcom in 1988. With the exception of the Baltic states, where the agricultural sector has become substantially smaller during the 1990s, none of the countries surrounding the Baltic Sea and the North Sea have succeeded in reaching the target of a 50% reduction of nitrogen losses to the sea (Naturvårdsverket 1997a). The effects of the Swedish measures to reduce nitrogen losses have been calculated by the Swedish EPA and National Board of Agriculture. The average nitrate-N leaching from arable land in Sweden is estimated to have decreased by approximately 25% (from 77 10^6 kg N to 55 10^6 kg N) between 1985 and 1994. This

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reduction is explained by an alteration of crops, less grain being grown due to different set aside programs, and improved N-efficiency (Johnsson and Hoffman 1996). The ammonia N emissions from agriculture have shown a small increase during the 1990s, from 43 10^6 kg N in 1990 to 46 10^6 kg N in 1995. The cause for this increase is explained by an increase in the production of pigs and a higher use of protein feed; an example of this is the substantially larger import of soymeal to Sweden since 1992 (Jordbruksverket 1997).

13.2 Nitrogen surplus

The nutrient balance is a useful tool when quantifying the flows of nutrients and emissions to air and water. This is a budget approach that can be used, since it is likely that most of today's agricultural soils will not accumulate further amounts of N under present cultivation practices. In table 13.1, a nutrient balance for the Swedish agriculture is shown. In the calculations used, all the inputs and outputs of nitrogen to the agriculture system are considered. Products that are circulating within the agricultural system are not included, which means that fodder such as silage, hay, grain, and manure, are not seen in the calculations.

Table 13.1 Input and output of nitrogen in Swedish agriculture in 1995 (Naturvårdsverket 1997b)

Input	10^6 kg		Output	10^6 kg
Fertilisers	188.3		Vegetable products	54.7
Imported feed	53.4		Animal products	42.8
Sewage sludge	4.3		N surplus	190.9
N-fixation	27.4		Total	288.4
N-deposition	15.0	a)	*N efficiency, %*	*34*
Total	288.4		*N surplus, kg per ha*	*69*

a) Ammonia N from domestic emissions is not included.

Bonde (1994) presents data on N surplus from ten North European countries where the Netherlands, Denmark, and Belgium show an N surplus of more than 150 kg N per hectare. Intensive animal production is characteristic for these countries. Such areas and farms can also be found in the south of Sweden, where a large share of the country's animal production takes place. Table 13.2 shows a nutrient balance for the municipal district of Halmstad, situated by the coast in Southwest Sweden. The arable land in this district, 23,038 ha, has an animal density which is 30% higher than Swedish average, hence the higher N surplus.

Table 13.2 Input and output of nitrogen in the agriculture system of the district of Halmstad (23 038 ha) in the south of Sweden (Cederberg 1997)

Input	1,000 kg		Output	1,000 kg
Fertilisers	1,810		Vegetable products	476
Imported feed	831		Animal products	514
Sewage sludge	13		N surplus	2,302
N fixation	348		Total	3,292
N deposition	230	a)		
Imp manure, seeds	60		*N efficiency, %*	*30*
Total	3,292		*N surplus, kg per ha*	*100*

a) All ammonia N included.

13.3 Data on nitrogen losses

Field trials for measuring nitrate leaching from agricultural soils started in the 1970's, and today there is fairly extensive data material for this nitrogen compound. Ammonia emissions have been measured in stables and during different field conditions. The variation of methods for manure handling is however significant, which seems to make it harder to correctly assess this nitrogen compound compared to nitrate N. Nitrous oxide discharges from arable soils have hitherto only been investigated in few field trials in Sweden.

13.3.1 Nitrate N

A model for estimating average N leaching from agriculture land was presented by Johnsson and Hoffman (1996) and it was devised to assess the Swedish goal to reduce the nitrogen load to coastal waters by 50%. It is based on a mathematical model SOIL/SOILN, which describes the dynamics and movements of N in agriculture soils. Johnsson and Hoffman have divided Sweden into nine regions of leaching which are characterised by climate, type of production, level of fertilising and production capacity (yields). For each region, the amounts of leaching have been estimating for typical situations for a combination of nine different crops, three soil types, and two fertilising strategies (with and without manure in the crop rotation).

Table 13.3 shows the average leaching from wheat and barley grown in the southernmost region of Sweden, the west and south coast of the province Skåne. This area is characterised by mild winters, normal precipitation, and an animal density higher than the Swedish average.

