ENVIRONMENTAL LIFE CYCLE ASSESSMENT
OF AGRICULTURAL SYSTEMS:
INTEGRATION INTO DECISION-MAKING

Thesis submitted for the Degree of Doctor of Philosophy
by
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Environmental Life Cycle Assessment (LCA) is an approach for assessing the comprehensive environmental impacts of human activities; effects are quantified along the life cycle from extraction of raw materials, through processing, manufacturing, transportation, use and on to final disposal. LCA was developed for assessing industrial systems, and agricultural systems are sufficiently different that this area of application introduces new methodological issues for all phases of LCA. These issues are addressed in the thesis; additionally, it explores issues related to the wider use of LCA in decision-making.

New methods are presented for assessing use of solar energy and water, soil quantity and quality, and biodiversity. Use of solar energy is assessed in relation to total incident radiation, and use of water in relation to average annual rainfall reaching land in a system under analysis. Soil quantity and quality are assessed assuming that soil is an ancillary item in LCA; this requires careful modelling, use of Organic Matter and Soil Compaction Indicators, and inclusion of eroded soil in assessing abiotic resource depletion. A method for assessing physical habitat maintenance and change is presented which highlights some generic features of LCA mitigating against its acceptance among some stakeholders. A case study of breadmaking wheat production demonstrates practical application of the methods. This suggests it may be equally, or even more, relevant to determine preferred locations of production rather than preferred farming practices in seeking to maximise the environmental performance of agricultural systems.

More attention is needed to ensuring the usefulness of LCA results. They should be accurate, relevant, understandable and meaningful to stakeholders, and the LCA approach must be accepted as a legitimate form of analysis. This requires more flexibility in LCA methodology to adapt it to different decision-making contexts, balanced by a greater focus on the process of undertaking LCA.
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CHAPTER I
INTRODUCTION

1. Introduction

In recent years, debate over the interaction of human activities with the “natural” environment has acquired a new focus which goes beyond recording and analysing environmental damage. One of the key concepts in this new focus is the idea of sustainable development as first articulated by the World Commission on Environment and Development (WCED) (the “Brundtland Commission”). They defined sustainable development as development that meets the needs of the present without compromising the ability of future generations to meet their own needs (WCED, 1987, p.43). In this form, their declaration is more of a statement of principle than a definition, and there continues to be much debate over definitions of “sustainability” and “sustainable development.” However, it is generally agreed that sustainable activities must be developed based on a consideration of environmental, social and economic factors. This is illustrated conceptually in Figure 1.

Figure 1. Aspects Considered In Sustainable Development

![Figure 1](image)

Source: Cowell et al., 1997.

The elusive definition of sustainable development, and the related difficulties of defining criteria for judging the sustainability of human activities, have led to the emergence of a number of environmental management approaches. These can be broadly organised into two categories:
• **Concepts** such as Cleaner Production, Clean Technology, and Industrial Ecology. A concept can be described as “an idea about how to achieve sustainability.”

• **Tools** such as Environmental Impact Assessment, Risk Assessment and Substance Flow Analysis. A tool is a more specific type of assessment, typically consisting of a systematic step-by-step procedure and a mathematical model (Baumann and Cowell, forthcoming; SETAC-Europe Working Group, 1997).

One approach that has received considerable attention, and which is now being integrated into company and governmental policymaking, is the environmental life cycle approach. At a conceptual level, it is called “life cycle thinking,” and approaches such as Product Stewardship and Producer Responsibility are built on this concept. However, the need for a more rigorous application of life cycle thinking to evaluate products and services has led to development of the tool Life Cycle Assessment (LCA).

With regard to defining the sustainability of different types of human activities, one sector with a diverse and important range of environmental impacts is food production, distribution and consumption. Various studies have provided some indicators of the magnitude of the overall impacts associated with the food chain. These impacts include:

• Agricultural land comprises 77% of the UK’s total land area (MAFF *et al.*, 1994), and hence has an important role to play in preserving biodiversity by providing a variety of habitats that support different species.

• Two different estimates of energy consumption throughout the food chain, undertaken in the 1960s and 1970s, calculated that the food system uses 21% and 28% of the UK’s total energy consumption (Leach, 1976, and Blaxter, 1977 respectively). These two values were based on energy consumption in agriculture, food processing, food distribution, and home preparation (and waste disposal in the latter estimate). Environmental impacts related to energy consumption include global warming, acidification, photochemical oxidant formation, and non-renewable resource depletion.

• Over 20% of landfilled and incinerated waste in the UK is related to the food system (authors’ calculations based on data in Atkinson and New, 1993, and DOE, 1993; see Appendix 1.1). The environmental impacts associated with management of this waste include space consumption, water and air emissions, and energy-related impacts during transportation.

• About one third of phosphate in rivers derives from agriculture (ENDS, 1996), and 65% from sewage (of which one third is from phosphate detergents) (ENDS, 1997); and the main source of
nitrate in water is agriculture (DOE, 1994, p.63). These nutrients both contribute to eutrophication (although their role is dependent upon local conditions).

- Chlorofluorocarbons (CFCs) are ozone depleting chemicals and potent global warming gases, and one of their major applications is refrigeration of food. Environmental impacts may arise through release of CFCs from faulty equipment and/or irresponsible disposal.

Given the significant role of the food chain, based on indicators such as those listed above, this sector deserves further consideration from an environmental management perspective. In particular, studies of this type are likely to be useful because of the variety of food production, processing, distribution, and consumption patterns. Furthermore, these patterns are capable of changing rapidly in response to government policy-making and regulation, commercial activity in the marketplace, and shifts in public perception. Thus, this sector is capable of responding to initiatives leading to more environmentally sustainable activity.

However, any discussion of sustainable activity usually raises a myriad of further questions. What are the goals of a sustainable food production system? What steps should be taken in moving along this development pathway? What actions should be prioritised? In many ways, we are only just beginning to grapple with these types of questions. It is the role of environmental analysts and managers to sift through the often disparate and conflicting ideas, to arrive at a better understanding of the underlying issues and values that shape our society, and to present the results in an accessible way for decision-makers.

In response to the issues sketched out above, there are therefore three themes that run through this thesis:

- Development of LCA as an environmental management tool
- Assessment of agricultural systems using LCA
- Application of LCA in decision-making.

I introduce each of these themes in more detail in the sections below.

2. Development of LCA Methodology

The basic approach in life cycle thinking and LCA is demonstrated in Figure 2. The conventional approach to environmental assessment is represented by system boundary 1, drawn around a manufacturing process, plant, or factory. However, the approach exemplified by system 1 is too
limited to assess sustainability. The materials and energy used in production must be obtained from primary resources and processed before use, while any products have further environmental impacts in the way they are used and ultimately recycled or disposed. Therefore, LCA extends the system to boundary 2 in Figure 2. In this way, the “cradle-to-grave” environmental impacts of any products or services under analysis are considered as part of the life cycle.

Figure 2. Generic Flow Diagram for Life Cycle Thinking and LCA

There are a number of unique advantages of defining the system boundary in this way, including:

- In evaluation of existing systems, “hot spots” in the environmental life cycle of any one system become obvious, facilitating prioritisation of activities to improve its environmental performance.
- In considering improvements to a system, any trade-offs are revealed between improvements at one stage, and increased impacts at another stage of the life cycle.
- In comparing two alternative systems with major environmental impacts at different life cycle stages, the assessment gives a comprehensive overview of the trade-offs between the two systems.

A more detailed explanation of LCA methodology is given in Chapter II. The overall approach can be described as a type of systems analysis (Baumann, 1995). In systems analysis, an object of study is taken to consist of a number of components that are mutually related. This object cannot be studied simply by sub-dividing it to analyse its components then putting them back together. Instead, the object is considered as a whole, and the inter-relationships of its components are studied alongside
their relationships to the whole object\(^1\) (Baumann, 1995, p.20-23). Describing LCA as a type of systems analysis also means that it is concerned with "the art of applying scientific methods and knowledge to complex problems arising in public and private enterprises and organisations and involving their interactions with society and the environment" (Quade and Miser, 1985, p.20). As such, this confirms the positioning of LCA in the shaded area of sustainable development shown in Figure 1, influencing — and being influenced by — the three lobes of the natural and physical sciences, social, and economic concerns. It means that relevant environmental impacts must be included in the analysis, and that the definition of "relevant" is rightly influenced by societal concerns. It is therefore appropriate to consider whether environmental impacts such as loss of biodiversity, deterioration in soil quality and animal welfare should be included in LCA. The "art" is in developing models of the real world that are simple and clear, yet as realistic as possible. In the words of Clayton and Radcliffe (1996), "the need is for an intelligent and sophisticated reductionism."

Development of LCA methodology as a type of systems analysis is therefore one focus of this thesis. Practically, it means that in assessing production of foodstuffs, relevant aspects for consideration include the role of crop rotations, location of production, and interactions between any one crop or livestock system and others in agricultural production. These types of issues are addressed as they arise throughout the thesis. In Chapter III, I discuss what types of issues should be included in LCA, and develop methods for inclusion of impacts on biodiversity, and soil quantity and quality in Chapters IV and V. Practical application of the methods is demonstrated in Chapter VI on a case study of breadmaking wheat production from intensive, integrated and organic farming systems.

3. Assessment of Agricultural Systems

Concerns about the environmental impacts of agricultural production have tended to focus on particular issues such as the toxicity of pesticides, excessive use of synthetic fertilisers, and human health risks from various livestock production systems. However, a number of other aspects of sustainability in food production have been overlooked by polarisation of the debate around these particular issues. Amongst others, these include the land areas required to feed a burgeoning global human population; the implications (environmental, social and economic) of globalising food trade; and the competing pressures on the limited resources of land, water and nutrients for alternative uses (for example, renewable sources of fuels and feedstocks for industrial use).

\(^1\) Spedding (1988, p.18) defines a system as "a group of interacting components, operating together for a common purpose, capable of reacting as a whole to external stimuli: it is unaffected directly by its own outputs and has a specified boundary based on the inclusion of all significant feedbacks."
A number of attempts have been made to provide a more integrated approach to assessing the sustainability of different agricultural systems. In the simplest form, the Food Miles concept has been put forward by the SAFE Alliance (Sustainable Agriculture, Food and Environment Alliance) (Paxton, 1994), and has received a considerable amount of attention in the UK. It puts forward the idea that the environmental and social impacts of food consumption patterns are linked to the distances travelled by different foodstuffs to their final destinations. However, due to the distances between points of production and consumption, consumers are often ignorant and/or unconcerned about the upstream social and environmental impacts associated with their consumption patterns because they are distanced from their immediate implications. The SAFE Alliance suggest that the Food Miles associated with our food consumption patterns should therefore be minimised by localising food production. In developing this argument, they draw on wider ideas about social justice and political systems in addition to environmental concerns, to support their thesis concerning localisation of food production systems.

A more sophisticated approach is the idea of Ecological Footprints. It was originally developed at the University of British Columbia by William Rees and others (Rees and Wackernagel, 1994), as a measure of the important role of land use in defining sustainability - or lack of it - in societies. In this approach, the amount of land required to produce the resources consumed by a region's population is calculated and termed the Ecological Footprint of that region. Land required for foodstuffs and forest products is calculated based on yields per hectare, and land for energy consumption is based on the land required to produce biomass energy equivalent to actual fossil fuel energy consumption. Using this approach, Rees and Wackernagel have estimated that the average Canadian requires 5.1 ha of productive land to meet his/her current consumption of food, forest products and energy. Since 1.6 ha of productive land is “available” on the Earth for each person, citizens in developed countries such as Canada are over-consuming and this represents a “sustainability gap” (Rees and Wackernagel, 1994).

Both these approaches have weaknesses linked with simplification of the environmental issues associated with food production. However, they also have strengths in putting across their messages using images that have resonance with the intended audience. In other words the messages are understandable and meaningful to stakeholders. In contrast, LCA is capable of giving a more comprehensive overview of the diverse environmental impacts associated with food production, and yet has some weaknesses in making its message understood and meaningful to stakeholders. The implications for development of LCA are discussed in Chapter VII.

Returning to the systems analysis approach utilised in LCA, there are clear links between using LCA to assess agricultural systems and the influential work of Professor Sir Colin Spedding (see, in particular, Spedding, 1988; also Spedding et al., 1981; and Spedding, 1989). Spedding applied
systems analysis to consider the efficiency of crop and livestock production. Aspects he investigated included land area requirements; both solar and fossil energy requirements; and the relative yields of dry matter, protein and energy in foodstuffs from different production systems given similar inputs. Interestingly, he suggested as long ago as 1979 that the efficient capture and use of solar radiation would be a major objective of future farming systems (Spedding, 1979).

He also provided a good example of the value of LCA, although in this case his analysis was restricted to energy use in the life cycle. The example concerns a comparison between energy supplied in the form of bread versus milk, and the data are shown in Table 1. Spedding makes the point that wheat is produced more efficiently per unit of support energy\(^2\) than milk when the analysis goes as far as the farm gate. However, if the two products are compared at the point of consumption, this difference between the two systems is reversed; in fact, the efficiency of use of support energy is almost identical for the two systems when post-agricultural processing for human consumption is included in the analysis. The example illustrates the need for life cycle thinking in comparing alternative foodstuffs.

Table 1. Efficiency of Energy Use In the Food Chain: The Example of Bread and Milk

<table>
<thead>
<tr>
<th></th>
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<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Wheat</td>
<td>To the farm gate</td>
<td>3.20</td>
</tr>
<tr>
<td></td>
<td>To loaf of white, sliced and wrapped bread</td>
<td>0.50</td>
</tr>
<tr>
<td>Milk</td>
<td>To the farm gate</td>
<td>0.65</td>
</tr>
<tr>
<td></td>
<td>To bottled milk delivered to the doorstep</td>
<td>0.60</td>
</tr>
</tbody>
</table>

Note: the wheat "to the farm gate" value calculated by Spedding above is somewhat lower than the value calculated in the study presented in Chapter VI (4.5 MJ in grain per 1 MJ support energy used). It is not possible to analyse the reasons for this difference because the data used for the calculation are not available.

There are therefore a number of studies that can usefully contribute to the debate on defining sustainable food systems. However, LCA has a potentially important role to play by integrating the strengths of the different approaches, and learning from their weaknesses. The research presented in this thesis has built upon these existing studies, developing LCA methodology to assess the comprehensive environmental impacts of agricultural systems (Chapters III to VI), and examining the role of LCA in decision-making processes (Chapter VII).

\(^2\) Support energy is defined as the additional energy (labour, fuel and electricity) used on the farm plus the "upstream" energy costs (manufacture of major inputs such as fertilisers, machinery, herbicides, etc., excluding human labour) and the "downstream" energy costs of processing and distribution (Spedding, 1988, p.163).
4. LCA and Decision-Making

Until relatively recently, the focus in LCA research has been on development of LCA methodology with the goal of providing "as complete a picture as possible of the interactions of an activity with the environment" (Consoli et al., 1993, p. 5). However, as discussed above in Section 2, the "art" in systems analysis is to develop models that are simple and clear yet as representative as possible of reality. Therefore, as LCA practitioners have continued to be frustrated by the lack of data and resource requirements of undertaking comprehensive LCAs, coupled with the perceived limited usefulness of LCA results among some users, attention has begun to shift towards analysing the role of LCA in decision-making. This implies a more reflexive approach in research on development of LCA methodology. It has involved examining aspects such as the evaluative frames implicit in current LCA methodology; the role of stakeholders in shaping LCA studies; and appropriate presentation formats for LCA results. Instead of focussing on collecting as much data as possible to construct a generic model of inputs and outputs associated with a system, the role of the procedural methodology in LCA is now increasingly recognised as a significant – if not dominant – component of LCA studies. This change of focus is examined in Chapter VII, along with its implications for assessment of agricultural systems.

5. Organisation of the Thesis

The preceding three sections have introduced the three themes that weave their way through the following pages. The order in which they are presented above reflects development of my research interests in this area, and this order is also reflected in the layout of the rest of the thesis. It begins with an introduction to LCA (Chapter II), showing how the methodology has been developed for using LCA as an environmental management tool. The need for further development of LCA methodology to assess agricultural systems is examined in Chapter III. This provides the justification for development of the methods for assessing biodiversity, and soil quantity and quality in Chapters IV and V. Practical application of these ideas is demonstrated in Chapter VI with a case study of breadmaking wheat production. In Chapter VII, I draw together some of the shortcomings identified in the previous chapters of using LCA as an environmental management tool, and take a more reflective look at the role of LCA in decision-making processes. In Chapter VIII, I conclude that life cycle thinking is an essential prerequisite to development of sustainable human activities, but that its operationalisation in the form of LCA requires a greater focus on responding to the requirements of different decision-making contexts. Application to the assessment of food systems is likely to be particularly far-reaching because these systems fulfill a fundamental need of the human species.
References


CHAPTER II
LIFE CYCLE ASSESSMENT

The world is vast and complex, and the human ability to process information is limited. All models of the world are reductionist, therefore, as information loss must be accepted in order to gain simplicity and clarity. The need is for an intelligent and sophisticated reductionism. (Clayton and Radcliffe, 1996, p.17)

1. Introduction

Life cycle assessment (LCA) dates back to the 1960s, and to energy analyses of industrial systems undertaken at that time and subsequently in response to the oil crises of the early 1970s. Although the original emphasis was upon consumption of energy resources, a number of studies also considered emissions (Fava et al., 1991). In the United States, the Midwest Research Institute (now Franklin Associates, Inc.) developed a methodology known as Resource and Environmental Profile Analysis (REPA), conducting its first analysis in 1969 on beverage containers for The Coca-Cola Company to "compare different containers to determine which produced the fewest effects on natural resources and the environment" (Hunt et al., 1992). However, interest in these studies declined in the late 1970s and it was not until the rise of environmental awareness in the late 1980s that attention was again focused on LCA as a potentially valuable environmental management tool.

A considerable amount of research has been undertaken in the last few years to develop LCA methodology. There have been four main foci for this work: the Society for Environmental Toxicology and Chemistry (SETAC), the Society for the Promotion of Life Cycle Development (SPOLD), various EU-funded projects, and the International Standards Organisation (ISO). Since the late 1980s, SETAC has been organising LCA conferences and workshops. Its booklet "Guidelines for Life-Cycle Assessment: A 'Code of Practice'" (Consoli et al., 1993) lays out general principles for conducting, reviewing, presenting, and using LCA. It has also produced a number of other guides to LCA methodology as shown in Table 1, and together they constitute an comprehensive framework for LCA practitioners. Between 1994 and 1997 five SETAC-Europe Working Groups held regular meetings to take forward the research agenda on LCA, and their reports were published towards the end of 1997. SPOLD is an industry-led organisation that aims to catalyse the development and application of LCA "by pooling the talent and resources of industry and other organisations interested
in LCA” (SPOLD, 1995). It has produced publications on data in LCA (Hemming, 1995), Impact Assessment (Grisel et al., 1994), and integration of social aspects into LCA (SPOLD, 1995).

A number of EU-funded projects have also played an important role in taking forward the research agenda for LCA. Most recently, the LCANET project has brought together LCA practitioners to define research priorities (Wrisberg et al., 1997), and a follow-up project called “CHAINET” has recently started to take forward work on integrating LCA with other environmental management tools. Meanwhile, the International Standards Organisation (ISO) has several Working Groups developing standards for LCA methodology.

Table 1. SETAC Publications On LCA

<table>
<thead>
<tr>
<th>Authors</th>
<th>Year of Publication</th>
<th>Title</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fava et al.</td>
<td>1991</td>
<td>A Technical Framework for Life-Cycle Assessments</td>
</tr>
<tr>
<td>SETAC-Europe</td>
<td>1992</td>
<td>Life-Cycle Assessment</td>
</tr>
<tr>
<td>Consoli et al.</td>
<td>1993</td>
<td>Guidelines for Life-Cycle Assessment: A ‘Code of Practice’</td>
</tr>
<tr>
<td>Fava et al.</td>
<td>1992</td>
<td>A Conceptual Framework for Life-Cycle Impact Assessment</td>
</tr>
<tr>
<td>Fava et al.</td>
<td>1994</td>
<td>Life-Cycle Assessment Data Quality: A Conceptual Framework</td>
</tr>
<tr>
<td>Huppes and Schneider</td>
<td>1994</td>
<td>Proceedings of the European Workshop on Allocation in LCA</td>
</tr>
<tr>
<td>Udo de Haes et al.</td>
<td>1994</td>
<td>Integrating Impact Assessment Into LCA</td>
</tr>
<tr>
<td>Udo de Haes</td>
<td>1996</td>
<td>Towards a Methodology for Life-Cycle Impact Assessment</td>
</tr>
<tr>
<td>Allen et al.</td>
<td>1997</td>
<td>Public Policy Applications of Life-Cycle Assessment</td>
</tr>
<tr>
<td>Christiansen</td>
<td>1997</td>
<td>Simplifying LCA: Just a Cut?</td>
</tr>
<tr>
<td>Clift et al.</td>
<td>1998</td>
<td>Towards a Coherent Approach to Life-Cycle Inventory Analysis</td>
</tr>
</tbody>
</table>

LCA is concerned with environmental impacts in the areas of ecological health, human health and resource depletion. The SETAC “Code of Practice” defines three prime objectives of undertaking an LCA (Consoli et al., 1993, p.5):

1. To provide as complete a picture as possible of the interactions of an activity with the environment.
2. To contribute to the understanding of the overall and interdependent nature of the environmental consequences of human activities.

1 The Working Groups focused on: Inventory Analysis, Screening and Streamlining, Impact Assessment, Case Studies, and Conceptually Related Programmes.
2 Working Group 1 is producing ISO 14040, the basic document for standardising LCA. Working Groups 2 and 3 focus on Goal Definition and Scoping, and the Inventory analysis (for ISO 14041). Working Group 4 is concerned with impact assessment (ISO 14042), and Working Group 5 is addressing “improvements and innovations.” ISO 14040 and ISO 14041 were published in 1997, and ISOs 14042 and 14043 are scheduled for production in 1998 (Marsmann et al., 1997).
3. To provide decision-makers with information which defines the environmental effects of these activities and identifies opportunities for environmental improvements.

It has been formally defined as:

a process to evaluate the environmental burdens associated with a product, process, or activity by identifying and quantifying energy and materials used and wastes released to the environment; to assess the impact of those energy and material uses and releases to the environment; and to identify and evaluate opportunities to effect environmental improvements. The assessment includes the entire life cycle of the product, process, or activity, encompassing extracting and processing raw materials; manufacturing, transportation and distribution; use, re-use, maintenance; recycling, and final disposal (Fava et al., 1991, p.1).

Four different phases of LCA can be distinguished:

- Goal Definition and Scoping
- Inventory Analysis
- Impact Assessment
- Improvement Assessment (or “Interpretation”).

These can be represented as shown in Figure 1. The diagram shows that, in practice, LCA involves a series of iterations as its scope is redefined on the basis of insights gained throughout the study. Sections 2 to 5 below describe the methodology for each phase, and Section 6 summarises the current status of LCA as an environmental management approach.

Figure 1. Phases of Life Cycle Assessment
2. Goal Definition and Scoping

This first LCA phase involves defining the purpose of the study, its scope, data quality goals, and functional unit. The **purpose** and **scope** of the study are shaped by its sponsor, and it is important that they are clearly defined to avoid any subsequent misunderstandings about the wider applicability of the results. For example, a study carried out by a company to compare two alternative production processes may be adequate for internal decision-making, but its results may not be appropriate for public policy-making if the data are not representative of the national situation. Having defined the purpose of the study, scoping involves defining boundaries for the study that are appropriate for its purpose. These boundaries are shaped by the desired geographical applicability of the results, time horizons over which the analysis is relevant, and the focus of the study or the comparisons to be made which may lead to omission of particular sub-systems or stages of the life cycle. I return to the issue of scoping and setting system boundaries in Chapter III.

**Data quality** tends to be a problem in LCAs, and has been the subject of a number of recent reports (for example, Fava *et al.*, 1994, and Hemming, 1995). Various “tagging” systems have been suggested for signalling data quality, and database formats put forward to standardise data collection and facilitate compilation of common datasets. However, often a complete absence of data is more of a problem than its quality. For all these situations, sensitivity analysis can be used to determine the impact of data deficiencies and/or omissions on the final LCA results.

An appropriate definition of the **functional unit** is fundamental to the credibility of the LCA, and may not be obvious at first glance: in fact, it relates to the **service(s)** provided by any product, process, or activity under analysis. Hence, an appropriate functional unit for an LCA study of disposable and cloth nappies for babies is “the quantity required to keep a baby in nappies for six months” rather than a certain number of nappies (due to the different rates of use for these two products). For paints, the functional unit may be “the quantity required to cover 10 m² of surface for a defined period of time,” and for beverage containers it may be “the quantity used to deliver 1000 litres of beverage,” rather than the same weight of different types of paint or packaging. Furthermore, for some products definition of the functional unit depends on the behaviour of the user or consumer, because alternative products which are packaged or dispersed in different ways may be used quite differently (Clift, 1994).
3. Inventory Analysis

At the Inventory phase the environmental burdens (or "interventions" according to ISO 14040 terminology) associated with the life cycle for the functional unit are quantified. These are the material and energy inputs, and product, waste, and emission outputs to air, water, and land. The methodology involves using a systems approach, drawing a boundary around the system under analysis and quantifying the inputs and outputs across the system boundary, as shown in Figure 2. Within the system, a number of discrete sub-systems are identified, and the relationships between inputs and outputs are modelled for each sub-system (including transportation). A generic life cycle for different products and their related services is shown in Figure 3.

Figure 2. Inputs and Outputs Across the System Boundary In LCA

![Figure 2](image)

Figure 3. Generic Flow Diagram for LCA Inventory Analysis

![Figure 3](image)

Source: Hodgson et al., 1997.
Sometimes in an analysis it is useful to draw a further boundary as shown in Figure 4. This makes a distinction between the Foreground System which is “the set of processes whose selection or mode of operation is affected directly by decisions based on the study,” and the Background System. The Background System comprises “all other processes which interact directly with the Foreground System, usually by supplying material or energy to the Foreground or receiving material or energy from it” (Clift et al., 1998). The usefulness of this type of distinction in an analysis is shown in Chapter VI for assessing use of manure in a wheat production system.

**Figure 4. The Foreground and Background Systems In LCA**

Although the methodology for this phase seems relatively straightforward, it is - in fact - complicated by two issues. The first relates to defining boundaries for the system under analysis, technically an issue that is resolved in Goal Definition and Scoping. However, one of the guidelines for inclusion of sub-systems in a study is that those contributing less than a certain percentage of the inputs or the final product can be excluded from the study (Fava et al., 1991, p.34; Vigon, 1992, p.55). Hence, some iteration is required to determine the sub-systems included in the study (as shown in Figure 1). A good example is capital equipment. In many LCAs, production and maintenance of capital equipment (such as industrial production line machinery) are omitted from the study. However, in some cases machinery may turn out to be significant in the LCA results because it has a relatively low use rate per functional unit (for example, agricultural machinery as shown in Chapter VI).

Allocation is the second issue. It is described as “partitioning the input or output flows of a process to the product system under study” (ISO 14040 draft). Consoli et al. (1993) identify three types of systems where inputs and outputs must be partitioned between different product systems:
• **Co-production**
In these systems, one process yields two or more useful products (Figure 5a). Allocation refers to the problem of partitioning the inputs and outputs between these products. Examples include production of chlorine and caustic soda from the chlorine manufacturing process, and wheat grain and straw from wheat production.

• **Waste treatment**
In this case, the function delivered is management of waste. Many waste treatment systems handle more than one type of waste stream (Figure 5b). In these cases, allocation refers to the problem of partitioning the inputs and outputs between the different waste streams.

• **Recycling**
In recycling, an output from a system becomes an input to a system or process. The recycle may go back into the same system from which it was produced: this is called **closed-loop** recycling. Alternatively, it may become an input into a different system or process: this is called **open-loop** recycling (Figure 5c). Closed-loop recycling does not present an allocation problem because the system can be modelled to take account of this type of recycling. However, open-loop recycling does present an allocation problem because the recycle becomes an input to another system producing a different product or products.

A hierarchy of preferred approaches to allocation has been developed by the Groupe des Sages (Clift et al., 1996), the SETAC-Europe Working Group on Inventory Analysis (Clift et al., 1998), the EU project on LCA and agriculture (Audsley et al., 1997), and is embodied in ISO 14041. In brief, this hierarchy is:

1. **Avoiding allocation by system extension**
The system is expanded to account for the fate of products ("co-products") other than the one of interest in the study (the "primary product"). The environmental burdens associated with the primary product are the remaining burdens after accounting for the fate of the other co-products, and subtracting any burdens displaced by use of these co-products in the economy (called the "Avoided Burdens")\(^3\).

2. **Allocation on the basis of physical causality (marginal allocation)**
The system inputs and outputs are partitioned between the co-products in a way that reflects the underlying physical relationships between these co-products. Where the options being compared represent marginal changes, the physical relationships are analysed by marginally - and

---
\(^3\) This approach is discussed in more detail in Tillman et al., 1994.
independently - varying the relative outputs of the co-products from the system, and allocating environmental burdens to each co-product in relation to these marginal changes.

3. **Allocation on the basis of composition**

   This approach depends upon determining a common property of the co-products that is representative of their functions; for example, their heat value or protein content. The burdens are then allocated in proportion to the relative values of that common property in the different co-products.

4. **Allocation on the basis of economic value**

   The burdens are allocated among the co-products according to their relative economic values at the point of division in the system. This represents a measure of the incentive for production.

Application of the hierarchy is discussed in more detail in the study of wheat production in Chapter VI.
Once all the data have been collected, and allocation issues resolved, they are normalised to the functional unit and compiled into an Inventory Table prior to Impact Assessment.

4. Impact Assessment

The environmental burdens calculated in the analysis are "translated" into environmental impacts during the Impact Assessment phase of LCA. The objective of this phase is to present the environmental impacts of the system under analysis in a form that is useful for the purpose of the study and that can be understood by users of the study results. It consists of a number of stages:

- **Classification**: each burden is linked to one or more environmental impact categories. Impact categories may vary from global warming and ozone depletion, to land use and physical ecosystem degradation. For example, CO₂ emissions may be linked with global warming and CH₄ emissions with global warming and photochemical oxidant formation.

- **Characterisation**: the contribution of each burden to any one environmental impact category is assessed by multiplying each burden by a relevant weighting factor (here called an Impact Assessment factor (IA factor)). The results within each impact category are then added to give scores for the different categories.

- **Normalisation**: each score is normalised in order to obtain an estimate of the relative significance of the result in the different impact categories. This is usually done by dividing the score in each impact category by the total score for each category in a given geographical area. For example, the Global Warming Potential (GWP) calculated for a system could be divided by the total GWP of gases released in the UK each year to give a normalised score. The normalised scores for different impact categories are then compared in order to gain an impression of the relative contribution made by the system to each impact category within a given geographical area.

- **Valuation**: the normalised result for each impact category is multiplied by a weighting factor representing the relative importance of the different impact categories. For example, global warming may be considered twice as important as photochemical oxidant formation, and so it is weighted in the ratio 2:1. The weighted results for each impact category are then added to give one final value for the environmental impact of the system under analysis.

At the present time, there is no agreement on one standard format for Impact Assessment (see, for example the discussion in Owens, 1996). Indeed, some people suggest that it should consist of just classification and characterisation, and most research effort has been focused on these two stages.
The most popular methods developed for these two stages are the Problem-Oriented and Critical Volumes\textsuperscript{4} approaches. Other approaches include the Eco-Indicator, EPS system and Eco-Points method.

Sections 4.1 and 4.2 below describe the Critical Volumes and Problem-Oriented approaches for classification and characterisation, and Section 4.3 introduces normalisation and valuation. In Section 4.4, I discuss an important and fundamental issue in Impact Assessment: assessment of actual versus potential impacts in LCA. In Section 4.5, I briefly introduce other methods of Impact Assessment.

4.1 Classification and Characterisation: The Critical Volumes Approach

The Critical Volumes approach was developed in Switzerland (Bundesamt für Umweltschutz, 1984; Habersatter and Widmer, 1991). The environmental burdens are classified into five categories:

- Energy consumption (MJ)
- Solid waste generation (kg)
- Emissions to air
- Emissions to water
- Emissions to land.

The first two categories, energy consumption and solid waste generation, are calculated by simple addition of relevant burdens. For the last three categories, each environmental burden is divided by a corresponding air/water/soil quality standard prior to addition of the burdens in each category. In other words, for these three categories the IA factors are the inverse of the quality standard for each substance. The quality standards may be political (for example, occupational exposure limits for air and EU drinking water standards) or based on toxicity experiments (for example, No Observable Effect Concentrations (NOECs) or LC\textsubscript{50} values). Adding the results gives, in effect, the volumes of air, water and soil required to dilute the emissions to an acceptable concentration.

Tables 2 and 3 give the IA factors calculated by Habersatter (1991) and Heijungs \textit{et al.} (1991) for air and water emissions using this approach (listed under "General Toxicity").

\textsuperscript{4} The Critical Volumes approach is also called the "weighted loads" approach (Grisel et al., 1994).
Table 2. Toxicity Factor for Air Emissions

<table>
<thead>
<tr>
<th>Source</th>
<th>Metals</th>
<th>Hydrocarbons</th>
<th>Benzene</th>
<th>HC excl. CH₄</th>
<th>VOCs</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Pb</td>
<td>Aldehydes</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ecotoxicity</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Guinée et al., 1996</td>
<td>Aquatic</td>
<td>1.2</td>
<td>11</td>
<td>0.0013</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Terrestrial</td>
<td>11000</td>
<td>450000</td>
<td>0.063</td>
<td>-</td>
</tr>
<tr>
<td>Human Toxicity</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Heijungs et al., 1992</td>
<td></td>
<td>160</td>
<td>120</td>
<td>3.9</td>
<td>-</td>
</tr>
<tr>
<td>Guinée et al., 1996</td>
<td></td>
<td>67000</td>
<td>4900</td>
<td>29</td>
<td>-</td>
</tr>
<tr>
<td>Audsley et al., 1997</td>
<td></td>
<td>8.9</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>General Toxicity</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Habersatter, 1991</td>
<td>(x10⁶)</td>
<td>1000</td>
<td>33.22</td>
<td>0.067</td>
<td>0.067</td>
</tr>
<tr>
<td>Heijungs et al., 1991</td>
<td>(x10⁶)</td>
<td>6.667</td>
<td>0.667</td>
<td>0.002</td>
<td>0.002</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Source</th>
<th>Others</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Cl₂</td>
</tr>
<tr>
<td>Ecotoxicity</td>
<td></td>
</tr>
<tr>
<td>Guinée et al., 1996</td>
<td>Aquatic</td>
</tr>
<tr>
<td></td>
<td>Terrestrial</td>
</tr>
<tr>
<td>Human Toxicity</td>
<td></td>
</tr>
<tr>
<td>Heijungs et al., 1992</td>
<td></td>
</tr>
<tr>
<td>Guinée et al., 1996</td>
<td></td>
</tr>
<tr>
<td>Audsley et al., 1997</td>
<td></td>
</tr>
<tr>
<td>General Toxicity</td>
<td></td>
</tr>
<tr>
<td>Habersatter, 1991</td>
<td>(x10³)</td>
</tr>
<tr>
<td>Heijungs et al., 1991</td>
<td>(x10³)</td>
</tr>
</tbody>
</table>

Table 3. Toxicity Factors for Water Emissions

<table>
<thead>
<tr>
<th>Source</th>
<th>Metals</th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>As</td>
<td>Cd</td>
<td>Cr</td>
<td>Cu</td>
<td>Fe</td>
<td>Hg</td>
<td>Ni</td>
<td>Pb</td>
</tr>
<tr>
<td>Ecotoxicity</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Heijungs et al., 1992</td>
<td>Aquatic</td>
<td>0.2</td>
<td>200</td>
<td>1</td>
<td>2.0</td>
<td>-</td>
<td>500</td>
<td>0.33</td>
</tr>
<tr>
<td></td>
<td>Terrestrial</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Guinée et al., 1996</td>
<td>Aquatic</td>
<td>190</td>
<td>4500</td>
<td>84</td>
<td>96</td>
<td>-</td>
<td>130000</td>
<td>2700</td>
</tr>
<tr>
<td></td>
<td>Terrestrial</td>
<td>0.0000097</td>
<td>0.025</td>
<td>0.00011</td>
<td>0.0001</td>
<td>-</td>
<td>8200000</td>
<td>0.000031</td>
</tr>
<tr>
<td>Audsley et al., 1997</td>
<td>Aquatic</td>
<td>-</td>
<td>520</td>
<td>2.6</td>
<td>-</td>
<td>-</td>
<td>1300</td>
<td>0.79</td>
</tr>
<tr>
<td></td>
<td>Terrestrial</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Human Toxicity</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Heijungs et al., 1992</td>
<td>1.4</td>
<td>2.9</td>
<td>0.57</td>
<td>0.02</td>
<td>0.00465</td>
<td>4.7</td>
<td>0.057</td>
<td>0.79</td>
</tr>
<tr>
<td>Guinée et al., 1996</td>
<td>51</td>
<td>130</td>
<td>9.3</td>
<td>1.1</td>
<td>-</td>
<td>18000</td>
<td>63</td>
<td>260</td>
</tr>
<tr>
<td>Audsley et al., 1997</td>
<td>3.1</td>
<td>0.62</td>
<td>0.022</td>
<td>-</td>
<td>7.8</td>
<td>0.062</td>
<td>0.86</td>
<td></td>
</tr>
<tr>
<td>General Toxicity</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Habersatter, 1991 (x10³)</td>
<td>-</td>
<td>10</td>
<td>-</td>
<td>2</td>
<td>0.5</td>
<td>100</td>
<td>-</td>
<td>2</td>
</tr>
<tr>
<td>Heijungs et al., 1991 (x10³)</td>
<td>50</td>
<td>666.666</td>
<td>20</td>
<td>20</td>
<td>2</td>
<td>3333.333</td>
<td>-</td>
<td>33.3333</td>
</tr>
</tbody>
</table>
Table 3. Toxicity Factors for Water Emissions (continued)

<table>
<thead>
<tr>
<th>Source</th>
<th>Others</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Ammonia</td>
</tr>
<tr>
<td><strong>Ecotoxicity</strong></td>
<td></td>
</tr>
<tr>
<td>Heijungs et al., 1992</td>
<td>Aquatic</td>
</tr>
<tr>
<td></td>
<td>Terrestrial</td>
</tr>
<tr>
<td>Guinée et al., 1996</td>
<td>Aquatic</td>
</tr>
<tr>
<td></td>
<td>Terrestrial</td>
</tr>
<tr>
<td>Audsley et al., 1997</td>
<td>Aquatic</td>
</tr>
<tr>
<td></td>
<td>Terrestrial</td>
</tr>
<tr>
<td><strong>Human Toxicity</strong></td>
<td></td>
</tr>
<tr>
<td>Heijungs et al., 1992</td>
<td></td>
</tr>
<tr>
<td>Guinée et al., 1996</td>
<td></td>
</tr>
<tr>
<td>Audsley et al., 1997</td>
<td></td>
</tr>
<tr>
<td><strong>General Toxicity</strong></td>
<td></td>
</tr>
<tr>
<td>Habersatter, 1991</td>
<td></td>
</tr>
<tr>
<td>Heijungs et al., 1991</td>
<td></td>
</tr>
</tbody>
</table>
4.2 Classification and Characterisation: The Problem-Oriented Approach

The Problem-Oriented approach has been pioneered by the University of Leiden (Heijungs et al., 1992a, 1992b). Using this method, a number of specific environmental impacts are identified, and environmental burdens are classified and characterised according to these impacts. Possible categories are listed in Table 4, although often only a subset of these categories is assessed in any LCA study.

Calculation of the impact in each category is considered in more detail below. For each category, IA factors are listed for substances assessed in the LCA presented in Chapter VI.

Table 4. Impact Assessment Categories Considered In the Problem-Oriented Approach

<table>
<thead>
<tr>
<th>Resource Depletion</th>
<th>Pollution</th>
<th>Disturbances</th>
</tr>
</thead>
<tbody>
<tr>
<td>Abiotic resources*</td>
<td>Global warming*</td>
<td>Physical ecosystem degradation</td>
</tr>
<tr>
<td>Biotic resources</td>
<td>Ozone depletion*</td>
<td>Landscape degradation</td>
</tr>
<tr>
<td>Land use</td>
<td>Ecotoxicity*</td>
<td>Desiccation</td>
</tr>
<tr>
<td>Water use</td>
<td>Human toxicity*</td>
<td>Direct victims</td>
</tr>
<tr>
<td></td>
<td>Photochemical oxidant formation*</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Acidification*</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Eutrophication*</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Radiation</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Dispersion of heat</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Noise</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Smell</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Working conditions</td>
<td></td>
</tr>
</tbody>
</table>

Source: adapted from Guinée et al., 1993.
* Categories most commonly included in LCAs.

4.2.1 Resource Depletion

A variety of different approaches and terminology have been used in assessment of resource depletion. Below I outline the different approaches that have been used in LCA, and then present an alternative perspective that integrates assessment of abiotic/biotic and non-renewable/renewable resources.

Abiotic Resource Depletion

Abiotic resources include energy and materials. Conventionally their depletion has been assessed as some ratio of use in relation to total reserves and, sometimes, extraction rates (Finnveden, 1994; Guinée et al., 1992; Heijungs et al., 1997).
The simplest method involves assessing use in relation to total reserves; this is the approach put forward by Heijungs et al. (1992a). The Abiotic Resource Depletion (ARD) factor for any substance X is therefore simply the inverse of the total reserves:

\[
\text{ARD factor (1)} = \frac{1}{R}
\]

where \( R \) = total reserves of substance X (tonnes)

Here, the "total reserves" may be assessed as either (Guinée, 1996; BP, 1994):

- Economic reserves: the quantity of substance X that is currently economically attractive to extract.
- Reserve base: the quantity of substance X meeting specified minimum physical and chemical criteria related to current mining practice. It includes that fraction of the total resource that is reasonably likely to become economically available within planning horizons beyond those that assume proven technology and current economics.
- Proven reserves: the quantity of substance X that "geological and engineering information indicates with reasonable certainty can be recovered in the future from known reservoirs under existing economic and operating conditions" (definition from BP, 1994).
- Ultimately extractable reserves: the total quantity of substance X in the Earth that can be technically extracted.
- Ultimate reserves (also known as geological reserves or resource base): the total quantity of substance X in the Earth's crust, oceans and atmosphere.

A more sophisticated approach involves assessment in relation to total reserves and extraction rates. The ARD factor is defined as:

\[
\text{ARD factor (2)} = \frac{P}{R}
\]

where \( P \) = annual production (extraction) of substance X (tonnes/year)
\( R \) = total reserves of substance X (tonnes).

In effect, this factor is the reciprocal of the number of years that the total reserves will last at current extraction rates, also called the "reserves-production ratio". This value is called the static reserve life by Lindfors et al. (1995, p.166).
However, the problem with this approach is that these ARD factors do not necessarily distinguish between resources that are relatively more or less scarce. This is demonstrated by Guinée (1996, p.97) who gives the example of choosing between using 1 kg of resource A or 1 kg of resource B, both of which have a reserves-production ratio of 20 years (i.e. the ARD factor for both resources is 0.05). On this basis, both resources seem equally attractive from a resource depletion perspective. However, it may be the case that resource A is both more abundant and more extensively exploited than B, for example:

- For resource A, \( R = 10^9 \) kg and \( P = 5 \times 10^7 \) kg/year
- For resource B, \( R = 100 \) kg and \( P = 5 \) kg/year.

These data suggest that resource A should be preferred over resource B because extraction of 1 kg represents a slight increment to current usage of A but a significant proportional increase in the use of resource B.

Guinée (1996) suggests that this problem can be overcome by squaring the reserve \( R \), and gives the following ARD factor:

\[
\text{ARD factor (3)} = \frac{P}{R^2}
\]

where \( P \) = annual production (extraction) of substance X
\( R \) = total reserves of substance X.

He then goes on to develop ARD factors using this approach but adding a reference substance (antimony\(^5\)) into the equation (in order to give dimensionless factors of order unity), and using the ultimate reserves as the measure of total reserves:

\[
\text{ARD factor (4)} = \left( \frac{P_X}{P_{sb}} \right) \left( \frac{R_{sb}}{R_X} \right)^2
\]

where \( P_X \) = annual production (extraction) of substance X
\( R_X \) = ultimate reserves of substance X
\( P_{sb} \) = annual production (extraction) of antimony
\( R_{sb} \) = ultimate reserves of antimony.

\(^5\) Antimony is not used for any special reason apart from that it is known as a scarce resource (Guinée, pers.comm.)
Examples of substances generally considered in this Impact Assessment category are listed in Table 5 together with relevant data. Some interesting comparisons can be made between the results using the different equations given above. For example, Table 6 shows the results obtained for ARD factors (1), (2) and (4) when assessing resource depletion for a system using one gramme each of copper, lead, tin and zinc. The results for each approach are normalised relative to tin in order to facilitate comparison. It can be seen that the relative results for each substance vary considerably between the different approaches, and indicate the importance of the chosen assessment method in determining the results of a study.

**Biotic Resource Depletion**

Biotic resources are living organisms. Relatively little attention has been given to assessment of this type of resource depletion in LCA. Heijungs et al. (1992b, p.69) suggest that use of a threatened species should be assessed in relation to its annual rate of exploitation ("annual use") and total population ("recoverable reserves"):  

\[
\frac{\text{Total annual use of Species Z}}{\text{Recoverable reserves of Species Z}} \times \frac{\text{Use of Species Z in System}}{\text{Recoverable reserves of Species Z}}
\]

A Biotic Depletion Factor (BDF) is therefore calculated for each species as:

\[
\text{BDF} = \left[ \frac{\text{Total annual use of Species Z}}{\left(\text{Recoverable reserves of Species Z}\right)^2} \right]
\]

However, BDFs for only five species are given by Heijungs et al. (1992a), and the method has not been developed elsewhere. Instead, more recently research attention has focused on assessing biotic resource depletion within land use categories (see below). I return to this issue in Chapter IV.
Table 5. Data To Calculate Factors for Assessing Abiotic Resource Depletion

<table>
<thead>
<tr>
<th>Substance</th>
<th>Reserves-Production Ratio</th>
<th>Mass of Reserves</th>
<th>ADP Using Method of</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>(Heijungs et al., 1992a, p.65) b</td>
<td>Guinée (1992) c</td>
</tr>
<tr>
<td>Oil</td>
<td>40</td>
<td>123,559 x 10^4 tonnes</td>
<td>4.36 x 10^-1</td>
</tr>
<tr>
<td>Natural gas</td>
<td>60</td>
<td>109,326 x 10^9 m^3</td>
<td>3.20 x 10^-1</td>
</tr>
<tr>
<td>Hard coal</td>
<td>390</td>
<td>-</td>
<td>6.00 x 10^-2</td>
</tr>
<tr>
<td>Soft coal</td>
<td>390</td>
<td>-</td>
<td>8.51 x 10^-3</td>
</tr>
<tr>
<td>Uranium</td>
<td>58</td>
<td>1,676,820 ton</td>
<td>2.87 x 10^-3</td>
</tr>
<tr>
<td>Bauxite</td>
<td>220</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Arsenic</td>
<td>21</td>
<td>-</td>
<td>9.17 x 10^-3</td>
</tr>
<tr>
<td>Cadmium</td>
<td>27</td>
<td>0.535 x 10^4 tonnes</td>
<td>3.30 x 10^-1</td>
</tr>
<tr>
<td>Chlorine</td>
<td>-</td>
<td>-</td>
<td>4.86 x 10^-4</td>
</tr>
<tr>
<td>Chromium</td>
<td>105</td>
<td>-</td>
<td>8.58 x 10^-4</td>
</tr>
<tr>
<td>Cobalt (land only)</td>
<td>90</td>
<td>-</td>
<td>2.62 x 10^-5</td>
</tr>
<tr>
<td>Copper (land only)</td>
<td>36</td>
<td>350 x 10^6 tonnes</td>
<td>1.94 x 10^-3</td>
</tr>
<tr>
<td>Fluorine</td>
<td>52 (fluorspar)</td>
<td>-</td>
<td>2.96 x 10^-6</td>
</tr>
<tr>
<td>Iron ore</td>
<td>119</td>
<td>-</td>
<td>8.45 x 10^-4</td>
</tr>
<tr>
<td>Lead</td>
<td>20</td>
<td>75 x 10^6 tonnes</td>
<td>1.35 x 10^-2</td>
</tr>
<tr>
<td>Manganese</td>
<td>(land only) 95</td>
<td>-</td>
<td>1.38 x 10^-4</td>
</tr>
<tr>
<td>Mercury</td>
<td>25</td>
<td>0.0057 x 10^4 tonnes</td>
<td>4.95 x 10^-1</td>
</tr>
<tr>
<td>Molybdenum</td>
<td>50</td>
<td>-</td>
<td>3.17 x 10^-2</td>
</tr>
<tr>
<td>Nickel (land only)</td>
<td>55</td>
<td>54 x 10^4 tonnes</td>
<td>1.08 x 10^-4</td>
</tr>
<tr>
<td>Phosphate</td>
<td>very large</td>
<td>-</td>
<td>8.44 x 10^-5</td>
</tr>
<tr>
<td>Potash</td>
<td>300</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Selenium</td>
<td>41</td>
<td>-</td>
<td>4.75 x 10^-1</td>
</tr>
<tr>
<td>Sulphur</td>
<td>24</td>
<td>-</td>
<td>3.58 x 10^-4</td>
</tr>
<tr>
<td>Thorium</td>
<td>-</td>
<td>-</td>
<td>2.08 x 10^-7</td>
</tr>
<tr>
<td>Tin</td>
<td>28</td>
<td>4.33 x 10^6 tonnes</td>
<td>3.30 x 10^-2</td>
</tr>
<tr>
<td>Titanium</td>
<td>70</td>
<td>-</td>
<td>4.40 x 10^-4</td>
</tr>
<tr>
<td>Tungsten</td>
<td>55</td>
<td>-</td>
<td>1.17 x 10^-4</td>
</tr>
<tr>
<td>Vanadium</td>
<td>135</td>
<td>-</td>
<td>1.16 x 10^-4</td>
</tr>
<tr>
<td>Zinc</td>
<td>21</td>
<td>147 x 10^6 tonnes</td>
<td>9.92 x 10^-4</td>
</tr>
</tbody>
</table>

aData in Lindfors et al. (1995, p.166) derived from Crowson (1992), and World Resources Institute (1992). The ARD factor is the inverse of this value (see text).

b These are the quantities of recoverable reserves. The ARD factor is the inverse of this value (see text).

c See text for calculation of these values.

Table 6. Relative Abiotic Resource Depletion Results for Four Substances Using Different Impact Assessment Methods

<table>
<thead>
<tr>
<th>Substance</th>
<th>ARD Factor (1) (Heijungs et al., 1992a)</th>
<th>ARD Factor (2) (Lindfors et al., 1995)</th>
<th>ARD Factor (4) (Guinée, 1996)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Copper</td>
<td>0.01</td>
<td>0.8</td>
<td>0.06</td>
</tr>
<tr>
<td>Lead</td>
<td>0.06</td>
<td>1.4</td>
<td>0.41</td>
</tr>
<tr>
<td>Tin</td>
<td>1.00</td>
<td>1.0</td>
<td>1.00</td>
</tr>
<tr>
<td>Zinc</td>
<td>0.03</td>
<td>1.3</td>
<td>0.03</td>
</tr>
</tbody>
</table>
Land Use

Use of land may be considered under “Resource Depletion” and/or “Disturbances” (see Section 4.2.3). As a resource issue, relevant aspects for consideration are:

- Occupation of land area
- Maintenance and/or changes in soil quantity and quality
- Maintenance and/or changes in ecosystems and biodiversity.

Occupation of land area is a resource issue because, although the total area of land available does not change (apart from in the worst global warming scenarios!), its use by a system under analysis means that it is not available for alternative uses. Since land is a limited resource, this should be recognised in LCA. This can be done easily by assessing land use in “m²-year” units in an analysis.

Maintenance and/or changes in soil quantity and quality had not been considered in LCA until the recent EU project on LCA for agriculture (Audsley et al., 1997); this aspect is considered in more detail in Chapter V. A few approaches for assessing ecosystems and biodiversity have been suggested that depend upon defining a limited number of land categories and assessing their “nature value” relative to a reference ecosystem (for example, Blonk et al., 1997). Land use is then measured within (and between) these categories (see, for example, Steen and Ryding, 1992; and Heijungs et al., 1992b, 1997). This aspect is discussed in more detail in Chapter IV.

Water Use

Use of water may be considered under “Resource Depletion” and/or “Disturbances (Section 4.2.3), similarly to land use. As a resource issue, relevant aspects for consideration are:

- Total use of water
- Use of surface water versus groundwater
- Temporary versus permanent removal of water from sources
- Total use of water in relation to water supplies in particular regions.

Lindfors et al. (1995, p.94) recommend that, if possible, LCA results for water use should account for all these aspects. In other words, water use should be presented in sub-categories of surface- and ground-water, temporary and permanent removal of water, and regional use of water. Heijungs et al.
(1992b, p.80-1), on the other hand, suggest that total water use should be excluded from an analysis because, in their opinion, water is not a scarce resource at global level. Instead, information on regional and local use of water should be included because its “desiccation” impacts can then be assessed in relation to regional and local conditions. This amounts to an argument in favour of assessing water use under “Disturbances” rather than “Resource Depletion” (see Section 4.2.3). I discuss this issue in more detail in Chapter III, Section 5.2.

An Integrated Approach To Resource Depletion

In comparing alternative methods, it is informative to first consider the difference between non-renewable and renewable resources. Here, I define non-renewable resources as elements in the Periodic Table; a finite mass of each element exists in the Earth’s crust, oceans and atmosphere. [This excludes nuclear reactions that transform one element into another, but these reactions would make only a negligible difference to the total mass of any element.] In contrast, renewable resources are those formed from more than one element; examples include water, fossil fuels, soil and trees. These are all resources that can be reformed (“renewed”) over a period of time by recombining elements. The time period varies according to the resource; fossil fuels take tens of millions of years to form, soils take hundreds of years, and trees take less than a hundred years.

In considering depletion of resources, it therefore follows that non-renewable resources, i.e. elements, cannot be depleted because they are always present on the Earth, i.e. the Law of Conservation of Mass holds true! Instead, I suggest that concerns about “depletion” of elements are centred on:

- **The total quantities available** (because of competing demands for alternative uses of any one element in the economy).
- **The location of resources** (because, for example, phosphorus in rock deposits in the USA has to be transported before use as agricultural fertiliser in the UK).
- **The concentration of resources** (because, for example, phosphorus at low concentrations in the Earth’s crust is less available for alternative uses than phosphorus concentrated in a bag of phosphate fertiliser).

It may therefore be more appropriate to refer to elements as “limited resources” where the extent to which an element is limited depends upon a combination of the above factors. Elsewhere, we have

---

6 The Law of Conservation of Mass states that matter can neither be created nor destroyed in a chemical reaction.
suggested that the dispersion of elements could be used to assess depletion (Cowell and Clift, 1997). This links with the idea of depletion being measured as a function of loss of exergy, although it has yet to be operationalised at a general level (Clift, 1993, 1995; Finnvveden, 1994; Finnvveden and Östlund, 1996). It also involves moving from a focus on “once-through” use to cyclical or metabolised use, equivalent to the difference between the First and Second Laws of Thermodynamics⁷.

For renewable resources, it could be argued that a definition of renewability should be linked with the time required for renewing each resource⁸ in addition to the factors listed above for non-renewable resources. I demonstrate application of this type of approach in Chapter III, Section 5.2, for assessment of water use.

