# Agricultural data for Life Cycle Assessments

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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 the 25 and 26 January 1999 at the Agricultural Economics Research Institute in The Hague. The papers 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.

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 reviewed for acceptance by two or more anonymous reviewers. The first volume of the book consists of chapter A, B, C, (topics), while the second volume of the book deals with chapter D, E (topics).

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## D. Data on other substance cycles

# 15. The role of the soil in phosphorus cycling

*W.J. Chardon* [1]

*Abstract*

On a world scale, a net transport of P to the oceans occurs, maintained by weathering of minerals and by erosion. The soil plays a different role regarding phosphorus in different agricultural systems. Without adding fertilisers, a soil can supply P for a limited time, by weathering or by mineralisation of indigenous organic matter. When a higher productivity is aimed at, fertilisers have to be added. In general, more P has to be added than is taken off by the crop to compensate for P becoming less available for plant uptake. In regions with a surplus of animal manure, P contents of the soil can become very high, creating problems due to eutrophication of surface waters.

## 15.1 Introduction

Phosphorus (P) makes up about 0.12% of the earth's crust. It is present in all soils and rocks, in surface waters and sediments, and in remains from plants and animals. The world's supply of P comes from mineral deposits, a non-renewable natural resource (Cathcart, 1980). A net transport to the oceans occurs (Tiessen, 1995): the use of mineral P fertilisers in 1990 was estimated as 16 Tg per year. Estimates of yearly P transport to the oceans vary between 21 and 39 Tg year with 23 Tg per year as the best estimate (Howarth et al., 1995). Thus, on a world-wide basis, the P cycle is not closed: the net P transport to the oceans is compensated by weathering of P containing minerals and erosion. However, as will be discussed in section 5, this situation differs strongly between agricultural regions: in many countries, a net input of P to agricultural soils takes place. In surface waters, enrichment of P can lead to eutrophication. The symptoms of eutrophication can differ between aquatic systems, and include turbidity, fish mortality, reduction of aquatic macrophytes, and growth of less desirable (toxic) algal species. In most inland waters (rivers and lakes), P is the limiting nutrient for algal growth, which makes reduction of P flow to surface waters important.

## 15.2 Input of P to soils

Input of P to soils can occur by atmospheric deposition (pollen or dust) and, in agricultural systems, by application of mineral or organic fertilisers.

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For Sweden, the amount of P deposition was estimated as 0.07 kg P per ha*year (Sepa, 1993). The P content of dust, mainly originating from wind erosion, will depend on the P content of the soil surface layer in the region where erosion took place.

Several kinds of inorganic fertilisers are used in agriculture. In developing countries, often rock phosphates are used, directly after mining or after acidulation, which releases P otherwise strongly bound to calcium. If applied to soils, rock phosphates will dissolve slowly, and are thus not directly available to crops. In developed countries, the most used fertiliser is superphosphate, which dissolves readily and is thus directly available to crops.

- When organic fertilisers (e.g. compost, animal manure, and sewage sludge) are applied, mineralisation has to take place before plants can take up P. This will proceed soon after application under wet conditions, with a relatively high ambient temperature, and more slowly at a low temperature or when the organic fertiliser dries out. When the fertiliser is mixed with the soil during or shortly after application, mineralisation will proceed more quickly.

## 15.3 Soil processes

After application of inorganic P fertilisers to a soil, or when P has been released from organic fertilisers by mineralisation, several reactions with the soil can take place:

- adsorption: fast reaction of P with the outside of soil particles;
- absorption: slow migration of P into the pores of soil aggregates;
- immobilisation: P is incorporated into soil organic matter; this is especially important on grassland soils, where a strong accumulation of organic P can be found;
- precipitation: binding of P with other chemical elements, e.g. calcium; this only occurs in soils with a high pH.

Adsorption of P mainly occurs onto hydroxides of iron or aluminium. The amount of these compounds in soils is limited. When the capacity is nearly used up, both the availability of P for uptake by plants and the possibility of P loss to the environment increases. In general, more P has to be added than is taken off by the crop to compensate for P becoming less available for plant uptake. Absorption (migration of P into soil aggregates) is considered to be the main process responsible for what can be called 'inevitable loss of P'.

## 15.4 Loss of P from soil

As will further be discussed in the paper of Heathwaite (this volume), loss of P from soils can proceed via different pathways:

- transport through the soil matrix, implying that P moves slowly through the soil with the rainfall surplus. After that, P can be bound in deeper soil layers or it can be transported with the percolating water to e.g. drainage ditches;

- transport over the soil surface with water from rainfall or snow melt, when the infiltration capacity of the soil is exceeded ('surface runoff') and the soil has a certain slope. This is a fast process, and the P content of the receiving surface water is raised shortly after rainfall occurs;
- transport via highly permeable parts of the soil ('preferential water flow'). This is also a fast process, leading to a quick rise of P in surface water at the start of rainfall. It may occur when:
  - clay soil contains cracks;
  - organic soils are dry;
  - the soil is artificially drained;
  - parts of the soil are water repellent; or
  - the soil contains permanent wormholes.

The transport routes mentioned above differ from an environmental point of view. When water transport occurs via the soil matrix, soil P content near the water table determines the environmental risk. In the case of surface runoff or preferential flow, the P content of the soil's upper layer is more important, implying that after excessive P application to the soil surface environmental problems by P loss can be expected in an early stage.

## 15.5 Balance of P

The difference between the amount of P applied to a field or on a farm via fertilisers, and the amount of P exported via crops, milk or meat, can be called the P balance (see also the paper of Withers in this volume). In developing countries, when farmers cannot afford the use of fertilisers, the balance can be negative: P in the soil becomes depleted, and yields are generally low. For sub-Saharan Africa, an average P loss of 2.5 kg P per ha*year was calculated (Stoorvogel et al., 1993). The P taken up by crops originates from weathering or mineralisation of indigenous organic matter. Often, after several years of exploitation, the soil will be left and new land will be cultivated.

When the yield pursued is higher, fertilisers have to be applied in order to build a reserve of P in the soil, and the P balance will be positive. This is the case in most developed countries. The situation changes dramatically when a farm (or region) has a limited soil surface area, a relatively large production of P containing manure exceeding crop offtake, and no possibilities to export the manure from the farm or region. The balance can become strongly positive through the purchase of P via animal feed. In this case, the aim of P application to the soil is not maintenance of soil fertility, but the disposal of the manure produced. This will lead to a fast build up of a P reserve in the soil, far exceeding crop needs, and creating a risk for the environment. For different European countries, the national P surplus was calculated; results for 1992 are given in table 15.1. It has to be kept in mind that also in countries with a low or moderate P surplus, regions may exist where confined animal production is concentrated, and a much larger surplus can be found. Regional differences are illustrated for the United Kingdom, France, and Spain.

It will be clear that P surpluses as shown for e.g. Belgium and the Netherlands will create, or already have created, environmental problems due to P loss. A positive aspect is that the use of chemical P fertilisers and P surpluses tend to decrease, as is shown in table 15.2.

Although the calculated surpluses of P have decreased in most countries, with a positive result of the P balance, the P reserves in the soils still increase.

*Table 15.1 Surplus of total-P (kg P per ha*year) for the average farm of some European countries and regions in 1990/1991*

| | Country | Region | | Country | Region |
|---|---|---|---|---|---|
| United Kingdom | 6 | | Ireland | 15 | |
| England West | | 8 | Switzerland | 16 | |
| Scotland | | 3 | Germany | 21 | |
| Denmark | 8 | | France | 28 | |
| Spain | 12 | | Bretagne | | 37 |
| Galicia | | 20 | Limousin | | 13 |
| Extremadura | | 10 | Belgium | 36 | |
| Greece | 15 | | Netherlands | 40 | |

Source: Data from Brouwer et al. (1995).

*Table 15.2 Reduction (%) in application rates of chemical P fertilisers and P surplus for some European countries*

| Country | Period, 1985- | Fertiliser | Period, 1985- | Surplus |
|---|---|---|---|---|
| Belgium | 1992 | 24 | 1992 | 5 |
| Denmark | 1992 | 29 | 1990 | 0 |
| France | 1992 | 19 | 1990 | 0 |
| Germany | 1993 | 58 | 1992 | 67 |
| Netherlands | 1992 | 11 | 1992 | 19 |
| Norway | 1992 | 18 | 1992 | 42 |
| Sweden | 1992 | 50 | 1990 | 20 |
| Switzerland | 1990 | 7 | 1990 | 20 |
| United Kingdom | 1993 | 17 | 1993 | 17 |

Source: Data from De Walle and Sevenster (1998).

In summary, depletion of soil P occurs in low productive agricultural systems in developing countries, with formation of deserts as a serious risk. A large build up of soil P occurs in regions where animal production is concentrated, and animal manure is produced in amounts which favour the use of soil as a place to dispose manure instead of use it as a fertiliser. This build up of soil leads to an increased risk of P transport to surface water, on a short term in case of surface runoff or preferential flow, and on a longer term in case of matrix flow.

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Tiessen, H., 'Introduction and synthesis', pp. 1-6, In: H. Tiessen (ed.) *Phosphorus in the global environment*. Transfers, cycles and management. Wiley, 1995.

Walle, F.B. de, J. Sevenster, *Agriculture and the environment*. Minerals, manure and measures. Kluwer, Dordrecht. 211 p., 1998.


## *Further reading*

Information on world P reserves, fertilizer manufacturing and use, reactions of P in soil, and plant nutrition can be found in:


Khasawneh, F.E., E.C. Sample and E.J. Kamprath, *The role of phosphorus in agriculture*. ASA, CSSA, SSSA, Madison USA, 910 p., 1980.


General reviews on P cycling can be found in:

H. Tiessen (ed.), *Phosphorus in the global environment*. Transfers, cycles and management. Wiley, 462 p., 1995.


Recent reviews on the relation between agriculture, soil P and eutrophication are given in:

Tunney, H., O.T. Carton, P.C. Brookes and A.E. Johnston (eds), *Phosphorus loss from soil to water*. CAB International 1997, 467 p., 1997.

# 16. Cycling and sources of phosphorus in agricultural systems and to the wider environment: a UK perspective

*P. J. A. Withers* [1]

*Abstract*

Inputs of phosphorus (P) in fertilisers and feeds often exceed P exports in harvested produce on intensively managed agricultural holdings, especially those operating with high livestock densities. The build-up of surplus P in the soil, together with frequent spreading of relatively large amounts of recycled excretal P on farms, are of environmental concern with respect to the transfer of P in land run-off to water causing eutrophication.

The frequency of P application, the amount of the P surplus and the soil depth over which surplus P is distributed varies considerably between different regions and farming systems with implications for P transfer. Trends towards continuous cultivation, slurry based livestock systems and the installation of tile under-drainage in arable and grassland systems are also considered to have increased the ease with which soil-accumulated and freshly-applied P are lost to surface waters. Accelerated P losses are not derived equally over the catchment area, and may originate only from fields with inherently high P loss risk, or which are mismanaged.

Climate, landscape, soil type, farming system and farm management data are required to define the transfer of soluble P and P associated with eroding soil particles to watercourses, but these show large regional variation due to the wide distribution of soil parent materials, climate and topography affecting natural P loss, and diverse patterns of farming systems with regard to land use, P inputs and land management. Expert systems are required to compare the relative importance of regional differences in site and agricultural management factors in order to quantify the P emissions associated with regionally produced agricultural products. In some areas, some form of control over agricultural P inputs, and/or the transport of P within the landscape, is, or will be, required in future to help maintain water quality for a range of uses.

## 16.1 Introduction

Agricultural crops require adequate amounts of phosphorus (P) for healthy growth and to maximise the utilisation of other nutrients, especially nitrogen (N). Similarly, livestock require adequate amounts of P in their diet to prevent against deficiencies, which might impair their health and performance. As a consequence, P fertilisers and minerals have been routinely and liberally imported onto farms in response to economic and political pressures to maximise agricultural production. Within the developed countries, farming systems have generally become more intensive, with a greater proportion of land

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under cultivation, with underdrainage and/or with increased animal densities. For example, agricultural census returns indicate that the numbers of animals kept on specialist livestock holdings in the UK typically increased 3-fold between 1965 and 1993.

Increased P imports on intensively farmed holdings, particularly livestock holdings, have led to a greater reliance on readily-available inorganic fertiliser and feed P products, larger amounts of faecal P requiring disposal onto land and an accumulation of surplus P in the soil, of which a greater proportion is in readily-exchangeable form (Isermann, 1990; Brouwer et al., 1995; Tunney et al., 1998). These trends have raised a number of environmental concerns; the mining of exhaustible rock phosphate reserves in developing countries and the air pollution associated with manufacture of inorganic P products; the accumulation of potentially harmful metals in soils from the repeated application of rock phosphate and its products, particularly cadmium (Cd), and the accelerated loss of freshly-applied and soil-accumulated P from agricultural land to water causing eutrophication (Withers and Sharpley, 1995). The relative importance of these concerns varies between different countries, but perhaps, the most widespread and increasing problem is that associated with eutrophication.

The extent of eutrophication problems in freshwaters is most commonly related to P inputs and only very low concentrations of P are required for eutrophication symptoms to appear (Gibson, 1997). The role of agriculture in the eutrophication process has rarely been clearly defined, largely because anthropogenic sources are usually the major source of P loads, and P losses in land run off are difficult to quantify due to their diffuse nature. They emanate from a number of source areas within the landscape, and their amount, form and timing are very variable as a result of short-term and often unpredictable changes in hydrological conditions and farming practices; rotational cropping, the application of fertilisers and manures, or the movement of animals from one field to another (Lennox et al., 1997). Recent monitoring of rural catchments suggests that the loads and concentrations of P in land run-off are sufficient to cause eutrophication and that they have increased under intensification (Foy and Withers, 1995; Heathwaite et al., 1996). This paper reviews the cycling of P within agricultural systems and the associated risks to environmental life cycles.

## 16.2 Phosphorus cycling within agriculture

### 16.2.1 Fertilisers and feeds

Unlike N, P is a conservative element whose inorganic forms become strongly bound to soil colloids. The degree of binding depends on the nature of the adsorbing surface and the ionic composition of the soil solution, but essentially P is relatively immobile in soil. Field experiments have consistently demonstrated that the proportions of fertiliser and manure P utilised by crops in the year of application is low (< 20%). Since crop P requirement is largely derived from the soil, it is the ease of exchangeability of soil P, as assessed by standard soil extraction tests, which forms the basis of fertiliser P recommendations world wide. Although the methods of soil analysis, and allowances for crop and soil type factors, differ between different countries (Tunney et al., 1997), it is widely recognised that there is usually no economic advantage to fresh fertiliser P inputs, once readily-exchangeable P reserves in the soil reach a certain level; C1 in figure 16.1. Above this critical level, P inputs need only to match crop P offtake, except perhaps for some soils where P becomes progressively unavailable to crops due to fixation processes (Withers et al., 1994; Bertilsson and Forsberg, 1997). On soils of adequate P fertility, the availability of P in different fertilisers and manures is therefore not significant in terms of crop production, and it is the total P input that must be regulated to prevent the excessive soil P accumulation which leads to accelerated P transfer to water (figure 16.1).

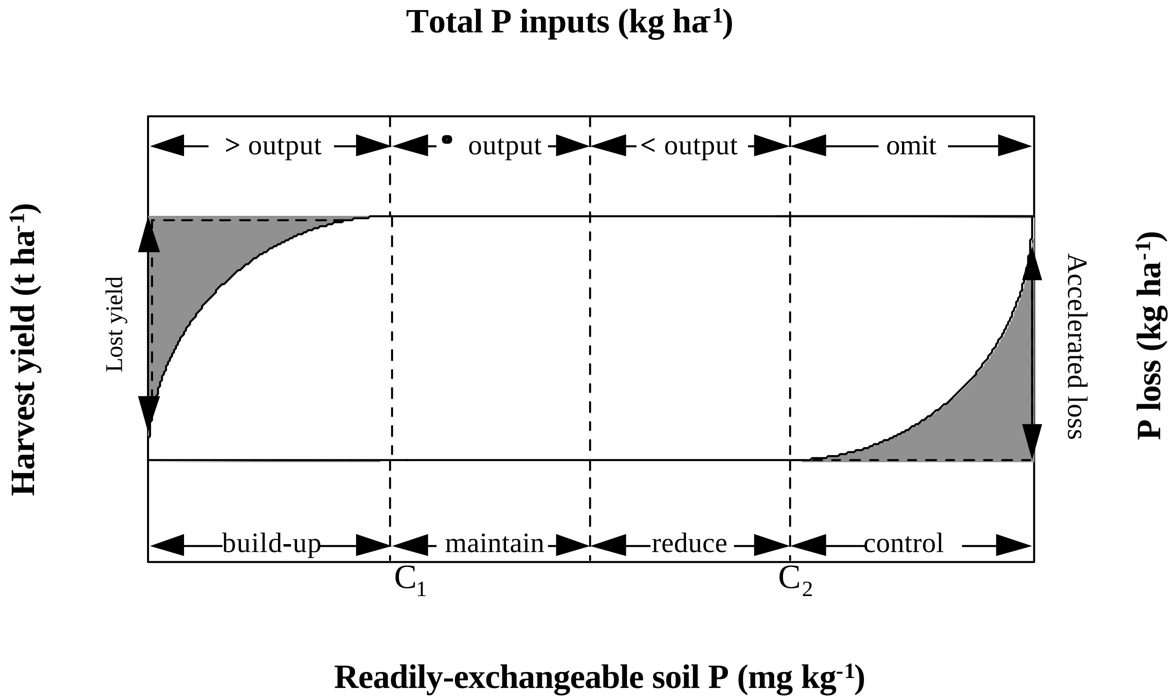


*Figure 16.1 Conceptual diagram of critical concentrations of readily-exchangeable P for optimum crop yield ($C_1$) and accelerated P loss to water ($C_2$) in relation to P input strategies*

Within animal production, the apparent efficiency of utilisation of feed P inputs is also low, with typically 70-80% of fresh P intake excreted in dung and urine (Lynch and Caffrey, 1997). Much of the dietary P intake of livestock on intensive holdings originates from imported concentrates, which are generally formulated to insure against deficiencies of P in other constituents of the diet and to overcome genetic variability in P absorption between animals. Feed compounders have, therefore, been less concerned with matching the actual P requirements of animal production, but have tended to include inorganic P supplements in generous quantities to avoid the possibility of reduced animal health and fertility. In a recent systems study in the UK, a 40% reduction in imported P fed to dairy cows yielding >6,000 l per year did not appear to impair milk production compared to a conventional dairy herd receiving commercially formulated inputs (table 16.1). Recent reviews indicate there may be scope to reduce P inputs, and/or improve P utilisation, through better dietary manipulation, since it is now well established that P fed in excess of dietary requirements is simply excreted without improving production (Valk et al., 1998; Damgaard-Poulson, 1998). Consequently, farmers have little or no idea of the amount of P ingested by their animals, or the extent to which it is being oversupplied. This is in direct contrast to the planning of fertiliser inputs for crop production. Unlike cropping systems on fertile soils, the P availability of individual feed ingredients is important to utilisation, especially for non-ruminants, which cannot utilise phytate-P effectively.

### 16.2.2 Surplus phosphorus

The relative proportions of feed and fertiliser imports compared to P output in farm produce determines the amount of surplus at the scale measured, and the degree of soil P accumulation. Within the UK, P imports exceed P exports by about 10 kg P per hectare, averaged over the total agricultural land area (table 16.1).

*Table 16.1 Inputs and outputs of phosphorus (kg per ha) in the UK and in different UK farming systems*

| | UK (1993) a) | Dairy systems b) | | Arable c) (1985-98) | Upland d) (hill sheep) |
|---|---|---|---|---|---|
| | | high output | reduced loss | | |
| Livestock density (LU $ha^{-1}$) | - | 1.9 | 1.6 | - | |
| Inputs | | | | | |
| Atmosphere | 0.3 | 0.5 | 0.5 | 0.5 | 0.1 |
| Imported feeds/minerals | 2.6 | 23.3 | 13.4 | - | 0.2 |
| Imported fertilisers | 9.3 | 13.6 | 0.8 | 24.8 | 0.5 |
| Sewage sludge | 0.4 | - | - | - | - |
| Sub-total | 12.6 | 37.4 | 14.7 | 25.3 | 0.8 |
| Outputs | | | | | |
| Milk and eggs | 0.7 | 11.6 | 10.2 | - | - |
| Meat and wool | 1.7 | 0.8 | 0.7 | - | 0.1 |
| Grain and straw | 0.6 | - | - | 16.7 | 0.1 |
| Misc e) | - | 1.7 | 1.1 | - | - |
| Sub-total | 3.0 | 14.1 | 12.0 | 16.7 | 0.2 |
| Surplus | 9.6 | 23.3 | 2.7 | 8.6 | 0.6 |
| Losses | | 0.26 f) | 0.36 f) | | 0.3 |

a) Withers (1996); b) Results from a study comparing an intensive high output dairy farming system with a dairy system receiving reduced P inputs in feeds and fertilisers and reduced stocking density after Withers et al. (1999); c) Commercial farm taken over in 1985 with detailed records of fertiliser P inputs and crop yields (Whinfield pers. comm.); d) Haygarth et al. (1998a); e) Sold silage; f) Measured losses of P in storm run-off and leaching.

The largest P import is fertiliser, without which the UK P surplus would be close to zero. However, P fertilisers are required in areas, which have no livestock, and it is probably the uneven redistribution of faecal P that is responsible for the P imbalance. Indeed, the exports of crop and animal products represent only about one quarter of the c. 210 $10^6$ kg of P recycled in UK agriculture either as excreta, or as home-grown feedstuffs fed to livestock (Withers, 1996). The redistribution of these large P amounts, which often require handling on more than one occasion, increases the opportunity for wastage and P loss at the farm scale. Comparing overall P inputs to outputs, UK agriculture would appear to be only 24% efficient (table 16.1). Most of the developed countries operate a national P surplus (Brouwer et al., 1995).

At the farm and field scale, surpluses of P may differ substantially from the national picture depending on the type of farming system (table 16.1). For example, upland grass/hill farms may operate a very small surplus compared to an intensive dairy farm with significant feed imports, although their P loss rates may be very similar. Unlike N, the link between the P surplus and the loss of P is not clear due to differences in the patterns of flow and retention of P in the soil. Arable farms without access to manure inputs may either be in balance, or be in surplus. For example the arable farm in table 16.1 has been trying to build-up soil P fertility with fertiliser P inputs in excess of P offtake, and has a surplus similar to the UK average. Horticultural holdings use relatively large amounts of fertiliser and manure P in their multi-cropping systems and often have high concentrations of readily-exchangeable P in the soil.

The greatest risk of P surplus is on farms, which generate or import far more manure than can be sensibly recycled to the available land area. For example, Brouwer et al. (1995) calculated an average P surplus of 269 kg per ha*year for granivore farms where stocking densities are very high and there is little land area for recycling of the manure. In addition, national surveys show that those farmers who do recycle manures on their farms do not take account of their fertiliser value, but still apply substantial amounts of inorganic P fertiliser (Edwards et al., 1997), a practice which limited research suggests is unnecessary (Smith and Van Dijk, 1987; Van Dijk and Sturm, 1983). Hence, the frequency of P application, or deposition for grazing animals, and the potential for large P surpluses, on individual fields is locally very variable. Fundamental differences in the distribution of surplus P with soil depth also exist between uncultivated and regularly cultivated land. Since the amount of P loss in storm run-off has been shown to be related to the rain-soil interaction within a shallow (1-2 cm) surface layer (Ahuja, 1986), differences in the distribution of soil P between arable land and grassland maybe environmentally significant.

## 16.3 Diffuse phosphorus loss from agricultural land to water

### 16.3.1 Sources and transfer of phosphorus

Since nutrient transport from land to water is a natural process, it implies that agriculturally driven P limiting eutrophication problems probably occur due to a sustained acceleration of diffuse P loss in land run-off (Gibson, 1997). Agriculturally derived point source P inputs, such as stormflow from farm waste stores or farmyards, may also contribute significant P loading to watercourses, but probably do not re-occur on a regular or widespread basis. Concentrations of P in rainfall can be large; for example Withers et al. (1999) recorded values > 700 µg per l associated with wind-blown soil particles in a dry arable area, but the loads landing directly on watercourses are comparatively small. Hence, agricultural P loss largely occurs diffusely in land run-off due to the transport of soil particles (erosion) and in soluble form (surface run-off and leaching; figure 16.2). Freshly-applied fertilisers and manures, including those deposited by grazing animals, which are not incorporated into the soil also contribute directly to incidental PP and DP loads in run-off producing land areas, especially on grassland (Heathwaite, 1997).

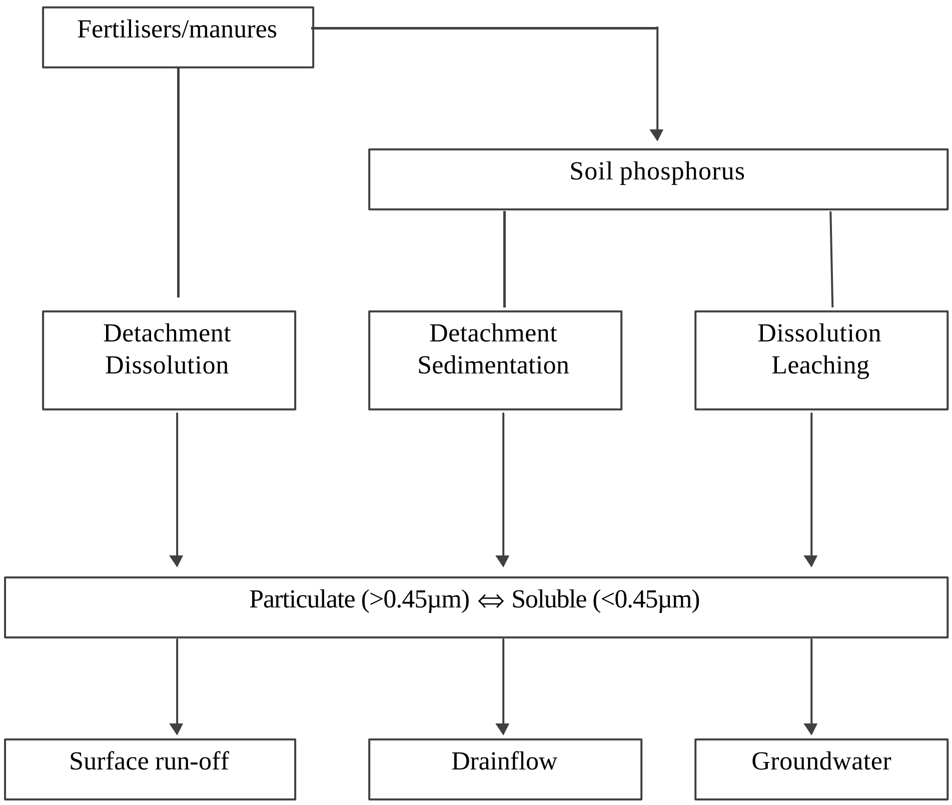


*Figure 16.2 Sources, processes, forms and pathways of P loss from agricultural land to water*

Analytically, the transfer of P in land run-off has been separated into that which occurs in association with soil particles passing >0.45µm (particulate P or PP) and that which occurs in soluble form (DP, <0.45µm). In reality, amounts of P in true solution may be very small, and most P is probably in particulate or colloidal form (Edwards and Withers, 1998). Although often viewed as a surface run-off problem, there is increasing evidence that tile under-drainage systems in fields act as important delivery channels for mobilised P from the land surface to the watercourse (Hodgkinson and Withers, 1996; Grant et al., 1996; Stamm et al., 1998). Hydrological conditions operating within the drainage basin or catchment are the main driving force for P transfer, and other important routes of connectivity to the stream include ditches, tracks and roads (Harden, 1992). The relative proportions of PP and DP in land run-off therefore depend on the complex interaction between climate, topography, soil type, soil P content, type of farming system, and farm management. The influence of agriculture and agricultural practices on the potential for P loss under the site hydrological conditions, and the identification of effective P loss control strategies, requires knowledge both of P inputs (nutrient management) and P transport (land management) (Withers and Jarvis, 1998).

### 16.3.2 Nutrient management

Phosphorus accumulation in the soil increases all P fractions, but is accompanied by an increase in the ratio of readily-exchangeable to total P. Linear and non-linear relationships between readily-exchangeable P (soil test P) and DP loss in surface and sub-surface run-off have been recently demonstrated in laboratory studies (Sharpley, 1995), plot studies (Heckrath, et al., 1995), and/or whole field comparisons (Smith et al., 1998). Such data suggests there is a critical soil test P level above which DP loss is greatly accelerated, or becomes unacceptable; $C_2$ in figure 16.1. Differences in the nature and/or slope of the relationship depend on soil type, land management and the depth and time of interaction between stormflow and the soil. Although current soil agronomic tests may not be the most appropriate method for assessing P loss potential in storm run-off, it is usually the only data which is normally routinely available on a regular basis to quantify soil P accumulation and P loss risk. However, the analytical methods used differ widely between different countries and comparison of soil test P concentrations within the EU is problematical (Tunney et al., 1997).

Impacts of total P accumulation on PP loss in storm run-off are more difficult to quantify and have not been extensively studied. Erosion is a selective process and particles which are transported long distances in run-off tend to be very fine-textured, highly P reactive and enriched with P compared to the bulk soil (Sharpley and Smith, 1990; McGuire et al., 1998). Annual P surpluses may only represent a very small proportion of the total soil P content, but depending on the soil depth over which P accumulation is measured. For example, a UK surplus of 10 kg P $ha^{-1}$ is only 0.4% of the median content of total P down to a plough depth of 25 cm, and may need to be continued for many years before increases in PP transport could be detected on arable soils. The relative contribution of agriculturally-derived P in the soil compared to the native P content under non-agricultural use is therefore unclear. Some parent materials (for example, chalks and limestones) are naturally rich in total P and these must be considered as natural hotspots of P. On other soils, surplus P inputs may greatly influence soil total P, especially since P inputs are taken up preferentially by fine aggregates (McGuire et al., 1998). There is currently no soil test available that quantifies the potential for particulate P loss from a given soil type.

When fertilisers and manures are applied to the surface of the soil, there is also a risk of incidental P loss in storm run-off, especially if applied to soils already at field capacity, to frozen soils, or to cracked or recently underdrained soils. Although the amounts of P lost are generally very small (<5%) in relation to the total P amounts applied, the concentrations are well above those required for eutrophication to occur, with up to 30 mg per l recorded in field experiments. Measured losses depend on the rate, time, method and frequency of application, the form in which the P is applied, hydrological conditions following application, and amounts of vegetative cover (Heathwaite, 1997; Smith et al., 1998). These factors are fundamentally different between arable and grassland farming systems. Regional information on the rate and type of fertiliser/manure and the method and timing of application is needed in relation to site factors, but it is unclear how widely available these are. In the UK, this information is derived by an annual survey of fertiliser practice (Burnhill et al., 1997).

### 16.3.3 Land management

Erosion is the main process of P transfer from agricultural land to water and is considered to have increased as a result of modern farming techniques. Field experiments and erosion surveys indicate that P export rates from arable land typically range from 0.1-30 kg P per hectare (Schonning et al., 1995; Chambers, 1997). In the UK, major contributing factors are the increase in the area sown to winter cereals, the introduction of tramlines which concentrate and increase the velocity of water flow, the removal of hedgerows which increases the length of slope, and the reduced soil stability arising from continuous cultivation (Evans, 1990; Spiers, and Frost 1985). For example, in a survey of 145 eroding fields in England and Wales monitored during 1989-1994, 80% were cropped to winter cereals (Chambers, 1997).

However, the development of rills and gullies at the surface of cultivated fields is not the only form of erosion from agricultural fields. Significant losses of particulate P also occur without any obvious disturbance of the soil surface, in sub-surface flow through tile drains and from poached grassland (Haygarth et al., 1998b; Heathwaite, 1997; Hodgkinson and Withers, 1996). For example, Hodgkinson and Withers (1996) measured particulate P losses of ca. 1.5 kg per hectare from a field drain in a dispersive silty soil under arable cropping. Using $^{137}$Caesium fingerprinting techniques, Grant et al. (1996) indicated that transported sediment-bound P in field drains originated from the topsoil and travelled down soil macropores and fissures. Heathwaite et al. (1990) found significantly greater loss of sediment bound P under high stocking densities compared to low stocking densities on grassland in a high rainfall area. Gateways and drinking troughs are high risk source areas for particulate P loss due to the effects of heavy poaching reducing ground cover and creating soil disturbance.

Quantification of particulate P emissions requires data on the amount of suspended sediment in run-off and the P concentration of the suspended sediment. Generally, transported particulate P is composed of silt and clay fractions, which are enriched in P compared to the bulk soil (Sharpley, 1985; Sharpley and Smith, 1990). The enrichment ratio will vary between sites (<1-5) according to soil type, P fertilisation history and the depth of P accumulation, but has been shown to be related to soil loss within a site (Sharpley, 1985). Models are available to predict these two parameters but they require validation at a wide range of field and catchment scale. In particular, there is little data on the differences in the spatial distribution of sediment-bound P within catchments of different size.

## 16.4 Conclusions

Recent research indicates that accelerated losses of P from intensively managed farmland to water may contribute to eutrophication problems within catchments. However the major source areas and pathways of diffuse P loss, and the specific contribution of agriculture or agriculture practices are difficult to identify and quantify accurately. There is marked spatial and temporal variation in the loads and concentrations of P in land run-off due to the complex interaction between the amount and intensity of rainfall, the susceptibility of the soil to detachment and erosion, the soil P content and its degree of saturation and the presence of fertilisers, manures, and crop residues at the soil surface.

Undesirable agricultural practices which accelerate PP and/or DP loss are the unnecessary accumulation of P, the application of P amendments at rates and at times which cause direct run-off, and the adoption of land management practices, which increase erosion risk on unstable soils. Other farm management practices, which have been introduced to improve agricultural production (increased winter cereals, tramlines, underdrainage, slurry-based livestock systems), probably also increase the potential for diffuse P loss but have other advantages, which makes their desirability more difficult to evaluate. Fundamental differences in the amount and frequency of P application, the amount of surplus P, the depth of soil P accumulation, inherent ground cover and hydrological conditions exist between cultivated and uncultivated systems which have implications for P transfer to water.

Although the precise impacts of diffuse P loss from agricultural land on water quality are poorly understood, it is clear that certain farm management practices can cause greatly accelerated P loss in land run-off, which has the potential to cause eutrophication problems and therefore are both unnecessary and unacceptable. Methodologies need to be established to quantify the total P load derived from agriculture, the impact of this load (or concentration) on the biotic equilibrium at critical times of the year, and to pinpoint the major source areas and pathways contributing the P loss so that control options can be implemented effectively.