Data on farmyard manure is based on the region's animal density, type of manure, and yearly statistic interviews with farmers on application rates. It is should be noted that the leaching data do not concern manure application to the single crop but the use of manure in the entire crop rotation.

Table 13.3 Present average leaching, kg N per hectare for wheat and barley along the coasts of Skåne. (From Jonsson and Hoffman, table 1:2 and 1:4)

Crop	Farmyard manure in crop rotation	Sandy soil a)	Silt a)	Clay a)
Winter wheat	No	33	21	12
Winter wheat	Yes	74	59	43
Spring barley	No	46	34	26
Spring barley	Yes	76	62	50

a) Average humus content is 4.5%.

The difference between regions becomes obvious when comparing the same crops grown in the area south central Sweden, 'Mälardalen,' the plains north and south of Lake Mälaren, west of Stockholm. This region has colder winters, lower precipitation, and lower animal density than the south west of Sweden.

Table 13.4 Present average leaching, kg N per hectare for wheat and barley in the middle of Sweden (From Jonsson and Hoffman, table 1:2 and 1:4)

Crop	Farmyard manure in crop rotation	Sandy soil a)	Silt a)	Clay a)
Winter wheat	No	18	10	4
Winter wheat	Yes	36	22	10
Spring barley	No	30	21	14
Spring barley	Yes	52	40	28

a) Average humus content is 4.5%.

In LCA studies, environmental effects are related to the products and not to the hectare. Although the average nitrate leaching is substantially higher in southern Sweden, the potential eutrophication per kilogram wheat varies less between different wheat production regions due to varying yields. The wheat yields are approximately 15% higher in the province of Skåne compared with the plains in the south central Sweden (Välimaa and Stadig, 1998).

13.3.2 Ammonia N

The Swedish National Board of Agriculture has developed the computer program STANK *(Manure and plant nutrients in recycling)* (1996) for calculating nutrient flows and losses on single farm enterprises. The program can be used for calculating nutrient balances, production of manure, and ammonia emissions from manure, nutrient content in manure, and to make economical analyses for changing systems for storing and spreading manure. This is a very useful tool when estimating ammonia emissions, which are important when analysing the production of milk, meat and eggs.

Ammonia losses in stables and manure storing are estimated from animal type (e.g. dairy cow, cattle, heifers, sows etc.) and kind of manure (e.g. solid, urine, slurry, thick bed of straw). For N losses connected to manure storing, techniques like covering and filling in top or bottom of tank also are considered. During the application of manure, a great number of combinations can occur that imply different N losses. Table 13.5 shows the estimated standard N losses in STANK for a few of these combinations.

Table 13.5 Data on ammonia losses during application, % lost NH_3-N of NH_4-N in the manure (From STANK 2.1)

	Solid manure	Urine	Slurry	Thick bed straw
Early spring, broad spread	20	40	30	20
Spring, broad spread incorporated 1 h	15	8	10	15
Spring, broad spread incorporated 12 h	50	20	20	50
Spring, band spread, incorporated 1h	-	7	5	-

13.3.3 Nitrous oxide

There have been only a few Swedish investigations performed on N_2O-losses from arable land. In the Mellby leaching experiment field in southern Sweden, discharges of nitrous oxides and ammonia were measured after spreading liquid manure (Weslien et al., 1998). The emissions of N_2O-N varied between 0.17-0.30% of applied N (corrected for ammonia losses) for slurry application in the spring and 0.79-0.91% of applied N for slurry application in the autumn. One reason for the lower losses in the spring can be that the field trial and measuring took place during a very dry spring. Additional investigations, also examining mineral fertilisers, are now in progress but no results have yet been published.

Due to very little data material on Swedish conditions, the IPCC methodology for assessing direct N_2O-emissions from agricultural soils must be used for the time being, calculating that 1.25% of applied N (synthetic fertiliser, manure, N fixated by fixating crops) is lost as N_2O-N.

13.4 Nitrogen losses in milk production

Nutrient balances were used in an LCA study of conventional and organic milk production (Cederberg 1998). Data were collected from two dairy farms in the west of Sweden. The functional unit was 1,000 kg milk (ECM) leaving the farm gate. Both farms studied were specialised on milk production and the only output products were milk and meat from culled cows and bull calves. Livestock density (including one heifer per cow) was 1.43 dairy cow per hectare on the conventional farm and 0.69 dairy cow per hectare on the organic farm. Table 13.6 and b show the calculated nutrient balances on the farms.