Land use does not fit in the non-renewable or renewable resource category. It should therefore be assessed separately as discussed above, and in Chapters IV and V.

4.2.2 Pollution

Global Warming

IA factors used for assessing global warming are the Global Warming Potentials (GWPs) from the International Panel on Climate Change (Houghton et al., 1996). These describe the radiative forcing of different global warming gases relative to that of CO₂, taking into account the absorption properties of the gases and their lifetimes. GWPs are available for 20, 100 and 500 year timespans, and it is generally recommended that results using all three timespans are presented in an LCA. Table 7 lists the GWPs for relevant air emissions.

<table>
<thead>
<tr>
<th>Source</th>
<th>Global Warming Potential</th>
<th>CO₂</th>
<th>CH₄</th>
<th>N₂O</th>
</tr>
</thead>
<tbody>
<tr>
<td>Houghton et al., 1996</td>
<td>GWP20</td>
<td>1</td>
<td>56</td>
<td>280</td>
</tr>
<tr>
<td></td>
<td>GWP100</td>
<td>1</td>
<td>21</td>
<td>310</td>
</tr>
<tr>
<td></td>
<td>GWP500</td>
<td>1</td>
<td>6.5</td>
<td>170</td>
</tr>
</tbody>
</table>

⁷ The First Law of Thermodynamics states that energy is neither created nor destroyed in any transformations. The Second Law of Thermodynamics states that energy becomes less available to perform useful work as it passes through successive transformations (i.e. entropy increases).

⁸ This has parallels with the EPS Enviro-Accounting Method where it is suggested that species can be valued in relation to the time taken for evolution of a “replacement” species. Large mammals take 100,000 to 1,000,000 years to evolve while insects may take 10,000 years or less (Steen and Ryding, 1992, p.20).
Stratospheric Ozone Depletion

Similarly to GWPs for global warming, Ozone Depletion Potentials (ODPs) are used to assess ozone depletion in LCA. These are generally the ODPs calculated by the World Meteorological Office. They describe ozone destruction in the stratosphere by different chlorinated or brominated compounds in relation to that of CFC-11, once they are in an equilibrium state in the stratosphere. However, it should be noted that the figures currently available exclude the effects of compounds such as CH₄, N₂O, CO, non-methane hydrocarbons and carbonyl sulphide (COS). The complexity of the processes by which these compounds contribute to ozone depletion precludes any estimate of their effects at the present time.

Since no atmospheric emissions contributing to ozone depletion were assessed in the LCA presented in this thesis, the ODPs are not listed in this chapter.

Acidification

Acidification is assessed in relation to release of H⁺ ions caused by different substances. Heijungs et al. (1992b, p.100) suggest that the potential to form H⁺ ions should be assessed, and therefore the following factors should be used:

- One mole SO₂ forms two moles H⁺
- One mole HCl forms one mole H⁺
- One mole NOₓ forms one mole H⁺
- One mole NO₃⁻ forms one mole H⁺
- One mole NH₃ forms one mole H⁺.

IA factors are then developed that relate these molar values of H⁺ production to the mass of relevant substances, in relation to the Acidification Potential (AP) of SO₂.

A modification of this approach involves assessing the most likely contribution to acidification made by different compounds. For example, Lindfors et al. (1995) explain that acidification of water bodies depends partly on the quantities of anions (SO₄²⁻, Cl⁻, NO₃⁻) in surrounding soils that leach into these water bodies. In central and northern Europe, most sulphates and chlorides are leached quite quickly from the soil but this is not the case for N-compounds where, typically, only a small proportion of these compounds will leach out. Instead, the nitrogen in the soil tends to be
incorporated into biomass when it is available because it is usually in short supply (see Chapter V, Section 2.1). Therefore, maximum and minimum scenarios can be defined for assessing acidification:

- Maximum scenario: uses the IA factors calculated by Heijungs et al. (1992a) for sulphur and halogenated compounds, and N-compounds.
- Minimum scenario: uses IA factors calculated by Heijungs et al. (1992a) for sulphur and halogenated compounds but assumes that the IA factors for N-compounds are zero.

Acidification factors for these different scenarios are presented in Table 8. They are the values for the maximum and minimum scenarios described above, including factors for Cl; and water emissions of Cl; F; nitrates and sulphates in addition to those listed in Heijungs et al. (1992a). The possibility of using different Impact Assessment factors to assess acidification suggests that site-dependent Impact Assessment may be desirable, and I return to this issue in Chapter III, Section 3; and Chapter 7, Sections 2.1 and 5.1.

**Photochemical Oxidant Formation**

The methodology developed for this approach assumes that ozone is representative of all oxidants that may be formed due to release of different substances. Photochemical Oxidant Creation Potentials (POCPs) are used as IA factors. They describe the change in ozone concentration due to a small increased release of a substance in relation to that caused by a small increased release of ethylene. Relevant substances are usually Volatile Organic Compounds (VOCs) but their action is dependent upon the presence of NOx and ultraviolet light; CO may also make a contribution to ozone formation.

POCPs have been calculated for a range of VOCs, and data are presented in Heijungs et al. (1992), Andersson-Sköld et al. (1992) and Finnveden et al. (1992). All these datasets present a number of different POCPs for each substance, depending upon assumptions about relevant background concentrations of NOx, whether maximum and/or minimum values are used in calculating the factors, and the time period used in assessing ozone formation. Using the different data, it is possible to calculate POCPs for scenarios such as:

- Minimum contribution (using lowest POCP values)
- Maximum contribution (using highest POCP values)
- Four days average measured under Swedish conditions (i.e. low NOx background concentration)
- Four days average measured with high NOx background concentration.
In addition, it may also be considered desirable to measure the contribution of NO\textsubscript{x} to POCP. Indeed NO\textsubscript{x} is more important than VOCs in determining ozone production in many parts of Europe (Lindfors \textit{et al.}, 1995, p.115). To date, the only suggestion for incorporating NO\textsubscript{x} into this Impact Assessment category requires making it a separate sub-category with NO\textsubscript{x} measured simply as "grammes NO\textsubscript{x}". In other words, two values are used to assess photochemical oxidant formation: NO\textsubscript{x} (as "grammes NO\textsubscript{x}") and VOCs (mass of all VOCs weighted using POCPs and then summed).

Lindfors \textit{et al.} (1995, p.117) also suggest that it may be relevant to consider CO and CH\textsubscript{4} as further sub-categories, although Finnveden \textit{et al.} (1992) give a POCP value for CO, and Heijungs \textit{et al.} (1992) give a POCP value for CH\textsubscript{4} that enable these substances to be aggregated within the VOCs sub-category.

Table 9 presents the data for a number of scenarios. The data for the minimum and maximum scenarios are taken as the minimum and maximum values from the ranges given in Heijungs \textit{et al.} (1992a) and Lindfors \textit{et al.} (1995).

\textit{Eutrophication}

Eutrophication refers to the addition of nutrients to soil or water, leading to increased biomass production and oxygen depletion in the receiving medium during organic matter decomposition. It is sometimes called "nutrification" because it is triggered by addition of nutrients to the receiving medium. IA factors are defined as the potential of a nutrient to form organic matter in relation to that of phosphorus. Organic matter is assumed to have the generic composition: $C_{106}H_{252}O_{116}N_{16}P$ (Heijungs \textit{et al.}, 1992a, p.87). Therefore, one mole P contributes to one mole organic matter, and one mole N contributes to $\frac{1}{16}$ mole organic matter. Releases of organic matter are also included in the assessment using COD as a unit of measurement. Since BOD and COD measure the oxygen consumed in decomposition of organic matter, and assuming that 138 moles O\textsubscript{2} are required for decomposition of one mole organic matter (Heijungs \textit{et al.}, 1992b, p.101), one unit BOD or COD (measured as O\textsubscript{2}) contributes to $\frac{1}{138}$ mole organic matter.
Table 8. Acidification Factors

<table>
<thead>
<tr>
<th>Source</th>
<th>Air Emissions</th>
<th>Water Emissions</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>NH₃</td>
<td>NOₓ</td>
</tr>
<tr>
<td>Heijungs et al., 1992a</td>
<td>1.88</td>
<td>0.7</td>
</tr>
<tr>
<td>Minimum</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Maximum</td>
<td>1.88</td>
<td>0.7</td>
</tr>
<tr>
<td>Preferred values</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Minimum</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Maximum</td>
<td>1.88</td>
<td>0.73</td>
</tr>
</tbody>
</table>

Table 9. POCP Factors

<table>
<thead>
<tr>
<th>Source</th>
<th>Air Emissions</th>
<th>HC excl. CH₄</th>
<th>CH₄</th>
<th>Chlorinated HC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Heijungs et al., 1992a</td>
<td></td>
<td>0.416</td>
<td>0.007</td>
<td>0.021</td>
</tr>
<tr>
<td>Minimum for 11 days, three scenarios</td>
<td>0.195</td>
<td>0</td>
<td>0.003</td>
<td></td>
</tr>
<tr>
<td>Maximum for 11 days, three scenarios</td>
<td>0.799</td>
<td>0.030</td>
<td>0.048</td>
<td></td>
</tr>
<tr>
<td>Finnveden et al., 1992, in Lindfors et al., 1995</td>
<td>High NOₓ background over 0-4 days (average)</td>
<td>0.032</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ordinary Swedish background over 0-4 days (average)</td>
<td>0.04</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Maximum difference in ozone formation</td>
<td>0.036</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Preferred values</td>
<td></td>
<td>0.032</td>
<td>0.195</td>
<td>0.003</td>
</tr>
<tr>
<td>Minimum</td>
<td>0.04</td>
<td>0.799</td>
<td>0.03</td>
<td>0.048</td>
</tr>
<tr>
<td>Maximum</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

* Scenarios for Germany-Ireland, France-Sweden and the UK.

* High NOₓ concentration is approximately sixteen times greater than ordinary Swedish background (Andersson-Sköld et al., 1992).

* In this case, the maximum is smaller than the average value for ordinary Swedish background conditions. The maximum is calculated as the POCP for the point where a maximum difference in ozone formation occurs between air with and without additional CO.

Table 10. Eutrophication Factors

<table>
<thead>
<tr>
<th>Source</th>
<th>Air Emissions</th>
<th>Water Emissions</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>NOₓ</td>
<td>P₂O₅</td>
</tr>
<tr>
<td>Heijungs et al., 1992a</td>
<td>0.13</td>
<td>0</td>
</tr>
<tr>
<td>Preferred values</td>
<td>N-limited</td>
<td>0.13</td>
</tr>
<tr>
<td>P-limited</td>
<td>0</td>
<td>1.34</td>
</tr>
</tbody>
</table>
Table 11. Toxicity Factors for Soil Emissions

<table>
<thead>
<tr>
<th>Source</th>
<th>As</th>
<th>Cd</th>
<th>Cr</th>
<th>Co</th>
<th>Cu</th>
<th>Fe</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Ecotoxicity</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Heijungs et al., 1992</td>
<td>Aquatic</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Terrestrial</td>
<td>3.6</td>
<td>13</td>
<td>0.42</td>
<td>0.42</td>
<td>0.77</td>
</tr>
<tr>
<td>Guinée et al., 1996</td>
<td>Aquatic (x10^3)</td>
<td>0.0018</td>
<td>0.27</td>
<td>0.00076</td>
<td>0.0044</td>
<td>0.00084</td>
</tr>
<tr>
<td></td>
<td>Terrestrial (/10^3)</td>
<td>270</td>
<td>470000</td>
<td>820</td>
<td>62</td>
<td>3400</td>
</tr>
<tr>
<td>Audsley et al., 1997</td>
<td>Aquatic</td>
<td></td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td></td>
<td>Terrestrial</td>
<td>2.3</td>
<td>9.6</td>
<td>0.27</td>
<td>-</td>
<td>0.27</td>
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<td><strong>Human Toxicity</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Heijungs et al., 1992</td>
<td></td>
<td>0.043</td>
<td>7</td>
<td>0.018</td>
<td>0.065</td>
<td>0.0052</td>
</tr>
<tr>
<td>Guinée et al., 1996</td>
<td></td>
<td>910</td>
<td>28000</td>
<td>470</td>
<td>2100</td>
<td>42</td>
</tr>
<tr>
<td>Audsley et al., 1997</td>
<td></td>
<td>78000</td>
<td>160000</td>
<td>32000</td>
<td>111000</td>
<td>1260</td>
</tr>
<tr>
<td><strong>General Toxicity</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Habersatter, 1991 (x10^3)</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Heijungs et al., 1991 (x10^3)</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Source</th>
<th>Hg</th>
<th>Mo</th>
<th>Ni</th>
<th>Pb</th>
<th>Se</th>
<th>Sn</th>
<th>V</th>
<th>Zn</th>
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<tbody>
<tr>
<td><strong>Ecotoxicity</strong></td>
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<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Heijungs et al., 1992</td>
<td>Aquatic</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Terrestrial</td>
<td>29</td>
<td>-</td>
<td>1.7</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>2.6</td>
</tr>
<tr>
<td>Guinée et al., 1996</td>
<td>Aquatic (x10^3)</td>
<td>16</td>
<td>-</td>
<td>0.036</td>
<td>-</td>
<td>-</td>
<td>0.0051</td>
<td>0.0026</td>
</tr>
<tr>
<td></td>
<td>Terrestrial (/10^3)</td>
<td>18000</td>
<td>-</td>
<td>710</td>
<td>-</td>
<td>-</td>
<td>1700</td>
<td>2400</td>
</tr>
<tr>
<td>Audsley et al., 1997</td>
<td>Aquatic</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
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</tr>
<tr>
<td></td>
<td>Terrestrial</td>
<td>19</td>
<td>-</td>
<td>1.1</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>1</td>
</tr>
<tr>
<td><strong>Human Toxicity</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Heijungs et al., 1992</td>
<td></td>
<td>0.15</td>
<td>0.7</td>
<td>0.014</td>
<td>0.025</td>
<td>-</td>
<td>-</td>
<td>0.007</td>
</tr>
<tr>
<td>Guinée et al., 1996</td>
<td></td>
<td>29000</td>
<td>-</td>
<td>1100</td>
<td>480</td>
<td>-</td>
<td>220</td>
<td>17</td>
</tr>
<tr>
<td>Audsley et al., 1997</td>
<td></td>
<td>400000</td>
<td>-</td>
<td>-</td>
<td>20000</td>
<td>560000</td>
<td>80</td>
<td>-</td>
</tr>
<tr>
<td><strong>General Toxicity</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Habersatter, 1991</td>
<td>-</td>
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<td>-</td>
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<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Heijungs et al., 1991</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

35
IA factors are then calculated that relate these molar values to the mass of relevant substances, in relation to the eutrophication potential of P. Using these values, the eutrophication potential of a system can be calculated by multiplying all relevant emissions by their IA factors, and adding the results. However, in practice only one nutrient is likely to limit biomass production at any one time. Therefore, a number of researchers have suggested that the results should be presented in sub-categories. Lindfors et al. (1995) suggest the following five sub-categories:

- For terrestrial ecosystems:
  1. N emissions to air
- For aquatic ecosystems:
  2. P emissions to water + organic matter to water
  3. N emissions to water + organic matter to water
  4. N emissions to water + organic matter to water + N emissions to air
  5. P emissions to water + N emissions to water + N emissions to air + organic matter to water.

These categories are suggested because N tends to be the limiting nutrient in terrestrial ecosystems, and either P or N can be limiting in aquatic ecosystems. N emissions to air are included in sub-category 4 because these emissions may have an effect if they precipitate onto water surfaces or are leached from terrestrial ecosystems. Sub-category 5 represents the maximum scenario where all relevant substances make a contribution.

Table 10 presents the eutrophication factors for different substances using this approach. The preferred values are those that account separately for N-limited and P-limited environments, because either nitrogen or phosphorus limits in any one environment rather than both nutrients. In the UK, phosphorus tends to be limiting in freshwater ecosystems and nitrogen in saline ecosystems.

Toxicity

Methods for toxicity assessment have been reviewed recently by the SETAC Working Group on Impact Assessment (Jolliet, 1996). Jolliet suggests that four different elements can be distinguished in development of toxicity IA factors:

1. The effect (i.e. the intrinsic toxicity) of a substance
2. The fate of a substance
3. The influence of background conditions
4. Geographical and time issues.

Impact Assessment methods for toxicity take account of these different elements to varying extents, and there is no consensus at present over the best approach. Earlier assessment methods focused on the first element (for example, Heijungs et al., 1992a, 1992b) but more recently researchers have been attempting to incorporate fate (Guinée et al., 1996) and background conditions (Potting and Hauschild, 1997a, 1997b; van Dokkum et al., 1997) into the IA factors. The fourth element, geographical and time issues, concerns the level of detail at which the toxicity of any one emission is assessed: for example, there may be only one generic IA factor for emissions to soil of substance $i$ or there may be separate IA factors for emissions to different types of soil of substance $i$. Thus Guinée et al. (1996) give IA factors for emissions to agricultural, industrial and generic soils.

Toxicity is usually assessed as both human and ecotoxicity. Ecotoxicity adds an additional layer of complexity in the analysis because it is concerned with impacts on all species rather than one species (i.e. *Homo sapiens*). Thus one must consider whether toxicity is assessed in relation to the most sensitive species in ecosystems or some hypothetical "average" species, and the extent to which bioaccumulation of toxic substances in food chains should be incorporated into IA factors. In practice, since there are relatively limited data available on the toxicity of most substances to any species, most approaches use the data that are available as a basis for developing IA factors (usually based on studies of organisms such as algae, crustaceans and fish).

Tables 2, 3 and 11 presents a number of IA factors calculated for both ecotoxicity and human toxicity; detailed data for pesticides are given in Chapter VI. The different methods are described in more detail in Appendix II.1. The factors are not directly comparable across different methods because each method calculates the factors in a different way. However, some interesting comparisons can be made concerning the relative differences between substances using each method. For example, considering human toxicity, the ratio of IA factors for NO$_x$ and lead for air emissions is 1:205, 1:257,692 and 1:4,564 for Heijungs et al. (1992), Guinée et al. (1996) and Audsley et al. (1997) respectively. For the ratio of IA factors for NO$_x$ and SO$_2$ for air emissions, the ratios are 1:1.54, 1:1.63 and 1:0.26 for the three methods. Although the differences between Heijungs et al. (1992) and Guinée et al. (1996) can be explained to a certain extent by the incorporation of fate (including bioaccumulation) in the latter method, there still seem to be many uncertainties in this type of analysis.
Other Factors

A number of other factors have been suggested as relevant for inclusion in LCAs, and are briefly described below.

- **Radiation**
  A method for assessing radiation has been proposed by Solberg-Johansen (Solberg-Johansen, 1998; Solberg-Johansen et al., 1997).

- **Waste Heat**
  Heijungs et al. (1992b, p.77) suggest measuring heat emissions to water in MJ.

- **Odour**
  Heijungs et al. (1992a, p.87-8; 1992b, p.78-9) define odour threshold values (OTVs) for a number of substances emitted into air (using ammonia as a reference substance). The OTV represents the "concentration of a given substance under defined standard conditions at which 50% of a representative sample of the population can just detect the difference between a sample of air mixed with that substance and a sample of clean air." IA factors are calculated using a Critical Volumes-type approach where relevant emissions are divided by their OTVs.\(^9\)

- **Noise**
  Heijungs et al. (1992b, p.37) suggest an approach based on the sound pressure level of a system (measured in decibels).

- **Working Conditions**
  Opinions vary about whether working conditions should be included in an LCA. Potential approaches are discussed in Potting et al. (1997). However, assessment of this aspect is outside the scope of this thesis.

4.2.3 Disturbances

Relatively little attention has been given to assessment of disturbances in LCA. Issues that have been suggested as relevant in this group are:

- **Physical Ecosystem Degradation**
  Current approaches are discussed and a new methodology is developed in Chapter IV.

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\(^9\) In the LCA presented in Chapter VI, ammonia was the only relevant substance in this Impact Assessment category; since it is the reference substance for the category, its OTV is unity.
• **Landscape Degradation**

Assessment of landscape degradation involves subjective judgements about the aesthetic desirability of alternatives (see, for example, Blonk *et al.*, 1997). For example, some people prefer landscapes with many trees while others prefer large open spaces. Another example is oilseed rape: in the Netherlands, people travel to areas where rape is grown to see the yellow fields, while in the UK the colour is perceived as a negative impact on the landscape (ETSU, 1996; Mattson *et al.*, 1988). This issue is therefore somewhat different from most other issues considered in LCA (which have a more objective basis for assessment), and may not be appropriate to include in an analysis. It is not considered in further detail in this thesis.

• **Desiccation**

Heijungs *et al.* (1992b, p. 80-81) suggest that “harmful” use of water should be assessed as use of water in relation to local and regional reserves of surface- and ground-water. “Harmful” use of water is defined as use that causes desiccation of “nature” but the rationale and methodology are not developed in further detail. Wegener Sleeswijk *et al.* (1996) suggest that desiccation should be assessed as groundwater abstraction apart from areas where desiccation is not a problem. I present an alternative method in Chapter III, Section 5.2.

• **Direct Victims**

Similarly to working conditions, opinions vary about the validity of including human injuries and fatalities in an LCA. It has not been considered in this thesis.

### 4.3 Normalisation and Valuation

The objective of normalisation and valuation in LCA is to present the environmental impacts of the system under analysis in a form that is useful for the purpose of the study and that can be understood by users of the study results. Normalisation is usually undertaken in relation to a defined geographical area; however, other alternatives include normalisation in relation to an average person’s annual contribution to each impact or in relation to impacts caused by familiar products or activities (such as electric fires or cars). Some alternatives are explored in more detail in Chapter VII.

Valuation raises the same methodological issues for LCA as exist in all types of multi-criteria analysis. Since a variety of alternative methods have been developed outside the LCA research community to address these issues, valuation is not considered in further detail in this thesis.

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10 The subject of Valuation in LCA is discussed extensively in Braunschweig *et al.* (1996).
4.4 Assessment of Potential Versus Actual Impacts (Site-Dependency In LCA)

Until relatively recently, the emphasis in LCA methodology development has been on assessment of potential impacts. However, a number of researchers have questioned the validity of this approach (see, for example, Owens, 1995 and 1996), arguing that actual, site-dependent impacts may be very different from potential impacts\(^{11}\).

The logic of this argument has been acknowledged in a number of Impact Assessment categories. For example, Lindfors et al. (1995) recommend assessment of eutrophication in a number of different sub-categories because actual impacts depend upon the ecosystems receiving the relevant emissions (see Section 4.2.2 above). For photochemical oxidant formation, they recommend assessing POCP in sub-categories defined by, among other factors, whether there are high or low NO\(_x\) background conditions.

This suggests that, in general, IA category results whose magnitude is affected by background conditions should be given as a range or set of alternative results rather than one single value. For example, for eutrophication and photochemical oxidant formation at least the highest and lowest values resulting from the different sub-categories should be presented in the LCA results. Furthermore, it also implies that where more data are available on actual background conditions, these should be used to calculate the Impact Assessment results rather than a theoretical potential impact. The principle is that LCA results should be as realistic as possible within the constraints of the data available for the study. This is discussed in more detail in Chapter VII, Section 5.1.

4.5 Alternative Impact Assessment Methods

A number of alternative method for Impact Assessment have been developed for use in LCA, and have been reviewed in Baumann and Rydberg (1994) and Grisel et al., (1994).

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\(^{11}\) There is some confusion about the meaning of the terms "potential" and "actual" in this context. Here, I take potential impact to mean "the potency of emissions or extractions as contributors under certain conditions to different types of impact" (Udo de Haes, 1996, p.12). I take "actual" impact to mean "the most accurate assessment of the potency of emissions or extractions as contributors under known conditions to different types of impact." For example, using this meaning of the terminology, assessing acidification as the potential of acidifying substances to form hydrogen ions would give a potential impact. Assessing acidification in relation to the location of release and fate of acidifying substances (i.e. site-dependent Impact Assessment) would give results closer to the actual impact. Measuring changes in the pH of environments affected by the release of these particular acidifying substances would give the actual impact, at least at this point in the cause-effect chain.
One method is the **EPS System** (Steen and Ryding, 1992). Two sets of weighting factors are given in this method: one for resources and one for emissions. These are used to weight all the resources used and emissions released as identified in Inventory Analysis. The weighting factors are calculated on the basis of i) society’s judgement of the importance of the environmental problem; ii) the intensity and frequency of the problem; iii) location and timing of the impact; iv) the contribution of the total effect in question; and v) the cost of decreasing the emission by one weight unit. This gives an “environmental load unit” (ELU) for the system under study, representing the “willingness to pay for avoiding negative effects on the safeguard subjects” (Steen and Ryding, 1992, p.1). The safeguard subjects are defined as: human health, biological diversity, production, resources and aesthetic values. A disadvantage of the method is its mixing together of ecological, sociological and economic effects in a way that lacks transparency (Grisel *et al.*, 1994).

A second approach is the **Eco-Points Method** (Ahbe *et al.*, 1990; Braunschweig, 1991). One set of weighting factors is given in this method, called “ecofactors,” used to assign relative weights to all the emissions released as identified in Inventory Analysis. These are then summed to give the “ecopoints” for the system under analysis. Each ecofactor is a ratio of the “critical load” of a pollutant to the actual emission within a defined geographical region (and so is more site-dependent than some other methods). The critical load may be defined as either an ecologically critical load (i.e. the load at which living organisms are affected) or as a politically maximum acceptable limit (political target). The only resources considered in the method are energy resources which are assigned an ecofactor of 1 ecopoint per MJ. Again, the method suffers from its lack of transparency.

It is also worth mentioning the **Eco-Indicator method** because it has received considerable attention and is used in a number of design applications (Goedkoop, 1995). It uses the standard LCA Impact Assessment methods described above but then applies weighting factors to the different Impact Assessment categories, calculating final “indicator scores” for different materials and processes used in products. The weighting factors are calculated using a Distance-to-Target method; in this method, the seriousness of an impact is related to the difference between the current and target values for Europe. The user simply has to multiply the quantities of materials and processes used in a system by the relevant indicator scores in order to obtain a total score representing the overall environmental impacts of the system. Of course, the weakness of this approach is that assumptions about the importance of different types of impacts are not transparent in the method.
5. Improvement Assessment (Interpretation)

The final phase of an LCA is Improvement Assessment (called Interpretation in ISO 14043). During this phase, the results of the analysis are discussed and opportunities for reducing the environmental impacts of the functional unit are identified and evaluated. Azapagic (1996) presents the only systematic methodology for undertaking Improvement Assessment proposed to date.

6. Current Status of LCA

The brief description of LCA presented in this chapter has shown that LCA is an evolving environmental management approach. Protocols for its use have developed relatively rapidly over the last few years but there are still a number of areas requiring further methodological development. These include:

- System modelling
- Role of uncertainty
- Impact Assessment methods
- Improvement Assessment (Interpretation).

An objective of LCA studies is to build up environmental impact models that are as representative as possible of reality, and good system modelling is the key to achieving this objective. The LCA approach currently makes the assumption that any system under analysis behaves as a linear homogeneous system; in other words, a change in the quantity of the functional unit leads to a linear change in all the environmental burdens associated with the system. While this may be true (or can at least be used as a working approximation) for incremental changes in the functional unit, the assumption is unlikely to apply for larger changes12. For example, an incremental increase in the quantity of a functional unit may require additional electricity. This can be modelled by assuming that extra electricity is generated at the margin; however, if the additional requirement is significant and sustained over a period of time such a modelling assumption is inappropriate because the electricity will be drawn from the base load rather than the margin. One response is to define alternative scenarios based on different assumptions about technology requirements to account for non-incremental changes due to the system under analysis (Clift et al., 1998). This is particularly

12 Indeed, Hofstetter (1996) describes LCA as a micro- rather than macro-instrument because it models incremental change.
important when LCA is used in national policymaking because alternative policy scenarios at this level of analysis are very unlikely to be linked with incremental changes in existing systems. The challenge for LCA is, therefore, how to account for such situations in a systematic way (Wrisberg et al., 1997, p.18). Other aspects related to improved system modelling include distinguishing between product- and process-related environmental burdens (Eggels and van der Ven, 1994; Clift et al., 1996), and accounting for changes in processes of secondary interest in a study using the Foreground/Background approach (Udo de Haes et al., 1994; Clift et al., 1997).

Uncertainty in LCA is often discussed but few LCAs systematically incorporate it in their results. Uncertainty may be related to data quality and/or omissions, methodological choices in LCAs (for example, concerning allocation and Impact Assessment methods), and assumptions made during the analysis (for example, location of system boundaries) (Wrisberg et al., 1997, p.22). As for system modelling, there is therefore a need to systematically account for uncertainty in LCA studies.

As shown in this chapter, there are still deficiencies and gaps in existing Impact Assessment methods. A certain uneasiness with current approaches has also led to increasing interest in the concept of safeguard subjects as a basis for developing alternative Impact Assessment methods (see, for example, Finnveden and Lindfors, 1997). However, progress in this area is most likely to be achieved when a greater emphasis is placed upon the purpose of LCAs, and development of Impact Assessment methods that are useful to users of the study results. The same applies for Improvement Assessment which has received very little attention.

This links to the other areas of LCA that have become a focus of attention. These are concerned with the positioning and applications of LCA in decision-making processes. The areas of interest can be defined as:

- Streamlining methods
- Integration with other environmental management tools
- Role of LCA in decision-making processes.

Streamlining in LCA was the subject of one of the SETAC-Europe Working Groups (the “Screening and Streamlining Working Group”) that operated between 1994 and 1997 (Christiansen, 1997). Recognising that all LCAs are streamlined to a greater or lesser extent, the Working Group aimed to define a systematic approach for undertaking streamlined studies. Meanwhile, another SETAC-Europe Working Group (on “Conceptually Related Programmes”) focused on the complementarity
(or otherwise) of LCA with other environmental management tools (SETAC-Europe Working Group, 1997). The work of both these groups was taken forward in the EU LCANET project which considered “positioning and applications of LCA” as one of its four themes (Cowell et al., 1997). All these initiatives have pointed to the importance of considering LCA within its decision-making context if it is to have a useful role in environmental management. However, the implications of this alternative perspective for development of LCA methodology require further consideration.

Thus LCA has reached an interesting point in its conceptual and methodological development. Two main interlinked themes have emerged as foci of research attention: rigorous methodological development using systems analysis as a modelling approach, and adaptation of LCA so that it is more responsive to decision-making contexts. Aspects of these two themes are developed in this thesis.

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Guinée, J.B. Pers.comm. E-mail from Jeroen Guinée, CML, Leiden University, dated 3 August 1998.


CHAPTER III

LCA AND AGRICULTURAL SYSTEMS

“There is a growing consensus that a spatial concentration of agricultural production on high-potential land would be the most sustainable strategy, leading to the most efficient resource use and the least damage to ecosystems. However, this option is often socially unacceptable.” (Fresco and Kroonenberg, 1992)

“There is a strong case for including a significant element of low input and organic agriculture in a sustainable agriculture strategy for Britain. This is necessary to maintain and enhance certain semi-natural habitats and protect sensitive areas as well as offering a system of farming based on sound management of the soil and very limited use of critical inputs such as fertilisers and pesticides. Consumer confidence in the rigour and credibility of claims made by proponents of different systems will also be crucial.” (Baldock et al., 1996, p.65)

1. Introduction

LCA was developed for the assessment of industrial systems. In the last few years, however, there has been increasing interest in applying LCA to assess the environmental impacts and, ultimately, the sustainability of agricultural systems. Agricultural systems are sufficiently different from industrial systems1 that this area of application introduces new methodological issues for all phases of LCA. These issues are discussed in three sections in this chapter: site-dependency (Section 3), general methodological issues (Section 4) and impact assessment issues (Section 5). However, firstly I outline the development of this new research area, showing how key initiatives and publications have shaped the current research agenda.

2. Development of a New Research Area: LCA and Agricultural Systems

The first internationally coordinated initiative on LCA and agricultural systems was an Expert Seminar held on 22-23 November 1993 in Lyngby, Denmark, organised by the Ecological Food

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1 Agricultural systems are those involving human activities carried out primarily to produce food and fibre (and, increasingly, fuels and other materials) by the deliberate and controlled cultivation of plants and animals (derived from definition in Spedding, 1988, p.5). Industrial systems are those involving human activities to produce products and deliver services without the cultivation of plants and animals. In the food chain, most
Project of the Interdisciplinary Centre, Technical University of Denmark. This brought together 34 European LCA researchers from universities, government organisations and companies to present their work on LCA and food production. The topics varied from national and international processing standards for "ecological" food, to waste management in brewery operations, to weed control on highways. The proceedings (Weidema, 1993a) and three other reports published around the same time (Andersson et al., 1993a, 1993b; Weidema, 1993b) summarised the existing research initiatives on LCA and food production. In the words of Andersson et al. (1993a, p.2), "reports on LCA incorporating whole food production systems are very scarce."

Subsequently an EU Concerted Action on "Harmonisation of Environmental Life Cycle Assessment for Agriculture" was funded, and held its first workshop in June 1995. It involved researchers from nine organisations in six countries who met for two one-week closed workshops in June 1995 and January 1996. Their research was presented at a two-day seminar at Silsoe Research Institute, Silsoe, UK, in June 1996, and was recently published in a comprehensive report (Audsley et al., 1997). The report includes an LCA of wheat production from three different farming systems, and summarises the current state-of-the-art in LCA methodology for assessment of agricultural systems.

In April 1996, a highly successful international conference was organised by VITO ("International Conference on Application of LCA Agriculture, Food and Non-Food Agro-Industry and Forestry: Achievements and Prospects"). At this conference, 20 papers and 15 posters were presented on topics ranging from definition of the functional unit, to crop rotations, to allocation among co-products (Ceuterick, 1996). Case studies were also presented on products ranging from pork and lamb meat to paper to cleaning agents. The conference was attended by 116 delegates from many European countries plus Nigeria, Brazil and Japan.

The next stage in consolidation of the growing research community on LCA and agricultural systems will be the EU Concerted Action project "LCANET-FOOD." This has recently been funded and will held its first meeting in April 1998. The aim of the project is to build a European network for LCA research on the food chain, evaluate state-of-the-art methodology in application of LCA to the food chain, and promote the formation of a cross-Europe database for use in LCAs of the food chain (LCANET-FOOD, 1996).

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food products are produced by a combination of agricultural and industrial processes. For example, both agricultural production of wheat and industrial processing of grain are required to produce a loaf of bread.
Table 1 lists key organisations and individuals involved in ongoing research concerning LCA and agricultural systems, alongside a selection of their publications. In the next two sections, I show how these research efforts, and those of others, have contributed to the development of LCA methodology for assessment of agricultural systems.

3. Assessment of Potential versus Actual Impacts (Site-Dependancy In LCA)

There is a continuing debate in the LCA research community about the extent to which LCAs should include site-dependent data in the Inventory Analysis and undertake site-dependent Impact Assessment. The issue is illustrated in Figure 1. This diagram shows siting decisions for a given facility (A) where Environmental Impact Assessment (EIA) is a typical environmental analysis approach, and choices between technologies independent of their sites (B) where LCA is a typical environmental analysis approach. However, it can be argued that LCA has a role to play in the A-type analysis, and indeed in the C-type analysis which involves assessing the environmental impacts of different technologies at specified different sites. In effect, this amounts to legitimising the choice of location as a valid difference between alternative systems, alongside choice of technology.

In the context of agricultural production, legitimising choice of location as a valid difference between systems is particularly important because site-dependent aspects can have a greater influence on LCA results than activity-dependent aspects (Cowell and Clift, 1998). In particular, the climate and soil type may determine final yield levels more than agricultural activities such as application of synthetic fertilisers and use of pesticides. This is recognised by the agencies that monitor soil quantity and quality, and “Land Capability Classification” maps have been developed by the Soil Survey in the UK. They grade land based on the crops suitable for cultivation and their levels of yield; this classification only takes into account physical properties of the area (such as texture of the soil, slope, drainage and climate) (Davies et al., 1993, p.263-6).

The question of site-dependency also deserves further attention in LCAs of industrial systems. Although inputs and outputs (apart from transport- and electricity-related burdens) may be unrelated to location, this is not necessarily true for assessment of impacts. Examples include the dependence of acidification and eutrophication impacts on the sensitivity of receiving media (see Chapter II, Sections 4.2.2 and Section 4.4), and the proximity of human populations to points of release of short-lived toxic substances in determining human toxicity values (Cowell, 1997). I discuss this issue in more detail in Chapter VII, Section 5.1.

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<tr>
<th>Country</th>
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In this thesis, therefore, I assume that site-dependency should be a valid consideration in LCA. A practical consequence of this assumption is that the results for all types of impacts whose magnitude depend upon location should be represented in LCA results as, preferably, a) specific values where location is known, or b) a range of values where actual location is unknown (see discussion in Chapter VII, Section 5.1). In effect, for the Problem-Oriented method, this means that site-dependent Impact Assessment should be included in the assessment for all the common Impact Assessment categories (marked with an asterisk in Chapter II, Table 4) apart from abiotic resources, global warming and ozone depletion (which have global rather than regional or local impacts).

4. Methodological Issues

Assessment of systems involving agricultural production raises new methodological issues at all phases of LCA. At the Goal Definition and Scoping, and Inventory Analysis phases of LCA, these issues concern definition of the functional unit and system boundaries for a study, and choices over inclusion or exclusion of ancillaries and atmospheric deposition of nutrients and heavy metals. Furthermore, agricultural systems characteristically produce more than one economic output (“co-products”) and have closely linked sub-systems (through flows of co-products and agricultural “wastes”). Therefore, decisions about allocation methodology have particularly important implications for the results of studies involving agricultural production. These issues are discussed in more detail in Sections 4.1 to 4.5 below.
4.1 Functional Unit

As for industrial systems, definition of the functional unit in systems involving agricultural production is not always obvious and this has been identified as an issue since the early 1990s (Weidema, 1993a, 1993b). Firstly, one must consider whether the focus of a study is production of foodstuffs or use of land area for different purposes. Although it may have been obvious that the focus should be production of foodstuffs in the years following World War II, when the primary aim of UK agricultural policy was to increase production, more recently the emphasis has switched to delivery of additional services through agricultural production. These include the recreational and aesthetic aspects of land use, maintenance of rural employment and infrastructure, and conservation of wildlife and ecosystems (Cowell, 1996). This change in emphasis is reflected in the government’s introduction of various agri-environment schemes providing financial incentives for farmers to conserve wildlife and threatened ecosystems, the recent White Paper on “Rural England,” and increasing public interest in conservation of existing landscapes threatened by road and other building developments.

Theoretically, these additional services could be included in an LCA whose functional unit is based on mass of a food product by including appropriate Impact Assessment categories. For example, impacts on wildlife and ecosystems can be incorporated into the Impact Assessment phase (see Chapter IV). However, the change in emphasis does actually beg a question about whether a more appropriate functional unit for some studies may be use of a specified land area (as suggested by Cowell, 1996; Gaillard, 1996; Udo de Haes, 1996; and Wegener Sleeswijk, 1993). Indeed, this type of functional unit may be particularly relevant in the context of review of the Common Agricultural Policy in Europe following concerns about over-production, and interest in alternative uses of set-aside land. The appropriateness of such a functional unit for some studies is illustrated by the example of a community (which may be a village, a county, a country or a union of different countries) reconsidering land use policy within its administrative jurisdiction. As part of the review, the community may wish to investigate the potential for reducing the environmental impacts of its activities through its land use policy. In such a context, an LCA might consider, for example, the relative environmental impacts of using a specified land area for:

- Option A: growing rapeseed for processing into biodiesel that displaces mineral oil-sourced diesel in engines of cars used in the community.
- Option B: growing potatoes that displace imported potatoes (with their related transportation energy requirements) eaten in the community.
Here, the functional unit would be the specified land area and the LCA would compare the changes in environmental impacts caused by Option A versus Option B. This type of analysis might conclude that the community should focus on growing crops locally to displace imported foodstuffs rather than growing energy crops on any surplus agricultural land (or vice versa).

If, alternatively, the focus is on production of foodstuffs rather than use of land, then a straightforward definition based on mass of consumed food is not always appropriate. This is because a quantity of food may not adequately represent the service provided by alternative products - and the "service provided" should be the rationale behind choice of the functional unit. To illustrate this point, Weidema (1993a, p.2) gives the example of spreads such as butter and margarine, where the "service provided" is covering a slice of bread, and the amount of spread required is very dependent upon the viscosity of the spread at refrigerator temperature. Wegener Sleeswijk et al. (1996) give another example of a comparison between beef steak and pork Wiener Schnitzel. In this case, the services provided may be provision of calories, protein and/or pleasure. An appropriate functional unit may be the actual portions of meat that would substitute for each other in a meal. Elsewhere we have noted that portions of food are an appropriate basis for comparison (Cowell and Clift, 1995). To summarise, an improved definition of the functional unit could be based on:

- Calories or grammes of protein delivered by alternative products (for example, X calories delivered by Y grammes meat versus Z grammes soyabeans) or
- Meal portions (for example, three slices of thin bread versus three slices thick bread, or one portion of chips versus boiled potatoes).

The most appropriate functional unit for a study will depend upon the behaviour of the consumer, and whether s/he considers the alternatives to be equivalent.

---

2 Actually, this is analogous to the debates in ecolabelling programmes about definition of product categories. For example, are mineral oil-based paints equivalent to water-based paints? Is a paper towel equivalent to a cloth towel? The answer depends upon the behaviour of the consumer, and is shaped by their perceptions about the substitutability of different products.
Gaillard (1996) has put forward a methodology for assessment of a functional unit that incorporates both production of foodstuffs and use of land area. In a comparison of alternative systems delivering a functional unit defined as equivalent quantities (or portions) of foodstuffs, the land areas used in the systems are made the same by cultivating a "complementary crop" on the "surplus" land in one of the systems, as illustrated in Figure 2a (where $P_a$ and $P_b$ are the equivalent foodstuffs from Systems A and B respectively). The production of output $Q$ from the complementary crop in System B is matched by additional production of an equivalent quantity of $Q$ in System A to make the two systems equivalent in their functional outputs (Figure 2b). The additional land area for production of $Q$ is then assessed as a resource issue for System A. In effect, this means that the difference in land areas between two systems producing the same functional output is allocated to the system requiring more land for cultivation of this functional output.

4.2 System Boundaries

As discussed in Chapter II, system boundaries are shaped by:
• Desired geographical applicability of the results.
• Time horizons over which the analysis is relevant.
• Focus of the study which may lead to omission of particular sub-systems or stages of the life cycle.

For **geographical boundaries**, assessment of agricultural systems does not raise any new issues compared with other LCA studies as regards the “geographical applicability” of the results (i.e. decisions about the countries or regions to be described by the system model). However, it does raise issues about the location of geographical boundaries on a more local scale, concerning inclusion or exclusion of soil and field margins in the system model. These issues are discussed in Section 4.3.

For **time-related boundaries**, the new issue that arises concerns crop rotations (as noted by Cowell and Clift, 1995; van Zeijts *et al.*, 1996; and Weidema, 1993a, p.3). Many crops are cultivated and livestock reared in rotations that enhance the overall productivity of the land over a three to six year cycle. This influence on productivity occurs because any one crop’s productivity is partially dependent upon previously cultivated crops. This dependency arises through the medium of the soil. In particular, nutrients, pathogens and weed seeds left in the soil after harvest may affect the productivity of subsequent crops. The importance of this aspect is illustrated by feedwheat: second wheats may have a yield reduction of 12.5% compared with first wheats, and third wheats a reduction of 10-15% below second wheats even with increased application of fertilisers (Nix, 1994, p.4). Another example is oilseed rape: this crop should only be grown once in a four to five year rotation in order to control club root and stem canker (Sandars, 1995, p.69).

Changes in the organic matter content of the soil due to reduced or increased incorporation of organic matter from crops and livestock manure may also affects its productivity, although this tends to occur over a longer time period than one crop rotation. For example, Audsley *et al.* (1997, p.57) point out that soil typically takes from 20 to 50 years to change from one carbon level to another given a change in cultivation pattern. Finally, a “green manure” crop may be grown in a rotation specifically to enhance the soil’s productivity. This crop does not have an economic output but contributes to the productivity of other crops in the rotation.

One way of incorporating these interactions in a crop rotation is to assess whole crop rotations in an LCA. In this type of study, the functional unit is specified quantities of a number of agricultural outputs from one cycle of a crop rotation, i.e. it is a multi-function system. Alternative systems (such as different types of crop rotations) for delivery of the same range and quantity of outputs can then be
compared using LCA. However, in many cases LCA studies are concerned with one type of foodstuff rather than a disparate range, and so functional units consisting of multiple foodstuffs are not helpful. In these studies, impacts on the productivity of future crops due to cultivation of the current crop must be allocated in some way to the system under analysis. This is discussed in more detail in Chapter V.

The omission of particular sub-systems or stages of the life cycle is perhaps the area where definition of system boundaries has most influence on the results. A particularly interesting one, revealing an inconsistency in approach between LCAs of industrial and agricultural systems, concerns the “grave” of studies. In LCAs of industrial systems, waste management of used products is considered an integral part of the analysis. However, LCAs of food products generally stop at the point of consumption of the food. Subsequent waste management, in other words sewage treatment after the food has been digested and egested, is excluded from the analysis. Now, one might argue that sewage treatment is not relevant for inclusion because sewage production (and therefore subsequent treatment) occurs regardless of the food under analysis (as suggested by Tillman, 1993, and Weidema, 1995, p.28, 65-66). However, there are different types of sewage treatment with different environmental implications, and exclusion of this stage of the life cycle could compromise the usefulness of an LCA in increasing understanding of the environmental consequences of human activities, and identifying opportunities for environmental improvements (Cowell and Clift, 1997).

Further sub-systems that form part of agricultural systems but not industrial systems can be classed as ancillaries and are discussed in the next section.

4.3 Ancillaries

Ancillary materials and equipment are included in studies of industrial systems if they make a significant contribution to the LCA results (where the definition of significance level is specific to the system under analysis). Generally, these will be items that wear out relatively frequently and have to be replaced; for example, they may be crates used to transport drink containers from manufacturing sites to filling sites. In other words, the study includes ancillaries whose degradation affects the future productivity of the system where they make a significant contribution to the LCA results. This is equivalent to accounting for the sustainability of the system in the sense that sustainability measures the ability of the system to produce the functional unit in the future as well as in the present.

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1 Ancillaries are defined as materials that contribute to maintenance of processes but are not intended to enter the product (Fava et al., 1991, p.39).
In agricultural systems, careful consideration is needed of what constitutes an ancillary item. Agricultural machinery is an obvious ancillary category, but there is less consensus on whether the soil can be regarded as an ancillary item (see, for example, Wegener Sleeswijk et al., 1996). Elsewhere, we have argued that soil should be considered an ancillary and included within the system boundary of an LCA involving agricultural production because its quantity and quality are closely linked with the farming activities taking place on the land (Cowell and Clift, 1995)4. Indeed, in the same way as iron is processed and formed for use in agricultural machinery, so the soil can also be regarded as a non-renewable resource5 that crosses the system boundary into the farming system, is “processed” and “formed” by agricultural practices such as fertilisation and tillage, and then leaves the system boundary at the end of the time period for the study. However, inclusion of soil as an ancillary also implies that changes in its quantity and quality that affect the future productivity of the system must also be taken into account in the LCA. This is analogous to inclusion of ancillaries in studies of industrial systems when their degradation affects future productivity. The idea is developed in Chapter V.

A further, less obvious category of ancillaries concerns farming infrastructure. Field boundaries such as hedges and fences, and field margins (the areas between the field boundary and crop) are integral parts of farming systems. They provide a number of functions ranging from keeping livestock and other animals in or out of fields, to reducing soil erosion, to providing a habitat for predators of agricultural pests. They have a similar role to buildings in industrial systems whose functions include security and protection of equipment from the weather. However, unlike many buildings in industrial systems, field boundaries and margins can make a significant contribution to the overall environmental impacts of systems under analysis due to their land use and impacts on biodiversity. Therefore, they are relevant for inclusion in LCAs involving agricultural production. However, very few studies have accounted for this aspect in their analysis. One example, at a very simple level, is a study of cleaning products in the metal industry which compared mineral-oil derived solvent products with vegetable oil products (Terwoert et al., 1996a). The study accounted for land use in four categories (natural, modified, cultivated and built land), and the Impact Assessment value for land use was calculated based on restoration of the land to a “reference situation.” Assessment of this aspect is discussed in more detail in Chapter IV.

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4 This is in contrast with some researchers who have regarded the soil as part of the “environment” (Udo de Haes, 1996; Wegener Sleeswijk, 1993).
5 For the purpose of LCA, soil can be regarded as a non-renewable resource because it takes between 200 and 1000 years to form one inch of topsoil under cropland conditions, and longer under pasture and forest conditions (Pimentel et al., 1995).
4.4 Atmospheric Deposition

Nitrogen and heavy metals are deposited onto agricultural land from the atmosphere in addition to inputs in fertilisers. It is debatable whether they should be considered in an LCA study involving agricultural production since they occur regardless of the farming system under analysis. However, it could be argued that nitrogen deposition should be included in the study because it is a consequence of choice of location for the farming system, since nitrogen deposition rates vary between different geographical regions. Van Zeijts et al. (1996) suggest that the technically usable fraction of the deposited nitrogen should be included (about 60% for arable crops and 72% as a maximum value, depending on the growing season of the crop). The implication is that the resulting emissions of N-compounds are also included in the Inventory Analysis. However, it is questionable whether emissions occurring regardless of the system under analysis should be relevant for consideration, and I do not include them in the case study in Chapter VI.

In the case of heavy metals, the same arguments are relevant. However, in addition a proportion of the metals may be taken up into the harvested foodstuff and subsequently contribute to the Human Toxicity Impact Assessment category for a system. This particular toxicity does occur due to the farming system under analysis, and therefore is relevant for inclusion.

4.5 Allocation

The allocation problem in LCA was introduced in Chapter II. Although it is not a new issue for consideration in LCAs involving agricultural production, it assumes a greater importance in these studies because agricultural systems characteristically produce more than one economic output and have closely linked sub-systems (as noted above). This is illustrated in Figure 3 which shows a simplified model of agricultural production in the UK. The model shows that livestock systems characteristically produce more than one economic output (co-products) as do some crop systems (grain and straw from cereals; oil and meal from oilseeds). As well as economic outputs, these agricultural sub-systems also produce waste or “near-to-waste” materials (manure from livestock systems, and harvested non-food parts and plant residues from crop systems). These co-products and waste/near-to-waste materials are recycled within the system through the media of the soil and/or livestock feed. In some cases, these nutrients are recycled for convenience; for example, plant residues may be ploughed back into the soil because this is an easier operation than their removal.
Figure 3. Simplified Flow Diagram To Show Agricultural Production In the UK

Note: “Plant residues” include the roots, leaves and stems of plants that are not harvested but remain on the land or in the soil.
Figure 4. Co-Products From a “Generic” Livestock Carcass

Meat and red offal

Further processing
and/or direct retailing

Wide range of chemicals

Hide/skin

Fellmongering, tanning

Footwear, leather goods, upholstery, clothing, bookbinding, etc.

Blood

Hearing and separation

Albumin and fibrin

Sterilization

Blood meal for animal feedstuffs

Offal not for human consumption

Pig intestines: soak in salt water and cook

Salt

Mincing

Feathers

Processing for pharmaceutical applications

Glands

Cattle feet

Wash and cook

Hydrogen peroxide solution

Soak cow heels in hydrogen peroxide solution

Heel bone and toenail for dog chews

Footwear, leather goods, upholstery, clothing, bookbinding, etc.

Boiling of bones

Stock

Grind bones and drying meal

Soup stock

Bonemeal for petfood

Steamting

Bones

Grinding bones

Bonemeal for fertilizer or petfood

Fat

Gelatin

Gelatin solution

Calcium phosphate

Filteration, concentration, chilling, drying, grinding

Bonemeal for bone china

Meat and bone meal

Animal feedstuffs

Calcium phosphate for animal feedstuffs

Usages ranging from plastics to pharmaceuticals

Human food and high grade soaps

Edible cooking fat

Note: Livestock may be cattle, sheep, pig or poultry. Fellmongerer's separate wool and hair from the skin and hide of livestock carcasses.
In other cases, the materials are recycled because they will contribute to the increased productivity of future crops and/or livestock; examples are manure from livestock systems, and blood and bone meal from rendering processes.

At a more detailed level, Figure 4 shows the typical co-products from a “generic” livestock carcass. More than 20 different products may be produced from an individual animal, only some of which are food products.

A further allocation issue arises with respect to crop rotations, as introduced in Section 4.2. Crop rotations can be viewed as a particular example of the allocation problem in LCA because their existence raises the issue of how to allocate environmental burdens between crops and livestock in a rotation (unless this is avoided by whole system modelling as outlined in Section 4.2).

For crops, the allocation problem is greatly reduced if soil is treated as recommended in Section 4.3. In other words, if the soil is regarded as an ancillary in an LCA study, and changes in its quality are assessed between the beginning and end of the study, straw and plant residues incorporated into the soil are not co-products of the system. Instead, their impacts on soil quality are assessed as part of the study, and all other impacts of agricultural production are “allocated” to the harvested product(s). However, allocation still remains an issue for crops producing more than one co-product (for example, cereals producing grain and baled straw, and oilseeds producing oil and meal).

For livestock systems, the allocation problem is not easily solved because livestock are typically fed compound feedstuffs (i.e. processed feed consisting of many co-products from different crops as shown in Table 2), and produce multiple co-products (as shown in Figure 4). The only systems where allocation is not an issue are ones where the animals feed on grass or forage from uncultivated systems, and produce only one product. Examples include livestock reared on grass solely for meat (or hide) production, and wild gamebirds.

In most LCAs involving agricultural production to date, the allocation issues raised by co-production have been solved by allocation according to economic value, probably because it is an allocation method that requires minimal collection of additional data (see, for example, Ceuterick and Spirinckx, 1997; Møller et al., 1996; Terwoert et al., 1996; and Weidema et al., 1995). System extension was

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6 Weidema et al. (1995, p.38 and 83) note that the economic values of the co-products must be calculated at the earliest point at which they can be regarded as separate products, i.e. as their selling value minus any processing
used by the German Federal Environment Agency in their study of biodiesel (Friedrick et al., 1993): they accounted for the by-products rapeseed meal and glycerine during biodiesel production by subtracting the "avoided burdens" for production of alternative livestock feed and glycerine from the system under analysis. A few studies have used physical composition as a basis for allocation. For example, Teulon (1996) used sugar content to allocate burdens between syrup and pulp produced from sugar beet, and fat content for allocation between cheese and whey produced from curdling milk. The importance of choice of allocation method has been illustrated for a number of cases by costs prior to sale but after co-production. For example, for wheat grain and straw the economic value of the grain is its selling price minus the drying costs.