Quantification of the P loss risk associated with regionally produced agricultural products requires assessment of the relative importance of regional differences in site and agricultural management factors. The availability of regional data is likely to vary considerably between different countries requiring inventories of such data within broad ecological zones. This information is also needed to highlight where in the production cycle most P emission occurs and how it might be controlled, since causal factors may be very different in different regions both within and between countries.

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# 17. Flows of phosphorous in the environment: identifying pathways of loss from agricultural land

*Louise Heathwaite* [1]

*Abstract*

Phosphorus (P) flows in the environment are hydrologically driven, and enable the transport of potentially mobile P from agricultural land to receiving waters. Two key modes of transport exist: surface runoff and subsurface flows. While surface runoff remains an important pathway of P loss, recent research demonstrates the potential for subsurface transport of P in macropore flow and from drained land. The forms of mobilised P differ according to the transport pathway. For grassland, dissolved P is transported in surface runoff but particulate P is proportionately more important in macropore and drainflow - especially during storm events. Tilled land generally shows high particulate P transport. Where livestock intensification has increased the rate of manure returns to land, there is clear evidence of enhanced P transport, both as incidental losses in surface runoff and through matrix or preferential flow in subsurface pathways.

*Abbreviations*: TP total P; TPP total particulate P; POP particulate organic P; PIP particulate inorganic P (adsorbed onto Fe/Al complexes and as Ca/Mg phosphate); TDP total dissolved P; DIP dissolved inorganic P (molybdate reactive P); DOP dissolved organic P (may include P oxides).

## 17.1 Introduction

The flow of phosphorus (P) from agricultural land depends on the coincidence of source and transport controls. Phosphorus source areas have a high potential to contribute P; they are often spatially limited and may include land of high soil P status or reflect agricultural land uses which increase surface P concentrations, for example, intensively grazed grassland or certain arable crops. Phosphorus source areas are dynamic and reflect agricultural land use and management. Transport factors describe the hydrological processes, which translate P source areas into P loss areas. Not all catchment areas are equally vulnerable to P loss; certain areas contribute runoff (both surface and subsurface) more readily than others do. For example, hillslope hollows become saturated through the confluence of subsurface water with the consequent rise in the local water table and increased risk of saturation-excess surface runoff (see later). In terms of P transport, such areas do not pose a risk unless they are *coincident* with P source areas. This means that within an agricultural catchment it is possible to have areas with a high potential to contribute P but no P transport if the hydrological connectivity does not exist; conversely we may have areas with high hydrological connectivity but no P transport because they do not link to P source areas. This paper will examine the hydrological pathways of P

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transport in an attempt to account for their significance in contributing to P flows from agricultural land to receiving waters.

Various conceptual models have been developed to provide an overall representation of the mechanisms of P export from agricultural land. Heathwaite (1995) suggests that here are two key controlling factors: soil (defining the initial chemical form of P export) and hydrology (initiating P mobilisation); these are implicit in the source control vs. transport control argument outline above. These ideas are developed further by Haygarth and Jarvis (in press). A modified version of their conceptual framework is given in figure 17.1. It highlights the significance of *hydrology* as the driving mechanism of P transport.

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*Figure 17.1 Conceptual model of phosphorus transport*

## 17.2 Background

This paper presents a UK perspective on the issues surrounding P flows from agricultural land to water. The general trends in P loads reaching freshwaters reflect greater control of point sources of P from industrial and urban areas, especially sewage treatment works, with a relative increase in the contribution from non-point, primarily agricultural sources. Total annual P loss from agriculture to surface waters in the UK is estimated at around 12.7 $10^6$ kg (Withers, 1998) or 1.4 kg P per ha*year. Approximately 57% of TP loss is derived from drained and undrained permanent grassland, with 23% from drained tilled land or grass leys and a further 18% from undrained tilled land or ley. Over the past 25 years, P inputs to land from fertilisers and manures have changed little but a greater proportion of tillage land is sown to winter cereals, more land is underdrained, and livestock density has increased (Withers, 1996). These land use trends have a potential to enhance P transport through soil erosion, subsurface P losses, and P enrichment of surface soils, respectively. The latter has received considerable attention because nutrient control is possible through regulation of fertiliser and manure inputs to land. Application of manures based on N demand results in overapplication of P because crop nutrient requirements are satisfied by a N:P ratio in the region 7-11:1 whilst manures generally fall in the range 2-6:1 (Smith et al., in press). Around 119 $10^6$ kg of P are returned annually to UK agricultural land as manures; an estimated 55% are applied to tillage land and 46% to grassland (Burnhill et al., 1994; Smith et al., in press). Part of the explanation of the current UK P surplus of circa 10 kg per ha*year may lie in P enrichment of surface soils because livestock manure P is undervalued (Sharpley and Withers, 1994).

## 17.3 Flow pathways in agricultural catchments

Figure 17.2 illustrates the main hydrological pathways important in P transport at the hillslope scale. Key P inputs to the systems are indicated. This scale has been selected because it enables some integration of current understanding of the mechanisms of P mobilisation with that of research on the magnitude of P loss.

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*Figure 17.2 Key hydrological pathways of phosphorus transport at the hillslope scale*

### 17.3.1 Surface pathways

Two starting points for the generation of surface runoff are recognised. The first, *infiltration-excess* flow is generated when the infiltration capacity of the surface soil is exceeded, usually as a result of high intensity storm events. In the UK, rainfall intensities are generally low and the soil infiltration capacity is unlikely to be exceeded (Kirkby, 1988) except where land management modifies the soil surface. Examples include intensive grazing of grassland or fodder crops (Heathwaite et al., 1989, 1990), which may generate infiltration-excess surface runoff on a field-wide scale. The second, *saturation-excess* surface runoff is topographically driven from spatially and temporally dynamic variable source areas (VSAs) (Beven and Wood, 1983). This pathway is triggered where the soil becomes saturated via lateral percolation above an impeding horizon. Saturation-excess surface runoff also occurs where the soil water table rises to the ground surface through convergent flow into hillslope hollows or where a rising stream water level result in saturation of near-stream zones. Under steady rainfall, saturation-excess flow requires much lower rainfall intensities to maintain it in comparison with infiltration-excess flow and is generally a more important mechanism of surface runoff generation.

### 17.3.2 Subsurface pathways

Subsurface flow may reach the drainage network via a number of pathways: (i) groundwater, (ii) lateral flow where soil layers have vertical conductivity < rainfall intensity, and (iii) where concave topographic contours create contributing areas because a high water table and/or subsurface impedance causes convergent flow. Where soils are deep and the bedrock permeable, percolation to groundwater rather than channelling of flow laterally will occur. The rate of subsurface flow depends on soil conductivity, which defines whether matrix flow (saturated/ piston flow) or preferential (macropore) flow predominates. Preferential flow defines a rapid pathway of water transit through the soil. Certain antecedent thresholds (e.g. rainfall intensity and duration > 10 mm per day; soil moisture ($\theta$) $\geq$ 0.3) must be satisfied before it occurs (Germann, 1986). It may occur naturally via soil macropores (Beven and Germann, 1982) or artificially via field drains (Armstrong and Garwood, 1991). Some soils, such as cracking clays, have a greater preponderance of macropores and hence more channelling of subsurface flow via this pathway.

## 17.4 Phosphorous fractionation in flows from agricultural land

Chemical fractionation procedures and soil P testing are considered by Edwards et al., (1997), Sims (1993, 1998), and Tunney et al. (1997). Recent research has focused on evaluation of P bioavailability in runoff (Dils and Heathwaite, 1998; Sharpley, 1993; Sharpley et al., 1992). Phosphorus is primarily mobilised as ions of inorganic orthophosphate or in association with organic or inorganic colloidal and particulate material. Phosphorus forms are commonly subdivided into particulate and dissolved fractions. The division is arbitrary; 0.45µm is the analytical divide (Johnes and Heathwaite, 1992). Recent research (Matthews et al., 1998) has questioned the viability of this division in assessing the consequences of P transport for the quality of receiving waters. Their work suggests that organic P fractions may form a larger part of transported P than previously thought and that attachment to soil colloids < 0.45µm may be particularly important.

Figure 17.3 indicates the main P fractions transported along the various hydrological pathways described in figure 17.2. In general, and primarily for tilled land, P transport in particulate form is associated with surface runoff. Here the selective adsorption of P onto clay and silt-sized soil particles (as Fe/Al complexes or Ca/Mg phosphate) enables mobilisation with soil eroded from agricultural land. Transport of P in particulate organic form is important in grassland systems (Heathwaite et al., 1990). Subsurface pathways are commonly associated with P transport in dissolved form. However, preferential flow may also be an important pathway of particulate P transport (Dils and Heathwaite, 1996; Heathwaite, 1997) particularly attached to colloidal material (Haygarth et al., 1997). At the receiving end of the conceptual diagram (figure 17.3) factors such as mineral formation and dissolution control P bioavailability (Lijklema, 1994).

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*Figure 17.3 Phosphorus fractions transported in hillslope hydrological pathways*

### 17.4.1 Surface pathways of P flow

Surface runoff has a strong affinity for P transport because the surface soil has the greatest effective depth of interaction (EDI) (Ahuja, 1986; Sharpley, 1985) and the highest concentrations of P (Haygarth et al. 1998). Phosphorus residing in the surface 0.5 mm of soil appears to be most vulnerable to export in runoff. Phosphorus transport in surface runoff is influenced by farming type, erosion potential, hydrologically effective rainfall, land use including fertiliser and manure amendments, the presence or absence of livestock, and soil total P (Chambers, 1997; Heathwaite, 1997). Surface runoff is important in physically transporting P via soil erosion (Sharpley and Smith, 1990). Even where erosion is minimal, elevated soil P can sustain high TP losses. For example, for grassland soils P transport in surface runoff may be exacerbated by high P concentrations at the soil surface as a result of organic matter inputs (Haygarth et al., 1996). Table 17.1 indicates the P fractionation in surface runoff from grassland in the Trent experimental catchment, Midlands, UK (Dils and Heathwaite, 1996). Here the concentration of dissolved P exceeds particulate P with most P transported in the DIP fraction. The large standard error indicates the wide spatial and temporal variation in TP transport in surface runoff. Similarly, Haygarth and Jarvis (1997) reported 70% TP transport in the dissolved fraction in surface runoff from grassland. Whilst Edwards and Daniel (1993) recorded TP loss in excess of 5 kg per hectare (95% dissolved P) from land receiving poultry litter. The rate, timing and form of manure applications are important (Heathwaite et al., 1998) as is the time interval between application and rainfall (Haygarth and Jarvis, 1997; Hooda et al., 1996; Sharpley et al., 1994).

*Table 17.1 Phosphorous fractionation (mg per l) in surface runoff from grassland (1994-1996) Trent catchment, Midlands, UK*

| | Total P | Total dissolved P | | Total particulate P | |
|---|---|---|---|---|---|
| | | DIP | DOP | POP | PIP |
| Mean | 1,136 | 488 | 214 | 341 | 93 |
| Standard error | 77 | 61 | 38 | 32 | 17 |
| N | 60 | 60 | 60 | 14 | 14 |

TDP = DIP (dissolved inorganic P or molybdate reactive P) + DOP (dissolved organic P); TPP = POP (particulate organic P) + PIP (particulate inorganic P).
Source: Modified from Heathwaite (1997).

Where land management has increased the incidence of infiltration-excess surface runoff, significant transport of P may occur during storm events - often on a field-wide scale (Heathwaite, 1997). Some land management practices or crop types present a greater risk of P transport than others do. Where P transport is linked to soil erosion, high risk crops include winter cereals and winter vegetables, with temporary grass (< 5 years old), potatoes, sugar beet and maize of medium risk, and other arable crops such as spring cereals and oilseed crops of low risk (Chambers, 1997). In the UK, around 60% high-risk land is actually eroding, with 20% for medium risk and 10% for low risk (Chambers et al., 1992). The authors estimate that only 33-43% of erosion events actually transport sediment and P to streams. Thus on average, for the UK, around 5,000 tonnes per year or 0.6 kg total P per ha*year may reach watercourses. Land management such as grazing fodder crops, may compact the soil surface and decrease the infiltration capacity of the surface soil leading to sheet erosion and associated P transport on a field-wide scale (Heathwaite et al., 1990). Chambers (1997) suggests that P loss by sheet erosion could be significant in the UK because, when triggered, it operates over large land areas. Phosphorus transport in surface runoff from critical source areas (CSAs) has recently been shown important. Pionke et al. (1997), for example, suggest that 90% of the P load in receiving waters is derived from just 10% of the catchment. These CSAs are commonly linked to areas generating saturation-excess surface runoff. Partial source areas (PSAs) may also be effective at entraining P in surface runoff (Dils and Heathwaite, 1996). PSAs include effluent leakage from silage clamps, runoff from farmyards, channelling of flow along roads or tracks and tractor wheelings or animal tracks within fields. However, hydrological connectivity with the stream must exist for them to be significant factor in P transport to receiving waters. Although a number of researchers have recognised the contribution from PSAs to P transport (Heathwaite et al., 1989; Dils and Heathwaite, 1996) it is difficult to quantify their actual contribution to the P load of receiving waters. Furthermore, the incidence of PSAs have a low frequency (although the P loss may be high) thus their impact remains under-researched.

### 17.4.2 Subsurface pathways of P flow

While the water reaching a stream via surface runoff largely constitutes rainfall falling during the event, subsurface flow reaching the channel is unlikely to be physically (or chemically) the same water as is actually falling as rainfall. Thus, evaluation of timelags in the system is crucial in understanding the mechanisms of P transport, especially as P transformations in transit through, for example, sorption of P from infiltrating water, will be far more important along subsurface pathways relative to surface pathways. To date there is little research on tracing P transformations during transit along subsurface pathways. This may be partly a reflection of the difficulty in isolating and measuring the P load along subsurface pathways and the perceived importance attached to surface pathways of P delivery during storm events. This assumption regarding the relative importance of surface vs. subsurface pathways of P transport may be well founded. Sharpley and Withers (1994), for example, compared P transport in surface runoff with losses in throughflow and artificial drainage and suggest that up to 9% of applied P fertiliser may be recorded in surface pathways compared to less than 1% in subsurface flow (although P loss in drainflow was higher). Hodgkinson and Withers (1996) demonstrated the importance of soil type, slope, and antecedent moisture on the incidence of surface vs. subsurface runoff and P transport. Their field losses of P in surface runoff and subsurface flow are presented in table 17.2a and 17.2b, respectively. Significant P transport (up to 1.76 kg TP per hectare) was recorded in subsurface flow; although the contribution in this pathway remained smaller than that in surface runoff. Grassland clay soils record highest TP and DIP transport with the exception of losses from sandy soils during wet years.

Three subsurface pathways are recognised as having potential for P transport: first, *near-surface lateral flow*, owing to higher soil P concentrations in upper soil horizons - although P present in this horizon may not necessarily be mobile. For example, Chambers and Smith (1998) found that whilst soils receiving high loadings of organic manures and inorganic fertiliser showed P enrichment in the upper 30 cm soil, there was no evidence of down profile mobilisation of P. Mobilisation appears to be dependent on the mechanism of subsurface flow. In general, matrix flow is unlikely to initiate significant P transport, whereas preferential flow may be important (see below). The significance of P transport to groundwater by leaching depends on the depth to the water table and P loading at the soil surface. Smith et al. (1998), for example, report P enrichment of subsoil (> 45 cm depth) where freely draining soils have a history of high organic manure loadings. For P rich sites they found that the concentration of DIP in soil water moving in matrix flow below 30 cm increased where the Olsen extractable P concentration in the soil exceeded 70 mg per l. Thus, the potential for high groundwater loss of P exists where there is significant down profile transport of P in P rich sites where the groundwater table is shallow. Heckrath et al. (1995), for arable soils, report enhanced P loss in drainage water where the Olsen extractable P concentration in the plough layer exceeds 60 mg per kilogram. In the Netherlands, for example, the shallow water table and high P loading at the soil surface has created a high potential for P transport to groundwater. Here, van Riemsdijk et al. (1987) suggest that breakthrough of high P concentrations to groundwater are likely within 20-30 years if manure P loadings at the soil surface continue at current rates.

*Table 17.2 Field losses of phosphorous (kg per ha) in surface runoff and subsurface flow for varying land use and soil types in England and Wales*

*(a) Surface runoff*

| Land use | Soil type | Slope | Total P (kg per ha) | Dissolved inorganic P (kg per ha) |
|---|---|---|---|---|
| Grassland | clay | 4° | 3.30 | 1.37 |
| Arable | silt | 5° | 0.07 | 0.02 |
| Arable | sand | 7° | 0.17 (dry year) | 0.01 (dry year) |
| Arable | sand | 7° | 9.33 (wet year) | 0.29 (wet year) |

*(b) Subsurface flow*

| Land use | Soil type | Total P (kg per ha) | Dissolved inorganic P (kg per ha) |
|---|---|---|---|
| Arable | clay | 0.70 | 0.25 |
| Arable | clay | 0.20 | 0.04 |
| Grassland | clay | 1.76 | 0.39 |
| Arable | silt | 1.64 | 0.25 |

Source: Modified from Hodgkinson and Withers (1996).

Second, *preferential flow* may enable rapid subsurface transport of mobile P through soil macropores. Macropore flow reduces the time for interaction and hence the degree of transformation of P forms during transit. This may affect the bioavailability of P reaching the stream network. To date there has been little work on solute movement via preferential flow and no studies on movement of P with the exception of some initial work by Dils and Heathwaite (1996). The majority of studies that have inferred macropore flow have no direct evidence. Thus, this flow pathway is often assumed by a process of elimination. For example, Thomas et al. (1997) and Heckrath et al. (1995) suggested P transport via macropores in the silty clay loams of the Broadbalk plots at Rothamsted (Herts, UK) because high P concentrations were measured in tile-drain flow but the soils had a large adsorption potential and P was absent in soil solution at depth. In addition to dissolved P forms, this pathway may be important for P transport in particulate and colloidal form - particularly from grassland soils (Heathwaite, 1997). Dils and Heathwaite (1996) found around 68% TP transport in macropore flow from a mixed grass/arable catchment was in the particulate fraction, with mean concentrations of 842, 265 and 576 mg P l-1 for TP, TDP and TPP, respectively. Within the particulate fraction, the organic phase dominated, accounting for around 62% TP transported in macropore flow in the upper 45 cm soil.

Finally, *artificial drainage* acts like preferential flow to encourage rapid transit of water from land to stream. Approximately 6.4 million hectare of agricultural land have been underdrained in England and Wales: 71% on arable land and 28% on grassland (Belding, 1971; Robinson and Armstrong, 1988). Phosphorus loss in drainflow is influenced by soil type (stability), soil total P, and excess winter rainfall (Chambers, 1997). Drained clay soils, for example, transmit water rapidly via cracks and mole channels; contact with subsoil is minimal and high P losses might be anticipated, particularly where such soils receive high fertiliser or manure amendments. Dils and Heathwaite (in press) monitored the P fractionation in drainflow and streamflow for a number of storm events in the Trent catchment, Midlands, UK. The physico-chemical fractionation of TP appeared to be dependent on flow: at low flow DIP dominated and TP concentrations were low (<100µg per l), at high drainflow (>10 l per minute) associated with storm events, PP dominated with concentrations up to 1 mg TP per l. Kronvang et al. (1997) found that up to 18% of annual particulate P loss from a lowland arable catchment in Denmark was transported in subsurface drainage. Total P loss from grazed underdrained land with high animal manure inputs was over 5 times greater than underdrained arable catchments (0.63 and 0.12 kg P per ha*year, respectively; Grant et al., 1996). A comparison of P export from drained and undrained agricultural land in England and Wales is given in table 17.3; P forms are not distinguished. Total P loss from agricultural land was estimated at 12,675 tonnes per year which is equivalent to 1.4 kg P per ha*year (Chambers, 1997). Underdrainage makes a significant contribution (38% of TP loss). The magnitude of loss via drainflow depends on the effective rainfall. Grassland makes the greatest contribution (43%) to TP loss although the P export coefficients are based on limited data. Lower P export around 0.5 kg per ha*year was recorded by Tunney et al. (1997) for Irish soils. Grassland drainage appeared to reduce the magnitude of P loss (Haygarth et al., 1998). Smith et al. (in press) recorded P transport in drainflow for tilled land receiving pig slurry, poultry litter or cattle FYM. The target rate of P application was 60 kg per hectare (range 37-103 kg P per hectare). Total P and DIP loss from pig slurry (1.15 kg TP per hectare; 0.44 kg DIP per hectare) exceeded that of poultry litter or cattle FYM (ca. 0.25 kg TP per hectare; 0.05 kg DIP per hectare). The magnitude of P loss was correlated with peak drainflow with higher P concentrations recorded in the first drainage event following manure application. Thus concentrated liquid manures significantly increased P transport in drainflow with DIP concentrations in drainage waters up to 1,000 μg per l. In summary, drain flow represents an important potential pathway of P loss, especially in grassland catchments. For drained soils, where the likelihood of surface runoff has been reduced as a result of drainage, subsurface pathways of loss may represent the main pathway of P loss. It is also important to recognise that this pathway does not require the high magnitude, high intensity storm events necessary to generate surface runoff. Thus, it may generate higher P losses than previously recognised.

*Table 17.3 Estimated phosphorous flow in surface runoff and drainflow*

| Land use | Drainage status | Hydrologically effective rainfall (mm) | Erosion risk | Phosphorus export coefficient (kg $h^{-1}$ $a^{-1}$) | Total P loss (tonnes $a^{-1}$) |
|---|---|---|---|---|---|
| Permanent | undrained | < 200 | - | 0.7 | 568 |
| grassland | | > 400 | - | 3.0 | 4,056 |
| Permanent | drained | < 200 | - | 0.5 | 195 |
| grassland | | > 400 | - | 2.0 | 1,296 |
| Tillage | undrained | - | very high | 28.0 | 73 |
| | | - | high | 6.0 | 204 |
| | | - | moderate | 6.0 | 433 |
| | | - | slight | 3.0 | 66 |
| Tillage | drained | < 200 | - | 0.4 | 263 |
| | | > 400 | - | 1.4 | 147 |

Source: Modified from Chambers (1997).

## 17.5 Conclusions and research needs

Table 17.4 presents a summary of the P 'signatures' recorded in different hydrological pathways for a mixed grassland/arable catchment in the Midlands, UK (after Dils and Heathwaite, in press). The data is a useful summary of the range and forms of P transport in different pathways. Highest P concentrations were recorded in surface runoff and near-surface lateral flow in macropores (0-15 cm). However, the P signatures differed: surface runoff was dominated by the DIP fraction whilst P transport in shallow macropore flow was primarily in the particulate fraction. It is possible that rapid P transport in macropore flow reduced the effective sorption capacity of the soil and turbid flow conditions generated high PP loss. Similar patterns are recorded for drainflow P transport during storm conditions where high PP loss is recorded. At low flow, minimal P transport takes place via this pathway. P transport in matrix flow was low (circa 100 μg P per l) and may indicate P adsorption in upper soil layers as long as flow is slow. The trends shown in table 17.4 do not reveal the extent of spatial or temporal variations at the field or catchment scale. An important research objective must be to integrate different scale of research from plots through to catchments. Of equal importance in terms of understanding and managing P flows is evaluating the thresholds of activation of different hydrological pathways. Hydrological processes within catchments reflect a continuum of pathways. These pathways may or may not be activated, depending on criteria such as antecedent moisture, topography, and rainfall intensity and duration. It is important to establish at what thresholds the balance shifts from subsurface to surface flow pathways, under what conditions matrix vs. macropore flow is initiated and what factors lead to infiltration-excess surface runoff rather than saturation-excess flow. These factors determine the significance of different pathways for P flows and ultimately the amount of P reaching the drainage network.

*Table 17.4 Average phosphorous concentration (μg P per l) in different hillslope hydrological pathways, Trent catchment, Midlands, UK*

| Hydrological pathway | Total P | Total dissolved P | | Total particulate P | |
|---|---|---|---|---|---|
| | | DIP | DOP | POP | PIP |
| Surface runoff | 1,136 | 488 | 214 | 341 | 93 |
| Matrix flow | 102 | 34 | 35 | 33 | (as TPP) |
| Macropore flow (0-15 cm) | 1,181 | 377 | 11 | 793 | (as TPP) |
| Macropore flow (15-30 cm) | 717 | 189 | 74 | 415 | 39 |
| Macropore flow (30-45 cm) | 578 | 82 | 48 | 277 | 171 |
| Drainflow (baseflow) | 33 | 16 | 7 | 10 | (as TPP) |
| Drainflow (stormflow) | 966 | 192 | (as TDP) | 774 | (as TPP) |
| Groundwater | 339 | 72 | 20 | 167 | 80 |

TDP = DIP (dissolved inorganic P or molybdate reactive P) + DOP (dissolved organic P); TPP = POP (particulate organic P) + PIP (particulate inorganic P).
Souce: Modified from Heathwaite et al. (in press).

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# 18. Soil and crop characteristics in relation to heavy metal cycling

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*Heavy metal inputs in agricultural areas*

Different sources can be identified for the input of heavy metals in agricultural areas. All input routes are diffuse, but with important local and regional variations and substantial point-source influence. Generally speaking, the following input routes are quantitatively the most important:

*1. Atmospheric deposition of heavy metal containing aerosols originating from industry, energy production and traffic*

Until recently, atmospheric heavy metal inputs in agricultural areas in a densely populated country like the Netherlands were quantitatively so important, that this sole input source accounted for a net average accumulation in agricultural soils: crops were unable to remove the annual atmospheric inputs. In recent years, environmental protection measures (lead-free fuels, more strict control on industrial exhaust pipes etc.) changed this situation substantially and reduced the relative importance of this input source. However, in Eastern European countries and industrialising third world countries, atmospheric heavy metal inputs in agricultural areas are still increasing.

It must be stressed that atmospheric inputs are a diffuse source, but important local and regional variation does exist. Higher input ratios were and are observed near highways, railroads, energy production plants, and metallurgical industries. This makes the general input pattern for atmospheric deposition quite complicated.

Besides direct inputs of heavy metals, atmospheric deposition also contains acidifying components (from industrial and agricultural origin) which have their influence on heavy metal dynamics in soils. This will be discussed later.

*2. Inputs originating from (organic) waste materials, used in agriculture*

The use of organic waste materials (dredged sediments, sewage sludge, composted household waste, animal manure etc.) in agriculture are another important input route for heavy metals in agricultural areas. Organic wastes are used for their soil-physical and soil-fertility benefits and their inputs are regulated by quality standards, including acceptable heavy metal contents, as well as maximum application rates. Legislation and law enforcement differs between countries, as does the traditions for direct disposal of organic wastes in agriculture.

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Recent measures have reduced the input rate of heavy metals by this route, e.g. the reduction of the use of copper-containing fodder additives in pig breeding in Europe, which resulted in generally lower copper contents in animal manures.

*3. Inputs through inorganic fertilisers*

Phosphate fertilisers from certain sources, contain natural high levels of heavy metals, cadmium being the most important one. To illustrate the importance of this input source, research results indicated that cadmium levels in agricultural soils in the Netherlands are about four times higher than in natural areas. This can be attributed mainly to phosphate fertilisers. New fertilisation schemes and new fertiliser production methods tend to reduce somewhat the impact of this specific input source in recent years.

*4. Inputs through river sediment transports*

The introduction of heavy metals in floodplain areas, as a consequence of the sedimentation of heavy metal polluted suspended matter from rivers, has been substantially reduced during the last decades in Europe. Environmental control measures in Germany and Switzerland, related to waste water treatment, have shown to be quite effective in the specific case of the river Rhine, resulting in bringing to an end floodplain heavy metal accumulation in the Dutch river Rhine floodplain area. Floodplain pollution with heavy metals continues in other countries like Poland.

## 18.1 The filter/buffer function of the soil

After being introduced to soils through one or more of the above-mentioned input routes, heavy metals tend to sorb strongly onto soil particles and become quite unavailable for environmental effects like (i) leaching to subsoil and surface waters, (ii) affecting soil (micro)biological ecosystem processes (soil respiration, carbon cycling, mineralisation) and (iii) uptake by vegetation and introduction into animal and human food chains.

Sorption of heavy metals onto soil particles is strong, but not complete, which results in a small portion that is present in the soil solution. It is generally accepted that only when appearing in the soil solution, heavy metals become available for the environmental effects mentioned above. So it is important to be able to estimate the available fraction of heavy metals in the soil, if one aims at determining the rate of heavy metal cycling in the source/soil/soil solution/plant/(animal)/food system.

The degree of sorption of heavy metals onto soils depends on the following main (groups of) determining factors:

*1. General soil characteristics*

Heavy metals tend to sorb most strongly on clay particles and soil organic matter. Therefore, heavy metals in sandy soils are quite available for environmental effects in contrast to heavy metals in clayish or peaty soils.

*2. Variable soil condition*

Sorption of heavy metals onto clay particles becomes less strong when the soil becomes more acidic. The presence of calcium in the soil profile regulates the solubilisation of soil organic matter; at low calcium contents soil organic matter tends to partially migrate into the soil solution, carrying with it heavy metals, making them more available for environmental effects. In general, liming of agricultural soils reduces heavy metal availability for environmental effects, both through increasing pH-values and through reducing soil organic matter solubilisation. On the other hand, afforestation increases their availability.

*3. Heavy metal characteristics*

- Some heavy metal availability patterns are more sensitive to soil organic matter dynamics (e.g. copper and lead) and some are more sensitive to pH-changes (e.g. cadmium and zinc).
- Especially at the short term, influences are expected from the chemical form of the heavy metals introduced in the soil. For example, it will make quite a difference if lead is introduced as an inorganic lead salt or as and organic lead-containing material; the latter will become available much later; (microbial) degradation of the organic part of the molecules must take place first.
- Heavy metals that are already present in soils for a very long period (including naturally present heavy metal levels) may become so strongly included in the soil solid chemical matrix, that they will not participate any more in chemical equilibrium processes involving solid and liquid soil phases, after turning definitely unavailable.

It will be obvious that the availability of soil heavy metals for environmental effects can be manipulated through fine-tuning of variable soil conditions (liming, addition of organic matter etc.).

## 18.2 Heavy metal uptake capacity of crops

Crops take up more heavy metals when exposed to higher actual heavy metal concentrations in the soil solution. Broadly speaking, a specific crop reacts in a more or less linear way to changes in the heavy metal availability caused by changing soil conditions, provided that undisturbed plant growth is not affected by (i) conditions where a toxic response takes place or (ii) too low pH values.

The response of different crops to different heavy metals, present in the soil solution, is quite straightforward but greatly dependent on crop characteristics, even at the variety level. To relate accurately the heavy metal concentrations in the soil solution to final crop heavy metal levels, *plant-specific heavy metal accumulation factors* must be known. These factors must quantitatively indicate the capacity of a certain crop variety to absorb a certain amount of heavy metals when exposed to a certain heavy metal concentration in the soil solution at a certain water uptake rate during the growing season.

Not only the 'natural' tendency of a certain crop variety to absorb heavy metals from the soil solution is important, but also which types of heavy metal chemical species, present in the soil solution, can be absorbed at all by a specific plant species. Certain plants only can take up 'free' ionic species from the soil solution. Others are able as well to take up heavy metals, which are present in the soil solution in an associated (higher molecular weight) chemical form, especially as heavy metal dissolved organic matter complexes.

## 18.3 Heavy metal Life Cycle Assessment

To be able to assess heavy metal cycles on a regional scale in the source/soil/soil solution/ plant/(animal)/food system - and especially in the source/soil/soil solution/plant part of it - the following information is necessary:

1. *Heavy metal soil maps* indicating total heavy metal contents in the plough layer of agricultural soils, as well as local general soil characteristics like clay content, organic matter content, calcium content and soil conditions like liming rates and pH. The latter in relation to local and regional acid deposition rates as well. Data must be average values over a limited area and related to some degree to soil composition homogeneity over the area.
2. *Heavy metal input maps* indicating (on the same scale as the soil maps) the estimated annual diffuse heavy metal input rates from different main sources. From the heavy metal content soil maps and the heavy metal input maps, *potential heavy metal accumulation maps* can be developed.
3. *Formulas and simple soil chemical models*, which relate total heavy metal contents in soils to soil characteristics/soil conditions in order to obtain plant-available heavy metal concentrations in the soil solution.
4. *Agricultural land use maps* indicating and quantifying the relative importance of main regional crops. Using plant accumulation factors for heavy metals and data on growing season average rainfall, the annual removal of heavy metals by crops can then be estimated at a regional scale. Agricultural crop-related heavy metal inputs in the animal/ human food chain can then be quantified in *heavy metal crop uptake maps*.
5. Annual heavy metal removal rates at a regional level can be used as a feed back correction factor for the heavy metal accumulation maps, mentioned under 2). This finally leads to *net heavy metal accumulation maps* at a regional level.

## 18.4 Scientific state-of-the-art

Soil maps, containing average regional data on heavy metal contents in combination with general soil characteristics and conditions (1), are being developed world-wide through national and regional monitoring programs. It must, however, be assured that these monitoring programs are and will be mutually comparable (scale, applied sampling and analysis methodologies etc.). Data on heavy metal input rates and acid deposition rates (2) are becoming available as well; here the need for mutual comparability with heavy metal soil maps must be stressed as well.

Statistics-based formulas (and simple models) transforming total heavy metal contents in soils into plant-available heavy-metal species concentrations in the soil solution, are rapidly developed and becoming available now.

The main constraint lies in the evaluation of plant heavy metal accumulation factors for different heavy metals and different crop varieties. Here years of work lie still ahead and this type of applied research needs a strong stimulation in order to make possible Life Cycle Assessment for heavy metals within the next decade.

The general methodology depicted above can be used for regional analysis of heavy metal life cycles, but also for heavy metal life cycles at the farm level, finally leading to proposing scientifically sound changes in farm management, which aim at equilibrating input/ output of heavy metals at the farm level.

## 18.5 Conclusion

Within a few years, regional maps can be made available which can be used to quantify heavy metal balances at a regional level to be used for Life Cycle Assessment; the same maps can be used to develop strategies to equilibrate heavy metals at the regional and possibly farm level, leading to a politically acceptable impact of heavy metals on food quality.

To reach this objective two main conditions must be met:

- special attention for efforts aimed at quantifying plant heavy metal accumulation factors, especially for the main agricultural crops in Europe;
- special attention for efforts aiming at mutually fine-tuning the development of data collection at the regional level, enabling effective direct links between soil maps, crop maps etc. These efforts must be strongly supported and co-ordinated at the political level.