Table 13.6a Nutrient balance on the conventional farm in the LCA milk study

Input, kg/ha	N	P	K	Output, kg/ha	N	P	K
Feed and seeds	134	19.8	46	Products	47	9.5	14
Fertilisers	86						
N fixation	15						
N deposition	10			*Nutrient surplus*	198	10.3	32
Total	245	19.8	46	Total	245	19.8	46

Table 13.6b Nutrient balance on the organic farm in the LCA milk study

Input, kg/ha	N	P	K	Output, kg/ha	N	P	K
Feed and seeds	29	5.2	9	Products	20	4.1	6
Fertilisers	0						
N fixation	46						
N deposition	10			*Nutrient surplus*	65	1.1	3
Total	85	5.2	9	Total	85	5.2	9

As seen, the fodder import is important for the nutrient input on both farms. The conventional farm buys grain from a neighbouring farm, protein feed (containing soymeal, rapeseed meal, maize gluten meal, beet pulp, sunflower meal, etc.) and super pressed beet pulp. The organic farm buys organic grain and peas from a neighbouring farm and only smaller amounts of concentrate feed. The rules for organic farming has a limit of a maximal use of 5% conventionally produced feed in the fodder ration in Sweden.

Research in Danmark shows an average N surplus of 240 kg N per hectare on 14 conventional dairy farms and 124 kg N per hectare on 16 organic dairy farms (Dalgaard et al., 1998). The Danish rules for organic farming allow a greater use of conventionally produced fodder than in Sweden and hence, livestock density can be higher on Danish organic dairy farms. Van der Werff et al. (1995) present Dutch data from three organic dairy farms with average surplus of 83 kg N per hectare compared to standard conventional milk production in the Netherlands with N surplus of 390 kg N per hectare. The differences in N surplus on the farms representing the two different production systems in the LCA milk study can also be found in other studies.

The methods described in section 3 for assessing N losses during Swedish conditions where used for calculating the N losses on the farms studied (see table 13.7). Nitrogen losses through denitrification are another important output of N from the soil system. Since N_2 is a natural component of the atmosphere, no environmental damage results from this loss. But denitrification can be one explanation of why a smaller part of the N surplus was found as emissions on the conventional farm

(48%). This farm has clay soils and larger mineral N flows and it is reasonable that the denitrification has been as high as 30-50 kg N per hectare. The organic farm in the study has lighter soils and low flows of mineral N in the soils, denitrification should not excess 10 kg N per hectare. However, even when N_2-losses of this magnitude are added to the total N emissions in table 13.7, there is still a share of N surplus that is not accounted for; 15-20% on the organic farm and 25-40% on the conventional farm. The fixing of N in the soil pool is yet another explanation for the unaccounted N in the balance.

Table 13.7 Data on N surplus and N losses in two production forms of milk in Sweden

	Conventional farm	Organic farm	
N surplus, kg N per ha	198	65	
Calculated N losses, kg N per ha			
NH_3-N	61	24	
NO_3-N	32	19	
N_2O-N	3.1	1.2	
Total estimated N losses		*96*	*44*
Share of nitrogen surplus found when calculating N losses	48%	67%	
Milk production, kg milk per ha	7,415	3,297	
a) N surplus, kg N per 1000 kg milk	22.8	16.8	

a) When allocating 85% to milk and 15% to meat.

Comparing the calculated N losses in an LCA study with a nutrient balance is an appropriate way to validate the data and emissions factor used in the calculations. Concerning N, which is found in a myriad of forms in soil, air, and water, it is very difficult to make correct estimations of losses. For example, of the total N surplus of 190 10^6 kg N in Swedish agriculture, 154 10^6 kg N is defined as losses of nitrate N and ammonia N, denitrification and soil input and this leaves 20% unaccounted for (Naturvårdsverket 1997b). The Swedish model used to calculate nitrate leaching has been tested against leaching data from experimental fields with continuous registration of discharges. The methods used for calculating ammonia losses appears to be less exact and there is a great variety in climate factors, manure handling etc., that grossly can influence the size of emissions in practice.

It seems that the N losses are underestimated in the conventional system and this especially concerns ammonia N. Dutch research shows a very strong connection between feeding intensity and ammonia losses (Smits et al. 1995). The use of high protein feed is much greater in conventional milk production than in organic production in Sweden. The Board of Agriculture also points out the increasing intensity in dairy cow feeding as one cause of increasing ammonia emissions in Sweden (Jordbruksverket 1997).