Table 2. Percentage Composition of Some Compound Feedstuffs

<table>
<thead>
<tr>
<th>Category of Feedstuff</th>
<th>Feedstuff</th>
<th>Standard</th>
<th>Pig Rearer</th>
<th>Broiler Starter</th>
<th>Poultry Layer</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cereals</td>
<td>Wheat</td>
<td>4</td>
<td>36</td>
<td>47</td>
<td>49</td>
</tr>
<tr>
<td></td>
<td>Barley</td>
<td>3</td>
<td>15</td>
<td>12</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Totals</td>
<td>7</td>
<td>51</td>
<td>59</td>
<td>52</td>
</tr>
<tr>
<td>Cereal by-products</td>
<td>Wheatfeed/other miller’s offals</td>
<td>11</td>
<td>7</td>
<td>-</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Extracted rice bran</td>
<td>5</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Confectionery/biscuit meal</td>
<td>1</td>
<td>1</td>
<td>-</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Maize gluten</td>
<td>12</td>
<td>-</td>
<td>-</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Maize germ</td>
<td>-</td>
<td>1</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Dried grains/grain screenings</td>
<td>1</td>
<td>-</td>
<td>-</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>Nutritionally improved straw</td>
<td>3</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Other cereal by-products</td>
<td>3</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Totals</td>
<td>35</td>
<td>9</td>
<td>1</td>
<td>11</td>
</tr>
<tr>
<td>Vegetable proteins</td>
<td>Soyabean meal</td>
<td>3</td>
<td>23</td>
<td>22</td>
<td>10</td>
</tr>
<tr>
<td></td>
<td>Sunflowerseed meal</td>
<td>9</td>
<td>-</td>
<td>-</td>
<td>8</td>
</tr>
<tr>
<td></td>
<td>Rapeseed meal</td>
<td>16</td>
<td>2</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Palm kernel</td>
<td>8</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Citrus pulp</td>
<td>1</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Peas</td>
<td>-</td>
<td>1</td>
<td>2</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Beans</td>
<td>2</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Other vegetable proteins</td>
<td>3</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Totals</td>
<td>42</td>
<td>26</td>
<td>26</td>
<td>20</td>
</tr>
<tr>
<td>Animal proteins</td>
<td>Herring meal</td>
<td>-</td>
<td>2</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Other fish meal</td>
<td>-</td>
<td>2</td>
<td>5</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Meat and bone meal</td>
<td>-</td>
<td>1</td>
<td>3</td>
<td>4</td>
</tr>
<tr>
<td></td>
<td>Other animal proteins</td>
<td>-</td>
<td>2</td>
<td>-</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Totals</td>
<td>-</td>
<td>7</td>
<td>8</td>
<td>5</td>
</tr>
<tr>
<td>Miscellaneous</td>
<td>Molasses</td>
<td>8</td>
<td>2</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Sugar beet pulp</td>
<td>1</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Limestone</td>
<td>2</td>
<td>-</td>
<td>-</td>
<td>9</td>
</tr>
<tr>
<td></td>
<td>Other minerals and vitamins</td>
<td>1</td>
<td>1</td>
<td>2</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Oils and fats</td>
<td>1</td>
<td>2</td>
<td>3</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>Other miscellaneous</td>
<td>1</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Totals</td>
<td>15</td>
<td>6</td>
<td>5</td>
<td>12</td>
</tr>
</tbody>
</table>

N.B. Columns may not add to 100 because of rounding up of values.
comparing the differences in allocation of burdens between co-products using different methods, as shown in Table 3. It can be seen that choice of allocation method can make a big difference to the results; this is most pronounced for the cheese study where allocation on the basis of economic value or fat content gives results more than five times greater than allocation on the basis of mass!

Table 3. Impact of Allocation Method On Results for LCA Studies

<table>
<thead>
<tr>
<th>Study</th>
<th>Co-Products</th>
<th>Allocation Method</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Mass</td>
</tr>
<tr>
<td>Ceuterick and Spirinckx, 1997</td>
<td>Rapeseed: straw</td>
<td>45.9</td>
</tr>
<tr>
<td></td>
<td>Oil: cake</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Biodiesel: glycerine</td>
<td>-</td>
</tr>
<tr>
<td>Maillefer, 1996 (soybean oil study)</td>
<td>Oil: cake</td>
<td>18.0</td>
</tr>
<tr>
<td>Teulon, 1996 (sugar study)</td>
<td>Syrup: pulp</td>
<td>83.0</td>
</tr>
<tr>
<td>Teulon, 1996 (cheese study)</td>
<td>Curdled milk: whey</td>
<td>15.0</td>
</tr>
</tbody>
</table>

Note: all values in table are percentage of environmental burdens allocated to the first-named co-product in the column titled “Co-Products.”

For use of waste/near-to-waste materials, the main focus of attention has been use of manure as a fertiliser in crop production. In one study, economic allocation was used as a basis for allocation between the previous livestock system and the system in which the manure was used (Weidema et al., 1995, p.109-13). In two other studies, the environmental burdens of manure were allocated 100% to the livestock system but those associated with its use were allocated 100% to the system in which it was used (Ceuterick and Spirinckx, 1996, p.66; Wegener Sleeswijk, 1993)

For crop rotations, a number of aspects have been considered in existing studies:

- **Nutrients in the soil**

  The role of nutrients in crop rotations has been considered by a number of researchers (Weidema et al., 1995, p.83; van Zeijts et al., 1996). Van Zeijts et al. (1996) point out that farmers generally apply nitrogen fertilisers separately to each crop; therefore generally the environmental burdens of nitrogen fertiliser production and use can be allocated to the crop to which the fertiliser is applied. However, for phosphorus and potassium the situation is different: surplus applied phosphorus and potassium may remain in the soil and be used by future crops. For example, in the Netherlands farmers on clay soils often do not use phosphorus fertilisers on winter wheat because the crop uses phosphorus applied to previous crops. The general recommendation is that the environmental burdens of fertiliser production and use should be allocated to crops according to the
recommended quantity for each crop if it is considered alone without knowing the previous crop (Audsley et al., 1997, p.25).

- **Green manure**
  Van Zeijts et al. (1996) suggest that the environmental burdens of cultivating green manure should be allocated among all crops in a rotation.

- **Organic matter**
  Van Zeijts et al. (1996) suggest that the environmental burdens of incorporating organic matter should be allocated among all crops in a rotation.

- **Fallow land**
  Gaillard (1996) identifies fallow periods between main crops as another point at which allocation is an issue because significant environmental impacts may occur during this time (such as soil loss and nitrate leaching). Audsley et al. (1997, p.47) divide environmental burdens occurring during this time equally between the preceding and subsequent crops.

These examples drawn from the literature show the variety of approaches that have been used to address allocation issues in LCAs involving agricultural production over the last few years. More recently, a hierarchy of approaches to allocation has been published along with guidelines on its operation (Audsley et al., 1997, p.18-19), and this is now suggested as the methodological standard for LCAs involving agricultural production. It follows the hierarchy developed for other LCA studies as outlined in Chapter II, Section 3.

### 5. Impact Assessment Issues Specific To Agriculture

In the SETAC "Code of Practice" the primary objectives of LCA are stated as (SETAC, 1993, p.5):

- To provide as complete a picture as possible of the interactions of an activity with the environment.
- To contribute to understanding of the overall and interdependent nature of the environmental consequences of human activities.
- To provide decision-makers with information which defines the environmental effects of these activities and identifies opportunities for environmental improvements.

As part of these objectives, the role of the Impact Assessment phase of LCA is to characterise and assess the effects of the environmental burdens identified in the Inventory phase of a study (as
discussed in Chapter II). However, the Impact Assessment categories defined and commonly used in existing LCA studies reflect the focus of LCA methodological development on industrial systems and their characteristic environmental impacts. This means that application of current LCA methodology to systems involving agricultural production gives results that do not reflect the primary objectives of LCA as listed above. It is therefore important to recognise gaps in existing Impact Assessment methodology and either account for the additional impacts caused by agricultural systems or “flag up” these omissions from the analysis. [Flagging up omissions means noting them in the final report of an LCA study, even though they may not be part of the analysis.] The gaps concern assessment of impacts related to:

- Ecosystems and biodiversity
- Soil
- Landscape degradation
- Use of solar energy
- Use of water
- Animal welfare.

In this thesis, I develop Impact Assessment methodologies for ecosystems and biodiversity (Chapter IV) and soil (Chapter V). As discussed in Chapter II (Section 4.2.3), landscape degradation may not be appropriate for inclusion in LCA but should be flagged up in a study. Other gaps are discussed in more detail below.

5.1 Use of Solar Energy

Use of non-renewable sources of energy, i.e. fossil fuels, is recognised as an important resource depletion issue in LCA. However, no attention has been given to use of the energy source from which almost all forms of energy - renewable and non-renewable - are ultimately derived: solar energy. This energy source can be regarded as a limited resource because its availability at the Earth’s surface cannot be increased, and it is used in a large number of “services” (ranging from natural lighting to space heating to electricity generation using photovoltaic cells). It therefore seems reasonable to assess use of this resource in agricultural systems because inefficient use implies wastage of a limited resource.

Use of solar energy can be assessed fairly simply in LCAs of agricultural systems by considering the proportion of total incident radiation reaching farmed land that is incorporated into the harvested
crop\textsuperscript{7}. Using this approach, use of solar energy by an agricultural system is calculated as the inverse of the proportion of incident solar radiation assimilated by the harvested crop:

\[
\text{Use of solar energy} = \frac{ARD}{G}
\]

where
- \( A \) = area required by functional unit (m\(^2\))
- \( R \) = incident radiation in this area (MJ/m\(^2\)/day)\textsuperscript{8}
- \( D \) = number of days crop is cultivated (i.e. time period for system under analysis)
- \( G \) = gross energy in harvested crop (MJ).

It is worth noting that use of this method implies that, all other things being equal, the harvested crop yield per hectare must be higher, or the cultivation period shorter, for areas with higher incident radiation to score equal to areas with lower incident radiation (Cowell and Clift, 1998). If this is not the case, it implies that areas with higher incident radiation are making less effective use of their solar radiation. In other words, site-dependent assessment forms an integral part of this method.

5.2 Use of Water

There has been debate about whether use of water is a relevant resource depletion issue in LCA (see Chapter II, Section 4.2.1). Elsewhere, we have suggested that a possible approach is to calculate the area over which average annual rainfall must fall in order to equal the quantity of groundwater used in the system under analysis (Cowell et al., 1996). An improved methodology is to assess total water use in the system under analysis (apart from rain that falls on the land in the system), i.e. surface- and ground-water rather than just groundwater (Cowell and Clift, 1998). This is because any requirement for additional water in a system means that water is a limited resource in that location whether it is

\textsuperscript{7} One possible criticism of this method is that it does not account for the role of solar energy in: i) wider hydrological cycles; ii) determining weather patterns (through wind formation); or iii) maintaining the temperature of the soil, water and atmosphere. It could be argued that these additional roles of incident radiation in other geographical areas make an equally important, albeit indirect, contribution to agricultural production in the system under analysis, and this should be recognised in assessment. Instead, this method of analysis implicitly “rewards” increased incorporation of solar energy into plant biomass (by lowering the IA result for use of solar energy in systems with higher yields per hectare). However, although this is true, it is very unlikely that increased incorporation of solar energy into agricultural products would have any noticeable effect on these other functions because this “sink” typically represents such a small proportion of total incident solar energy. Photosynthetically active radiation (PAR) is 44\% of total incident radiation and, typically, between 0.01 and 3\% of PAR is incorporated into above-ground Net Primary Productivity (NPP) (Begon et al., 1996, p.720). Crop plants under ideal conditions can incorporate up to 10\% of PAR into above-ground NPP (Cooper, 1975), i.e. a maximum of 4.4\% but more typically less than 1.4\% of total incident radiation.

\textsuperscript{8} Oliver and Jackson (1998) quote incident radiation values varying from 800 kWh/m\(^2\)/year in Iceland, to 1000 kWh/m\(^2\)/year in the UK (average), to 2500 kWh/m\(^2\)/year in small parts of California, Chile, Peru and Niger.
above or below the ground, and the extent to which it is "limited" is defined by local rainfall patterns. One approach to assessing water use is then:

\[ \text{Use of water (1)} = \frac{W}{R} \]

where \( W \) = total quantity of water used in system excluding rainfall (m\(^3\))

\( R \) = average annual rainfall per m\(^2\) in area under cultivation (m\(^3\)/m\(^2\)/year).

In effect, this calculation gives the additional land area (in m\(^2\)-years) required to supply the water used by the system under analysis, and is therefore similar to the Ecological Footprints approach introduced in Chapter I, Section 3.

An alternative approach is to assess water use as:

\[ \text{Use of water (2)} = \frac{W}{A} \]

where \( W \) = total quantity of water used in system excluding rainfall (m\(^3\))

\( A \) = average annual rainfall in total area under cultivation in system under analysis (m\(^3\)/year in area under cultivation).

This calculation gives the number of years in the area under analysis required to supply the water used by the system. Hence it demonstrates application of the idea in Chapter II, Section 4.2.1, of assessing renewable resources in relation to the time required for renewing each resource. The assessment method chosen for any one study is a matter of personal preference; however, both methods demonstrate promising alternative approaches to assessment of resource use that are worth further research attention.

In both these approaches, I do not distinguish between use of surface- and ground-water because such a distinction is not relevant from a resource depletion perspective. Also, as for use of solar energy, it is worth noting that site-dependent assessment forms an integral part of this method (since assessment of water use depends upon the average annual rainfall in the area under cultivation). This is important because the extent to which water can be regarded as a limited resource is defined by the location of use.
Concerning use of water as a disturbance issue (Chapter II, Section 4.2.3), any desiccation impacts within the area under cultivation are automatically assessed under Physical Habitat Depletion (PHD) (see Chapter IV). Theoretically, impacts of water abstracted elsewhere for use in the system under analysis can also be assessed under the PHD Impact Assessment category. For example, the physical habitat impacts of water abstracted from a lake for use elsewhere can be assessed by considering physical habitat changes in the lake due to water abstraction from the time at which abstraction began until the present using an appropriate Physical Habitat Index. The resulting PHD value (in m²-years) is then multiplied by the quantity of water used in the system divided by the total quantity of water abstracted in the time period. This gives a PHD value that can be added to the PHD value for the area under cultivation. However, in many cases it will not be possible to obtain relevant data to make the analysis. For example, for water abstracted from a river the desiccation impacts may be diverse and dispersed over large areas (ranging from a reduced volume of freshwater habitat to physical changes in river banks and wetlands). In these cases, the non-quantifiable impacts of water use should be flagged up in a study. Furthermore, it is worth noting that this part of the analysis is likely to make an insignificant contribution to the PHD value compared with the area under cultivation apart from in exceptional circumstances (i.e. abstraction of large quantities of water from a small or particularly vulnerable ecosystem).

5.3 Animal Welfare

Animal welfare is commonly perceived as a legitimate issue for consideration in environmental management. Opinions differ about whether it should be included in LCA. It may be argued that welfare of animals is an anthropocentric issue and that there is no reliable way of determining the actual welfare of animals under different conditions. Therefore this is not an issue amenable to analysis in LCA. Another argument can be made from a systems analysis perspective. Using this perspective, it can be argued that farm animals are part of the system under analysis. They do not cross the system boundary in or out of the system, apart from as meat, hide and other animal products. Therefore, their state of welfare within the system is not a relevant consideration because only system components crossing the system boundary are assessed in an LCA analysis (see Chapter II). A third line of argument recognises the legitimacy of including animal welfare as a factor for consideration in decision-making but advocates its inclusion as part of the wider LCA-based decision-making process within which quantitative, systems-based LCA is located as an environmental management tool. This requires a modification in operationalisation of LCA, and is discussed in more detail in Chapter VII.
6. Conclusions

The research area concerned with application of LCA to the assessment of agricultural systems has developed over the last few years, and now involves researchers from most European countries (Table 1). LCA has a potentially valuable role to play in assessment of systems involving agricultural production by identifying the trade-offs, for example, between use of non-renewable and renewable materials, and comparing different agricultural systems delivering the same product or service. The conflicting quotes at the beginning of this chapter provide ample proof of the need for such an approach. However, a number of methodological issues must still be addressed if LCA is to assess the comprehensive environmental impacts of agricultural systems, as otherwise it may give misleading results. It is important to resolve these methodological issues so that the models constructed in LCA give results as representative as possible of reality, and ones that can guide decision-makers towards more sustainable solutions.

The main methodological issues requiring attention have been discussed in this chapter, and a new approaches have been suggested for assessing use of solar energy and water (Section 5). In Chapters IV and V, I develop approaches for assessment of biodiversity, and soil quantity and quality in LCA. In Chapter VI, I describe a study of wheat production from three different farming systems which shows how the issues identified in this chapter can be addressed in LCA studies.

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CHAPTER IV

BIODIVERSITY AND LIFE CYCLE ASSESSMENT: OPTIONS FOR INTEGRATION

"Noting that, where there is a threat of significant reduction or loss of biological diversity, lack of full scientific certainty should not be used as a reason for postponing measures to avoid or minimise such a threat” (Preamble, United Nations Convention on Biological Diversity)

1. Introduction

Biodiversity has been described as:

The variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems (UNEP, 1992).

In other words, biodiversity is concerned with genetic variability within species, numbers of different species, and existence of different ecosystems. Within these broad areas, biodiversity can be further considered at a number of different scales as shown in Table 1. This Table shows that genetic diversity can be measured from the level of populations through individuals, to chromosomes and down to nucleotides. Species diversity can be measured at the level of kingdoms of different organisms, phyla, classes, and so on down to individual species. Ecological diversity exists from the level of biomes down to individual communities within ecosystems. [A glossary of terms and acronyms used in this chapter can be found in Appendix IV.1.]

For the human species, biodiversity is important for a range of reasons (Lovejoy, 1995). From a scientific perspective, biodiversity provides a reservoir of “genetic insurance” to guard against future changes: a greater diversity of species and individuals means that at least some are likely to possess characteristics that facilitate their survival under changed conditions. Also, biodiversity of ecosystems, and of species and individuals within ecosystems, is necessary for their continued

1 Perfect (1991) provides an interesting analysis of this aspect. He describes the paradox that exists between natural selection leading to dominance of a restricted number of species that are well adapted to their
functioning. Although the actual "quantity" of biodiversity required in an ecosystem is a subject of
debate, almost all ecologists argue that the current loss of biodiversity should be a subject of concern
precisely because we cannot predict the exact point at which ecosystem functioning is compromised
as a result of decreased biodiversity. From a human economy perspective, many of the materials
contributing to the economy are obtained from organisms, ranging from building materials to fabrics
to foods to fuels to medicines. A diversity of sources for these materials helps to ensure their
continued availability in changing conditions (as mentioned above), and conserves the opportunity to
develop new alternatives. Additionally, eco-tourism is a growing industry, and economic rewards
can be gained by countries conserving their wildlife and ecosystems for the benefit of tourists.
Finally, from an ethical perspective, it can be argued that other life-forms have intrinsic value and
deserve protection from destruction.

Table 1. Components of Biodiversity

<table>
<thead>
<tr>
<th>Ecological Diversity</th>
<th>Species Diversity</th>
<th>Genetic Diversity</th>
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</thead>
<tbody>
<tr>
<td>Biomes</td>
<td>Kingdoms</td>
<td>Populations</td>
</tr>
<tr>
<td>(Ecosystems)</td>
<td>Phyla</td>
<td>Individuals</td>
</tr>
<tr>
<td>Communities</td>
<td>Classes</td>
<td>Chromosomes</td>
</tr>
<tr>
<td>(Habitats)</td>
<td>Orders</td>
<td>Genes</td>
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<tr>
<td></td>
<td>Families</td>
<td>Nucleotides</td>
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<td>Genera</td>
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<tr>
<td></td>
<td>Species</td>
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<td></td>
<td>Subspecies</td>
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</tr>
<tr>
<td></td>
<td>Populations</td>
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</tbody>
</table>

Source: adapted from Heywood, 1996.
Note: Ecosystems and habitats are in brackets because they describe the non-living components as well as living
organisms found in different environments. Ecologists suggest that, instead, the term "community" should be
used to describe living organisms from the level of ecosystems down to habitats (Begon et al., 1996). However,
the term "ecosystem" is still widely used to describe assemblages of living organisms, and was used in the
definition of biodiversity in the Convention on Biological Diversity. Therefore it has been retained in this
chapter to describe the characteristic assemblages of living organisms found in forests, grasslands and so on.

However, there is increasing concern that biodiversity is being depleted at all levels due to human
activities that range from the implementation of EU and national Seed Registers for food and other
crops, to pollution, to large-scale destruction of ecosystems such as tropical rainforests. Although it

environment, and the need of these species to retain genetic insurance (i.e. genetic diversity) against future
changes in their environment.

2 Again, Perfect (1991) provides an alternative argument, suggesting that humans are fast becoming able to meet
their requirements by genetic engineering and that this may be a more realistic approach than "a random search
through the Amazon Basin." However, many would regard this view of genetic engineering as rather optimistic.
is impossible to measure actual depletion rates (see below), it is estimated that species are becoming extinct at rates hundreds or thousands of times the estimated average background extinction rate (i.e. the rate at which species become extinct in the absence of human-induced changes) (Heywood, 1996). Indeed, in the UK it is estimated that over 100 species have been lost this century including 7% of dragonflies, 5% of butterflies and more than 2% of fish and mammals (UK Biodiversity Steering Group, 1995a, p.5).

These concerns have led to international agreements such as the Bern Convention on the Conservation of European Wildlife and Natural Habitats, the EC’s Habitats and Species Directive, the Bonn Convention on the Conservation of Migratory Species of Wild Animals, the Convention on International Trade in Endangered Species (CITES), and the Convention on Biological Diversity. In particular, the Convention on Biological Diversity, agreed at the 1992 Earth Summit in Rio de Janeiro, has stimulated a number of actions at national level to conserve biodiversity. In the UK, a Biodiversity Action Plan Steering Group was set up and published two reports in December 1995 (UK Biodiversity Steering Group, 1995a, 1995b). The reports list 1,250 species of conservation concern, and state that conservation plans should be developed for about 300 of these species plus 38 key habitats.

Given the increasing recognition of biodiversity depletion as an issue of concern, it therefore seems reasonable to consider how - and whether - it should be assessed in LCA methodology. LCA aims to assess environmental impacts in the areas of ecological health, human health and resource depletion (Consoli et al., 1993, p.5). Although the exact nature of the impacts relevant to these areas is open to debate, generally most people would agree that ecological health is intrinsically related to the biodiversity of ecosystems: an ecosystem cannot be regarded as ecologically healthy unless it maintains an appropriate level of biodiversity. Therefore assessment of biodiversity should be included in LCA methodology. The question of how it should be included is the subject of the rest of this chapter. Throughout the chapter, the approach developed is based on the following premises:

1. Assessment in LCA requires valuation of different ecosystems in a system under analysis for their relative contributions to global biodiversity. This is achieved by development of Physical Habitat Factors (PHFs) for different ecosystems (Section 4). The method is analogous to development of

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3 “Physical Habitat” is used rather than “Physical Ecosystem” because the term “habitat” describes the place where organisms live whereas the term “ecosystem” also includes all the organisms in the ecosystem in its definition. The distinction is rather blurred, since the habitat of an organism may include other organisms; for example, the habitat of an insect species might be an oak tree. However, use of the term “habitat” puts the emphasis upon the environment surrounding the organisms of interest in an analysis. Therefore, since we are
Global Warming Potentials, Acidification Potentials, and so on for other Impact Assessment categories. To calculate Physical Habitat Depletion for any system, the land areas used in the system are multiplied by a combination of the relevant PHFs (see Section 7); this is analogous to calculating Global Warming, Acidification, and so on by multiplying the emissions of substances from any system by the relevant Impact Assessment factors.

2. A low value in the Physical Habitat Depletion (PHD) category for a system indicates that the particular physical habitats supported by existence of that system are more beneficial for global biodiversity than another system with a higher PHD value. This is analogous to low values in the other Impact Assessment categories being more beneficial than higher values from an environmental perspective. The implication is that the system can be improved by:
   a) Reducing use of land area (because any surplus land can then be used to support physical habitats that enhance global biodiversity), and/or
   b) Changing the type of physical habitat in the system under analysis, i.e. replacing the ecosystem in the system by others that make a greater contribution to global biodiversity.
   This is analogous to, in other Impact Assessment categories, a) reducing emissions, and/or b) changing the type of emissions from the system under analysis to those with less environmental impacts.

3. PHFs can be defined for ecosystems at different levels of detail. In some studies, generic ecosystem classes are appropriate (for example, tropical rainforest, temperate deciduous forest, and boreal forest). In other studies, more detailed classes may be appropriate (for example, different types of temperate deciduous forest). The level of detail to be used in a study depends upon the purpose and scope of the study. Section 5 discusses assessment at these different levels of detail.

4. In each type of ecosystem, human management may have a significant effect on biodiversity. However, differences in biodiversity within an ecosystem due to varying management practices have less impact on biodiversity than differences between ecosystems. Therefore, Physical Habitat Depletion can be measured at the level of differences between ecosystems, using PHFs, or at the level of differences between management practices within any one ecosystem. In the latter case, Physical Management Factors (PMFs) rather than PHFs are used to calculate Physical Habitat Depletion. This is discussed in Section 6.

_4 An alternative approach is to include the "surplus land" in the analysis so that the actual physical habitat supported on that land is assessed as part of the analysis.
5. As well as changes between ecosystems, or management practices within any one ecosystem, assessment in LCA should also account for conservation of existing ecosystems due to human activities. Incorporation of this aspect is discussed in Section 7.

Firstly, however, I begin by outlining current approaches to measurement of biodiversity (Section 2), and examine the extent to which current LCA methodology accounts for biodiversity (Section 3). This provides the justification for development of the PHFs (Sections 4 and 5), PMFs (Section 6) and their application to assessment of Physical Habitat Depletion (Section 7). Use of the PHFs and PMFs is discussed in Section 8. In the final section (Section 9), it is concluded that this method provides a flexible and practical approach to assessment of physical habitat degradation, but requires further research effort for its implementation. However, it also highlights a number of conceptual issues that are not addressed, and which can be construed as valid criticisms of the LCA approach.

2. Theoretical Measurement of Biodiversity

The definition of biodiversity given in Section 1 shows that measurement of biodiversity is a complex task, and it is currently the subject of considerable research attention (see, for example, Hawksworth, 1995, Reaka-Kudla et al., 1997, and Mooney et al., 1996).

For biodiversity at the level of ecosystems (ecological diversity), different groupings of organisms can be described at increasing levels of detail throughout the world. At the most general level, a restricted number of biomes (see Table 1) can be described that may actually bear little resemblance to the ecosystems found in these areas. For example, Begon et al. (1996, p.29) define eight terrestrial and two aquatic biomes. According to this classification, the UK falls within the “temperate forest” biome. However, obviously there is relatively little temperate forest in the UK today, although this would be the climax vegetation throughout much of the country in the absence of human activities.

A more realistic picture of the UK’s land surface can be described using ecological categories at a greater level of detail. An example is the Natural Vegetation Classification in the UK which distinguishes between ecosystems at a relatively detailed level of analysis. For example, it differentiates between deciduous woodlands characterised by the presence of different key species such as:
• *Alnus glutinosa, Fraxinus excelsior, Lysimachia nemorum* (common alder, common ash, yellow pimpernel) woodland
• *Fraxinus excelsior, Acer campestre, Mercurialis perennis* (common ash, maple, Dog’s mercury) woodland
• *Fagus sylvatica, Mercurialis perennis* (common beech, Dog’s mercury) woodland
• *Quercus petraea, Betula pubescens, Oxalis acetosella* (sessile oak, downy birch, wood sorrel) woodland.

The assumption behind this categorisation is that typical assemblages of species in communities can be identified by the presence of key species.

Most attention has been devoted to assessing biodiversity at the level of numbers of species. However, although estimates of the total number of species on the Earth range from 7 to 20 million, only 1.75 million species have been described scientifically representing just 13% of the estimated total number (Heywood, 1996). Moreover, there is no comprehensive listing of these 1.75 million described species. Indeed, it has been said that species are more likely to become extinct than to be named by taxonomists (Jermy *et al.*, 1995, p.13)! Nevertheless, existing studies of the numbers of species of particular groups of organisms can be used as indicators of overall biodiversity in different areas. For example, the UK Countryside Survey 1990\(^6\) showed that arable fields have an average of 4.8 plant species and upland grasslands an average of 23.4 plant species per plot (see Figure 1), indicating that overall biodiversity is likely to be higher in upland grasslands than arable fields.

For genetic diversity, it can be inferred from the lack of data on species that there is even less data on measurement of genetic diversity. In theory, differences in the DNA of individuals in the same species or in different species can be used to assess the genetic diversity within and between species. A small number of studies exist that demonstrate the approach (see, for example, Templeton, 1995, and O’Donnell *et al.*, 1995)\(^7\). A complementary approach is phylogenetics, where variation within

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\(^5\) Arctic tundra, Northern coniferous forest, temperate forest, tropical rainforest, tropical seasonal forest, temperate grassland, tropical savannah/grassland/scrub, Mediterranean vegetation/chaparral, desert, mountains, freshwater and marine biomes.

\(^6\) The UK Countryside Survey 1990 was undertaken by the Institute of Terrestrial Ecology (ITE) and the Institute of Freshwater Ecology. It collected comprehensive data on land cover, habitats and species, using an extensive field study of different habitats and satellite mapping. The results are presented in a Main Report (Barr *et al.*, 1993) and a Summary Report (DOE, 1993).

\(^7\) For example, one bacterial “species” (*Legionella pneumophila*) has nucleotide sequence homologies of less than 50%. This is as large as the characteristic genetic distance between mammals and fishes (May, 1995, p.15).
(and between) species in features or attributes is used to quantify and estimate biodiversity (Faith, 1995). The assumption behind this approach is that features and attributes are determined by the genetic coding in individuals and species, and therefore they can be used as proxies for genetic diversity. However, again there are very few data available that use this approach to assess biodiversity within and between species.

Comparing the three levels (ecosystem, species and genetic) for their usefulness in measuring impacts on biodiversity in specific areas due to different human activities, the advantage of measuring biodiversity at the level of ecosystems is that it is relatively easy to undertake, particularly with advances in satellite imaging systems. Its disadvantage is that it classifies areas on the basis of key vegetation features. As a result, it cannot account for the presence or absence of additional species (and individuals) that may or may not be present within that type of vegetation. Furthermore, it might be assumed that equal distribution of land area among different ecosystems will maximise overall biodiversity. However, this is not necessarily the case as illustrated by the following example. Consider a hypothetical world containing only three ecosystems: tropical rainforest, intensively cultivated farmland, and urban land. The latter two ecosystems have much lower numbers of species and individuals than the tropical rainforest. Therefore, maximum biodiversity will not be achieved by an equal distribution of land among the three ecosystems. Instead, the best distribution (from a biodiversity perspective) is likely to be achieved by weighting the land areas in favour of tropical rainforest. Therefore, measuring biodiversity by land area of different ecosystems requires additional judgements about the "best" distribution of total land area among different ecosystems, i.e. judgements informed by the contributions made by number of species and number of individuals in each ecosystem to global biodiversity.
Measurement of biodiversity by counting **numbers of species** is problematic because of a lack of data, and the time (and financial) implications of conducting such studies. Furthermore, even if such data are available, there are a number of other complications. Firstly, this type of assessment does not account for the rarity of different species. For example, one can imagine a theoretical case in which two sites are compared for their biodiversity. One site contains two species of *Ranunculus* (buttercup) and the other one contains one species of *Ranunculus* and *Cypripedium calceolus* (Lady's-slipper orchid), a critically endangered species in Europe. Any measurement that describes these two sites as equal in terms of their biodiversity seems inadequate. Secondly, genetic diversity is ignored when counting numbers of species and therefore closely related species are given the same weighting in the assessment as other species that are not closely related. For example, consider a case in which four sites each have two species: one is a species of *Ranunculus*, and the other is either another species of *Ranunculus* from the same genus, a rabbit, or a protozoan of the genus *Amoeba*. Again, any measurement that describes all these sites as equal in terms of their biodiversity is not very informative (Harper and Hawksworth, 1995, p.7). Thirdly, this approach does not account for genetic diversity within each species.

Theoretically, measurement of biodiversity at the level of **genetic diversity** makes sense because it accounts for biodiversity at its most detailed unit of analysis, hence it incorporates species- and ecosystem-level biodiversity. However, as noted above, its practical implementation seems unlikely given the enormous research effort required for its operationalisation. This is amply illustrated by considering soil where one gram may contain $10^9$ organisms and $10^4$ species (O'Donnell et al., 1995), and mammals where a typical animal has about 100,000 genes comprising in the order of four billion nucleotide pairs (Wilson, 1988).

In summary, then, there are shortcomings with all the methods for measuring biodiversity outlined above:

- **Areas of different ecosystems**: maximising the land area of all ecosystems does not necessarily maximise global biodiversity because different ecosystems have varying numbers of species and individuals.

- **Number of species**: maximising the number of species in any one area does not necessarily maximise global biodiversity because these species may all be common throughout the world, or may be closely related and therefore show low genetic diversity.

- **Genetic diversity**: although theoretically a good method, implementation is impractical.
There is a further aspect of measuring biodiversity that also requires consideration at all three levels of assessment. This concerns the value to humans of different ecosystems, species, and/or attributes and features of organisms. Indeed, many of the arguments for conservation of biodiversity are made on the basis that threatened ecosystems and species are, or may be, “useful” to human societies (where “useful” may be defined using economic, health and/or aesthetic justifications, as outlined above). Should this be recognised in measuring biodiversity? For example, should conservation of “furry animals” be given a higher priority than conservation of endangered insect species or even the smallpox virus? This is a question that cannot be answered by scientific analysis because it is concerned with value judgements.

All these considerations suggest that objective measurement of biodiversity is impractical at the present time – and may not even be desirable if one acknowledges the role of human values in assessing the relative importance of different ecosystems and species. However, since LCA attempts to be more objective rather than subjective in assessing different impacts associated with a system under analysis8, in this chapter I examine the feasibility of objective assessment (as opposed to measurement) of biodiversity. In other words, I do not include a role for value judgements in prioritising initiatives to conserve selected species and/or ecosystems (but see Section 9 below). The approach developed is based on defining appropriate indicators of biodiversity. However, firstly I discuss existing methods of accounting for biodiversity in LCA.

3. LCA Methodology and Biodiversity: Current Status

Section 1 highlighted the relevance of assessing biodiversity in LCAs. It is therefore appropriate to ask if current LCA methodology does account for biodiversity, and - if not - how it can be developed to take account of this aspect.

Impacts on biodiversity due to human activities may occur by three routes. The first route is chemical changes in the environment due to, for example, releases of air and water pollutants or addition of nutrients to soils and water bodies. The second route is non-chemical changes in the environment

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8 Theoretically, LCA methodology consists of both objective and subjective parts. The objective parts are the Inventory Analysis, and Classification and Characterisation during the Impact Assessment phase. The subjective parts are the Goal Definition and Scoping phase, relative weighting of the different Impact Assessment categories, and the Improvement phase. In reality, this distinction is blurred because subjective decisions are taken about which burdens are quantified during Inventory Analysis, and which methodology and Impact Assessment factors are used during Impact Assessment. However, I retain the distinction in development of methodology for the PHD category for consistency with the rest of LCA methodology.
caused by factors such as noise, heat, radiation and smell. The third route is physical change in the environment (sometimes called habitat change), in other words change in ecosystems due to removal or addition of vegetation, livestock, landscape features (such as ponds and ditches), and so on. Human activities may affect biodiversity by one, two or all three routes. For example, building a power station on agricultural land will lead to: i) chemical changes in the environment due to release of air and water pollutants; ii) non-chemical changes in the environment due to, for example, heat pollution if cooling water is released into a lake or river; and iii) physical changes in the environment due to replacement of agricultural ecosystems by buildings and other industrial infrastructure. However, physical changes in ecosystems due to human activities are considered to make the greatest single contribution to impacts on biodiversity\(^9\).

For chemical changes, existing LCA impact categories such as Acidification, Eutrophication and Ecotoxicity do attempt to account for impacts on biodiversity, and these impact categories can be described as lower order measures of biodiversity (Fava et al., 1992). The same applies for non-chemical changes, although the LCA impact categories accounting for these changes (such as Noise, Waste Heat, Radiation, Odour) have not received much research attention (see Chapter II). For physical changes, a number of assessment methodologies have been proposed. They are discussed below in two sections: Section 3.1 discusses approaches for assessing the value of different ecosystems, and Section 3.2 discusses approaches to using these values in LCA. In Section 3.3, I evaluate the usefulness of these different approaches.

3.1 Approaches for Assessing the Value of Different Ecosystems

A distinction can be made between two types of approaches in assessing the value of different ecosystems. In the first type, a reference “natural” ecosystem is defined and other ecosystems are assessed according to the difference in “nature value” between these ecosystems and the reference ecosystem. The “nature value” of an ecosystem has been defined as a measure of the “development space for nature through land use.” Possible indicators of nature value include: biomass production or state, diversity, topsoil erosion or state, energy or substance balance, relaxation time\(^10\), rareness, and landscape perception (Blonk et al., 1997). Thus actual biodiversity may be assessed to a greater

---

\(^9\) According to data in Park (1997, p.445), 30% of species extinctions related to human activities are due to habitat alterations. Other contributors are hunting for commercial products (20%), hunting (18%), introduction of alien species (16%), pest and predator control (7%), pet trade (5%), pollution (2%), and religious and cultural practices (2%). Bennett (1991) states that “The single most important cause of species decline is habitat change, and it is the increasing intensity, scale and dynamism of human activities which are chiefly responsible for the degradation of natural and semi-natural habitats in Europe” (quoted in Pienkowski, 1993).

\(^10\) This is the time “nature would need to return by itself to a reference situation” (Blonk et al., 1997).
or lesser extent depending on the indicators chosen in the analysis. In the second type of approach, assessment is based on use of one or more indicators to assess biodiversity specifically for a range of ecosystems.

The first type of approach has developed from an IUCN report (IUCN/WWF/UNEP, 1991) that classified ecosystems into five types: natural, modified, cultivated, built and degraded systems. Heijungs et al. (1992) suggest categorisation of different ecosystem types in this implied hierarchy. After weighting the different categories, they suggest that changes from one type of ecosystem to another could be measured on this scale. This concept has been taken forward by Frischknecht et al. (1995) who use the same categorisation (minus the "degraded" category), assessing land use in terms of time taken for restoration from a current land use category to a "reference situation" chosen from the other land use categories. Blonk et al. (1996) and Weidema et al. (1996) also take a similar approach, suggesting that changes in ecosystems due to human activities should be measured against a reference ecosystem on a scale measuring the "nature value" of different types of ecosystems. The definition of an appropriate reference ecosystem is a matter for debate; Blonk et al. (1997) suggest the "would-be natural situation." Furthermore, after reviewing a number of nature value indicators, they suggest that the most appropriate indicator is:

\[
\text{Net Primary Production (NPP) - Net Community Production (NCP)}
\]

where NCP measures human food production. This could be supplemented by an indicator of other more "descriptive and irreversible aspects of ecosystem degradation, like biodiversity or erosion."

A particularly simple use of the second type of approach is demonstrated in the EPS Enviro-Accounting Method. A monetary value that describes "the whole world's willingness to pay to preserve diversity" is normalised to each one square metre cultivated land, irrespective of the type of cultivation. The authors suggest that an improvement would be to value different activities according to the number of species threatened with extinction due to these activities, and the value of each species to humans. However, this is not possible due to lack of information (Steen and Ryding, 1992, p.20, 52).

Biewinga and van der Bijl (1996), at CLM in the Netherlands, developed a more detailed method for assessing biodiversity in different agricultural crops. For each of four groups of species (birds, mammals, insects and flora), they scored cultivation of any one crop according to three indicators:
• Number of species (using Simpson’s Index where possible)
• Number of threatened species (using Red Lists)
• Number of characteristic species (using regional lists of species preferring habitat provided by that crop).

Each indicator was given a score —1, 0, +1 or +2, where —1 was a negative contribution and +2 was a positive contribution to biodiversity in that species group. The indicators were then weighted, and aggregated for each species group in a particular agricultural crop. Finally, the results for the different species groups in any one agricultural crop were weighted and aggregated to give an overall score for that crop. An example of some results is shown in Table 2.

Table 2. Assessment of Biodiversity In Northern Netherlands Using Method of Biewinga and van der Biil (1996)

<table>
<thead>
<tr>
<th>Crop</th>
<th>Species Group</th>
<th>Weighted average</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Birds</td>
<td>Mammals</td>
</tr>
<tr>
<td>Silage maize</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Willow</td>
<td>0</td>
<td>0.17</td>
</tr>
<tr>
<td>Sugar beet</td>
<td>0</td>
<td>0.5</td>
</tr>
<tr>
<td>Poplar</td>
<td>0.5</td>
<td>0.33</td>
</tr>
<tr>
<td>Grass fallow</td>
<td>0</td>
<td>0.5</td>
</tr>
<tr>
<td>Hemp</td>
<td>0.5</td>
<td>0</td>
</tr>
<tr>
<td>Miscanthus</td>
<td>0.67</td>
<td>0.5</td>
</tr>
<tr>
<td>Oil seed rape</td>
<td>1.83</td>
<td>0</td>
</tr>
<tr>
<td>Winter wheat</td>
<td>1.83</td>
<td>0.5</td>
</tr>
</tbody>
</table>

Note: a high value on this scale is indicative of high biodiversity value, in contrast to the usual approach in LCA where a high value is indicative of negative environmental impact.
Source: Biewinga and van der Bijl, 1996, p.189.

3.2 Approaches To Using Values of Different Ecosystems In LCA

Having defined the nature value, or biodiversity value, of different ecosystems, a further question concerns how these values are used to assess impacts in any system under analysis.

As mentioned above, Heijungs et al. (1992) suggest that changes from any one ecosystem type to another due to the system under analysis are measured on the scale of the five land use categories. However, they do not give details of how to weight changes between the categories. Wegener Sleeswijk et al. (1996), on the other hand, suggest that types of ecosystems existing due to the activities under analysis should be assessed rather than changes in ecosystems. Blonk et al. (1997) also support assessment of the ecosystem existing during the activities under analysis.
Both Wegener Sleeswijk et al. (1996, p. 80) and Blonk et al. (1997) advocate calculation of the final value for biodiversity assessment by multiplying the land area during the time period under consideration (i.e. m²-year) by the relevant ecosystem value(s).

### 3.3 Evaluation of the Different Approaches

A number of approaches to assessing the value of different ecosystems have been outlined in Section 3.1. The main problem occurs with the first type of approach, involving definition of a reference ecosystem. Heijungs et al. (1992) imply they should be those ecosystems which have experienced minimum human interference, Weidema et al. (1996) suggest climax ecosystems as reference states, and Blonk et al. (1997) advocate use of the "would-be natural situation." The use of these reference ecosystems is problematic because they fail to acknowledge the contribution of ecosystems other than "natural" or climax ecosystems to global biodiversity. In fact, many ecosystems containing rare species require active human management for their conservation, so that cessation of these activities would have a negative impact on biodiversity. Examples in the UK include chalk downlands maintained by less intensive grazing regimes, unimproved meadows traditionally managed for hay production or as pastureland, and coppiced woodlands. Furthermore, they imply that land such as that in parts of the Netherlands should cease to exist because its "natural" or "climax" state is seabed rather than terrestrial ecosystem(s). Also, use of reference ecosystems means that biodiversity is not assessed on a global scale because there is no distinction between climax or "natural" ecosystems in different parts of the world. In other words, the implication is that, for example, the climax ecosystem in South America (say, tropical rainforest) has the same value as the climax ecosystem in central Africa (say, desert scrub).

The second type of approach is more promising because it focuses on indicators of the biodiversity value of different ecosystems without reference to any one particular reference ecosystem. However, here the choice of indicators is critical to the credibility of the method. For example, a monetary value describing the whole world's willingness to pay to preserve biodiversity at some defined and, presumably, agreed level (Steen and Ryding, 1992) seems highly unlikely to achieve any level of credibility. In Section 4 I develop this indicator approach, selecting four indicators to reflect the biodiversity value of different ecosystems.

The debate over assessing changes in ecosystems versus existence of ecosystems due to a system under analysis (Section 3.2) is an interesting one. It arises because LCA conventionally only assesses changes due to a system under analysis, yet this approach does not seem adequate for assessing
physical habitat degradation. For example, consider a comparison between two biofuels: biomass from *Miscanthus*, and wood from a coppiced deciduous forest. For the sake of argument, the change in physical habitat value of the land use associated with production of both biofuels is of a similar magnitude\(^\text{11}\), and they both require the same amount of land to produce equivalent quantities of energy. Therefore, if only changes in land use are considered, there is no basis for differentiating between the two systems in terms of physical habitat degradation. Yet the existence of the deciduous woodland is likely to be more beneficial for biodiversity because it supports a wider range of species, including rare species, than the *Miscanthus* crop. The implication is that the continued existence of different ecosystems, as well as changes in ecosystems, should be assessed in LCA. This is the justification for the assessment method developed in Section 7.

4. Development of Physical Habitat Factors

The previous discussion has suggested that it is appropriate to develop weighting factors to assess existence of, and changes in, different ecosystems. Ideally the factors should reflect:

- The contribution of each ecosystem to global biodiversity
- The existence of internationally rare species in certain ecosystems
- The number of species in each ecosystem
- The number of individuals in each ecosystem.

These considerations can be regarded as indicators to be included in development of weighting factors for different ecosystems\(^\text{12}\). Possible ways of measuring these indicators are shown in Table 3. Operationalisation requires choosing an appropriate measurement method for each indicator, and weighting each indicator relative to the other indicators’ contribution to biodiversity. Development of the method is discussed below.

\(^{11}\) For example, the *Miscanthus* may be grown on land formerly used for cultivation of oilseed rape and wheat, and the deciduous woodland may have formerly been a mix of coniferous plantations and deciduous woodland.

\(^{12}\) The role of different ecosystems in functioning of the biosphere at a global level is a further consideration. It is not addressed here due to our poor understanding about these types of interactions. However, it does raise a question about the role of uncertainty in LCA, and I return to this in Section 9.
Table 3. Possible Measurements for Different Indicators In Assessing Physical Habitat Degradation

<table>
<thead>
<tr>
<th>Indicator Number</th>
<th>Indicator</th>
<th>Possible Measurements</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Contribution to global ecosystem diversity</td>
<td>Area of ecosystem as a proportion of total land area in world.</td>
</tr>
<tr>
<td>2</td>
<td>Number of rare species</td>
<td>Number of species listed in Red Data books(^a) found in ecosystem. Number of rare species listed elsewhere that are found in ecosystem.</td>
</tr>
<tr>
<td>3</td>
<td>Number of species</td>
<td>Number of species in ecosystem. Number of species from representative group of species in ecosystem.</td>
</tr>
<tr>
<td>4</td>
<td>Number of individuals</td>
<td>Number of individuals in ecosystem. Gross Primary Productivity (GPP)(^b) of ecosystem. Net Primary Productivity (NPP)(^c) of ecosystem.</td>
</tr>
</tbody>
</table>

\(^a\) The Red Data books are catalogues published by the International Union for the Conservation of Nature (IUCN) or by national authorities listing species that are rare or in danger of becoming extinct globally or locally.

\(^b\) Gross Primary Productivity (GPP) is the total fixation of energy by photosynthesis in an area, expressed as units of energy (J/m\(^2\)/day) or dry organic matter (kg/m\(^2\)/year).

\(^c\) Net Primary Productivity (NPP) is the energy or organic matter accumulated by plants during photosynthesis (GPP minus respiration), measured using the same units as for GPP.

**Choice of Measurement Method for Each Indicator**

**Indicator 1** is straightforward: the area of each ecosystem relative to the area of other ecosystems in the world can be used to assess its contribution to global ecosystem diversity. A relatively small area implies that the ecosystem has a higher physical habitat value than other ecosystems with larger areas (albeit given the shortcomings of this assessment method as outlined in Section 2, hence necessitating the need for more than one indicator).

In theory, assessment on the basis of listed numbers of rare species seems relatively straightforward for **indicator 2**. However, its use in this context implies that the physical habitat value of an ecosystem increases with an increase in numbers of rare species in that ecosystem; in other words, the greater the number of rare species in an ecosystem, the greater priority should be given to its conservation. Although this is a reasonable assumption for many ecosystems, it does not apply to all ecosystems. This can be illustrated by considering bird species whose habitat is farmland in the UK. Many of these species have experienced a reduction in numbers greater than 50% in the 25 years from 1969 to 1994 (Gibbons *et al.*, 1993). A biodiversity assessment method that gives a high rating to the farmland ecosystem because of these threatened species is misleading as it implies that a further increase in area of this ecosystem would be an appropriate response to the situation. In fact, this response would not improve overall biodiversity; a far more appropriate response would be to alter farming practices in the existing ecosystem, for example changing from autumn to spring-sown.
cereals (UK Biodiversity Steering Group, 1995a, p.31). This example suggests that assessment for
dicator 2 should only account for species that are threatened due to a decrease in area of their
habitat. Species that are threatened primarily due to a change in management of the habitat require
assessment using an alternative approach (as discussed in Section 6). Therefore, use of this indicator
requires additional judgements about why species are rare in specific ecosystems: if the reason is
primarily the management regime in the ecosystem rather than a decrease in area of that ecosystem,
this indicator should not be used.

Theoretically, one possible approach for indicators 3 and 4 is the Simpson’s Index developed for
ecological studies. This index accounts for species richness (i.e. numbers of species) and evenness
(or equitability) of distribution of individuals between species in the following way:

\[
\text{Simpson's Index, } D = \frac{1}{\sum_{i=1}^{S} p_i^2}
\]

where

- \( S \) = total number of species
- \( p_i \) = proportion of individuals that species \( i \) contributes to the total number of
  individuals in the ecosystem.

Operation of the Index can be illustrated by the following example. Consider six trees:

- Tree A: supports two species of insect, and 100 individuals of each species.
- Tree B: supports two species of insect; one species has 160 individuals while the other has 40
  individuals.
- Tree C: supports two species of insect, and 20 individuals of each species.
- Tree D: supports two species of insect; one species has 32 individuals while the other has 8
  individuals.
- Tree E: supports five species of insect, and 40 individuals of each species.
- Tree F: supports five species of insect; one species has 160 individuals while the others each have
  10 individuals.

The data are summarised in Table 4 together with the Simpson’s Index value for each tree. They
show that the tree with the highest number of species and most equitable distribution of individuals
among the species (Tree E) has the highest D value. They also show how changes in the equitability
of distribution of individuals among species changes the D value (Tree A versus Tree B, or Tree C
versus Tree D, or Tree E versus Tree F). Furthermore, they show how an increase in the number of species increases the D value (Tree A versus Tree E). However, the Index fails to distinguish between ecosystems containing the same number of species and the same equitability but different total numbers of individuals, and therefore with different genetic diversities. For example, Tree A (200 individuals) and Tree C (40 individuals), and Tree B (200 individuals) and Tree D (40 individuals), have the same D values. Therefore, this Index can only make a partial contribution to assessment of indicators 3 and 4 because it does not account for genetic diversity within species (as represented by number of individuals in a species).

Table 4. Application of the Simpson’s Index: A Theoretical Example

<table>
<thead>
<tr>
<th>Tree</th>
<th>Number of Species</th>
<th>Number of Individuals of Each Species</th>
<th>Total Number of Individuals</th>
<th>Simpson’s Index D Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tree A</td>
<td>2</td>
<td>100, 100</td>
<td>200</td>
<td>2.00</td>
</tr>
<tr>
<td>Tree B</td>
<td>2</td>
<td>160, 40</td>
<td>200</td>
<td>1.47</td>
</tr>
<tr>
<td>Tree C</td>
<td>2</td>
<td>20, 20</td>
<td>40</td>
<td>2.00</td>
</tr>
<tr>
<td>Tree D</td>
<td>2</td>
<td>32, 8</td>
<td>40</td>
<td>1.47</td>
</tr>
<tr>
<td>Tree E</td>
<td>5</td>
<td>40, 40, 40, 40, 40</td>
<td>200</td>
<td>5.00</td>
</tr>
<tr>
<td>Tree F</td>
<td>5</td>
<td>160, 10, 10, 10, 10</td>
<td>200</td>
<td>1.54</td>
</tr>
</tbody>
</table>

Anyway, in practice LCA studies are unlikely to include ecological studies - if at all - at this level of detail. A feasible alternative for assessing total number of species (i.e. indicator 3) depends upon generalisation from existing studies of restricted groups of species. Those groups considered to be representative of overall biodiversity are used as indicators of species richness, as discussed by Pearson (1995).

For indicator 4 (number of individuals), the Net Primary Productivity (NPP) of an ecosystem seems to be a suitable indicator given the lack of data for any other approach. NPP is preferred rather than Gross Primary Productivity (GPP) because it is more representative of the total number of organisms in an ecosystem (since it measures accumulation of living organic matter rather than the quantity produced plus the amount lost in respiration). I do not subtract Net Community Production (NCP) as Blonk et al. (1997) recommend (see Section 3.1) because plants cultivated and livestock reared for human consumption represent sources of – albeit limited – genetic diversity alongside other more “natural” species.
Weighting Each Indicator Relative To the Other Indicators

Weighting of the different indicators requires assessment of their relative contributions to biodiversity. As a first estimate, I suggest that indicators 1 and 4 (ecosystem diversity and number of individuals) should be given weighting factors of one each, and indicators 2 and 3 (number of rare species and number of species) weighting factors of two each. The reason is that indicators 2 and 3 are likely to be better indicators of genetic diversity than indicators 1 and 4, and assessment of biodiversity on the basis of genetic diversity makes most sense (see Section 2). The shortcomings of assessing genetic diversity by land area (i.e. indicator 1) have been discussed in Section 2. Assessment of genetic diversity by number of individuals (i.e. indicator 4) is also less indicative of high genetic diversity than the number of rare species and number of species because generally there is greater diversity between species than within species (although there are, of course, exceptions as shown in footnote 7).

Development of the Weighting Factors for Different Ecosystems

Based on the discussions above, I use the following measurement parameter for each indicator:

- Indicator 1: area of ecosystem
- Indicator 2: number of listed rare species found in ecosystem
- Indicator 3: number of species in ecosystem
- Indicator 4: Net Primary Productivity of ecosystem.

The final weighting factor (Physical Habitat Factor, PHF) for any ecosystem e, on a scale of 0 (high physical habitat value) to 1 (low physical habitat value), is then:

\[ \text{PHF}_e = \frac{1}{6} \left[ (\text{indicator 1}) + 2 \left( \text{indicator 2} \right) + 2 \left( \text{indicator 3} \right) + (\text{indicator 4}) \right]. \]

\[ \text{PHF}_e = \frac{1}{6} \left[ \left( \frac{A_e}{A_{\text{max}}} \right) + 2 \left( 1 - \frac{R_e}{R_{\text{max}}} \right) + 2 \left( 1 - \frac{S_e}{S_{\text{max}}} \right) + \left( 1 - \frac{P_e}{P_{\text{max}}} \right) \right], \]

where

- \( A_e \) = Area of ecosystem e in world
- \( A_{\text{max}} \) = Largest area of any one ecosystem in world
- \( R_e \) = Number of rare species in ecosystem e (apart from physical habitats where species are rare primarily due to management practices rather than a change in land area; in these cases, indicator 2 is set to unity)
5. Compilation of Physical Habitat Factors At Different Levels of Detail

In the last section, a theoretical approach was developed for assessing the relative physical habitat values of different ecosystems according to their contributions to global biodiversity. In this section, practical calculation of Physical Habitat Factors (PHFs) is demonstrated at global and national levels in Sections 5.1 and 5.2. At each of these two levels, a Physical Habitat Index can be calculated independently that consists of the PHFs for the different ecosystems at that level of detail. The most appropriate level of detail (i.e. global or national) to use for any study depends upon the purpose of the study, and the availability of data; this is discussed in Section 8.

5.1 Compilation of PHFs for a Physical Habitat Index At Global Level

In order to show how a Physical Habitat Index can be calculated, some initial estimates have been made in Table 5 for major ecosystems in the world. The Index results are calculated from existing data on areas of different ecosystems and NPP, and “first guess” estimates of numbers of species and numbers of rare species in these ecosystems (see Appendix IV.2 for further details). As such, the results should be considered as preliminary values requiring further verification. I return to the role of uncertainty in this type of assessment in Section 9.