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# 19. Modelling of nutrient losses to waters and the atmosphere for different farm types

*Juha Grönroos and Seppo Rekolainen* [1]

*Abstract*

Nutrient runoff from fields and ammonia and methane emissions to the atmosphere from livestock farming make a major contribution to the total environmental effects of food production. It is important to develop methods that can be used to assess these emissions and to improve data availability/accessibility and quality used as input data. A major problem of deterministic runoff models is that the small-scale (e.g. soil profile or field-scale) models do not contain all the relevant process descriptions necessary in larger areas (e.g. channel and lake processes). Moreover, the material transfer from one modelled parcel to the next (e.g. from one field to another) is frequently lacking. This means that regional assessments can be considered rather as potential risk assessments than as predictions in absolute terms. In the case of nutrient balance calculations, availability of the input data is perhaps not such a great problem as processing the output data in order to determine what part of the nutrient surplus may potentially cause environmental problems. Assessment of gaseous emissions from agriculture to the atmosphere requires accurate data about manure handling methods. Accurate information is also needed on ammonia volatilisation during different phases of manure handling systems.

## 19.1 Introduction

Nutrient losses from cultivated fields to waters, and ammonia and methane emissions from livestock farming to the atmosphere, make a major contribution to the total environmental effects of the life cycle of food products. Assessing these emissions to waters and to the atmosphere is complex, because of the heterogeneity of production systems and local differences between soil properties and climatic conditions. In this paper different approaches to assess these losses and the input data requirements for these assessment systems are described. Data availability and data quality are also discussed using Finland as a case study.

## 19.2 Nutrient balances as indicators of nutrient losses

Nutrient balance calculations are used to estimate differences between the input and output of nutrients in farm systems (farm gate balance), fields (surface balance) or animal production systems (cattle

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balance). The farm gate balance describes the balance between purchased nutrient inputs to a farm and the nutrient content of products exported from a farm, whereas surface balance calculates the difference between nutrient input and output in single field parcels. Cattle balance indicates the difference between the nutrients in the animal feeds and in the animal products, theoretically equalling the nutrient contents of manure in the production system.

Nutrient balances have been calculated in order to make international comparisons between different countries (e.g. Brouwer et al., 1995; OECD, 1997) and national comparisons between different regions (e.g. Pirttijärvi, 1998). Moreover, in the case of available historical time series, trends in nutrient balances have been calculated (e.g. Pirttijärvi, 1998).

Nutrient balance estimates do not equal observed losses from agricultural production systems, since the balance calculations do not take into account the natural processes, which also contribute to the actual nutrient losses to waters and to the atmosphere. A mismatch in annual balance estimates and observed loss estimates may also be caused by long lags between changes in inputs and responses in actual losses. Moreover, nutrient balances do not tell whether the potential surpluses remain in the soil or enter the waters or the atmosphere. Thus, nutrient balances can only be used as an indicator of potential pressures on the environment, not as an input to systems assessing e.g. water and air quality.

### 19.2.1 Data requirements

In this context, only the requirements of the surface balance calculation method are discussed. In order to obtain comparable balance assessments it must be agreed what terms are to be included in input/output calculations. Depending on this, the most important data requirements are:

- nutrient quantities of fertilisers applied (both inorganic and organic);
- atmospheric deposition;
- biological nitrogen fixation;
- nutrient quantities within the harvested plant tissues.

Occasionally, nitrous oxide and $N_2$ emissions due to denitrification, ammonia emissions, and losses to waters are included in balance calculations. However, it is questionable to include these loss estimates in the balance calculations if they are used as indicators for losses.

### 19.2.2 Data availability

Depending on the scale of an assessment, the resolution of the data is of crucial importance. In the case of preparing national assessments, mean input and output values are usually sufficient. However, for regional or local studies or comparisons, local or regional data based on administrative units or natural units (e.g. river basins) are required. Usually, the availability/accessibility of the data becomes more complex when progressing towards smaller units.

If only inorganic fertilisers are used and the application levels are known, the quantities of nutrients applied per hectare can easily be calculated. However, the common statistics about inorganic fertilisers usually contain only sales of the fertilisers as a national average or as an average for larger regions within a country. For example in Finland, farm-scale fertiliser amounts can be assessed by farmers, but no comprehensive study about the distribution of fertiliser use between farms can be performed due to the lack of national statistics.

In addition to the lack of national statistics, amounts of nutrients in manure application are difficult to assess, because the nutrient contents of manure are often very variable and poorly known. Single analysis of the nutrient contents within one livestock farm does not necessarily help since nutrient contents of manure may vary widely even within one farm. This is a problem particularly in the case of solid manure. In Finland, livestock farms participating in the Finnish Agri-Environmental Programme (about 90% of Finnish farms) are obligated to perform manure nutrient analysis for nitrogen but phosphorus is only recommended. Farms can also use the manure nutrient coefficients, which represent average nutrient contents of different manure types in Finland. Of course, those coefficients do not take into account yearly changes or regional differences, which may be considerable.

In order to calculate the nutrient output within harvested plant tissues, dry matter yields and nutrient contents must be known. As with inorganic fertiliser, data at the field scale is not always available. Moreover, the accuracy of the information concerning yields of different crops is very variable. Usually, estimates that are more accurate are available for crops, particularly for cereals, which are exported from farms, while yield estimates for crops used for fodder inside a farm are more uncertain. Additionally, temporal and regional differences in yields may cause further problems. Crop-specific data, such as dry matter and nutrient contents, needed to calculate output nutrient flows, can be collected from the literature. In Finland, this data is also available at the regional level and for different years, but with a rather small sample size.

Other input parameters, which must be taken into account in balance calculations, are deposition of nutrients and biological nitrogen fixation. Deposition is usually well known through measurements or modelling. If biological nitrogen fixation is to be taken into account, it must usually be assessed for each field separately. To do this, the amount and the fixation capacity of each type of legume must be known.

Most of the data needed is available on the regional level, for different years and for different crops, but it still includes notable uncertainties. The greatest problems are the manure nutrient coefficients and the yields of different crops, which - if farm-specific surface balances are calculated - should be known at the farm level.

### 19.3 Deterministic models for assessing losses to waters

Several deterministic, dynamic models have been developed for predicting and assessing erosion and nutrient losses from agricultural land to surface and ground waters: CREAMS (Knisel 1980), GLEAMS (Leonard et al., 1987), SWRRB (Williams et al., 1985), EPIC (Williams et al., 1984), ANSWERS (Beasley et al., 1980), AGNPS (Young et al., 1987), GWLF ( Haith and Shoemaker 1987), WEPP (Lane and Nearing, 1989), ANIMO (Groenendijk and Kroes, 1998) and SWAT (Arnold et al., 1995). Of these models ANIMO is a one-dimensional soil-profile model, CREAMS/GLEAMS, SWRRB, EPIC and WEPP are field-scale models, whereas ANSWERS, AGNPS, GWLF and SWAT are basin scale models.

These models differ extensively from each other, but a common feature is that they use meteorological time series as driving variables and soil physical and chemical data to calculate losses in space and time. The scales of these models vary from one-dimensional soil profile models to three-dimensional watershed models. Many differences in these models also exist in the description of soil-water-atmosphere-plant processes.

### 19.3.1 Data requirements

Despite the differences between these models, the data required for reliable predictions are rather similar. The driving variables consist of historical time series for meteorological variables, such as precipitation, temperature, radiation, wind speed, and humidity. The time step for these data varies from very detailed (time interval shorter than one day, e.g. breakpoint rainfall data) to more aggregated monthly or annual time series. However, the most common requirement is daily data.

Physical and chemical information of soils consists of soil texture, hydraulic and hydrological properties, as well as chemical composition of soil, particularly in terms of carbon, nitrogen, and phosphorus pools of the soil. Most of the models also include a crop growth submodel, which in turn requires crop-specific data determining the development of yields, root growth, and residues.

Furthermore, data on management operations, such as planting, fertilisation, harvesting, and tillage are needed.

### 19.3.2 Use of the models

Deterministic models can be used for making predictions of the nutrient losses from the modelled system and for comparing various management practices. These models usually have different numbers of parameters, the estimation of which cannot be totally based on measurements or on available information. This fact gives rise to a need for calibration and validation, which in turn requires observed data sets of nutrient losses. The need for calibration is perhaps greater when the objective is to obtain loss predictions in absolute terms, whereas for comparative studies more general information on losses might be sufficient.

Frequently, transfer functions or regression equations are used to estimate erosion and nutrient losses from non-monitored basins. Usually the losses can roughly be predicted as a function of land use, and soil and topographical characters within the basin. As an example, fraction of agricultural land use have been used to predict nutrient losses from drainage basins in Finland (Rekolainen, 1989). The use of such regression equations is restricted, since they are valid only under similar conditions, from where the statistical relationships are originally derived.

For soil erosion assessments, the Universal Soil Loss Equation (USLE, Wischmeier and Smith, 1958, 1978) has been widely used. The erosion prediction in many dynamic models is also based on the USLE (e.g. CREAMS/GLEAMS, AGNPS, EPIC and SWRRB), while in WEPP, erosion is described more physically-based. However, both approaches require parameters, whose values are often not readily available.

### 19.3.3 Regional estimates

Many attempts have been made to scale up the small-scale (e.g. soil profile or field-scale) model results to obtain loss assessments for larger areas. One of the limitations is that the small-scale models do not contain all the relevant process descriptions necessary in larger areas (e.g. channel and lake processes). Moreover, the material transfer from one modelled parcel to the next (e.g. from one field to another) is frequently lacking. This means that the regional assessments can be considered rather as potential risk assessments than as predictions in absolute terms.

Furthermore, regional estimates require spatial soil, topographic and management data of the whole area. Topographic data (e.g. digital elevation models) often exists, but data on soil textural classes and other relevant soil properties is frequently not satisfactory. Statistics on management (e.g. cultivated crops, use of manure and fertilisers) is also often missing, or is based on average data for larger areas, not for all the individual field parcels within the area to be modelled.

## 19.4 Assessing ammonia emissions to the atmosphere

In the Finnish Environment Institute, an ammonia emission model has been created which can be used to calculate ammonia emissions from livestock farming and from the use of inorganic fertilisers (Grönroos et al., 1998). The model follows the paths of nitrogen excreted by each animal type during the manure handling system and during the pasture period. The model can be used to calculate emissions on the national, regional or farm level. The model calculates emission quantities directly as output, but specific emission coefficients per animal type are also produced. These coefficients can then be further used if information about manure handling methods is not available.

Changes in nitrogen content of manure during the manure handling procedure are calculated. The first input data are the numbers of animals and the volume of nitrogen (nitrogen coefficients) in the manure excreted by each animal type. This data is used to calculate the amounts of nitrogen excreted during one year by each animal category. With the information about pasturing, it is possible to calculate the share of manure, which is excreted inside animal shelters and on pasture. Manure excreted inside animal shelters is divided into different manure types in accordance with current manure handling methods. The nitrogen fractions excreted on the pasture and in animal shelters are followed separately until nitrogen enters the ground and is in non-volatile form or is used by plants.

Ammonia emission assessments for the use of inorganic fertilisers are based on the application rate of fertiliser nitrogen and the information about volatilisation of nitrogen in the form of ammonia after application. Application rates are usually well known on the farm level, whereas on the regional level information is available only based on statistics about fertiliser sales in different areas in Finland. The information about ammonia volatilisation from inorganic fertilisers has been gathered from the same sources as mentioned in the case of manure.

### 19.4.1 Data requirements

The model is based on:
- numbers of different animal types;
- nitrogen content of manure excreted per animal per year (nitrogen coefficient);
- data about manure handling methods;
- data about nitrogen volatilisation in each stage of manure handling;
- abatement efficiencies of the different emission reduction measures.

### 19.4.2 Data availability

For the nitrogen concentration in manure, it is often possible to use farm-level data because manure nutrient analyses have often been performed, but when missing, average coefficients must be used, which do not take into account the farm-specific characters, for example differences in animal feeding regimes. Data on distributions between different manure handling methods in Finland have been gathered from the literature and by expert judgements.

Data about nitrogen volatilisation from each stage of manure handling (if no emission abatement measures are used) have been gathered mainly from studies conducted in conditions, which might differ largely from Finnish conditions. Because of the differences between Finland and the countries where the emission studies have been made, using this data as such may not give correct emission estimates for Finland because of the lower mean outdoor temperature and the lower soil alkalinity in Finland. For this reason a temperature correction factor has been introduced in order to correct the volatilisation data.

Data about reduction efficiencies of the different emission abatement measures have also been gathered from foreign literature. It is assumed that the efficiencies of similar abatement measures are similar in different countries. However, uncertainties might be considerable for abatement efficiencies too.

The greatest uncertainties for volatilisation data are caused by changes in climatic conditions: for example, ammonia volatilisation is greatly affected by temperature and wind. For emission reduction data, the greatest uncertainties are caused by the quality of realising the reduction measures.

### 19.4.3 Regional estimates

In cases when no farm level data are available, it is possible to use regional data about the nitrogen content of manure using statistics based on obligatory manure analyses. Data about manure handling methods are currently available only on the national level, but in future regional data may also be available. At present, data about nitrogen volatilisation in each stage of manure handling are not differentiated regionally. In future, however, it should be possible to assess the regional differences caused by climatic differences. Data about abatement efficiencies of the different emission reduction measures are available only on the national level. However, there is no need to differentiate it regionally.

## 19.5 Missing data

More information about manure handling in different parts of the country is needed for making more accurate estimates of regional emissions to the atmosphere. To realise this, a manure handling data base should be created, in which farm-scale manure handling data could be stored.

Regional nutrient loss estimates require spatial soil, topographic and management data of the whole area. Topographic data (e.g. digital elevation models) often exists, but data on soil textural classes and other relevant soil properties are frequently not satisfactory. Furthermore, statistics on management (e.g. cultivated crops, use of manure and fertilisers) are often missing, or are based on average data for larger areas, not for all single field parcels within the area to be modelled. Even municipality-level information is not available on tillage practices or soil properties. The availability of such data is likely to vary considerably between different countries.

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# 20. Conclusions of the working group on non-nitrogenous substances

*Sarah J. Cowell* [1]

This paper presents a summary of the discussions and results for the working group on 'other (non-nitrogenous) substance cycles. In the Working Group, most time was spent discussing modelling flows of phosphorus in LCA; heavy metals were also discussed but in less detail.

## 20.1 An approach for modelling use of phosphorus and other substances for LCAs involving agricultural systems

For a number of substances that pass through agricultural systems, existing scientific data are inadequate for accurately modelling the flows of these substances. In other cases, there are models but they may be based on different assumptions and farming conditions. Grönroos and Rekolainen (this volume) discuss some of the relevant issues; they include problems such as data availability, scaling up from field to regional models, and differences between potential and actual environmental impacts.

Since the inventory phase of LCA aims to account for the flows of all environmentally relevant substances, it is necessary to identify and characterise these flows in as much detail as possible using existing models and appropriate assumptions when data are missing (Cowell, 1999). The working group therefore developed an approach for systematically addressing this issue. It involves asking four questions about any substance X considered in an LCA:

1. What environmental impacts are associated with substance X?
2. Are there characteristic patterns in the flow of X through agricultural systems leading to environmental impacts?
3. Are there characteristic geographical and management practices in farming systems leading to the environmental impacts associated with X?
4. Are there likely to be differences between the potential and actual impacts of X and, if so, what are the determinants?

The results are discussed below for phosphorus and heavy metals.

## 20.2 Modelling flows of phosphorus

Use of phosphorus (P) in agricultural systems may subsequently contribute to eutrophication in water bodies (Withers, this volume). The flow of P through these systems can be characterised as shown

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in figure 20.1 where the two widths of arrows represent the major and minor inputs and outputs of P through the system (see also papers by Chardon, Withers, and Heathwaite, this volume). The dotted arrows represent the flows of P to and from agricultural land outside the system boundary.

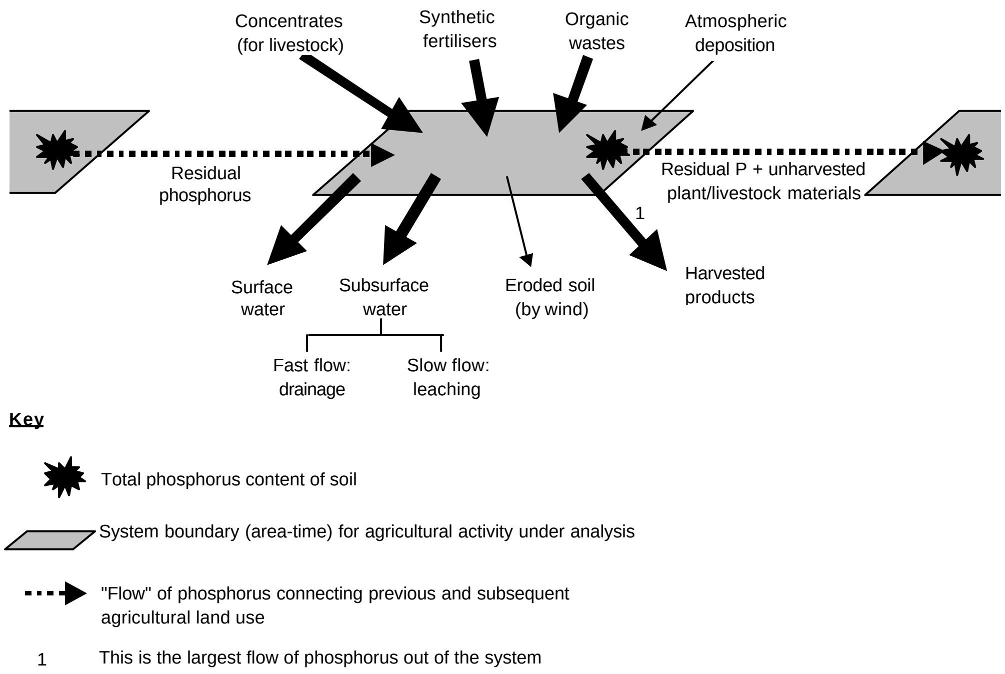



*Figure 20.1 Flow of phosphorus through agricultural system*

The diagram shows that the major flows of P imported into agricultural systems are via livestock concentrate feeds, synthetic fertilisers, and/or organic wastes (excluding P in the soil). One-off events, such as flooding of agricultural land, may also contribute P. The major flows out of agricultural systems are via surface and/or subsurface water (excluding P in the soil).

A number of geographical and management factors determine the magnitude of the flows (see figure 20.2 and papers by Chardon, Heathwaite and Withers, this volume), and may be related to specific distinguishable regions (Hughes and Larson, 1988). Through discussion, the group selected a restricted range of key factors characterising the major flows of P through agricultural systems (see figure 20.3). These are, therefore, the minimum data required to model flows of P through an agricultural system prior to impact assessment.

At impact assessment, there may be a difference between the potential and actual impact on eutrophication of water bodies due to P emissions from the system under analysis. This is because the actual impact depends on:

| Geographical factors | Management factor |
|---|---|
| Soil | Crop type |
| - P status | Livestock density and grazing management |
| - Parent material | Tillage |
| - Texture | - Reduced versus conventional |
| - OM content | - Timing |
| - Moisture | - Direction (e.g. contour ploughing) |
| - Hydraulic regime | Drainage |
| Climate | Surface features (e.g. hedgerows) |
| - Rainfall: total, distribution | Fertiliser/organic waste(s) |
| - Temperature | - Type |
| Water table | - Quantity |
| - Height | - Rate |
| - Fluctuation | - Timing |
| Topography/landscape features | - Application method |
| - Slope | Crop residue management |
| - Hedgerows | Applications of other nutrients, lime, and |
| Altitude | pesticides (because yield influences P uptake) |

*Figure 20.2 Geographical and management factors determining flows of phosphorus*

- the P content of vulnerable water bodies prior to addition from the system under analysis (i.e. the baseline);
- the distance between the system under analysis and a water body;
- the type of water receiving the P. For example, is it freshwater, estuarine or marine water? Is it surface- or ground-water? Is it standing or flowing water?
- the form of the P emission. For example, is the P dissolved or particulate?

The group thought that the FADN typology would be adequate for describing farming systems analysed during LCA, augmented by the management factors listed in figure 20.3. For the geographical factors, a GIS-based model defining regions using the factors listed in figure 20.3 would be appropriate. P emissions from any system under analysis could then be predicted using this combination of data. Impact assessment would then proceed on the basis of assessing potential impacts, or actual impacts given the availability of site-dependent data to modify the impact assessment factor for P limited eutrophication.

In developing such an approach, existing and/or new P models need to be developed. Although research on these models is at a relatively early stage (compared with, for example, models of nitrate leaching), existing projects include:

- ICECREAM model (details available from Seppo Rekolainen, Finnish Environment Institute, Finland);
- Soil Survey and Land Research Centre (SSLRC) model (details available from Tim Harrod, Soil Survey, IGER, UK);

- SC (ANIMO) model for acid-sandy soils (details available from Oscar Schoumans, Winand Staring Center, SC, Netherlands; email: o.f.schoumans@sc.wag-ur.nl);
- Export Coefficient Models (details available from Louise Heathwaite, University of Sheffield, UK; see also Johnes and Heathwaite, 1997).

| Farm typology | Geographical factors | Management factors |
|---|---|---|
| Crop or livestock type<br>Stocking density for livestock | Soil type (including typography)<br>P status<br>- Total P<br>- % saturation of P sorption capacity<br>Hydrologically effective rainfall (excess rainfall) quantity, intensity and duration | Quantity of concentrates, fertilisers, and/or organic waste(s)<br>Tillage<br>- Timing<br>- Type<br>Drainage |

*Figure 20.3 Key factors characterising magnitude of major flows of phosphorus through agricultural systems*

Furthermore, an EU Concerted Action project (COST832, 'Quantifying the Agricultural Contribution To Eutrophication') has recently started to develop methodologies for predicting P emissions in land run-off within catchments, and it may be beneficial to co-operate with this project on development of the approach outlined above. Paul Withers (ADAS Bridgets Research Centre, UK) is the Lead Co-ordinator for this project. Its website is: www.ab.dlo.nl/eu/cost832/welcome.html.

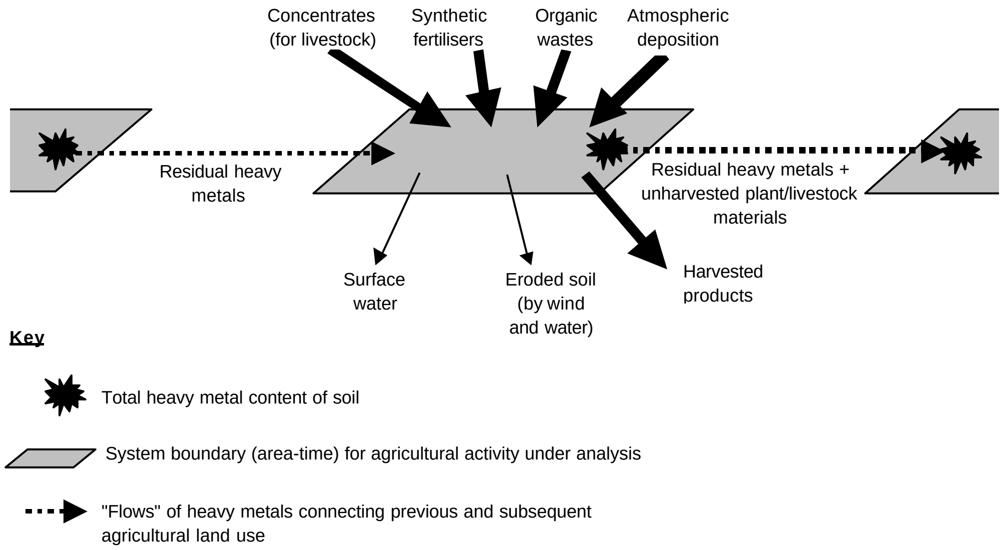
## 20.3 Modelling flows of heavy metals

Toxicity is associated with emissions of heavy metals to air, water, and soil. The toxic effects may be experienced by micro-organisms in the soil, grazing livestock, and/or crops on agricultural land, aquatic ecosystems, and humans via the food chain.

The characteristic flows of heavy metals in agricultural systems are shown in figure 20.4 (see also Japenga, this volume). The diagram shows that the major flows of heavy metals into agricultural systems are via concentrates for livestock, synthetic fertilisers, organic wastes, and/or atmospheric deposition. Cadmium (in phosphate fertilisers) and copper (in livestock feedstuffs) are the two heavy metals of primary concern associated with agricultural inputs, but a much wider range of heavy metals should be considered when accounting for atmospheric deposition and other non-agricultural inputs. One-off events, such as flooding of agricultural land, may also contribute heavy metals. The main flow out of the system is via harvested crops.

As for phosphorus, the group went on to define the key factors required to characterise the major flows of heavy metals through agricultural systems (figure 20.5). These are, therefore, the minimum data required to model flows of heavy metals through an agricultural system prior to impact assessment (see also Japenga, this volume).

At impact assessment, there may be a difference between the potential and actual toxicity of heavy metals due to aspects such as their tendency to bioaccumulate and synergistic interactions with other substances.




| Farm typology | Geographical factors | Management factors |
|---|---|---|
| Crop or livestock type | Atmospheric deposition<br>Soil characteristics<br>- pH<br>- Calcium content<br>- Clay content<br>- OM content<br>Parent material | Quantity of coincentrates, fertilisers, organic waste(s) and/or other inputs<br>Liming |

*Figure 20.4 Flow of heavy metals through agricultural system*

*Figure 20.5 Key factors characterising magnitude of major flows ofheavy metals through agricultural systems*

The group felt that a similar approach to that proposed for phosphorus would be appropriate, using the FADN typology and a GIS-based approach for the geographical factors. Such an approach should make use of existing data on patterns of atmospheric deposition in Europe, and the heavy metal content of fertilisers and manure. For outputs, there are models to describe movement of heavy metals into soil solution from bound forms; however, currently there are very few models for uptake by plants of heavy metals in soil solution. This is an aspect requiring further research attention (see Japenga, this volume). For sources of information, see Del Castilho et al. (1993), Driel and Smilde (1981, 1990), Groot et al. (1996), Groot et al. (1998), Römkens and Salomons (1998), and Römkens et al. (1999).

Impact assessment can then proceed on the basis of assessing potential impacts of heavy metals passing into the human food chain or remaining in the soil, which crosses the system boundary at the end of the time period under consideration (as shown in figure 20.4). Alternatively, the potential impact assessment factors can be modified to account for additional aspects affecting the actual toxic effects of heavy metals, such as those mentioned in the previous section.

## 20.4 Conclusions

Through its discussions, the working group developed a practical approach for modelling flows of substances through agricultural systems. This facilitates identification of the major flows contributing to potential environmental impacts, and subsequent modelling to account for relevant agricultural and geographical factors. Operationalisation requires development of datasets where these are missing, and integration with existing models. The approach can be extended to consider the 'cradle to grave' life cycle of foodstuffs (Cowell, 1999). Ultimately, the choice of restricted or expanded system boundaries for a study will depend upon the question(s) being asked in any given decision-making situation.

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# E. Data on other environmental aspects

# 21. Introduction

*Bo P. Weidema* [1]

Many environmental aspects of agriculture are related to the energy and substance cycles dealt with in the previous chapters. However, to obtain a complete picture of the environmental effects of agriculture a number of other aspects need to be addressed.

In this section of the book, we include three papers covering issues *not* discussed at the 2nd European Invitational Expert Seminar on Life Cycle Assessment of Food:

- pesticides;
- physical impacts on ecosystems (from land use);
- occupational health.

These papers reflect state-of-the-art in their respective areas and give recommendations for future data collection efforts.

We do not claim that an exhaustive description of all relevant environmental aspects of agriculture can be obtained, even when including the recommendations from these three papers. Issues that are not covered in this book include:

- methane emissions;
- water consumption in areas where the ground water table may be affected;
- veterinary medicine and its fate in the environment;
- animal welfare.

Methane is the subject of intense interest as a greenhouse gas, and much research is therefore performed in this area. Different models are proposed by e.g. Johnson and Ward (1996), Kirchgessner et al. (1995), Matthews and Knox (1999).

Water use may be an environmental problem in areas where the ground water table is affected. The effect of a falling ground water table on the ecosystems is similar to the physical impacts described in Cowell and Lindeijer (this volume) and it seems reasonable to suggest that this issue should be integrated in the methods and indicators suggested by these authors.

The environmental aspects of the use of veterinary medicine has received increasing attention during the last few years. The most obvious reason for concern is the issue of antibiotics resistance (see e.g. Commission on Antimicrobial Feed Additives 1997, Committee on Drug Use in Food Animals 1999, Frimodt-Moller et al. 1998, Khachatourians 1998) but concern has also been raised (see e.g. Commission on Antimicrobial Feed Additives 1997) regarding the effects of some of the involved toxic substances on the farmers' occupational health and the scarce knowledge on the environmental fate of pharmaceuticals and their metabolites (Addison 1984, Halling-Soerensen et al. 1998, Römbke

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et al. 1996). Surveys of agricultural use of medicine are few (for USA, see Dewey et al., 1997), which makes it unlikely that a model of medicine consumption can be developed at present. It seems reasonable to suggest a monitoring system for use of veterinary medicine in parallel to the system suggested for pesticide use (see Hauschild, this volume).

Animal welfare is a controversial issue in Life Cycle Assessments of agricultural products. In parallel to occupational health, it has been questioned whether it is an environmental concern to be included in Life Cycle Assessments. Even when accepting animal welfare as an issue of concern, its measurement is very complicated. And finally, animal welfare depends so much on management factors that it is very difficult to make a clear relation between animal welfare and e.g. a specific housing or feeding system. Thus, the most effective way to ensure improvements in animal welfare seems to be through a certification scheme at the individual farm level. In this respect, animal welfare may be just an extreme example of a general problem with some of the environmental parameters used in Life Cycle Assessments. Not only animal welfare, but also emissions of hazardous substances (in agriculture e.g. the handling of pesticides and medicine) and occupational health issues may vary more between individual enterprises (farms) than between processes, technologies or products. This may be used as an argument for not including these issues in Life Cycle Assessments in the traditional way, but rather as a certification requirement (e.g. of an environmental management scheme with certain minimum annual improvement criteria) for the enterprises throughout the life cycle. However, it should not be used as an argument for leaving these issues out of Life Cycle Assessments altogether. The advantage of Life Cycle Assessment is exactly that different environmental issues can be seen in proportion to each other throughout the product life cycle. Thus, such issues with a large local variation should be included in Life Cycle Assessments with a range, exactly showing how large this variation may be, thus indicating when a local, site specific assessment or certification is of importance.

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# 22. Estimating pesticide emissions for LCA of agricultural products

*Michael Hauschild* [1]

*Abstract*

Emission data for pesticides from agricultural product systems may be based on national and international pesticide usage statistics, but these only provide information on the applied dose. When the field is considered as part of the technosphere, the emissions from the system are those quantities, which reach the environment surrounding the field. The routes of emission may be direct through wind drift or indirect through evaporation, leaching, or surface run-off. Models are presented that will allow estimation of emission factors based on substance characteristics normally available for pesticide ingredients.

## 22.1 Introduction

There are a several reasons why pesticides as a substance group need special attention in the Life Cycle Assessment of agricultural products.

Firstly, they are distinguished by the fact that while most other chemicals reach the environment as an unintentional consequence of their application, pesticides are spread on purpose in parts of the biosphere to control certain life forms.

Secondly, pesticides have been designed to have strong and rather specific effects on selected groups of organisms in the environment while chemicals at large often have weaker and more unspecific effects.

Thirdly, the use of pesticides is one of the main differences between conventional and organic agriculture. For a comparative Life Cycle Assessment of the products of these two forms of cultivation, it is therefore crucial, that the impacts of the pesticides be represented well.

As a consequence of the first two characteristics of pesticides as a group, the requirements are rather strong on testing and documentation of their environmentally relevant characteristics. Pesticides are therefore among the best examined chemicals as regards properties like:

- biodegradability in different environmental compartments;
- degradability through hydrolysis and photolysis;
- formation of environmentally persistent degradation products;
- adsorptive properties and mobility in soil;
- human toxicity;
- ecotoxicity to terrestrial and aquatic species.

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This allows a more qualified - although still generic - modelling of the fate of pesticides in and outside the field.

It is the aim of this paper to present a generic procedure to be used in life cycle inventory analysis for estimation of the fate and hence the emissions of pesticides after application to a field. Parameters that depend on soil types and climatic conditions are typically chosen to represent North European conditions. The use of the procedure is illustrated through an example at the end of the paper.

## 22.2 Estimating the usage of pesticides in the product system

Pesticide formulations have dosage instructions on their labels, which might be used as an estimate of the actually applied dose for the inventory analysis of LCA. However, experience shows that it is not sufficient to assume that the farmer applies the pesticide at the label-recommended rate. In Great Britain, the average rate for application of fungicide products to wheat in 1996 was thus found to be around half the recommended rate (Thomas, 1998). It thus seems more reliable to base the collection of inventory data on regional or national usage statistics where available. In any case, since the applied rate will often influence the overall results of the study, a sensitivity analysis should be used to check the assumptions made here.

Usage statistics for pesticides are collected regularly in many European countries (e.g. Danish EPA, 1998) and have also been published on an EU level (Eurostat, 1992; Stanners and Bourdeau, 1995, EMEP/CORINAIR, 1998). The purpose of the usage statistics is primarily to support the regulation of pesticide usage and monitor the changes over time as a consequence of different measures. Usage statistics may also provide useful information for the review process of pesticide approvals and for the approval of new pesticides (Thomas, 1998). The applicability in the inventory analysis of LCA is not among the main goals of the statistics, but it may still provide a good impression of the average usage of pesticides in the studied product system, provided that the proper information is gathered together with the usage statistics:

- it is crucial that total usage can be split into the quantity applied to different individual crops. Most pesticides will be used on more than one crop and the mere collection of the total annual national pesticide usage based on production, import or sale will thus in general be of little use for LCA purposes;
- additional data must be gathered which allows the quantity used per functional unit to be deducted from the total usage figures, e.g. data on total treated area and application frequency per crop life;
- preferably, the usage statistics should allow differentiation according to regional variations in usage patterns within the area that is covered. This will allow estimation of the usage pattern under the conditions that are relevant for the product system under study. For the same crop, important regional differences may arise as a consequence of different soil types, climatic conditions, and cultivation practices.