The LCA methodology is product oriented, focusing the output products and their environmental loading and not the land area were the production takes place. The nutrient balance is area based and when used for calculating the N surplus in an agricultural system, it gives an indicator of the potential environmental impact from the agricultural system studied. In table 13.8, the N losses per hectare have been converted to N losses per 1,000 kg milk. Because of the higher milk production per hectare in the conventional system studied, the difference between the two farms is much smaller when the N losses per hectare are converted to N losses per functional unit (1,000 kg milk).

Table 13.8 Calculated N losses on the farms in the LCA study converted to kg N per 1,000 kg milk

	Conventional	Organic
Ammonia as NH_3-N	6.97	6.13
Nitrate as NO_3-N	3.62	4.85
Nitrous oxide as N_2O-N	0.36	0.30
Total	10.95	11.28
N surplus according to nutrient balances a)	22.8	16.8
Share of N surplus found	48%	67%

a) 85% of total N surplus and losses is allocated to milk.

The nutrient balance shows the flow of nutrients through the entire farm unit. However, it does not provide the entire lifecycle perspective of the product milk as is done with the LCA methodology. Since the dairy farms studied as most dairy farmsimport feed, there has been nitrate losses in the cultivation of the crops yielding this feed, resulting in N emissions outside the farm border. There are also N_2O emissions of importance outside the farm border due to N fertiliser production.

When interpreting the LCA study on milk production, the importance of N losses to major environmental impact categories is obvious. Emissions of N_2O connected to the nitrogen cycle on the farms studied (losses from soils) and N_2O-emissions from synthetic fertiliser production play a larger role in terms of the potential contribution to global warming than CO_2-emissions from the use of fossils. Due to higher fertiliser rates, the conventional system here shows larger negative effects.

Approximately 90% of the maximal potential contribution to acidification from milk production could be derived from ammonia emissions in both system studied; the conventional system showed a higher potential impact due to higher NH_3-N emissions. The maximal potential contribution to eutrophication was slightly higher for the organic alternative due to higher nitrate losses per ton feed. However, this impact category was also greatly influenced by discharges of ammonia N and correct data for this compound seem to be crucial in environmental analysis of animal products.

13.5 Conclusion

We believe that there are good data available for making fairly correct estimations of *nitrate leaching* from arable land in Sweden. Most of the present data material concern conventional agriculture where both synthetic fertiliser and organic manure are used, but as organic farming is increasing, field trials on nitrate leaching from this farming system are now underway.

The computer program STANK offers a good tool for calculating *ammonia losses* on individual farms. The program does not consider different feeding intensity with protein, which is a deficiency. Smits et al. (1995) show urea content of urine from cows fed with low protein diet to be 42% lower than that for cows fed on high protein diet which resulted in a reduction of ammonia emissions by 39%. Variations of this magnitude for this important nitrogen emission will have a big impact on the acidification as well as the eutrophication potential from animal products. Therefore, it seems to be of vital importance to obtain data on the connection between protein feeding and ammonia emissions.

For emissions of *nitrous oxide*, there are very little data from Swedish field trials. The few data so far published seem to show lower N_2O-emissions than the IPCC guidelines for assessing this greenhouse gas.

Comparing the calculated N losses in an LCA study with a nutrient balance is an appropriate way to validate the data and emissions factors used in the calculations. However, after calculating N losses, there is often an unexplained part of N surplus and how to deal with this unaccounted part is not a straightforward question to answer. When the models used for calculating N losses in a Swedish LCA milk study are validated, the estimations of ammonia N losses seem to be the most uncertain and possibly underestimated.

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14. Conclusions of the working group on the nitrogen cycle

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14.1 Nitrogen modelling

Ideally, a holistic model would be available, allowing you to model all nitrogen flows at the same time for a given situation, even taking into account the dependency between those N fluxes. However, since no such model yet exists, each flux must be considered in isolation. The following approaches have been extracted from the discussion of the working group:

N fixation

N fixation is a relevant issue, especially when looking at organic farming systems. It was concluded that for the time being, a simple Danish empirical model (Høgh-Jensen et al., 1998) presented by Loges et al. (this volume) might be applied. Input data for this model are harvested drymatter yield of leguminosae and information on the production system (partitioning leguminosae/grass, ...).