Using this method and the “first guess” data, the results suggest that tropical forests and temperate evergreen forest are a priority for conservation (with PHF values of 0.13 to 0.15). These are followed by temperate deciduous forests, boreal forest and grasslands (PHF values of 0.34 to 0.52). Tundra and cultivated land have lower conservation value (PHF values of 0.71 and 0.85 respectively), and the lowest values are for desert and unvegetated land (PHF values of 0.95 and 1.00). The most unexpected result is for tundra, and is due to the estimated low numbers of species and rare species, and low Net Primary Productivity, of this ecosystem type. This value may change if the first guess estimates are inaccurate for total number of species and rare species. However, an alternative explanation is that
Table 5. Example To Show Compilation of the Physical Habitat Index At Global Level

<table>
<thead>
<tr>
<th>Ecosystem Type</th>
<th>Area(^a)</th>
<th>Number of Rare Species</th>
<th>Number of Species</th>
<th>Net Primary Productivity(^a)</th>
<th>Physical Habitat Factor (PHF)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>10(^b) square kilometres</td>
<td>Indicator 1</td>
<td>Number of rare species</td>
<td>&quot;First guess” estimate for indicator 2(^b)</td>
<td>Number of species</td>
</tr>
<tr>
<td>Tropical rainforest</td>
<td>17.0</td>
<td>0.71</td>
<td>-</td>
<td>0.1</td>
<td>-</td>
</tr>
<tr>
<td>Tropical seasonal forest</td>
<td>7.5</td>
<td>0.31</td>
<td>-</td>
<td>0.1</td>
<td>-</td>
</tr>
<tr>
<td>Temperate evergreen forest</td>
<td>5.0</td>
<td>0.21</td>
<td>-</td>
<td>0.5</td>
<td>-</td>
</tr>
<tr>
<td>Temperate deciduous forest</td>
<td>7.0</td>
<td>0.29</td>
<td>-</td>
<td>0.5</td>
<td>-</td>
</tr>
<tr>
<td>Boreal forest</td>
<td>12.0</td>
<td>0.50</td>
<td>-</td>
<td>0.5</td>
<td>-</td>
</tr>
<tr>
<td>Woodland and shrubland</td>
<td>8.5</td>
<td>0.35</td>
<td>-</td>
<td>0.5</td>
<td>-</td>
</tr>
<tr>
<td>Savannah</td>
<td>15.0</td>
<td>0.63</td>
<td>-</td>
<td>0.1</td>
<td>-</td>
</tr>
<tr>
<td>Temperate grassland</td>
<td>9.0</td>
<td>0.38</td>
<td>-</td>
<td>0.5</td>
<td>-</td>
</tr>
<tr>
<td>Tundra and alpine</td>
<td>8.0</td>
<td>0.33</td>
<td>-</td>
<td>0.8</td>
<td>-</td>
</tr>
<tr>
<td>Desert and semi-desert shrub</td>
<td>18.0</td>
<td>0.75</td>
<td>-</td>
<td>1.0</td>
<td>-</td>
</tr>
<tr>
<td>Extreme desert, rock, sand</td>
<td>24.0</td>
<td>1.00</td>
<td>-</td>
<td>1.0</td>
<td>-</td>
</tr>
<tr>
<td>and ice</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cultivated land</td>
<td>14.0</td>
<td>0.58</td>
<td>-</td>
<td>1.0</td>
<td>-</td>
</tr>
<tr>
<td>Swamp and marsh</td>
<td>2.0</td>
<td>0.08</td>
<td>-</td>
<td>?</td>
<td>-</td>
</tr>
<tr>
<td>Lake and stream</td>
<td>2.0</td>
<td>0.08</td>
<td>-</td>
<td>?</td>
<td>-</td>
</tr>
<tr>
<td>Estuaries</td>
<td>1.4</td>
<td>0.06</td>
<td>-</td>
<td>?</td>
<td>-</td>
</tr>
</tbody>
</table>

\(^a\) From Begon et al. (1996, p.715) (after Whittaker, 1975). Note that data are 20 years old and so may not be representative of the current situation.

\(^b\) "First guess" estimate for number of species based on latitude of ecosystems as indicated by a number of ecological studies (Begon et al., 1996, p.900-904), adjusted to (partially) account for other factors. See Appendix IV.2 for further details.
Table 6. Data for Compilation of a Physical Habitat Index for the UK

<table>
<thead>
<tr>
<th>Ecosystem Type</th>
<th>Number of Species of Conservation Concern</th>
<th>Area in Hectares</th>
</tr>
</thead>
<tbody>
<tr>
<td>Broadleaved and yew woodland</td>
<td>232</td>
<td>800,000 (Britain)</td>
</tr>
<tr>
<td>Planted coniferous woodland</td>
<td>-</td>
<td>1,516,000 (Britain)</td>
</tr>
<tr>
<td>Native pine woodland</td>
<td>37</td>
<td>16,000 (UK)</td>
</tr>
<tr>
<td>Lowland wood pastures, parkland</td>
<td>38</td>
<td>10,000-20,000 (?UK)</td>
</tr>
<tr>
<td>Boundary features</td>
<td>65</td>
<td>Hedges: 450,000 km (UK)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Dry stone walls: 112,500 km (England)</td>
</tr>
<tr>
<td>Arable</td>
<td>72</td>
<td>-</td>
</tr>
<tr>
<td>Improved grassland</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Unimproved neutral grassland</td>
<td>-</td>
<td>&lt;15,000 (UK)</td>
</tr>
<tr>
<td>Acid grassland</td>
<td>-</td>
<td>1,230,000 (Britain)</td>
</tr>
<tr>
<td>Calcareous grassland</td>
<td>112</td>
<td>40,000-50,000 (UK)</td>
</tr>
<tr>
<td>Lowland heathland</td>
<td>82</td>
<td>58,000 (UK)</td>
</tr>
<tr>
<td>Grazing marsh</td>
<td>-</td>
<td>300,000 (UK)</td>
</tr>
<tr>
<td>Fens, carr, marsh, swamp, reedbed</td>
<td>73</td>
<td>Reedbeds: 5,000 (UK)</td>
</tr>
<tr>
<td>Lowland raised bog</td>
<td>-</td>
<td>6,000 (UK)</td>
</tr>
<tr>
<td>Standing open water</td>
<td>136</td>
<td>-</td>
</tr>
<tr>
<td>Rivers and streams</td>
<td>75</td>
<td>-</td>
</tr>
<tr>
<td>Canals</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Montane (alpine/subalpine types)</td>
<td>70</td>
<td>600,000 (UK)</td>
</tr>
<tr>
<td>Upland heathland</td>
<td>74</td>
<td>56,658,000 (UK)</td>
</tr>
<tr>
<td>Blanket bog</td>
<td>-</td>
<td>1,500,000 (UK)</td>
</tr>
<tr>
<td>Maritime cliff and slope</td>
<td>91</td>
<td>-</td>
</tr>
<tr>
<td>Shingle above high tide mark</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Boulders and rock above high tide</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Coastal: strandline</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Machair</td>
<td>-</td>
<td>5,000 (UK)</td>
</tr>
<tr>
<td>Saltmarsh</td>
<td>-</td>
<td>45,000 (UK)</td>
</tr>
<tr>
<td>Sand dune</td>
<td>52</td>
<td>47,118 (Britain)</td>
</tr>
<tr>
<td>Estuaries</td>
<td>54</td>
<td>-</td>
</tr>
<tr>
<td>Saline lagoons</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Islands and archipelagos</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Inlets and enclosed bays (including sea lochs, rias, voes)</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Open coast</td>
<td>61</td>
<td>-</td>
</tr>
<tr>
<td>Open sea water column</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Shelf break</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Offshore seabed</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Limestone pavements</td>
<td>-</td>
<td>&lt;3,000 (UK)</td>
</tr>
<tr>
<td>Urban</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>


a Species of conservation concern were identified using specific criteria by the Biodiversity Steering Group. These included: threatened endemic and globally threatened species, species where the UK has more than 25% of the world’s population, and species listed in various international conventions (UK Biodiversity Steering Group, 1995b, p.2).
the "unexpectedness" of the result is a reflection of human value systems which attach greater subjective value to certain ecosystems, regardless of their actual contribution to global genetic diversity. Another unexpected result is for temperate evergreen and deciduous forests, with PHF values respectively of 0.37 and 0.38. It implies, for example, that coniferous forests in the UK have higher conservation value than deciduous forests. Although this seems logical for the Caledonian pinewoods, it seems harder to justify in other areas of Britain. These two examples show that any use of these values should be subject to expert judgement guided by additional information where this is available.

This Index provides a very general assessment of the physical habitat value of different ecosystems throughout the world. If nothing else, it does indicate a general trend in physical habitat value from tropical forests (high physical habitat value), to temperate and boreal forests, to grasslands, to cultivated lands, to unvegetated land (low physical habitat value). However, in most LCA studies, a more detailed level of assessment is required for meaningful comparisons between land use in different systems. Therefore, a possible approach for calculation of a Physical Habitat Index at country-level is discussed in the next section.

5.2 Compilation of PHFs for a Physical Habitat Index At National Level: The UK

The Index described in Section 5.1 provides a basis for comparison between different ecosystems. However, more often than not LCAs compare different systems using land within the same major ecosystem category. For example, livestock production may take place on different types of grassland, and wood may be harvested from different types of forest. Is it possible to account for the more detailed differences in ecosystems occurring at levels of detail below those identified for ecosystems at a global level? In theory, the answer is "yes" because the ecosystem categories described so far can be envisaged as consisting of nested categories of ecosystems at different levels of detail. For example, the temperate grassland category in Table 5 could be composed of improved, unimproved, neutral, acid and calcareous grasslands. However, obtaining relevant data on a global scale at these greater levels of detail is impractical. Instead, one possible approach is to calculate PHFs at national level rather than global level, since more detailed data on different ecosystems are generally available at this level. This approach is outlined below for the UK.

In the UK, the Biodiversity Steering Group reports (1995a, 1995b) provide a good basis for compilation of more detailed PHFs. The Steering Group developed a basic framework of 37 broad ecosystem types that include the whole land surface of the UK and the surrounding sea to the edge of
the continental shelf in the Atlantic Ocean. The terrestrial ecosystem types developed by the Steering
Group are listed in Table 6, along with data on the areas occupied by many of the ecosystems. In
addition, the Steering Group estimated the number of species of conservation concern found in each
ecosystem, and again some data are given in the reports as shown in Table 6. Therefore, in order to
calculate the PHFs for these more detailed ecosystem types, two further data sets are required:
numbers of species and numbers of individuals in each ecosystem. For numbers of species, the
Countryside Survey 1990 carried out detailed field studies to estimate the numbers of plant species in
32 Land Use classes (following the Institute of Terrestrial Ecology classification) (Barr et al., 1993;
DOE, 1993). Although the data in the Main Report do not enable the reader to extrapolate the results
to the Biodiversity Steering Group’s framework of habitat types, presumably the detailed field study
results should facilitate this type of analysis. For numbers of individuals, data are required on the Net
Primary Productivity (NPP) of the different ecosystems, and it should be possible to develop such a
dataset from existing information.

Using these data, PHFs could be developed using the method presented in Section 4. In this case, the
\( A_{\text{max}}, R_{\text{max}}, S_{\text{max}} \) and \( P_{\text{max}} \) values would represent the highest values at national level rather than global
level. Following on from this, a shortcoming of this approach is therefore that it assesses each
ecosystem’s contribution to UK biodiversity rather than its contribution to global biodiversity. As a
result, an ecosystem that is, for example, relatively widespread in the UK but rare in the world may be
assessed as having low physical habitat value, although at an international level it may be important
for conserving global biodiversity. However, in order to address this aspect it would be necessary to
compile an international Physical Habitat Index at the same level of detail as the UK Biodiversity
Steering Group’s recent assessment. This is extremely unlikely to happen, and so use of a country-
specific Index seems a practical alternative option at the present time.

It should also be noted that this Index can only be applied to land use within the UK. Assessment of
land areas used in two or more countries within any one system under analysis, is only possible if a
common Index is compiled for all these countries. Otherwise, the Physical Habitat Depletion values
for the system would only reflect impacts on physical habitats at the national rather than international
level.
6. Compilation of a Physical Management Index To Account for Management Practices Within Ecosystems: Agricultural Systems

In the discussion above, Physical Habitat Indices each consisting of a range of PHFs have been discussed for different ecosystems at the global and national levels. If this categorisation does not provide enough detail to distinguish between systems in an LCA analysis, a further level of categorisation within the relevant type of ecosystem is desirable (given the availability of sufficient data in a study). However, rather than attempting to develop sub-categorisations of ecosystems at this level of detail, I suggest that a more practical approach is to identify the relevant management practices that are the primary determinants of Physical Habitat Depletion (PHD) within the categories identified at national level. These can be used as indicators of PHD, and can be assessed and weighted for the systems in an LCA study. Given the focus of this thesis, I develop this approach below for assessment of agricultural systems.

In the UK, management practices with relevant impacts on cultivated land include:

- **For all agricultural systems:**
  a) *Existence of additional features such as ponds, ditches and rocky outcrops*: Often these features will enhance biodiversity on agricultural land as they provide additional habitats for species.
  
  b) *Mosaics of habitats*: Many species require mosaics of habitats at different geographical scales for their survival. For example, frogs need ponds for the tadpole stage of their life cycle, butterflies require sunny patches in woodlands, and invertebrates in general need a variety of habitats (or microhabitats) to complete the different stages of their life cycles (Pienkowski, 1993). Additional features, as well as field boundaries and margins, are important components of these mosaics, but their geographical distribution on farmland is important as well as their existence.

- **For arable crops:**
  a) *Timing of sowing crops*: In recent years, there has been an increase in areas of cereals sown in the autumn rather than the spring in the UK. As a result, the area of stubble left over the winter period has decreased. As this stubble is the winter feeding ground for a number of farmland

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13 However, Pienkowski (1993) warns against simplification of recommendations with regard to addition of such features, giving the example of farm pools. Construction of these pools has been promoted in intensively farmed lowland areas of the UK in order to increase the nature value of these areas. However, this advice was also passed on to some upland farmers, where the most obvious places to dig pools would often be in semi-natural wetlands leading to the loss of habitats of high nature conservation value.
birds, its disappearance is thought to have contributed to the decline of these species since the 1960s (UK Biodiversity Steering Group, 1995a, p.31).

- **For grasslands:**
  a) *Timing of cutting grass*: Early cutting of grass destroys the nests of farmland birds that breed in grassland (such as the corncrake and skylark). This can be avoided by delaying the first cut of grass each year (DOE, 1994, p.74).
  b) *Grazing density of livestock*: The characteristic plant species found in different grasslands are sensitive to the grazing intensity of livestock. Thus lowland calcareous grasslands require fairly heavy grazing whilst neutral grasslands require low grazing levels in order to maintain their characteristic flora and associated fauna (Pienkowski, 1993, p.193).

- **For boundaries:**
  a) *Types of field boundaries and margins*: Field boundaries include hedges, walls, fences and banks. Field margins are the associated strips of uncultivated land that run beside these features and additional “headlands” at the corners of fields. In recent years, fences have replaced many other types of field boundaries, and field margins have been minimised in order to increase the areas of cultivated land. This has a negative impact on biodiversity because field boundaries such as hedges and dry stone walls, and field margins provide important habitats for a wide range of species. For example, over 600 plant, 1,500 insect, 65 bird and 20 mammal species are known to live or feed in hedgerows (UK Biodiversity Steering Group, 1995b, p.276).
  b) *Maintenance of field boundaries and margins*: The management of existing field boundaries and margins has important implications for biodiversity of farmland. For example, it is estimated that bad management of hedges between 1978 and 1990 led to the loss of an average of one plant species from each 10 metre length of hedge, an 8% decrease in plant species diversity in these habitats (DOE, 1994, p.97). Appropriate management of field margins is particularly important, as illustrated by The Game Conservancy’s ongoing research on the grey partridge. This work has shown the importance of insects and weed seeds in the early diet of grey partridge chicks. On arable land, much of this food is found in field margins, and there is a positive correlation between grey partridge populations and the size and siting of field margins.

In this approach, relevant indicators are selected for assessing alternative management practices in the systems under analysis. A provisional framework for assessing arable ecosystems is shown in Table 7. Each indicator is scored and weighted, and the values are added to obtain an overall Physical
Management Factor (PMF) for each management regime in the systems under analysis. The PMF for any particular management regime $r$ is:

$$ PMF_r = 1 - \left[ \frac{\sum_{i} I_{i} W_{i}}{\sum_{i} W_{i}} \right] $$

where $I_{i}$ = Score for indicator $i$ (on a scale of 0 to 1 where 0 is least beneficial and 1 is most beneficial management for biodiversity in the system under analysis)

$W_{i}$ = Weighting for indicator $i$ (on a scale of 1 to 10 where 1 is least important and 10 is most important for enhancing biodiversity in the ecosystem category).

Using this equation, a PMF value approaching 0 is most beneficial for biodiversity and a value approaching 1 is least beneficial for biodiversity within the ecosystem under consideration.

The advantage of this method for assessing alternative management regimes is that the number of indicators can be tailored to the study in question. Thus:

- For a specific farm: a full range of indicators are used to calculate the PMFs.
- For an average farm representative of the national situation: a restricted set of indicators is used based on the information available. For example, in a comparison between intensive and organic farming it may be known that most organic farms maintain a one metre uncultivated strip next to their boundaries whilst most intensive farms do not practise this management technique. Therefore, this indicator is used in the analysis. However, there may be no information on the average state of hedges in intensive versus organic farms, and so it is not possible to include this indicator in the analysis.
- For an hypothetical farm: a very restricted set of indicators will be appropriate. For example, it will not be possible to make judgements about the use of one metre uncultivated strips or the state of hedgerows between an hypothetical intensive and organic farm because there is no inherent reason why they should differ in these management practices.
Table 7. Example of Framework for Calculating Physical Habitat Degradation for Arable Ecosystems

<table>
<thead>
<tr>
<th>Ecosystem Type</th>
<th>Indicator (i)</th>
<th>Preferred Management Practice</th>
<th>Weighting of Indicators (W_i)</th>
<th>Score for System s for Each Indicator (I_i)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Arable</td>
<td>1. Additional features</td>
<td>Presence of ponds, ditches, etc. beneficial to wildlife.</td>
<td>W_1</td>
<td>I_1</td>
</tr>
<tr>
<td></td>
<td>2. Mosaic of habitats</td>
<td>Rich mosaic of habitats at all geographical scales.</td>
<td>W_2</td>
<td>I_2</td>
</tr>
<tr>
<td>2. Boundary features</td>
<td>1. Additional features (ponds, ditches, etc.)</td>
<td>Additional features beneficial to wildlife.</td>
<td>W_4</td>
<td>I_4</td>
</tr>
<tr>
<td></td>
<td>2. Mosaic of habitats</td>
<td>Rich mosaic of habitats at all geographical scales</td>
<td>W_5</td>
<td>I_5</td>
</tr>
<tr>
<td></td>
<td>3. Type of field boundaries</td>
<td>Hedges and dry stone walls rather than fences.</td>
<td>W_6</td>
<td>I_6</td>
</tr>
<tr>
<td></td>
<td>4. Maintenance of field boundaries</td>
<td>Hedges: mature, sizeable, no gaps, containing trees.</td>
<td>W_7</td>
<td>I_7</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Dry stone walls: maintained rather than falling down.</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

7. Use of Physical Habitat Factors and Physical Management Factors To Calculate Physical Habitat Degradation for Any System

Use of Physical Habitat Factors

Options for use of either the global or national Index can be demonstrated with an example. Suppose that we wish to assess the impacts on biodiversity of four alternative scenarios (Systems A to D) for delivery of a particular service. All the scenarios use the same amount of land for the same amount of time but they have different land use implications. During the time period under analysis:

- System A involves a change of land use from urban land to forest
- System B involves no change of land use: the land remains as forest
- System C involves no change of land use: the land remains as urban land
- System D involves a change of land use from forest to urban land.
For this example, urban land has a Physical Habitat Factor (PHF) value of 1 and forest has a PHF value of 0. Table 8 shows four alternative ways of assessing Physical Habitat Degradation (PHD) for these alternative systems (prior to multiplication by the area-time value):

- Option 1: $\text{PHD} = \text{PHF}_f - \text{PHF}_i$
- Option 2: $\text{PHD} = \text{PHF}_f - \text{PHF}_i + \text{PHF}_f$
- Option 3: $\text{PHD} = \frac{1}{3} [1 + 2\text{PHF}_f - \text{PHF}_i]$
- Option 4: $\text{PHD} = 10^{-\left[2 - 2\text{PHF}_f + \text{PHF}_i\right]}$ i.e. $\log_{10} \text{PHD} = -\left[2 - 2\text{PHF}_f + \text{PHF}_i\right]$

Option 1 does not distinguish between maintenance of different ecosystems due to the system under analysis; thus Systems B and C have the same PHD value despite the fact that System B contributes to conservation of an ecosystem with higher physical habitat value than System C. This shortcoming is addressed in Option 2 by adding the PHF value for the ecosystem at the end of the study to the PHD equation. Option 3 shows how the equation in Option 2 is altered to give a final PHD equation with results on a linear scale from 0 to 1. In Option 4, the results are on an exponential scale which runs from 0.001 to 1.0; thus the relative weighting between the four systems is different from Option 3.

Table 8. Example To Show Alternative Assessment Methods for PHD of Different Systems

<table>
<thead>
<tr>
<th>System</th>
<th>Land Use Implications</th>
<th>Results Using Different Options for Assessing Physical Habitat Degradation</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Land use at beginning of study (PHF value)</td>
<td>Land use at end of study (PHF value)</td>
</tr>
<tr>
<td>System A</td>
<td>Urban (1)</td>
<td>Forest (0)</td>
</tr>
<tr>
<td>System B</td>
<td>Forest (0)</td>
<td>Forest (0)</td>
</tr>
<tr>
<td>System C</td>
<td>Urban (1)</td>
<td>Urban (1)</td>
</tr>
<tr>
<td>System D</td>
<td>Forest (0)</td>
<td>Urban (1)</td>
</tr>
</tbody>
</table>

N.B. The results in the table for the different options in Systems A to D need to be multiplied by the areas used in the systems for the final results for Physical Habitat Degradation. This is not shown here for clarity of presentation.

The preferred PHD equation for any system under analysis ($s$) is therefore either:

$$\text{PHD}_{s} (1) = \frac{1}{3} [1 + 2\text{PHF}_f - \text{PHF}_i] x [A_s]$$

or

$$\text{PHD}_{s} (2) = 10^{-\left[2 - 2\text{PHF}_f + \text{PHF}_i\right]} x [A_s]$$

where $\text{PHF}_f$ = Physical Habitat Factor for ecosystem in system $s$ at end of study
PHFi = Physical Habitat Factor for ecosystem in system s at beginning of study
Ai = Area of land in system s for time t (ha-year).

The implications of choosing one or other of these equations are explored in Section 9.

Use of Physical Management Factors

Calculation of Physical Habitat Depletion (PHD) using the Physical Management Factors (PMFs) is more straightforward than using the PHFs. Here, only management during the time period under analysis is assessed. As a result, the PHD equation for the system under analysis (s) is:

\[ \text{PHD}_s = \text{PMF}_s \times A_i \]

where

PMFs = Physical Management Factor for system s
Ai = Area of land in system s for time t (ha-year).

8. Choice of Global or National Physical Habitat Index, Or Physical Management Index for a Study

The choice of an appropriate Index is dependent upon the systems under analysis and the data available in a study. In comparative studies (i.e. studies comparing two alternative systems delivering the same functional unit), the following guidelines apply:

- If ecosystems in compared systems are in different major ecosystem categories defined at the global level, then assessment of Physical Habitat Depletion at the global level is appropriate. For example, a comparison between beef production in South America and the UK may involve conversion of tropical rainforest to grassland in South America, and continued use of cultivated land in the UK.
- If ecosystems in compared systems are in the same major ecosystem category defined at the global level, then assessment of Physical Habitat Depletion at the national level is appropriate provided that the ecosystems are all in the same country. For example, beef production may occur in the UK on improved grassland or unimproved neutral grassland.
- If ecosystems in compared systems are in the same ecosystem category defined at the national level, then assessment of Physical Habitat Depletion using PMFs is appropriate. For example,
beef production may occur in the UK on improved grassland but using either farmland divided by hedges into small fields with many additional features, or large fields divided by wire fences with no additional features.

In analysis of a single system, the purpose is to identify “hot spots” in the life cycle, and strategies for improvement of the system’s environmental performance. Therefore, here again there is a comparative aspect to the assessment (although the comparison is within one product’s life cycle). Therefore, the same guidelines apply here as outlined above for comparative studies.

9. Discussion and Conclusions

Biodiversity is a complex issue but that is not a legitimate reason for excluding it from assessment in LCA. In this chapter, I have discussed alternative approaches to measurement of biodiversity and examined how it is currently assessed in LCA methodology. This has shown that, although impacts on biodiversity are partially addressed through existing Impact Assessment categories, the impact of physical habitat maintenance and/or change has not been adequately addressed. Therefore I have developed a method for assessing physical habitat maintenance and change that makes maximum use of the data currently available, and which is flexible enough to be adapted to the purpose and data available for any LCA study. I have shown how the approach can be operationalised at global and national levels, and within any one type of ecosystem. Operationalisation of the Indices at the global and national levels requires further research effort to refine my initial estimates of the relative important of the different indicators, and of the magnitude of the different indicators. Within any one type of ecosystem, operationalisation requires further development of appropriate indicators for management practices.

However, the discussion has also highlighted a number of possible criticisms with this method:

1. The method does not account for the role of human values in assessing the relative importance of different ecosystems and species (Section 2).
2. There is inevitable uncertainty in measuring at least two of the indicators (number of species and number of rare species), and weighting the different indicators used in compilation of Physical Management Factors. This uncertainty can be reduced by consultation with experts in the field, but cannot be removed because of the limits to human knowledge. However, use of the method
developed in the chapter does not make these uncertainties transparent to users of the LCA results.

3. By quantitatively valuing each ecosystem for its contribution to biodiversity and the land area it occupies in a system, the implicit assumption is that trade-offs can be made between different systems under analysis. This may not be an acceptable assumption for some users of LCA results.

These three criticisms are valid not just for assessment of biodiversity in LCA but also for assessment of other types of impacts in LCA. The first point is important because if users do not agree with the basis for assessment in an LCA they will not regard LCA as a legitimate form of analysis. With respect to biodiversity, maybe it is worth asking whether this impact should be assessed in terms of threats to “furry animals.”

The second point raises a question about the role of uncertainty in LCA, and whether all uncertainties should be made more explicit in presentation of results. For example, perhaps each Impact Assessment result should be presented as a range of values to account for unknown and/or uncertain data. Alternatively, it may be asked whether unquantified “expert judgement” may actually provide a better assessment of the impact than quantitative methods.

The assumption about trade-offs (point 3 above) is a fundamental one, and a basic premise upon which the rest of LCA methodology has been developed (see Chapter VII, Section 5.2). However, while some people are prepared to accept trade-offs between different types of pollutants, they may find it more difficult to accept trade-offs between land areas in different ecosystems. For example, consider the four systems (A to D) in Table 8, and assume that they require different land areas for delivery of the same functional unit as shown in Table 9. The table shows that, using equation (1) from Section 7, the order of preference in the Physical Habitat Depletion category is System A (most preferable), System C, System D and then System B (least preferable). Using equation (2) from Section 7, the order of preference is System A (most preferable), Systems B and C, and then System D (least preferable). The results using equation (2) demonstrate clearly that there is a point at which larger land areas of ecosystems with high biodiversity value become equally desirable as smaller land areas of ecosystems with low biodiversity value. In this particular example, it is when forested land takes up ten times as much area as urban land, i.e. the difference in land area requirement between Systems B and C. Whether this is an acceptable trade-off depends upon basic values and attitudes towards decision-making among stakeholders but, by choice of a particular equation during
development of the LCA methodology, in practice the person developing the LCA method integrates a specific set of values into the assessment.

Table 9. Example of Trade-Offs Between Physical Habitat Values and Land Areas In Assessment Methods for Physical Habitat Depletion of Different Systems

<table>
<thead>
<tr>
<th>Land Use At Beginning of Study (PHF, Value)</th>
<th>Land Use At End of Study (PHFf Value)</th>
<th>Land Area Requirement (ha)</th>
<th>Physical Habitat Depletion Result Using Equation (1)</th>
<th>Physical Habitat Depletion Result Using Equation (2)</th>
</tr>
</thead>
<tbody>
<tr>
<td>System A: Urban (1) Forest (0)</td>
<td>Forest (0)</td>
<td>10,000</td>
<td>0</td>
<td>10</td>
</tr>
<tr>
<td>System B: Forest (0) Forest (0)</td>
<td>Forest (0)</td>
<td>10,000</td>
<td>3,300</td>
<td>100</td>
</tr>
<tr>
<td>System C: Urban (1) Urban (1)</td>
<td>Urban (1)</td>
<td>1,000</td>
<td>660</td>
<td>100</td>
</tr>
<tr>
<td>System D: Forest (0) Urban (1)</td>
<td>Urban (1)</td>
<td>1,000</td>
<td>1,000</td>
<td>1000</td>
</tr>
</tbody>
</table>

I return to all three of these criticisms in Chapter VII of this thesis.

References


CHAPTER V
INCORPORATION OF SOIL QUANTITY AND QUALITY INTO LCA

"The way we exercise our stewardship of soil will be of critical importance in determining whether sustainable development can be achieved." (Royal Commission on Environmental Pollution, 1996, p.3)

1. Introduction

The idea of assessing soil quantity and quality in LCA was introduced in Chapter III. It is based on the assumption that soil can be treated as an ancillary item in LCAs of agricultural production. Therefore, changes in its quantity and quality as a result of the activities under analysis are relevant for consideration.

Factors that affect the soil are listed in Table 1. The table also shows whether a change in each factor affects future agricultural productivity, availability of resources, biodiversity and/or human health. These are four of the five safeguard subjects defined under the EPS system (Steen and Ryding, 1992); the fifth safeguard subject, aesthetic values, is not considered here as it is somewhat different from the usual types of issues assessed in LCA (see comments about landscape degradation in Chapter II, Section 4.2.3). Furthermore, barring extreme impacts such as desertification, it is difficult to see how soil quantity and quality contribute directly to aesthetic values. These four safeguard subjects are used here on the assumption that they are suitable endpoints for assessing environmental impacts in LCA, and are complementary to those listed in the SETAC Code of Practice for LCA (resource depletion, ecological health and human health) (Consoli et al., 1993, p.5).

In the context of LCA, we are concerned with assessing the effects of changes in these factors due to the activities under analysis. Therefore, the next question to ask is the extent to which changes in each of these factors are already assessed in current Impact Assessment methodology (assumed to be the Problem-Oriented method). For these changes, there is no need to develop any additional Impact Assessment methodology. This leads to the following observations:

- Human health impacts of pesticides and heavy metals are already assessed under the "Human Toxicity" category (see Chapter II, Section 4.2.2).
- Biodiversity impacts of the various factors are already assessed under the Pollution group of Impact Assessment categories and "Physical Ecosystem Degradation" (as defined in Chapter IV).

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Table 1. Factors Affecting Soil Quantity and Quality

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Living organisms</td>
<td>Weed seeds</td>
<td>✓</td>
<td>x</td>
<td>✓</td>
<td>x</td>
</tr>
<tr>
<td></td>
<td>Micro- and meso-organisms</td>
<td>x¹</td>
<td>x</td>
<td>✓</td>
<td>x</td>
</tr>
<tr>
<td></td>
<td>Pathogens</td>
<td>✓</td>
<td>x</td>
<td>✓</td>
<td>x</td>
</tr>
<tr>
<td>Non-living matter</td>
<td>Organic matter</td>
<td>✓</td>
<td>x</td>
<td>✓</td>
<td>x</td>
</tr>
<tr>
<td></td>
<td>Water in soil</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Trace substances</td>
<td>Nutrients</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td>x</td>
</tr>
<tr>
<td></td>
<td>Heavy metals</td>
<td>x²</td>
<td>x</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td></td>
<td>Pesticide residues</td>
<td>x²</td>
<td>x</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td></td>
<td>Salts</td>
<td>✓</td>
<td>x</td>
<td>✓</td>
<td>x</td>
</tr>
<tr>
<td></td>
<td>pH of soil</td>
<td>✓</td>
<td>x</td>
<td>✓</td>
<td>x</td>
</tr>
<tr>
<td>Form of soil</td>
<td>Texture</td>
<td>✓</td>
<td>x</td>
<td>✓</td>
<td>x</td>
</tr>
<tr>
<td></td>
<td>Structure</td>
<td>✓</td>
<td>x</td>
<td>✓</td>
<td>x</td>
</tr>
<tr>
<td>Mass of soil</td>
<td>Loss from erosion</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td>x</td>
</tr>
<tr>
<td></td>
<td>Addition from incorporation</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td>x</td>
</tr>
</tbody>
</table>

¹ Impacts on productivity occur via changes in availability of nutrients and soil compaction.
² No impact when levels of heavy metals and pesticide residues are within reasonable limits.

Impacts of changes in the different factors on resources and future productivity therefore remain for further consideration. These are discussed in Section 3 (resources) and Section 4 (productivity). Firstly, however, it is necessary to consider a number of other aspects related to assessment of soil quantity and quality. These are: i) the quantity of soil crossing the system boundary at the beginning and end of a study, i.e. the soil under analysis in a study; ii) allocation issues related to crop rotations; iii) the scale of analysis; and iv) the information available for a study. These are discussed in Section 2.

2. Methodological Aspects In Assessment of Soil Quantity and Quality

2.1 Soil Under Analysis

Following the logic for inclusion of soil (Chapter III, Section 4.3), the soil under analysis in a study should be soil that is “processed” and “formed” by agricultural activities. This is the furrow slice on agricultural land: the soil that is turned by a plough. It is also referred to as the “topsoil.” Its typical volume composition is shown in Figure 1. By mass, 1 m³ typically contains 250 kg water, 65 kg organic matter (dry weight), and 1,205 kg minerals (giving a total mass of 1.52 tonnes)¹. Since the

¹ This is equivalent to a bulk density of 1.27 tonnes/m³. It assumes that organic matter has a bulk density of 1.3 tonnes/m³ (dry matter) and mineral solids 2.68 tonnes/m³ (Brady, 1990, p.102-3).
furrow slice usually extends to a depth of 20 cm, a hectare-furrow slice typically weighs about 3,040 tonnes. This is, therefore, the soil whose quantity and quality is subject to assessment in LCAs of agricultural systems. An exception is subsoiling to treat compaction at a depth below 20 cm in the soil; since subsoil compaction affects the future productivity of the land, it is relevant for assessment (see Section 4.6).

**Figure 1. Soil Composition By Volume**

![Soil Composition By Volume](image)


In the discussion in this chapter, I focus on mineral rather than organic soils. Mineral soils typically contain 1 to 6% organic matter by mass, and are the predominant soil type on agricultural land. Organic soils typically contain over 50% organic matter by mass, and are usually found in wetland areas such as swamps, bogs and marshes (Brady, 1990, p.10).

### 2.2 Scale of Analysis

Scale of analysis in LCA may be concerned with time and/or geographical scales. In previous chapters, I have discussed the role of geographical scale in influencing the results of an LCA (see Chapter II, Section 4.4, and Chapter III, Section 3). Here, I am concerned with the role of time scale in LCA which inevitably in agricultural systems links to the role of crop rotations and whole system modelling approaches (Chapter III, Section 4.2).

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2. For consistency, this should be the same volume (and/or mass as applicable) of soil as that used in assessing toxicity (see Appendix II.1). In order to compare the data, the dry matter content of soil must be calculated: according to these data, it is \((1.52-0.25)= 1.27 \text{ tonnes/m}^3\) topsoil. Heijungs et al. (1992b, p.88) use a depth of 15 cm and a dry matter content of 1.2 tonnes/m$^3$ in assessing toxicity. Guinée et al. (1996, p.26) consider the "top layer" of soil (the depth is not specified). The Critical Surface-Time method uses a soil depth of 20 cm (Audsley et al., 1997, p.82). Therefore, there seems to be reasonable consistency between these approaches.
Different approaches are illustrated in Figure 2. This figure shows a simplified system consisting of four crops grown in rotation. Crop A is barley, Crop B is oilseed rape, Crop C is winter wheat, and Crop D is a three year grass/clover ley (referred to as D1, D2 and D3 for each year). Different fungicides are used by each arable crop (F1, F2 and F3), and in addition herbicides (H1, H2 and H3) are applied to control grass and broadleaved weeds. Lime is applied every six years, and repair of the drainage ditches is undertaken every twelve years.

System boundaries can be drawn for a number of alternative scenarios:

- Assessment of two crop rotations over a twelve year period (Case 1).
- Assessment of one crop rotation over a six year period (Case 2).
- Assessment of one crop over a one year period (Case 3).

In this simplified system, Case 1 represents a whole-system modelling approach, over a twelve year period, and Cases 2 and 3 are examples of "sub-system" modelling on time scales of six years and one year respectively. Case 1 involves no allocation issues, and in Cases 2 and 3 an increasing number of allocation issues require resolution. The results for Case 1 and Case 2 would be given as the total impacts associated with production of the four crops, while the results for Case 3 would be given for just one crop.

Figure 2. Sequence of Activities In Assessment of Agricultural Systems: Time Perspective

Decisions about the appropriate timescale for an LCA involving agricultural production depend upon the purpose of the study. In some studies, such as those informing longer-term agricultural
policymaking, it may be appropriate to use the Case 1 or Case 2 timescales. In other studies, the Case 3 timescale may give results that are more relevant to the purpose of the study. For example, it is more appropriate to use a Case 3 timescale when comparing the relative environmental impacts of purchasing alternative foodstuffs in the supermarket because the consumer is choosing between products A and B rather than crop rotations X and Y. It is also worth noting that an alternative – and equivalent – approach is to consider a larger area producing a range of outputs in any one year, as illustrated in Figure 3. Here, Cases 1 or 2 may represent a typical farm or a region (for example, Devon or East Anglia) while Case 3 represents a single crop (as in Figure 2).

In this chapter, I assume that LCA methodology should be developed for analysis at all these different scales, and so I discuss resolution of allocation issues for Case 2 and Case 3 studies in Section 2.3 below.

Figure 3. Sequence of Activities In Assessment of Agricultural Systems: Area Perspective

2.3 The Allocation Issue In Assessment of Soil Quantity and Quality

Allocation is a central consideration in assessing soil quantity and quality because soil is an ancillary item shaped by cultivation of previous crops as well as the crop under consideration. The types of activities that may raise allocation issues in assessing soil are illustrated in Figure 2 and concern:
• Activities specific to different crops: fungicides F1, F2 and F3
  
  These are specific to fungal infections occurring in each crop. Therefore, it is reasonable to allocate the burdens associated with use of these substances to the crop under cultivation at the time of application of the fungicide. In other words, for these activities there is no allocation issue requiring resolution.

• Activities occurring less frequently than once per crop and with equal benefits for more than one crop in the rotation: herbicide H2

  This is a herbicide that is applied to Crop B but equally benefits Crop C. Therefore, the burdens associated with its use should be allocated among both these crops according to their respective area-time (ha-year) requirements. In this case, half of the burdens associated with application of herbicide H2 to one hectare of Crop B are allocated to one hectare of Crop C, assuming each crop is cultivated for the same time period.

• Activities occurring less frequently than once per crop and with variable benefits for some or all crops in the rotation: lime

  This is applied every six years in order to stop the soil becoming acidic. Its use benefits all crops in the rotation. However, some crops are more responsive to changes in the soil’s pH than others, and so obtain a greater benefit from use of lime. For example, barley is sensitive to acidic soils and so liming usually takes place before the barley crop in a rotation (Jellings and Fuller, 1995, p.162). Therefore, the burdens associated with use of lime should be allocated among the crops in a rotation according to their sensitivity to acidic pH (see Section 4.5 below).

• Activities occurring less frequently than each crop rotation and with equal benefits for all crops in the rotation: drainage ditches

  Repair of the drainage ditches is undertaken every twelve years. Each crop in each crop rotation benefits from maintenance of the drainage ditches. Therefore, the burdens associated with these activities are allocated among all the crops in both rotations according to their respective area-time (ha-year) requirements.

• Activities occurring less frequently than each crop rotation and with variable benefits for some or all crops in the rotation

  Although not illustrated in this example, some activities undertaken infrequently may have variable benefits for different crops over several crop rotations. For example, flooding and drainage of saline soils to reduce their salinity may benefit some crops more than others, depending upon their tolerance of salinity. In these cases, as for the minimum pH tolerance example above, the burdens associated with the activities should be allocated among the crops in relation to their tolerance of salinity (see Section 4.4 below).
Of course, this example is a simplification in that the five types of activities listed are not distinct and separate categories in reality. For example, well-drained land may benefit some crops more than others in different rotations. However, given that all models are simplifications of reality, conceptualising allocation issues in this way gives insights for more realistic allocation of burdens affecting soil quantity and quality.

The example also illustrates that allocation remains an issue even when system boundaries are extended to include whole crop rotations. It is therefore worthwhile to develop an approach that can be applied at timescales varying from a crop, to a crop rotation, to several crop rotations. Hence, detailed application of the approach to allocation illustrated above for Case 2 and Case 3 studies is discussed in Sections 3 and 4 below.

2.4 Information Available For a Study

In Chapter IV, I discussed assessment of biodiversity when different amounts of data are available for a study. The question of data availability is also relevant in assessing soil quantity and quality for crops in rotations. Obviously the Case 1 and Case 2 studies shown in Figure 2 are only possible when data are available for whole crop rotations. However, Case 3 studies also require data on other crops in the rotation for realistic modelling, as demonstrated in Section 2.3 for assessing use of the herbicide H2 and lime. However, sometimes data are not available on other crops in a rotation. In these cases, a range or set of alternative scenarios should be used to represent the most realistic alternatives in a crop rotation. Thus, for example, assessment of Crop B might include two alternative scenarios where the burdens of using herbicide H2 are allocated 100% and 50% to Crop B. In other words, if it is not known that the benefits of using herbicide H2 are shared by one subsequent crop (Crop C in this example), alternative scenarios are constructed that involve application of the herbicide in a crop rotation every one or two years. In this way, the results of an LCA are not biased by the lack of data available for the study.

3. Accounting For Resource Depletion Aspects of Changes In Soil Quantity and Quality

The following sections (Sections 3.1 to 3.3) discuss the background and approaches for assessing changes in soil mass, water and nutrients.
3.1 Mass of Soil

Changes in the soil’s mass between the beginning and end of a cultivation sequence under analysis are dependent upon a number of factors, as illustrated in Figure 4. The diagram shows that the soil’s mass is increased by addition of organic matter and lime, and the weathering of underlying rocks. It is decreased by the erosion of soil, leaching, and the activity of micro-organisms. In addition, water enters and leaves the soil (see Section 3.2).

Figure 4. Factors Affecting Soil Quantity

Considering changes in organic matter (OM) firstly, one might assume from this diagram that, in the absence of soil erosion, the mass of soil on agricultural land will tend to increase over time due to continual addition of OM. However, this tends not to be the case because of a feedback mechanism mediated largely by the micro-organisms. As OM is added, the numbers of these organisms increase rapidly, OM is broken down, carbon dioxide and water are released, and simpler organic compounds are formed. Once the easily digested OM is gone, the numbers of organisms decrease, leaving a heterogeneous mass of organic compounds that are together referred to as humus.

As well as being dependent upon the addition of OM, the activity of these micro-organisms is also controlled by the amount of available nitrogen in the soil. As they multiply, the organisms take up available nitrate from the soil, and it only begins to be released again once the numbers of organisms

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3 These micro-organisms include micro-fauna (nematodes, protozoa and rotifers) and micro-flora (algae, fungi, actinomycetes and bacteria). Between 60 and 80% of the total soil metabolism is due to the micro-flora (Brady, 1990, p.257). Meso-organisms, such as earthworms, mites and woodlice, also contribute to the breakdown of organic matter.

4 In waterlogged conditions, methane may be formed in addition to carbon dioxide because aerobic micro-organisms are inhibited and methanogenic bacteria become active (Brady and Weil, 1996, p.372). This is particularly a concern in rice paddies, since methane is a potent global warming gas.

5 The majority of the organic matter (60-80%) is completely decomposed and released as carbon dioxide within one year of incorporation. The remainder is decomposed more slowly, and indeed some studies have shown that humus contains organic carbon thousands of years old! (Brady and Weil, 1996, p.377-9).
start to decline, and their decay results in the release of nitrates back into the soil. These relationships are illustrated in Figure 5.

**Figure 5. Relationship Between Organic Matter, Micro-Organism Population and Nitrate Level of Soil**

![Graph showing the relationship between organic matter added to soil, micro-organism population, and soluble nitrogen in soil over time.]

Source: adapted from Brady and Weil, 1996, p.371 and 376 (assuming a high carbon:nitrogen ratio in the organic matter added to the soil).

Actual changes in the OM level in the soil depend, therefore, upon the activities of micro-organisms, and are affected by a range of activity-dependent factors:

- Types of crops and rotations over time: crops add OM to the soil, and nitrogen-fixing crops release nitrogen to the soil. Typical levels of OM under different types of vegetation are given in Table 2, and the amounts added by the roots of various crops are given in Table 3; in addition, above-ground OM may be incorporated into the soil after harvesting (for example, a typical value for a crop of winter wheat is 4 tonnes above-ground, dry matter per hectare (Audsley, pers.comm.)).
- Addition of other OM such as manure.
- Addition of fertilisers: they stimulate plant growth and hence production of more OM.
- Fallow periods in rotation: this allows OM levels in the soil to increase because no OM is removed from the land as harvested crops.
- Drainage of soils: poorly drained soils tend to have higher OM levels because the poor aeration and high moisture levels inhibit microbial activity (Brady, 1990, p.297). Brady cites examples of a difference of approximately 2% in OM levels (as a percentage of the total soil’s mass) at the
surface between a well drained and poorly drained soil in Minnesota, and 9% between a well drained and poorly drained soil in Indiana.

- Type of tillage: minimising tillage tends to lead to higher OM levels because crop residues are left at or near the surface, where they are subject to less microbial activity. Brady (1990, p.299) cites an example of a 0.2% difference in OM levels (as a percentage of the total soil’s mass) five centimetres below the soil surface between conventional and conservation tillage.

Table 2. Levels of Organic Matter In Soil Under Different Types of Vegetation

<table>
<thead>
<tr>
<th>Type of Vegetation</th>
<th>Organic Matter (% In Soil)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Arable, straw/residues removed</td>
<td>3 - 5</td>
</tr>
<tr>
<td>Arable, straw/residues incorporated</td>
<td>4 - 7</td>
</tr>
<tr>
<td>Grass/arable rotation</td>
<td>5 - 10</td>
</tr>
<tr>
<td>Permanent grass</td>
<td>10 - 20</td>
</tr>
<tr>
<td>Woodland</td>
<td>15 - 30</td>
</tr>
</tbody>
</table>

Source: Parkinson, 1995, p.110

Table 3. Quantities of Organic Matter Added To Soil By Roots of Crops

<table>
<thead>
<tr>
<th>Crop</th>
<th>Mass of Dry Roots In Top 20 cm of Soil (kg/ha)</th>
<th>% Increase In Soil Due To Organic Matter In Top 20 cm of Soil Before Decomposition</th>
</tr>
</thead>
<tbody>
<tr>
<td>1-year grass ley</td>
<td>4,500 - 5,500</td>
<td>0.2 - 0.3</td>
</tr>
<tr>
<td>3-year grass ley</td>
<td>6,500 - 9,500</td>
<td>0.3 - 0.5</td>
</tr>
<tr>
<td>Winter cereals</td>
<td>2,500</td>
<td>0.1</td>
</tr>
<tr>
<td>Spring cereals</td>
<td>1,450</td>
<td>Less than 0.1</td>
</tr>
<tr>
<td>Sugar beet</td>
<td>550</td>
<td>Less than 0.1</td>
</tr>
<tr>
<td>Potatoes</td>
<td>280</td>
<td>Less than 0.1</td>
</tr>
<tr>
<td>Red clover</td>
<td>2,200</td>
<td>0.1</td>
</tr>
</tbody>
</table>

Source: Davies et al., 1993, p.198.

Longer-term changes in mean OM levels (as opposed to short-term changes such as those shown in Table 3 for the immediate increase in OM due to crop roots) take place relatively slowly due to the feedback mechanisms described above. This has been demonstrated in a famous, long-running experiment at the University of Illinois where the “Morrow plots” have been cultivated since 1876 and measurements of soil organic carbon taken since 1903. [Soil organic carbon is the most usual way of measuring the OM content of soil; the approximate amount of OM is calculated by multiplying this value by 1.7 (Brady, 1990, p.294). This is based on the fact that the carbon content of OM is 58% (Brady and Weil, 1996, p.372).] In a worst-case scenario, where a plot has been under a continuous corn crop since 1876, without any addition of manure or synthetic fertilisers, the level of soil carbon decreased from about 48 tonnes/ha in 1903 to about 32 tonnes/ha in 1958 when it levelled off. This is equivalent to a loss of approximately 500 kg OM/ha/year over this time period, i.e. less

6 Conservation tillage refers to agricultural practices that minimise disturbances of the soil, and keep organic residues at or near the soil surface (Brady and Weil, 1996, p.124).
than 0.016% of the total topsoil per year. However, when lime and synthetic fertilisers were added to the plot, the organic carbon content changed from about 32 tonnes/ha to 38 tonnes/ha in 18 years (Brady, 1990, p.299; Brady and Weil, 1996, p.388). This is equivalent to an increase of approximately 570 kg OM/ha/year, i.e. an increase of 0.019% in the mass of topsoil per year. In other words, changes per year in the mass of soil under arable cultivation due to its OM content are likely to be small.

However, changes from arable to grass cultivation have a more marked effect on OM levels. Table 4 shows the results of a 12 year experiment undertaken by ADAS. It shows the effect of rotational grass (leys) on OM levels in topsoils at five farms in the UK. It can be seen that the OM levels in the different soils increased between 0.36 and 1.0% after nine years under grass. At the upper end of this range, then, the change in OM levels may contribute to a >0.1% annual increase in soil mass. However, interestingly the experiments also showed that much of the additional OM in the soil under grass was quickly lost when the land returned to arable cultivation. The exception was the sandy soil at Gleadthorpe. This soil had a low OM level at the beginning of the trial and retained most of its OM when it was returned to arable cultivation (final column in Table 4) (Davies et al., 1993, p.199).

Table 4. ADAS Experiment To Demonstrate Effect of Leys On Organic Matter Level In Soils

<table>
<thead>
<tr>
<th>Location</th>
<th>Type of Soil</th>
<th>% Organic Matter In Topsoils</th>
<th>At Start of Trial</th>
<th>After 3-Year Ley</th>
<th>After 9-Year Ley</th>
<th>After 3-Year Arable Following 9-Year Ley</th>
</tr>
</thead>
<tbody>
<tr>
<td>Boxworth, Cambs.</td>
<td>Clay loam</td>
<td></td>
<td>3.1</td>
<td>+ 0.1</td>
<td>+ 0.5</td>
<td>+ 0.0</td>
</tr>
<tr>
<td>Bridgets, Hants.</td>
<td>Chalk loam</td>
<td></td>
<td>4.4</td>
<td>+ 0.1</td>
<td>+ 0.6</td>
<td>+ 0.1</td>
</tr>
<tr>
<td>Gleadthorpe, Notts.</td>
<td>Sand</td>
<td></td>
<td>1.6</td>
<td>+ 0.1</td>
<td>+ 0.36</td>
<td>+ 0.3</td>
</tr>
<tr>
<td>Rosemaund, Hereford</td>
<td>Silt loam</td>
<td></td>
<td>3.5</td>
<td>+ 0.2</td>
<td>+ 1.0</td>
<td>+ 0.0</td>
</tr>
<tr>
<td>High Mowthorpe, Yorks.</td>
<td>Chalk loam</td>
<td></td>
<td>3.8</td>
<td>+ 0.4</td>
<td>+ 1.0</td>
<td>+ 0.5</td>
</tr>
</tbody>
</table>

Source: Davies et al., 1993, p.199.

Lime is typically added to croplands at a rate of no more than 9 tonnes/ha every four years (Brady, 1990, p.240). Over this time, the limestone is gradually leached and removed in the harvested crop. Therefore, on average it can be assumed that use of limestone does not increase the mass of soil.

For weathering of underlying rocks, Pimentel et al. (1995) state that the average rate of soil formation is 1 tonne/hectare/year (i.e. this is the rate of conversion of parent material into soil). This is equivalent to an increase of 0.03% in the mass of topsoil per year. In contrast, the mass of soil lost
by erosion can be much larger than the rate of formation. In Asia, Africa and South America, rates of soil erosion average 30 to 40 tonnes/ha/year; and in the United States and Europe they average 17 tonnes/ha/year (Pimentel et al., 1995). In intensively-farmed parts of central and eastern England, fields commonly lose 20 tonnes soil/ha/year soil (Arden-Clarke and Hodges, 1987) and rates as high as 100 tonnes/ha/year have been measured (Silsoe Review, 1993). This is equivalent to a decrease of between 0.6 and 3.3% in the mass of topsoil per year.

Method of assessment

The discussion above has shown that when changes in soil mass over several years are normalised to one year, the only factor likely to make a difference greater than 0.1% per annum change in soil mass is soil erosion. The exception is changes between arable and grass cultivation, where the associated increase or decrease in soil mass due to changes in the OM level may be of the same order of magnitude as for soil erosion over a one year period. It therefore seems pragmatic to assess changes in soil mass as a function of soil erosion unless it is known that previous cultivations involved an arable/grass rotation, in which case an allowance can be made - if necessary - for changes in OM levels.

Quantities of soil eroded can be:

1. Measured directly.
2. Calculated for a particular site using the Revised Universal Soil Loss Equation or other model (Morgan, 1995, p.63-83) based on factors such as slope gradient, cover and management, and erosion control practices.
3. Estimated from existing data on quantities of soil eroded in different areas.

The method chosen will depend upon the data available for a study. However, as discussed in Chapter II, Section 4.4, the aim should be to use data that are as realistic as possible; i.e. the methods listed above should be regarded as a hierarchy of approaches, Method 1 being the most preferable and Method 3 the least preferable approach.

7 From the data in Table 4, the increase in OM in the topsoil over nine years ranged from 14% (Bridgets) to 29% (Rosemaund). Assuming 1 m$^3$ soil contains 65 kg OM (Section 2.1), this is equivalent to a 0.6 to 1.2% increase in soil mass over the nine years, and a 0.07-0.14% annual increase in soil mass.
Once the actual or estimated mass of eroded soil has been obtained, resource depletion can be assessed using the method of Lindfors et al. (1995) (see Chapter II, Section 4.2.1). A static reserve life for soil is calculated as:

\[
\text{Soil static reserve life} = \frac{R}{E}
\]

where \( R \) = Global reserves of agricultural soil (i.e. total topsoil in world)

\( E \) = Current annual global net loss of soil mass by erosion.

The current annual global loss of soil mass by erosion from agricultural land is \( 50 \times 10^9 \) tonnes (Pimentel et al., 1995), a hectare-furrow slice weighs 3,040 tonnes, and the total area of agricultural land worldwide is between \( 1.5 \times 10^9 \) hectares (Royal Commission, 1996, p.4) and \( 1.8 \times 10^9 \) hectares (Graetz, 1994, p.134). The average rate of soil formation is 1 tonne/ha/year, i.e. \( 1.5 \times 10^9 \) tonnes/year worldwide. The soil static reserve life follows as:

\[
\frac{1.5 \times 10^9 \times 3040}{(50 \times 10^9) - (1.5 \times 10^9)} \quad \text{to} \quad \frac{1.8 \times 10^9 \times 3040}{(50 \times 10^9) - (1.8 \times 10^9)}
\]

This gives a static reserve life of between 94 and 114 years (average 104 years). This is intermediate between manganese and chromium with static reserve lives of 95 and 105 years respectively (Lindfors et al., 1995, p.166). Of course, the use of this method for assessment of soil depletion is crude because soil is not globally available in the same way as resources such as copper, lead or fossil fuels: soil is not generally transported around the world for use in areas where there is a shortage of this resource (apart from on a small scale in, for example, adding topsoil to the gardens of new houses). However, the calculation does indicate that soil depletion should be ranked alongside concerns about depletion of other resources.