Recently, Eurostat established a set of guidelines for the collection of pesticide usage statistics within agriculture and horticulture directed towards the national bodies responsible of collecting information on pesticide usage (Thomas, 1998). The guidelines require that the national sales and import statistics be supplemented by collection of usage statistics on individual crops. This collection may proceed through questionnaires (as currently used in the Netherlands), telephone interviews (as currently used in Sweden) or even personal visits (currently used in United Kingdom, France, Sweden, and the USA) to individual farmers.

The guideline requires that the usage statistics cover all important crops as ranked according to the total area covered by the crop, the treated area covered by the crop, the weight of pesticide applied to the crop or average application rate of pesticide to the crop.

According to the Eurostat guideline, the data to be collected comprise:

- the crop (name, development stage);
- the area grown for the crop;
- the pesticide product (rather than active ingredient - other constituents may cause harmful environmental impacts);
- the amount used or the rate of application;
- the area actually treated for the crop (if different from the area grown).

The guideline stresses the importance of breakdown of the national usage statistics on regions and on different farm size groups as usage patterns may vary widely between these.

It seems that the guidelines from Eurostat will ensure European pesticide usage statistics, which meet the needs for establishing relevant average usage estimates for life cycle inventory analysis of most agricultural crops.

## 22.3 Converting usage statistics into emission estimates

The field system is a kind of ecosystem albeit strongly manipulated by man. Nevertheless, in Life Cycle Assessment, the field system is normally considered to be part of the technosphere, i.e. the production system, rather than the ecosphere. This means that an emission (of nutrients or pesticides) is not considered an emission to the environment before it crosses the border between technosphere and ecosphere by leaving the field, unless its impact damage the productivity of the field system. As a consequence of this, LCA performed on agricultural products traditionally disregard the strong (and intentional) impacts on target organisms within the field as well as the unintended but often inevitable impacts on *non-target* organisms within the field.

The inventory analysis of an agricultural system will typically provide information on the quantities of different pesticides or active ingredients that are applied to a crop and possibly about the equipment used. This is information about quantities applied within the technosphere but it does not in itself provide insight in the quantities that are emitted. Depending on substance properties and characteristics of the cultivation system, a large or small fraction of the pesticide ingredients will cross the border of the field system and reach the different compartments of the environment as emissions.

The following sections review methods for determining redistribution factors that together allow estimation of the fractions of the applied quantity of pesticide that reach the different compartments.

### 22.3.1 Dispersion routes from the field

When applied to a crop, the pesticide can follow different routes as illustrated in figure 22.1.

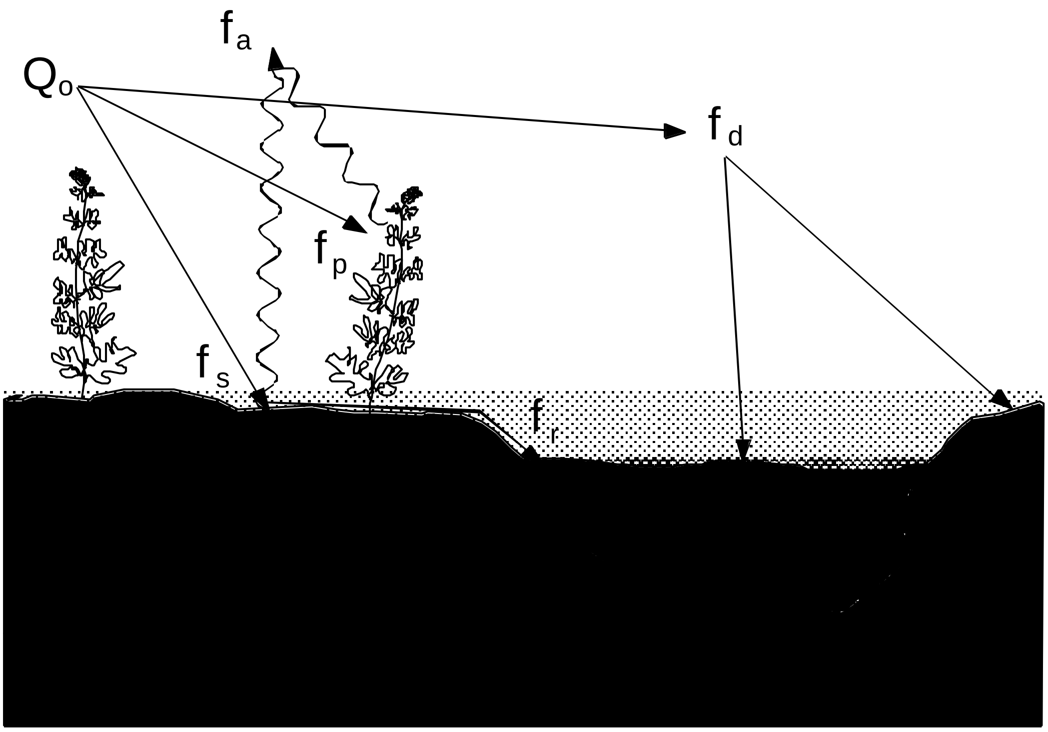


*Figure 22.1 Dispersion routes for a pesticide applied to a field crop with redistribution factors for the different routes*

The total quantity that is applied ($Q_0$) is initially divided into fractions that deposit on the crop plants ($f_p$), on the soil ($f_s$), or drift off the field as particles or vapour to reach the surrounding environment ($f_d$). Depending primarily on the properties of the pesticide ingredients, a fraction of what reaches the plants or the soil of the field may volatilise ($f_a$). From the part that deposits on the soil surface, a fraction may reach surrounding surface waters through surface run-off ($f_r$). Another fraction may leach ($f_l$) and reach the groundwater ($f_g$) or surface waters via drain pipes ($f_{dr}$) if the soil is drained.

Once the redistribution factors, $f_i$, are known, the emissions to the different environmental compartments can be determined from the usage statistics' information on $Q_0$, as:

Emission to air (particles or vapour): $Q_0 \cdot ( f_a + f_d )$

Emission to water: $Q_0 \cdot (f_{dr} + f_r)$

Emission to ground water: $Q_0 \cdot f_g$

Direct emission to soil outside the field may occur during spraying of the border of the field. It is thus determined by the shape and the area of the field. As default, it is assumed negligible while deposition of wind drifting or volatilised pesticide is considered.

The dispersion of pesticide through the different routes depends on application techniques, characteristics of the field-crop system and meteorological conditions. In the following sections, a review is given of methods to determine redistribution factors for estimation of pesticide emission from usage statistics for use in LCA.

### 22.3.2 Wind drift

Wind drift is the dispersion outside the field of pesticide in the form of wet and dry particles that have not yet reached the crop or the field soil. It occurs immediately after the pesticide has left the spraying nozzle and its extent is influenced by the application technique, the distance from the edge of the field, the morphology of the crop and the local meteorological conditions at the time of spraying. It is less dependent on the physical and chemical characteristics of the pesticide.

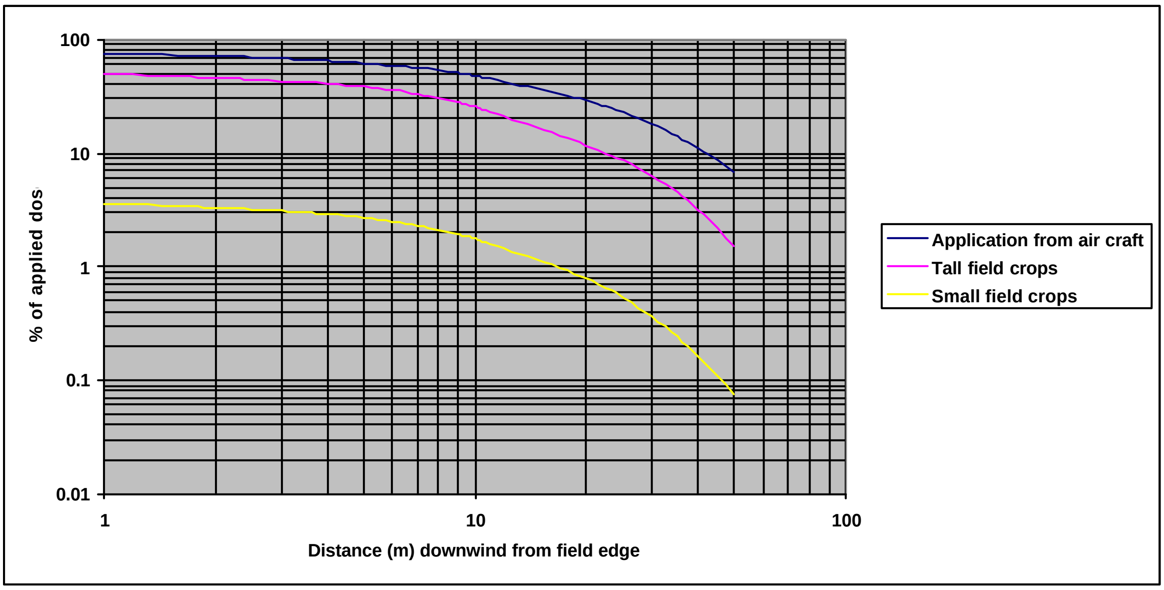


*Figure 22.2 Deposition curves showing the fraction of pesticide deposited after wind drift as a function of the distance from the edge of the field for three different application scenarios: Small field crops (lower than 1 m, lower line), tall crops (bushes and trees taller than 1 m, middle line) and application from an aircraft (upper line); an uncertainty factor of +/-20% should accompany any value taken from either curve*

Source: EPPO (1996).

The fraction that leaves the field system through wind drift ($f_d$) may be estimated using a model developed by the European and Mediterranean Plant Protection Organisation, EPPO for risk screening of pesticides (EPPO, 1996). For different application techniques and crop morphologies, the model predicts the fraction of the applied pesticide that will deposit with increasing distance from the edge of the field. Results of the model are shown in figure 22.2.

For a known application scenario, $f_d$ can be estimated using figure 22.2 and assuming an average distance from the edge of the field to the nearest terrestrial or aquatic ecosystem. It is clear from the figure that the extent of deposition from wind drift decreases strongly with the distance from the edge of the field. In Denmark, the distance from the field to a stream must be at least 2 m. The model predicts that up to 3.2% of the applied quantity will drift this far for application to field crops. Terrestrial ecosystems will per definition start just outside the field. In a distance of 1 m $f_d$ will assume a value up to 3.5% or 0.035 for field crops.

The deposition curves in figure 22.2 have been calculated for a 'realistic worst case' situation, i.e. all parameters of the model have been set at that value within their normal range that results in the largest predicted wind drift. The figure will thus tend to give a conservative estimate of the wind drift, which may be unwanted in LCA.

For comparison, the USES-model that was developed for risk screening of chemicals within EU comprises a module for assessment of pesticides (Jager and Visser, 1994, Emans et al., 1992). Here, the applied value for $f_d$ lies in the range of 0,01-0,1 for field crops depending on the application.

### 22.3.3 Deposition on crop plants and field soil

The substance that reaches the field system is divided between the crop plants and the field soil. The relative partitioning between the two compartments is determined predominantly by the crop species and growth stage. The more extensive the foliage, the larger the fraction ($f_p$) that is intercepted by the crop and hence the less the fraction ($f_s$) that will reach the soil of the field. $f_s$ can be expressed as a function of the leaf area index L which is defined as the total leaf area over the field divided by the area of the field ($m^2/m^2$). For average application conditions, an expression is given by Gyldenkærne et al. (1999):

A crop of oilseed rape will at the time of bloom have a leaf area index of 5-7 (Gyldenkærne, 1999) giving $f_s$ a value of 0.03-0.08 according to this expression.

$$f_s = e^{-0.5 \cdot L}$$

$f_p$ can be determined from $f_s$ and $f_d$ since the three must sum up to one.

$$f_p + f_s + f_d = 1$$
$$\Leftrightarrow \quad f_p = 1 - (f_d + f_s)$$

For comparison, the pesticide module of the USES model gives default values for $f_p$ and $f_s$ representative of different growth stages of various field crops (assuming an $f_d$-value of 0.1):

*Table 22.1 Suggested values in the USES model for the pesticide fraction, which is intercepted by the crop, and the fraction that reaches the field soil*

| Crop | Growing stage | Fraction intercepted by crop, $f_p$ | Fraction reaching the soil, $f_s$ |
|---|---|---|---|
| Potato or beet | 2-4 weeks | 0.2 | 0.7 |
| | full growth | 0.8 | 0.1 |
| Peas | shortly after emergence | 0.1 | 0.8 |
| | around bloom | 0.7 | 0.2 |
| Corn | 1 month | 0.1 | 0.8 |
| | full growth | 0.5 | 0.4 |
| Grassland | | 0.4 | 0.5 |
| Sprouts | full growth | 0.7 | 0.2 |
| Onions | full growth | 0.5 | 0.4 |

Note: Jolliet et al. (1998) assume as a general default value that 85% of the applied quantity of pesticide is deposited on the field soil, i.e. $f_s = 0.85$.
Source: After Jager and Visser (1994).

### 22.3.4 Volatilisation

If ingredients of the pesticide are sufficiently volatile, they may evaporate after reaching the crop plants or the soil of the field. The extent of volatilisation also depends on local meteorological conditions at the time of application, notably the temperature and wind velocity.

The fraction of the initially applied dose that volatilises ($f_a$) can be expressed as the sum of fraction that volatilises ($f_{sa}$) upon reaching the soil and the fraction that volatilises ($f_{pa}$) upon reaching the crop plants:

$$f_a = f_{sa} \cdot f_s + f_{pa} \cdot f_p$$

*Table 22.2 Evaporation rates for pesticides on soil as determined by the volatility of the substance*

| Volatility | Vapour pressure, Pa | Daily loss $\alpha_s$ through evaporation (fraction of $f_s$), $d^{-1}$ |
|---|---|---|
| High | $> 10^{-1}$ | 0,50 |
| Low | $10^{-3}$-$10^{-1}$ | 0,10 |
| Not volatile | $< 10^{-3}$ | 0,01 |

Source: EPPO (1996).

The evaporation from soil and plants can be determined by a model developed by EPPO as part of a risk screening model for pesticides (EPPO, 1996). Table 22.2 and 22.3 give recommended values for the daily evaporation losses from soil respectively plant surface as a function of the vapour pressure of the substance.

*Table 22.3 Evaporation rates for pesticides on crop surfaces as determined by the volatility of the substance*

| Volatility | Vapour pressure, Pa | Daily loss $\alpha_p$ through evaporation (fraction of $f_p$), $d^{-1}$ |
|---|---|---|
| High | $> 10^{-3}$ | 0,50 |
| Low | $10^{-5}$-$10^{-3}$ | 0,25 |
| Not volatile | $< 10^{-5}$ | 0,10 |

Source: EPPO (1996).

In the determination of $f_{sa}$ and $f_{pa}$ from this information, the evaporation process is considered to follow a first order kinetics with $\alpha_s$ and $\alpha_p$ from Table 22.2 and 22.3 respectively as rate constants. If $\tau_s$ and $\tau_p$ are the expected residence times in the soil and on the crop plants respectively, $f_{sa}$ and $f_{pa}$ can be determined as:

$$f_{sa} = 1 - e^{-a_s \cdot t_s}$$
$$f_{pa} = 1 - e^{-a_p \cdot t_p}$$

The residence time in soil ($\tau_s$) is generally determined by the microbial degradation rate of the substance which must be known to obtain a permit for the pesticide in many countries. The residence time of the pesticide on the crop ($\tau_p$) is typically determined by its rate of photolysis or photochemical oxidation. Also this information is available for most pesticides.

For comparison, Jolliet et al. (1998) assume in their model for life cycle impact assessment of pesticides that as an average approximately 10% of the applied substances remain in the air (wind drift) or return to the air upon volatilisation. This means that in their model, $f_d + f_a = 0.1$.

In their Emission Inventory Guidebook, EMEP/CORINAIR provide emission factors for a small group of the most environmentally problematic pesticides (many of which have today been banned for use in Europe). The emission factors are derived from the vapour pressure of the active ingredients and give values for $f_a$ ranging from 0.05 to 0.95 with an uncertainty of a factor 2-5 (EMEP/CORINAIR, 1998).

### 22.3.5 Surface run-off

In case of precipitation the substance that reaches the soil may experience surface run-off with rainwater in dissolved form or absorbed to soil particles. The extent of surface run-off must thus be expected to depend on the properties of the substance:

- water solubility and sorptive properties influence how much can be carried with the water; and

- substance degradability and volatility influence how much is still on the soil surface when the hydrological conditions that allow surface run-off arise as a consequence of heavy rain fall or melting of snow on frozen fields.

In addition, the slope of the field has a strong influence on the extent of surface run-off.

No attempts have been found to express the dependence of that fraction which undergoes surface run-off as a function of the properties of the substance. Under Danish conditions, where most fields are flat and horizontal and water erosion of the fields is a minor problem, the surface run-off is dominated by the transport of dissolved substance. Based on empirical data from monitoring of Danish fields (Felding et al., 1997), a default value of $f_r = 0.0001$ is suggested. This value is not expected to be representative of more hilly country.

### 22.3.6 Leaching

Leaching is important as a transport route to surface waters and ground water. In regions with a precipitation surplus, water movement in the soil will leach substances from the top of the soil to the deeper layers. Leaching can be seen as the combination of percolation through the soil and preferential transport through the macropore structure of the soil.

Percolation allows dissolved substance in the soil liquid to interact in a sorptive manner with the solid phase of the soil as the liquid phase moves downwards. The fraction ($f_l^*$) that reaches deeper layers of the soil through percolation is thus strongly influenced by the substance's sorptive equilibrium between soil liquid and soil particles.

Preferential transport takes place through soil macropores in the form of cracks and other voids e.g. created by biological activity or decay of plant roots. Macropores are created and destroyed continuously in the soil showing the highest stability in clayey soils. Through macropores the water moves much quicker downwards than through percolation and the fraction ($f_l^{**}$) of a substance that is transported through macropores is hence not to any significant degree influenced by the sorptive properties of the substance.

Leaching is also influenced by the substance's propensity to undergo microbial degradation. The dominant degradation capacity is present in the ploughing layer in the top 30-40 cm of the soil, and in the fate-modelling, degradation below this depth is generally disregarded. The degradability is very important for the substance's probability of percolative transport, while the transport through macropores is so quick that degradation will be of minor significance once the hydrological conditions allowing macropore transport are present.

Parameters influencing the potential for leaching of pesticides in soil are thus:

- characteristics of the pesticide:
  - water solubility;
  - sorptive properties;
  - persistence in the top layer of the soil;

- meteorological conditions. The duration between application of the pesticide and the first precipitation event is decisive. The longer the time, the larger the fraction that will be degraded, the stronger the sorption of the residual to soil particles and the lower the potential for leaching;
- soil texture. The coarser the soil texture, the quicker the leaching and the larger the fraction that leaches from the ploughing layer through percolation. Percolation is thus quicker through sandy soils than through clayey soil. Particularly for clayey soils, the existence of macropores may enhance leaching substantially because percolation is slow and at the same time the macropore structure is stable compared to sandy soils. The higher the clay content, the more important the macropore transport (DHI, 1996).

For estimation of leaching, the USES model for risk screening of chemicals draws on the PESTLA model that estimates the fraction of a substance which will leach below 1 m depth in the soil through percolation under typical Dutch conditions (soil type and precipitation). The PESTLA model determines the fraction ($f_l^*$) that percolates from knowledge of the experimentally determined half life of the substance in soil and its adsorption coefficient $K_{oc}$ to the organic material in the soil. High $f_l^*$-values of nearly 0.5 are found for substances that combine a long half life (500 days) and a low adsorption coefficient (10-20 l/kg), while low $f_l^*$-values are found for substances displaying a combination of high biodegradability and strong sorption (Jager and Visser, 1994).

With its weaker dependence on substance characteristics, preferential transport through macropores can be important for many substances, particularly on soils of fine texture. The extent of preferential transport will be governed by the soil texture, the frequency of macropores and the probability of precipitation occurring shortly after application of the pesticide (particularly for short-lived pesticides). It should thus be expected to vary over the year with a maximum during autumn. No general models have been found, but model simulations of preferential transport of selected pesticides on different Danish soil types have been performed at the Danish Hydrological Institute (Thorsen, 1995). The results show that on a sandy loam, the fraction undergoing macropore transport ($f_l^{**}$) reaches an autumn maximum of between 0.001 and 1.4%. During spring, the values are in the interval 0-0.04%.

A default value of $f_l^{**}$ = 0,001 may be appropriate for Danish conditions with predominantly clayey soils. On more sandy soils $f_l^{**}$ will be lower as macropores will be less stable.

The fraction that leaches from the ploughing layer is determined as the sum of percolation and macropore transport:

$$f_l = f_l^* + f_l^{**}$$

In case, the field is drained, the leaching substance may through drain pipes be directed to surface waters, typically streams. If the frequency of drainage is δ the fraction that goes with drainage water to surface waters can be determined as:

$$f_{dr} = \delta \cdot (f_l^* + f_l^{**})$$

If it is assumed that all degradation occurs in the top layer of the soil, the fraction that reaches ground water can be determined as that part of the leaching fraction which is not drained off:

$$f_g = (1 - \delta) \cdot (f_l^* + f_l^{**})$$

For Denmark, δ assumes an average value of 0.55 (Nielsen, 1997).

### 22.3.7 Emissions to surface water, groundwater and soil

For calculation of the overall fractions of the applied dose that reach the different environmental compartments outside the field system, it is necessary to assume how large a part *a* of the surface to which deposition occurs, that is covered by aquatic systems.

The overall fraction that reaches surface water outside the field system processes is determined

$$f_w = a \cdot (f_d + f_a) + f_r + f_{dr}$$

as:

The overall fraction reaching soil outside the field system processes is determined as:

$$f_s = (1 - a) \cdot (f_d + f_a)$$

The overall fraction reaching the groundwater compartment is $f_g$.

In Hauschild et al. (1998a), a relative frequency of water systems of 0.2 is proposed as a global default for *a* for land-borne activities.

### 22.3.8 Persistent degradation products

Sometimes, it is not the active ingredient itself, but rather a degradation product that is of environmental concern. If degradation of a pesticide ingredient on the way to full mineralisation proceeds through formation of a stable intermediary compound, which may leave the field system as an emission to the environment, this emission should also be estimated and enter into the inventory of the product system.

## 22.4 Life cycle impact assessment of pesticides

The redistribution factors presented above do not necessarily represent the final fate of the substances. The airborne fraction may undergo physical and chemical degradation (photolysis, photochemical oxidation, hydrolysis) while in the air compartment, deposition to the soil or surface water compartment, and on plant and soil surfaces and in the soil and water compartments it may be subjected to further microbial degradation.

The intention of the presented redistribution factors is merely to allow estimation of the quantities that reach the main environmental compartments: air, surface water, soil and ground water and thus help convert usage statistics into inventory data that may serve as input to the next phase of LCA in which the potential impact of the pesticides on the environment is determined.

The main anticipated impacts are toxicity to humans and to the exposed ecosystems as a consequence of direct as well as indirect exposure (through food chains). As mentioned in the introduction, pesticides are - as a group - among the best examined chemicals with regard to their final environmental fate and their potential effects, also to non-target organisms. This means that any method developed for life cycle impact assessment of human and ecotoxicity of chemicals will be feasible for most pesticides based on the existing and available data. Several methodologies exist to model the further environmental fate of the pesticide ingredients and the reader is referred to these for the impact assessment of the pesticide emissions (Guinée et al., 1996; Jolliet et al., 1998; Jolliet and Crettaz, 1997; Wenzel et al., 1997; Hauschild et al., 1998a,b).

## 22.5 Example of estimation of pesticide emissions

For illustration of the proposed procedure, pesticide emissions are estimated for the application of the herbicide Kerb in a oilseed rape field. The applied formulation of Kerb contains 500 g/l Propyzamide and 57 g/l ethylene glycol (Danish EPA, 1992). It is applied in the field during early spring when the crop has a leaf area index L of around 3 (Gyldenkærne, 1999).

For the active ingredient, propyzamide, the following substance-specific data is required:

- vapour pressure: $1{,}1a*10^{-2}$ Pa;
- half life determined by photolysis: 2 d;
- half life in soil: 33 d;
- adsorption coefficient to organic material in soil: $K_{oc} = 1587$ (Danish EPA, 1992).

Using these data, the redistribution factors are determined below using the proposed procedure.

*Wind drift*

In the classification of figure 22.2, oilseed rape qualifies as a small field crop. Assuming a distance of 1 m from the edge of the field, 3.5% of the applied dose will deposit after wind drift, i.e. $f_d = 0.035$.

*Deposition on field soil and crop plants*

The fraction that reaches the soil within the field is determined by the leaf area index L of the crop as: $f_s = e^{-0.5a*L}$. Given a crop leaf area index of 3, the fraction that deposits on the soil within the field can thus be estimated as: $f_s = e^{-0.5a*3} = 0{,}223$.

Since wind drift outside the field, deposition on soil and deposition on plants must sum up to one, the fraction that deposits on the crop plants can now be determined as $f_p = 1-(e^{-0.5a*3}+0{,}035) = 0{,}742$.

*Volatilisation*

The fraction that is lost daily through volatilisation from field soil or crop plants is determined from knowledge of the vapour pressure of the substance, using Table 22.2 and 22.3. With a vapour pressure of 1,1a*10$^{-2}$ Pa the volatility of propyzamide from soil is qualified as low with a typical daily loss fraction of 0.1 (table 22.2). Its volatility from crop plants is high and the typical daily loss fraction is 0.5 (table 22.3).

The fraction $f_{sa}$ that volatilises from the soil can be determined from the daily loss fraction and the residence time in soil. A typical half life of propyzamide in soil is 33 d giving a residence time of 33/ln(2) d = 48 d. From this information, $f_{sa}$ can be determined as:

$$f_{sa} = 1 - e^{-0.1 \cdot 48} = 1 - 0.008 = 0.992$$

Apart from volatilisation, the residence time of propyzamide on crop leaves is determined by photolysis giving a half life of 2 d equivalent to a residence time of 2/ln(2) = 2.9 d. This allows $f_{pa}$ to be determined as:

$$f_{pa} = 1 - e^{-0.5 \cdot 2.9} = 1 - 0.235 = 0.765$$

The fraction of the applied dose which undergoes volatilisation is thus:

$$f_a = 0.223 \cdot 0.992 + 0.742 \cdot 0.765 = 0.789$$

*Surface run-off*

The fraction undergoing surface run-off is determined as a fixed value, $f_r = 0.0001$.

*Leaching*

Of the part of the applied pesticide that deposits on the field soil, a fraction will leach and reach the ground water, either through percolation or through preferential transport via macropores. Given the relatively short residence time in soil (a half life of 33 d) and a strong adsorption to the organic material of the soil (expressed through a $K_{oc}$ value of 1587), PESTLA predicts the fraction of the applied propyzamide that leaches through percolation to be $f_l^* = 0.00$ (Jager and Visser, 1994). If the field soil is a sandy loam, the fraction that undergoes macropore transport is $f_l^{**} = 0.001$ and the total fraction undergoing leaching is $f_l = f_l^* + f_l^{**} = 0.001$.

If the frequency of draining is ∗ = 0.55, the fraction that will reach surface water through drain pipes is $f_{dr} = 0.55a*0.001 = 0.00055$ while the fraction reaching ground water is
$f_g = (1-0.55)a*0.001 = 0.00045$.

*Emissions to surface water, ground water and soil*

For the inventory, the overall fraction of the applied dose that reaches surface water is calculated as:

$$f_w = 0.2 \cdot (0.035 + 0.789) + 0.0001 + 0.00055 = 0.165$$

The overall fraction reaching soil ecosystems outside the field system processes is determined as:

The overall fraction reaching the groundwater compartment is $f_g = 0.00045$

$$f_s = (1 - 0.2) \cdot (0.035 + 0.789) = 0.659$$

From the calculations, it is clear that the predominant route to the surrounding environment for an active ingredient with the properties of propyzamide is via volatilisation and re-deposition.

In addition, propyzamide has a stable degradation product which may have a potential for leaching (Miljøstyrelsen, 1992). The required substance-specific data have not been found for this metabolite and it is therefore not considered in this example although it should be included in a real life inventory analysis.

## 22.6 Variables

| | |
|---|---|
| $\alpha_s$ | Daily loss of pesticide ingredient through evaporation from soil surface ($d^{-1}$) |
| $\alpha_p$ | Daily loss of pesticide ingredient through evaporation from crop surface ($d^{-1}$) |
| $\tau_s$ | Expected residence time of the pesticide in the soil |
| $\tau_p$ | Expected residence time of the pesticide on the crop plants |
| $\delta$ | Frequency of drainage |
| a | Relative share of deposition area covered by aquatic systems |
| $f_a$ | Fraction of initially applied dose which volatilises from the crop or field soil |
| $f_d$ | Fraction of initially applied dose which reaches areas outside the field through wind drift |
| $f_{dr}$ | Fraction of that part of the initially applied dose which leaches and through drain pipes reaches surface waters outside the field |
| $f_g$ | Fraction of that part of the initially applied dose which leaches to reach the ground water |
| $f_l$ | Fraction of initially applied dose which leaches from the top-layer of the field soil |
| $f_l^*$ | Fraction of initially applied dose which leaches through percolation |
| $f_l^{**}$ | Fraction of initially applied dose which leaches through preferential transport |
| $f_{sa}$ | Fraction of that part of the initially applied dose which reaches the field soil which later volatilises |
| $f_p$ | Fraction of initially applied dose which deposits on the crop plants |
| $f_{pa}$ | Fraction of that part of the initially applied dose which reaches the crop plants which later volatilises |
| $f_r$ | Fraction of initially applied dose which leaves the field through surface run-off |
| $f_s$ | Fraction of initially applied dose which deposits on the soil of the field |
| $f_t$ | Overall fraction of initially applied dose which ends up in terrestrial ecosystems outside the field |
| $f_w$ | Overall fraction of initially applied dose which ends up in aquatic ecosystems outside the field |

$K_{oc}$ Adsorption coefficient expressed relative to the organic matter content of the soil (l/kg)
L Leaf area index defined as the total leaf area over the field divided by the area of the field ($m^2/m^2$)
$Q_0$ Initially applied dose per functional unit

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# 23. Impacts on ecosystems due to land use: Biodiversity, life support, and soil quality in LCA

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*Abstract*

With a focus on data for agriculture, this paper deals with data for impact assessment of land use in LCA. The methodological framework of assessing land use impacts is given, making a distinction between traditional inventory data and data for impact assessment. For the impact assessment, three stages are discussed: selection of relevant end-points, choice of representative category indicators and integrating indicators into a quantitative impact assessment. Special attention is given to the possible scales of analysis and the implication of the above for agricultural data collection, resulting in a more or less prioritised list of indicators for agriculture.

## 23.1 Introduction

Incorporation of land use impacts into LCA has been a subject of considerable discussion in the last few years; a number of methods have been suggested but no one approach has been accepted as the preferred alternative. In fact, a major cause for its late inclusion in the LCA methodology has been the difficulty of making this impact category operational. The availability or, more appropriately, non-availability of data is a particularly important issue in assessment of land use impacts, and has influenced development of methods. Therefore, this paper gives an overview of the common issues related to land use impacts and, in particular, those relevant for agriculture.

Section 2 describes the generalised methodological approach for assessing land use impacts and the corresponding data requirements. One particular aspect whose consideration is driven by data limitations is the scale of analysis used in assessing physical habitat depletion, and this is discussed in section 3. The implications for agricultural data are summarised in section 4, with a conclusion relevant for the LCANET Food working group on data in section 5.

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## 23.2 Methodological Approach

The two main phases in LCA relevant to the issue of land use impacts are inventory analysis and impact assessment. At *inventory analysis* (process data collection), several researchers have noted the need to account for two aspects of land use (Lindeijer et al., 1998):

- occupation of land area;
- change in land use quality.

These two aspects are illustrated in figure 23.1 below. The horizontal axis depicts the course of time and the vertical axis depicts the quality (change). The change aspect is measured as the vertical area (in the 3D picture at the final quality line) showing the difference between the initial and the final state, and the occupation is measured as the 'body' enclosed by the lines between the initial and the final state.

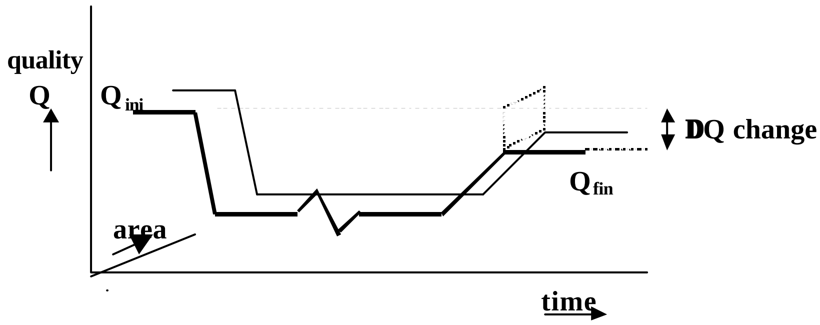


*Figure 23.1 Land use typology*

Occupation is measured in 'hectare-year' (ha*y) units, or as ($m^2$*y). Additional information may be added at Impact Assessment, and the additional data required depends on the impact assessment approach applied. The core impact of 'occupation' is the competition for (scarce) land area with other possible uses. This land use always has a time dimension, as it is about land occupied for one type of use during a certain time, excluding other uses. Examples are $m^2$*y of forestry, agriculture or roads for a certain output (x $m^3$ roundwood, y kg crops or z km*kg of goods transported, respectively). Thus occupation may be linked to qualitative statements on the type of land use in the Impact Assessment phase of LCA, or assessed quantitatively using the same indicators as for land use change (see below).

Land use change is measured in area units ([$m^2$]), and is thus without a time dimension. The core impact here is the change in quality of the land. This change is independent of the time required to perform the change, as this aspect only considers the impacts of the quality difference (the time spent to perform the change should be taken into account via occupation). One class of land use change may be a distinct change from one land use to another (say from forestry to agriculture, or from agriculture to industrial area). This change should be allocated to the output causing the change, based on yearly trends (see Lindeijer et al., 1998 for dealing with this problem). Another class of land use changes is about the more subtle changes, which occur due to continuous management practices with long-term impacts. An example is the decline in organic matter content of agricultural soil over a number of years due to practices such as crop rotations that exclude green manure crops or grass leys. The allocation principle is the same here, although quantitative data may be more difficult to determine.

To determine the quality of the occupied or changed land use is a more complex issue because it is not immediately obvious exactly what quality information is relevant for the *impact assessment* phase of LCA. This assessment requires consideration of the end-points of the analysis: what is the quality we want to assess? In fact, data requirements at inventory analysis must be defined by consideration of the Impact Assessment phase. For instance, it is not straightforward that $m^2$*y or $m^2$ are the most important data for the impact assessment. Additional quality information linked to these $m^2$*y or $m^2$, or even without multiplying with the area, may be as important or even more important. In this section, we focus on assessment of this quality aspect, and the associated data requirements.