Ammonia

The British MARRACAS-model (Cowell and ApSimon 1998) seems to be best practicable means for making a first approximation of ammonia emissions for LCA purposes. By means of this model, NH_3 emissions of different types of animals can be modelled. Basis of the model is the nitrogen excreted by animals (literature values). The model is currently in use in the UK (validation at level of UK, no onfarm validation yet) and is state-of-the art in Europe. This model covers both organic fertiliser and mineral fertiliser (relevant here is the urea and ammonium content). Input data might be taken from FADN and study farms.

The following key parameters are taken into account in the MARACCAS-model:

- temperature (housing of animals and storage of manure);
- application method (surface spreading versus injection);
- type of manure applied (N excreted, i.e. N in manure);
- land coverage.

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Parameters which are not dealt with in the model, but which are regarded as being important are:

- wind speed;
- radiation;
- infiltration rate (soil characteristics, season);
- precipitation;
- time between application precipitation, or incorporation into the soil;
- soil chemistry (pH, cation exchange capacity, buffer capacity).

A rough attempt was made to assess what effect those parameters might have upon the outcome of the ammoniamodel. Efforts for improving ammonia modelling should focus those factors that have the largest influence on the outcome of the model. However, the conclusion was that many factors seem equally important. Thus, when improving ammonia modelling, all the above factors should be incorporated in a more sophisticated approach. In addition, an attempt should be made to include possible interrelations between those parameters. One of the problems that arise when attempting to model real ammonia emissions is that there are few real time data available (e.g. wind speed, temperature, rainfall, etc.). However, not all of these data are relevant for LCA purposes, as LCAs most often relate to larger geographical and temporal averages.

Nitrate

The British SLIMMER-model (Anthony et al., 1996) was considered to be best practicable means for LCA purposes. This model assumes a non-linear relationship between the remaining nitrogen on the plot (N surplus) and the amount of nitrogen that leaches. Basis for the calculation and key issue here is the amount of nitrogen in the soil in autumn. This model is applied in the UK. It can be used both for arable farming and grassland.

It seems not to be fit for use in regions where irrigation is applied (Southern Europe) and in areas with heavy rainfall (effects of by pass flows, etc.). Key parameters in nitrate modelling are soil and climate related parameters (e.g. field capacity, drainage water rate). There is a need for a harmonisation of calculation procedures at EU level, especially with the nitrate Directive in mind.

Nitrous oxide (N_20)

The IPCC-calculation procedure (Bouwman formula, IPCC 1997) should be applied for LCA applications since this is a broadly accepted approach in the area of greenhouse gas emissions. However, the Bouwman formula (Bouwman, 1996) simplifies the complex dependencies of the various important soil, climate, and management related parameters very much. Unfortunately, to date it is not possible to take account of all key parameters influencing the N_2O emissions caused by agriculture.

The IPCC approach is regularly updated. Thus, the most recent version should always be used for calculating nitrous oxide emissions.

Nitrogen gas (N_2)

No good models available. There seems to be a ratio between N_2/N_2O which is influenced by the soil condition (e.g. on sandy soils almost no N_2, on clay soils more N_2). It is often assumed that the deficit in N balance is lost as N_2. However, this approach is likely to over estimate the N_2 emission. It was suggested instead to distribute the deficit over all the above sources in the proportion to the amounts resulting from the above calculation procedures. Since N_2 is not calculated, it will then participate in this distribution with the same proportion as the deficit itself.

Not covered

Not covered in the modelling approach is NO_x since this N flux is considered to be small compared to the other N fluxes. Few data are available on this matter, and there is a large uncertainty in this area.

Also not covered are nitrogen losses by soil and wind erosion. This however seems to be only relevant in cases where mineral fertilisers are applied.

14.2 Needs for future research

The members of the work group identified two important needs concerning nitrogen modelling.

a) Need for model improvement and validation

Many of the models that are available at this moment seem to be developed for use under specific circumstances. In addition, many of these have not been validated thoroughly what makes their use and application within the framework of LCA difficult. Models should be applicable in different regions and for different farm types. Furthermore, the development of a more holistic model that covers all N fluxes at the same time would be desirable.

It was suggested to prepare a note with recommendations towards the EU 5th framework programme, for a cluster of N projects, including issues on quality assessment and control of models.

b) Data availability

Data availability is a crucial issue when models and calculation procedures have to be applied. Data from the FADN network can be combined with other data for use in LCA and for performing sensitivity analyses. The results from these LCA's can be used as input for improving data gathering within the FADN framework.

c) Experience

Experience has to be gained with the approach that was agreed upon above. It was suggested to make a case study, e.g. on a mixed livestock farm.

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