An alternative method of assessing soil depletion estimates the time required to renew this resource (see Chapter II, Section 4.2.1). Using this method, soil depletion is also an important resource depletion issue because of the relatively long time taken for its renewal (see above).

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8 Net loss of soil by erosion is analogous to annual extraction rates for other resources in this method because its use in this equation gives the number of years that the global soil reserve (i.e. the topsoil) will last at current rates of erosion.
3.2 Water In Topsoil

The mass of water in the topsoil is determined by the soil’s structure and associated drainage patterns, composition, rainfall, and withdrawal of groundwater by cultivated crops. Changes in water content as a result of changes in the soil’s structure and associated drainage patterns are accounted for by assessing soil structure (see Section 4.6). The soil’s composition does not vary between years (because it is determined by the underlying substrate) apart from the level of OM, and assessment of OM is discussed in Section 4.3. Rainfall is a site-dependent aspect and so is implicitly assessed as part of choice of location for the system under analysis. Withdrawal of groundwater by cultivated crops is relevant and such water use should be assessed using one of the methods put forward for resource depletion in Chapter III, Section 5.2.

Therefore, it is concluded that this is not a relevant factor for separate assessment in LCA.

3.3 Nutrients

In all agricultural systems, a basic prerequisite for production is an adequate supply of nutrients essential to plant growth. The main nutrient requirements are nitrogen, phosphorus and potassium. In conventional and integrated systems, these nutrients are supplied by regular application of synthetic fertilisers. In organic systems, the nutrients are supplied by growing nitrogen-fixing crops in the rotation and regular applications of materials such as animal manures, natural phosphate rock, natural rock potash, wood ash, and dried seaweed meal.

In organic systems, the supply of nutrients from livestock-derived products (such as manure and bone meal) raises a further question about the origin of nutrients in these livestock products. The answer to this question depends upon the legislative guidelines in the area under agricultural production. In the UK, organic farmers can use livestock products from more intensive farming systems, although use is “restricted” and a number of criteria must be met (see The Soil Association, 1996, p.19-20). This implies that the ultimate origin of nutrients in organic farming systems may include synthetic fertilisers, albeit these nutrients are one step further down the production chain, having passed through (or into) livestock prior to use in the organic system. However, in Switzerland, for example, organic farmers are not allowed to use livestock products other than those from other organic farms. In this case, the origin of nutrients must be the types of materials listed in the previous paragraph.
This suggests that changes in nutrient levels in the soil can be assessed as an increased or decreased requirement by future crops, since agricultural soils must be supplemented by regular inputs of nutrients in order to remain productive.

**Method of assessment**

It seems reasonable to account for a decrease of nutrients in the soil due to the crop under analysis as an increased requirement by future crops. In the same way, an increase in nutrients in the soil implies a decreased requirement by future crops. This can be assessed by adding or subtracting the burdens associated with production, delivery and spreading of the appropriate quantity of nutrients. For intensive and integrated farming systems, it can be assumed that these nutrients are supplied as synthetic fertilisers. However, for organic systems the assumption about the source of nutrients will depend upon country-specific conditions, as discussed above. A practical application of this approach is given in Chapter VI (for use of manure on an organic farm in Switzerland).

**4. Accounting For Impacts On Productivity of Changes In Soil Quantity and Quality**

In Table 1, most of the factors were identified as affecting the soil's future productivity. However, since assessment methods for nutrients and the quantity of soils were developed in Section 3, they are not considered in this section. Also, water in the topsoil is excluded from this discussion since it was concluded in Section 3.2 that it is not relevant for inclusion. However, it is worth noting in passing that soil erosion can have a particularly significant impact on future productivity. Pimentel *et al.* (1995) calculated that average soil erosion in the United States (17 tonnes/ha/year) reduces corn yields by 8% over one year, and 20% over a 20 year period. This is mainly due to increased water runoff from the eroded land⁹ and differential loss of nutrients in the eroded soil¹⁰. [However, others have suggested that this is an overestimate of yield reductions due to extrapolation of data from a small number of sampling points (Royal Commission, 1996, p.5)].

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⁹Moderately eroded soils absorb just 7 to 44% of total rainfall, and absorption rates less than 70% can result in significant water shortages for crops depending upon the area under consideration (Pimentel *et al.*, 1995).

Erosion affects the infiltration rate of water because it reduces the mass of soil available to absorb water, and differentially removes OM which has a high water-holding capacity. [Eroded soil typically contains 1.3 to 5 times more OM than soil left in the field (Pimentel *et al.*, 1995).]

¹⁰Eroded soil typically contains three times more nutrients than soil left in the field, and so this loss of nutrients should be included when modelling use of nutrients in LCA (see Appendix VI.3).
In the following sections, I discuss assessment of impacts on future productivity caused by changes in numbers of weeds and weed seeds; pathogens; the level of OM; salts; the soil's pH; and the form of the soil.

4.1 Weeds and Weed Seeds

Many non-chemical techniques have developed for weed control over the centuries (called "cultural" control as opposed to chemical control). They include:

- Rotation of crops
- Disturbance of the soil during cultivation
- Manual destruction of weeds
- Preventive activities.

**Rotation** of crops means that the local environment of a field changes from year to year. Different cultivation methods for each crop (such as timing of sowing and harvesting, preparation of seedbed, and so on), and variation in the seasonal presence of the crop canopy, mean that the competitive environment for weeds changes each year. As a result, no one weed species has the opportunity to become established and dominant (Lockhart *et al.*, 1982, p.38). Furthermore, weed suppressing crops (for example, grass leys) and weed susceptible crops (for example, cereals) can be alternated in a rotation in order to control weeds (Moule, 1995, p.257).

Several strategies fall under the category of **disturbance of the soil during cultivation**:

- Preparation of false seedbeds ("stale seedbed" techniques): the seedbed is prepared early and weed seeds are allowed to germinate. They are then killed by harrowing\(^\text{11}\). This process may be repeated several times before sowing of the crop seed.
- Stubble cleaning: this is carried out in the same way as for false seedbeds, and works particularly well on autumn-germinating weeds such as blackgrass (*Alopecurus myosuroides*) and sterile brome (*Bromus sterilis*) (Moule, 1995, p.257).
- Fallows: the soil is broken up with a rotary cultivator to encourage weeds to grow. When they reach a certain height, the land is again cultivated to kill the weeds. The process is repeated several times to exhaust the weeds (Moule, 1995, p.258).

\(^\text{11}\) A harrow consists of vertical spikes on a metal frame that are dragged through the topsoil, dislodging seedlings.
Manual destruction of weeds can be undertaken by hoeing between crop rows or by destruction with a heat source ("flame weeding"). Previously, stubble burning also contributed to control of weeds; it could halve the subsequent population of blackgrass and reduce by a third the quantity of wild oat seeds (Tottman et al., 1982, p.275). However, this practice was banned in the UK in the early 1990s.

As well as direct actions to control weeds, preventive activities for weed control are also important. They vary from keeping crop seeds free from weed seeds, to avoiding use of manure and straw bedding contaminated with weed seeds, to controlling weeds in field margins, to cleaning machinery from a weedy field before it is used in a "clean" field (Tottman et al., 1982, p.46-48, 273-4).

From the 1940s, these strategies have been supplemented and, in many cases, replaced by use of herbicides. As a result, there has been a decrease in the number of tillage operations required per crop, and a decline in the diversity of crop rotations. Instead, herbicides may be applied at one or more stages in a crop's cultivation:

- Pre-sowing in autumn: herbicides such as glyphosate and amitrole are applied to kill perennial grass weed seeds.
- Pre-emergence of autumn-sown crops: herbicides such as isoproturon and cyanazine are applied for control of annual grasses and broadleaved weeds.
- Post-emergence of crop in spring: herbicides such as clodinafop-propargyl and cyanazine are applied to control wild oats and broadleaved weeds (Moule, 1995, p.258).

It can be seen that many of these strategies for weed control take place on an annual basis. Therefore, these activities do not raise any allocation issues. However, in some cases activities may be undertaken less frequently; examples include occasional use of herbicides (as illustrated for herbicide H2 in the example in Figure 2) and fallow periods in a crop rotation. In these cases, allocation is a relevant issue.

**Method of assessment**

In all three Cases illustrated in Figure 2, assessment requires an initial assumption that the average number of weed seeds does not change between the beginning and end of a crop rotation. In other words, it is assumed that the agricultural system under analysis is part of a sustainable system of crop
production as regards control of weed populations. This is a reasonable assumption in areas where agriculture has been practised over a number of years, i.e. in most of Europe. [If the initial assumption about the average number of weed seeds remaining constant over a crop rotation is not made, this implies that the productivity of future crops will be compromised or enhanced due to the increased or decreased number of weed seeds. Analysis will then require some further type of assessment to account for this future loss or gain in productivity, perhaps similarly to the method proposed for assessment of nutrient depletion in Section 3.3.]

In Case 1 and Case 2, then, analysis can proceed without any need to consider allocation issues. However, in Case 3 further consideration must be given as to whether activities to control weeds occur annually or less frequently, the latter implying benefits for more than one crop and therefore a need for allocation. If this is the case, then, as discussed in Section 2.3 of this chapter, the burdens associated with these activities should be allocated among the affected crops in the rotation.

4.2 Pathogens

Pathogens may build up in the soil when a crop is grown repeatedly on the same land, crop rotations are too short to kill the organisms before the same crop is cultivated again, or the weather and other environmental factors are favourable to particular organisms. Pathogens are controlled by appropriate crop rotations and, since the 1940s, the use of pesticides ranging from generic insecticides and fungicides to the more specialised bactericides, molluscicides (for slugs and snails) and nematicides.

Pesticides may be applied at various stages of a crop’s cultivation from pre-sowing applications to the soil through to spraying the growing crop. Also, crop seeds may be treated prior to drilling with insecticides (for example, gamma-HCH or methiocarb) and/or fungicides (for example, guazatin or carboxin).

Method of assessment

Assessment of use of pesticides raises the same issues as for use of herbicides (see Section 4.1). Therefore, the same approach should be used as recommended in Section 4.1.

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12 Examples of pathogens include potato wart (Sychitrium endobioticum), clubroot (Plasmodiophora brassicae) in brassicas, and sugar beet rhizomania.
However, in particular for pathogens, the validity of an initial assumption about agricultural systems maintaining a "sustainable" level of pathogens must be carefully considered in development of scenarios for some studies. For example, the requirement to grow oilseed rape in no less than four year rotations due to soil-borne diseases was a relevant factor in a study examining the potential for biodiesel production from rapeseed oil to displace mineral oil-sourced diesel in Germany (Federal Environment Agency, 1993, p.129). Here, the potential for biodiesel production was limited by the amount of land available for cultivation of oilseed rape in a four year rotation. It was estimated that a maximum of 10% of total agricultural land could be used on average per year for oilseed rape cultivation due to rotational restrictions, and climate and soil suitability in different areas.

4.3 Organic Matter

The amount of OM in the topsoil has an important effect on its productivity, even though it is typically only about 4% of arable soils (Royal Commission, 1996, p.53). This is because:

• OM is an important source of nutrients, and it increases the cation adsorption capacity\(^{13}\) of the soil.
• It contributes to a good soil structure.
• The populations of micro-and meso-organisms are sustained by OM.
• The soil's temperature is maintained by heat released during the metabolic activities of microorganisms as they digest OM.
• OM has a good water-holding capacity.

It is therefore beneficial to maintain soil OM at "as high a level as is economically feasible" (Brady, 1990, p.301). The actual level achieved is dependent upon a number of activity- and site-dependent factors. Activity-dependent factors were listed in Section 3.1. Site-dependent factors include:

• Temperature: areas with lower mean temperatures tend to have higher OM levels in the soil. If other factors are held constant, for each 10°C decrease in mean annual temperature the average OM level in soil increases by two to three times (Brady and Weil, 1996, p.384).
• Rainfall: areas with higher rainfall tend to have higher OM levels in the soil. For example, an arable soil near the coast of Lancashire is likely to have 1-2% more OM than a similar soil in Norfolk (Davies et al., 1993, p.197).

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\(^{13}\) The cation adsorption capacity (also called the cation exchange capacity) is the total of exchangeable cations that a soil can adsorb.
• Soil texture: soils high in clay and silt tend to have higher OM contents than sandy soils. For example, in one area of the United States, the OM content changes by about 0.9% (as a percentage of the soil’s total mass) for each 10% rise in clay content of the soil (Brady, 1990, p.297). In trials, clay loam soils at ADAS Boxworth (an experimental farm operated by ADAS) had a 3% OM level after 12 years arable cultivation, whereas on lighter soils at ADAS Gleadthorpe the equivalent OM level was 1.6% (Davies et al., 1993, p.198).

**Method of assessment**

Given the discussion in Section 3.1, four approaches can be taken to assessment of OM in LCA:

1. Actual changes in the OM level of the soil are measured directly between the beginning and end of the system under analysis.
2. Models are used to estimate the change in the OM level in the soil due to the crop under analysis (Smith et al., 1997, give a guide to soil organic matter models and long-term experimental datasets).
3. All inputs of OM are credited to the system under analysis, on the assumption that any increase in the OM level of the soil can only take place if OM is added to the system. In effect, this is a precautionary approach, using total input of OM as an indicator of maintenance or improvement in the OM level of the soil.
4. The OM level of the soil is not considered in the analysis. The argument in favour of this approach is that the OM level changes by relatively small amounts with changes in inputs of OM, and tends to be dominated by site- rather than activity-dependent factors in any given arable rotation or on grassland. Hence, its effects on productivity are implicitly included in a study by choice of location for the system under analysis.

The method chosen will depend partly upon the data available for a study and partly upon the perceptions of the analyst conducting the study. Obviously, Method 1 is most preferable as it provides actual “real life” data, and Method 2 should be the second choice provided that the model used is judged to be appropriate and reliable. However, in many studies data required by these two methods are not available. The choice of Method 3 or Method 4 is then likely to be influenced by whether the analyst tends towards the “nature is fragile” or “nature is robust” approach in environmental management (see Chapter VII, Section 2.4). The relevance of peoples’ perceptions

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14 Brady and Weil (1996, p.384) state that “Climatic conditions, especially temperature and rainfall, exert a dominant influence on the amounts of organic carbon (and nitrogen) found in soils.”
about the environment in shaping environmental analysis approaches is discussed in more detail in Chapter VII.

I therefore conclude that Method 1 is most preferable, followed by Method 2, and, in the absence of the required data, either Method 3 or Method 4 can be used but the reason for the choice should be made clear in the report of a study. In presenting the results using Method 3, the inverse of the mass of OM should be used for consistency with other Impact Assessment category results (where higher values imply greater negative environmental impacts). In other words, the OM indicator for Method 3 is:

\[
\text{OM Indicator} = \frac{1}{M}
\]

where \( \text{OM Indicator} \) is the result in this Impact Assessment category for the system under analysis and \( M \) is the total mass of OM added to the soil in the system under analysis (above-ground matter, roots and other OM such as manure).

4.4 Salts

Soils in arid and semi-arid regions tend to be saline (i.e. contain salts) because the low rainfall means that salts are not flushed out of the upper soil layers. The salts are primarily chlorides and sulphates of calcium, magnesium, sodium and potassium, and include soluble as well as insoluble salts where there is poor internal drainage of the soil (Brady and Weil, 1996, p.307, 309). The source of salts is the weathering of rocks and minerals, rainfall, groundwater and irrigation waters (Brady, 1990, p.243). In humid regions, soils tend to be less saline but irrigation may increase their salinity.

Plants have varying levels of tolerance to saline soils. For example, barley and sugar beet are tolerant; wheat is moderately tolerant; and potatoes and tomatoes are sensitive to salinity. This tolerance depends on factors such as the physiological constitution of the plant, its growth pattern and rooting habits (Brady and Weil, 1996, p.316-7).

It has been found that plants are additionally sensitive to the level of sodium in the soil. Sodium affects plants through both physical and chemical pathways. Physically, sodium ions in the soil contribute to the breaking up of soil aggregates through ionic double-layer effects, causing them to repel each other. This leads to a breakdown in the soil's structure. Chemically, sodium raises the
soil’s pH\(^{15}\), interfering with the plant’s metabolism and nutrition, and having a toxic effect through the formation of bicarbonate and other anions (Brady and Weil, 1996, p.315-6). Soils with high levels of sodium are described as sodic.

Some soils share characteristics of both saline and sodic soils, and are called saline-sodic soils. They have higher levels of salts but also have high levels of sodium. If additional sodium is added in irrigation waters, the soil may begin to lose structure as described above for sodic soils.

Salinity of soils is treated by carefully managed flooding with water low in soluble salts to leach out excess salts from the soil. It must be coupled with good drainage, and so flooding may have to be preceded by installation of drainage systems. For saline-sodic and sodic soils, the level of sodium ions must first be reduced, and then excess salts are removed in the same way as for saline soils. The level of sodium ions can be reduced by mixing gypsum into the soil surface (at a rate of several tonnes per hectare); it leads to the formation of sodium sulphate (Na\(_2\)SO\(_4\)) which can then be leached out. Alternatively, elemental sulphur or sulphuric acid may be used for the same purpose (White, 1987, p.219-20).

**Method of assessment**

In areas where soil salinity is a problem, measures to control it should be included in an LCA. Hence, a proportion of the burdens associated with activities such as flooding, building and maintenance of drainage ditches, and application of gypsum, should be allocated to the crop under analysis. The proportion is determined by the time required for cultivation of the crop under analysis against the time interval between activities to control salinity. For example, if a crop is cultivated for one year and flooding to control salinity takes place every five years, the crop under analysis is allocated 20% of the total burdens of the flooding activities. This approach is equivalent to that outlined for repair of drainage ditches in Section 2.3. [Alternatively, if appropriate, some allowance may be made for the variable salt tolerance of different crops cultivated on the land; this is equivalent to the approach taken for assessing use of lime (see Section 4.5).]

\(^{15}\) The high pH is largely due to the hydrolysis of sodium carbonate (Brady and Weil, 1996, p.316).
4.5 pH of Soil

As noted in Section 4.4, agricultural soils in humid regions tend to be acidic while those in arid and semi-arid regions tend to be alkaline (Brady and Weil, 1996, p.20). In this section, I focus on soils in humid regions.

The typical pH of most cultivated soils in humid regions is in the range 5.5 to 7.5 (Fitzpatrick, 1986, p.144). It is determined by:

- The soil’s parent material
- Rainfall (because it leaches out base-forming cations from the surface layers of the soil)
- Addition of lime (which raises the soil’s pH)
- Decomposition of OM (forming organic and inorganic acids)
- Acidic deposition from the atmosphere in acid rain or particulates.

As discussed in Section 2.3, different crops have different pH requirements. For example, potatoes can tolerate a pH down to a minimum of 4.9 before productivity is affected; swedes and turnips start to be adversely affected at a pH of 5.4; wheat and rape require a pH of 5.6 or more; and beans require a minimum pH of 6.0 (Brockman, 1995, p.147). Since most humid area soils tend to be acidic, the soil’s pH typically has to be raised to maximise productivity. This is achieved by the use of agricultural limes, primarily the carbonates, oxides or hydroxides of calcium and magnesium (Brady and Weil, 1996, p.294).

Method of assessment

The burdens of changing the pH to meet the needs of the crop under analysis can be assessed as those related to production and use of agricultural lime. For example, the four crops shown in the example in Figure 2 have the following minimum pH requirements: Crop A, pH 5.9; Crop B, pH 5.6; Crop C, pH, 5.6; and Crop D, pH 4.7 (Brockman, 1995, p.147). If we assume that the pH of the soil without use of lime is 4.7, the burdens associated with use of lime are allocated among Crops A, B, C and D in the proportions 1.2 : 0.9 : 0.9 : 0.0 (calculated as the difference between the soil pH without liming and the minimum pH requirement for each crop). [This method assumes that the mass of lime required to change the soil pH by a defined amount is the same regardless of the initial soil pH. While this is not true for the full range of pH values, it is a sufficiently accurate approximation for the
pH values most common in agricultural soils of humid areas (see, for example, data given in Figure 9.23, Brady and Weil, 1996, p.298).]

4.6 Form of Soil: Texture and Structure

Soil texture refers to the proportions of different sized mineral particles in the soil. These particles may range from sand (diameter 50 μm - 2 mm) through silt (2 – 50 μm) to clay (<2 μm) (Brady and Weil, 1996, p. 101). Since the types of mineral particles are largely determined by the parent material of the soil, and are not changed by human activities, this is not a relevant factor for assessment in LCA. In effect, it is a site-dependent factor whose influence on productivity is determined by choice of location for agricultural production.

Soil structure refers to the arrangement of particles within the soil. Distinct layers can be distinguished at different depths in the soil, referred to as horizons, and these are often the subject of interest in discussions on soil structure. Ploughing and cultivation of the soil destroys these layers in the furrow slice, however, and so these horizons are not a relevant consideration in LCAs assessing cultivated soils. But the structure of the soil itself is relevant because it is partially influenced by farming activities, and can affect productivity. Here, structure is concerned with pore space and the aggregation of particles in the soil. It is affected by:

- **Soil texture**: sandy soils have less pore space than clayey soils.
- **Cations**: aggregate formation is influenced by the types of cations adsorbed by soil particles. For example, higher concentrations of sodium ions adsorbed onto soil particles do not effectively reduce the electronegativity of these particles which therefore continue to repel each other, mitigating against formation of structural aggregates. On the other hand, calcium, magnesium and aluminium ions promote aggregation of particles by forming electropositive links between the electronegative particles (Brady, 1990, p.111-2).
- **Organic matter**: organic compounds bind together mineral particles, and are the major agent promoting formation and stabilisation of aggregates (Brady and Weil, 1996, p.125).

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16 Pore space is the proportion of soil volume occupied by air and water, and aggregates (or peds) are groupings of soil particles. Pores and aggregates are important for productivity because, among other things, they affect the water- and air- holding capacity of the soil, its stability, its ability to absorb cations, and the ease with which plant roots can penetrate the soil.

17 Pore space can be divided into macro- and micro-pores. Sandy and clayey soils tend to have similar volumes of macro-pores (which are found between soil aggregates). However, sandy soils have few micro-pores, which are usually found within soil aggregates, while clayey soils have large numbers of micro-pores within each soil aggregate (Brady and Weil, 1996, p.115-7).
- **Tillage and other cultivation operations**: in the short-term, tillage can promote aggregation by mixing particles together. However, in the longer term, use of heavy farm machinery for different operations can also compact soils, breaking down soil aggregates and thereby removing some interparticle void space (Brady and Weil, 1996, p.124-9).

Soil texture and cations are not affected by farming activities and so are not a relevant factor for assessment (apart from use of lime which has been discussed in Section 4.5). In effect, their impacts on productivity are implicitly assessed through choice of location for agricultural production. Assessment of organic matter has been discussed in Section 4.3. This leaves tillage and other cultivation operations as a relevant factor for assessment. In this section, I focus on assessing compaction of the soil during cultivation because this can negatively affect future productivity, as distinct from the shorter-term mixing of particles which mainly benefits just the crop under analysis.

Compaction of soil is an issue of particular relevance for the sustainability of agricultural production because it affects yields of future crops, and subsoil compaction in particular is extremely difficult to treat once it has taken place. Yield reductions due to soil compaction of up to 20% in the year following compaction have been measured for a number of crops (Håkansson, 1990; Chamen et al., 1992a, 1992b; Chamen and Longstaff, 1995). These yield reductions may persist for a number of years or even indefinitely if the subsoil has been compacted (see, for example, Etana and Håkansson, 1994; Håkansson and Reeder, 1994), as illustrated in Figure 6. The diagram shows how soil compaction to different depths affects subsequent crop yields. The actual depth to which compaction occurs varies depending on factors such as the weight of vehicles, the soil type, and the water content of the soil. For example, Håkansson and Reeder (1994, p.284) state that “when driving a vehicle on moist, arable soil, measurable compaction may be expected to a depth of at least 30 cm at an axle load of 4 Mg, 40 cm at 6 Mg, 50 cm at 10 Mg, and 60 cm or deeper at an axle load of 15 Mg or higher”.

However, it should also be noted that a certain degree of compactness is beneficial for crops because otherwise the soil is too loose, and this optimal degree of compactness varies between crops (Arvidsson and Håkansson, 1991; Håkansson, 1990).

The negative effects of soil compaction on crop yields arise from a number of interrelated impacts. These can be traced back to a reduction in the soil’s porosity that reduces movement of water and air in the soil, and impedes the growth of the crop roots. This restricts the uptake of water and nutrients by the plants, and may also create more anaerobic conditions in the soil leading to denitrification.

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18 Note that 1 Mg is 1 tonne.
(Whalley et al., 1995; Håkansson, 1990). In addition to effects on crop yield, soil compaction also increases the risk of soil erosion and raises the energy demand for cultivation (see review by Chamen et al., 1992a; and Chamen and Longstaff, 1995).

**Figure 6. Influence of Soil Compaction On Crop Yield**

The problem of soil compaction has been recognised for a long time, and can be caused by draft animals as well as machinery. Jirlow (1958) describes a ploughing operation in South Sweden where a one-furrow plough was pulled by two horses and twelve oxen, and accompanied by eight men! However, it particularly became a problem when steam engines were introduced into agriculture in the second half of the nineteenth century; the machines had a very high mass/power ratio as they were made from iron rather than steel. As a result, cable-farming was introduced where the engine was situated at one end of the field, and the cultivation equipment was drawn across the field by cable (Soane and van Ouwerkerk, 1994b, p.4); indeed, the viability of using this method today on soils sensitive to compaction has been investigated in Sweden (Håkansson, 1990, p.235). The introduction of the internal combustion engine and use of steel subsequently improved the mass/power ratio, and led to common use of direct traction with trailed equipment on fields. Continuing concerns in the early twentieth century about the impacts of compaction on productivity were overshadowed by the productivity increases achieved with these more efficient cultivation systems. It is only more recently that interest has revived in the problem of soil compaction as part of a more general concern about soil degradation. This has stimulated research on gantry systems, where the machinery has a wide track (for example, 12 metres) on which an implement is loaded, supported between two sets of wheels. As a result, typically only half the number of wheel tracks are produced in the field for any given operation compared with a conventional tractor system (Chamen et al., 1992b). Other alternatives for reducing the risk of soil compaction include:
• Using set tramlines so that only restricted areas of the soil become compacted (Soane and Ouwerkerk, 1994b, p.17).
• Conservation tillage: minimising the number of tillage operations.
• Combining operations: for example, combining harrowing and fertilising the seedbed with sowing.
• Modification of machinery: options include lightweighting machinery and reducing tyre inflation pressure (Soane and Ouwerkerk, 1994b, p.17).

Method of assessment

There are three approaches to measurement of compaction:

1. Direct measurement.
2. Estimation of compaction using models.
3. Use of an indicator to represent the predicted compaction.

Method 1 is very expensive and unlikely to be feasible in LCA studies due to time and financial constraints. For Methods 2 and 3, a number of models have been developed and are reviewed by Kuipers and van de Zande (1994). For these models it is useful to distinguish between two components in predicting compaction:

• Compaction capability of operations
• Compactability of the soil.

This is analogous to distinguishing between the effect and fate factors in assessing toxicity in LCA (see Appendix II.1). The “effect factor” here is the potential of any operation to cause compaction in soil, and the “fate factor” modifies this potential value in accordance with various properties of the soil. Aspects considered in developing models to assess the risk of compaction are listed in Table 5.
Table 5. Aspects Assessed In Models To Predict Risk of Compaction

<table>
<thead>
<tr>
<th>Compaction Capability of Operation</th>
<th>Compactability of Soil</th>
</tr>
</thead>
<tbody>
<tr>
<td>Weight of machinery (wheel load)</td>
<td>Water content</td>
</tr>
<tr>
<td>Distance covered by machinery (rut length)</td>
<td>Clay content</td>
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<tr>
<td>Time spent in field by machinery (load time)</td>
<td></td>
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<tr>
<td>Working width of machinery</td>
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<tr>
<td>Properties of wheels:</td>
<td></td>
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<tr>
<td>- wheel width</td>
<td></td>
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<tr>
<td>- track width</td>
<td></td>
</tr>
<tr>
<td>- diameter of wheel</td>
<td></td>
</tr>
<tr>
<td>- tyre inflation pressure</td>
<td></td>
</tr>
</tbody>
</table>

After reviewing the various models, Kuipers and van de Zande (1994) suggest that the Field Load Index (FLI) “is likely to be an effective criterion for quantifying the compaction risk from field traffic on the scale of a farmer’s field” for typical field traffic. The FLI is defined as:

\[
\text{FLI} = W \times T
\]

where \( W \) = weight of vehicle plus implement (tonnes/ha)

\( T \) = field time of the vehicle (hours/ha)

This Index gives results that are in good agreement with other models for quantifying the risk of compaction from field traffic.

It is possible to go beyond this model to include soil compactability criteria, and predict specific crop yield reductions from soil compaction. For any one crop, this could be, for example (Kuipers and van de Zande, 1994, p.423):

\[
\text{YL} = 0.00154 \times \text{TI} \times \text{CF} \times \text{CC}
\]

where \( \text{YL} \) = yield loss (%)

\( \text{TI} \) = traffic intensity (tonnes-km/ha)

\( \text{CF} \) = correction factor, ranging from 0 to 1.5 depending on soil water content and tyre inflation pressure

\( \text{CC} \) = clay content (% w/w).

It is worth noting that the clay content of the soil, and its water content, may have a greater influence on the actual compaction of the soil than the different characteristics of the machinery (such as weight.
or tyre inflation pressure) (for example, see Arvidsson and Håkansson, 1991; and Soane and van Ouwerkerk, 1994b). In other words, site-dependent aspects may have a greater influence on the actual compaction of the soil than the chosen “technology” (i.e. types of machinery and operations).

However, this latter equation requires a high level of detail in data requirements. Instead, it seems sensible to use the FLI as it quantifies the risk of soil compaction using data usually available in an LCA. If more detailed data are provided on the compactability of the soil in any system under analysis, the FLI can be modified to account for this aspect. This is analogous to modification of the effect factor using a fate factor in assessing toxicity in LCA. Therefore, the Soil Compaction Indicator (SCI) can be defined as:

$$SCI = A \sum_i^n W_i \times T_i$$

where $SCI = \text{Soil Compaction Indicator value (tonnes-hours)}$

$A = \text{area (ha)}$

$i = \text{operation } i$

$W = \text{weight of vehicle plus implement for operation } i \text{ (tonnes)}$

$T = \text{field time of the vehicle for operation } i \text{ (hours/ha)}$.

5. Discussion and Conclusions

Through systematic consideration of the different factors affecting soil quantity and quality (Table 1), a restricted number of additional factors have been defined for assessment in LCAs involving agricultural production. They are:

- **Quantity of soil**: generally measured as the actual or predicted mass of eroded soil during the time period under analysis (Section 3.1), and assessed as part of the existing abiotic resource depletion impact category.

- **Organic matter**: changes in the OM level of the soil may be assessed by direct measurement, modelling or use of indicators (Section 4.3).

- **Soil compaction**: assessed using a Soil Compaction Indicator (Section 4.6).

It has also been shown that a number of activities affecting other factors must also be modelled carefully when assessing agricultural systems that are sub-systems of a larger “whole system,” i.e.
single crops or crop rotations where the larger “whole system” involves activities occurring less frequently than the time period for the system under analysis. An allowance should be made for these infrequent activities where they benefit the crop(s) under analysis. Relevant activities are:

- **Weed control practices** (Section 4.1)
- **Measures to reduce soil salinity** (Section 4.4)
- **Measures to control soil pH** (Section 4.5).

Finally, care must be taken to account for the impacts of soil erosion in studies that are additional to resource depletion. These impacts arise from export of organic matter, nutrients and heavy metals in the eroded soil, and should be assessed as part of the Impact Assessment. For example, loss of nutrients from the system in the eroded soil can be assessed as an increased requirement by future crops (Section 3.3).

In summary, then, inclusion of soil quantity and quality in LCA involves integration of soil erosion into the existing abiotic resource depletion impact category, and use of two additional impact categories (Organic Matter and Soil Compaction Indicators). Additionally, it requires careful modelling of the system under analysis to account for infrequent activities that benefit the crop(s) under analysis and for changes in soil nutrient levels.

To paraphrase the quote at the beginning of the chapter, our stewardship of the soil must be a central consideration in developing sustainable patterns of human activities. Exclusion of these aspects from LCA, therefore, limits LCA’s usefulness in contributing to the goal of sustainable development. Instead, studies should aim to include as many aspects as possible, accepting and allowing for the constraints imposed by data availability. The method put forward in this chapter facilitates this approach, and practical implementation is demonstrated in the next chapter.

**References**


AN LCA STUDY OF BREADMAKING WHEAT PRODUCTION

In a sense, the sustainability of any subsystem of the global system – be it a state, a firm, a region, or even an individual – can only be defined in terms of a sustainable global system, and cannot be meaningfully said to exist in the absence of its links to the greater whole. (Allenby, 1998, p.18)

1. Introduction

This chapter presents the methodology and results for an LCA of breadmaking wheat production from three different farming systems. An earlier version of this study was undertaken as part of an EU Concerted Action project on “Harmonisation of Environmental Life Cycle Assessment for Agriculture.” The project involved researchers from nine research institutes, and took place between 1995 and 1997. Further details can be found in Audsley et al. (1997). As part of the project, the different researchers separately undertook LCAs of wheat production, and then compared and discussed their results at two one-week workshops. This chapter, therefore, has been developed from the final methodology and results I calculated for my study, based on discussion with other participants in the EU project.

The organisation of the chapter follows the format recommended by Consoli et al. (1993). It begins with a description of the systems analysed in the study (Section 2). This is followed by sections on Goal Definition and Scoping, Inventory Analysis and Impact Assessment (Sections 3 to 5). Section 6 presents the results of the study, and the results of the sensitivity analyses are discussed in Section 7. Conclusions are drawn from the study in Section 8.

2. Details of the Systems Considered in the Study

Three breadmaking wheat production systems were studied:

System A An intensive production system on a large arable farm without animal production, typical of East Anglicia in the UK. It has a high input level of synthetic fertilisers and
pesticides. 40% of the straw is baled and the remainder incorporated. The grain yield is 8 tonnes/ha with a protein content of 12%.

**System B** A reduced input (integrated) system on a non-livestock farm, typical of wheat grown in Switzerland. Thomas meal (a waste product from steel production) is used as phosphate fertiliser together with other synthetic fertilisers and pesticides. All the straw is incorporated. The grain yield is 6 tonnes/ha with a protein content of 11%.

**System C** A low input system on an organic farm in Switzerland. Farm manure and mechanical or manual weed control are used instead of synthetic fertilisers and pesticides. All the straw is baled. The grain yield is 4 tonnes/ha with a protein content of 12%.

Figure 1 shows the different processes considered in the analysis, and Table 1 summarises the main differences between the systems. The processes include energy and material production; fertiliser and pesticide production; agricultural machinery production, maintenance and storage; on-farm activities; and soil-related processes. Fuller details for each system are given in Appendix VI.1.

![Diagram of processes considered in the study](image)

**3. Goal Definition and Scoping**

As recommended by Consoli *et al.* (1993), this stage of LCA requires consideration of the purpose, scope, functional unit, and data quality goals for the study. They are discussed below in Sections 3.1 to 3.4.
<table>
<thead>
<tr>
<th>Type of Characteristics</th>
<th>Characteristic</th>
<th>System A</th>
<th>System B</th>
<th>System C</th>
</tr>
</thead>
<tbody>
<tr>
<td>Crop</td>
<td>Grain yield</td>
<td>8 tonnes</td>
<td>6 tonnes</td>
<td>4 tonnes</td>
</tr>
<tr>
<td></td>
<td>Protein content</td>
<td>12%</td>
<td>11%</td>
<td>12%</td>
</tr>
<tr>
<td></td>
<td>Straw yield</td>
<td>5 tonnes</td>
<td>3.5 tonnes</td>
<td>5 tonnes</td>
</tr>
<tr>
<td></td>
<td>Straw incorporated</td>
<td>60%</td>
<td>100%</td>
<td>0%</td>
</tr>
<tr>
<td>Human labour</td>
<td>Hours of labour</td>
<td>15.0 hours</td>
<td>17.2 hours</td>
<td>32.5 hours</td>
</tr>
<tr>
<td>Machinery</td>
<td>Hours of tractor and combine harvester</td>
<td>13.4 hours</td>
<td>13.0 hours</td>
<td>21.7 hours</td>
</tr>
<tr>
<td></td>
<td>Diesel fuel use</td>
<td>126 litres</td>
<td>105 litres</td>
<td>126 litres</td>
</tr>
<tr>
<td>Fertilisation</td>
<td>Nitrogen applied</td>
<td>240 kg</td>
<td>132 kg</td>
<td>86 kg</td>
</tr>
<tr>
<td></td>
<td>Phosphorus applied</td>
<td>26 kg</td>
<td>22 kg</td>
<td>24 kg</td>
</tr>
<tr>
<td></td>
<td>Potassium applied</td>
<td>50 kg</td>
<td>10 kg</td>
<td>215 kg</td>
</tr>
<tr>
<td></td>
<td>Total mass of synthetic fertilisers</td>
<td>971 kg</td>
<td>930 kg</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Manure</td>
<td>-</td>
<td>-</td>
<td>15 tonnes farmyard manure</td>
</tr>
<tr>
<td></td>
<td>Lime mass applied</td>
<td>3 tonnes every 3 years</td>
<td>-</td>
<td>3 tonnes every 3 years</td>
</tr>
<tr>
<td>Pesticides</td>
<td>Active ingredients</td>
<td>6.76 kg</td>
<td>3.03 kg</td>
<td>-</td>
</tr>
<tr>
<td>Fields</td>
<td>Size of fields</td>
<td>Larger</td>
<td>Smaller</td>
<td>Smaller</td>
</tr>
<tr>
<td>Soil erosion</td>
<td>Soil mass eroded</td>
<td>6 tonnes</td>
<td>3 tonnes</td>
<td>3 tonnes</td>
</tr>
<tr>
<td>Timing of activities</td>
<td>Primary cultivation</td>
<td>September</td>
<td>September</td>
<td>October</td>
</tr>
<tr>
<td></td>
<td>Base fertilisation</td>
<td>September</td>
<td>September</td>
<td>October</td>
</tr>
<tr>
<td></td>
<td>Sowing</td>
<td>October</td>
<td>October</td>
<td>November</td>
</tr>
<tr>
<td></td>
<td>Top fertilisation</td>
<td>March-June</td>
<td>March-June</td>
<td>March-April</td>
</tr>
<tr>
<td></td>
<td>Harvesting</td>
<td>August</td>
<td>July</td>
<td>August</td>
</tr>
<tr>
<td></td>
<td>Fallow</td>
<td>-</td>
<td>August</td>
<td>September</td>
</tr>
<tr>
<td>Rotation</td>
<td></td>
<td>Six year rotation: wheat, wheat, beans, wheat, barley, rape</td>
<td>Four year rotation: maize, sugar beet, wheat, rape, green manure</td>
<td>Six year rotation: maize, potatoes, wheat, grass ley, grass ley, grass ley</td>
</tr>
<tr>
<td>Physical habitat details</td>
<td>Presence of additional features in fields</td>
<td>Few additional features.</td>
<td>Many additional features such as lone trees, ditches and ponds.</td>
<td>Many additional features such as lone trees, ditches and ponds.</td>
</tr>
<tr>
<td>Field boundaries</td>
<td>Hedge with 2m field margins.</td>
<td>Field size = 20 ha.</td>
<td>Additional features include lone trees and ditches.</td>
<td>Path with 1.5m field margins.</td>
</tr>
</tbody>
</table>

Source: see Appendix VI.1.

a Magnesium and sulphur were also added in the fertilisers.

b Green manure cultivated for five months over winter.
3.1 Purpose

The immediate purpose of the study was to compare breadmaking wheat production from three different farming systems. Since the EU project's objective was to develop LCA methodology for agricultural systems, the systems were selected as interesting examples for analysis rather than to be representative of typical farming practices. Therefore, the results should not be viewed as comparing, for example, the relative environmental benefits of intensive, integrated and organic farming systems.

The more important purpose of undertaking the study was, however, to explore the methodological and application-related issues associated with LCAs assessing agricultural systems, as outlined in previous chapters. For this reason, the "graves" of the systems under analysis were defined as the farmgate (rather than the final product, bread), and Improvement Assessment was not considered explicitly in the study.

3.2 Scope of the Study

Defining the scope of a study involves decisions about inclusion or exclusion of different processes. For this study, issues related to scoping arose concerning:

- **Soil-related processes**
  A decision to include or exclude soil from the system can make a significant difference to the LCA results because a number of processes with environmental impacts take place through the medium of the soil (see Chapter III). In this study, soil down to the depth of ploughing was regarded as crossing the time boundary into the system under analysis at the beginning of the time period studied, and leaving it at the end of the relevant time period (one year later). Therefore, differences in the soil's quantity and quality between the beginning and end of the study were relevant for inclusion.

- **Atmospheric deposition of nitrogen and heavy metals**
  Nitrogen and heavy metals are deposited onto agricultural land from the atmosphere in addition to inputs in fertilisers. It is debatable whether atmospheric deposition should be included in an agricultural LCA study (see Chapter III, Section 4.4). In this study, the influence of including or excluding heavy metal deposition in the study was explored using sensitivity analysis (Section 7).
• Human labour

Human labour was excluded from the study on the basis that people exist regardless of their occupation. In other words, the environmental burdens arising from a farm worker's lifestyle (food consumption, use of energy, and so on) occur whether he or she works on a farm, in an office or in a shop. This is a simplification to the extent that those engaged in manual labour are likely to consume larger quantities of food than those in more sedentary occupations. However, these differences in food consumption patterns will make an insignificant contribution to the LCA results. [This should not be interpreted as underrating the importance of human labour in farming systems: human labour should be considered alongside other socio-economic factors in the overall decision-making process to which environmental LCA makes a contribution.]

• Capital equipment

Construction and maintenance of agricultural machinery were included because other studies have indicated that it can make a significant contribution to the LCA results (see, for example, Weidema et al., 1995).

• Manure

The use of manure as fertiliser in System C raised the issue of whether manure and slurry production should be included within the system boundary. This was resolved by including the burdens arising from equivalent nitrogen production using biological nitrogen fixation in a clover crop (see Section 4.2.2 below). However, inputs of phosphorus and potassium to System C were not accounted for in this study. The influence on the results of choosing this modelling approach to account for use of manure was investigated using sensitivity analysis (Section 7).

• Crop rotations

One particular crop was considered in this study, but it interacts with other crops in a crop rotation as discussed in Chapter V. This aspect is discussed in more detail in Section 4.2.4.

• Transport of harvested grain to breadmaking facility

As this was a comparative study of different wheat production systems, realistic comparison should include transport to the breadmaking facility because the transportation distance will be different for System A (located in the UK) compared with Systems B and C (located in Switzerland). The influence on the results of including transportation of the dried grain was investigated using sensitivity analysis (Section 7).

In summary, therefore, the following assumptions were made in scoping the study:

• Inclusion of soil for the time period under consideration (one year)
• Influence of including atmospheric deposition of heavy metals and nutrients investigated using sensitivity analysis
• Exclusion of human labour
• Inclusion of capital equipment
• Inclusion of the environmental impacts of manure production by modelling equivalent nitrogen production using a nitrogen-fixing crop
• Inclusion of crop rotations using the approach developed in Chapter V (see Section 4.2.4)
• Influence of including transport of harvested grain investigated using sensitivity analysis.

3.3 Functional Unit

The Functional Unit for the study was defined as “that quantity of wheat grain containing 120 kg protein used to make one tonne 12% flour, delivered to the breadmaking facility.” Use of this definition recognises that:

• The three systems produce grain of equivalent quality apart from their protein contents: Systems A and C produce grain with 12% protein content while System B produces grain with 11% protein content.
• Grain of a lower protein content is blended with grain of a higher protein content to obtain the required protein level in the flour. Thus, a breadmaker requiring 12% protein-content flour may use grain from the three systems as shown in Figure 2. For System B, Figure 2 shows that the grain is supplemented by higher protein-content grain from another system (System D) which may have different burdens associated with its production.
• This higher protein-content grain may be imported and so there will be additional burdens related to its transportation to the breadmaking facility.

Figure 2. Quantities of Grain Required From Systems A, B and C In Breadmaking

```
System A
1000 kg grain
12% protein
1 tonne 12% flour

System B
670 kg grain
11% protein
1 tonne 12% flour

System D
330 kg grain
14% protein
1 tonne 12% flour

System C
1000 kg grain
12% protein
1 tonne 12% flour
```
For this study, the environmental burdens of protein production (in the form of wheat grain) in System D were assumed to be the same as for System B. Of course, in reality these burdens may vary between different farming systems but there is no reason why this assumption is unrealistic as one scenario.

3.4 Data Quality Goals

The criteria used to select data were:

- Age of data: more recent data were preferred to older data.
- Specificity of data: more detailed data were preferred to generic data.
- Expert judgement of likely accuracy of data: in some cases, data were selected based on discussions about likely accuracy with other researchers in the EU project.

4. Inventory Analysis

Having defined the scope of the study, data were collected and environmental burdens calculated for processes included in the study (as shown in Figure 1). The methods used for calculating these burdens are discussed in Section 4.1; fuller details can be found in Appendix VI.2. In addition, a number of allocation issues required resolution, and these are discussed in Section 4.2. The source of data for materials, energy and transport was the PEMS database (Version 3.2, produced by Pira, 1995); the “Average European” mix was used for electricity and “Heat-Middle Distillate” for energy from fuel oil and diesel oil. For transport, the “Large truck - average” data were used for road transport and “Rail - electric” for rail transport.

4.1 Calculation of Environmental Burdens for Processes in Agricultural Production

4.1.1 Production and Maintenance of Agricultural Machinery

The burdens associated with steel and rubber production, and energy utilisation were calculated for the study. Energy utilisation was calculated for each machine related to manufacture, repairs and maintenance, and transport of the manufactured machines to the farm. The methodology is described in Appendix VI.2.
4.1.2 Use of Agricultural Buildings for Storage of Machinery

Only the burdens associated with energy utilisation were calculated in this study, using an energy utilisation value of 419 MJ/m²/year (Kohler, 1994, assuming buildings last 80 years). This value accounts for construction, maintenance and demolition of industrial buildings and so represents the upper limit of the value expected for agricultural buildings. It was allocated to each item of machinery according to its space requirements (see Appendix VI.2). The energy mix was taken as 100% fuel oil.

4.1.3 Production of Fertilisers

The study accounted for:

- Energy used in production of different fertilisers
- Natural gas as feedstock for nitrogen fertilisers
- Process emissions for nitrogen fertilisers and triple superphosphate (TSP) production.

Data are given in Appendix VI.2. The study did not account for any process emissions during potassium fertiliser production: according to Pybus (1995), dust emissions are small and “harmless to humans and beneficial to vegetation,” while sodium chloride discharged into the North Sea is diluted so rapidly that any effects can be discounted. Issues related to production of Thomas meal are discussed in Section 4.2.3.

4.1.4 Production of Pesticides

Energy used in production of different pesticides was included in the study. For other emissions during pesticide production, average emissions from data on total annual production of pesticides in the UK were used to give an indication of their influence on the final results. More details are given in Appendix VI.2.

4.1.5 Production and Use of Energy Carriers, Including Transportation

Data for different energy carriers (oil, gas, electricity, and so on) were taken from the PEMS database. For each energy input in the study, the specific energy carriers required were selected based on the literature and/or discussions with participants in the EU project; they are listed in the other sections of
this chapter and in Appendix VI.2. For transportation, default distances of 1000 km by rail and 200 km by road were used for fertiliser, pesticide and agricultural machinery delivery to the farm. A default distance of 200 km by road was used for delivery of lime to the farm.

### 4.1.6 Production of Seeds

There are two ways of accounting for seeds:

- If the method of seed production is very different from crop production (for example, grass and clover crops), seed production should be modelled separately.
- If it is similar to crop production, the same burdens can be used as for crop production, and the yield of seeds per hectare adjusted where appropriate.

For this study, the difference in production methods for seed and crop was assumed to be small and therefore the same burdens were assumed as for crop production.

### 4.1.7 Use of Fertilisers

Three aspects were considered related to use of fertilisers: fate of nutrients, fate of heavy metals and the role of the soil as a sink for methane. These are discussed in Sections 4.1.7.1 to 4.1.7.3 below.

#### 4.1.7.1 Fate of Nutrients

The outputs of nutrients considered in the study were:

- **Nitrogen:** harvested crop; nitrate to water; nitrogen oxides, nitrous oxide and ammonia to air.
- **Phosphorus:** harvested crop; phosphate to water; surplus remaining in soil.
- **Potassium:** harvested crop; potassium to water; surplus remaining in soil.

Each output was quantified using one or more methods based on formulae from, or derived from, literature or complex models. Further details about modelling the fate of nutrients are given in Appendix VI.3, and the data used in the study are listed in Appendix VI.2. The resulting balances of nutrients in the three systems are shown in Table 2. It should be noted that nutrients lost in eroded

---

1 In the case of herbage seed production, herbicides are used regularly to minimise growth of weeds, and high rates of seedbed phosphorus and potassium are applied to encourage vigorous growth (Brockman, 1995, p.203).
soil were not included in the analysis but the effects on the results of including this aspect were considered during sensitivity analysis (see Section 7).

Table 2. Flows of Nutrients In Farming Systems Per Hectare

<table>
<thead>
<tr>
<th>Nutrient</th>
<th>Input or output</th>
<th>System A</th>
<th>System B</th>
<th>System C (Nitrogen-Fixing Crop)</th>
<th>Use of Manure</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nitrogen</td>
<td>Fertilisers</td>
<td>+240.0</td>
<td>+132.0</td>
<td>+222.6</td>
<td>+141.0^a</td>
</tr>
<tr>
<td></td>
<td>NO₃-N to water</td>
<td>-31.0</td>
<td>-7.8</td>
<td>-47.4</td>
<td>-7.3</td>
</tr>
<tr>
<td></td>
<td>N₂O-N to air</td>
<td>-3.1</td>
<td>-1.2</td>
<td>-0.12</td>
<td>-0.66</td>
</tr>
<tr>
<td></td>
<td>NOₓ-N to air</td>
<td>-0.3</td>
<td>-0.1</td>
<td>-0.01</td>
<td>-0.07</td>
</tr>
<tr>
<td></td>
<td>NH₃-N to air</td>
<td>-12.4</td>
<td>-4.8</td>
<td>-7.7</td>
<td>-21.6</td>
</tr>
<tr>
<td></td>
<td>Harvested grain</td>
<td>-133.6</td>
<td>-91.8</td>
<td>(-141.0)^b</td>
<td>-66.8</td>
</tr>
<tr>
<td></td>
<td>Baled straw</td>
<td>-8.7</td>
<td>0</td>
<td>-</td>
<td>-21.7</td>
</tr>
<tr>
<td></td>
<td>Balance: N₂ to air</td>
<td>+50.9</td>
<td>+26.3</td>
<td>+13.2</td>
<td>+11.5</td>
</tr>
<tr>
<td></td>
<td>Balance: N to soil</td>
<td>0</td>
<td>0</td>
<td>+13.2</td>
<td>+11.5</td>
</tr>
<tr>
<td>Phosphorus</td>
<td>Fertilisers</td>
<td>+26.0</td>
<td>+22.2</td>
<td>+24.0</td>
<td>+24.0</td>
</tr>
<tr>
<td></td>
<td>P to surface water</td>
<td>-1.0</td>
<td>-1.0</td>
<td>-1.0</td>
<td>-1.0</td>
</tr>
<tr>
<td></td>
<td>Harvested grain</td>
<td>-22.3</td>
<td>-16.7</td>
<td>-14.1</td>
<td>-3.0</td>
</tr>
<tr>
<td></td>
<td>Baled straw</td>
<td>-1.2</td>
<td>0</td>
<td>-</td>
<td>-3.0</td>
</tr>
<tr>
<td></td>
<td>Balance: in soil</td>
<td>+1.4</td>
<td>+4.2</td>
<td>+</td>
<td>+8.8</td>
</tr>
<tr>
<td>Potassium</td>
<td>Fertilisers</td>
<td>+50.0</td>
<td>+55.0</td>
<td>+215.0</td>
<td>+215.0</td>
</tr>
<tr>
<td></td>
<td>K to surface water</td>
<td>-25.0</td>
<td>-25.0</td>
<td>-37.0</td>
<td>-37.0</td>
</tr>
<tr>
<td></td>
<td>Harvested grain</td>
<td>-29.9</td>
<td>-22.4</td>
<td>-15.0</td>
<td>-15.0</td>
</tr>
<tr>
<td></td>
<td>Baled straw</td>
<td>-14.4</td>
<td>0</td>
<td>-36.1</td>
<td>-36.1</td>
</tr>
<tr>
<td></td>
<td>Balance: in soil</td>
<td>-19.3</td>
<td>+7.6</td>
<td>+127.0</td>
<td>+127.0</td>
</tr>
</tbody>
</table>

^a In farmyard manure, 41 kg N is available for the current crop and 50 kg is available in subsequent years; the remaining 50 kg is available N in liquid manure.

^b Nitrogen to wheat production system.

4.1.7.2 Fate of Heavy Metals

Heavy metals reach the soil by atmospheric deposition, and in synthetic fertilisers and manure; the quantities are given in Appendix VI.2. In modelling this aspect for the study:

- Atmospheric deposition was only considered during sensitivity analysis (see Section 7).
- For Thomas meal, the allocation method used in the LCA determines whether the environmental impacts related to heavy metals in Thomas meal are allocated to the farming system using the fertiliser or to the steel production system from which it derives (see Section 4.2.3). In this study, the chosen allocation method (see Section 4.2.3) resulted in the total heavy metal content of the Thomas meal being allocated to System B.
- For manure, the same situation applies as for Thomas meal: the allocation method determines whether the environmental impacts are allocated to the farming system using the manure or to the former livestock production system. In this study, the chosen allocation method resulted in the
total heavy metal content of the manure being allocated to the livestock production system rather than to System C (see Section 4.2.2 below).

Heavy metals may leave the system in the harvested crop(s), in eroded soil and leached into water. The amounts taken up by the crop are determined by variables that include the concentration of the heavy metal in the soil, the soil type, and the weather. Each heavy metal may respond in a different way to these variables (see Alloway, 1990). For this study, limited data on the heavy metal content of soils and plants were used to derive a first estimate of the fate of the heavy metals applied in the systems under analysis (see Appendix II.1). Heavy metals in the harvested straw were assumed to be returned to the soil (either as livestock bedding or feedstuffs).

4.1.7.3 Impact of Fertiliser Use On the Role of Soil as a Sink for Methane

Application of ammonium fertilisers affects the function of the soil as a sink for methane. Figures quoted are reductions of 30-70% from a natural level of 2 kg methane absorbed per hectare per year in unfertilised soils (Duxbury et al., 1993; Knowles, 1993; Lelieveld and Crutzen, 1993). For this study, it was assumed that there is a linear decrease in sink strength such that for application of 150 kg nitrogen per hectare, the sink strength is reduced to 1 kg methane per hectare. Thus the reduction in sink strength is N/150 kg methane per hectare where N is the amount of nitrogen applied as fertiliser or manure (after Audsley et al., 1997, p.58).