Development of methodology for assessing the impacts of land use takes place in three stages. As noted above, the first stage involves selection of relevant end-points affected by occupation or change in land use. In general, endpoints are related to basic environmental concerns, such as ecosystem quality, human health, or more general human welfare. Several end-points have been suggested for land use, including impacts on biodiversity, life support, productivity, abiotic resources and aesthetics (based on Steen and Ryding, 1992, and Udo de Haes, 1999). Once these end-points have been defined, relevant indicators are chosen to represent the value of different ecosystems in relation to these end-points (the second stage). The third stage involves integration of the relevant indicators into equations used to calculate results for one or more impact assessment categories. This integration generally requires a weighting step, when more than one indicator is chosen. These three stages are discussed below.

### 23.2.1 Selection of relevant end-points

As noted above, relevant end-points may be impacts on biodiversity, productivity, abiotic resources, life support, and aesthetics. For impacts on *biodiversity*, it is important to distinguish between those impacts already assessed in conventional LCA methodology (i.e. resulting from emissions to air, water and land) and ones currently omitted from the conventional methodology (i.e. related to physical impacts in habitats). For land use, a method is required that accounts for impacts on biodiversity due to physical interventions in habitats as opposed to pollution in order to avoid double-counting (see Cowell, 1998). *Productivity* refers to the ability of the soil to support, for instance, agricultural production. Cowell (1998) suggests that loss of soil should be related to the end-point *abiotic resources*, emphasising its irreversible character. In Udo de Haes (1999) it is considered as part of the degradation of life support functions. *Life support* refers to processes in the natural environment which have broad regulation functions, such as cycling of nutrients and generation of stable microclimates (see Udo de Haes et al., 1999).

With regard to LCA methodology, the aspects not considered here are related to *aesthetics* (landscapes), cultural and historic values. These are all part of the cultural environment that is created by human society, and have not been main concerns for LCA up to now, due to the focus of most LCA methodology on the 'natural' environment, resources and human health. A generally applicable methodology for these aspects seems hard to establish. Only the fact that diversity of landscapes is appreciated can be mentioned, but no further approaches have been proposed for this within LCA.

Merely as an illustration, the relationship between environmental interventions, indicators, endpoints, and safeguard subjects (or areas of protection) for land use as sketched roughly in (Lindeijer et al., 1998) is shown as figure 23.2. A similar scheme for all impact categories is given in (Udo de Haes et al., 1999).

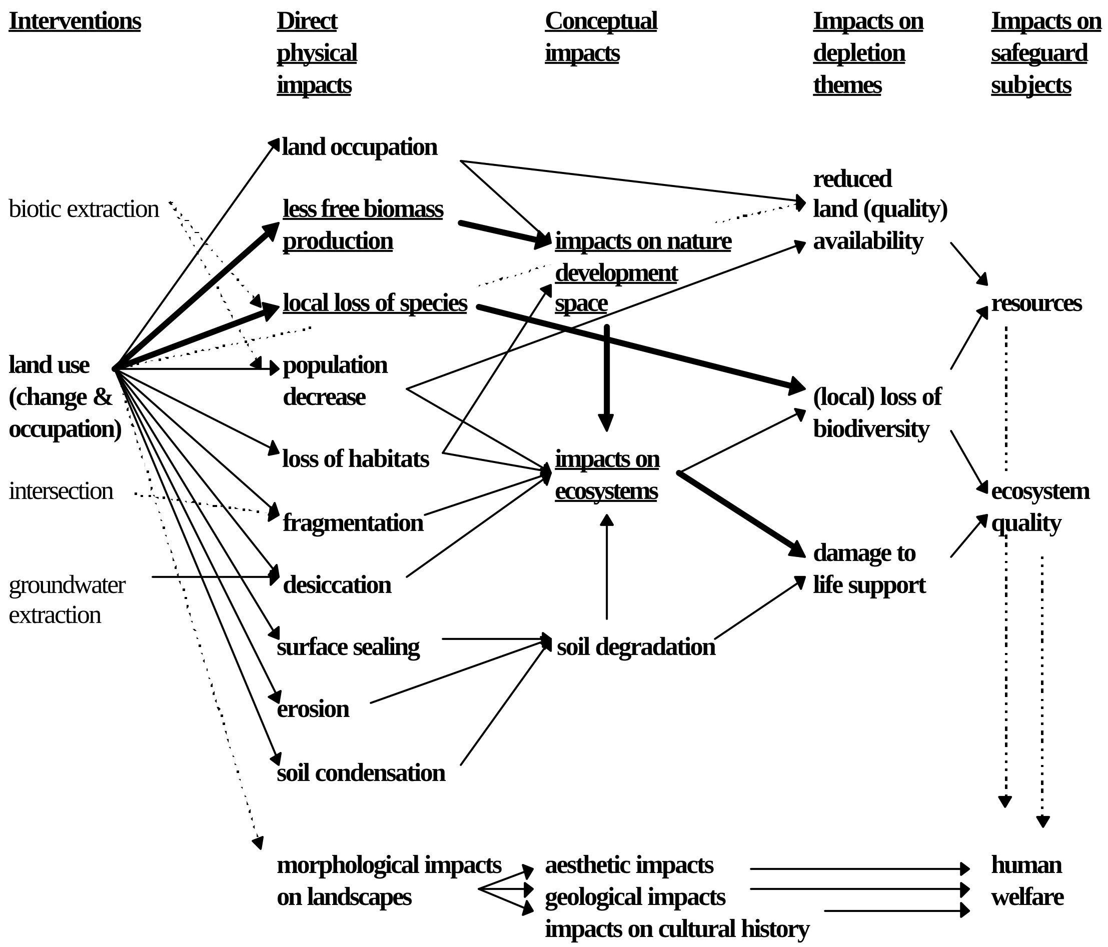


*Figure 23.2 Rough overview of the cause-effect chain related to land use*

### 23.2.2 Choice of relevant indicators for end-points

Various researchers have proposed a number of indicators for the different end-points described in the last section. For impacts on the physical habitat aspect of *biodiversity,* Lindeijer et al. (1998) have suggested starting with one indicator for biodiversity, namely number of vascular plant species per unit area. This seems a good indicator for most other aspects of biodiversity and data seem fairly well available for this indicator. On a global scale, rough data are collected in (Lindeijer et al., 1998) and some other studies. More detailed data are available in several European countries and are currently being collected for a Ph.D. study on biodiversity indicators for land use impacts (Koellner, 1999). Cowell (1998) uses four biodiversity indicators for each ecosystem type to assess physical habitat depletion. The indicators are: area, number of listed rare species, number of species, and number of individuals (measured by the Net Primary Productivity (NPP as biomass)) of each ecosystem. Data requirements for these indicators currently constrain their operationalisation. Specifically, research is required to identify up-to-date sources of data on areas of different ecosystems on a global scale, and to quantify the number of listed rare species found in these ecosystems. Also, data on the total number of species in each ecosystem are unknown in detail, and so alternative data must be identified for this aspect (a first guess is given in Cowell, (1998)).

For impacts on *productivity*, Cowell (1998) has developed two additional indicators: an Organic Matter Indicator, and a Soil Compaction Indicator. Recommendation of a particular indicator is partially based on the types of data likely to be available in an LCA study. Examples are data on types of machinery and their field times, as opposed to other indicators requiring more detailed data relating to wheel widths, tyre inflation pressure, and so on.

For impacts on *abiotic resources* - in this case, loss of soil is proposed to fit under this end-point - Cowell (1998) suggests that loss of soil can be treated in the same way as depletion of other abiotic resources. For example, using the method of Lindfors et al. (1995), loss of agricultural soil is assessed in relation to global reserves of agricultural soil. Actual soil losses can theoretically be quantified using the Revised Universal Soil Loss Equation (RUSLE) (see, for example Renard et al., 1994), although it is unlikely that most LCA studies will have the data required for this equation. Instead, it may be more appropriate to use existing data on quantities of soil eroded in different areas to predict losses due to alternative agricultural practices.

The competition over land as a flow resource may also be considered under abiotic resource depletion, resembling the competition of available surface water (see Udo de Haes et al., 1999). The $m^2$*year of land use themselves may be indicators for this. A quality factor may be added, to express the extent to which competition with nature occurs, if this quality factor expresses the extent to which nature is suppressed during the land use.

For land use impacts on *life support*, Lindeijer (1998) has developed an indicator based on the free net primary production of biomass (fNPP). fNPP is a measure of the amount of biomass left for organic material cycling in ecosystems and to contribute to the development of nature. The amount of biomass taken off the land in agri- or silviculture is therefore subtracted. Impact indicators on soil (see above) may also reside under this heading.

All these approaches have been developed in full realisation of the fact that such indicators are very crude estimates of the impacts which require assessment, and that the LCA modelling of land use impacts is necessarily poor due to lack of scientific knowledge and data.

Earlier approaches did not express their indicators in terms of the above end-points. Some were similar to the above biodiversity indicators (see Feldman and Glod, 1996). Others were close to the above life support indicators (Knoepfel, 1996). Finally, mere classification of the land use has been proposed in various forms, all based on the concept of 'Hemerobiestufen' (naturalness). None of these considered the distinction between occupation and change, and most are not based on scientific measurable, continuous scales, which are considered necessary for assessment in LCA (see section 3). See (Lindeijer et al., 1998) for a more extensive literature overview.

### 23.2.3 Integration of indicators in assessing land use impacts

Once indicators have been selected, two further issues concern a) relative weighting of these indicators if they are to be integrated into a single scale of analysis for different ecosystems, and b) development of equations used to calculate the results. These issues are then related to one or more Impact Assessment categories representing occupation and/or change in land use.

For example, the approach for biodiversity developed by Cowell (1998) requires relative weighting of the four indicators used in assessing the physical habitat value of different ecosystems (issue (a)), followed by choice of an appropriate equation to represent occupation or change in land use for the Physical Habitat Depletion impact category (issue (b)). The relative weightings of the indicators should be representative of the magnitude of the contribution of each indicator to the end-point, and their definition requires the involvement of experts in the assessment of biodiversity. For the development of equations, it is questionable whether current levels of scientific knowledge about biodiversity are a sufficient basis upon which to define and make use of these equations.

In Lindeijer et al. (1998) no relative weighting of the biodiversity and the life support indicators is proposed due to the same lack of scientific knowledge. For both end-points, however, similar formulas have been developed to quantify the impacts separately for change and occupation. The one for life support is in absolute terms whereas the one for biodiversity is in relative terms.

Some other methodological studies operationalise quite different indicators (see, for example, Baitz et al., 1998, and Schweinle, 1998). Data collection is here even more a severe problem, and weighting is not performed due to the same lack of scientific justification as in the above approaches. It is an issue for debate whether current levels of scientific knowledge about biodiversity are sufficient to make the first steps in application using the more simple approaches, allowing for further expansion of the number of indicators when data are available.

The last issue under this heading relates to the linking of $m^2$ or $m^2*y$ to the quality indicators discussed above. When the $m^2$ or $m^2*y$ is multiplied with the quality indicator scores, an aggregation of the area/time and quality aspect(s) is performed. In former studies and in the more recent (Cowell, 1998), (Lindeijer et al., 1998) and (Udo de Haes et al., 1999), this is the generally applied approach. The $m^2$ or $m^2*y$ is considered part of the inventory data, to which an equivalency factor (the quality score) is applied for characterisation in the Impact Assessment phase.

However, it can be argued that for land occupation (competition; depletion of land availability), the $m^2$*y is clearly part of the impact assessment, as it is exactly these $m^2$ and the use time, which cause land availability to be depleted. When additional quality aspects are added to be more specific on what kind of land is less available, it is not obvious that the $m^2$*y and the quality aspects should be weighted equally by multiplication. This issue is especially relevant for LCAs where wood or other renewable resources are compared with use of fossil fuels or minerals in delivery of the functional unit. Renewable resources can easily require more than a factor of one hundred more $m^2$*y per functional unit compared with fossil fuels or minerals.

A related problem with multiplying area and time aspects with quality aspects is the different position of the $m^2$ and years related to traditional inventory data. For both occupation and change, the land use is often a deliberate choice, whereas emissions are generally not. The quality impacts are often less deliberate, but for land use changes still quite explicitly included in the decision making for the change in most cases. This makes the traditional LCA aggregation by multiplication questionable. However, this issue of combining the different types of information is not yet settled.

Nevertheless, it seems quite clear that concerning data requirements both the $m^2$ or $m^2$*y data and the quality-related information are to be gathered in the inventory analysis. The types of quality information needed are still under discussion and also depend on the application, as is discussed further in the next section.

## 23.3 Scale(s) of analysis for assessing impacts on biodiversity

In the Impact Assessment categories commonly used in LCA, a single scale of analysis is used for assessing each type of impact. For example, all emissions of global warming gases are assessed relative to the global warming potential of carbon dioxide, and all emissions of acidifying substances are assessed relative to the acidification potential of sulphur dioxide. This implies disregarding regional differences completely and not performing assessments in more detail. If one follows this approach for assessing impacts on biodiversity due to occupation or change in physical habitats, a single scale of analysis should be defined for physical habitat depletion in LCA. In other words, occupation or change in physical habitats should be assessed against a globally relevant scale defined using the types of indicators discussed in the last section. However, such a global scale is not very detailed, risking low discrimination between alternative cases. For other aspects of land use, the issue of requiring different scales of assessment for different purposes also holds.

In order to facilitate assessment, Cowell (1998) has suggested that different scales of analysis may be used in studies rather than a single globally relevant scale. The scale of analysis may be defined for a country or even a specific type of physical habitat, and choice of a particular scale is determined by the scope of the study. For example, in a comparative study of two alternative agricultural systems for wheat production it is not necessary to assess physical habitat depletion (PHD) of the agricultural systems in relation to tropical rainforests. Indeed, in this type of study it is unlikely that data of sufficient accuracy will be available for carrying out the type of assessment outlined in section 23.2. Instead, it may be more appropriate to define management practices that are the primary determinants of PHD; examples may include the timing of sowing crops, and types of field boundaries and margins. These can be used as indicators of PHD, and can be assessed and weighted for the systems in an LCA study.

In the COST E9 Action on LCAs for forestry products, this issue has also been recognised by the 'land use' working group. Various types of LCA information are distinguished

| Type of information | Indicator | Basic information required | Reference |
|---|---|---|---|
| *Inventory Data* | | | |
| Traditional inventory data | Land occupation | Average $m^2$ required for producing 1 unit of output | Various, see (Udo de Haes et al., 1999) |
| Traditional inventory data | Land change trends | Yearly changes to agriculture (caused by agriculture demands), specifying the type converted, in quality terms (see below) and $m^2$ per type | (Lindeijer et al., 1998) |
| Traditional impact assessment data | Land use type | Type according to Hemerobiestufen-approach | (Knoepfel, 1994), (Renner and Klöppfer, 1996) |
| *Biodiversity Data* | | | |
| Global biodiversity data | Vascular plant species diversity | Number of vascular plant species per $km^2$, for agriculture and nearby highest diversity region (reference situation); for change also the situation before the change | (Lindeijer et al., 1998), (Baitz et al., 1998), (Schweinle, 1998) |
| Global biodiversity data (change only) | Relative area of ecosystem changed to agriculture | Area of classical ecosystem type changed | (Cowell, 1998) |
| Global biodiversity data | Relative number of rare species | Number of rare species per $km^2$ for the ecosystem under consideration, and same numbers for the related global ecosystem type | (Cowell, 1998) |
| Local biodiversity data | Simpson index, national Red List and regional characteristic birds, mammals, insects and vascular plants | Number of species and individuals, number of rare and characteristic species on regional lists (but note qualification in Cowell (1998)) | (Biewinga and Van der Bijl, 1996) |
| Local biodiversity data | Criteria and indicators for sustainability of agriculture at a local level | Many parameter scores, in sections: stability, productivity, vitality, biodiversity and protective functions | Similar to Helsinki criteria for sustainable forest management |
| *Life Support Data* | | | |
| Life support data | Free net primary biomass productivity (fNPP) | Yearly amount of dry matter biomass produced per unit of output, not withdrawn for human consumption (leaves, roots etc.) | (Lindeijer et al., 1998) |
| *Soil Quality Data* | | | |
| Life support/ abiotic depletion data | Soil erosion (RUSLE) | Soil loss, rainfall-runoff, erodibility, slope length, slope steepness, cover-management, supporting practices | (Cowell, 1998), (Renard et al., 1994) |
| Life support/ productivity data | Soil compaction (FLI) | Weight of vehicle and field time, no. of drives and percentage of compacted area | (Cowell, 1998), (Schweinle, 1998) |
| Life support/ productivity data | Soil organic matter (OM) | Mass of organic material added | (Cowell, 1998), |

*Figure 23.3 Overview of proposed indicators for agricultural land use assessment in LCA*

according to the application area. In particular, it is noted that for assessments within one sector (in this case, forestry/silviculture) a detailed assessment is required, whereas comparisons between wood products and minerals can be undertaken with less detailed data. Ideally, it should be possible to translate detailed data to the global scale. In Lindeijer (1998) this issue is addressed, with as possible solution to use one or two indicators on both the global and the local scale. Then detailed, local data can also be used in global assessments in the rough way, and global land use data on background processes can be compared to the detailed foreground data in local assessments. At present, vascular plant species, ecosystem area, and number of rare species seem promising biodiversity indicators for this integration of the global scale into the regional.

## 23.4 Implications for Agricultural Data Collection for LCA

In general, one could conclude that it is not yet clear which kinds of data are required to include land use in LCAs considering agricultural products or processes. As a minimum, the area changed or used during a certain time is required together with some qualitative assessment of the land use impacts. On the additional quality aspects, there are several possible indicators for which data could be collected. The extent to which this is possible depends on the available time and money. Priorities should therefore be set for data collection. Below, an overview is given on the various proposed indicators, more or less in order of priority (including feasibility) within four categories of types of information according to the personal views of the authors. Ideally, all end-points should be covered, and all data should be collected and made available with their uncertainty ranges.

## 23.5 Discussion and conclusions

*Indicator selection*

The above review has shown that data issues are particularly important in assessing land use impacts because there are many data gaps in this area. All approaches for assessing land use impacts must therefore be developed with an awareness of limitations due to current data availability, and the feasibility of obtaining data in the near future for any proposed approach. This suggests that the criteria for selection of indicators for any one end-point should include:

- inventory and impact assessment data availability;
- relevance of the indicator to the end-point;
- comprehensiveness (i.e. how comprehensively the indicator represents the value of different ecosystems in relation to the end-point).

For example, it could be argued that genetic diversity should be assessed as a relevant and comprehensive indicator for assessing biodiversity. However, it is extremely unlikely that sufficient data will be available in the foreseeable future on the genetic diversity found in different physical habitats. Therefore, there is little value in developing an approach for assessing biodiversity based on a requirement for these data.

*Data requirements for agricultural products*

The data required includes at least the land use itself ($m^2$*y), including the quality information related to the land (see below). For detailed assessments on changes, the quality situation before and after the change should also be specified. In order to be able to allocate macro-level changes due to a process type to a process occupying land, stable trends in yearly changes due to this process type could also be assessed. An example is the allocation of the yearly decrease of pristine Russian forests to wood production from Russia.

For the quality aspects, data should be gathered for the end-points biodiversity, productivity/life support, and abiotic resources. Aesthetics can only be assessed qualitatively. For agricultural products, the easiest single life support indicator to assess on a global or regional level seems fNPP. However, a combination including other biodiversity indicators (relative areas of different ecosystems, number of rare species, and the total number of species from a representative group of species (such as vascular plants) might be considered more important, along with the soil erosion indicator (which is also relatively easy to use). Other management-related indicators can be gathered relatively easily, but are not applicable to non-agriculture land use types. They are relevant only when comparing agricultural systems, and when non-agricultural processes do not make a significant contribution to the systems under analysis.

*Interpretation of results*

The approaches developed for assessment of land use impacts, some of which are mentioned above, have generally taken a pragmatic approach based on the availability of current data - or its potential availability given some additional research. The presently insufficient scientifically verified modelling makes authoritative assessment of physical habitat depletion based on LCA 'models' difficult. Therefore, qualitative judgement (or at least a rigorous interpretation) of the LCA results for land use impacts on especially biodiversity and life support must remain a viable alternative approach at the present time. Generally, one can state that present LCA results on land use can, at the most, indicate where important land use impacts are to be considered, and where a more detailed assessment (probably outside the present scope of LCA) is required.

The issue of determining the best available practice for impact categories and indicators for land use in LCA is presently subject of a task group of the SETAC-Europe Working Group on Impact Assessment (see Udo de Haes et al., 1999). This task group will probably continue working for at least two years. The subject is also handled in Workgroup 2 of the COST E9 Action on LCIA for forestry products.

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# 24. Occupational health data in agriculture

*Peter Lundqvist* [1]

*Abstract*

The work environment should be integrated in Life Cycle Assessments (LCA) to avoid a development, which create work environment problems when solving external environmental problems. Agriculture is the most hazardous industry in Sweden. A literature survey shows that a number of life-cycle assessments have been completed in which the work environment has been included to some extent. The most promising method to integrate work-environment factors seems to be the Swedish WEST method. The method has been used in various case studies in the manufacturing industry and is based on assessing nine different work environment factors. To be used in agriculture, it has to be adapted for this type of production environment. To include work environment in LCA calculations for agriculture, special projects and research groups must be initiated to focus on this important issue.

## 24.1 Introduction

Life Cycle Assessments (LCA) are becoming increasingly common as a tool for evaluating the environmental impact of materials and products. To avoid LCA leading to a development, which create work environment problems when solving external environmental problems, the work environment should be integrated in LCA. Until recently, there has not been any tradition for including matters relating to the work environment. A number of Nordic projects have started in which the work environment has been included to some extent in the life-cycle assessments. These studies have been performed in a number of sectors, mostly manufacturing industries and in assessments of new materials. To our knowledge, no LCA studies of work environment in agriculture have so far been performed or published.

## 24.2 Work environment in agriculture

Working in agriculture may involve much joy from interesting work tasks, being able to see the result of your own work, to see the crop grow and mature, king with the nature and follow the changes in the seasons. However it may also be a dangerous and harmful environment causing occupational inju-

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ries and diseases through exposure to occupational accidents and to physiological, physical, biological, chemical, psychological and sociological factors (Lundqvist, 1988).

The first European survey on the work environment (Paoli, 1992) showed that agriculture, building, and transports are clearly the sectors where, overall, the highest amount of constraints appear. This is reflected in the proportion of workers feeling at risk: 51% in agriculture, 46% in construction, and 37.5% in transport. In Sweden, the fatality rate for the agricultural industry (including forestry work) in 1996 was 23.0 deaths per 100,000 workers including both employed workers and self-employed farmers (Statistics Sweden, 1998). Comparing with other main industries and figures for all industries, this makes agriculture the most hazardous industry in Sweden.

Noise and vibrations is highly connected to the use of different machinery. May et al. (1990) found substantial hearing loss among farmers, especially in the high-frequency ranges. A study by Lindén (1986) showed that 58% of the work-related injuries in agriculture affected the musculoskeletal system as compared with 49% in all other Swedish industries.

Cow milking is reported to involve a number of ergonomical problems and a high frequency of musculoskeletal disorders (Lundqvist et al., 1997).

Working in confined animal buildings with poultry, pigs and cows involves exposure to air pollution such as ammonia, carbon dioxide, hydrogen sulphide and organic dust. Health effects from exposure of the gases in animal buildings range from mild irritations of the respiratory system to lethal effects due to exposure of high concentrations of sulphide gas in systems with liquid manure handling systems (Donham et al., 1982). Inhalation of organic dust occurring in agriculture can give rise to diseases of the airways and lungs. Dust of this kind comes, for example, from hay, grain, fuel chips, straw or other types of bedding, and from the livestock themselves. Many tasks in agriculture give rise to large quantities of dust. Lung diseases are more prevalent among farmers and farm workers than in the rest of the population (Swedish National Board of Occupational Safety and Health, 1994).

The use of chemicals in agriculture is very widely spread. The predominant chemicals are pesticides used to fight pests of different types, such as insects and fungi, and weed-killers.

The medical effects of these chemicals may be intoxication due to acute exposure or chronic effects caused by long-term low dose exposure. Acute toxicity of pesticides is considerable. The most toxic items are usually insecticides like the organic phosphorus compounds. Weed-killers and fungicides are usually less acutely toxic. The effect of organic phosphorus compounds is deadly in high doses (Hoglund, 1997).

Farming has been included on the National Institute for Occupational Safety and Health's (NIOSH) listing of the ten most stressful occupations, as well as its recently published research agenda (NIOSH, 1996). Work by many investigators has linked occupational stress in farmers to a variety of adverse outcomes. Distressed farmers and spouses commonly experience sleep disturbances, family conflict and concentration problems (Walker and Walker, 1987). A study comparing male farmers in Ohio with data from a sample of all employed males (National Health Interview Survey) found that the farmers showed elevated levels of emotional stress and depressive symptoms (Elliott et al., 1995). Suicide has been documented to occur in farmers at a rate higher than that of the general population (Gunderson et al., 1993).

## 24.3 Methods to include work environment in LCA

In order to be able to perform an LCA in which effects on the work environment are also considered, there has to be ways of measuring these effects. It must be possible to compare different working situations with each other. The work environment consists of many different physical as well as psychological and social factors.

A limited literature survey shows that a number of Life Cycle Assessments have been completed in which the work environment has been included to some extent.

A Danish report (Broberg et al., 1993) describes six projects in which the work environment has been included: 1) Environmental assessment of materials ('The water-tap project'), 2) The frame programme for integrated assessment of environmental and work environment effects (part of the Danish Materials Technology Development Programme), 3) Environmental Design of Industrial Products, 4) 'The recycled house', 5) Environmental assessment of new metropolitan trains, and 6) Life-cycle costs of products. The report concludes that the six projects have so diverse objectives that they require different data, assessment parameters, and methods.

The Swedish Institute of Production Engineering Research (IVF) has developed an interesting method for evaluation called the WEST method (Bengtsson and Berglund, 1997). The method has been used in various case studies and is based on assessing nine different work environment factors (six physical factors and three psychological/social factors) and assigning point scores to them:

- risk of accidents;
- physical work load;
- noise;
- chemical health risks;
- vibration;
- general physical environment;
- work atmosphere;
- work content;
- freedom of action.

The method attempts to estimate how a particular work situation or work place affects an individual relative to the effect on him/her if he/she did not perform that particular work. This is not, however, a direct comparison with unemployment: instead, it is concerned with understanding exactly what the particular work situation involves for the person, both positive and negative. Certain factors give positive points and other negative, while some factors can give points that are either positive or negative depending on the particular work situation. The method has been used in more than 40 manufacturing industries.

## 24.4 Available occupational health data in agriculture

Statistics Sweden (1998) publishes every year data on occupational diseases and accidents. The data is based on injury forms sent by employers to the social insurance offices. Copies are sent on to the Labour Inspectorate where specialised staff examine, codify, and register the information from the forms. Occupational injuries are divided, according to the type of injury, into three groups, namely occupational accidents, commuting accidents, and occupational diseases. The data presented for each occupation, such as agriculture, is only presented for the whole group and is not broken down into different types of work operations, type of machinery or material involved. It is also well known that there is an underreporting of occupational injuries in Swedish agriculture. This has clearly been shown by Jansson (1988).

Another source of occupational health data is the type of facts presented through regulations and occupational exposure limit values (Swedish National Board of Occupational Safety and Health, 1993). This type of data gives the rules on noise exposure, exposure to hazardous substances etc.

In order to collect specific data on occupational health aspects in agriculture, a number of surveys and research projects has been carried out (Hoglund, 1997). However, the agricultural sector is so diverse that it is hard to standardise occupational health data for production of meat, milk, or grain. We still miss many important data on the level of health data related to products.

When it comes to data on time spent on different work operations, it is also a problem with enough up to date data. In Sweden there used to be published a yearly book of production data, including labour data, by the Swedish University of Agricultural Sciences. An ongoing project will make it available through Internet, but the lack of relevant new data will still be a problem.

## 24.5 What is needed to establish occupational health data in agriculture

One important issue is to decide on the level of accuracy. We need accuracy comparable to that used in the manufacturing industry, and we need to perform new studies in the agricultural sector, with methods similar to or adapted from the manufacturing industry, such as the Swedish WEST method (Bengtsson and Berglund, 1997).

To include work environment in LCA calculations for agriculture, special projects and research groups must be initiated, which in detail plans on how to make it possible to answer questions such as: What is the health effect of producing one ton of milk, meat or grain?

Work environment has to be involved in Life Cycle Assessments for agriculture. Too often, changes have been introduced by politicians and authorities to promote animal welfare or external environment without calculating the health effects for the people involved in production.

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# F. Farm typologies and farm accountancy data for LCA

# 25. Farm types - How can they be used to structure, model, and generalise farm data?

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*Abstract*

In this study, a method for farm typing is proposed. The ability of the method to structure, model, and generalise farm data is illustrated by three examples, where differences between farm types are analysed. The examples are calculation of 1) nitrogen-surplus, 2) phosphorous-surplus, and 3) use of fossil energy.

A farm type is defined as a relatively homogeneous farming system, described by a set of system variables. Three basic farm types are identified: 1) Stockless, cash crop production, 2) Production of cattle and other ruminants, and 3) Production of pigs and other non-ruminants. The classification into farm types is made on the basis of economic criteria according to the distribution of average gross margins for produced farm products. A subdivision of farm types is proposed into small, mixed and specialised farms, further subdivided into farms with different livestock types. However, depending on the actual purpose of the investigation, the farms may be subdivided or aggregated into other suitable farm types.

The system variables needed for farm typing are available in Denmark, but the use of these data for modelling and allocation of resource use to the products of Danish agriculture, has just begun. Comparison of the defined farm types across EU countries are possible, as EUROSTAT uses a farm classification compatible with the one presented. Difficulties may arise if comparison with farming outside the EU is desired.

Example 1 shows the use of farm typing for analysis of nitrogen-surplus, and shows how farm type modelling can help to identify important system parameters. Example 2 shows that two different classifications of farm types are suitable for analyses of respectively nitrogen- and phosphorous-surplus on study farms. Finally, example 3 shows how modelled farm data can be generalised to a larger geographical scale than the farm.

## 25.1 Introduction

Quantification of the use of resources in agriculture and the following environmental impact is of increasing interest (Brown et al., 1998). Both consumers and the authorities are interested in documentation. Life Cycle Assessment (LCA) is a method for such documentation and specification of the use of resources in each step of the production. One of the major problems for performing LCAs for agricultural products is the lack of access to structured farm data and farm models. In this

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study, a method for farm typing is proposed, which can help to structure and model farm data, and the use of the method is illustrated with three examples.

Farm typing is a useful method for classifying farms and farm data. Because each farm type is relatively homogeneous, it can be described and understood as a system, which can be modelled separately (Sørensen and Kristensen, 1988). Consequences of changes in the system environment can then be modelled for each farm type and aggregated to larger scales, as well as compared with consequences of similar changes for other farm types.

Several approaches for farm typing are used for different purposes in Denmark. Some of the approaches use economic criteria for division into farm types (e.g. Denmark's Statistics, 1998; Danish Institute of Agricultural and Fisheries Economics, 1998c; Schou et al., 1998). Others use physical criteria like crop rotations and soil type (Mikkelsen et al., 1998), amount and type of animal manure spread (Østergård and Mamsen, 1990), fodder feeding intensity (Dalgaard et al., 1998) or simply type and number of animals per ha (Danish Agricultural Advisory Centre, 1998). A common feature for all these approaches is that they distinguish between three, basic, farming systems: 1) cash crops, stockless, 2) cattle and other ruminants, and 3) pigs and other non-ruminants. Division into further types depends on the purpose of the individual approach. Determination of the criteria for this further division is interesting and will be discussed in this article.

The aim of this paper is to review existing methods and data available for farm-typing in Denmark and on this basis to define and evaluate a method for farm type classification to be used at Danish Institute of Agricultural Sciences. The aim with this method is to structure the available farm data into relatively homogeneous units (farm types), model input and output for each farm type, so that they can be allocated to crops, fields, animals etc. (bottom-up modelling), and generalise the modelled data to a larger geographical scale than the farm (upscaling).

## 25.2 Materials and methods

### 25.2.1 Farm data available in Denmark

Farm data are available from several sources in Denmark (table 25.1). The most comprehensive data set is the General Agricultural Register, GLR/CHR (Ministry of Food, Agriculture and fisheries, 1997). Here all farm holdings are registered, including the number and type of their farm animals, field sizes, types of crops, and key data for their use of manure and fertiliser. The data are geo-related, which means that each farm has a co-ordinate, and each field is related to a field block. A field block is defined as an area with a static boundary in the landscape e.g. hedges, roads, or streams. Each field block has a number which in a Geographical Information System (GIS) can be pointed out to an area consisting of one to ten fields (figure 25.1). The geo-related data makes it possible to combine farm data with other geographical data such as soil type, climate, and topography (table 25.3).

Data in the GLR/CHR are gathered in one database as a part of the administration of the EU scheme for crop and animal subsidies and national regulations on pollution from fertilisation. The quality of the data is high, and is controlled by the authorities via satellite control of registered field crops and by farm control of fertiliser plans and animal counts. However, the data in the general register are not very detailed when it comes to description of internal flows on the farm and do not include any economic data, except for the official land evaluation of the farm, and subsidy paid.

In special study areas, more detailed data are collected for all farms within a geographical region. As for the GLR/CHR, the data can be combined with other geographical data, and interactions between e.g. farm and soil type can be investigated. The duration of data collected from study areas is limited by the period of the research project collecting these data. In contrast, the GLR/CHR data will continue to be collected each year, as long as Danish legislation regarding the data collection is unchanged.

The most detailed farm data available are from study farms (Danish Agricultural Advisory Centre 1998). These are private farms where data on animal and field level are collected in cooperation with the farmer and the advisory service. The study farm data are gathered in a database and the quality of the data is checked at the farm level against input and output from the farm accounts. They are therefore well suited for detailed analyses, and analyses of connections within the farm. The study farms are not a statistical representative sample of Danish farms, but cover the most common farm types.

*Table 25.1 Examples of farm data available in Denmark; the total number of farms in Denmark is around 65,000*

| | No. of animals | Crop types | Detailed data a) | Economic data | Individual farm data | Geo-related | No. of farms |
|---|---|---|---|---|---|---|---|
| GLR/CHR | x | x | | | x | x | all farms |
| Study Areas | x | x | x | | x | x | 30-500 |
| Study Farms | x | x | x | x | x | | 70 |
| Economical Statistics | x | x | | x | | | 2,000 |
| National Statistics | x | x | | x | | | 26,000 |

a) E.g. field data about yields, number of treatments and amount of manure used, data on fodder used per animal, weights on animals, prices on farm products and input, status for fodder stocks etc.

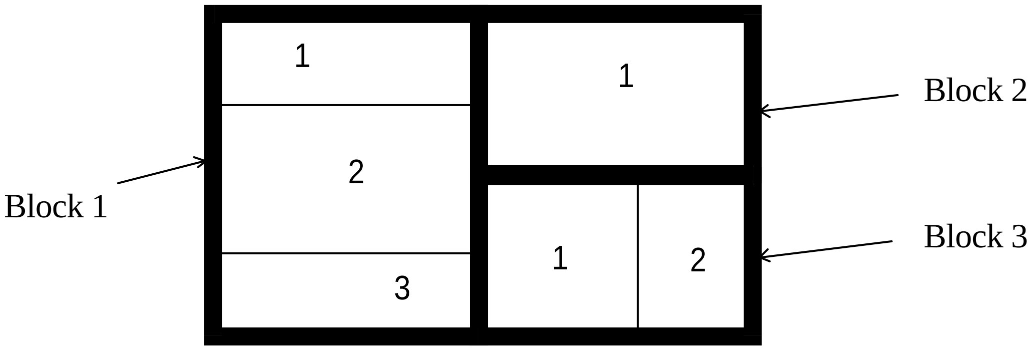


*Figure 25.1 Schematic drawing of three field-blocks; block 1 consists of three fields, block 2 of one field and block 3 of two fields*

Statistically representative farm data are available from Denmark's Statistics (1998) and from Danish Institute of Agricultural and Fisheries Economics (1990, 1998a, b, c). Here, a yearly status is given in form of averaged input and output distributed on predefined farm types or classes. The statistics are comparable with statistical data from other EU countries, as all the statistical bureaus in the EU use the same basic data classes (Denmark's Statistics 1998). Statistics divided into other classes than the predefined are not readily available, and demand expensive extra queries in the statistical databases.

### 25.2.2 A method for farm typing

A farm type is defined as a relatively homogeneous farming system, which can be described by a set of system variables. Dependent on an actual investigation, farms can be divided into a suitable number of farm types, which each is described by a suitable number of system variables.

In this study, the classification into farm types is made from economical criteria analogous to the criteria used by Denmark's Statistics (1998), Danish Institute of Agricultural and Fisheries Economics (1998c), and EUROSTAT. Here, the farms are classified according to the distribution of their average gross margin (AGM), which is a measure for the total income to the farm minus the variable costs. AGM for a whole farm (AGMtot) is calculated as the sum of AGM from crop production (AGMcrp), AGM from the livestock production of cattle and other ruminants (AGMrum), and from pigs and other non-ruminants (AGMnrum) (equation 1). AGMrum is calculated as AGM from dairy animals (AGMdairy) plus AGM from beef animals (AGMbeef) (equation 2). AGMnrum is calculated as the sum of AGM from slaughter pigs (AGMslaug), breeding pigs (AGMbreed), poultry (AGMpoul), and fur animals (AGMfur) (equation 3). Each of the AGM are calculated as the average AGM according to the Institute of Agricultural and Fisheries Economics (1998c) statistics in the period 1992-94 (see table 25.3 and 25.4).

$$\mathrm{AGMtot} = \mathrm{AGMcrp} + \mathrm{AGMrum} + \mathrm{AGMnrum} \qquad (1)$$
$$\mathrm{AGMrum} = \mathrm{AGMdairy} + \mathrm{AGMbeef} \qquad (2)$$
$$\mathrm{AGMnrum} = \mathrm{AGMslaug} + \mathrm{AGMbreed} + \mathrm{AGMpoul} + \mathrm{AGMfur} \qquad (3)$$

Farms are then classified into three basic types: 1) Production of cash crops without livestock, 2) Production of cattle and other ruminants, and 3) Production of pigs and other non-ruminants. Each of these types can again be divided into subtypes dependent on the actual analysis. Here the three basic types are subdivided into small, mixed or specialised farms, which again are subdivided into farms with different livestock types (figure 25.2). This division is compatible with the EU-statistics and therefore farm data classified into these types can be compared with at least these statistics.

| | Farm type | Criteria for Average Gross Margin (AGM) |
|---|---|---|
| 1 | Crop production, stockless | AGMcrp>1/3*AGMtot, AGMrum<1/3*AGMtot, |
| 1.1 | Small | AGMnrum<1/3*AGMtot |
| 1.2 | Mixed | AGMtot<30000 |
| 1.3 | Specialised | AGMcrp<2/3*AGMtot |
| | | AGMcrp>2/3*AGMtot |
| 2 | Cattle and other ruminants | AGMrum>1/3*AGMtot, AGMcrp<1/3*AGMtot, |
| 2.1 | Small | AGMnrum<1/3*AGMtot |
| 2.2.1 | Mixed, dairy | AGMtot<30000 |
| 2.2.2 | Mixed beef | AGMrum<2/3*AGMtot, AGMdairy>AGMbeef |
| 2.3.1 | Specialised, dairy | AGMrum<2/3*AGMtot, AGMdairy<AGMbeef |
| 2.3.2 | Specialised, beef | AGMrum>2/3*AGMtot, AGMdairy>AGMbeef |
| | | AGMrum>2/3*AGMtot, AGMdairy<AGMbeef |
| 3 | Pigs and other non-ruminants | AGMnrum>=1/3*AGMtot and not type 1 or type 2 |
| 3.1 | Small | AGMtot<30000 |
| 3.2.1 | Mixed, slaughter pigs | AGMnrum<2/3*AGMtot, AGMslaug>AGMbreed, not 3.2.3 or 3.2.4 |
| 3.2.2 | Mixed, breeding pigs | AGMnrum<2/3*AGMtot, AGMslaug<AGMbreed, not 3.2.3 or 3.2.4 |
| 3.2.3 | Mixed, poultry | AGMnrum<2/3*AGMtot, AGMpoul>1/2*AGMnrum |
| 3.2.4 | Mixed, fur animals Special- | AGMnrum<2/3*AGMtot, AGMfur>1/2*AGMnrum |
| 3.3.1 | ised, slaughter pigs | AGMnrum>2/3*AGMtot, AGMslaug>AGMbreed, not 3.3.3 or 3.3.4 |
| 3.3.2 | Specialised, breeding pigs | AGMnrum>2/3*AGMtot, AGMslaug<AGMbreed, not 3.3.3 or 3.3.4 |
| 3.3.3 | Specialised, poultry | AGMnrum>2/3*AGMtot, AGMpoul>1/2*AGMnrum |
| 3.3.4 | Specialised, fur animals | AGMnrum>2/3*AGMtot, AGMfur>1/2*AGMnrum |
| 3.4 | Others | Others |

*Figure 25.2 Farm type classification into systems. In this example farms are subdivided into three levels, which each again can be subdivided, or can be aggregated to a lower level of subdivision*

### 25.2.3 Bottom-up modelling and farm type models

In this context, bottom-up modelling is defined as an approach where the agriculture of a region is modelled as an assembly of separately modelled farm types. The contrast to bottom-up modelling is a top-down approach where the whole agricultural sector is modelled as one average farm or as a combination of models which each model different parts of the agricultural sector (Walter-Jørgensen, 1998).

Bottom-up modelling demands comprehensive farm input data, which are available in Denmark (table 25.1). However, it is not practical to set up a specific model for each farm. Instead, models can be set up for each of the defined farm types, which can be modelled by a set of system input variables (figure 25.3).

Farm type models differ in two respects: Physical structure (number of fields, animal housing etc.) and management (crop rotations, fertilisation etc.) This means that when given the same input data the models will generate different output (figure 25.3).

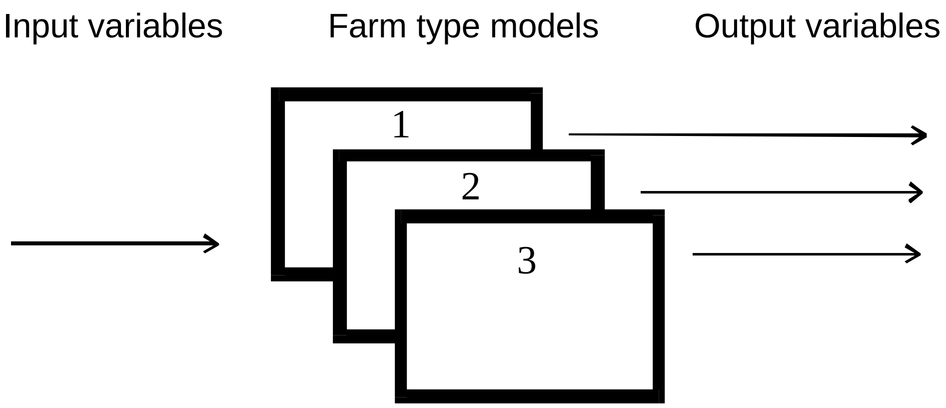


*Figure 25.3* *Farm type models, input- and output variables; each farm type is described by different models; the same set of input variables results in different sets of output variables according to the farm type*

### 25.2.4 Farm type system variables

The farm type system variables are the set of data, which are required to drive the farm type models and to compare model output and reality. Some of the system input variables are part of the physical structure of the farm and are therefore relatively fixed (e.g. soil type and number of fields). Other inputs depend on the actual farm management (e.g. fertiliser use and crop rotation), and can therefore vary from year to year e.g. as a result of changed prices or political conditions (figure 25.4). Some examples of the resulting system output variables are listed in figure 25.5.

### 25.2.5 Upscaling

One of the main questions relating to bottom-up modelling is how the modelled results can be aggregated to different levels and compared with regional statistics. A useful tool for aggregation is a Geographical Information System (GIS), which can handle large amounts of geographically linked data and the distribution of these geographic themes on the farm types located in the landscape.

In this study, upscaling means aggregation and generalisation of agricultural data from the farm level to a region, where calibration against regional statistics are possible, or where more generalised analysis are relevant.

| | Available nation-wide a) | Available for study farms a) |
|---|---|---|
| Fixed | | |
| ↑ | | |
| Soil type | (x) | (x) |
| Climate | x | x |
| Topography | (x) | (x) |
| | | |
| Administrative borders | x | x |
| Farm area | x | x |
| Placement of fields | (x) | x |
| Manure system | - | x |
| Machinery | - | x |
| Draining and irrigation | - | (x) |
| Windbreaks | (x) | (x) |
| | | |
| Farmers age | x | x |
| Farm income | (x) | x |
| Farmers education | - | - |
| External relations | - | x |
| | | |
| Import/export of manure | (x) | x |
| Import/export of fodder | - | x |
| Number of animals | x | x |
| Fodder plan | - | x |
| Use of energy | - | (x) |
| Use of manure and fertiliser | (x) | x |
| Use of pesticides | - | x |
| Field crops | x | x |
| ↓ | | |
| Variable | | |

*Figure 25.4 Example of farm type system input variables and their availability in Denmark*
a) 'x' = available, '(x)' = partly available, '-' = not available.

| | Available nation-wide a) | Available for study farms a) |
|---|---|---|
| Crop yield | (x) | x |
| Kg meat and milk produced | (x) | x |
| | | |
| Farm balances for nutrients | - | x |
| Direct and indirect energy use | - | x |
| Emission of green-house gasses | - | (x) |
| Loss of nutrients | - | (x) |

*Figure 25.5 Examples of farm type system outputs and their availability in Denmark*
a) 'x' = available, '(x)' = partly available, '-' = not available.

## 25.3 Case studies

In this section, three small examples are given of how farm typing can help to structure, model, and generalise farm data.

The first two examples illustrate an analysis of differences in surplus of nitrogen (N) and phosphorous (P) for different farm types. This analysis is interesting because N and P surpluses on the farm scale indicate loss of nutrients to the environment (Vatn, 1996; Dalgaard et al., 1998). It is therefore interesting to know the total surpluses, so that they can be allocated to the products of the actual farm type, and be used for an LCA.

The third example illustrates an analysis of differences in use of fossil energy. A quantification of the use of fossil energy is interesting because it leads to the emission of carbon dioxide, and therefore to global warming (IPCC, 1997). Again, for the purpose of LCA, the question is to allocate the use of fossil energy to the different products, so that their energy costs can be compared.

### 25.3.1 Example 1: N surplus on study farms

The study farms (table 25.1) are classified into the three main farm types and the N surpluses are modelled according to the method in Dalgaard et al. (1998) (figure 25.6). The farm type classification reveals a significantly lower N surplus for crop production than for livestock production farms. Pig farms have the highest average N surplus, but also the highest variation.

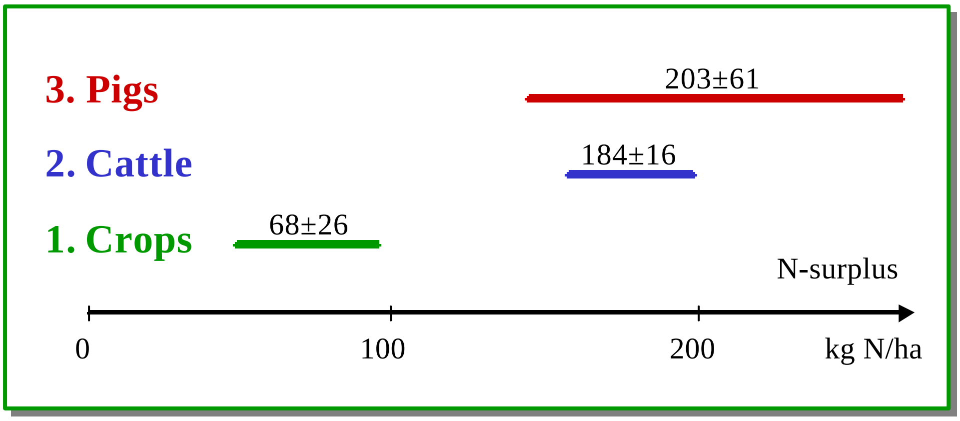


*Figure 25.6 Average N surplus for the three main farm types among the study farms (95% confidence intervals); all cattle farms are dairy farms*

Further investigation showed a significant linear correlation between kg N in animal manure per hectare per year and calculated N surplus per hectare per year for the study farms (figure 25.7). Thus, an important system indicator for N surplus for the classified farm types was kg N in animal manure spread.

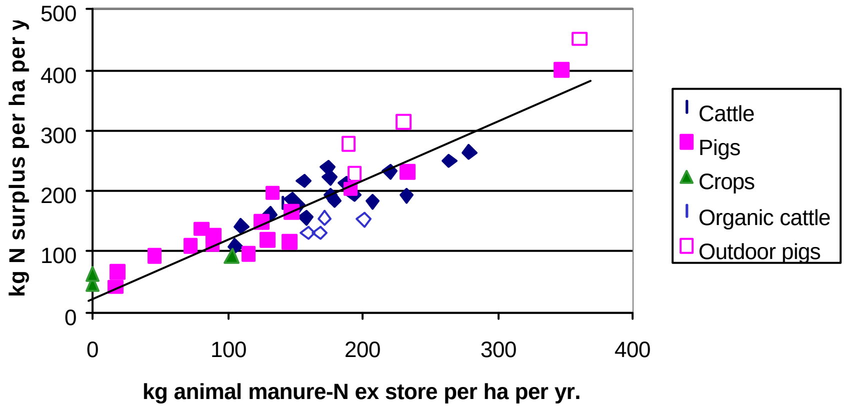


*Figure 25.7 Kg N surplus per hectare per yr. as a function of kilogram animal manure-N ab store per hectare per year for different farm types; data from study farms in 1997 (Dalgaard, 1998); the line is the best linear fit to all points*

The possibilities for subdivision of the three basic farm types (crop-, cattle-, and pig production) were investigated. The investigation showed that organic cattle farms had a lower N surplus than the average, and that outdoor pig production had a higher surplus than the average (figure 25.8). However, N surpluses for the conventional pig farms did not clump, which means that there can not be found one single value for N surplus per kilogram pig meat produced. A division into more pig farm types is therefore necessary, if such a value is desired for an LCA.

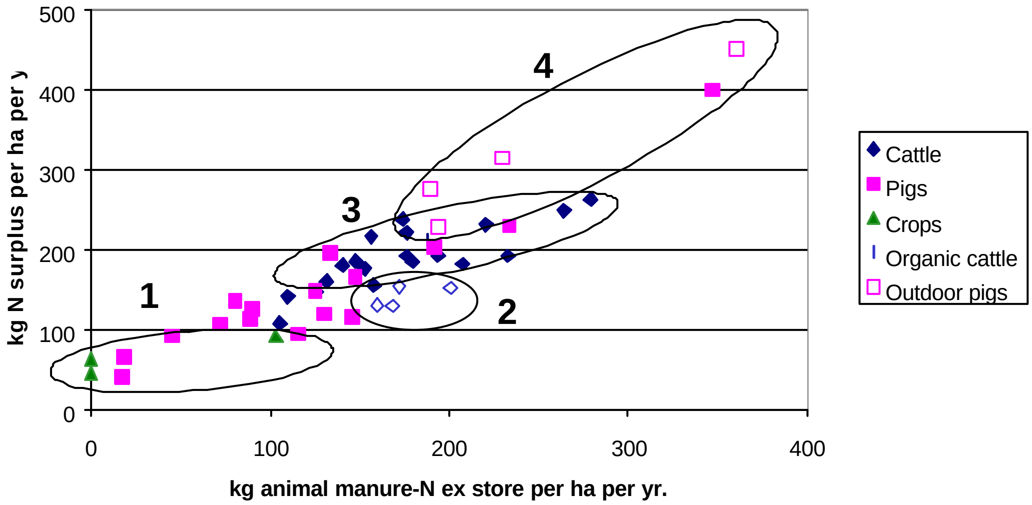


*Figure 25.8 Identification of four different farm types relating to N surplus: 1) cash crops, 2) organic cattle, 3) conventional cattle, and 4) outdoor pig production*

The N surpluses for respectively conventional and organic dairy farming fell into two nice clumps (figure 25.8). These two farm types were therefore chosen for exemplification of farm type modelling, where N surpluses for the two types were modelled for farming on three different soil types (figure 25.9).

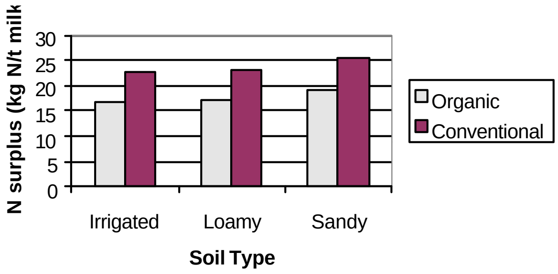


*Figure 25.9 The sensitivity of N surplus (kg/t milk) to soil type (irrigated, loamy, and non-irrigated sandy soil) modelled for organic and conventional dairy farming*
Source: Dalgaard et al. (1998).

The modelling shows that differences in the physical structure, with differences in soil type as indicator, only have a small effect on the N surplus per kilogram of milk produced, and that the difference between soil types is practically the same for both farm types. Alternatively, the differences between organic and conventional farming could be interpreted as differences in management. In that case, management style is shown to have a relatively high influence on the N surplus per kilogram milk produced.

### 25.3.2 Example 2: P surplus on study farms

The P surplus for the study farms is modelled analogous to the N surplus (figure 25.10). Here, crop production also shows a significantly lower surplus than livestock production, and the cattle farms have a lower P surplus than the pig farms (almost significant at the 95% level).

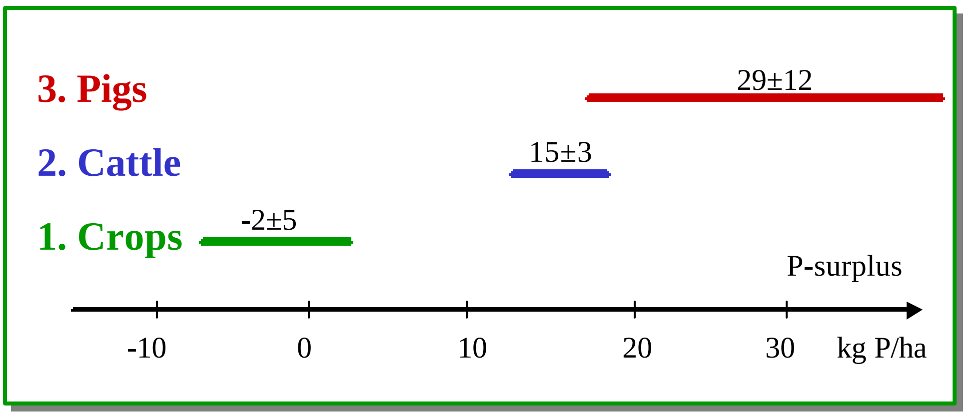


*Figure 25.10 Average P surplus' for the three main farm types among the study farms (95% confidence intervals); all cattle farms are dairy farms*

When P surplus' are plotted against the amount of manure spread (with kg N in manure per hectare per year as indicator), a similar correlation to the one discovered in figure 25.7 is found. However, for P there seems to be no significant difference between conventional and organic dairy cattle production, or between conventional and outdoor pig production. Therefore, in the case of P surplus, there is no reason to divide the cattle farms (type 2) into organic and conventional cattle farms, or to divide the pig farms (type 3) into conventional and outdoor pig production (figure 25.11).

Another subdivision may be relevant if P surpluses are to be allocated to e.g. kilogram of milk or meat produced. Such a division is not exemplified here, where the main purpose with example 2 was to show that two different sets of farm types are suited for analysis of respectively N surplus and P surplus.

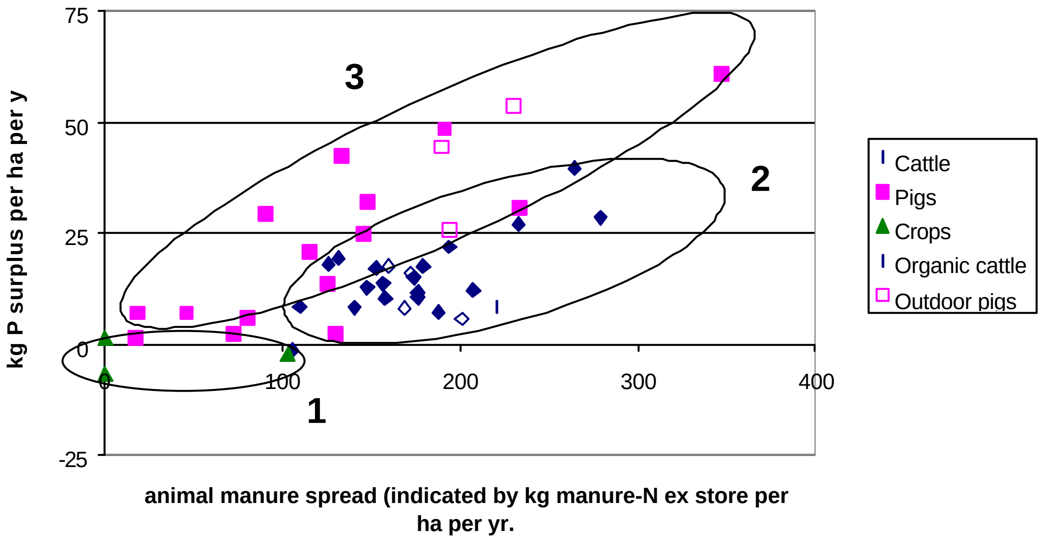


*Figure 25.11 Kg P surplus per hectare as a function of the amount of animal manure spread (with kilogram manure-N ab store per hectare per year as indicator) on the three different farm types: 1) crop production, 2) cattle production, and 3) pig production*

### 25.3.3 Example 3: Use of fossil energy

In a study by Dalgaard et al. (1998b), the use of fossil energy were estimated for the 1996-situation and for three national scenarios for conversion to 100% organic farming in Denmark:

A) Self-sufficiency in fodder and feed (pig production is limited).

B) Maximum fodder import according to the national rules for organic farming (15% of the own fodder production for ruminants and 25% for non-ruminants).

C) The same animal production as in 1996 (this means a high import of fodder).

The national energy balances were estimated by aggregating energy costs for each type of crop and animal (figure 25.12 and 25.13). The energy costs in figure 25.13 can be interpreted as the energy cost for production of one livestock unit (LSU) on two times four different farm types (organic and conventional dairy, beef, pig and poultry production). The total use of energy for livestock production in Denmark can then be modelled from the aggregated energy used on these farm types (table 25.2).

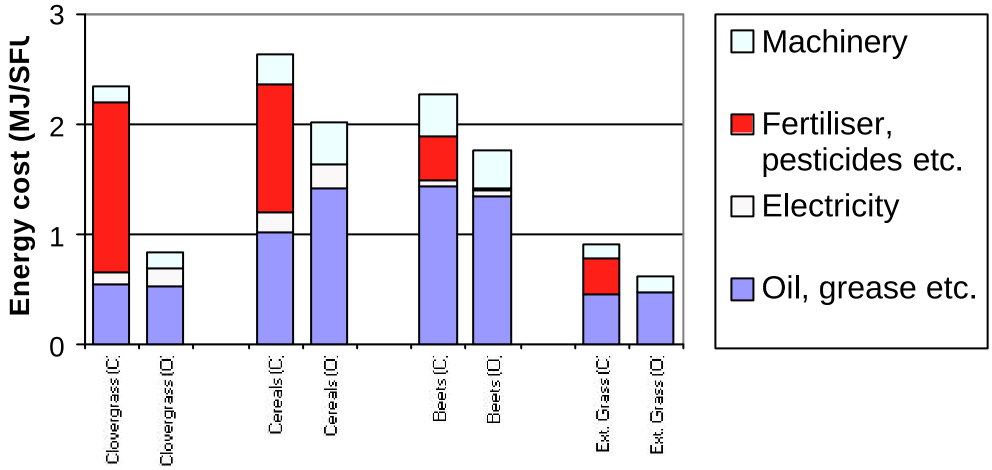


*Figure 25.12 Estimated energy costs for typical crops: clover-grass, cereals, beets, and extensive grassing. Conventional farming marked with (C), and organic farming marked with (O) (Dalgaard et al., 1998b); 1 SFU (Scandinavian fodder unit) equals the fodder value in 1 kg barley = 12.5 MJ metabolisable energy*

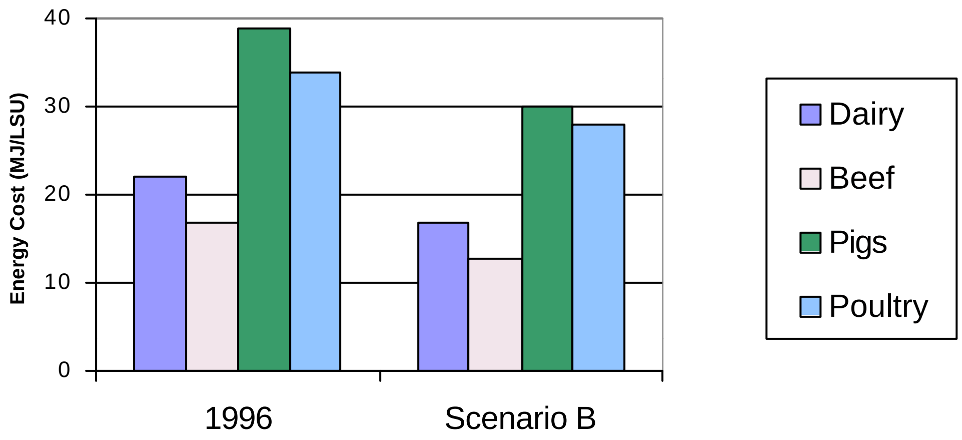


*Figure 25.13 Estimated energy cost per livestock unit (LSU) dairy cattle, beef cattle, pigs, and poultry (Dalgaard et al., 1998b); examples for the 1996-situation and for scenario B. (1 LSU corresponds to 1 cow of large race, 30 porkers produced or 2,500 slaughter hens)*

The aggregated modelled use of each energy carrier (oil, electricity, fertilisers, machinery etc.) for the 1996-situation, was compared with the national use according to Denmark's Statistics (1997). For each energy carrier, correction factors were calculated as energy use according to the statistics divided with energy use according to the modelling. These correction factors were then used for correction of the summed, modelled use of energy carriers in all the scenarios. Consequently, the figures in table 25.2 are corrected figures while the figures in figure 25.12 and 25.13 are non-corrected figures.

*Table 25.2 Fossil energy balance for Danish agriculture in the 1996-situation and for the three organic scenarios*

| | The 1996-situation $10^{15}$ J | Scenario A $10^{15}$ J | Scenario B $10^{15}$ J | Scenario C $10^{15}$ J |
|---|---|---|---|---|
| *Crop production (C)* | | | | |
| Oil, grease etc. a) | 16.4 | 11.7 | 11.7 | 11.7 |
| Electricity | 1.8 | 1.8 | 1.8 | 1.8 |
| Other inputs | 13.9 | 0.1 | 0.1 | 0.1 |
| Machinery | 4.6 | 3.3 | 3.3 | 3.3 |
| Total | 36.8 | 16.9 | 16.9 | 16.9 |
| *Animal production (A)* | | | | |
| Electricity for stables | 10.8 | 8.2 | 8.9 | 9.3 |
| Climate stables (oil and straw) a) | 3.0 | 0.0 | 0.0 | 0.0 |
| Buildings, inventory etc. | 6.3 | 4.6 | 5.6 | 6.4 |
| Fodder import | 16.3 | 0.0 | 14.0 | 25.6 |
| Total | 36.3 | 12.8 | 28.5 | 41.4 |
| *Net Energy production (E)* | 6.6 | 2.1 | 2.0 | 0.5 |
| Energy balance (C+A-E) | 66.5 | 27.7 | 43.5 | 57.8 |

a) Including energy costs for distribution, refining etc.
Source: Dalgaard et al. (1998b).

Example 3 shows two things: 1) how farm type modelling can be used for allocation of fossil energy costs to single crops types or animal types, and 2) how these modelled data can be used for generalisation to a larger geographical scale than the farm.

## 25.4 Discussion

A method for farm typing on the basis of economic criteria was proposed, and the use of this method was illustrated by three small examples.

A farm type was defined as a relatively homogeneous farming system, described by a set of system variables. Three basic farm types were identified: 1) stockless, cash crop production, 2) production of cattle and other ruminants, and 3) production of pigs and other non-ruminants. The classification into farm types was made on the basis of economic criteria according to the distribution of average gross margins for produced farm products. A subdivision of farm types into small, mixed and specialised farms, further subdivided into farms with different livestock types, was proposed, but depending on the actual investigation, the farms may be subdivided or aggregated into other suitable farm types. In particular, the three examples evaluated the division into farm types according to livestock type. An evaluation of the suitability of the other levels of farm typing is an interesting task for future work. Such an evaluation is relevant, because the proposed method for farm type classification is compatible with the method used for farm typing in the EU statistics. It is therefore well suited for comparison of farming within the European Union, but difficulties may arise if comparison with farming outside the EU is desired.

In Denmark, many farm data are available, both on the detailed farm level, and on the general geographical distribution of farm types and the corresponding biophysical factors like soil types, climate, and topography. We have just started to use all these data, and the three examples in this study are only appetisers for the future use of farm typing to structure, model, and generalise farm data.

The farm system variables listed in figure 25.4 and 25.5 mainly included strict physical variables like soil type, manure system, field sizes, and field numbers (hard system variables). However, non-physical factors like management style and the farmer's agronomic knowledge (soft system variables) also influences the system output, and may be included in the farm type models. These soft system variables are difficult to measure and therefore difficult to find in the statistics. The presented list of input variables (figure 25.4) therefore only includes a few soft system indicators (farmer's age, education etc.). These may be used as dummies, indicating soft system variables like the management style (e.g. an old farmer will tend to phase out his farm, while a young farmer may tend to expand). The soft system variables are difficult to include in the type of bottom-up farm models presented in this paper. It is therefore important to discuss the importance of the key soft system variables for the results given by hard system models. Alternatively, the soft system variables could be included in the farm models, but this would require help from economists or sociologist with expertise in e.g. behaviour-models or farm management. This is not within the reach of the present work.

The proposed division into three basic farm types, may not be the best suitable division for all purposes. However, the classification is compatible with most other divisions of farms into types. Moreover, if another farm type classification than the one described in figure 25.2 is needed, the method for classification is flexible, and new classes can be defined on the basis of calculated average gross margins. An alternative way to classify farm types is a factorial analysis (Sørensen and Coreia, 1998), where all available farm data via a statistical ordination procedure are ordinated, e.g. to a co-ordinate system with two axes (Manly, 1990). Farm types can then be identified as clumped data analogous to the identification of farm types in figure 25.8 and 25.10, and afterwards the types can be described by the characteristics for each clump of farms. However, one can not be sure that the clumped farms are homogeneous farming systems, which can be described and modelled by separate farm type models. The factorial analysis method is therefore not investigated further in this paper.

All in all, the proposed method for farm typing was suited to structure, model and generalise farm data. This was demonstrated in the three examples: example 1 showed the use of farm typing for analysis of N surplus, and how farm type modelling could be used to identify important system parameters for determination of differences in N surplus. Example 2 showed that two different classifications of farm types were suitable for analyses of respectively nitrogen- and phosphorous-surpluses. Finally, example 3 showed how modelled farm data could be generalised to a larger geographical scale than the farm.

## 25.5 Acknowledgements

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*Table 25.3 Average Gross Margins (AGMcrp) for production of crops in the two regions of Denmark; region East and Region West*

| Crop | AGM (ECU per ha) | |
|---|---|---|
| | region East | region West |
| Wheat | 999 | 852 |
| Rye | 659 | 536 |
| Barley | 776 | 636 |
| Oats and mixed cereals | 636 | 547 |
| Legumes | 916 | 786 |
| Potatoes | 3,156 | 1,841 |
| Sugar Beets | 1,866 | 1,386 |
| Industrial Seeds | 891 | 754 |
| Seeds for sowing | 826 | 600 |
| Fodder Crops | 0 | 0 |

Source: Denmark's Statistics (1998).