4.1.8 Use of Pesticides

The Impact Assessment factors for toxicity calculated by Jolliet and Crettaz (in Audsley et al., 1997, p.79, 83) include an allowance for the fate of pesticides (see Appendix II.1). Since these factors were used at the Impact Assessment stage of this study, the modelled fate of the pesticides was also an implicit part of the study.

4.1.9 Use of Water

No additional irrigation water was required by any of the systems in this study.

---

2 Since most of the heavy metals in straw will eventually return to the soil (via livestock bedding or feedstuffs), this seems a reasonable assumption.
4.1.10 Use of Lime

The quantities of lime used in each system are listed in Table 3. It was assumed that the soil pH without liming was 5.6 in System A and 5.4 in System C. Following the method developed in Chapter V (Section 4.5), the burdens associated with use of lime were allocated according to the pH requirements of each crop in the rotation. Relevant data are given in Table 3.

Table 3. Use of Lime

<table>
<thead>
<tr>
<th>System</th>
<th>Use of Lime</th>
<th>pH of Crops in Rotation</th>
<th>Lime Allocated to Wheat Crop</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Crop</td>
<td>pH</td>
</tr>
<tr>
<td>System A</td>
<td>3 tonnes every 3 years</td>
<td>Wheat</td>
<td>5.6</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Wheat</td>
<td>5.6</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Beans</td>
<td>6.0</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Wheat</td>
<td>5.6</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Barley</td>
<td>5.9</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Rape</td>
<td>5.6</td>
</tr>
<tr>
<td>System B</td>
<td>–</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>System C</td>
<td>3 tonnes every 3 years</td>
<td>Maize</td>
<td>5.6</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Potatoes'</td>
<td>4.9</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Wheat</td>
<td>5.6</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Grass ley</td>
<td>4.9</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Grass ley</td>
<td>4.9</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Grass ley</td>
<td>4.9</td>
</tr>
</tbody>
</table>


4.1.11 Changes In Organic Matter Content of Soil

Addition of organic matter in manure and straw was assumed to have no effect on the organic matter content in the systems under analysis (see discussion in Chapter V, Section 4.3). However, the influence of this assumption on the final results was explored during sensitivity analysis (Section 7).

4.1.12 Soil Compaction

Soil compaction was assessed using the method developed in Chapter V, Section 4.6. Data are given in Appendix VI.2.

4.2 Allocation Methods

The allocation methods chosen for relevant parts of the study reflect the hierarchy of approaches to allocation outlined in Chapter II, and the practical availability of data for undertaking the analysis. The methods used are outlined below. A discussion of the different approaches can be found in
Appendix VI.4, and the influence of choice of approach on the final results is demonstrated in Section 7.

4.2.1 Wheat Grain and Straw

Wheat grain and straw are co-products of the wheat production system. In Appendix VI.4, different methods for allocation of burdens between the grain and straw are described, according to the hierarchy of approaches to allocation (see Chapter II, Section 3). System extension is the preferred option in the hierarchy. However, it was not possible to use it in this case because there was no obvious way of extending the system to account for the fate of the straw (whose most likely subsequent use was as livestock bedding). It was therefore decided to use “allocation on the basis of physical causality” (the second approach in the hierarchy). The following burdens were allocated to the grain:

Total burdens for wheat production to harvest

*Plus* additional burdens for straw incorporation (3.76 litres diesel and “0.2 ha” of straw chopper per tonne straw for straw chopping and deep ploughing)

*Plus* additional soil organic matter (850 kg dry matter per tonne straw)

*Minus* avoided burdens from nutrients added to soil in the straw (4.3 kg nitrogen, 0.6 kg phosphorus and 7.2 kg potassium per tonne straw (Appendix VI.2, Table 7, numbers adjusted to allow for water content of straw)).

For System A, the burdens were calculated assuming that the quantity of straw incorporated was 0.256 tonnes straw per additional 1 tonne grain. This is because a marginal increase in the grain output from 7.815 tonnes/ha (i.e. yield after accounting for seed yield) to 8.815 tonnes/ha by increasing the cultivated land area would require incorporation of an additional 0.256 tonnes straw in order to hold the straw output constant. System B did not produce baled straw, and so allocation was not an issue in this system. For System C, the burdens were calculated assuming that the quantity of straw incorporated was 1.316 tonnes straw per additional 1 tonne grain. This was because a marginal increase in the grain output from 3.8 tonnes/ha to 4.8 tonnes would require incorporation of 1.316 tonnes straw. It is worth noting that this approach is equivalent to assuming that all the straw is incorporated at some point in its life cycle.

---

3 Data taken from Grant *et al.* (1995, p.18, 22).
4.2.2 Manure

Manure is a co-product of livestock production systems; other co-products from these systems include milk, meat and hide (see Chapter III, Figure 4). It is regarded as a product because it has economic value. Therefore, its use in System C raises a question about how the burdens of manure production can be included in the analysis. Appendix VI.3 describes the different approaches, according to the hierarchy of approaches to allocation. In this case, it was possible to use the system extension method, assuming that the alternative source of nitrogen in the manure was a nitrogen-fixing crop (see Appendix VI.2 for data on the nitrogen-fixing crop). This gave the total burdens associated with use of manure in System C as:

Total methane emissions during treatment and storage of manure: 5.5 kg methane per tonne farmyard manure and 0.55 kg methane per cubic metre liquid manure (Hausheer, 1997, pers.comm.)

Plus burdens associated with use in System C

Plus burdens associated with production and use of nitrogen equivalent to the quantity in manure, using a nitrogen-fixing crop.

The burdens associated with use in System C (the Foreground System) were included as a worst-case scenario. In other words, it was assumed that alternative use in the Background System would have negligible burdens associated with it compared with use in the Foreground System. The influence of this assumption on the results was investigated using sensitivity analysis (Section 7).

4.2.3 Thomas Meal

Thomas meal is a waste product from steel production. Assuming that the Thomas meal is in excess supply, and using the system extension approach, the agricultural system under analysis should be credited with the avoided burdens from its disposal in the Background System. Any additional burdens due to further processing, transport and use of Thomas meal in the agricultural system (the Foreground System) should be added to this system. The following additional burdens were therefore used in the study for System B: i) a processing energy requirement of 9.6 MJ/kg P (Audsley et al., 1997, p.32), ii) transportation for use in the agricultural system (1000 km by rail, 200 km by road) (see Section 4.1.5), and iii) the estimated quantities of heavy metals in the Thomas meal used in System B (see Appendix VI.2). No avoided burdens were credited to System B on the assumption
that the waste slag from steel production would have minimal environmental burdens associated with its disposal; the effect of this assumption on the results is discussed in Section 7.

### 4.2.4 Crop Rotations

Allocation of burdens among different crops in a rotation may be relevant for a number of processes during the specific time period under analysis. In this study, they included use of lime, the balance of nutrients in the soil at the end of the time period, and the green manure crop in System B. Use of lime has been discussed in Section 4.1.10. For the balance of nutrients, the method described in Chapter V, Section 3.3, was used in the study. Thus:

- The surplus P nutrient in Systems A and B (see Table 2) was credited against the quantity of phosphate fertiliser applied in Systems A and B.
- The deficit of K nutrient in System A was assessed as an increased requirement for potassium fertiliser by future crops. The surplus K nutrient in System B was credited against the quantity of potash fertiliser applied in System B.
- For System C, no credit was given for the surplus P and K because their supply cannot be separated from the supply of N (as they all occur together in the manure). Instead, System C was credited with surplus N in the soil (see Table 2) on the assumption that this would become available to future crops.

The burdens associated with the green manure crop in System B were allocated between the maize, sugar beet, wheat and rape crops in the rotation according to the area-time requirements of these crops following the method developed in Chapter V, Section 2.3. The burdens were taken as equivalent to those for cultivation of a nitrogen-fixing crop (see Appendix VI.4).

### 5. Impact Assessment

Having quantified the burdens for the three systems, their environmental impacts were assessed in the Impact Assessment phase of the LCA. This involved definition of appropriate Impact Assessment categories; use of the new methodological approaches developed in Chapters III, IV and V; and calculation of impact values. The categories considered in the study are listed in Table 4. Sections 5.1 to 5.5 describe the methodologies used in the study for the new Impact Assessment categories and
for those categories where there was a choice of more than one methodology (marked with an asterisk in Table 4).

5.1 Solar Energy

The incident radiation at the Earth's surface for the southern part of the UK is 1,100 kWh/m²/year, and for Switzerland is 1,200 kWh/m²/year (average values cited in Oliver and Jackson, 1998). These values were used to calculate use of solar energy by each system, using the methodology described in Chapter III, Section 5.1.

### Table 4. Summary of Impact Assessment Categories Considered In the Study

<table>
<thead>
<tr>
<th>Type of Impact Assessment Category</th>
<th>Impact Assessment Category</th>
<th>Method and Source of Data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Abiotic resources</td>
<td>Non-renewable energy (fossil fuels)</td>
<td>MJ total extracted energy</td>
</tr>
<tr>
<td></td>
<td>Solar energy*</td>
<td>See Chapter III, Section 5.1</td>
</tr>
<tr>
<td></td>
<td>Water use</td>
<td>See Chapter III, Section 5.2</td>
</tr>
<tr>
<td></td>
<td>Others</td>
<td>IA factors calculated as annual extraction rate divided by total reserves (Lindfors et al., 1995). Soil loss calculated using method in Chapter V, Section 3.1.</td>
</tr>
<tr>
<td>Land use</td>
<td>Land area</td>
<td>Area under cultivation (m²-year)</td>
</tr>
<tr>
<td></td>
<td>Physical ecosystem degradation*</td>
<td>See Chapter IV.</td>
</tr>
<tr>
<td></td>
<td>Soil quality: organic matter content*</td>
<td>See Chapter V, Section 4.3</td>
</tr>
<tr>
<td></td>
<td>Soil quality: soil compaction*</td>
<td>See Chapter V, Section 4.6</td>
</tr>
<tr>
<td></td>
<td>Landscape degradation</td>
<td>See Chapter II, Section 4.2.3</td>
</tr>
<tr>
<td>Pollution</td>
<td>Global warming</td>
<td>Lindfors et al. (1995); see Chapter II, Table 7</td>
</tr>
<tr>
<td></td>
<td>Acidification</td>
<td>Lindfors et al. (1995); see Chapter II, Table 8</td>
</tr>
<tr>
<td></td>
<td>Photochemical oxidant formation</td>
<td>Heijungs et al. (1992a); see Chapter II, Table 9</td>
</tr>
<tr>
<td></td>
<td>Eutrophication</td>
<td>Heijungs et al. (1992a), see Chapter II, Table 10</td>
</tr>
<tr>
<td></td>
<td>Ecotoxicity*</td>
<td>Jolliet in Audsley et al. (1997); see Appendix II.1 and Appendix VI.5</td>
</tr>
<tr>
<td></td>
<td>Human toxicity*</td>
<td>Jolliet in Audsley et al. (1997); see Appendix II.1 and Appendix VI.5</td>
</tr>
<tr>
<td></td>
<td>Odour</td>
<td>Heijungs et al. (1992a); see Chapter II, Section 4.2.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td>* = methodology described in Sections 5.1 to 5.5.</td>
</tr>
</tbody>
</table>

5.2 Physical Habitat Degradation

In Chapter IV, three methods for assessing physical habitat degradation were developed. For this study, the first two methods (physical habitat degradation at the global and national levels) were not suitable because both methods fail to distinguish between the types of habitats considered in the study. The third method was also unsuitable because the data available for the study (Table 1) were
not sufficiently detailed to develop a Physical Management Index. For example, the lack of data on “additional features” in fields and field boundaries would make comparison between the three systems impossible. Therefore, this impact category was not considered in the assessment.

5.3 Soil Quality

Following the method developed in Chapter V, two factors were assessed:

- Organic matter: assessed using the method developed in Chapter V, Section 4.3 (initially using Method 4 but comparing with results using Method 3 as part of the sensitivity analysis).
- Soil compaction: assessed using the method developed in Chapter V, Section 4.6.

5.4 Ecotoxicity

Ecotoxicity was assessed using the Critical Surface-Time methodology (see Appendix II.1). The range of ETPs used in the study are shown in Table 5. Data for each substance can be found in Appendix VI.5. According to this approach, ecotoxicity is most likely to be associated with heavy metals reaching both aquatic and terrestrial ecosystems where heavy metals occur as inputs to agricultural systems. Pesticides and other substances such as oil and phenol may also make a contribution to aquatic ecotoxicity.

The results using this method were compared with the results using the Guinée et al. (1996) method as part of the sensitivity analysis (Section 7).

Table 5. Range of Ecotoxicity Potentials (ETPs) Used In Study

<table>
<thead>
<tr>
<th>Substance</th>
<th>Aquatic Ecotoxicity Potentials (ETP&quot;)</th>
<th>Terrestrial Ecotoxicity Potentials (ETP&quot;)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Heavy metals</td>
<td>0 - 1300</td>
<td>0 - 19</td>
</tr>
<tr>
<td>Pesticides</td>
<td>0 - 43</td>
<td>Less than 0.01</td>
</tr>
<tr>
<td>Other substances (e.g. oil and phenol)</td>
<td>0 - 15</td>
<td>-</td>
</tr>
</tbody>
</table>

N.B. For heavy metals and other substances, the quantity of each substance reaching water is multiplied by the ETP" and the quantity reaching soil is multiplied by the ETP" for that substance. For pesticides, the quantity of each active ingredient applied is multiplied by the ETP" and ETP" for each active ingredient.

5.5 Human Toxicity

Human Toxicity was also assessed using the Critical Surface-Time methodology (see Appendix II.1). The range of HTPs calculated for substances used in the systems are shown in Table 6; data for each
substance can be found in Appendix VI.5. According to this approach, the HTPs show that human toxicity is most likely to be associated with heavy metals, particularly those in food; this is due to their persistence in the soil and subsequent uptake in future crops. Pesticide residues in foods may also make a contribution where pesticides are used in farming systems.

The results using this method were compared with the results using the Guinée et al. (1996) method during sensitivity analysis (Section 7.5).

Table 6. Range of Human Toxicity Potentials According To Critical Surface-Time Method

<table>
<thead>
<tr>
<th>Substance Category</th>
<th>Range of HTPs in Air</th>
<th>Range of HTPs in Water</th>
<th>Range of HTPs in Food via Soil</th>
</tr>
</thead>
<tbody>
<tr>
<td>Heavy metals</td>
<td>1 - 9</td>
<td>1 - 8</td>
<td>80 - 560,000</td>
</tr>
<tr>
<td>Other substances (e.g. CO₂, particulates, fluorides)</td>
<td>Less than 0.01</td>
<td>Less than 2.1</td>
<td>-</td>
</tr>
<tr>
<td>Pesticides</td>
<td>Less than 0.22</td>
<td>Less than 0.004</td>
<td>0.3 - 47.0</td>
</tr>
</tbody>
</table>

N.B. For heavy metals and other substances, units for HTPs are relative to the effect of lead in air per kg substance emitted to each medium. For pesticides, units for HTPs are also relative to lead but are per kg pesticide applied to field.

6. Results of Impact Assessment

Final Impact Assessment results are shown in Figure 3. In this Figure, the result in each impact category for System A is given a value of one, and the results for Systems B and C are shown relative to System A. It shows that System A has the highest impact for abiotic resource depletion ("other"), acidification (minimum) and ecotoxicity (aquatic). System B has the highest impact for human toxicity. System C has the highest impact for abiotic resource depletion (energy), solar energy, land area, global warming, acidification (maximum), photochemical oxidant formation, eutrophication (N- and P-limited), ecotoxicity (terrestrial) and odour. No results were recorded for water use as no water other than rainfall was used in any of the systems. The results are discussed in more detail in Section 6.1 below; it is worth reiterating that they should not be interpreted as necessarily representative of typical farming systems but as examples of some farming systems.
Figure 3. Relative Impact Assessment Scores for Systems B and C In Relation To System A

- Abiotic Resource Depletion - Energy
- Solar Energy
- Abiotic Resource Depletion - Other
- Land Area
- Physical Ecosystem Degradation
- Soil Compaction
- Soil Organic Matter
- Global Warming (GWP20)
- Global Warming (GWP100)
- Global Warming (GWP500)
- Acidification (Minimum)
- Acidification (Maximum)
- Photochemical Oxidant Formation (Minimum)
- Photochemical Oxidant Formation (Maximum)
- Eutrophication (N-limited)
- Eutrophication (P-limited)
- Ecotoxicity (Aquatic)
- Ecotoxicity (Terrestrial)
- Human Toxicity
- Odour

Result Relative to System A

Key:
- ■ System A
- ○ System B
- □ System C
6.1 Assessment of the Results

In this section, the results for each Impact Assessment category are shown in Figure 4 subdivided into different life cycle stages:

- Machinery production: production, delivery and repairs/maintenance of agricultural machinery used in the field.
- Fertiliser production: production and delivery of synthetic fertilisers.
- Pesticide production: production and delivery of pesticides.
- Diesel fuel: diesel used by agricultural machinery in the field.
- Use of fertilisers: fate of fertilisers after application.
- Use of pesticides: fate of pesticides after application.
- Drying fuel oil: fuel oil required for drying grain.
- N-fixing crop/fallow: cultivation of a N-fixing crop in System C or fallow ("green manure") in System B.
- Use of lime: production, delivery and use of lime in System C.

In the following sections, I discuss the results in each Impact Assessment category, showing which substances contribute more than 10% of the Impact Assessment result for at least one of the systems.

6.1.1 Resource Depletion

*Abiotic Resource Depletion - Non-Renewable Energy (Fossil Fuels)*

*Contributing substances: fossil fuels.*

Figure 3 shows that energy utilisation is similar for the three systems (varying from 3,315 MJ for System A to 3,700 MJ for System C). Table 7 and Figure 4a show that, for Systems A and B, energy utilisation is highest for fertiliser manufacture (51% and 41% respectively of total energy use), followed by diesel use (23% and 27%), fuel oil (11% and 12%), machinery manufacture and maintenance (9% and 12%), and pesticide manufacture (7% and 4%). For System C, energy utilisation is greatest for diesel use (43%), followed by machinery manufacture and maintenance (22%), the nitrogen-fixing crop (11%), fuel oil (10%) and lime (15%). In Systems A and B, over 90% of the energy used in fertiliser manufacture is for nitrogen fertiliser production.
Figure 4. Contribution of Different Subsystems To Impact Assessment Categories

4a) Abiotic Resource Depletion - Energy

4b) Global Warming (GWP20)

4c) Global Warming (GWP100)
Figure 4 (continued)

4d) Global Warming (GWP500)

Percentage contribution to IA category

4e) Acidification (Minimum)

Percentage contribution to IA category

4f) Acidification (Maximum)

Percentage contribution to IA category

- System A
- System B
- System C
Figure 12 (continued)

4g) Photochemical Oxidant Formation (Minimum)

4h) Photochemical Oxidant Formation (Maximum)

4i) Eutrophication (N-limited)
Figure 4 (continued)

4j) Eutrophication (P-limited)

4k) Ecotoxicity (Aquatic)

4l) Ecotoxicity (Terrestrial)
A comparison of the results for the functional unit with the results per hectare yields a number of interesting insights. Table 7 shows that, although the differences in total energy utilisation per functional unit are relatively small for the three systems (1 : 0.99 : 1.12 for Systems A, B and C), the differences per hectare are much larger (1 : 0.69 : 0.55 for Systems A, B and C). This shows the important role of the final yield in influencing the results (since the functional unit results are calculated by dividing the results per hectare by the yield). If the yield changes (for example, due to different weather conditions) while the farming activities (seedbed preparation, base fertilisation and so on) remain constant, then this could have a larger influence on the final results than differences in farming activities between the three systems.

Apart from the influence of yield, the energy utilisation per functional unit is similar for the three systems because the non-use of synthetic fertilisers and pesticides in System C (i.e. an energy saving
compared with Systems A and B) is offset by the additional energy utilisation associated with agricultural machinery, diesel use, lime and cultivation of the nitrogen-fixing crop.

Table 7. Energy Utilisation In Systems A, B and C

<table>
<thead>
<tr>
<th>Sub-System</th>
<th>System A</th>
<th>System B</th>
<th>System C</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>120 kg</td>
<td>120 kg</td>
<td>120 kg</td>
</tr>
<tr>
<td></td>
<td>protein</td>
<td>protein</td>
<td>protein</td>
</tr>
<tr>
<td>Fertiliser manufacture</td>
<td>1,677</td>
<td>1,334</td>
<td>-</td>
</tr>
<tr>
<td>Diesel</td>
<td>748</td>
<td>4,835</td>
<td>1,575</td>
</tr>
<tr>
<td>Fuel oil</td>
<td>352</td>
<td>2,063</td>
<td>363</td>
</tr>
<tr>
<td>Machinery manufacture, maintenance and buildings²</td>
<td>314</td>
<td>2,037</td>
<td>821</td>
</tr>
<tr>
<td>Pesticide manufacture</td>
<td>223</td>
<td>732</td>
<td>-</td>
</tr>
<tr>
<td>Lime</td>
<td>-</td>
<td>-</td>
<td>540</td>
</tr>
<tr>
<td>Nitrogen-fixing crop/green manure</td>
<td>-</td>
<td>145</td>
<td>401</td>
</tr>
<tr>
<td>Total</td>
<td>3,315</td>
<td>17,584</td>
<td>3,700</td>
</tr>
<tr>
<td></td>
<td>25,621</td>
<td>14,057</td>
<td></td>
</tr>
</tbody>
</table>

1 The values are for the total quantity of protein from 1 hectare used to cultivate breadmaking wheat, i.e. they exclude the energy utilisation associated with straw production and harvesting.
2 Although the same amount of fuel oil is used per tonne grain in each system (6.89 kg), the values per 120 kg protein vary slightly between the systems because each system uses a different quantity of seeds (which are themselves dried) and the protein content of the grain is lower in System B leading to a higher fuel oil requirement per 120 kg protein in this system.
3 Out of the total energy utilisation in this sub-system, 18, 31 and 34% is for buildings in Systems A, B and C respectively. The larger value for System C is due to the greater number of machines used in this system and the higher number of tractor and combine harvester hours (see Table 1).

Abiotic Resource Depletion - Solar Energy

Use of solar energy increases from System A to System C in line with the increased land area requirement for each system (Figure 3); in particular, the value for System C is larger than for Systems A or B because of the additional area required for the nitrogen-fixing crop.

Water Use

Since none of the systems used irrigated water, this category is not relevant in the assessment.
Abiotic Resource Depletion - Other

Contributing substances: soil.

In this category, the ranking order of the three systems is System A (highest value) followed by System C then System B (lowest value). Loss of soil dominates this result, contributing more than 99% of the final result for each system.

6.1.2 Land Use

Land Area

The land area requirements per functional unit increase from System A to System C (1,280, 2,086 and 4,075 m² respectively). The over-threefold increase in land area for System C compared with System A is due to the additional area required for the nitrogen-fixing crop as well as the reduced yield in System C. In effect, the land area used to grow the crop must be increased by more than half to account for nitrogen fixation. Obviously the modelling approach used for manure has a big influence on this result, and the implications of assumptions at this stage are explored in Chapter 7.

Physical Habitat Degradation

This category was not included in the assessment (see Section 5.2).

Soil Quality: Soil Organic Matter

Using Method 4 from Chapter V, Section 4.3, there is no difference in this category between the three systems. The influence of using an alternative method is explored in Section 7.6.

Soil Quality: Soil Compaction

Figure 3 shows that Systems A and B have similar Soil Compaction Indicator values but that the result for System C is more than double these values. The data used to calculate these results (see Appendix VI.2, Tables 13 and 14) show that the operations making the greatest contribution are ploughing and harvesting (in all three systems), and manure spreading in System C. Again, as for abiotic resource depletion (non-renewable energy), the yield has an important role in influencing the results.
Landscape Degradation

This category was not included in the assessment.

6.1.3 Pollution

Global Warming
Contributing substances: CO₂, CH₄, and N₂O.

For all the global warming timespans, the systems have the same ranking order in their contribution to global warming: System C (highest value), followed by System A then System B (lowest value). In Systems A and B, CO₂ and N₂O both make the largest contribution to the result. CO₂ contributes between 36% and 60% and N₂O contributes between 39% and 63% to each global warming timespan result. In System C, CH₄ contributes 83% at GWP20 and 39% at GWP500 of the total value for global warming, demonstrating the importance of choice of time horizon for this impact category.

The main life cycle stages contributing to this impact are fertiliser production, use of fertiliser and diesel fuel, as shown in Figures 4b, 4c and 4d. Energy use is the main source of CO₂ emissions at these different life cycle stages, while N₂O and CH₄ are process emissions from production and use of nitrogen fertilisers, and storage and use of manure respectively. Changes in the function of the soil as a sink for CH₄ (see Section 4.1.7.3) make a negligible contribution to this impact category.

Acidification
Contributing substances: NOₓ, SO₂, and NH₃ to air; fluorides and NO₃⁻ to water.

The ranking order of the three systems for the acidification (minimum) scenario is System A (highest value) followed by System C then System B (lowest value) (Figure 3). The ranking order for the acidification (maximum) scenario is System C (highest value) followed by System A then System B (lowest value). Thus the ranking order of the three systems depends on the acidification scenario chosen for the study.

Although eleven substances were assessed in this category (Cl₂, HCl, HF, NH₃, and NOₓ to air; Cl⁻, fluorides, NO₃⁻, and SO₄²⁻ to water), only five (listed above) make a contribution greater than 10% to the final value for each system in either scenario. For the acidification (minimum) scenario, SO₂ makes the largest contribution apart from System A where fluorides to water make the largest contribution; however, as noted in Appendix VI.2, Table 4, in reality these fluorides are unlikely to
make a large contribution because most will be emitted as stable fluorosilicates. For the acidification (maximum) scenario, the largest contribution is from NO₃⁻ in all three systems.

Figures 4e and 4f show that the main life cycle stages contributing to this impact are different for each scenario. For acidification (minimum), the main contributing life cycle stages are machinery production, fertiliser production and diesel fuel. This reflects the fact that SO₂ is the main contributing substance in the acidification (minimum) scenario, and emissions are related to energy use. For acidification (maximum), the main contributing life cycle stages are use of fertiliser and the nitrogen-fixing crop (System C) or fallow period (System B). This reflects the fact that NO₃⁻ makes the largest contribution, and emissions are related to nitrogen processes in the soil.

**Photochemical Oxidant Formation (Photochemical Oxidant Creation Potential)**

*Contributing substances: CO, HC (excluding CH₄), CH₄.*

For this category, the ranking of the three systems remains the same for both scenarios: System C (highest value), followed by System B then System A (Figure 3). Only three types of substances make a contribution greater than 10% in this category: CO, hydrocarbons (HC) excluding CH₄, and CH₄. CH₄ makes a negligible contribution apart from in System C for the POCP (maximum) scenario where it contributes 87% of the final value. This is because CH₄ has a minimum POCP of 0 and a maximum POCP of 0.03 (see Chapter II, Section 4.2.2); therefore, POCP (maximum) is much larger when greater quantities of CH₄ emissions are associated with the system under analysis. The POCP (maximum) result demonstrates the importance of the choice of modelling approach for manure (see Section 7).

Figures 4g and 4h show that the main life cycle stages contributing to this impact in Systems A and B are fertiliser production and diesel fuel. For System C, the main life cycle stage contributing to this impact is use of fertiliser. This is because CO and HC (excluding CH₄) emissions are linked with energy use, and CH₄ is related to storage and use of manure.

**Eutrophication**

*Contributing substances: NOₓ and NH₃ to air; NO₃⁻ and PO₄³⁻ to water.*

The results for eutrophication (N- and P-limited) show that System C has the highest value followed by System A then System B (Figure 3). Although seven substances were considered in the analysis (COD, NH₃ (air), NH₄⁺ (water), NO₂⁻ (air), NO₃⁻ (air), P₂O₅ (air) and PO₄³⁻ (water)), only four made a
contribution greater than 10% to any of the results: NH₃ and NOₓ to air, NO₃⁻ and PO₄³⁻ to water. However, the magnitude of the differences between the systems change for each scenario; for example, for eutrophication (N-limited) the ratio of the results for System A compared with System C is 1:2.7, and for eutrophication (P-limited) it is 1:1.08. In other words, the choice of eutrophication scenario significantly affects the relative magnitude of the results for the three systems.

The main life cycle stages contributing to eutrophication (N-limited) are use of fertilisers and the nitrogen-fixing crop (System C) or fallow land (System B) (Figures 4i and 4j). The main life cycle stages contributing to eutrophication (P-limited) are use of fertiliser, and additionally fertiliser production for System A. This is because the emissions contributing to this impact arise from processes in the soil, and phosphate emissions from TSP production for System A.

Ecotoxicity

**Contributing substances:** Cd, Hg, oils and greases, and phenols to water; Cd, Cr, Ni and Zn to soil; cypermethrin, chlorothalonil and isoproturon.

For ecotoxicity (aquatic), System A has the highest value followed by System B then System C (lowest value) (Figure 3). Seven types of substances make a contribution greater than 10% to this category in at least one of the systems (Cd, Hg, oils and greases, and phenols to water; and cypermethrin, chlorothalonil and isoproturon). In Systems A and B, the main contributions are from use of pesticides (in particular, cypermethrin contributes 71% of the final value in System A, and chlorothalonil and cypermethrin contribute 87% of the final value in System B).

For ecotoxicity (terrestrial), the ranking of the three systems is reversed (Figure 3). System C has the highest value followed by System B then System A (lowest value). In all three systems, the main contribution is from heavy metals in the soil due to use of TSP in System A, Thomas meal in System B, and lime in System C (in particular, Cd, Cr, Ni and Zn contribute more than 10% to the final value for at least one of the systems).

Figures 4k and 4l confirm these results, showing that the main life cycle stages contributing to this impact are use of pesticides and fertilisers (Systems A and B), and use of lime (System C). [The result for System C in Figure 4k is due to oils and greases released as water emissions.]
**Human Toxicity**

*Contributing substances: Cd, Cr, Co, Pb and Se to soil.*

The results for human toxicity show that System B has the highest value followed by System A then System C (lowest value) (Figure 3). The results are dominated by the contribution from heavy metals in food which have been taken up from the soil (over 99.99% in all three systems). The source of these heavy metals is use of TSP in System A, Thomas meal in System B and lime in System C.

**Odour**

*Contributing substance: NH$_3$ to air.*

In this study, the only substance classified in this category is NH$_3$. The ranking of the three systems is System C (highest value) followed by System A then System B (lowest value) (Figure 3). The main life cycle stages contributing NH$_3$ are production and use of nitrogen fertilisers in Systems A and B, and treatment and use of manure in System C.

6.2. Summary of the Results

6.2.1 Contributing Substances

In the resource depletion and land use categories, use of fossil fuels and soil erosion are the major relevant factors, alongside use of land and its incident solar radiation. Soil compaction is due to use of machinery in all three systems, in particular ploughing and harvesting operations, and spreading of manure in System C.

The results for the different pollution categories show that 26 emitted substances contribute more than 10% to at least one Impact Assessment category in one of the systems, and these are listed in Appendix VI.6. Originally, approximately 100 substances emitted to different media were considered in the analysis. This suggests that it may be possible to simplify assessment of agricultural systems by focusing on a restricted number of substances making the greatest contribution to the impacts. Of course, choice of these substances requires verification through assessment of other agricultural systems.
6.2.2 Life Cycle Stages

The results shown in Figure 4, together with data from the spreadsheets used for the analysis, indicate that a number of life cycle stages make a major contribution to the Impact Assessment results:

- Production and use of nitrogen fertilisers: contribute to a range of Impact Assessment categories.
- Production and use of phosphate fertilisers: contribute to a range of Impact Assessment categories due to process emissions during production, and leaching from the soil.
- Use of manure: contributes to a range of Impact Assessment categories due to CH₄ emissions and N-emissions during storage and use.
- Production, maintenance and repair of agricultural machinery: contributes to a range of Impact Assessment categories.
- Production and use of diesel fuel: contributes to a range of Impact Assessment categories.
- Use of pesticides: contributes to ecotoxicity (aquatic).
- Use of lime: contributes particularly to ecotoxicity (terrestrial) and human toxicity due to heavy metals in lime.
- Nitrogen-fixing crop and fallow ("green manure"): contributes to acidification (maximum) and eutrophication (N-limited) due to N-emissions from the soil.

A number of life cycle stages make a contribution less than 10% to any Impact Assessment category:

- Production of pesticides
- Production and use of potassium fertilisers
- Role of the soil as a sink for CH₄
- Production of seeds.

The implication is that it may be possible to simplify assessment of agricultural systems by focusing on the restricted number of life cycle stages that make the greatest contribution to the impacts. As for choice of a restricted number of substances (Section 6.2.1), however, choice of these life cycle stages requires verification through assessment of other agricultural systems.
7. Sensitivity Analysis

7.1 Atmospheric Deposition of Heavy Metals

Atmospheric deposition may add heavy metals to agricultural soil in addition to those from phosphate fertilisers and lime. In order to investigate the influence of this source of heavy metals on the Impact Assessment toxicity results, atmospheric deposition was added to the system model for System A. The data used in the analysis are given in Appendix VI.2, Table 11. Although the data are from the 1970s and 1980s, and are therefore likely to overestimate current atmospheric deposition rates, they provide an indication of the contribution of atmospheric deposition to the Impact Assessment results.

The results are shown in Figure 5; in this figure, the results are shown relative the System A without atmospheric deposition (the "Base Scenario"). It can be seen that atmospheric deposition makes a large difference to the ecotoxicity (terrestrial) and human toxicity values for System A; the values for these two categories are 8.5 and 5.7 times larger respectively than the values without atmospheric deposition. In both cases, the differences are sufficiently large to alter the ranking of the three systems in these two Impact Assessment categories. This suggests that choice of location may be more important in determining ecotoxicity (terrestrial) and human toxicity than use of TSP, Thomas meal and/or lime (i.e. the other sources of heavy metals). In other words, restriction of farming to areas with low atmospheric deposition of heavy metals is likely to be the most effective improvement option for the categories of ecotoxicity (terrestrial) and human toxicity. Based on the data available, focusing exclusively on heavy metals in fertilisers, for example, fails to address the main cause for concern in these categories.

Figure 5. Influence of Atmospheric Deposition On Impact Assessment Results for System A
7.2 Transport of the Harvested Grain To the Breadmaking Facility

In order to investigate the influence of transportation distance for the dried grain on the LCA results, it was assumed that grain from System A travelled 800 km by road from East Anglia in the UK to the breadmaking facility in Switzerland while the distances travelled by the grain from Systems B and C to the breadmaking facility was sufficiently small to disregard in the analysis.

The influence on selected Impact Assessment categories of the additional transportation for System A is shown in Figure 6; here, the “Base Scenario” is System A without this additional transportation. It can be seen that the largest differences are for abiotic resource depletion (energy) (13% increase), global warming (6% increase), acidification (6% increase for both scenarios) and photochemical oxidant formation (20% and 17% increase for minimum and maximum scenarios respectively). For abiotic resource depletion (energy) and photochemical oxidant formation, the additional transportation alters the relative ranking of the three systems under analysis.

Figure 6. Influence of Transportation of Grain On Impact Assessment Results for System A

Note: Soil Compaction and Soil Organic Matter Indicators not shown because they are not influenced by changes in transportation.
The result suggests that choice of location for agricultural production in relation to consumption can be more important in determining the magnitude of some environmental impacts than choice of technology, such as increasing the energy efficiency of operations on the farm. This influence of choice of location is further illustrated in Figure 7 which shows energy utilisation for the different life cycle stages of System A, compared with energy for transporting the grain to different destinations. It can be seen that when grain is transported as far as southern Europe or across to Russia, the energy utilisation for transportation equals or exceeds that required for fertiliser manufacture. In other words, it may be appropriate to focus on minimising the distance between location of agricultural production and points of consumption rather than fertiliser manufacture when seeking to maximise energy efficiency in this system.

Figure 7. Influence of Choice of Location On Energy Utilisation In System A

7.3 Approaches To Account for Use of Manure

In order to investigate the influence of the approach used to account for use of manure in System C on the results of the LCA, the Impact Assessment results for System C were recalculated for two scenarios:

1. Assuming that the burdens associated with use of manure in System C (the Foreground System) are the same as those associated with its use in the Background System. [Previously it was assumed that the burdens were negligible in the Background System (see Section 4.2.2).] In other words, no burdens are associated with use of manure in System C because the avoided burdens in the Background System are equal to the additional burdens from use of manure in System C.
2. Treating manure as a waste product from a livestock system. In other words, no burdens are associated with production and treatment of the manure in System C (i.e. only the burdens associated with its use in System C are assessed in the LCA).

The results are shown in Figures 8 and 9. For Scenario 1, Figure 8 shows that an assumption of equal burdens associated with manure use in the Foreground and Background Systems reduces the Impact Assessment results by more than 10% for acidification (maximum) and eutrophication (N-limited). This is due to the reduction in N-compound emissions. However, the differences are not sufficient to alter the ranking of the three systems.

Figure 8. Scenario 1. Influence of Assumptions About Avoided Burdens On LCA Results for System C

For Scenario 2, Figure 9 shows that the results for System C are significantly changed in a number of Impact Assessment categories, although the ranking of the three systems is altered for only one category (global warming, GWP100). In particular:

- Use of solar energy, land area and the Soil Compaction Indicator are reduced due to the absence of the N-fixing crop from the system under analysis.
• The global warming (GWP100) result means that System C has the lowest value out of the three systems. Previously it had the highest value, and the difference is due to the exclusion of CH₄ emissions from System C when manure is treated as a waste output⁴.

• For acidification (maximum) and eutrophication (N-limited), the value is reduced due to the exclusion of NO₃⁻ emissions to water associated with the N-fixing crop.

• For photochemical oxidant formation (maximum), the value is reduced mainly due to the the exclusion of CH₄ emissions from System C when manure is treated as a waste output.

Figure 9. Scenario 2. Influence of Treating Manure as a Waste Product In System C

The analysis suggests that assumptions about the avoided burdens in the Background System, and whether manure is a waste or a co-product from livestock systems, are important in determining the magnitude of the results for a number of Impact Assessment categories in this farming system. In reality, these choices are at least partly determined by the location of production (as discussed in Appendix VI.4, Section 2).

⁴ It is debatable whether the burdens associated with treatment and storage of the manure should be allocated to the livestock system and/or system in which the manure is used. However, the "best case" scenario was used here (i.e. no CH₄ emissions associated with use of the manure) to demonstrate the greatest possible difference due to treating manure as a waste output.
7.4 Approach To Account for Use of Thomas Meal

In the analysis, the avoided burdens associated with waste disposal of Thomas meal rather than its use in System B were not included in the analysis. In order to test the influence of this omission on the final results for toxicity, the avoided burdens associated with leaching from a landfill of 50% of the heavy metals from the Thomas meal used in System B were included in the analysis. The results are shown in Figure 10. It can be seen that there is a negligible influence on the results, apart from ecotoxicity (aquatic) where System B has a net positive result in this category (i.e. avoided ecotoxicity (aquatic)). There is no difference in ecotoxicity (terrestrial) because this category only assesses emissions to soil, and the difference is negligible for human toxicity because the result in this category is dominated by the emissions to agricultural soil.

Figure 10. Influence of Including Avoided Burdens for Waste Management of Thomas Meal In System B

The sensitivity analysis shows, therefore, that including the avoided burdens for waste disposal of Thomas meal has the potential to significantly alter the Impact Assessment results for ecotoxicity (aquatic). Furthermore, it shows that heavy metals have the potential to make an important contribution to this Impact Assessment category in addition to pesticides. However, realistic modelling of the fate of these heavy metals is necessary in order to draw conclusions, and such modelling is currently problematic due to lack of data. In such a case, it may be more practical to conceptualise the issue without quantitative modelling. Essentially, the analysis concerns the relative environmental benefits of disposing Thomas meal to landfill versus spreading it onto agricultural land. For ecotoxicity (aquatic), the result is determined by the leaching of heavy metals from landfill as opposed to agricultural soil; leaching is likely to be greater from agricultural soil because leachate is not managed and treated from this system as it is in (most) landfills. Therefore, the result for ecotoxicity (aquatic) is likely to be in the range between zero and the value calculated in Section 6 for
System B (where it was assumed that there was no ecotoxicity (aquatic) associated with waste disposal of Thomas meal). For ecotoxicity (terrestrial) and human toxicity, the results are determined by the heavy metals reaching soil and entering the human food chain after initial deposition in landfill as opposed to application to agricultural soil. It seems obvious that the greatest impacts in both these categories will be associated with application to agricultural soil. Therefore, the results in both these categories are likely to be in the range between zero and the values calculated in Section 6 for System B (i.e. assuming no ecotoxicity (terrestrial) or human toxicity associated with waste disposal of Thomas meal).

Having reached this conclusion, the ranking of the three systems for the toxicity categories shown in Figure 3 seems reasonable. Moreover, the implication here is that it is not always necessary to undertake complicated quantitative modelling exercises in order to determine the environmental impacts of some parts of systems under analysis.

### 7.5 Assessment of Toxicity

In order to investigate the influence of choice of toxicity assessment method on the results, toxicity was calculated using both the CST method and the Guinée method (see Appendix II.1 for a description of these two methods). Since Guinée et al. (1996) do not include Impact Assessment factors for the pesticides used in Systems A and B, these were excluded from the analysis. Therefore, the results using the two methods are for assessment of heavy metals and hydrocarbons.

Figure 11 shows that the ecotoxicity (aquatic) results are similar for the two methods, i.e. the relative ranking of the three systems is similar for both methods. However, the results for ecotoxicity (terrestrial) and human toxicity are very different. For ecotoxicity (terrestrial), System C has the highest ranking (i.e. greatest impact) using the CST method and the lowest ranking (i.e. lowest impact) using the Guinée method, although Systems A and B are similar in their relative magnitudes in both methods. For human toxicity, System B has the highest ranking using the CST method and is intermediate between System A and System C using the Guinée method; Systems A and C are similar in their relative magnitudes in both methods.

These results support the examples given under “Toxicity” in Chapter II, Section 4.2.2, suggesting that there are still many uncertainties associated with assessment of toxicity in LCA. These uncertainties are related to assessing the inherent toxicity of different substances (the “effect factors” in LCA terminology) and modelling their fate. Indeed, in commenting on the health implications of a
recent MAFF study on heavy metal contamination of milk and vegetables produced near industrial sites, the Committee on Toxicology of Chemicals in Food, Consumer Products and the Environment (COT) stated that (MAFF, 1998):

“In evaluating the implications for human health, we note the following assumptions and limitations:

a) the chemical forms of the elements in food are not known. The relevance of the available toxicity data is therefore uncertain;

b) the estimates of intake assume that, where an element has not been detected, it is present at the limit of detection. Intakes in these cases are therefore dependent on the limit of detection or other limit assigned and can be regarded as overestimates, possibly by a considerable margin;

c) the toxicity data available to us are inadequate for complete evaluation of any of the elements in the diet, particularly indium;

d) the data are insufficient to allow the identification of groups of individuals who might be particularly susceptible to any adverse effects from dietary intakes of these elements. Consequently, our evaluation applies only to healthy adults.”

Given these uncertainties, it may be questioned whether detailed quantitative modelling of the fate of toxic substances is the most appropriate approach to use in LCA. A possible alternative is to develop an indicator related to properties such as the inherent toxicity, spatial and/or temporal range of substances (see, for example, the work of Scheringer and colleagues: Berg and Scheringer, 1994;
Scheringer and Berg, 1994; Scheringer, 1996; Scheringer, 1997; and Scheringer and Hungerbühler, 1997).

7.6 Changes In Soil Quality: Organic Matter

Method 4 from Chapter V, Section 4.3, for assessment of changes in soil organic matter was used in the analysis. However, if Method 3 is used instead, all inputs of organic matter are credited to the system under analysis. The relevant calculations are in Appendix VI.2, and the results are shown in Figure 12. It can be seen that assessment using Method 3 gives quite different results from Method 4. As discussed in Chapter V, Section 4.3, choice between these two methods is likely to be influenced by the perceptions of the analyst (or others involved in undertaking the analysis).

![Figure 12. Assessment of Soil Organic Matter Using Methods 3 and 4](image)

7.7 Changes In Soil Quantity: Eroded Soil

The environmental impacts associated with eroded soil are due to depletion of this resource and export of organic matter, nutrients and heavy metals in the eroded soil (see Chapter V, Section 5). Although not quantified explicitly in this study, an example illustrates the potential importance of this aspect based on the soil erosion rates for the three systems given in Table 1 and the generic data on quantities of nutrients in eroded soil quoted in Appendix VI.3. Table 8 shows the quantities of nutrients applied and lost in soil erosion for the three systems (per hectare) based on these data. For nitrogen, 10% or more of the applied nitrogen is lost in eroded soil from each system. For phosphorus, over 20% of the applied phosphorus is lost in eroded soil from each system. For potassium, Systems A and B have a deficit of potassium due to loss in eroded soil, and in System C 73% of the applied potassium is lost in eroded soil. These losses are of the same order of magnitude as the losses of nutrients from the farming systems via other routes (see Table 2). Since many of the environmental impacts of the three systems are related to production and use of nutrients (Section
6.2.2), actions to minimise soil erosion are likely to improve the overall environmental performance of these farming systems.

Table 8. Losses of Nutrients in Eroded Soil (Per Hectare)

<table>
<thead>
<tr>
<th></th>
<th>System A</th>
<th>System B</th>
<th>System C</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nitrogen Applied</td>
<td>240 kg</td>
<td>132 kg</td>
<td>141 kg</td>
</tr>
<tr>
<td>Lost in eroded soil</td>
<td>27 kg</td>
<td>14 kg</td>
<td>14 kg</td>
</tr>
<tr>
<td>Phosphorus Applied</td>
<td>26 kg</td>
<td>22 kg</td>
<td>24 kg</td>
</tr>
<tr>
<td>Lost in eroded soil</td>
<td>9 kg</td>
<td>5 kg</td>
<td>5 kg</td>
</tr>
<tr>
<td>Potassium Applied</td>
<td>50 kg</td>
<td>55 kg</td>
<td>215 kg</td>
</tr>
<tr>
<td>Lost in eroded soil</td>
<td>312 kg</td>
<td>156 kg</td>
<td>156 kg</td>
</tr>
</tbody>
</table>

8. Discussion and Conclusions

The study presented in this chapter has demonstrated the methodology developed in previous chapters for assessment of agricultural systems. It has shown that this methodology can be operationalised, apart from assessment of Physical Habitat Degradation which remains problematic due to lack of knowledge (and other factors as discussed in Chapter IV). However, the sensitivity analyses have also shown the dependency of the results on the chosen approach (Section 7). Decisions about how to account for atmospheric deposition of heavy metals, use of manure and Thomas meal, and incorporation of organic matter can make a significant difference to the final results.

The study has also provided insights and examples related to issues raised in previous chapters of this thesis, including:

**System modelling: crop rotations** (see Chapter III, Section 4.2):
- Use of lime contributes more than 10% of the final result in some Impact Assessment categories in System C, and its inclusion in a system under analysis is dependent upon modelling of crop rotations.

**System modelling: functional unit** (see Chapter III, Section 4.1):
- Definition of the functional unit requires consideration of the protein content of the grain, and the downstream processes associated with bread production.
System modelling: soil erosion (see Chapter V)

- Loss of soil makes the greatest contribution to abiotic resource depletion (other).
- Loss of nutrients in eroded soil is likely to make a significant difference to the results (Section 7.7).

Assessment of potential versus actual impacts (site-dependency in LCA) (see Chapter II, Section 4.4; Chapter III, Section 3):

- The magnitude of a number of Impact Assessment category results are dependent upon processes in the soil that often vary between different locations. In particular, the fate of nitrogen applied to the soil (as either synthetic nitrogen fertilisers or farmyard manure) may vary with location, and determines the magnitude of impacts such as global warming (\(\mathrm{N}_2\mathrm{O}\) to air), acidification (maximum) (\(\mathrm{NH}_3\) to air, \(\mathrm{NO}^+\) to water) and eutrophication (N-limited) (\(\mathrm{NH}_3\) to air, \(\mathrm{NO}_3^-\) to water). Nitrogen-related emissions from land used for nitrogen-fixing crops and fallow land ("green manure") also determine the magnitude of these impacts in systems involving these additional land requirements.
- The magnitude of the eutrophication results for different systems is highly dependent upon assumptions about the background conditions, i.e. whether substances are emitted to N- or P-limited environments. Although the difference in results was not sufficient to change the ranking of the three systems considered in this study, it made a big difference to the relative magnitude of the results (see Figure 3).
- The acidification results show that the ranking order of the three systems (Figure 3) depends upon the assumption about the role of N-compounds in acidification. This is a site-dependent factor (see Chapter II, Section 4.2.2).
- Inclusion of the burdens associated with transportation of the dried grain to the breadmaking facility makes a difference greater than 10% to the results for abiotic resource depletion (energy) and photochemical oxidant formation in System A (Section 7.2). This supports the assertion in Chapter III, Section 3, that choice of location should be legitimised as a valid difference between alternative systems alongside choice of technology.
- Inclusion of atmospheric deposition of heavy metals alters the ranking of the three systems for ecotoxicity (terrestrial) and human toxicity (Section 7.1). This suggests that choice of location (i.e. in areas with higher or lower rates of atmospheric deposition of heavy metals) may have a greater influence on these Impact Assessment results than choice of technology.
Uncertainty in modelling systems (see Chapter II, Section 6):

- The magnitude of the toxicity impacts are largely dependent upon a) the Impact Assessment method used in the analysis (as shown above for the CST and Guinée methods (Section 7.5)), and b) the fate of pesticides and heavy metals in the system(s) under analysis (as demonstrated for heavy metals in Thomas meal (Section 7.4)). Since modelling the fate of these substances is difficult due to a lack of knowledge, assumptions made at this stage can have a large impact on the results. This has implications for the acceptability of LCA as a legitimate form of analysis among different stakeholders (see Chapter VII, Section 2.4).

- Lack of knowledge also has an important effect on the Photochemical Oxidant Formation results. The study has shown that the POCP factors chosen for the analysis have a large impact on the results (i.e. the difference between the minimum and maximum scenarios shown in Figure 3 for Photochemical Oxidant Formation).

Simplification in LCA studies (see Chapter II, Section 6):

- The summary of the results in Section 6.2 suggests that there is potential for simplifying LCA methodology by considering a) a restricted number of substances, and/or b) a restricted number of life cycle stages. [Although, of course, comprehensive assessment is preferable given the availability of sufficient time and financial resources.] It may be possible to derive a limited list of substances that contribute most of the environmental impacts, or focus on particular life cycle stages that make the greatest contribution to the different impact categories. However, development of this approach requires further research to confirm whether the results of this study are replicated for other crop and livestock production systems.

LCA and decision-making (see Chapter I, Section 4):

- Assessment of organic matter incorporation in LCA is dependent upon the perceptions of different stakeholders (Section 7.6).

In summary, I would highlight four insights that have been gained by undertaking this study. Firstly, the study has shown that capital equipment makes a contribution greater than 10% to a number of Impact Assessment categories. This is in contrast to most industrial production systems where capital equipment is ignored because it makes a negligible contribution to the overall results. The explanation is found in the fact that agricultural equipment tends to be large, subject to considerable "wear and tear," and is used relatively infrequently compared with, for example, industrial production line machinery. As a result, the environmental burdens associated with its production, maintenance
and repair tend to be relatively high per unit of agricultural output (i.e. the most common functional unit in LCA studies of agricultural systems).

Secondly, total energy use in farming systems requires assessment of solar as well as fossil fuel energy. Previously, only fossil fuel energy has been assessed in LCA studies. Figure 13 shows how the assessment changes when both energy sources are included in the assessment. Use of fossil fuel energy is similar in all three systems whilst use of solar energy increases from System A to System C. For each 1 GJ in the equivalent quantities of harvested grain, the total solar energy input is 338 GJ in System A, 549 GJ in System B, and 1,174 GJ in System C. One interpretation of these results is that, given the similar fossil fuel utilisation in all three systems per functional unit, use of fossil fuel energy in the form of synthetic fertilisers and pesticides (Systems A and B) rather than as additional machinery and diesel requirements (System C) increases the efficiency of conversion of solar energy to energy for human nutrition.

Thirdly, the study has demonstrated that choice of location is more important in determining impacts than choice of technology (i.e. different farming practices) for a number of Impact Assessment categories. The study has demonstrated how choice of location intrinsically determines the lime requirement, the quantity of energy used for transporting products from the farm to the point of consumption, and the toxicity associated with atmospheric deposition of heavy metals in any system under analysis. Geographical variation in all these factors has been shown to alter the LCA results by more than 10% in some Impact Assessment categories. Furthermore, it also determines the magnitude of the eutrophication and acidification results. The implication is that it may be equally, or even more relevant to determine preferred locations of production rather than preferred farming practices in seeking to maximise the environmental performance of farming systems.

Fourthly, the study has suggested that the organic system (System C) has a worse environmental performance than the intensive and integrated systems (Systems A and B) in a majority of Impact Assessment categories. This is linked with use of lime to alter the soil’s pH, the lower yield in System C, use of farmyard manure as a source of nutrients, and use of more agricultural machinery in System C than in Systems A and B. As discussed above, use of lime is a location-dependent aspect and so cannot be regarded as an intrinsic requirement for the organic system as opposed to the intensive and integrated systems. However, the other factors are intrinsically related to the organic farming system because it does not use synthetic fertilisers or pesticides. In other words, its yield is likely to be lower due to pests and less available nutrients, farmyard manure (or
other “natural” sources of nutrients) must be used rather than synthetic fertilisers, and the farmyard manure takes longer to apply than specially formulated synthetic fertilisers (i.e. agricultural machinery must be used for longer periods of time). Thus, the environmental preferability of the organic system over the intensive and integrated systems in this case study can be questioned for a number of Impact Assessment categories. However, the organic system does have a better environmental performance than the other two systems in the toxicity categories (excluding the contribution to these categories from application of lime and atmospheric deposition which, as noted above, are not intrinsically related to this type of farming system). It seems, therefore, that there is a basic trade-off here between the greater toxicity related to use of synthetic fertilisers and pesticides
for Systems A and B, and greater impacts in all other Impact Assessment categories for System C. Ultimately, then, the overall preferability of System C over Systems A or B is determined by the relative weighting of toxic impacts against other impacts. Since attitudes towards toxicity are often determined more by peoples’ perceptions than (the inadequate) scientific evidence, it is not surprising that the debate over the environmental preferability of organic farming as opposed to more conventional, intensive farming is an emotive one.

Although these four insights are derived from a specific case study, it is likely that they would be replicated in other agricultural systems; an exception is the comparison between organic and intensive farming systems because here the results are dependent upon the specific farming operations and sources of nutrients in the compared systems. As well as insights into the environmental impacts of farming systems, the case study has also demonstrated a number of implications for the development of LCA methodology. I explore some of these in the next chapter on “LCA and Decision-Making.”

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1. Introduction

In the previous chapters, I have developed a methodology for Life Cycle Assessment of agricultural systems. This has involved consideration of aspects such as system boundaries, definition of the functional unit, allocation, and assessment methods for biodiversity and soil quality. Use of this methodology facilitates comprehensive analysis of the overall environmental impacts of systems involving agricultural production.