*Table 25.4 Average Gross Margins (AGM) for farm animals in Denmark*

| Animal | AGM-class | AGM (ECU per head) |
|---|---|---|
| Horses | beef | 330.00 |
| Calves <1 yr. | beef | 107.00 |
| Bulls and bullocks 1-2 yr. | beef | 140.00 |
| Bulls and bullocks >2 yr. | beef | 137.00 |
| Heifers 1-2 yr. | dairy | 114.00 |
| Heifers >2 yr. | dairy | 176.00 |
| Dairy cows | dairy | 1,438.00 |
| Beef cows | beef | 265.00 |
| Sheep | beef | 11.00 |
| Pigs < 20 kg | breed | 71.00 |
| Sows and porkers for breeding | breed | 424.00 |
| Other pigs | slaug | 71.00 |
| Slaughter chickens | poul | 1.28 |
| Hens | poul | 2.86 |
| Turkeys, ducks and geese | poul | 3.74 |
| Fur animals | fur | - |

Source: Denmark's Statistics (1998).

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# 26. Using a Farm Accountancy Data Network in data management for LCA

*Krijn J. Poppe and Marieke J. G. Meeusen* [1]

*Abstract*

This paper discusses the usefulness of Farm Accountancy Data Networks for Life Cycle Assessment (LCA). It discusses which data can be found in the Farm Accountancy Data Networks and which are relevant for LCA. Several users are distinguished, each having their own requirements towards data. This forms the basis for requirements towards the data collection systems, of which the Farm Accountancy Data Network is one. The Farm Accountancy Data Networks can provide data from many farms at once and it can be used as a base for environmental models that help to estimate the emissions, which are necessary inputs for LCA.

## 26.1 Introduction

At the dawn of the third millennium, the agricultural sector faces two challenges: new scientific developments (e.g. biotechnology, information and communication technology) and - not unrelated - new demands from the society (e.g. requirements on environmental performance of products and production). Both challenges form the background of this paper where we discuss the use of the Farm Accountancy Data Network (FADN) for Life Cycle Assessment (LCA).

Executing LCAs of a(n) (agricultural) product requires a lot of effort and energy. In principle, a process sheet has to be made for each process that contributes to the environmental burden of the product. The process sheet covers the environmental and economic flows in and out the process. A lot of processes contribute to the whole life cycle of a(n) (agricultural) product and collecting data of all those processes is time and energy consuming, which makes LCAs still rather expensive. Even when we focus our attention just on the agricultural processes, where much more data are available than in other sectors, there is still a problem with data collection.

This paper discusses the use of the FADN for LCAs of agricultural products with the focus on the processes that occur within the agricultural sector. It starts with a discussion on the FADNs, as these data and the concepts used to collect them (large representative samples, typology, risk analysis) might be a useful additional source for LCA data management. After this introduction to FADN, we focus on the use of FADN data for LCA; we compare this type of data with data from non-accounting sources in agriculture, especially the engineering approach. The main theme of this paper is to suggest that the approaches used in (farm) accounting might be useful in the discussion

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'how to collect data for LCA process sheets', and to develop some suggestions on the circumstances under which the FADNs have the preferred data collecting systems for the LCA practitioner.

The paper is written in a provocative way to make an exchange of ideas between LCA practitioners and FADN managers possible. Based on experiences of the authors, the paper provides suggestions and ideas that now guide the authors in setting up their discussion within the group of FADN managers (with LCA practitioners). The background of first author (Krijn J. Poppe) is not in engineering, but in farm accounting and auditing and our institute LEI bases a lot of its research on accounting data. This might colour our paper.

## 26.2 Introduction to the FADNs

### 26.2.1 General

Farm Accountancy Data Networks (FADNs) exist in all EU member states, as well as in Norway and in Switzerland and they are set up in Central and East European Countries. Non-European countries often have comparable business surveys. The background of these FADNs is the need for micro economic farm level data to monitor and analyse the agricultural policy. A FADN is a representative sample of farms. In the EU 60,000 farms are sampled on request of the European Commission (CEC, 1989; Abitabile, 1999).

The data collection on these farms is based on farm accounting. The results are available in the form of a number of statements, e.g. a farm structure statement, a balance sheet, a profit and loss account, a cash flow statement, and (in some countries) a gross margin statement and a mineral balance. Such statements describe the situation of an individual farm in a certain year. Although research institutes have access to data on (individual) farm level (which allows them to investigate e.g. the income and wealth distribution), the results are available to the public only as aggregated or average results for e.g. a certain farm type and the results are used as statistical information.

In a number of statements, values, as well as quantities are available. Monetary values dominate in balance sheets and profit and loss accounts, but often quantities on e.g. production and number of animals are also available. However, between member states the FADNs vary a lot with respect to the availability of these data.

### 26.2.2 FADN and LCA data about inputs and emissions

In order to execute an LCA of agricultural products one has to collect data about the inputs required for agricultural production and the emissions that are caused by agricultural production. Both can be found with the help of FADN.

*FADN and data on inputs*

The current European FADN provides only a limited number of data that could be useful for those who are studying the environmental impact of farming. These data typically include stocking rates, the area cultivated with irrigation, production levels (for the main products also in volumes) and monetary inputs of pesticides and fertiliser. These data can be mapped to (crossings of) farm types, regions, altitude level, farm size etc. Although this amount of data and its usefulness is somewhat limited (especially for those researchers who have access to a much more detailed national FADN as in the Netherlands), interesting studies can be carried out to assess the environmental impact of farms. An example is Brouwer et al. (1995) estimating mineral balances for all FADN-holdings.

However, in recent years, FADNs showed an interest in collecting data for environmental purposes. These data can also be useful as an input to LCA. We mention the results of a recent project for the European Commission on the future of the EU-FADN, called RICASTINGS. It is clear from that study that five member states now collect mineral balances (Netherlands, Italy, Luxembourg, Portugal, Finland) and that another seven countries assess this as feasible if the finance would be available. It is also clear that eight member states have data on organic production, another six think this will be feasible. About half of the member states think that there is an interest in collecting data on pesticides, energy, and water, and that this is technically feasible (Abitabile et al., 1998). Van Lierde (1998) already made a study on energy use in the Belgian horticulture by using the Belgian FADN. This suggest that finance and organisation (bringing users and data providers together) are the main bottlenecks to have more data from FADNs available for LCA.

In the Netherlands there is quite a lot of experience in collecting data on the use of minerals, pesticides, energy, and water (Poppe, 1992). Already for many years now, accounting software in FADNs of research institutes like LEI in the Netherlands have collected these data. Data on the inputs mentioned above can be used to estimate emissions by using environmental models (see below). The data are collected on farm level. They are allocated to products, but this is not always done in the recording stage. Inputs are not recorded per activity, although Activity Based Costing (Schoorlemmer and Welten, 1998) could support this. In the Netherlands, this type of software has moved to the level of the farm or commercial accounting office. This is especially true for mineral accounting, where farmers are obliged to keep records on mineral flows, and have to pay a levy on surpluses. Compilation of these accounts benefits from special statements on the mineral content of products that are provided by farm suppliers, sometimes in an electronic data interchange (EDI) format. These statements are also used to audit the farmers' accounts (see Breembroek et al, 1996 for a detailed description of the system). The Dutch examples show that it is technically feasible to collect data on the environmental performance of a farm, on farm level.

### *FADN and data on emissions*

Farm accounting typically collects data on inputs and outputs that are potentially environmental damaging. However, FADN does not necessarily provide information on the emissions towards air, soil, and water. To estimate such emissions, agronomists use environmental models. Figure 26.1 shows that these models form an important link between farm level data and an LCA. Where an LCA could use some data (e.g. production volumes) directly from a FADN, models would be needed to estimate the emissions that result from e.g. a surplus of N on the mineral balance. It seems that most of these models are difficult to generalise and can not easily be linked to data of individual farms without further calibration.

In the Netherlands, we have experience with the development and use of environmental models based on Farm Accountancy Data Network, e.g. the MAM and the Stofstromenmodel, the results of which are being used in national environmental monitoring programmes.

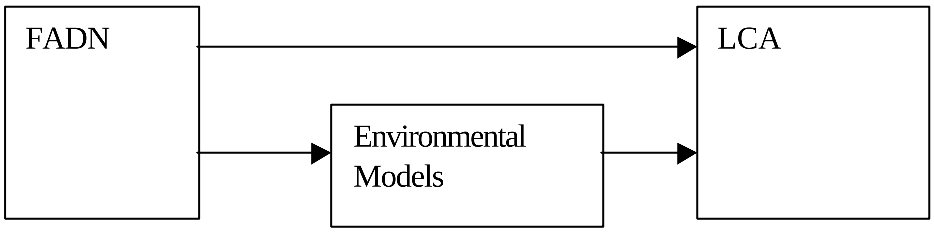


*Figure 26.1 Relationship between FADN, environmental models and LCA*

### 26.2.3 FADN-typology and LCA

The FADNs use typology as a system that classifies farms. The classification method largely depends on the application and the user. In agricultural statistics farms are classified to their technical-economic orientation, based on the share of the different technical production activities (e.g. sugar-beets, potatoes, wheat, eggs) in the estimated added value of the farm (Tiainen, 1998). This orientation (e.g. arable farms, specialist dairy farms) is calculated by multiplying the area of the crops and the number of animals with a standard gross margin (a 3-year average, standard for a region) and than looking to the share of different activities in that total farm added value (expressed in ESU - European Size Units). The total added value of the farm is also used to classify a farm in a certain size class. Typologies are also used for several regional dimensions (less favoured areas, 5B-regions, administrative regions like the NUTS nomenclature).

The FADN typology has an output-oriented component, which gives a useful link to the functional unit of LCA. However, for each study and each purpose of the study one has to ask whether the FADN typology is useful for the identification of farm systems on which the LCA has to be carried out. It can be assumed that better results for LCAs will be reached by dedicated typologies (e.g. intensive dairy farms on sandy soils). The only way to find this out is to perform these classifications and to look with multivariate statistical techniques whether better typologies can be developed.

### 26.2.4 The use of risk analysis [1] in accounting

Traditionally, LCA is very much an engineering tool where data are considered about all the processes that contribute to the total environmental burden. The same goes for process sheets: all the factors that contribute to the total emission of a process are taken into account. This makes data collecting neither very cost efficient, nor very easy with the data collecting systems that farmers have. We should consider more cost effective methods to collect data to carry out an LCA: 'Why spending 80% of the costs on the last 20% of the data?'

Accountants use a technique called risk analysis to investigate the relevance of the data collection activities. By examining the risks of making errors in a data collection activity, and studying the causes of these risks, it can be determined how the data collection should be organised so that the information needed is as accurate as possible, given a certain amount of costs for the data collection. This can for instance result in spending a lot of costs on making data collection as error-free as possible for an important process (e.g. applying $P_2O_5$ fertiliser in spring time or on the previous crop) and not much, or even nothing, on a process that only contributes marginally to the end result (e.g. use of phosphate in plant potatoes used as seed).

It is obvious that such techniques can only be used under two conditions:

- first, a certain knowledge about the contribution of different sources and inputs to the emissions and their environmental impact is necessary. This requires the availability of many LCAs: then there is a base for the selection which factors are really important and which factors are less important;
- secondly, it should be noted that risk analysis requires a clear priority of the environmental issues that have to be considered and those who have less priority. To illustrate: destroying a few trees might be less problematic in Finland than in the Netherlands.

As long as one can not distinguish one or a limited number of relevant factors that cause emissions and environmental effects and one has no general knowledge about the contribution of several activities to the emissions and their environmental effects, applying a risk analysis to improve quality (versus costs) does not make sense. Relative to other sectors, like the building industry, the packaging sector, and the automobile sector, the agricultural sector seems to have less experience that allows for risk analysis.

## 26.3 A closer look at FADN data and LCA

### 26.3.1 LCA-users and requirements on LCA data

[1] The use of the term risk analysis might be confusing in an LCA context. It is not an environmental risk analysis, but an analysis to improve the quality of the accounting data.

To design a data collection system that provides LCA data, it is necessary to take into account the user and the background of the information-need of that user. There are two reasons for this necessity:

1. Information systems must provide information *of a certain quality at an acceptable cost.* However, quality is a user-based concept: There is no such thing as 'absolute' quality - and quality comes at a certain price. Quality can be defined as 'the totality of features and characteristics of a product or service that bear on its ability to satisfy stated or implied needs' (ISO 8402).
2. LCA and data collection systems have a cost, and the user has alternatives if the collection system becomes too costly.

In this paper, we follow the user-based approach to answer the question: 'which data collection systems to use for with application?'

*LCA users and applications*

An LCA is carried out to provide a user or a group of (different) users with information. We distinguish two categories of users:

1. the agribusiness (retailers, food industry and farmers); and
2. the government.

For the agri-business (retailers, food industry and farmers) there is a number of decisions in which environmental issues and LCA play a role. There are decisions on product level - these decisions influence every firm in the production chain - and decisions on individual firm level.

1a. Negotiations with the government on environment regulations. The aim of the government is to reduce environmental effects of production. Therefore, the government wants to push firms to lower their emissions. However, firms are not always happy with such governmental interventions and need two sorts of information: Insight in potential strategies to cope with the effects of governmental intervention (see point d), and insight in their environmental performance compared to that of other sectors. By benchmarking with other sectors, they find arguments to reduce governmental interventions. This benchmark can be useful on product level (food compared to cars) or on sector-level (dairy industry compared to paint industry).

1b. Tracing and tracking. The issue of product liability and requirements from consumers for tracing and tracking becomes increasingly important. In case of food safety, the buyer exactly wants to know what activities have influenced the product(safety) and whether the quality control of such processes have worked (e.g. by installing ISO or HACCP procedures). This is not yet the case with environmental issues. However, one might expect this will be the case on longer term.

1c. Communication with the consumer - (Eco)labels. The environmental performance plays an increasing role in the quality-concept of (agricultural) products and an increasing number of consumers base their buying-decision on the environmental effects of the product. Therefore, consumers need information about the environmental impacts of products. The agribusiness considers two strategies to inform the consumer: Ecolabels, which cover only the environmental performance and where LCA is considered as the main tool in the procedure of developing labels (Green Goods V International Conference, 1998), and brand labels, which are based on the assumption that environment is not an issue as such, but an issue that forms part of the total quality of the product (safety, taste etc.), which is guaranteed by brand labels. For both strategies, firms within the agricultural production chain, especially those that produce for green-labelled products, face contractual obligations to report on the environmental aspects of their production. Until now, these obligations often centre on one product, for which (in an engineering approach) the activities have to be recorded. It has been argued elsewhere (Udo de Haas, 1996; Poppe, 1998) that - for farms - an ISO 9001 or 14000 procedure for the whole farm could make more sense, and can lead to audits that provides more guarantees and cheaper data (Meeusen-van Onna and Poppe, 1996).

1d. Improvement of environmental performance. All firms within the agribusiness are actively looking for options to improve the environmental performance of their agricultural products. In this decision making process several levels or stages can be identified: In the planning phase, the agribusiness needs information about the contribution of each activity/process to the overall environmental performance (Which processes contribute the most? e.g. the milk industry could ask: 'Is it the use of feed, the use of fertilizer, the use of energy in the milk production that contribute significantly?'). One can imagine that the outcome of such a study might lead to a revision of contracts (e.g. other criteria for labels), the implementation of other house-keeping systems (pigs in the Netherlands), purchase of other machines, lower stocking rates, to another transport system, moving production to regions were effects are smaller or to less environment friendly regions and moving out of this type of production. etc. These decisions often have a long term element. In the operational control in which 'every day' decisions are made. When the agribusiness knows that the use of fertiliser largely determines the environmental performance of potatoes, one needs information about the impact of the use of fertiliser in certain places, times and under certain weather circumstances. This information could be involved in the 'every day' decision processes. Consequently one needs information about the contribution of each process to the overall environmental performance plus information about how the contribution of each process can be lowered. Furthermore benchmark-information can be relevant. One can learn by benchmarking with (e.g.) the best 20% firms.

The governments need for information depends on the stage of the political process (see Meeusen-van Onna en Poppe, 1996 for more details). In the stage of problem recognition, there is mainly a need for fact finding to define and locate the problems. In policy formulation, representative monitoring systems and statistics have to be used in order to estimate the costs and effects of a proposed policy. The government should adapt its statistics and databases according to new realities as topics and policies change (Fletcher and Phipps, 1991). In the stage of implementing solutions, policies often evolve from extension and soft policies that include compensation for negative consequences of a policy, towards more severe, including the questioning of the necessity of production as such. Economic effects on micro-level are often a central issue in the discussion, as well as the efficiency of the policy. This requires detailed information. For example, the mineral accounting in the Netherlands, where farmers have to record the environmental impacts of their farm in an auditable way. The fulfilment of policies by e.g. farmers can demand simple taxes and auditable data for these levies. In the stage of control of the policy, the main need for information is monitoring, that leads to less detailed information needs than in the previous stages.

*Requirements*

The previous section provided an overview of the users of LCAs and the applications in which they use the LCA information. As every other information system, information systems must provide the required information of a certain quality at an acceptable cost (see section 1). The quality concept can be broken down into seven main criteria (Abitabile et al., 1998):

1. relevance: data are relevant when they meet the users needs;
2. accuracy: the closeness between the estimated value and the (unknown) true population value;
3. timeliness and punctuality: the need for up to date figures;
4. accessibility and clarity of the information: Accessibility is the best when data are available in the forms that users desire and when data are adequately documented;
5. comparability: Data of a certain characteristic have the greatest usefulness when they enable reliable comparisons of values taken by the characteristic across space and over time;
6. coherence: Common definitions, classifications, and methodological standards;
7. completeness: Users want a complete information system: the information system has to provide information on 'all vital aspects';

Finally, we consider the costs of the data collecting system in order to assess 'price-quality' ratio.

The quality criteria mentioned above are relevant for all LCA users and the applications of LCA in their decision making. However, in some applications certain criteria look more relevant than others. For example, an agribusiness needs data with much more detail when it uses an LCA for tracing and tracking or to improve environmental performance, than when using it for negotiations with the government to discuss an environmental bill.

Figure 26.2 provides some ideas about the relative importance of the quality criteria per user/application. We emphasise the fact that the scores are not based on scientific research; it is what we (and our colleagues) have experienced in our work.

| User/ application a) | 1 | 2 | 3 | 4 | 5 | 6 | 7 |
|---|---|---|---|---|---|---|---|
| Agribusiness: For negotiations with the government | | | Future-oriented | | X | X | |
| Agribusiness: Tracing and tracking | | X | Historical | X | | | X |
| Agribusiness: Communication with the consumer | | X | Historical | X | | | X |
| Agribusiness: Improvement of environmental performance | | | Future-oriented | | X | X | |
| Government: Stage of Problem recognition | X | | | | | | |
| Government: Stage of Policy formulation | | X | | X | | | |
| Government: Stage of Implementation policy | | X | | X | | | |
| Government: Stage of Control | | X | | X | | | |

*Figure 26.2 An estimate of the relative importance of criteria per user/application*
a) 1: Relevance; 2: Accuracy; 3: Timeliness and punctuality; 4: Accessibility and clarity of the information; 5: Comparability; 6: Coherence; 7: Completeness.

### 26.3.2 FADN versus other data collection systems

This section describes two data collection systems for LCA process sheets. It is based on the way data on costs of production for an individual product on farms are collected in FADN. This section translates the experiences in that area towards the way data about the emissions in process sheets can be collected. We distinguish two methods:

1. the (farm) accounting approach; and
2. the engineering approach.

We want to emphasise that these methods are in reality more complementary to each other than competitive. The so-called hybrid method that has been developed and applied at several Dutch universities, use of the top-down economic-statistical I/O-analysis *is combined with* the 'bottom-up' process analysis. They are used in a complementary way. However, in order to help the discussion and to make the differences more clear, we characterise them on their own and probably a bit distorted.

#### *Farm accounting approach*

The farm accounting approach (or the survey approach), is based on accounting information collected from a large sample of farms in a FADN. Every farm in the sample is representative for a group of farms (with more or less the same characteristics). This is secured by using a farm typology (see section 2). Information about these farms is (very) detailed recorded by the farmers themselves or the accountants.

Data on quantities are collected on both inputs and outputs, and can - with the help of the farmer - be allocated to products in the case that farms produce different products. Internal flows within farms (manure from animals to crops, straw from crops to cattle) are also recorded by farmers, although this can imply estimations at the farm. The emissions, specified to impact categories, is estimated by environmental models.

The (environmental) accountant is trained to work top-down, by looking mainly to relevance. Translated to LCA-data for process sheets one might think about the following procedure: the accountant tracks the 'responsible' factors for the emission and then assesses which factors have the highest contribution to the emission. He will then concentrate his efforts on collecting and auditing detailed data about these factors. Other factors will be recorded on a more aggregated level with fewer efforts.

Advantages of this approach is that results are representative for a (well defined) group of farms, that there is information about distribution of emissions, and that the information is auditable.

Disadvantages of this approach is that harmonisation of data is required (each farmer and/or accountant has to use the same rules to fill in the forms and information system) and that in some cases the environmental impact has to be estimated, especially if environmental models are not available.

*The engineering approach*

The engineering approach is based on technical coefficients for the (processes on an) average farm in a given region. Coefficients are often provided by experts, based on their experience and on a one-time questionnaire among farmers (that have to remember their 'normal' yearly practice when they answer the questionnaire).

The engineering approach works bottom-up. It is an inventory of all factors contributing to the total emission of a process.

The advantage of the approach is that the effort of data-collection is focused just on the technical coefficients. Consequently, one has only to know the (changes and developments of the) technical coefficients in order to draw up the process sheet.

Disadvantages of the approach are:

- that it can only be used for a short period of time because in the long run structural changes happen that go far beyond the change of individual technical coefficients so that other formulas have to be developed.
- that the results are not necessarily representative for the country/region as a whole. The average farm does not necessarily have average production and/or an average emission. When this causes too many problems, one has to define another type of firm (see section 5: typology). For example, to calculate the production costs, a 'modern farm' is chosen in stead of the average farm.

- the lack of information about the distribution of the emissions among farms, assuming that only a small number of farms are surveyed.

*Comparison of data collection systems*

The information provided above can be summarised in a comparison of the two approaches in data collection versus the quality aspects of information that we discussed before. Figure 26.3 provides our estimate of the relative differences between the two approaches.

In figure 26.3 we suggest that the engineering approach in data collecting for an LCA leads to very complete and relevant data at low costs, but the accuracy and coherence with other information can not always be guaranteed. The accounting approach has the disadvantage that it can be expensive and provides a historical view. It becomes even more expensive if one does not support the accountants' practice to concentrate on the most relevant emissions ('spot-light administration', see section 2), and wants data on all emissions allocated to all products and processes. The advantage is that distribution is available, representativeness is documented, data are audited, and integration with other types of data is facilitated.

The high costs of the accounting approach is a point for further discussion. If the data-collection for LCA can be seen as a by-product of the accounting information, a marginal cost

| Quality aspect | Engineering approach | Accounting approach |
|---|---|---|
| Costs of collection | Relatively low | Relatively high, due to high number of farms surveyed |
| Relevance | OK- (no distribution of variance of data is available) | OK+ (distribution of variance of data is available) |
| Accuracy | OK- (technical coefficient are often been estimated by experts - at best by questionnaires) | OK+ (data are audited, description of the representativeness if possible using the typology of the FADN) |
| Timeliness | Often more actual data or even future data | Based on historical data (unless extrapolation is carried out) |
| Punctuality | OK | Less, there is a risk of delay in the accounting process |
| Accessibility | OK | OK- (sometimes data are not available due to privacy restrictions) |
| Clarity | OK | OK+ (methods are often better documented) |
| Comparability in space | OK if well defined typology of farms | OK if well defined typology of farms |
| Comparability in time | OK on short term<br>Not OK on long term | OK |
| Coherence with other data | Often not, but definitions of emission models and LCA can easier be taken into account | OK |
| Completeness | OK+ (very complete;<br>all (sub) processes have been considered) | OK- (less complete due to the focus on major (relevant) processes, with a category 'other' for less relevant processes) |

*Figure 26.3 Scores a) of two data collection systems on quality criteria*

a) Symbols: OK stands for: a good score on this criterion; OK- stands for a good score but with one minor point (compared to the other data collection system); OK+ stand for a good score with an extra point (compared to the other data collection system).

calculation can be defended. This is especially the case in FADNs that provide policy makers and researchers with data. A second point is that the huge changes in information and communication technologies have a bigger effect on the accounting approach than the engineering approach.

### 26.3.3 Conclusion: Towards a contingency theory?

The information presented above raises the obvious question: 'Can suggestions be made on the choice of the best data collection systems to perform an LCA, given certain circumstances?' Based on the analysis of the use of LCA in section 26.3.1 and the analysis of the differences between the two data collection systems in section 26.3.2, figure 26.4 provides our suggestions for a contingency approach: 'In which case is the engineering approach superior to the accounting approach of data collection, and vice versa?'

The analysis suggests that the engineering approach for the LCA data collection systems is especially interesting for strategic producer decisions and in the problem recognition stage of governmental policy making. The engineering approach then delivers future oriented and rather cheap data on an average farm system. Accuracy of data is less important. A risk of applying the engineering approach in these cases is that too much time is spend on collecting information on processes that do not contribute to the overall assessment of the environmental impact of the production process.

| | Engineering approach | Accounting approach |
|---|---|---|
| Agribusiness: For negotiations with the government | X | |
| Agribusiness: Tracing and tracking | | X |
| Agribusiness: Communication with the consumer | | X |
| Agribusiness: Improvement of environmental perfomance | X<br>(strategic level) | X<br>(operational level) |
| Government: Stage of Problem recognition | X | |
| Government: Stage of Policy formulation | | X |
| Government: Stage of Implementation policy | | X |
| Government: Stage of Control | | X |

*Figure 26.4 A contingency approach in the choice of a data collection system for LCA*

The accounting approach for the LCA data collection system is especially interesting if a close look into the data of more than one firm is needed. If a food company would like to monitor the production process of all its supplier or even (as a chain leader) would like to improve the environmental performance in the food chain, or if the government would like to formulate and defend efficient policies, an accounting system is superior. A striking insight is that this need for detailed information will not always lead to a requirement for detailed data on the environmental effects of separate processes.

As environmental decisions are more and more incorporated into all types of decisions, and as the incorporation of environmental aspects in accounting is within reach with only marginal cost increases due to a number of innovations (see Poppe et al., 1997; Beers et al., 1999), a move from the engineering approach towards the accounting approach can be expected in the data supply for LCA.

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# 27. Use of farm accountancy data for monitoring energy consumption in agriculture

*Dirk van Lierde* [1]

## 27.1 Introduction

Although LCA, and more in particular LCA in agriculture, and farm accountancy at first sight have widely divergent objectives, they have at least one point in common: the search for and collection of data on inputs and outputs of agricultural productions. The collection of data on the farm level is usually very expensive. It is recommended to investigate if there are possibilities to collect the data for both disciplines at the same time instead of collecting data separately. In most of the countries, and especially those of the European Community, there are farm accountancy data networks (FADNs), especially used to observe the profitability of the agricultural holdings. In this paper the term FADN includes not only the collection of the accountancy data but it includes also the analysis and the research activities that are based on the collected data. These accountancies, which are not kept for tax purposes, mostly do more than just gathering the necessary financial values. Often there is also a technical aspect connected with the accountancies and attention is paid to physical outputs, use of raw materials, production systems, etc. The last few years, more attention is paid to the role of the farm accountancy networks for studies about the influence of the agricultural activity on the environment. The data model of the accountancy is in some cases extended from a pure accounting model to a data model that also includes data useful for, e.g. the environmental policy. Especially the mineral balances, the use of pesticides, and the use of energy have been focused (Poppe and Beers, 1996).

The FADNs also gather data about the consumption of raw materials just as fertilisers, pesticides, energy, etc. This paper examines the extent to which the data collected in the FADN can also be used for LCA, and in what way the processes of the FADN can be adapted to produce data for LCA. Specific attention will be paid to the collection of information about energy consumption. At the Centre of Agricultural Economics (CAE), a project is going on about the energy use in the Belgian agriculture and horticulture. At the moment, the energy consumption and the energy efficiency in the greenhouse horticulture are studied. This study is mainly based on data collected in the Belgian FADN. Later, the analysis will be further extended to energy consumption in other sectors of horticulture and agriculture (Van Lierde and De Cock, 1998).

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## 27.2 Objectives of FADN and LCA

Strongly simplified, it can be said that the FADNs study the economical aspects of the farm activities, while LCAs study the environmental aspects and potential impacts throughout the life of a product, from raw material acquisition through production, use, and disposal. As already mentioned, LCA and farm accountancies show an important common characteristic, namely the search for and the collection of data on inputs and outputs of agricultural productions. In a FADN these data are worked out in the economical field, while LCA translates the data to environmental aspects.

According to ISO, the LCA includes four phases: Goal and scope definition, Inventory Analysis, Impact Assessment and finally Interpretation (ISO, 1997). In the first two phases some common interests with the farm accountancies can be found. The first phase of LCA includes among other things the description of the system to be studied, the functional unit, the requirements with regard to data and data quality, and the need for critical review. These elements are also found in the farm accountancy, where a proper description of the system allows a better understanding of the coherence between the different system components. This facilitates the development of data models and a better control of the data. The second phase of the LCA, namely the inventory analysis, contains a.o. data collection and calculation procedures to quantify relevant inputs and outputs associated with the product system under study (Ceuterinck, 1998). In the farm accountancies, the inventory of the inputs and outputs and all the necessary data for this purpose are described. Although LCA and FADN partly gather data about the same items, there are some differences between them. FADNs mainly focus on the collection of quantitative data, expressed in monetary values. The LCA collects qualitative (what kind of raw materials and outputs?) as well as quantitative data (how much?). The quantitative data in LCA refer to the quantities of the products and not to the monetary values. Further on the information in LCA is completed with other data so the whole life cycle of the production process is included (emission coefficients, environmental effects upstream and downstream, etc.).

Although in the farm accountancy most of the attention is focused on the collection of monetary values, the collection of the quantities of the products used in the production process is becoming more important. These quantitative data allow a better control of the economic data, resulting in an improvement of the quality of the accountancy. On the other hand, the FADNs often have a double objective: on the one hand to supply economical data and on the other hand to gather technical data useful for the farm management. Despite the similarities in the collection of data on the farms for the farm accountancy and LCA, there are also some differences. An example is the drawing up of the use of minerals coming from chemical fertilisers and based on data of the farm accountancy (a first step to draw up mineral balances). For this purpose, one only needs the number of units N, P, and K used. In LCA, however, it will be also of major importance to know what kinds of fertilisers are used, because emissions to the environment during the use of the fertiliser and during the manufacturing of the fertiliser are dependent on the kind of fertiliser used. In fact these data are also available in the files of the accountants but as they were not considered very useful for accountancy purposes, they were not implemented in the accountancy data model and for that reason not transferred to the central databases.

## 27.3 Collection and processing of farm accountancy data

In the process of keeping farm accountancies, different stages can be distinguished. The situation, described shortly below, is the actual situation of the Belgian FADN and can be different from these of other FADNs (Van Lierde, 1998).

A. *The collection of the data on the farm*
The accountant frequently visits the farm to discuss with the farm manager the coherence of the data sent by the manager to the accountant by mail. At this occasion, the data are completed via invoices, by asking questions, etc.

B. *The processing of the data by the accountant*
In his office, the accountant notes down the data in the foreseen books, and forms and processes the data to a series of semi-gross results. Software developed for this purpose makes it possible that the accountant introduces the semi-gross results in his personal computer. This results in a computer file containing the data model of the accountancy. This computer file is sent to the Central Office of the FADN.

C. *The processing of the data in the Central Office of the FADN*
In the Central Office, the computer files coming from the offices all over the country are stored into a central database. The data in the central database, and only these data, are available for the purposes of further analyses and studies.

It has to be stressed, that only part of the information available in the files of the accountant is sent to the central database. At the moment, only the data necessary for the accountancy are sent to the central database. The rest of the information, not imported in the personal computer of the accountant, is not available for further research. However, the completion of the central database with this extra information could strongly extend the research possibilities of the central database. The availability of these data in the central database would offer research possibilities that would go much further than the field of the pure economical research.

But even if the data model is adapted for other purposes than pure economical purposes, this does not mean that it is suitable for LCA use. Again, the drawing up of mineral balances can illustrate this. For the moment, the data model of the Belgian FADN is adapted to collect the consumption of fertilisers per crop. The accountant records in his files for each fertiliser an inventory of the bought quantities, the used quantities, and the crop the fertiliser is used for. Finally, the accountant calculates how many units of N, P, and K are used for each crop. Together with the monetary value of the fertilisers, he imports the number of units in the data model. So, in the central database only the number of units per crop is available, but there is no information about the kind of fertilisers used. To make it possible to include in the central data model also information about the kinds of fertilisers, a much broader data model is necessary, together with a code list of all the possible fertilisers available on the market. Because of the wide variation in mineral fertilisers on the market, a complete inventory seems to be quite difficult. Besides, the system can be completed with and extended to the different kinds of organic fertilisers and manure originating from the own farm as well as from other farms. It is clear that even for this simple example of fertilisers, where all the information is available in the files of the accountants, it is not easy to transfer the information to the central database and a thorough adaptation of the software and data models is necessary.

## 27.4 Determination of the energy use for the individual farm

On a farm, energy is used in many different production processes. This energy can be used for different purposes: tractors, other machinery, lighting, heating, drying, cooling, etc. The accountant makes an inventory of the machinery and processes that use energy on the farm; this allows a better judgement of the data collected in relation with the energy consumption on the farm. On the other hand, an inventory is made of the different kinds of energy sources, such as electricity, fuel oil, gasoline, coal, etc.

In the beginning, only the monetary values were saved and sent by the accountant to the central database of the CAE. When the project on energy consumption started, the data model was adapted, so for the most important energy sources not only the monetary value but also the used quantities were sent to the central database. For the other fuels, which were not frequently used for heating greenhouses, the accountant was asked to note down the primary energy content in Joule. For this purpose, a list of the energy content of these fuels was made. Based on this information, the primary energy content of all the used fuels could be calculated. Later, in the second part of the study, the emission caused by this energy consumption was estimated. Since the emission coefficients are dependent on the kind of energy sources used, it seemed necessary to know separately, the quantity of each used source. For this purpose, an additional survey was necessary. The extra costs for this survey could have been avoided if the data model was immediately adapted to deliver the information of all kind of energy sources, and not just those who at that moment were the most important fuels used. So if the data model is adapted to gather data in a particular research field, one should aim for completeness of the data. The omission of what originally is seen as less important can give difficulties and inaccuracy when a later thorough study of the problem is made. Usually the collection of the missing data via extra inquiries asks extra efforts and costs (Van Lierde and De Cock, 1998).

The method that was used by the CAE is also very applicable for LCA purposes. With an eye on the calculation of emissions to the environment, it is recommended, for the completeness and the accuracy of the data, to note down some characteristics of the heating system. The quality of the heating system and the presence of filters or flue gas catalysts will influence the emissions released by combustion of fuels.

For most of the energy sources, there are no difficulties in measuring the used quantities; the delivered quantities are mentioned on the invoices. Also, the chemical composition of most of the fuels is quite stable. For other fuels, it is necessary to give a good description of the energy sources because different kinds exist, for example for heavy fuel oil. For a number of other energy sources, it is quite difficult to determine the quantity or the energy content. In the overview below, the most important energy sources and their characteristics are given:

*Energy sources with a stable composition and quantities easy to measure*
*Paraffin, gasoline, diesel* (litre).
*Propane, butane* (litre or kilogram).
*Electricity* (kWh, recorded by an electricity meter and mentioned on the invoice).

*Energy sources with different composition and quantities easy to measure*
*Heavy fuel oil* (kilogram). There are different kinds of heavy fuel oils on the market. These differences are especially the result of differences in sulphur content (1%, 3%, 3.5%), each kind of heavy fuel oil has his own energy content.
*Natural gas*. The quantity of natural gas measured in m³ does not give a good indication because the composition of the gas can differ in energy content dependant on the origin of the natural gas. Natural gas has to be mentioned in Joule in the data model; on the invoices, the consumption of natural gas is indicated in Joule.
*Coal* (kilogram). A clear difference must be made between the different kinds of coal, each with an own composition and energy content and resulting in other emissions (sulphur and dust). It is not always easy to obtain the right denomination of the coal, as the farmer often does not know this.

*Energy sources with different compositions and quantities difficult to measure*
*Straw* (kilogram, bales). If the straw is produced on the own farm the quantities must be estimated, if the straw is bought the quantity is mostly known.
*Wood.* The estimation of the right quantity and energy content of wood is difficult because the energy content differs from species to species. Moreover, there is a difference between blocks of wood and waste of wood or brushwood. Also the energy necessary to gather the wood has to be taken into account.
*Paper.* The quantities must be known and the right estimation of the energy content must be available.
*Waste products* (as waste oil). The determination of the quantities is not always obvious and even more problematic is to know the energy content and the composition of the products.

On Belgian agricultural holdings, especially energy sources of the first group are used, for the horticultural sector this group is completed with heavy fuel oil and natural gas. In other countries, the use of wood, coal, and straw can be important. To have enough information available in the central database about the kinds of energy used, an inventory of all the energy sources that can be used must be made and coded so each accountant can use them. Further, the data model must be adapted in such way that it can process the new energy source codes. The development of a very flexible data model that adapts the new added energy source codes (or other codes) without problems, requires an important investment in new software. New software for the personal computer of accountants as well as new software for the central processing department of the FADN has to be developed.

Finally, one should not forget that many activities on a farm (sowing, harvesting, drying, etc.) are executed by contractors. The contractors use their own machines, so the fuels necessary for their machines are not included in the farm accountancy. The accountant should keep a good inventory of the activities executed by contractors. Later, it will be necessary to estimate the energy use of the contractors on the farm. For this purpose the use of norms per unit (hectares, tons, etc.) is indicated. This means that for contractors' work, the accountant not only needs to collect the paid costs, but also the nature of the activities of the contractor and the numbers of units (hectares, tons, etc.) that were treated by the contractor. For the moment, this kind of information is not included in the CAE accountancy (and probably it is not included in most other FADNs).

### 27.5 Determination of the energy consumption for an average farm

If the energy consumption of the individual farm is determined, including the energy consumption of the contractors, the total of the energy consumption of an average farm of a country, a region, or a particular type of farm can be calculated. For this purpose an aggregation model is developed. The aggregation model is based on a stratified sample. The Belgian accountancy network is a representative stratified sample, and contains about 1,600 accountancies (on a population of about 45,000 farms). The stratification is done based on the farm type, the farm dimension, and the different agricultural regions in Belgium. Based on the data of the yearly agricultural census and the results of the accountancies different aggregation systems were developed. These aggregation systems make it possible to calculate averages for the different agricultural regions and farm types. In this way economical and technical parameters of the farms are aggregated (Goffinet, 1988; Mineur and Van Lierde, 1991). A same method, possibly refined to restrict aggregation errors, can be used for the determination of the average energy consumption of farms.

The types of farms, used for the aggregation, are those used by the European Community in the agricultural statistics and the accountancy (European Commission, 1985). The typology of the farms is used in many countries, which makes comparison between countries possible. However, it must be mentioned that this calculated energy consumption is related to the whole farm, and that the specification per product or production is not foreseen.

### 27.6 Determination of the energy consumption of a production

The determination of the total energy consumption of a farm can be done in a rather simple way. But LCA is not only interested in the total energy consumption of farms, there is also a great interest in the energy consumption of the separate productions. This is also a point of interest in the FADN, as one is also interested in the cost prices of the products. To make this possible, the costs have to be divided over the different productions. To do this as accurate as possible, an analytic accountancy is necessary where each cost item is assigned to one production. However, this method is very expensive. For a number of cost items, an allocation key must be used, and so the accuracy depends on the accuracy of the allocation key. Determining this allocation key necessitates additional research, and thus extra costs.

Many FADNs go further than a global accountancy system, but do not go as far as an analytic accountancy. It is mostly semi-analytical accountancies, which determine gross margins for different productions. The gross margin of a production is the difference between the gross production and the direct costs of the productions (seeds, fertilisers, pesticides, energy, etc.). The gross margin corresponds with the remuneration that is received for the input of labour and capital goods. For LCA purposes, it is interesting that in those accountancies most of the direct costs are known per crop or production, and that it is also possible to extend this to the quantities of each direct cost. For seeds, fertilisers, and pesticides, the farmer mostly knows the quantities uses for each crop. For seeds, this is obvious, for fertilisers it can be measured how many kilograms of each fertiliser are used on the field. For pesticides, this is also known, and the pesticides are mostly as specific for one crop so a good control is possible. Unfortunately, this is not always possible for energy consumption. The farmer knows how much diesel he puts in his tractor but it is not possible to measure the consumption of the fuel after each activity for a particular production. For electricity, the farmer knows how many kWh he uses. However, he does not know how much goes to the milking machine, to the lighting, to the heating lamps for the piglets, etc. To obtain all this information, a separate meter should be installed on each machine that uses electricity, which is not realistic. So it is clear that the partitioning of the total energy consumption of a farm over the different productions is not easy. Measuring the energy consumption per activity asks additional research and additional costs. To illustrate this, a number of possibilities to gain more information about the energy consumption per production are enumerated:

- the global energy consumption of the farms, expressed in Joule, can be allocated over the most important production groups by linear programming or multiple regression. The more detailed the production groups are, the less accurate the allocation is. Moreover, it is not possible to make an allocation of the different energy sources over the productions;
- in collaboration with the farm manager, one could try to allocate the total energy consumption of the farm over the different productions. It is suggested to focus on the activity that consumes most of the energy on the farm. Eventually an inventory of the capacity (power) of every machine and installation can be drawn up. In combination with the number of hours every machine or installation works during the year, this can be a helpful tool to allocate the energy consumption. However this is a very time-consuming and rather inaccurate method, and requires a lot of experience of farm manager and accountant;
- the previous method gives the best results on farms that are much specialised. On these farms, the total energy consumption of the farm is not much higher than the energy consumption of the most important specialisation, and energy consumption for this production can be estimated with a good precision. For example, on farms specialised in pig breeding, with few or no other activities, the largest part of the energy consumption is destined for the pig breeding. If there are a lot of such farms present in the farm accountancy network a lot of rather precise information on energy consumption in pig breeding will be available;
- also, extra instruments to measure energy consumption can be installed on the farms to focus on the energy consumption of particular machines or installations. However, this is expensive and belongs no longer to the activity or tasks of an accountancy network.

So, in a FADN the precision of the estimation of energy consumption will be less precise if one wants more detailed information. Estimations of the energy consumption with more precision will require more expensive methods.

## 27.7 Conclusion

The use of FADN data for LCA purposes offers a lot of possibilities. During the normal activities of a FADN a lot of information is collected that can be used for LCA research. Extending these activities would enlarge these possibilities. Data for the global energy consumption are available in the FADN (or can be obtained with little additional efforts). Data on energy consumption for crops and productions are much more difficult to obtain, as this requires a better measuring of the energy consumption on every farm. This is not a specific problem of FADN; it will always be difficult to obtain the energy consumption per crop or production, whatever the method is. The more precise one wants to work, the more expensive the collection of data will be. This means that it is probable that only a restricted number of observations will be available. As we know that farm activity is characterised by a great variability one will have to balance the advantage of more precise data with less observations, and the availability of a lot of observations with less precision.

Collecting data on farms is very expensive. In a FADN the infrastructure exists to collect data on farms, so it would be less expensive to extend the activities of the FADN for this purpose then to develop a totally new instrument that would almost do the same thing for LCA. A Life Cycle Assessment of farm data will probably confirm that collaboration is better than working apart. Collaboration of FADN's and LCA is suggested, also concerning budgets. Stakeholders of FADNs and LCA research should be aware of these possibilities. In the future, LCA and economics will work more and more in the same domain, because LCA will come up with choices for policies. These choices will have an economic impact that will be studied by economists (based on FADN data). So it is of interest to aim at collaboration between LCA and Farm Accountancy Data Networks. However, this will require a lot of energy.

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# 28. Development of a new management tool by combining LCA and FADN

*Stephan Pfefferli and Gérard Gaillard* [1]

*Abstract*

Life Cycle Assessment (LCA) for agricultural products is characterised by high data requirements. The quantity of all inputs must be known. The question is how these high requirements towards data can be met. All Western European countries have implemented efficient Farm Accountancy Data Networks (FADN). Monetary data is fully available, whereas physical information is usually only given for the outputs of agricultural production. It is proposed that existing experience and infrastructure of FADN should be used for LCA purposes. There could be synergistic benefits, e.g.

1. by obtaining the supplementary data at marginal costs; and
2. by creating a consistent decision basis for which all environmental and economic data could conjointly be established.

In order to be competitive in a liberalised market, the farmer can differentiate his product from that of other suppliers, e.g. by choosing a farming system with less environmental impacts. Life Cycle Assessment (LCA) can provide the necessary information, e.g. within the scope of an environmental management system. By aggregating the results of farm types on regional or national level, the environmental impacts of agriculture could be monitored.

## 28.1 Swiss FADN

The Swiss Federal Research Station for Agricultural Economics and Engineering (FAT) analyses every year the accounts of about 4,000 farms from all over Switzerland. The consolidated data and the results are used

- for the evaluation of the situation and the development of the farms' income;
- for the representation of the competitiveness of different farm enterprises;
- for the determination of the economic effects of planned and implemented agro-political measures by means of calculations and models;
- as information basis for the quantitative research in agricultural economics, for establishing the planning bases used in agricultural consulting, as well as for agricultural and fiscal taxation;
- in the field of agricultural training on all levels; and
- in the field of farm management (horizontal farm comparison).

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Although there are some differences between the FADN of EU-member countries and the Swiss one, the results can be compared as shown in the report 'Comparison of Farm Accountancy Data between Switzerland and the EU' (Meier, 1996).

## 28.2 LCA in Swiss agriculture

The LCA has been successfully applied to Swiss agriculture on process, farm and sector level. The following questions were examined:

- *Is the intensive, integrated or organic wheat production environmentally sounder?*
  Based on the study of Audsley et al. (1997), Gaillard and Hausheer (1999) have compared the environmental impacts of intensive wheat production in the U.K. to those of intensive, integrated and organic wheat production in Switzerland. Wheat cropping systems with a reduced input of fertilisers and plant protection products proved to be most favourable from an ecological point of view, provided that the yield reaches a certain level.

- *Are products made of renewable raw materials more favourable than conventional products from an economic and ecological point of view?*
  In the study Wolfensberger and Dinkel (1997), twelve alternatives were investigated. Where possible, the comparison covered the whole life cycle of the products. The comparison between 'rape for fuel (RME)' and diesel lead to the following conclusion: the great advantage of the 'rape for fuel' scenario lies in the technical feasibility not only as regards agricultural production but also from the point of view of its commercialisation and application. As far as the environmental aspects are concerned, there are no significant advantages or disadvantages. With regard to energy consumption, RME is somewhat more advantageous, however, not economically efficient.

- *Can Life Cycle Assessment be applied to entire farms?*
  The environmental impacts of 13 farms in the Western part of Switzerland were analysed. In addition, suggestions in view of improving their environmental compatibility have been made (Rossier 1998). Table 28.1 illustrates that the environmental impacts of the farms highly depend on the farm type. Within the scope of the project 'FADN and LCA' (see section 4), questions concerning the appropriate functional unit, the variance of the results of farms with comparable production and of those with various productions will be analysed in detail. If Life Cycle Assessment could be applied to farms, it could be used as a method for environmental review (Erb and Gerth 1997) within the scope of environmental management systems, according to ISO 14001.

- *Can the development of the environmental compatibility of the agriculture of a country be represented by means of Life Cycle Assessment?*

Rossier (1995) analysed this question, taking Switzerland as an example. For this purpose, he investigated the environmental impacts of the Swiss agriculture for the years 1939, 1970, 1980, and 1990. According to the ISO Standard issued in the meantime, the applied method does not meet the requirements of a Life Cycle Assessment because two important prerequisites are missing: a division of the examined system into process units and the fulfilment of the requirements in the field of data quality. The study is updated within the scope of the project 'FADN and LCA' (see chapter 4). Within this context, the influence of different functional units on the result is examined.

*Table 28.1 Global evaluation of environmental impacts of 13 Swiss farms*

| | Farm no | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 2 | 6 | 12 | 8 | 4 | 9 | 10 | 3 | 11 | 13 | 7 | 5 | 1 |
| Altitude (m ASL) | 1,030 | 1,000 | 760 | 650 | 400 | 480 | 800 | 500 | 500 | 450 | 400 | 370 | 400 |
| Utilised agric. area (ha) | 39 | 25 | 31 | 20 | 25 | 34 | 39 | 32 | 45 | 19 | 30 | 47 | 22 |
| Area used for crops (%) | 10 | 16 | 18 | 24 | 44 | 67 | 67 | 71 | 71 | 92 | 93 | 94 | 98 |
| Livestock units per ha | 0.9 | 1.3 | 1.2 | 0.7 | 1.1 | 1.0 | 1.7 | 0.6 | 0.9 | 0.4 | 1.1 | 0.0 | 0.0 |
| | Farm type | | | | | | | | | | | | |
| | milk production | | | | combined farms | | | | | crops and beef fattening | | crops only | |
| Environmental category *(emissions in % of the median/ha)* | | | | | | | | | | | | | |
| Fossil resources | **58** | **58** | 88 | 92 | 100 | 189 | 150 | 148 | 99 | 137 | 117 | 71 | 107 |
| Land use | 93 | 97 | 100 | 100 | 104 | 124 | 109 | 95 | 100 | 120 | 110 | 87 | 88 |
| Global warming | 63 | 55 | 87 | 84 | 131 | 160 | 166 | 133 | 100 | 119 | 160 | 57 | 75 |
| Photo-oxidant formation | 45 | 36 | 73 | 73 | 78 | 131 | 100 | 106 | 100 | 130 | 100 | 56 | 83 |
| Acidification | 88 | 67 | 107 | 100 | 107 | 173 | 200 | 106 | 83 | 83 | 135 | 14 | 30 |
| Aquatic eutrophication | 7 | 7 | 89 | 3 | 100 | 194 | 152 | 126 | 144 | 264 | 33 | 41 | 136 |
| Terrestrial eutrophication | 91 | 68 | 109 | 100 | 108 | 171 | 207 | 101 | 79 | 79 | 136 | 12 | 25 |
| Total eutrophication | 55 | 51 | 90 | 82 | 101 | 195 | 225 | 127 | 98 | 127 | 187 | 100 | 97 |
| Aquatic ecotoxicity | 45 | 43 | 35 | 35 | 189 | 181 | 100 | 184 | 64 | 260 | 118 | 167 | 92 |
| Terrestrial ecotoxicity | 88 | 43 | 130 | 40 | 361 | 662 | 100 | 285 | 25 | 164 | 37 | 118 | 68 |
| Human toxicity | 37 | 32 | 49 | 54 | 71 | 142 | 133 | 131 | 100 | 119 | 139 | 100 | 141 |

a) Very favourable; Favourable; Comparable; Unfavourable; Very unfavourable.

The adaptation of the LCA methodology for application in agriculture is in progress. In collaboration with other partners in Switzerland, the FAT currently investigates the following issues:

- *Are mechanical weed control techniques environmentally sounder than chemical ones?* The first results (Gaillard and Irla, 1998) show that mechanical or mechanical-chemical weed control techniques are particularly favourable for maize and rape whereas in the case of potatoes no significant difference exists between the investigated techniques.

- *Which is the optimum quantity of nitrate in wheat production from an ecological point of view?*
  Given the surface as functional unit, an input of 100 kg N per hectare and year entails less environmental impact than an input of 220 kg. If the functional unit is defined as 1 equivalent tonne grain with 13% protein, the environmental impacts decrease per produced unit of wheat with increasing nitrate input (100, 140, 180, 220 kg N per hectare and year, Charles et al. 1998).
- *Which are the specific problems of the Life Cycle Assessment applied to organic farming?*
  One of the main items is the implementation of additional environmental categories in comparison to the study of Audsley et al. (1997), such as biodiversity, landscape and soil fertility, in order to globally assess the environmental impacts of organic farming.

The aim of all projects mentioned is to achieve a global assessment of the environmental impacts of the agricultural production. With the exception of the study of Wolfensberger and Dinkel (1997), which examines the ecological and economic impacts, the studies focus on ecological impacts only. This is, however, not sufficient for a global assessment according to the concept of sustainability: ecological, economic and social aspects are to be taken into account to an equal degree. By combining Farm Accountancy Data Networks with Life Cycle Assessment, the two first-mentioned dimensions of sustainability could at least be represented. Experience showed that projects which combine the LCA analysis with economic tools are at best in position to act on the agricultural decision-makers to improve the processes under consideration.

### 28.3 How to combine FADN with LCA?

Swiss FADN receives the following data from farms:
- general information (location, quotas, ...);
- monetary data in a consistent form (receipts and expenses of the whole farm household: farm enterprise, para- and non-agricultural enterprises, salaried off-farm work, private household);
- area sizes of different crops and green land;
- number of animals for different species and categories;
- crop yields and animal performances;
- working days per person of all people working on the farm.

FADN provides a framework for consistent data analysis. Monetary data is fully available, whereas physical information is only given for the outputs. Fully available monetary data and partially available physical data allow to carry out extensive plausibility and consistency checks. In comparison to restricted and specific inquiries, this quite good data situation could even be improved by getting more input data (a present lack of FADN systems) needed for the LCA analysis. The infrastructure of the FADN should be used to get the necessary data regarding agricultural production.

Environmental improvement of production methods usually implies higher production costs. The advantage of the combination between LCA and FADN is to create a consistent decision basis for which all environmental and economic data is conjointly established.

Like accounting, LCA is a management tool and could also be used as a monitoring instrument. The present state of methodological development, as described in detail in the ISO-norms or norm drafts 14040 to 14043, combined with experience already gained for agricultural processes, is basically sufficient for an implementation in a FADN system. The integration of LCA in the FADN framework could generate synergies between both applications.

## 28.4 How to achieve synergies between LCA and FADN?

The aim of the new project 'FADN and LCA' (Pfefferli 1998) is to develop a management tool allowing to support the self-responsible acting of the farmers instead of creating new legal restrictions. Taking into account the economic and political environment, the farmer determines which inputs are to be used and implicitly which environmental impacts they involve. The combination of FADN and LCA is supposed to achieve a responsible and well-informed farm management.

For a broader application of LCA on farm level, the following problems have to be solved:

- improving the data available for emissions related to agricultural buildings and machinery;
- developing new environmental impact categories such as biodiversity, landscape and soil fertility;
- simplifying data collection and calculation of a LCA on farm level;
- showing the farmers the utility of such a management tool;
- choosing an appropriate functional unit.

In order to use LCA as a monitoring instrument together with a FADN system, which is another goal of the project 'FADN and LCA', the following challenges have to be mastered as well:

- defining a sample of farms;
- elaborating a method to aggregate the LCA results of selected farms to relevant and significant results on regional level.

These issues are topics of current projects (section 2). Some reflections related to the five last mentioned points are presented in the following paragraphs.

*a) Simplifying data collection and calculation of a LCA on farm level*

As stated above, the main lack in FADN is missing physical input data. The invoice provides indeed monetary as well as physical data concerning the inputs bought by the farmer. However, only monetary information are recorded in the accounts. On the other hand, an increasing number of farmers use a plot register (PC software) to record the physical amounts of used fertilisers and pesticides. In Switzerland, farmers receive direct payments only, if they are able to prove their environmental performances. Requirements for this are e.g. equilibrated nutrient balances for N and P or an ecological compensatory area of at least 7% per cent of the utilised agricultural area. Required data is still available, but not in an adequate form for data-processing. To simplify the administration and handling of the data, extension services in Switzerland are developing a farm database (Graf and Keller, 1998). Their objective is to collect all relevant physical data of the farm. If the farmer has to apply for direct payments, the required data concerning area sizes and number of livestock are retrieved from the databank and filled in the corresponding form. The same procedure can be used for preparing the tax declaration or the data (physical amounts of inputs) needed to calculate an LCA. If there was another database containing the emissions of the inputs used in agricultural production (see Gaillard et al., 1997), it would be possible to automate the calculation of an LCA on a farm level together with a FADN system and the corresponding economic assessment. The question remains to know whether a standard LCA software supplied on the market can be adapted (interface for data input) or a specific application software has to be developed.

*b) Showing farmers the utility of such a management tool*

The more the market for agricultural products is liberalised, the more it is important for each farmer or branch association (i.e. dairy farmers) to differentiate the own product from that of other suppliers. This can be made by means of the chosen production method and the related environmental impacts. LCA is able to provide the latter information. In order to assess his own farm, the farmer can compare his present LCA results to those of former years (vertical comparison) or of farms with a similar production structure (horizontal comparison). The horizontal comparison is relevant to prove that his production is environmentally sounder than that of his competitor. Here the question rises, how a farm typology should be defined to allow a correct comparison.

*c) Choosing an appropriate functional unit on farm level and creating an farm typology*

The results of the LCA primarily depend on the chosen functional unit, the amounts and the types of the needed inputs (e.g. mineral fertilising or manure) and the site-specific environmental impacts. If the area (a hectare) is chosen as functional unit, the type of production (determined by the composition of the crops and livestock and the relation between them) is a relevant criteria to get an appropriate farm typology, whereas the size of the farm, the ownership or the production method should only play a minor role. The location seems to be relevant as well, because for example the environmental impact of nitrogen input on ground water (nitrate leaching) is not the same for sandy soil as for clay soil.

The area does not take into account the quantity and quality of the products of a farm. For this, a solution could be to go more in detail and to perform the LCA on the product level, but this procedure implies very complex allocation problems already known from the calculation of production costs. The most promising solution seems to be the use of a limited number of units corresponding to the different functions of the farm. Their number depends on the diversity of the production program of the respective farm. Possible units would be kilogram of energy corrected milk (ECM), kilogram meat without bones, kilogram eggs, joules of energy in crop products, kilogram dry matter of fruits, berries and vegetables. This approach would offer the advantage, on the one hand, that all similar products (e.g. veal, beef, pork) could be summed up in an ecological assessment (unit meat) and, on the other hand, that these units could be used together with the location of the farm in order to constitute different farm types. The question of a combined unique functional unit is however not solved.

*d) Defining a sample of farms for monitoring the environmental impacts of agriculture on a regional level*

The ideal case would be to draw a sample of farms at random. However, as long as the LCA is not applied for a high percentage of farms and if it is not possible to fully indemnify the concerned farmers for their additional effort, this method is not practicable. The same problem is known from most of the FADN in European countries. For LCA purposes, the solution could be the same as for the FADN, i.e. a stratified selection plan. As the calculation of LCA on a farm level is relatively recent and therefore still time-consuming, only very few results are available.

The number of farms needed for monitoring depends on the variability of LCA results in the farm groups. Therefore, this parameter should be evaluated for farms with a similar production structure. To aggregate the results of farm types on regional level, they have to be weighted according to the number of each farm type per region.

## 28.5 Conclusion

The combination of the LCA and FADN tools offers a number of new perspectives in order to efficiently promote a sustainable agriculture by putting at the farmer's disposal a useful, complete and consistent management tool in the economic and environmental fields. If the objective is to develop the LCA in view of developing a broadly used management tool on farms and also to take advantage of the resulting data for horizontal farm comparison and for a monitoring system of environmental impacts of agriculture on regional level, then the existing infrastructure of FADN systems should be used. However, a number of problems remain to be solved which are similar to those which arose when FADN were developed. Therefore, a lot of experiences are available.

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# 29. Conclusions of the working group on farm typologies

*Erwin Lindeijer [1] and Bo P. Weidema [2]*

A farm typology is a stratification of the farm population with the following aims:
- to lower data variability;
- to allowing a better selection of representative farms for detailed research;
- to better determine the marginal effects of a studied change.

Input data should ideally be available through the Farm Accountancy Data Network (FADN), which use a typology that is based on technical-economic criteria, i.e. a combination of production activities and the gross margins of those activities related to the total gross margin of the farm.

Consequently, FADN uses a product output related approach, which partly fits with the LCA requirements of relating environmental effects to the product outputs.

However, several organisational problems need to be overcome before more detailed information can become available:
- additional costs, both in data collection and centrally;
- farmers acceptance of more Farm Accountancy documentation;
- privacy issues in relating Farm Accountancy Data with Geographical Information Systems for correlations with soil types and economic efficiency.

By-products with near-to-zero value are not included in the Farm Accountancy Data Network.

Outputs to the environment are derived from models, which in some countries are now linked to the Farm Accountancy Data Network. In some cases, 'models' may consist of simple emission factors. At present, models are not well harmonised and they are dependent on the available input data. Ideally, the same models and farm types should be used in all countries, although possibly with country-specific deviations.

It is proposed to base further harmonisation on the achievements of the countries most advanced in integrating Farm Accountancy Data Network with modelling. Default estimates for less advanced countries may be derived by combining these models with the available data in these countries.

Links to Geographical Information Systems for correlation with soil type is possible in the Netherlands and Switzerland (and soon also in Austria). Further harmonisation, also of nomenclatures, is needed.

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It is proposed to combine data on soil type with a detailed farm typology from the Farm Accountancy Data Network, based on economic outputs and efficiency. Additionally, for each environmental issue, the best, average and worst fractals may be distinguished. Remaining gaps should be filled in with data from experimental farms, e.g. for heavy metals.

# 30. Conclusions

*Marieke J.G. Meeusen [1] and Bo P. Weidema [2]*

The papers presented in this book give an overview of the state of the art. In this final chapter, we will not repeat the facts and conclusions that have been presented in the separate papers and the conclusions of each preceding chapter. Instead, this concluding chapter will analyse how well the questions stated in our introductory chapter has actually been answered, i.e. focussing on:

1. the state of the art concerning availability of parameters and models across environmental impact categories, i.e. across the issues treated in the preceding chapters;
2. the way data can be aggregated at different levels and calibrated against regional statistics;
3. the immediate recommendations for Life Cycle Assessment practitioners;
4. the future research needs.

## 30.1 State of the art concerning availability of parameters and models

The first two questions we set out to answer were:

- How can the environmental data best be modelled to the outputs of individual crops and animals?
- What are the most important parameters determining differences in product related environmental data?

For energy consumption, key parameters that determine the energy use and energy consumption have been identified in the papers and conclusions presented in section B (chapters 4-7). The relevant parameters are known and their influence is to some extent modelled, especially for field operations, while energy use in stables is less known. The models often have a bottom-up approach, considering all processes that use energy and for each process requiring the consumption per hour, the time required for operating, etc. This means that processes that have negligible contributions are also being considered. This could lead to unnecessary use of time and budget in LCAs. Furthermore, the models have been developed independently in each country with very little harmonisation in methodology.

Also for the nitrogen cycle (section C; chapters 8-14), the emission types are identified, and for each emission type, it is known what parameters that influence the emissions. Models are available for most fluxes of nitrogen emissions, but these models are often applicable only within certain boundary conditions (certain soil types, climates, geographical situations), typically relevant for a specific country. There is a need both for more generalised models per emission type, allowing comparisons

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across several countries, and for an integrated model that treat all the nitrogen emissions at the same time.

The research on phosphorus and heavy metals (section D; chapters 15-20) is more recent and therefore not yet as far as for nitrogen and energy. On several levels (globally as well as detailed) the knowledge about the way phosphorus moves and cause environmental effects is being developed. No general models are available yet although several projects are underway, including an EU concerted action to stimulate the development of models across EU. For heavy metals, the state-of-the art is quite similar to that of phosphorus.

In general, one can distinguish different types of key-parameters that play a role:

- utput parameters (product types, product characteristics);
- eographical parameters (soil type, climate, slope);
- arm management parameters;
  - on long term (farm type; available labour, land and capital; farm structure),
  - on mid term (purchase of machinery),
  - on short term (use of fertiliser, use of pesticides).

We have seen (section F; chapters 25-29) that the Farm Accountancy Data Networks use a typology that is based on technical-economic criteria, i.e. a combination of production activities and the gross margins of those activities related to the total gross margin of the farm. Consequently, FADNs use a product output related approach, which partly fits with the LCA requirements of relating environmental effects to the product outputs. FADNs also have data for many of the key parameters mentioned above, especially the output and farm management parameters. The conclusion is that the FADN-typology can be a good basis, although additional criteria are necessary to make the typology suitable for LCA covering all the key parameters mentioned above. Soil type, climate (for example rainfall), and other geographical characteristics (e.g. slope), may be some of the additional criteria that have to be considered.

## 30.2 How can data be aggregated at different levels and calibrated against regional statistics?

This was the third question stated in our introductory chapter. The conclusions of the seminar can best be expressed by quoting the words of Halberg et al. (this volume):

'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 representativity 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:

1. 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;
2. check the representativity 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;
3. 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;
4. calculate emissions based on the farm models and best knowledge of emission processes;
5. check and adjust partial emissions of nutrients with balances at farm and enterprise level;
6. 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 FADNs could be a very useful data-source also for identifying the different farm types.

## 30.3 Immediate recommendations for LCA practitioners

The fourth question we set out to answer is the one most interesting for Life Cycle Assessment practitioners

- What data are available today? or 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?

In spite of the relatively large amount of knowledge on the factors influencing energy consumption in agriculture, surprisingly few data are readily available for LCA purposes, i.e. on crop and/or product level. Nielsen and Luoma (this volume) give some data on field operations (fuel consumption in litres per hectare), generally based on actual measurements. Another recent source of data, not cited in section B, is Borken et al. (1999), using a modelling approach taking into account also the different loads on the machinery, as also suggested by (Audsley, this volume). Of the different parameters influencing the fuel consumption for field operations, soil type was identified as one of the more important (see e.g. Vitlox and Michot, this volume) and it was suggested that data from soil maps may be included in the models (Cortijo, this volume). For the modelling, a key parameter is the number and type of field operations. Today, the default source of such data are national farmers' handbooks, like the KTBL (1994) cited by Moerschner and Gerowitt (this volume). Working depth for soil cultivation is a local parameter of large importance, for which local expert knowledge is typically the only readily available source of data. For energy use in stables, the model developed by Dalgaard et al. (1998, see also Halberg, this volume) seems to give a valid representation of actual energy consumption, although the model has not yet been validated outside its country of origin.

For nitrogen, the recommendation is to distribute the N surplus (N input minus N in crops) over the possible outflows using the currently best available models for each flux: The MARRACAS-model for ammonia, the SLIMMER-model for nitrate, the IPCC-procedure for nitrous oxide (see Ceuterick and Weidema, this volume).

For phosphorous, the link between surplus and loss is not as clear as for N, due to differences in the patterns of flow and retention of P in the soil. Nevertheless, it can be concluded that the greatest risk of P loss is on farms, which have a large P surplus due to inputs from manure, and that erosion is the main process of P transfer from agricultural land to water (Withers, this volume). Until specific models become available (see Cowell, this volume), the best default values available seem to be those of Chambers (1997, see also Heathwaite, this volume).

For heavy metals, no default values can be recommended yet, mainly due to the lack of plant heavy metal accumulation factors (Japenga and Römkens, this volume).

For pesticides, statistics on actually applied amounts are still only available for a few countries, implying that estimates for the time being often must be based on recommended doses and experts judgement, possibly with the aid of the calculation method suggested by Audsley (this volume). The fractions of the applied quantity of a pesticide that reach the different environmental compartments can be estimated by the method suggested by Hauschild (this volume), which includes default values and is based on readily available data.

## 30.4 Future research needs

This leads us to the ultimate question of our introductory chapter:

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

For all the environmental aspects discussed, it is agreed that the Farm Accountancy Data Networks (FADNs) should play a larger role:

- FADNs can already now be used as a data-source. FADNs covers several data, which are useful for LCAs, e.g. inputs of energy, fertilisers, pesticides etc.;
- FADNs comprise some data that can be considered as a (key)parameters for calculating product specific energy consumption, emissions of nitrogen, phosphorus etc.;
- FADNs use an output oriented typology, which fits with the product oriented approach of LCA. Therefore, FADNs form a good base for farm typology, which can be used within LCA. However, in some cases some adjustments and additions in some cases;
- data from FADNs form a good base for modelling the emissions that occur within agricultural processes. Examples in the Netherlands and Switzerland have shown that emission models often can be linked to a FADN. Also links to the Geographical Infor-mation Systems (GIS) for correlation with soil type (another key-parameter for many environmental aspects) is possible;
- finally, FADN can be used for stratifying farms. For each environmental issue, (a) the best; (b) the average and (c) the worst fractiles can be distinguished based on physical efficiency: the

physical output in relation to the physical inputs (of for example energy, nitrogen, phosphorus, etc.).

Several organisational problems need to be overcome before more detailed information can become available through the FADNs:

- additional costs, both in data collection and centrally;
- farmers acceptance of more Farm Accountancy documentation;
- privacy issues in relating Farm Accountancy Data with Geographical Information Systems for correlations with soil types and economic efficiency.

For a number of environmental aspects, development of models is still hampered by lack of data on which to base the models, e.g. for heavy metals, physical habitat disruption, and occupational health.

For all the environmental aspects discussed, models are often empirically based (and dependent on the locally available input data) rather than based on proven, general relationships. Also, the models are not well harmonised. Thus, models still need to be developed, and existing models need to be improved, integrated across substances, and harmonised across Europe. Ideally, the same models and farm types should be used in all countries, although possibly with country-specific deviations.

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