However, such studies have substantial associated resource implications - of both time and money. Therefore, it is reasonable to ask whether the usefulness of these studies outweighs their human and financial resource requirements. Indeed, there is growing concern about the eco-efficiency of LCAs in general, where eco-efficiency measures the ecological benefits of using an environmental management tool in relation to its economic costs. Here, the “ecological benefits” are defined as a tool's ability to provide correct, representative information and to support ecologically beneficial decisions (Schaltegger, 1997).

In a recent survey, it was found that common barriers to the adoption of life cycle approaches by European industry included (Berkhout, 1997):

- Methodological differences across firms and sectors
- High cost
- Poor access to data
- Mismatch between the needs of firms and the results of studies
- Problems with communication of results.

The LCA research community has, until recently, focused almost exclusively upon addressing the first of these barriers. Indeed, that has been the main subject of this thesis up to this point. In this chapter, therefore, I step back and take a more arms-length look at LCA and its role in environmental
management. Rather than starting with the *de facto* assumption that it is beneficial to use LCA, it is useful to take a second look at the extent to which LCA actually fulfills users' requirements. Therefore, firstly I ask, "How useful are LCA results?" (Section 2). This is a slightly different approach from that used by a number of other LCA practitioners who have focused on developing criteria for comparing existing LCA methodologies, in particular at the Impact Assessment phase (see Lindeijer, 1996, p.87). By initially focusing on the usefulness of the results obtained from existing methods rather than the methods themselves, the role of the decision-making context in defining "usefulness" becomes apparent.

This leads into a discussion about the use of LCA in different decision-making contexts (Section 3) and the structure of LCA as an environmental management approach (Section 4). I then draw the various arguments together to comment on the implications for future development of LCA methodology (Section 5). I conclude in Section 6 that "LCA and decision-making" is a research area that should have a much higher priority in the LCA research community.

In the following sections, I take it as a given that the life cycle concept is valuable in environmental management. Instead, the discussion focuses on the usefulness of LCA as an environmental management tool. I distinguish between the life cycle concept and LCA based on the definitions put forward by the SETAC Working Group on Conceptually Related Programmes. Here, we define an environmental management concept as an idea about how to achieve sustainability, and a tool as typically consisting of a systematic step-by-step procedure and a mathematical model (SETAC Working Group, 1997).

2. How Useful Are LCA Results?

In order to measure the usefulness of LCA results, I use four criteria:

- Are the results accurate?
- Are the results relevant?
- Are the results understandable and meaningful?
- Is the approach accepted as a legitimate form of analysis?

1 Powell *et al.* (1997) do acknowledge this dimension in their criteria for assessing different weighting schemes in Valuation, suggesting that two of the criteria should be goal consistency and goal acceptability. They define goal consistency as meaning that the "weights must be consistent with the goal." Goal acceptability is defined as...
The first three criteria are adapted from the literature on environmental indicators (UK Round Table on Sustainable Development, 1997). The fourth criterion is based on the fact that LCA results will lack credibility if the method is not understood and accepted by the users of the results. These criteria are used to assess the usefulness of LCA results in the next four sub-sections. I assume use of the methodology developed through SETAC activities (as presented in Chapter II), and the Problem-Oriented approach to Impact Assessment as a basis for discussion.

2.1 Accuracy of the Results

In the last three years, a number of researchers have questioned the precision of LCA results. For example, Pohl et al. (1996) state that, “The total error of an LCA can easily become larger than the calculated differences of ecological impacts of products and services.” The precision of an LCA study can be compromised at a number of points. At the Inventory Analysis phase, these may include:

- **Use of single values for burdens**
  Burdens associated with processes analysed at the Inventory Analysis may be given as single values when, in reality, the actual magnitude of burdens may be variable. For example, the magnitude of nitrate leached from agricultural land may vary according to weather conditions, and emissions from a flue stack may vary with small changes in the operating conditions.

- **Use of generic rather than specific data**
  For example, “default” generic data on emissions from combustion plant may be used rather than data on emissions from combustion of specific fuels such as gas or fuel oil. Obviously, the latter are more representative if the heat source is known.

- **Omission of relevant data**
  Relevant data may be omitted due to lack of availability.

At the Impact Assessment phase, the accuracy of a study can be compromised by:

- **Assessment of potential rather than actual impacts**
  This issue was introduced in Chapter II, Section 4.4. Impact Assessment methods were originally developed to assess potential impacts but more recently there has been interest in developing site-

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meaning that “the goal should be rooted in some form of social acceptability, that is the goal should not reflect unrepresentative interests. Goals should not be arbitrary, nor should they reflect ‘false concerns.’”
dependent Impact Assessment methods (see, for example, Potting and Hauschild, 1997a, 1997b). Examples of assessment of potential impacts include: i) use of generic Impact Assessment factors for eutrophication rather than factors for nitrogen- or phosphorus-limited environments, and ii) calculating acidification potential without reference to the sensitivity of receiving media.

- **Inferior Impact Assessment characterisation methods**
  For example, it was shown in Chapter II, Section 4.2.1 (Table 6) that resource depletion results vary widely depending upon the Impact Assessment factors used in the analysis. The drawbacks of assessing resource depletion by reference to reserves are widely acknowledged, but no practical alternative is available at the present time. Another example is Physical Habitat Degradation where uncertainty in the data required for the analysis may overshadow the results (see Chapter IV, Section 9).

- **Missing data**
  For example, in the USES 1.0 model used to characterise the fate of toxic substances, a number of inappropriate default values have been used, according to Tukker (1997, p.21).

- **Wrong data**
  Again according to Tukker (1997, p.21), the USES 1.0 model calculates the fate of some substances using data from out-of-date literature, or data that are just wrong.

Of course, many of these inaccuracies can be eliminated by use of good data. However, given that many of these data are not currently available, one must consider whether efforts to collect such data should be promoted as an eco-efficient - or even realistic - approach in environmental management. In Section 5, I discuss some alternatives. The inaccuracies can also be reduced by changing the presentation of LCA results to make the uncertainties inherent in the results more obvious, and this is also discussed in Section 5.

### 2.2 Relevance of the Results

The relevance of LCA results cannot be assessed without reference to the decision-making context for an LCA study. This is because results relevant in one decision-making context may be irrelevant in another one. The decision-making context is defined by a) the type of question being asked, i.e. the purpose of the study, and b) the stakeholders involved in, or affected by, the decision (Wrisberg and Gameson, 1998, p.10). For example, a company using LCA for internal decision-making about modification of existing products may only be interested in certain prioritised environmental impacts,

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2 Here, a stakeholder is defined as "someone with a legitimate interest in the decision" (Cowell et al., 1997, p.44).
or even a restricted number of emissions rather than impacts. In this decision-making context, an LCA that required the company to assess a comprehensive range of impacts and/or emissions would yield at least some irrelevant results.

Another example might be a Non-Governmental Organisation (NGO) that wants to influence consumers’ purchasing decisions about the environmental trade-offs in egg production between free-range and battery chickens. In this case, conventional LCA (as developed in previous chapters of this thesis) can be used to assess the environmental impacts of differences in feed requirements, land use, and fertiliser production (from chicken litter) between the two production systems. However, a study that ignores the implications for animal welfare has limited relevance to the decision-making context. In this case, including a quantitative or qualitative estimation of the trade-offs in animal welfare between the two systems will be essential if the study is to meet its purpose.

These two examples suggest that the usefulness of an LCA can be enhanced by considering the purpose of the study, and the needs of stakeholders, in the process of undertaking an LCA. In other words, this is an argument for greater flexibility in LCA methodology to adapt it to the needs of stakeholders. However, adoption of such an attitude raises another set of questions: How much can a study be simplified before it can no longer be regarded as an LCA? How restricted can be the definition of system boundaries? Can a study involving both quantitative and qualitative data be regarded as an LCA? In other words, when is an LCA not an LCA? Furthermore, beyond the immediate requirements for a study to support decision-making, there is another role for LCA: raising awareness about the life cycle impacts of activities. Use of restricted system boundaries interferes with this role of LCA because certain sub-systems and/or impacts are consciously omitted from a study.

This suggests that there is a tension between making LCA more responsive to the needs of stakeholders and its role in raising awareness about the overall environmental impacts of human activities. Christiansen (1997) addresses this aspect in discussing simplification of LCA, underlining the importance of “reliability assessment” (checking that the results are reliable enough to justify the conclusions drawn) in the process of simplifying an LCA study. I return to this discussion in Section 4 below.
2.3 Understandability of the Results

LCA results can only be useful if they are understood by users of a study. Responsibility for making the results understandable rests with the Impact Assessment phase of an LCA. As Udo de Haes (1996, p.11) notes, "The Impact Assessment phase is necessary because information on the inputs and outputs of the product system, as identified in the Inventory Analysis, is often not sufficiently relevant for environmental management, if it is not explicitly interpreted in relation to environmental problems." This statement makes it clear that an LCA consisting of an Inventory Analysis but no Impact Assessment can be sufficient if its results are understandable to users, but that in most cases Impact Assessment is required to interpret the Inventory Analysis results.

It is easy to lose sight of this purpose of Impact Assessment in developing technically elegant assessment methods. For example, development of multiple Impact Assessment categories may increase confusion rather than understanding among users of a study. For panel assessment methods used in Impact Assessment, Hofstetter (1996, p.18) refers to the "cognitive stress" suffered by panellists asked to weight different environmental burdens. Cognitive stress may also be suffered by users of LCA results if they do not have a good understanding of the different implications of environmental problems.

In considering choice of indicators for conveying concepts of sustainable development to the public, the UK Round Table on Sustainable Development has suggested that "public resonance" should be an important requirement. In other words, indicators should be understandable and meaningful (UK Round Table, 1997, p.15). They cite the example of using the population of salmon in the Sustainable Seattle project as an indicator of a variety of issues such as river pollution, biodiversity and the state of the local economy. This indicator had particular resonance with the public because salmon have a special status in the region's cultural heritage (UK Round Table, 1997, p.17). In the same way, it can be argued that LCA results should have resonance with users of studies.

It may be that existing approaches to Impact Assessment are not the most appropriate ones for minimising cognitive stress among users or increasing the resonance of LCA results. I return to this issue in Section 5.3.

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3 In defining cognitive stress, Hofstetter (1997, p.18) states that people suffer from it if "the questions or problems are not clearly stated, the information is not available or present on all relevant aspects, the attributes or alternatives are conflicting with too much [sic] different objectives, the decision maker is not used to compare the objectives at issue, etc."
2.4 Acceptability of LCA As a Legitimate Form of Analysis

If LCA results are to inform and guide decision-making, LCA must first be accepted as a legitimate form of analysis by users of studies. Establishment of LCA as authoritative suffered a number of setbacks in the early 1990s when LCA was used by some companies to promote their products: in a number of cases LCAs of equivalent products produced by market competitors gave contradictory results. The implication was that a sponsor paid for the results more than for the study itself. This led to increased efforts to standardise LCA methodology, and the promotion of Peer Review as part of the process of undertaking an LCA. However, in the late 1990s the debate has come full circle and the benefits of standardising LCA, and in particular Impact Assessment methodology, are under review. Researchers such as Finnveden (1997) and Tukker (1997, p.24-7) argue against standardisation for Impact Assessment methods. They suggest that peoples' acceptance or rejection of Impact Assessment methods is determined by their fundamental attitudes towards human interactions with the environment. Since people have different attitudes, widely acceptable Impact Assessment methods are unlikely to emerge. However, on the other hand, Udo de Haes (1996, p.13) states that current SETAC and ISO activities are aiming to increase the level of standardisation for Impact Assessment methods.

An interesting example is provided by Bras-Klapwijk (1997) and Tukker (1997), related to the ongoing debate on the environmental impacts of using PVC in Sweden. They suggest that LCAs related to this issue have been of limited use because of the different frames of actors in the debate. A frame is “a perspective from which an amorphous, ill-defined, problematic situation can be made sense of and acted on. Framing is a way of selecting, organising, interpreting, and making sense of a complex reality to provide guideposts for knowing, analysing, persuading, and acting” (Rein and Schö, 1993, p.146). Frames are rooted in peoples’ evaluative paradigms, i.e. basic attitudes to the nature of reality4. Use of LCA as an analytical approach has not been accepted by all actors in the debate because they have different frames, and LCA is rooted in a particular frame. Bras-Klapwijk (1997, p.51) describes this frame as the rational approach. Some basic assumptions of the frame used in LCA are listed in Table 1, alongside the corresponding attitudes of some environmental NGOs who have been vocal in their opposition to use of PVC.

4 A paradigm is defined in the Chambers English Dictionary as “a conceptual framework within which scientific theories are constructed.” This sense, which is appropriate to the discussion here, derives originally from the work of Thomas Kuhn (1962).
Table 1. Basic Assumptions Underlying LCA Compared With Basic Assumptions of NGOs (Drawn From the Debate Over Phasing Out PVC In Sweden)

<table>
<thead>
<tr>
<th>Category of Assumptions</th>
<th>Basic Assumptions In LCA</th>
<th>Basic Assumptions By Some Environmental NGOs</th>
</tr>
</thead>
<tbody>
<tr>
<td>Conceptual</td>
<td>Trade-offs can be made between the environmental impacts of different emissions.</td>
<td>Trade-offs cannot be made between releases of some substances: their elimination should be the only guideline. In particular, prevention of irreversible contamination of the environment should be given top priority, i.e. avoidance of releases of toxic substances with very long residence times should be a priority.</td>
</tr>
<tr>
<td></td>
<td><strong>Generic weighting factors for different environmental impacts can be defined</strong></td>
<td>Use of weighting factors is a political decision and should not be “imposed” by any one interest group in society.</td>
</tr>
<tr>
<td>Attitudes to data</td>
<td>Inventory data are of reasonably good quality.</td>
<td>Inventory data may be inaccurate and/or generalised, and presenting single values gives a false appearance of objectivity.</td>
</tr>
<tr>
<td></td>
<td>Impact Assessment predicts the effect, fate and final impact of substances with reasonable precision.</td>
<td>We have a very incomplete understanding of the fate of substances, particularly persistent ones. Synergistic effects between substances can be important.</td>
</tr>
<tr>
<td></td>
<td>Reasonable assumptions are made where there are data gaps.</td>
<td>Uncertainties in data should be weighted more heavily (i.e. more negatively) in environmental analysis.</td>
</tr>
<tr>
<td></td>
<td>All environmentally relevant substances are included in the LCA.</td>
<td>Emissions of micro-pollutants may be omitted. For example, processes involving chlorine may miss out emissions of chlorinated micro-pollutants.</td>
</tr>
<tr>
<td>Inclusion of different types of environmental impacts</td>
<td>LCA is not concerned with inherently qualitative, subjective aspects such as animal welfare and landscape degradation.</td>
<td>Qualitative aspects are important and should be part of any environmental analysis.</td>
</tr>
<tr>
<td></td>
<td><strong>Risks due to accidents and other adverse events are not relevant for inclusion in LCA.</strong></td>
<td>Risks due to accidents and other adverse events should be an important consideration in any environmental analysis.</td>
</tr>
</tbody>
</table>

aN.B. This is not a consensus view in the LCA research community!

Although Table 1 lists differences in attitudes between the frame used in LCA and the frame used by some environmental NGOs, examples of differences can also be identified between the LCA frame and that of some commercial companies. One is the current debate over the “less-is-best” approach used in Impact Assessment and the alternative “above-threshold” approach (White et al., 1995). In the “less-is-best” approach, all burdens are assessed as contributing to environmental impacts whether they raise concentrations in receiving media above threshold levels or not. However, in the “above-threshold” approach, described by White and colleagues (White et al., 1995), and subsequently developed by consultants at two environmental consultancies in the United States (Hogan et al., 1996;

5 In the above-threshold approach, emissions with local and regional impacts are assessed at Impact Assessment only if they are released in geographical areas that fail to meet standards for ambient concentrations of these emissions. Emissions released in areas that meet these standards are not included in the Impact Assessment (Hogan et al., 1996).
Tolle, 1997), only burdens raising concentrations above the threshold at a local/regional level are assessed in the LCA. It could be argued that the different attitudes towards emissions into the environment represented by these two approaches, are rooted in different frames and their underlying evaluative paradigms.

In fact, there is an interesting link here with the research work of anthropologists linking different cultural perspectives with ways of perceiving nature. It builds on the work of Mary Douglas (1970) who suggested that ways of seeing the world are linked with different forms of social organisation. She suggested that these forms of social organisation are arranged along two continuums: a “group” dimension describing peoples’ membership of societal groups, and running from strong individualism to strong collectivism; and a “grid” dimension describing the extent to which peoples’ activities are open to individual choice, and running from restriction to independence. Figure 1 shows this “Grid-Group” model. The combination of these two continuums produces four different forms of social organisation, each of which is characterised by certain ways of thinking and acting (Milton, 1991):

- **The fatalist** has a high degree of individualism but is subject to prescription. This person’s freedom of choice is restricted and there is a lack of support from a group. These people feel manipulated by a system over which they exercise no control.
- **The entrepreneur** is highly individualistic and independent. The main motivation is personal profit, and the market is the principal mechanism through which it is achieved.
- **The hierarchist** belongs to groups whose actions are prescribed by others or by “the system.” There is a strong emphasis on central control, and procedures are more important than end results.
- **The egalitarian** belongs to a collective with a strong group membership but which defines its own membership criteria and makes its own rules. Personal profit is less important than the “general good” which may be interpreted as narrowly as “the continued existence of the group.”

Michael Thompson and other anthropologists used the Grid-Group model as a basis for categorising ecologists’ different views of nature. They suggested that there are four “myths of nature” that are found among natural resource ecologists managing forests, fisheries and grazing lands. The myths can be illustrated by a ball acted on by gravity on a surface, as shown in Figure 2. The fatalist views nature as capricious: “there is no knowing what nature will do next and no use theorising about it.”

The entrepreneur views nature as robust, and is therefore not particularly concerned about environmental impacts because s/he regards nature as able to stabilise itself. The hierarchist views nature as robust within certain limits, and is therefore concerned about keeping environmental
Figure 1. The “Grid-Group” Model of Social Organisation

GRID
prescription

Fatalist

Hierarchist

Entrepreneur

Egalitarian

GROUP
collectivism

individualism

independence


Figure 2. The Four Myths of Nature

GRID
prescription

Fatalist

Hierarchist

Entrepreneur

Egalitarian

GROUP
collectivism

individualism

independence


impacts within these limits. Finally, the egalitarian views nature as fragile, and seeks to avoid perturbations of nature (Milton, 1991; Douglas, 1992).

There has been much debate over the usefulness of this model, given that any one person may act in ways that are consistent with a number of the different myths of nature. For example, an individual
might a) drive a car to work because it is more convenient than using public transport (entrepreneurial perspective); b) lobby the government for tighter pollution controls (hierarchist perspective); and c) buy a more expensive phosphate-free washing powder because it is good for the environment, and the good of the environment should take precedence over personal profit (egalitarian perspective) (Milton, 1991). Nevertheless, this model is useful not because it is an accurate description of reality but because it yields a greater understanding of the reality of any situation.

Applying the model to the observations noted above about attitudes towards LCA of different sectors in society, there are some interesting parallels. For example, the objections to LCA voiced by some environmental NGOs (as listed in Table 1) suggest that their attitude to pollution is based on the “nature is fragile” perspective. The “less-is-best” approach in Impact Assessment is also more consistent with a “nature is fragile” perspective than the “above-threshold” approach which falls within the “nature is robust within limits” perspective. Those adopting the “nature is robust” or “nature is capricious” perspectives are unlikely to perceive any value to use of LCA. This is because those adopting these perspectives will regard environmental management in general as unnecessary, either because they believe nature is capable of adapting to different impacts (“nature is robust”), or because they regard nature as completely unpredictable and so there is no point attempting to manage it (“nature is capricious”). This indicates that conflicting views about development of LCA methodology are related to differences between the “nature is fragile” and the “nature is robust within limits” perspectives.

Accepting the existence of different perspectives, Tukker suggests that LCA practitioners should therefore focus on constructing alternative sets of indicators reflecting peoples’ different paradigms (Tukker, 1997, p.24-5). Bras-Klapwijk suggests that LCA methodology should “be able to accommodate more perceptions, in other words become a hybrid, or alternative methods should be developed that can accommodate perceptions which are not accommodated by the current LCA methodology” (Bras-Klapwijk, 1997, p.58). They both emphasise the dangers of presenting LCA as an apparently objective and neutral tool because use of LCA results may then undermine the arguments advanced by others using different – and equally valid – frames that are perceived as more subjective and, therefore, biased towards the values of those advancing these arguments. Of course, in a society where “objective” and “subjective” arguments are perceived as equally valid, this type of discrimination is not relevant. However, at least in Western societies, the more objective, rational, quantitative approach is often regarded as superior to alternative approaches (see, for example, the discussion in Burningham (1996, p.9-11) on assessment of objective and subjective impacts in Social Impact Assessment).
It follows from this discussion, therefore, that further consideration should be given to accommodation of different frames in LCA if LCA results are to be regarded as useful by different stakeholders in society; I return to this issue in Section 5.2. However, it is also important to consider how LCA can, or should, be adapted for different purposes, and this is the subject of the next section.

3. The Purpose of LCA Studies

In Section 2.2, it was suggested that the relevance of an LCA’s results cannot be assessed without reference to its decision-making context, and that this decision-making context is defined by a) the purpose of the study, and b) the stakeholders involved in, or affected by, the decision. In this section I consider the purpose of LCA studies, and in Section 4 I discuss the role of stakeholders.

Two types of purpose can be defined in LCA studies:

1. Decision-making
   In this context, LCA results are used to support specific decisions. These decisions may be categorised in different ways. A common categorisation is into operational and strategic decisions (Wrisberg and Gameson, 1998). Operational decisions are concerned with small changes of small-scale systems with a short time horizon; strategic decisions are concerned with large and possibly qualitative changes of large-scale systems with long time horizons. Examples of operational decisions include modifications in the design of an existing product by a commercial company, or the design or operation of a manufacturing process by a chemical company. Examples of strategic decisions are found in companies and governments developing policy. For example, a company may decide to move from a product-oriented to service-oriented focus for its operations, and this could be described as strategic decision-making.

2. Awareness-raising
   In this context, the main focus is on the learning experience and educational benefits of undertaking an LCA study rather than making a decision; the desired outcome is “increased understanding.” It is characterised by open-ended enquiry and objectives concerned with gaining insights into the system under analysis rather than arriving at a definitive set of results. Studies may be used to raise the awareness of the person(s) conducting the study (“learning”) and/or to raise the awareness of others by communication of the study results. In both cases, it can be expected that the information provided by the study will affect subsequent decisions by those
receiving the LCA results. For example, a company may conduct an LCA to identify the most significant environmental impacts associated with its products ("learning") but at some point this may lead to a decision to phase out use of a particular material.

Having identified these contexts for use of LCA, the question now arises as to whether LCA should be used in the same or different ways in these contexts. In other words, should there be flexibility in LCA methodology to adapt it to different contexts? And in relation to the earlier discussion (Section 2.1), when is an LCA not an LCA? To address this issue, it is useful to revisit the SETAC "Code of Practice" and its listed objectives for LCA. These are defined as:

- To provide as complete a picture as possible of the interactions of an activity with the environment
- To contribute to the understanding of the overall and interdependent nature of the environmental consequences of human activities, and
- To provide decision-makers with information which defines the environmental effects of these activities and identifies opportunities for environmental improvements (Consoli et al., 1993, p.5).

It can be seen that these objectives fit well with the use of LCA in an awareness-raising context. However, interestingly LCA to date has mainly been applied in operational decision-making. It is not surprising, therefore, that there is some discontent with current LCA methodology among decision-makers who are attempting to use a methodology which has been developed to satisfy objectives that are not be entirely relevant to their decision-making contexts. Indeed, the current interest in development of simplified LCA methods can be explained as an attempt to overcome this misfit between current LCA methodology and the needs of decision-makers.

However, to return to the earlier question, how far can LCA be simplified before it can no longer be called LCA? In order to answer this question, it is necessary to define the essential characteristics of an LCA study that differentiate it from other environmental management approaches. I tentatively suggest they are:

- A unit of analysis (functional unit) defined as the service provided by the system under investigation in the study.
- A system model consisting of a quantitative representation of the flows of matter and energy from cradle-to-grave associated with provision of the functional unit.
- Consideration of multiple environmental impacts in the study (as opposed to a single-issue based study focused on, for example, waste generation or energy use).
• Detailed justification for omission of relevant sub-systems or impacts given in a definitive list of categories of environmental impacts.

The flexibility in LCA methodology to adapt it to different decision-making contexts is provided by the last characteristic, and it is this characteristic that requires further research effort. For example, what procedure should be used to justify omission of particular sub-systems and/or impacts? What is the role of stakeholders in this procedure? What should be the definitive list of environmental impact categories? The purpose in attempting to answer these questions should be to enhance the usefulness of LCA results while at the same time maintaining the credibility of LCA among as wide a group of stakeholders as possible. It is no easy task!

4. The Role of Stakeholders In LCA: LCA As a Process

In Chapter 1, I suggested that LCA is a type of systems analysis. In systems analysis, models are used as an integral part of studies. Models can be described as simplified representations of reality. A clear purpose for a study must be defined prior to building a model so that relevant components are included within the system model boundaries. It therefore follows that many types of models can be defined for a system, depending upon the purpose of a study (Baumann, 1995, p.22). Indeed, the procedure by which the boundaries for a system model are selected is just as integral to the success (i.e. usefulness) of a study as the model itself. The procedure by which the results of the model are interpreted is also integral to the success of the model. This implies an iterative approach in a study so that the final model closely fits the study’s purpose and delivers the most useful results.

Relating this to LCA, it becomes clear that the system model in LCA is the “quantitative representation of the flows of matter and energy from cradle-to-grave in relation to the functional unit” (Baumann, 1995, p.31). The procedure consists of the Goal Definition and Scoping, Inventory Analysis, Impact Assessment and Interpretation phases of the LCA, as discussed in Chapter II. Some might argue that Impact Assessment should also be regarded as part of the model, and this is at the heart of the current debate about standardisation of Impact Assessment (as discussed above). However, we suggest elsewhere that the quantitative flows of matter and energy should be regarded as the core LCA model, and that analysts should be able to choose between different “bolt-on” Impact Assessment models to fit the purpose of their studies (Baumann and Cowell, forthcoming).

6 Udo de Haes (1997) gives a default list but notes that it is provisional because the categories are not yet sufficiently systematically defined.
This distinction between model and procedure is generally not clearly articulated in LCA methodology guides (Baumann, 1995, p.40). Indeed, until recently the emphasis has been upon development of the LCA model, and it is only in the last couple of years that more attention has been given to the role of Goal Definition and Scoping, and Impact Assessment in shaping this model7.

One way of emphasising the importance of the procedure in LCA is to describe LCA as a process rather than as a tool in environmental management. Here, a tool is defined as “a means of combining information in a form which can be used in decision-making processes,” and a process as “a way of using and integrating different tools with stakeholder expectations and other decision parameters to meet one or more of the requirements for a decision” (Cowell et al., 1997, p.7). The relationship of tools and processes in decision-making is shown in Figure 3. It is worth noting from this diagram that all stages of decision-making are shaped by the Goals and Requirements, and I return to this aspect in Section 7.

The distinction between tools and processes helps to clarify the different ways in which LCA methodology is being developed. On the one hand, LCA has been developed to assess potential environmental impacts without regard to site-specific conditions, and in some cases using generic weighting factors (see, for example, Heijungs et al. (1992a, 1992b) and the approach used in the Ecolndicator manual (Goedkoop, 1995)). This is analogous to using LCA as a tool. On the other hand, some practitioners argue that the more site-specific and subjective, evaluative components of LCA are crucial to the results, and this should be recognised in the methodology. This is part of using LCA as a process. It implies that the process of undertaking an LCA cannot be separated from the deliberative process shown in Figure 3 that leads to a decision.

Both approaches have strengths and weaknesses. As a tool, LCA facilitates easier analysis (because all potential effect factors can be supplied in a “Do-It-Yourself LCA Manual”), and requires practitioners to incorporate a defined set of impacts so that trade-offs between alternatives in the final decision are more transparent. It may give misleading results, however, because potential impacts are often different from actual impacts: local conditions may be critical in determining actual impacts (as noted in Chapter II, Section 4.4, and Chapter III, Section 3). As a process, LCA may produce results that are considered more relevant to the decision under consideration and which have greater acceptance by stakeholders because their concerns have been incorporated into the process. On the other hand, it may allow a smaller group of stakeholders to “hijack” the process and exclude

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7 This can be contrasted with the development of Design for the Environment as an environmental management approach, where the main emphasis has been on procedure (Hodgson et al., 1997).
consideration of environmental impacts that fall outside their immediate interests (but see Section 3). However, given the discussion in Section 2, I suggest that development of LCA as a process is critical to its continued use in supporting environmental decision-making.

5. Implications for LCA

In the previous four sections, a number of ideas have been outlined related to the practical usefulness of LCA studies. The main points emerging from this discussion are:

- There is concern among those who commission LCA studies about the usefulness of comprehensive studies in relation to their high costs.
- The usefulness of LCA studies can be defined in terms of the accuracy of the results, their relevance to users, their understandability, and the acceptability of the LCA approach.
- Apart from the accuracy of results, assessing the usefulness of LCA results implies a need to focus on decision-making contexts.
This suggests a need for greater flexibility in LCA to make it more responsive to the different purposes for carrying out LCAs, and to users’ values (i.e. frames and evaluative paradigms) - in other words, the needs of users.

However, there is a tension between making LCA more responsive to the needs of users and its role in raising awareness about the overall environmental impacts of human activities.

Therefore, in order to maintain acceptance of the legitimacy of LCA as an environmental management approach, definitive guidelines must be developed for the process of undertaking an LCA.

This constitutes a new research area for LCA practitioners. With respect to decision-making contexts, researchers such as Henrikke Baumann (1995, 1998), the study cited above by Frans Berkhout (1997), and the work undertaken in LCANET (Cowell et al., 1997) have begun to explore the inter-relationships between LCA and decision-making processes. With respect to LCA and evaluative paradigms, in particular, publications such as those by Finnveden (1997) and Hofstetter (1996) have shown how LCA is built upon certain fundamental ethical and ideological values, and have demonstrated how these affect the acceptability or otherwise of the approach among different actors in society. The new SETAC Working Group on “LCA and Decision-Making” will take forward this agenda.

In the meantime, I give three examples below of how the issues raised in this chapter can begin to be addressed in LCA.

5.1 Assessment of Potential Versus Actual Impacts

At various points in this thesis, assessment of potential versus actual impacts has been raised as an issue in LCA. In this chapter, I have suggested that such choices can affect the accuracy of LCA results (Section 2.1). This can be illustrated by the example shown in Figure 4, which illustrates a decision-making situation where a choice is to be made between two alternative manufacturing routes for a product requiring a particular chemical. The chemical is manufactured at two specific locations shown as boxes on the arrows in Figure 4; there are also upstream and downstream processes related to production of the final product. In this example, we are concerned with emission of just one substance, “X,” which is assumed to be short-lived with no cumulative effects, and moderately toxic to humans. The numbers on the diagram give the mass of X emitted during chemical manufacture and downstream processing to produce the final product. It can be seen that three units of X are released during chemical manufacture for both systems. Downstream emissions of X vary between
the two systems: five units of X are emitted by System A and two units of X by System B. In System A, the manufacturing site for the chemical is in a remote location while in System B the manufacturing site is adjacent to a large human population. However, no information is available for the location of populations in relation to downstream processes involving emissions of X.

In assessing human toxicity of these two systems, only emissions of X are considered for the sake of clarity. This substance is given a potential Impact Assessment toxicity factor value of three, and so the potential impacts of the two systems are calculated by simply multiplying the total emissions of X by three for each system. This is the conventional approach to Impact Assessment, here called the “Potential Impacts” (PI) approach. An alternative approach is to account for the proximity of human populations to the sites of release of X, called the “Best Practical Estimate” (BPE) approach in this example. Using this approach, emissions of X from the manufacturing site in System A are multiplied by an Impact Assessment factor of zero because no human populations are exposed to these emissions; in System B, an Impact Assessment factor of three is used for these emissions because of the proximity of human populations. For downstream emissions of X in both systems, no data are available on adjacent populations and so the Impact Assessment factor is a range from zero to three.

The results using the two approaches are shown in Figure 4 underneath the arrows. According to the PI approach, System B is preferable because it has a lower Impact Assessment value for toxicity (15 compared with 24 for System A). However, using the BPE approach, the results are different and depend upon the attitude of the person interpreting the results (the “user”). If the user prefers to make decisions based on “worst-case scenarios,” s/he has no reason to prefer either System A or System B. However, if the user wants to choose the system that minimises the likelihood of actual human toxicity impacts, s/he will choose System A because the range of human toxicity values is from zero to 15 while in System B it is from 9 to 15.

Hence, the choice of assessment method and interpretation of the result can give three different results for this decision-making situation: System B is preferable according to the PI approach, and either System A or neither system is preferable according to the BPE approach. However, the BPE approach gives results that are closer to the actual impacts and that make the best use of the information available for the study (i.e. the data on proximity of human populations to the processes under analysis). Therefore, I suggest that this should be the preferred approach given the availability of relevant data.
This example also illustrates another important point concerning presentation of results derived from a combination of generic and more site-dependent data sources. In such cases, it is important that ranges of Impact Assessment factors are used for the burdens where site-dependent information is not available. This is because use of potential Impact Assessment factors in these cases would introduce a bias against studies in which no site-dependent information is available through presentation of "worst-case scenarios" for these systems. Instead, use of ranges of values avoids this bias, leaving users to interpret the results depending upon their attitudes towards the unknown magnitude of environmental impacts.

Figure 4. Example To Illustrate Assessment of Potential Versus Actual Impacts

For agricultural systems, in particular, the role of site-dependency has been illustrated by the case study in Chapter VI. Here, it was shown that the LCA results are influenced by site-dependent aspects ranging from soil quality, to transportation distances, to atmospheric deposition of heavy metals, to background conditions determining the magnitude of eutrophication and acidification. It was demonstrated that these site-dependent factors can have a greater influence on the results than the choice of "technology" (in this case, the range of agricultural practices for cultivating wheat) for a number of Impact Assessment categories. One of the implications of this result is that it may be possible to define zones of preferred geographical areas for production of foodstuffs, based on environmental criteria. This links with the idea behind the Food Miles concept as introduced in Chapter I, but suggests a more sophisticated development of this concept based on life cycle thinking.

5.2 Adaption of LCA Methodology To Accommodate Different Frames of Stakeholders

In Section 2.4, Table 1, a number of basic assumptions in LCA were compared with the basic assumptions of some environmental NGOs. In this section, it was suggested that further
consideration should be given to accommodation of different frames in LCA if results are to be regarded as meaningful and useful by different stakeholders in society. The first assumption listed in the table related to the acceptability of trade-offs in environmental impacts related to different substances included in an analysis; this is a basic assumption in LCA but it is one that is not accepted by some stakeholders. Therefore, it can be asked whether it is possible to adapt LCA methodology so that it can operate without this assumption. In this case, the answer is “no” because a decision to undertake an LCA in the first place implicitly includes an assumption that trade-offs can be made between different types of impacts. If this were not the case, there would be no need to conduct the study because absolute priorities would guide any decisions, and trade-offs would not be a factor in the decision-making process (Finnveden, 1997).

On the other hand, there is scope for adapting LCA methodology to respond to the different assumptions concerning data and inclusion of different types of environmental impacts. For example, supplementary data could be supplied in an LCA on the risks due to accidents and other adverse events associated with different processes in the system under analysis. Qualitative data on animal welfare and landscape degradation could be listed alongside the quantitative data in LCA results. And a range of values can be used to indicate the variability in data in an analysis, as illustrated in Section 5.1 above.

One proposal has been put forward by Hofstetter (1996) for incorporating uncertainty in Impact Assessment data into the Impact Assessment phase. He suggests that weighting factors should be developed for use in Impact Assessment based on at least two main criteria: extent of damage, and inertia (unknown damage). [A third criterion, “need and possibilities for reduction,” may also be incorporated into the methodology depending upon the outcome of future research.] Uncertainty is measured in the criterion “inertia (unknown damage)” by reference to properties of substances such as accumulation tendency and persistency in the environment. This proposal is not developed in detail by Hofstetter, but will be a subject of future research activity.

The argument concerning accommodation of different frames in LCA advanced in Section 2.4 is particularly valid for LCAs involving agricultural production. This is because assessment of Physical Habitat Degradation is an important impact category in assessing agricultural production. Changes in physical habitats are among the most easily perceived impacts of alternative farming practices among the general public (as opposed to, for example, greater use of energy or pesticides on farms). Yet this impact category is one of the most difficult to measure because of lack of knowledge concerning its impacts on biodiversity. As a result, inevitably the weaknesses of adopting the rational, “quantifying”
frame of conventional LCA methodology to develop a method for assessing this aspect become very obvious (as discussed in Chapter IV, Section 9). It is therefore reasonable to question whether this category of impact should be assessed in the same way as other Impact Assessment categories. Perhaps the lack of knowledge, and the diversity of value-based perspectives among stakeholders concerning this type of impact mean that quantitative assessment is inadvisable – and would only undermine the legitimacy of using LCA to raise awareness of the environmental impacts of alternative food production systems. An alternative is to include a qualitative discussion for this Impact Assessment category in the LCA results. This leaves open the possibility of different interpretations of the factors contributing to the impact, and the importance of the impact itself, by stakeholders with different frames.

5.3 Flexibility In Impact Assessment Methods

The importance of the understandability of the results to users of LCA studies was discussed in Section 2.3. It was suggested that the presentation format for the results should aim to minimise “cognitive stress” among users and increase the resonance of the results among users. One possible approach may be to investigate the role of indicators other than the Problem-Oriented Impact Assessment categories for presentation of results. A good example is transportation. At present, the environmental impacts of transportation are assessed as those associated with energy production and use. The burdens are then added to others associated with the system under analysis, and are assessed using the conventional Impact Assessment categories of global warming, photochemical oxidant formation, and so on. However, if the general public are asked about the environmental impacts of transportation, they cite problems such as noise, vibration, risk of accidents, and landscape degradation due to transport infrastructure, alongside direct energy-related impacts such as photochemical smog formation. Therefore, in presenting LCA results to the public, more useful information may be conveyed about environmental impacts by presenting transportation data as “number of kilometres travelled,” or even showing a map of transportation routes, than by subsuming the energy-related burdens within the generic Impact Assessment phase.

6. Conclusions

The emphasis in this chapter has been on building flexibility into LCA methodology to meet the needs of different users and their decision-making contexts. One of the dangers in adopting this approach, however, is that LCA may be over-simplified and compromised in its role of identifying
overall life cycle impacts. As noted above (Section 3), this can only be addressed by developing procedures that require transparent listing and justification of all omissions.

The desire among decision-makers in companies, in particular, to restrict the scope of LCAs to consideration of impacts more directly related to their own operations merits further consideration in itself. Although this type of simplification provides a way of overcoming some of the barriers outlined in Section 1 to adoption of life cycle approaches, it also has its disadvantages. Indeed, it may be that the greatest benefits arise from adopting a “comprehensive” life cycle perspective because it can stimulate new ways of thinking about the delivery of products and services. This may be the key to the long-term financial stability and success of a company in a competitive marketplace. Ralph Keeney, who works in operational research, suggests that decision-makers usually think of decision situations as problems to be solved rather than opportunities that can yield rewards. He goes on to say, “There are two ways to create decision opportunities. One is to convert an existing decision problem into a decision opportunity. Often this involves broadening the context of the problem ... The other way to create decision opportunities is from scratch. You use your creative genius, which can be stimulated by value-focused thinking, to examine whether and how you can better achieve your objectives” (Keeney, 1994). LCA has a role to play in creating decision opportunities by widening the context for consideration of a “problem,” so that creative alternatives can be identified. This role is compromised if, for example, the system boundaries for LCA are restricted, priorities are narrowly defined, and/or greater understanding is not valued as one of the outcomes of a study.

The importance of comprehensive LCA studies of the “awareness-raising” type is illustrated by considering the role of the “Goals and Requirements” shown in Figure 3, Section 4. This diagram shows that the whole decision-making process is shaped by these goals and requirements. However, what shapes the goals and requirements themselves? I suggest that, in fact, the three types of purposes for carrying out LCAs identified in Section 3 (operational decisions, strategic decisions and awareness-raising) are actually linked via the shaping of goals and requirements for each purpose, as shown in Figure 5. In this diagram, the major influence is exerted by studies carried out in an awareness-raising context, through shaping the goals and requirements for strategic and operational decisions. A smaller influence is exerted in the opposite direction by studies conducted to support operational and strategic decisions.
This diagram, then, shows the far-reaching value of creative life cycle thinking. It also suggests that the keys to future sustainability may well be found in the types of "awareness-raising" studies that are typically conducted by academic researchers and others who are granted creative freedom in their work activities. LCA plays a role by helping to shape the world seen by designers, engineers, policymakers and consumers. This should not be forgotten in attempts to standardise and simplify LCA methodology.

References


CHAPTER VIII
CONCLUSIONS

1. Introduction

The preceding chapters have integrated information from a diverse and multi-disciplinary range of sources to address the issues raised by LCA of agricultural systems. Inevitably, this means that some aspects have been addressed in less detail than a specialist in one particular subject might consider desirable. However, the value of this research is to be found instead in the unified approach it provides for environmental assessment, and the insights it yields for improved environmental management. Since environmental management is inherently multi-disciplinary, it is absolutely appropriate to draw on literature ranging from chemical engineering, to ecology, to operations research to sociology. It is equally inevitable that one thesis cannot give an in-depth account of all subjects relevant to this research area. Instead, I have attempted to focus on integration of relevant data and approaches within an environmental life cycle framework.

The layout of the thesis has also reflected development of my research interests in this area. As such, it began by describing the more quantitative, objective tool of LCA, and went on to develop methodology for assessment of agricultural systems. At a number of points, this raised questions about the apparent objectivity of LCA, and the role of value-based judgements in determining results. This led into a consideration of how LCA influences, and is influenced by, different decision-making contexts.

As noted in Chapter I, then, three themes have run through the thesis:

- Development of LCA methodology
- Assessment of agricultural systems
- LCA and decision-making.

In the next three sections, I summarise how the work laid out here has contributed to the research agenda in these areas.
2. Development of LCA Methodology

In Chapter III it was explained that LCA was developed for the assessment of industrial systems; agricultural systems are sufficiently different from industrial systems that this area of application introduces new methodological issues for all phases of LCA. In particular, I have developed new methods for assessing use of solar energy and water, soil quantity and quality, and biodiversity. Within a life cycle framework, these aspects are likely to be most relevant in assessing agricultural rather than industrial systems because use of solar energy and water, and impacts on soil and biodiversity, tend to be greater in agricultural rather than industrial systems. However, the methods can be applied to any system under analysis.

Use of solar energy (Chapter III, Section 5.1) was assessed as the proportion of total incident radiation on the farmed land incorporated into the system under analysis. This implies that, all other things being equal, the harvested crop yield per hectare must be higher and/or the cultivation period shorter for areas with higher incident radiation to score equal to areas with lower incident radiation. In other words, site-dependent assessment forms an integral part of this method.

Use of water (Chapter III, Section 5.2) has been assessed as total use of water (apart from rain that falls onto land in the system) in relation to average annual rainfall per unit area or per year in the cultivated area. Site-dependent assessment, therefore, also forms an integral part of this method, and implies that the extent to which water is a limited resource is defined by local rainfall patterns.

For soil quantity and quality (Chapter V), I have developed Organic Matter and Soil Compaction Indicators, and demonstrated how eroded soil can be assessed as part of abiotic resource depletion. Additionally, I have shown how inclusion of soil quantity and quality requires careful modelling to account for infrequent activities that benefit the crop(s) under analysis and for changes in soil nutrient levels. This was demonstrated using a case study of breadmaking wheat production.

With respect to biodiversity (Chapter IV), I have developed a method for assessing physical habitat maintenance and change. However, this method is problematic due to peoples’ framing of biodiversity issues, and the inevitable uncertainties in data used in the analysis. In fact, the challenge of attempting to develop a method has highlighted some generic features of LCA that mitigate against its acceptance as an environmental management approach among some stakeholders (see Chapter IV, Section 9, and Chapter VII).
3. Assessment of Agricultural Systems

Application of LCA to agricultural systems, as well as highlighting the need for assessing the aspects discussed in the last section, also shows the importance of systems analysis in developing realistic life cycle models. As the quote at the beginning of Chapter VI said, “In a sense, the sustainability of any subsystem of the global system – be it a state, a firm, a region, or even an individual – can only be defined in terms of a sustainable global system, and cannot be meaningfully said to exist in the absence of its links to the greater whole” (Allenby, 1998, p.18). For agricultural systems, the implications are that LCA should account for:

- Interactions of a crop under analysis with other crops in a rotation via the medium of the soil (Chapter III, Section 4.2).
- Field boundaries and field margins because they can be regarded as ancillaries (to use LCA terminology) in agricultural systems (Chapter III, Section 4.3).

Additionally, the case study has illustrated the influence on the LCA results of the method used to account for manure (Chapter VI, Sections 4.2.2 and 7.3), and shown practical implementation of the hierarchy of approaches to allocation developed by LCA practitioners. System extension, the preferred approach, gives a more realistic representation of the overall impacts associated with this source of nutrients. The case study has also shown that capital equipment makes a contribution greater than 10% to the results in a number of Impact Assessment categories, and is therefore a relevant subsystem for assessment in LCAs of agricultural production.

In Chapter III, Section 3, it was asserted that site-dependency of results should be a valid consideration in LCA, and particularly in LCAs of agricultural systems. This was demonstrated in the case study where it was shown how choice of location influences the LCA results through intrinsically determining the lime requirement, the quantity of energy used for transporting products from the farm to the point of consumption, and the toxicity associated with atmospheric deposition of heavy metals. This suggests that it may be equally, or even more, relevant to determine preferred locations of production rather than preferred farming practices in seeking to maximise the environmental performance of farming systems. Using LCA to make such assertions represents a novel application because LCA is conventionally used to inform choices between technologies largely independently of their sites.
4. LCA and Decision-Making

Turning to the role of LCA in decision-making, the research on agricultural systems has provided some insights into the strengths and weaknesses of using LCA in different decision-making contexts. The case study has demonstrated how LCA sets issues in context by using a comprehensive environmental life cycle approach. For example, it has shown how the perceived environmental preferability of organic over more conventional, intensive farming systems can be questioned for some types of environmental impacts. However, ultimately the choice is determined by value-based decisions about the relative weighting of toxic impacts against other impacts.

Attempting to assess biodiversity in LCA has highlighted the problem of quantifying impacts in cases where there is a high level of uncertainty, and where people have strong opinions about the importance of an impact. It has also demonstrated an implicit assumption in LCA: trade-offs can be made between different impacts in systems under analysis. This assumption may be questioned by some stakeholders in society, and goes some way towards explaining why LCA, and LCA studies, have not been accepted and used by, for example, some environmental NGOs.

In Chapter VII, I suggested that more attention is needed to making uncertainties in LCA more obvious and increasing the resonance of LCA results with the intended audience. For uncertainties, this could involve more attention to the site-dependency of impacts, and presenting results as a range of values. For increasing the resonance of LCA results, use of indicators provides a possible way of increasing the understandability and meaningfulness of results.

Increased use of LCA in decision-making is also dependent upon perceptions about its relevance, and the relevance of an LCA study cannot be assessed without reference to its decision-making context. I have suggested that LCA methodology should be more flexible for use in different decision-making contexts. However, this requires more attention to the procedural aspects of LCA, and in particular the process of omitting sub-systems or impacts, so that the credibility of results is maintained through the process of simplification. However, there is a tension between making LCA more flexible and responsive to the needs of users, and its role in raising awareness about the overall environmental impacts of human activities. In fact, the more comprehensive, “awareness-raising” LCA studies that attempt to assess the overall impacts of human activities, shape the goals and requirements for strategic and operational decisions. Used in this way, LCA widens the context for consideration of problems so that creative alternatives can be identified. Indeed, the keys to future sustainability may well be found in these awareness-raising studies.
5. Implications for Future Research

This research has contributed to development of a comprehensive methodology for LCA of agricultural systems. In order to operationalise the approach, further work is needed on development of the Physical Habitat Indices, and this requires the involvement of specialists in biodiversity assessment. However, otherwise, the only barriers to its use are data, time and financial resources, and the perceived usefulness of the approach among stakeholders.

Perhaps the greater remaining challenge concerns how LCA of agricultural systems can be integrated into different decision-making contexts. These contexts may vary from a consumer’s choice between two products, to a farmer’s choice between alternative farming operations, to a company’s decision about using an agricultural versus a mineral oil-derived product in its manufacturing operations, to the EU’s interest in reform of the Common Agricultural Policy.

Therefore, I suggest that the focus of LCA research on agricultural systems should now shift to:

- Research on application of LCA within these different contexts; and

- Studies asking specific questions about the environmental preferability of different farming systems (such as intensive versus organic systems), and the role of site-dependency in such assessments.

Such studies have the potential to make an important contribution to the development of sustainable human activities, and are particularly timely given current interest and concern over the environmental impacts of agricultural systems at local, national and international levels.

References

APPENDIX I.1
CONTRIBUTION OF FOOD CHAIN TO SOLID WASTE ARISINGS
IN UK

Tables 1 and 2 below give data for the types of solid waste sent to landfill and incineration in the UK in the early 1990s. These data are combined in Table 3 to calculate the proportion of waste derived from the food chain; this is between 24.3 and 24.6 million tonnes per year. According to these data, therefore, 23% of landfilled and incinerated waste is derived from the food chain.

Table 1. Sources of Controlled Waste Sent To Landfill and Incineration (Estimated, Early 1990s)

<table>
<thead>
<tr>
<th>Waste Type</th>
<th>Mass Landfilled (tonnes/year)</th>
<th>Mass Incinerated (tonnes/year)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Household*</td>
<td>13.3</td>
<td>2.2</td>
</tr>
<tr>
<td>Civic amenity waste*</td>
<td>3.6</td>
<td>0.0</td>
</tr>
<tr>
<td>Commercial*</td>
<td>14.3</td>
<td>0.8</td>
</tr>
<tr>
<td>Sewage sludge*</td>
<td>1.2</td>
<td>0.2</td>
</tr>
<tr>
<td>Construction and demolition</td>
<td>13.2</td>
<td>0.0</td>
</tr>
<tr>
<td>Asphalt planings</td>
<td>1.6</td>
<td>0.0</td>
</tr>
<tr>
<td>Industrial (blast furnace ash)</td>
<td>1.8</td>
<td>0.0</td>
</tr>
<tr>
<td>Industrial (power station ash)</td>
<td>6.5</td>
<td>0.0</td>
</tr>
<tr>
<td>Industrial (general)</td>
<td>9.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Industrial (miscellaneous processing)</td>
<td>15.3</td>
<td>0.0</td>
</tr>
<tr>
<td>Industrial (food processing)*</td>
<td>14.4</td>
<td>0.0</td>
</tr>
<tr>
<td>Clinical</td>
<td>0.1</td>
<td>0.2</td>
</tr>
<tr>
<td>Hazardous</td>
<td>3.5b</td>
<td>-</td>
</tr>
<tr>
<td>Fragmentiser Residues</td>
<td>0.5</td>
<td>0.0</td>
</tr>
<tr>
<td>Meat processing residues*</td>
<td>1.2</td>
<td>0.0</td>
</tr>
<tr>
<td>Poultry – wastes*</td>
<td>1.4</td>
<td>0.4</td>
</tr>
<tr>
<td>Mushroom compost*</td>
<td>0.3</td>
<td>0.0</td>
</tr>
<tr>
<td>Tyres</td>
<td>0.3</td>
<td>0.0</td>
</tr>
<tr>
<td>Wood waste</td>
<td>0.5</td>
<td>0.1</td>
</tr>
<tr>
<td>Totals</td>
<td>102.0</td>
<td>3.9</td>
</tr>
</tbody>
</table>

* = wastes categories that include food chain waste.

a Landfilled waste is after pre-disposal treatment to remove moisture.
b This is the upper end of the range of hazardous waste arisings of two to four million tonnes and using 90% as estimated disposal percentage to landfill.
c Less than 0.1 million tonnes.

<table>
<thead>
<tr>
<th>Waste Category</th>
<th>Sub-Category</th>
<th>Weight (%)</th>
<th>Food-Related (%)</th>
<th>Minimum</th>
<th>Maximum</th>
</tr>
</thead>
<tbody>
<tr>
<td>Paper and card</td>
<td>Newspapers</td>
<td>11.44</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Magazines</td>
<td>4.61</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Other paper</td>
<td>9.53</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Liquid containers</td>
<td>0.64</td>
<td>-</td>
<td>0.32</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Card packaging</td>
<td>3.79</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Other card</td>
<td>3.10</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Plastic film</td>
<td>Refuse sacks</td>
<td>1.16</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Other plastic film</td>
<td>4.18</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Dense plastic</td>
<td>Clear beverage bottles</td>
<td>0.63</td>
<td>0.63</td>
<td>0.63</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Coloured beverage bottles</td>
<td>0.12</td>
<td>0.12</td>
<td>0.12</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Other plastic bottles</td>
<td>1.12</td>
<td>-</td>
<td>0.56</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Food packaging</td>
<td>1.91</td>
<td>1.91</td>
<td>1.91</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Other dense plastic</td>
<td>2.14</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Textiles</td>
<td>Textiles</td>
<td>2.13</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Miscellaneous</td>
<td>Disposable nappies</td>
<td>4.21</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>combustibles</td>
<td>Miscellaneous (other)</td>
<td>3.90</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Miscellaneous</td>
<td>Miscellaneous non-combustibles</td>
<td>1.81</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>non-combustibles</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Glass</td>
<td>Brown glass</td>
<td>1.31</td>
<td>1.31</td>
<td>1.31</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Green glass</td>
<td>2.39</td>
<td>2.39</td>
<td>2.39</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Clear glass</td>
<td>5.37</td>
<td>5.37</td>
<td>5.37</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Other glass</td>
<td>0.20</td>
<td>0.20</td>
<td>0.20</td>
<td></td>
</tr>
<tr>
<td>Putrescibles</td>
<td>Garden waste</td>
<td>3.40</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Putrescibles (other)</td>
<td>16.77</td>
<td>16.77</td>
<td>16.77</td>
<td></td>
</tr>
<tr>
<td>Ferrous metal</td>
<td>Ferrous beverage cans</td>
<td>0.53</td>
<td>0.53</td>
<td>0.53</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Food cans</td>
<td>3.74</td>
<td>3.74</td>
<td>3.74</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Batteries</td>
<td>0.06</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Other cans</td>
<td>0.40</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Ferrous (other)</td>
<td>0.98</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Non-ferrous metal</td>
<td>Non-ferrous beverage cans</td>
<td>0.43</td>
<td>0.43</td>
<td>0.43</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Foil</td>
<td>0.47</td>
<td>0.47</td>
<td>0.47</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Other non-ferrous</td>
<td>0.71</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Fines</td>
<td>&lt;10 mm fines</td>
<td>6.77</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Totals</td>
<td></td>
<td>100.00</td>
<td>33.87</td>
<td>34.75</td>
<td></td>
</tr>
</tbody>
</table>

### Table 3. Estimated Quantities of Food Chain Waste In Solid Waste Arisings for the UK

<table>
<thead>
<tr>
<th>Waste Type</th>
<th>Mass Landfilled (tonnes/year)</th>
<th>Mass Incinerated (tonnes/year)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Household</td>
<td>4.5 - 4.7</td>
<td>0.7 - 0.8</td>
</tr>
<tr>
<td>Civic amenity waste</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Commercial</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Sewage sludge</td>
<td>1.2</td>
<td>0.2</td>
</tr>
<tr>
<td>Construction and demolition</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Asphalt planings</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Industrial (blast furnace ash)</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Industrial (power station ash)</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Industrial (general)</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Industrial (miscellaneous processing)</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Industrial (food processing)</td>
<td>14.4</td>
<td>0.0</td>
</tr>
<tr>
<td>Clinical</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Hazardous</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Fragmentiser Residues</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Meat processing residues</td>
<td>1.2</td>
<td>0.0</td>
</tr>
<tr>
<td>Poultry – wastes</td>
<td>1.4</td>
<td>0.4</td>
</tr>
<tr>
<td>Mushroom compost</td>
<td>0.3</td>
<td>0.0</td>
</tr>
<tr>
<td>Tyres</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Wood waste</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td><strong>Totals</strong></td>
<td><strong>23.0 - 23.2</strong></td>
<td><strong>1.3 - 1.4</strong></td>
</tr>
</tbody>
</table>

### References


APPENDIX II.1
ASSESSMENT OF TOXICITY IN LCA

A number of methods have been developed for calculation of Impact Assessment (IA) factors for toxicity in LCA. The earlier methods have been described as using the “critical volumes” approach, and this approach has also formed the basis for assessment in subsequent methods. In the three sections below, I describe the different approaches.


In both these methods, assessment is based on the initial receiving media (air and water) for emissions, and IA factors are calculated as the inverse of certain quality standards. Total toxicity is then assessed as the sum of all the weighted immissions into the two media. For air emissions, Habersatter (1991) uses Maximale Immissions-Konzentration (Maximum Immission Concentration (MIK)) values (or Maximale Arbeitsplatzkonzentration (MAK), i.e. occupational exposure limit, values if MIK values are not available). For water emissions, he uses Swiss directives for immissions into surface waters. Heijungs et al. (1991) use Dutch Maximum Accepted Concentration (MAC) values for air immissions, and EC directives for drinking water standards for water immissions.

In effect, these approaches give the volume of air (in m³) and water (in litres) required to dilute the emissions to acceptable levels. Hence they are described as “critical volumes” approaches.

2. Provisional Toxicity Assessment Using The Problem-Oriented Approach (Heijungs et al. (1992a, 1992b)

Assessment using this method gives three different toxicity values: aquatic ecotoxicity, terrestrial ecotoxicity, and human toxicity. Like the critical volumes approaches described above, this method also calculates toxicity based on the initial receiving media for different emissions. For aquatic toxicity, the only initial receiving medium considered is water. For terrestrial ecotoxicity, the only initial receiving medium considered is soil. For human toxicity, the media considered are air, water and soil.
IA factors for ecotoxicity are calculated using the critical volumes approach. However, the method is proposed as an improvement of the previous approach because the values are based on actual toxicological data rather than quality standards developed through the political process (although still shaped by toxicological data). The method is discussed in more detail below.

IA factors for human toxicity are calculated using two parameters: an effect factor and an exposure factor. The effect factor is similar to the IA factor calculated using the critical volumes approach; it therefore represents the inherent toxicity of a substance. The exposure factor is a first step towards accounting for the fate of a substance in the IA factor. Indeed, Heijungs et al. (1992) describe this method as “provisional” because the exposure factors are based on models and data that were available to them at the time, and which they recognised had a number of shortcomings. The updated method with altered IA factor values has recently been published and is described below (Section 3).

The method, as published in 1992, is developed in terms of the following IA factors:

\[
\begin{align*}
\text{Aquatic ecotoxicity (ECA)} &= E_{aq} \\
\text{Terrestrial ecotoxicity (ECT)} &= E_t \\
\text{Human toxicity for air emissions (HCA)} &= E_a \times X_a \\
\text{Human toxicity for water emissions (HCW)} &= E_w \times X_w \\
\text{Human toxicity for soil emissions (HCS)} &= E_s \times X_s
\end{align*}
\]

where \( E_{aq}, E_w, E_s, E_t \) and \( E_e \) = effect factors for emissions to water (aq) and soil (t) in ecosystems, and emissions to air (a), water (w) and soil (s) that subsequently reach humans.

\( X_a, X_w \) and \( X_s \) = human exposure factors for air, water and soil emissions.

**Effect factors for ecotoxicity**

The effect factors for ecotoxicity are calculated as the inverse of the “maximum tolerable concentration” (MTC) for each substance (after Slooff, 1992). The MTCs are based on toxicological data extrapolated to account for uncertainty, and are meant to provide a rough approximation of the concentration at which 95% of species in an ecosystem are protected (Heijungs et al., 1992b, p.98). The toxicological data are taken from experimental data on specific species giving parameters such as the No Observable Effect Concentration (NOEC), the lethal concentration for 50% of the organisms (LC\(_{50}\)), the effect concentration for 50% of organisms (EC\(_{50}\)), or the “Quantitative Structure-Activity Relationship” (QSAR) of one of these parameters. The extrapolation factors are shown in Tables 1
and 2 for aquatic and terrestrial ecotoxicity. If the toxicity data available gave different estimates of toxicity (for example, different LC50 values), then Heijungs et al. used the geometric mean of the data to calculate the effect factor (provided that all the data sources were equally reliable). If enough data were available to calculate two or three effect factors using the different toxicological parameters, then the lowest value was chosen as the effect factor.

Table 1. Extrapolation Factors for Aquatic Ecotoxicity

<table>
<thead>
<tr>
<th>Toxicological Parameters</th>
<th>Extrapolation Factor</th>
<th>Range of Uncertainty in Toxicological Data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lowest acute LC50, EC50 or QSAR estimate of acute toxicity</td>
<td>0.001</td>
<td>Most uncertain</td>
</tr>
<tr>
<td>Lowest acute LC50, EC50 or QSAR estimate of acute toxicity to at least one representative of three of the four groups: algae, crustaceans and fish</td>
<td>0.01</td>
<td></td>
</tr>
<tr>
<td>Lowest chronic NOEC or QSAR estimate of chronic toxicity to at least one representative of three of the four groups: algae, crustaceans and fish</td>
<td>0.1</td>
<td>Least uncertain</td>
</tr>
</tbody>
</table>

Source: Heijungs et al., 1992b, p.98.

Table 2. Extrapolation Factors for Terrestrial Ecotoxicity

<table>
<thead>
<tr>
<th>Toxicological Parameters</th>
<th>Extrapolation Factor</th>
<th>Range of Uncertainty in Toxicological Data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lowest acute LC50, EC50 or QSAR estimate of acute toxicity</td>
<td>0.001</td>
<td>Most uncertain</td>
</tr>
<tr>
<td>Lowest acute LC50, EC50 or QSAR estimate of acute toxicity to at least one representative of three of the four groups: microbial processes, earthworms, anthropoda and plants</td>
<td>0.01</td>
<td></td>
</tr>
<tr>
<td>Lowest chronic NOEC or QSAR estimate of chronic toxicity to at least one representative of three of the four groups: microbial processes, earthworms, anthropoda and plants</td>
<td>0.1</td>
<td>Least uncertain</td>
</tr>
</tbody>
</table>

* Sub-order of primates including monkeys, gibbons and great apes.


Thus each IA factor is calculated as:

\[
E_{aq\ or\ t} = \frac{1}{MTC_{aq\ or\ t}} = \frac{1}{\text{Toxicity factor} \times \text{Extrapolation factor}}
\]

where \( E_{aq\ or\ t} \) = effect factor for aquatic or terrestrial ecotoxicity

\( MTC_{aq\ or\ t} \) = maximum tolerable concentration in water (for aquatic ecosystems) or soil (for terrestrial ecosystems)

Toxicity value = Lowest NOEC, LC50, EC50 or QSAR value from data for as many species as possible
Extrapolation factor = value ranging from 0.001 to 0.1 depending upon uncertainty in toxicological data1.

In effect, use of these IA factors in an LCA gives the volume (m³) of water and mass (kg) of soil that is required to “dilute” the emissions to a “safe” concentration.

**Effect factors for human toxicity**

Heijungs *et al.* (1992a, 1992b) calculate different effect factors for emissions to air, water and soil. However, the method of calculation is the same for all the factors:

\[
E_{a, w, or s} = \frac{1}{TDI_{a, w, or s}}
\]

where \(E_{a, w, or s}\) = effect factor for emissions to air, water or soil

\(TDI_{a, w, or s}\) = tolerable daily intake via air water or soil.

The TDI values are calculated using data provided by the National Institute of Public Health and Environmental Protection (RIVM) in the Netherlands. These data are based on toxicity data multiplied by uncertainty factors (similar to calculation of effect factors in ecotoxicity). Where these data are not available, Heijungs *et al.* (1992a, 1992b) use alternative data on toxicity such as the air quality guidelines values set by the World Health Organisation to calculate the TDI. The effect factors calculated in this way can be interpreted as the amount of body mass (in kilogrammes) required to “dilute” one kilogramme of substance \(i\) to a safe concentration given a daily intake of one kilogramme of substance \(i\) via air, water or soil.

**Exposure factors for human toxicity**

In order to calculate exposure factors, Heijungs *et al.* (1992a, 1992b) firstly construct a model world (based on the concept of an area of one square kilometre representing a unit world developed by Mackay (1991)). This model world has the following characteristics:

- Total air volume = \(3 \times 10^{14}\) m³ (taking 10 km as the depth of the troposphere)

---

1 For example, common uncertainty factors are: i) a factor 0.1 for uncertainty due to extrapolation from laboratory animals to humans; ii) a factor 0.001 as a safety factor (Heijungs *et al.*, 1992b, p.90, citing Vermeire *et al.*, 1991).
• Total water volume = 3.5 x 10^{18} litres (assuming only top 10 m of water will be polluted in a relatively short time period)
• Total soil weight = 2.7 x 10^{16} kg (dry matter)
• Total world population = 5 x 10^9 persons, each weighing 70 kg
• Total volume air inhaled per person = 20 m³ air/day
• Total volume water consumed per person = 2 litres water/day.

This model world is based on “real world” data, and represents the volumes of air and water, and weight of soil that can become contaminated if an emission is dispersed throughout the world. For example, the Earth’s total surface area is taken as 5 x 10^8 km², and 30% is covered with soil (i.e. this is the total land area). Assuming that pollution will largely be contained within the top 15 cm of soil, and that the average soil density is 1,200 kg dry matter per m³, the model world contains:

\[ 5 \times 10^8 \times 10^4 \times 0.3 \times 0.15 \times 1,200 = 2.7 \times 10^{16} \text{ kg soil (dry matter)} \]

Once this model world has been defined, the exposure factors are calculated by firstly assuming that an emission to air, water or soil is distributed uniformly throughout the world’s air, water or soil. It then reaches humans by inhalation in air, consumption of drinking water or via direct dermal exposure and a number of indirect routes from the soil.

Therefore, the exposure factor for emissions to air is:

\[ X_a = \frac{V_a \times W}{V_a} = 3.33 \times 10^{-8} \]

where \( V_a \) = volume of air inhaled per person per day (20 m³)
\( W \) = world population (5 x 10^9 persons)
\( V_a \) = total volume of air in world (3 x 10^18 m³).

This can be interpreted as the fraction of total air in the world that the world population inhales during one day.

The exposure factor for emissions to water is:

\[ X_w = \frac{V_w \times W}{V_w} = 2.86 \times 10^{-9} \]
where \( \nu_w \) = volume of water consumed per person per day (2 litres)
\( W \) = world population (5 x 10^9 persons)
\( V_w \) = total volume of water in world (3.5 x 10^14 litres).

This can be interpreted as the fraction of total water in the world that the world population consumes during one day.

For emissions to soil, calculation of the exposure factor is more complicated than for air or water. This is because humans do not directly inhale or consume soil. Instead, once a uniform distribution of substance \( i \) throughout the world’s soil has been assumed, a further calculation must be undertaken to determine the proportion of substance \( i \) that is “consumed” by humans via food or other routes (for example, by migration across plastic water pipes and into drinking water). Heijungs et al. (1992a, 1992b) used existing data (Van den Berg, 1991) to calculate this proportion (described as the “extent of exposure \( p^* \)” (Heijungs et al., 1992b, p.95)). Its value varies between different substances because the extent of exposure by humans to each substance is partly related to the inherent properties of the substance (such as its propensity to migrate through plastic pipes into drinking water). However, Heijungs et al. (1992b) do not give further details about these differences.

The exposure factor for any substance \( i \) emitted to soil is therefore:

\[
X_{i,j} = \frac{\nu_{i,j} \times W}{V_s}
\]

where \( X_{i,j} \) = exposure factor for substance \( i \) in soil
\( \nu_{i,j} \) = mass of soil containing substance \( i \) “consumed” per person per day (reflecting extent of exposure to humans of substance \( i \) in the soil)
\( W \) = total world population (5 x 10^9 persons)
\( V_s \) = total mass of soil in world (2.7 x 10^16 kg dry matter).

This can be interpreted as the proportion of the total potentially contaminable soil in the world to which the world’s population is exposed, corrected to allow for the proportion of a substance in that soil which will actually reach humans.
Calculation of final IA factors for human toxicity

Having calculated effect and exposure factors for substances emitted to air, water and soil, final IA factors are calculated as:

- **Aquatic ecotoxicity (ECA)**: \( E_{\text{aq}} \)
- **Terrestrial ecotoxicity (ECT)**: \( E_t \)
- **Human toxicity for air emissions (HCA)**: \( E_a \times X_a = E_a \times 3.33 \times 10^8 \)
- **Human toxicity for water emissions (HCW)**: \( E_w \times X_w = E_w \times 2.86 \times 10^9 \)
- **Human toxicity for soil emissions (HCS)**: \( E_s \times X_s \)

where \( E_{\text{aq}}, E_t, E_a, E_w, E_s \) = effect factors for emissions to water (aq) and soil (t) in ecosystems, and emissions to air (a), water (w) and soil (s) that subsequently reach humans

\( X_a, X_w, X_s \) = human exposure factors for air, water and soil emissions.

Calculation of aquatic and terrestrial ecotoxicity, and human toxicity in an LCA

Once the effect and exposure factors have been calculated, final toxicity scores in an LCA are calculated as:

- **Aquatic ecotoxicity** = \( \sum_{i} E_{\text{CAi}} \times m_{\text{w},i} \)
- **Terrestrial ecotoxicity** = \( \sum_{i} E_{\text{CTi}} \times m_{\text{s},i} \)
- **Human toxicity** = \( \sum_{i} [(H_{\text{CAi}} \times m_{\text{a},i}) + (H_{\text{CWi}} \times m_{\text{w},i}) + (H_{\text{CSi}} \times m_{\text{s},i})] \)

where \( E_{\text{CAi}}, E_{\text{CTi}}, H_{\text{CAi}}, H_{\text{CWi}}, H_{\text{CSi}} \) = aquatic ecotoxicity IA factor for substance \( i \) emitted to water, terrestrial ecotoxicity IA factor for substance \( i \) emitted to soil, human toxicity IA factors for substance \( i \) emitted to air (a), water (w) and soil (s)

\( m_{\text{w},i}, m_{\text{a},i}, m_{\text{s},i} \) = mass of substance \( i \) released to water (w), soil (s) and air (a).
3. Updated Toxicity Assessment Using Problem-Oriented Approach (Guinée et al., 1996)

The method proposed by Heijungs et al. (1992) was described as "provisional," and in 1996 a more complete version of this method was published by Guinée et al. (1996). The method still gives three different toxicity values: aquatic ecotoxicity (AETP), terrestrial ecotoxicity (TETP) and human toxicity (HTP). However, it goes beyond the earlier method in more detailed modelling of the fate of substances as part of the toxicity assessment. Thus the IA factors for emissions of substances into different media take into account subsequent processes such as transfer to other media (air, water, soil, and so on), degradation rates, and bioconcentration through food chains.

The initial media for different emissions are defined as air, surface water and three types of soil: agricultural, industrial and generic soil\(^2\). For AETP, toxicity assessment is based on emissions into all five media and subsequent transport into surface water. For TETP, it is based on emissions into all five media and subsequent transport into agricultural soil. For human toxicity, it is based on emissions into all five media, and transport via various media to humans (as shown in Figure 1).

Figure 1. Assumed Transport Routes Through Various Media for Assessment of Human Toxicity

![Figure 1](source)  
Source: Guinée et al., 1996, p.28.

Calculation of the fate of different substances is undertaken using the USES (Uniform System for the Evaluation of Substances) 1.0 model. This computer model was developed by the National Institute of Public Health and Environmental Protection (RIVM) in the Netherlands for the quantitative hazard

\(^2\) The IA factor for a substance released into "generic soil" is calculated from the weighted IA factors for agricultural and industrial soils, assuming that generic soil is 73% agricultural and 27% industrial soil (Guinée et al., 1996, p.63).
and risk assessment of mainly organic substances. However, it has been adapted by RIVM and CLM using a "country-file editor" that converts the model to describe an "LCA world," so that relevant IA factors can be calculated for use in LCA. This LCA world is based on Western Europe, and it is represented diagrammatically in Figure 2. Its features include:

- Land area is $3.56 \times 10^6$ km$^2$ (the area of continental Western Europe)
- The area fraction of water is set to 3%, natural soil to 60%, agricultural soil to 27%, and industrial soil to 10%.

Figure 2. "LCA World" Modelled Using USES 1.0 Computer Model

Source: Guinée et al., 1996, p.25.

As in the provisional method for human toxicity assessment, IA factors are calculated in this updated method for both ecotoxicity and human toxicity using two parameters: an effect factor and an exposure factor.

**Effect factors for ecotoxicity**

Effect factors for ecotoxicity are calculated in a similar way to those in the provisional method. However, in the updated method a "Predicted No Effect Concentration" (PNEC) value is used instead of a MTC value. Again, these values are defined as those concentrations at which 95% of the species in an ecosystem are protected (Guinée et al., 1996, p.29), and are based on toxicological data.
However, the extrapolation factors proposed by the EU are used in calculating values rather than those put forward by Slooff (1992)\(^3\).

**Effect factors for human toxicity**

A “No Observed Adverse Effect” level (NOAEL) is calculated for humans based on toxicological data, and extrapolated where necessary to give a “No Effect Level” (NEL). The effect factor for human toxicity via different routes of exposure (see Figure 1) is then the inverse of the NOAEL or NEL for exposure specific to each of those routes. This is analogous to use of the “Tolerable Daily Intake” value in the provisional method but the routes of exposure are modelled in greater detail.

**Exposure factors for ecotoxicity**

For aquatic ecosystems, exposure in aquatic ecosystems is via surface water (Guinée et al., 1996, p.27). Exposure values are calculated by immiitting 1000 kg/day of a substance into one of the initial media in the USES 1.0 model (i.e. air, surface water, agricultural soil or industrial soil), and then letting the model calculate the final steady state concentration of the substance in surface water. This is called the “Predicted Environmental Concentration” (PEC).

For terrestrial ecosystems, exposure is via agricultural soil (Guinée et al., 1996, p.27). Exposure values are calculated in the same way as for aquatic ecotoxicity.

**Exposure factors for human toxicity**

Exposure of humans to a substance emitted to any of the initial media (air, surface water, agricultural soil, industrial soil or generic soil) occurs via the routes shown in Figure 1, according to the USES 1.0 model. The model calculates flows of the substance through these different routes to humans, and then calculates a value similar to the “Predicted Daily Intake” (PDI) based on these data\(^4\).

\(^3\) In fact, these two sets of factors are very similar.

\(^4\) Guinée et al. (1996) do not give further details about how this PDI value is linked with the different routes of exposure.
Calculation of final IA factors

Having calculated effect and exposure factors for aquatic and terrestrial ecotoxicity, and human toxicity, for different substances released to different media, the preliminary toxicity factors for any substance \( i \) released to any of the five initial media are:

- **Aquatic ecotoxicity**
  \[
  \text{PEC}_{w,i} \quad \text{PNEC}_{w,i} \\
  \]

- **Terrestrial ecotoxicity**
  \[
  \text{PEC}_{i} \quad \text{PNEC}_{i} \\
  \]

- **Human toxicity**
  \[
  "\text{PDI}_{i}" \quad "\text{ADI}_{i}" \quad \frac{1}{\text{MOS}_{i}} \\
  \]

where \( \text{PEC}_{w,i} \) = Predicted Environmental Concentration in surface water for substance \( i \)

\( \text{PNEC}_{w,i} \) = Predicted No Effect Concentration in surface water for substance \( i \)

\( \text{PEC}_{i} \) = Predicted Environmental Concentration in agricultural soil for substance \( i \)

\( \text{PNEC}_{i} \) = Predicted No Effect Concentration in agricultural soil for substance \( i \)

"\text{ADI}_{i}\" = value analogous to Acceptable Daily Intake for substance \( i \)

"\text{PDI}_{i}\" = value analogous to Predicted Daily Intake for substance \( i \)

\( \text{MOS}_{i} \) = Margin of Safety for substance \( i \), analogous to the reciprocal of the \( \text{PEC}/\text{PNEC} \) value in ecotoxicity.

These results are then normalised to the \( \text{PEC}/\text{PNEC} \) or \( 1/\text{MOS} \) values for a reference substance within each toxicity category (aquatic and terrestrial ecotoxicity, and human toxicity). The reference substance chosen by Guinée et al. (1996) is 1,4-dichlorobenzene. For aquatic toxicity, the relevant \( \text{PEC}/\text{PNEC} \) value for 1,4-dichlorobenzene is that calculated from an initial release of this substance to surface water. For terrestrial ecotoxicity, the relevant \( \text{PEC}/\text{PNEC} \) value is that calculated from an initial release of this substance to industrial soil. For human toxicity, the relevant \( 1/\text{MOS} \) value is that calculated from an initial release of this substance to air.

As a result, the final IA factors for any substance \( i \) emitted to any one medium \( c \) (which may be air, water, agricultural soil, industrial soil or generic soil) is:
\[
\begin{align*}
\text{AETP}_c,i & = \frac{\left( \frac{\text{PEC}_{w,i}}{\text{PNEC}_{w,i}} \right)}{\left( \frac{\text{PEC}_{w,r}}{\text{PNEC}_{w,r}} \right)} \\
\text{TETP}_c,i & = \frac{\left( \frac{\text{PEC}_{t,i}}{\text{PNEC}_{t,i}} \right)}{\left( \frac{\text{PEC}_{t,r}}{\text{PNEC}_{t,r}} \right)} \\
\text{HTP}_c,i & = \frac{\text{MOS}_{a,r}}{\text{MOS}_{a,i}}
\end{align*}
\]

where 
- \(c\) = initial medium: air, water, agricultural soil, industrial soil or generic soil (a, w or t) 
- \(r\) = reference substance (1,4-dichlorobenzene) 
- \(i\) = substance \(i\) 
- \(w\) = surface water 
- \(t\) = agricultural, industrial and/or generic soil 
- \(a\) = air 

\(\text{AETP}_c,i\) = aquatic ecotoxicity factor for substance \(i\) emitted to initial medium \(c\) 

\(\text{TETP}_c,i\) = terrestrial ecotoxicity factor for substance \(i\) emitted initial medium \(c\) 

\(\text{HTP}_c,i\) = human toxicity factor for substance \(i\) emitted initial medium \(c\) 

\(\text{PEC}_{c,i}\) = Predicted Environmental Concentration in medium \(c\) for substance \(i\) 

\(\text{PNEC}_{c,i}\) = Predicted No Effect Concentration in medium \(c\) for substance \(i\) 

\(\text{PEC}_{c,r}\) = Predicted Environmental Concentration in medium \(c\) for 1,4-dichlorobenzene 

\(\text{PNEC}_{c,r}\) = Predicted No Effect Concentration in medium \(c\) for 1,4-dichlorobenzene 

\(\text{MOS}_{c,i}\) = Margin of Safety in medium \(c\) for substance \(i\), analogous to the reciprocal of the PEC/PNEC value in ecotoxicity. 

\(\text{MOS}_{a,r}\) = Margin of Safety in air for 1,4-dichlorobenzene, analogous to the reciprocal of the PEC/PNEC value in ecotoxicity. 

Calculation of aquatic and terrestrial ecotoxicity, and human toxicity in an LCA 

Once the effect and exposure factors have been calculated, final toxicity scores in an LCA are calculated as:
Aquatic ecotoxicity  $$= \sum_{c=1}^{5} \left[ \sum_{i}^{n} \text{AETP}_{c,i} \times m_{c,i} \right]$$
Terrestrial ecotoxicity  $$= \sum_{c=1}^{5} \left[ \sum_{i}^{n} \text{TETP}_{c,i} \times m_{c,i} \right]$$
Human toxicity  $$= \sum_{c=1}^{5} \left[ \sum_{i}^{n} \text{HTP}_{c,i} \times m_{c,i} \right]$$

where  
- $i$ = substance $i$
- $c$ = initial medium: either air ($c=1$), water ($c=2$), agricultural soil ($c=3$), industrial soil ($c=4$), or generic soil ($c=5$)
- $\text{AETP}_{c,i}$ = aquatic ecotoxicity IA factor for substance $i$ emitted to initial medium $c$
- $\text{TETP}_{c,i}$ = terrestrial ecotoxicity IA factor for substance $i$ emitted to initial medium $c$
- $\text{HTP}_{c,i}$ = human toxicity IA factor for substance $i$ emitted to initial medium $c$
- $m_{c,i}$ = mass of substance $i$ emitted initial medium $c$.


The Critical Surface-Time (CST) method has also been proposed as an improvement on the method of Heijungs et al. (1992a, 1992b) because it includes fate as well as effect factors in calculating all IA factors. However, while the method of Guinée et al. (1996) uses a computer model to calculate fate (or “exposure”) factors, the CST method uses empirical data to calculate these factors. In other words, it is based on actual data collected on the fate of substances while the Guinée et al. (1996) method uses properties of substances (such as vapour pressure, solubility, hydraulic residence time, and air-water partition coefficients) to predict final concentrations in different media.

Assessment is undertaken for the same toxicity categories as for Heijungs et al. (1992a, 1992b) and Guinée et al. (1996): aquatic ecotoxicity, terrestrial ecotoxicity and human toxicity. The IA factors available for human toxicity assess the comprehensive impacts of i) pesticides used in the field (through their subsequent transport to air, water, and via soil into food); ii) heavy metals emitted to air and water, and transported from the soil into food; and iii) other substances emitted directly into air and water. For aquatic ecotoxicity, the IA factors assess the impact of pesticides used in the field through transport into water, and for other substances the IA factors assess direct emissions into water. For terrestrial ecotoxicity, the IA factors assess the impact of pesticides used in the field through transport into soil, and for other substances the IA factors assess direct emissions into the...
soil. In other words, a comprehensive model of inter-media transport of substances is only used for pesticides and in assessing the human toxicity of heavy metals transported into food via the soil. For other substances, only direct emissions into each medium (air, water and soil) are considered in calculating the IA factors. This is different from the method of Guinée et al. (1996) where the comprehensive inter-media transport of all substances from five initial media (air, surface water, agricultural soil, industrial soil and generic soil) is modelled in calculating IA factors.

A summary of IA factors currently available using the CST method is shown in Table 3. The model used for transport to different media of pesticides after initial application in the field is shown in Figure 3, and the fractions reaching water and food are given in Table 4. The fractions of pesticides reaching water are taken from the literature (Weber et al., Jury et al., 1987). For pesticide residues in food, very limited data are available. For chlorothalonil, Eilrich (1991) observed that 5% of the "tolerable value" remains on fresh vegetables after peeling, washing and processing. In this case, the tolerable value is defined as the maximal concentration of pesticide observed on the grain for a correct pesticide application. Using this 5% value for all the pesticides (in the absence of other data), a first estimate of the proportion of the applied quantity that occurs as food residues can be calculated as:

\[
F = \frac{0.05 \times V \times Y}{D}
\]

where:
- \( F \) = fraction of active ingredient occurring as food residue
- \( V \) = tolerable value (kg active ingredient/kg grain)
- \( Y \) = average yield of grain per hectare (kg grain/ha)
- \( D \) = recommended dose of active ingredient per hectare (kg active ingredient/ha)

Swiss data on tolerable values, average yield of grain (6,000 tonnes/hectare) and recommended doses of active ingredients were used to calculate the proportions of each pesticide occurring as residues in food (using data from EDMZ, 1996).

For heavy metals emitted to soil, the assumptions made about transport from the soil to food are shown in Table 5.

Calculation of the effect and fate factors using the CST method is described below.
Table 3. Types of Impact Assessment Factors Available Using the CST Method

<table>
<thead>
<tr>
<th>Toxicity Category</th>
<th>Initial Emission Medium</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Air</td>
<td>Water</td>
<td>Soil</td>
</tr>
<tr>
<td>Human toxicity</td>
<td>Pesticides¹</td>
<td>Heavy metals</td>
<td>Heavy metals³</td>
</tr>
<tr>
<td></td>
<td>Heavy metals</td>
<td>Other substances⁵</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Other substances²</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Aquatic ecotoxicity</td>
<td>Pesticides⁴</td>
<td>Heavy metals</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Heavy metals</td>
<td>Other substances⁶</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Other substances⁷</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Terrestrial ecotoxicity</td>
<td>Pesticides⁴</td>
<td>-</td>
<td>Heavy metals</td>
</tr>
</tbody>
</table>

¹ Includes transport of pesticides to drinking water (via surface and ground-water) and food (via soil) for human toxicity.
² Substances are CO, NOₓ, particles and SO₂ (Audsley et al., 1997, p.78).
³ Only assesses impact of pesticides subsequently transported to water.
⁴ Only assesses impact of pesticides subsequently transported to soil.
⁵ Substances are fluoride, nitrate, phenol, phosphate and sulphide (Audsley et al., 1997, p.78).
⁶ Only oil and phenol.
⁷ Only assesses impact of heavy metals transported to food.

Figure 3. Modelling the Fate of Pesticides Applied In the Field (CST Method)

- Pesticide applied
  - 80% Field
  - 20% Air
  - 10% Crop
    - 2% Water
    - 88% Soil
  - 90% Air after 10 mins.
  - 10% Air after 10 mins.

a Half of this assumed to be in the next field.
b Residence time of pesticides in air assumed to be 1.6 days (based on value for lindane in Dinkel et al., 1996).
c 2% of pesticides in Netherlands are applied directly to water.
d 0-1.9% of applied quantity depending on pesticide (derived from Weber et al., 1980, and Jury et al., 1987; see Table 3).
e 0.0003-0.13% of applied quantity depending on pesticide (see Table 3).
f Assumed to be 85% of quantity applied.

Source: adapted from Audsley et al., 1997, p.53.
Table 4. Transport of Applied Pesticides To Surface Water, Groundwater and Food

<table>
<thead>
<tr>
<th>Type of Pesticide</th>
<th>Active Ingredient</th>
<th>Percentage of Applied Quantity to Surface Water</th>
<th>Percentage of Applied Quantity to Groundwater</th>
<th>Percentage of Applied Quantity to Food</th>
</tr>
</thead>
<tbody>
<tr>
<td>Growth regulator</td>
<td>Chlormequat</td>
<td>1.9</td>
<td>0</td>
<td>0.13</td>
</tr>
<tr>
<td></td>
<td>Mepiquat chloride</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>2-chloroethylphosphonic acid</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Fungicide</td>
<td>Carbendazin</td>
<td>-</td>
<td>0.036</td>
<td>0.036</td>
</tr>
<tr>
<td></td>
<td>Chlorothalonil</td>
<td>0.5</td>
<td>0</td>
<td>0.006</td>
</tr>
<tr>
<td></td>
<td>Fenpiclonil</td>
<td>0.6</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Fenprodinid</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Flusilazole</td>
<td>1.1</td>
<td>0</td>
<td>0.015</td>
</tr>
<tr>
<td></td>
<td>Hexaconazole</td>
<td>0.5</td>
<td>0.015</td>
<td>0.016</td>
</tr>
<tr>
<td></td>
<td>Tebuconazole</td>
<td>1.1</td>
<td>0.006</td>
<td>0.006</td>
</tr>
<tr>
<td>Herbicide</td>
<td>Diflufenican</td>
<td>0.5</td>
<td>0</td>
<td>0.011</td>
</tr>
<tr>
<td></td>
<td>Fluroxypyr</td>
<td>0.8</td>
<td>0</td>
<td>0.019</td>
</tr>
<tr>
<td></td>
<td>Ioxynil</td>
<td>0.5</td>
<td>0.0086</td>
<td>0.0086</td>
</tr>
<tr>
<td></td>
<td>Isoproturon</td>
<td>0.5</td>
<td>0.0012</td>
<td>0.0012</td>
</tr>
<tr>
<td></td>
<td>Mecoprop-P</td>
<td>1.0</td>
<td>0.3</td>
<td>0.0003</td>
</tr>
<tr>
<td>Insecticide</td>
<td>Cypermethrin</td>
<td>0.5</td>
<td>0</td>
<td>0.004</td>
</tr>
<tr>
<td>Aphicide</td>
<td>Pirimicarb</td>
<td>0.6</td>
<td>1.09</td>
<td>0.004</td>
</tr>
<tr>
<td>Slug pellet</td>
<td>Methiocarb</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>


Table 5. The Fate of Heavy Metals Applied To Agricultural Soil (CST Method)

<table>
<thead>
<tr>
<th>Heavy Metal</th>
<th>Mass in Grain (mg per kg dry weight) (Meyer, 1991)</th>
<th>Mass in Soil (mg per kg dry weight) (Meyer, 1991)</th>
<th>Fraction Exported in Grain¹</th>
<th>Residence Time (years) (Audsley et al., 1997, p.52)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cadmium</td>
<td>0.12</td>
<td>0.75</td>
<td>54%</td>
<td>1260</td>
</tr>
<tr>
<td>Copper</td>
<td>5.9</td>
<td>27.5</td>
<td>61%</td>
<td>1080</td>
</tr>
<tr>
<td>Lead</td>
<td>&lt;1.3</td>
<td>29.5</td>
<td>24%</td>
<td>1870</td>
</tr>
<tr>
<td>Zinc</td>
<td>31</td>
<td>67.5</td>
<td>77%</td>
<td>760</td>
</tr>
<tr>
<td>Average</td>
<td>-</td>
<td>-</td>
<td>54%</td>
<td>1260²</td>
</tr>
</tbody>
</table>

¹ Calculated by assuming annual yield of 6,000 kg dry matter per hectare, and annual soil erosion of 820 kg per hectare.
² The average is 1,243 years, but value of 1,260 years used by Audsley et al. (1997, p.52) in calculating Impact Assessment factors.

Source: Audsley et al., 1997, p.52.

Effect factors for ecotoxicity and human toxicity

Effect factors are calculated in a similar way to that used by Heijungs et al. (1992a, 1992b), based on toxicological data and an extrapolation factor to account for uncertainty.
Fate factors for ecotoxicity

For aquatic ecotoxicity, the fate factor for any substance \( i \) emitted into water or reaching water from another medium is based on two empirical measurements: residence time and dilution volume. [The term “residence time” here has a different meaning from its use in chemical engineering, and could alternatively be described as the “life-time” of a substance (see Solberg-Johansen, 1998, p.51)]. It is calculated as:

\[
F_i = \frac{R_i}{V_i}
\]

where \( F_i \) = fate factor for substance \( i \) released into water
\( R_i \) = residence time of substance \( i \) in water
\( V_i \) = dilution volume for substance \( i \).

The residence time for pesticides in water is taken from data in Linders et al. (1994) and Gaillard (1995), corrected to allow for replenishment of water sources by rainwater (Audsley et al., 1997, p.81). The residence time for heavy metals in water is taken as 0.23 years (the time taken for rainwater to replace the dilution volume (see below), given an annual rainfall of 0.76 m³/m²) (Audsley et al., 1997, p.81). No details are given about calculation of the residence time for other substances.

The dilution volume is the volume of a medium that is found in an area of 1 m² of the Earth’s surface. It is taken as 0.18 m³/m² for water (which is the volume of surface freshwater per m² of world area⁵) (Audsley et al., 1997, p.81).

For terrestrial ecotoxicity, a similar calculation is undertaken. Residence times for pesticides in soil are taken from Gaillard (1995); for heavy metals, residence times are calculated from data in Meyer (1991) assuming export in harvested crops and eroded soil (see Table 3). The dilution volume is taken as 0.2 m³/m² for soil (Audsley et al., 1997, p.82).

Fate factors for human toxicity

For human toxicity, the fate factors are also based on the residence time and dilution volume for different substances. For emissions into air, residence times are taken from the literature (and reproduced in Jolliet and Crettaz, 1997). For dilution volume, rather than using a standard value as

---

Footnotes:
⁵ The world area is taken as 5.1 x 10¹⁴ m² (Jolliet and Crettaz, 1997).
for aquatic and terrestrial ecotoxicity, Jolliet and Crettaz (1997) have calculated dilution volumes specific to different substances based on empirical data about their heights of dilution. The “height of dilution” of a substance \( i \) is the greatest height above the Earth’s surface at which the substance is found (measured on a global scale) after an initial emission from the Earth’s surface. Based on data for 17 substances, they derived the following equation for calculating the height of dilution for any substance \( i \) with a residence time less than 60 days:

\[
\text{Height of dilution} = 30,100 \times R^{0.61}
\]

where \( R = \) residence time in years of substance \( i \) (provided \( R \) is less than 0.164 years (60 days)).

For substances with residence times greater than 60 days, the height of dilution is taken as 10,000 metres (Jolliet and Crettaz, 1997).

These heights of dilution also represent the dilution volumes of different substances, measured as m\(^3\)/m\(^2\) world area (i.e. a height of dilution of 10,000 metres is equivalent to a dilution volume of 10,000 m\(^3\) world area). The fate factor can then be calculated in the same way as for aquatic and terrestrial ecotoxicity.

For emissions into water, the fate factor is calculated in the same way as for aquatic ecotoxicity except for two alterations. Firstly, the dilution volume accounts for groundwater as well as surface water (and is given as 20 m\(^3\)/m\(^2\) (Audsley et al., 1997, p.76)). Secondly, the fate factor is adjusted to account for the quantity of water consumed per day per kg body mass. This gives a final fate factor for emissions into water of:

\[
F_i = \frac{R_i}{V} \times \frac{Q}{B} = R_i \times 1.4 \times 10^{-6}
\]

where

- \( F_i = \) fate factor for substance \( i \) emitted to water
- \( R_i = \) residence time of substance \( i \) in water
- \( V = \) dilution volume (20 m\(^3\)/m\(^2\))
- \( Q = \) volume of water consumed per person per day (0.002 m\(^3\)/person·day)
- \( B = \) body mass of one person (70 kg/person).
For emissions into soil, no details are given about calculation of the fate factor apart from that it is a function of the body mass per m² in the world and “the number of days per year ... according to the second equivalency principle” (Audsley et al., 1997, p.75).

**Calculation of final IA factors**

Having calculated effect and fate factors for aquatic and terrestrial ecotoxicity for different substances, the preliminary toxicity factors for any substance \( i \) are:

\[
\begin{align*}
\text{Aquatic ecotoxicity} & = E_{w,i} \times F_{w,i} \\
\text{Terrestrial ecotoxicity} & = E_{t,i} \times F_{t,i}
\end{align*}
\]

where \( E_{w,i}, E_{t,i} = \text{effect factor for substance} \ i \ \text{emitted into water (w) or soil (t)} \)

\( F_{w,i}, F_{t,i} = \text{fate factor for substance} \ i \ \text{emitted into water (w) or soil (t)} \).

In addition, for pesticides, the need to model their transport to water and soil is removed by incorporating transfer coefficients (as shown in Figure 3) into the preliminary toxicity factors. Thus aquatic and terrestrial ecotoxicity factors for pesticides are only listed under the initial emission medium of “air” because it is assumed that all pesticides are sprayed in the field onto crops through the air.

For human toxicity, the preliminary toxicity factor for any substance \( i \) is:

\[
\text{Human toxicity} = E_{c,i} \times F_{c,i}
\]

where \( c \) = initial medium (either air, water or soil)

\( E_{c,i} = \text{effect factor for emissions to medium} \ c \)

\( F_{c,i} = \text{fate factor for emissions to medium} \ c \).

In addition, for pesticides, as for ecotoxicity, the need to model the transport of pesticides to drinking water and soil is removed by incorporating transfer coefficients into the preliminary toxicity factors. This has also been done for the transfer of heavy metals from soil to food. Thus human toxicity

---

6 It is worth noting that this is different from the “unit world” approach in Heijungs et al. (1992a, 1992b) where it is assumed that all substances are dispersed throughout a standard volume of air.
factors for pesticides are only listed under the initial emission medium of air, and for heavy metals under soil.

The preliminary IA factors obtained in this way are then normalised to the IA factors for a reference substance within each toxicity category (aquatic and terrestrial ecotoxicity, and human toxicity). The reference substance chosen for aquatic ecotoxicity is zinc emitted to water; for terrestrial ecotoxicity zinc emitted to soil; and for human toxicity, lead emitted to air. As a result, the final IA factors for any substance $i$ are:

$$
ETP^w (\text{aquatic ecotoxicity}) = \frac{E_{w,i} \times F_{ew,i}}{E_{w,Zn} \times F_{ww,Zn}}
$$

$$
ETP^s (\text{terrestrial ecotoxicity}) = \frac{E_{a,i} \times F_{ea,i}}{E_{a,Zn} \times F_{as,Zn}}
$$

$$
HTP (\text{human toxicity}) = \frac{E_{c,i} \times F_{ec,i}}{E_{a,Pb} \times F_{ea,Pb}}
$$

where $c =$ medium of air, water or soil

$w =$ water

$s =$ soil

$a =$ air

$E_{ci} =$ effect factor for substance $i$ in medium $c$

$E_{w,Zn}, E_{s,Zn} =$ effect factor for zinc (Zn) in water or soil

$E_{a,Pb} =$ effect factor for lead (Pb) in air

$F_{ci} =$ fate factor for substance $i$ released into any medium $c$ and staying in or transferred to another medium $c$ (but note that transfer between media is only considered for pesticides, and for heavy metals transferred from soil to food, in current IA factors)

$F_{ew,Zn}, F_{ea,Pb} =$ fate factor for zinc (Zn) or lead (Pb) released into medium $c$ and staying in or transferred to another medium $c$.

**Calculation of aquatic and terrestrial ecotoxicity, and human toxicity in an LCA**

Once IA factors have been calculated, final toxicity scores in an LCA are calculated in the same way as for Guinée et al. (1996):
Aquatic ecotoxicity = \sum_{c=1}^{3} \left[ \sum_{i}^{n} ETP_{e,c}^{w} \times m_{c,i} \right] \\
Terrestrial ecotoxicity = \sum_{c=1}^{3} \left[ \sum_{i}^{n} ETP_{e,c}^{t} \times m_{c,i} \right] \\
Human toxicity = \sum_{c=1}^{3} \left[ \sum_{i}^{n} HTP_{e,c} \times m_{c,i} \right] 

where i = substance i \\
c = medium: either air (c=1), water (c=2) or soil (c=3) \\
ETP_{e,c}^{w} = aquatic ecotoxicity IA factors for substance i emitted to medium c \\
ETP_{e,c}^{t} = terrestrial ecotoxicity IA factors for substance i emitted to medium c \\
M_{c,i} = mass of substance i emitted to medium c.

5. Summary of Impact Assessment Methods for Toxicity In LCA

This review of existing toxicity assessment methods shows that all methods depend upon calculation of IA factors using at least one of two parameters: an effect factor and a fate factor. The earlier methods used just the effect factor (Habersatter, 1991; Heijungs et al., 1991), but subsequent methods have incorporated the fate factor using more or less detailed models and empirical data. The method of Guinée et al. (1996) uses the most detailed modelling approach. However, it depends upon a number of assumptions that are not transparent to the user because they are embedded in the USES 1.0 computerised model. In this respect, the approach used by Jolliet and Crettaz (1996; Audsley et al., 1997) has provided useful data for cross-checking with the Guinée et al. (1996) results, and indeed this was done during development of the USES 1.0 model (Jolliet and Crettaz, 1996).

Table 6 shows the IA toxicity factors that are provided using the Heijungs et al. (1992a, 1992b), Guinée et al. (1996) and CST method. It can be seen that the greatest number of factors are available for the Heijungs et al. (1992a, 1992b) method, followed by the Guinée et al. (1996) method.

Final choice of IA factors to use in a study will, therefore, be a compromise between availability of relevant factors and likely accuracy of the factors. In future, the best IA factors are likely to be derived from the USES 1.0 model, verified by comparison with the results obtained using empirical data such as those calculated using the CST method. However, in the meantime it seems prudent to use more than one approach for assessing toxicity in a study.
<table>
<thead>
<tr>
<th>Type of Substances</th>
<th>Heijungs et al. (1992a, 1992b)</th>
<th>Guinée et al. (1996)</th>
<th>CST Method</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Ecotoxicity</td>
<td>Human toxicity</td>
<td>Ecotoxicity</td>
</tr>
<tr>
<td>Heavy metals</td>
<td>As, Cd, Cr, Co, Cu, Pb, Hg,</td>
<td>As, Ba, Cd, Cr³⁺,</td>
<td>As, Cd, Cr³⁺,</td>
</tr>
<tr>
<td></td>
<td>Ni, Zn</td>
<td>Cr⁴⁺, Co, Cu, Pb, Hg, Ni,</td>
<td>Cr⁴⁺, Co, Cu, Pb, Hg, Ni,</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Va, Zn</td>
<td>Va, Zn</td>
</tr>
<tr>
<td>Pesticides</td>
<td>About 120</td>
<td>About 195</td>
<td>About 15</td>
</tr>
<tr>
<td></td>
<td></td>
<td>CS₂</td>
<td>NH₄⁺, CS₂</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>NO₂⁻, PCB</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>SO₄²⁻</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>SCN</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hydrocarbons</td>
<td>About 105</td>
<td>About 65</td>
<td>About 65</td>
</tr>
</tbody>
</table>

References


# APPENDIX IV.1

## GLOSSARY OF TERMS AND ACRONYMS

### Terms

**Biome**
Major ecological complex or set of ecosystems possessing characteristic vegetation. Biomes occupy large areas of land surface, and typically occur on more than one continent.

**Chromosome**
Linear sequence of genes found in cells, consisting of DNA and a number of proteins.

**Class**
A taxonomic category in classification of living organisms. For example, humans are in the class *Mammalia*.

**Community**
The species that occur together in space and time.

**DNA**
Deoxyribonucleic acid. The nucleic acid forming the genetic material of all cells, consisting of a chain of nucleotides.

**Ecosystem**
The plants, animals, and physical and chemical components of their immediate environment or habitat which together form a recognisable self-contained entity.

**Endemic**
Having a habitat in a specified district or area.

**Gene**
Smallest physical unit of heredity, passing on genetic information from one individual to its offspring. It consists of a sequence of nucleotides.

**Habitat**
Place where a micro-organism, plant or animal lives.

**Kingdom**
The highest taxonomic category in the classification of living organisms. For example, humans are in the kingdom *Animalia*.

**Nucleotide**
Specific types of molecules found in DNA.

**Phylum**
A taxonomic category in classification of living organisms. For example, humans are in the phylum *Chordata*.

### Acronyms

**GPP**
Gross Primary Productivity (kg/m²/year or J/m²/year)

**NPP**
Net Primary Productivity (kg/m²/year or J/m²/year)

**PHD**
Physical Habitat Depletion (Impact Assessment category analogous to categories such as Global Warming and Ozone Depletion) (-)
<table>
<thead>
<tr>
<th>Code</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>PHF</td>
<td>Physical Habitat Factor (Impact Assessment factor analogous to factors such as Global Warming Potentials and Ozone Depletion Potentials) (-)</td>
</tr>
<tr>
<td>PHI</td>
<td>Physical Habitat Index (-)</td>
</tr>
<tr>
<td>PMF</td>
<td>Physical Management Factor (-)</td>
</tr>
<tr>
<td>PMI</td>
<td>Physical Management Index (-)</td>
</tr>
</tbody>
</table>
APPENDIX IV.2

CALCULATION OF GLOBAL PHYSICAL HABITAT FACTORS

Indicator 1 (Area)

The area of different ecosystems is taken from Begon et al. (1996, p.715), based on Whittaker (1975).

Indicator 2 (Number of Rare Species)

In order to obtain a "first guess" estimate for rare species, the value of Indicator 2 has been calculated for each ecosystem based on three factors:

1. Changes in land areas of different ecosystems between pre-agricultural times and the present (Graetz, 1994). The reasoning behind this assumption is that ecosystems which have decreased markedly in size over this time period are likely to have a greater number of rare species.
2. The proportion of locally endemic species in the ecosystem type: ecosystems with greater numbers of locally endemic species are likely to have greater numbers of rare species associated with a decrease in land area.
3. Total number of species in ecosystem: the absolute number of rare species is likely to be higher if the ecosystem contains an initially greater number of species.

The data and estimated results are shown in Table AIV.1.

Indicator 3 (Number of Species)

One of the most widely recognised patterns in species richness is the increase that occurs from the poles to the tropics. This is seen in a wide variety of species groups, and also occurs in terrestrial, marine and freshwater habitats (Begon et al., 1996, p.900). For example, a one hectare area of equatorial rainforest may contain 40-100 different tree species, while the same area of deciduous forest in eastern North America has 10-30 species, and coniferous forest in northern Canada has 1-5 species. Exceptions to this general pattern include:

- Particular groups of species that are more diverse in polar regions (e.g. penguins and seals).
- Coniferous trees and ichneumonid parasitoids that are most diverse in temperate latitudes.
• Deserts which are species-poor even when close to the Equator, probably because the climate is very extreme.
• Saltmarshes and hot springs which are relatively species-poor (although productive in terms of NPP), probably because they represent harsh environments.

However, the general pattern is one of an increase in numbers of species from the poles to the tropics. Therefore I use latitude as a “first guess” estimate of the numbers of species in different ecosystems. The data are shown in Table AIV.2.

**Indicator 4 (Net Primary Productivity)**

The Net Primary Productivity of different ecosystems is taken from Begon *et al.* (1996, p.715), based on Whittaker (1975).

**Table 1. Calculation of Indicator 2**

<table>
<thead>
<tr>
<th>Ecosystem Type</th>
<th>Percentage Change In Land Area: Pre-Agricultural Times To Present</th>
<th>Proportion of Species That Are Locally Endemic&lt;sup&gt;a&lt;/sup&gt;</th>
<th>Total Number of Species In Ecosystem&lt;sup&gt;b&lt;/sup&gt;</th>
<th>“First Guess” Estimate for Indicator 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tropical rainforest</td>
<td>-3.9%</td>
<td>Higher</td>
<td>High</td>
<td>0.1</td>
</tr>
<tr>
<td>Tropical seasonal forest</td>
<td>-3.9%</td>
<td>Higher</td>
<td>High</td>
<td>0.1</td>
</tr>
<tr>
<td>Temperate evergreen forest</td>
<td>-20.6%</td>
<td>Lower</td>
<td>Medium</td>
<td>0.5</td>
</tr>
<tr>
<td>Temperate deciduous forest</td>
<td>-20.6%</td>
<td>Lower</td>
<td>Medium</td>
<td>0.5</td>
</tr>
<tr>
<td>Boreal forest</td>
<td>-20.6%</td>
<td>Lower</td>
<td>Medium</td>
<td>0.5</td>
</tr>
<tr>
<td>Woodland and shrubland</td>
<td>Woodland = -18.6%, Shrubland = -8.6%</td>
<td>Lower</td>
<td>Medium</td>
<td>0.5</td>
</tr>
<tr>
<td>Savannah</td>
<td>-19.4%</td>
<td>Higher</td>
<td>High</td>
<td>0.1</td>
</tr>
<tr>
<td>Temperate grassland</td>
<td>-19.4%</td>
<td>Lower</td>
<td>Medium</td>
<td>0.5</td>
</tr>
<tr>
<td>Tundra and alpine</td>
<td>0.0%</td>
<td>Lower</td>
<td>Medium</td>
<td>0.8</td>
</tr>
<tr>
<td>Desert and semi-desert shrub</td>
<td>-1.9%</td>
<td>Lower</td>
<td>Low</td>
<td>1.0</td>
</tr>
<tr>
<td>Extreme desert, rock, sand and ice</td>
<td>-1.9%</td>
<td>Lower</td>
<td>Low</td>
<td>1.0</td>
</tr>
<tr>
<td>Cultivated land</td>
<td>+1,760%</td>
<td>Lower</td>
<td>Low</td>
<td>1.0</td>
</tr>
<tr>
<td>Swamp and marsh</td>
<td>?</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lake and stream</td>
<td>?</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Estuaries</td>
<td>?</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>


<sup>a</sup> Myers (1997) states that the tropics and subtropics tend to have greater numbers of locally endemic species than the temperate and boreal zones.

<sup>b</sup> See Table AIV.2 below.

**Note:** The values for percentage changes in land area should be treated with caution. For example, Graetz (1994, p.134) quotes a -3.9% change in tropical forest land area from pre-agricultural times to the present. On the other hand, Adger and Brown (1994, p.75) present data from the FAO 1990 Assessment of tropical forests. This gives an 8% decrease in total forest area in the tropics between 1980 and 1990 alone.
Table 2. Calculation of Indicator 3 Based On Latitude of Ecosystems

<table>
<thead>
<tr>
<th>Ecosystem Type</th>
<th>Latitude</th>
<th>Latitude Where Ecosystem Type Most Widespread</th>
<th>&quot;First guess&quot; Estimate for Indicator 3</th>
<th>Notes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tropical rainforest</td>
<td>30°N to 20°S</td>
<td>0°N</td>
<td>0.0</td>
<td></td>
</tr>
<tr>
<td>Tropical seasonal forest</td>
<td>35°N to 30°S</td>
<td>20°N, 20°S</td>
<td>0.0</td>
<td></td>
</tr>
<tr>
<td>Temperate evergreen forest</td>
<td>40 to 55°N</td>
<td>50°N</td>
<td>0.3</td>
<td></td>
</tr>
<tr>
<td>Temperate deciduous forest</td>
<td>30 to 50°N</td>
<td>45°N</td>
<td>0.3</td>
<td></td>
</tr>
<tr>
<td></td>
<td>30 to 50°S</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Boreal forest</td>
<td>40 to 65°N</td>
<td>55°N</td>
<td>0.5</td>
<td></td>
</tr>
<tr>
<td>Woodland and shrubland</td>
<td>30 to 40°N</td>
<td>40°N</td>
<td>0.3</td>
<td></td>
</tr>
<tr>
<td>Savannah</td>
<td>30°N to 50°S</td>
<td>15°N, 15°S</td>
<td>0.3</td>
<td>Fewer species than tropical forests</td>
</tr>
<tr>
<td>Temperate grassland</td>
<td>25 to 55°N</td>
<td>40°N</td>
<td>0.5</td>
<td>Fewer species than temperate forests</td>
</tr>
<tr>
<td></td>
<td>25 to 30°S</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Tundra and alpine</td>
<td>60 to 90°N</td>
<td>60°N</td>
<td>0.7</td>
<td>Low biodiversity because of extreme climate</td>
</tr>
<tr>
<td>Desert and semi-desert shrub</td>
<td>20 to 40°N</td>
<td>30°N</td>
<td>1.0</td>
<td>Low biodiversity because of extreme climate</td>
</tr>
<tr>
<td></td>
<td>15 to 35°S</td>
<td>30°S</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Extreme desert, rock, sand and ice</td>
<td>20 to 40°N</td>
<td>30°N</td>
<td>1.0</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>30°S</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cultivated land</td>
<td>N/A</td>
<td>N/A</td>
<td>0.9</td>
<td>Range to be used because biodiversity depends on latitude</td>
</tr>
<tr>
<td>Swamp and marsh</td>
<td>N/A</td>
<td>N/A</td>
<td>?</td>
<td>As above</td>
</tr>
<tr>
<td>Lake and stream</td>
<td>N/A</td>
<td>N/A</td>
<td>?</td>
<td>As above</td>
</tr>
<tr>
<td>Estuaries</td>
<td>N/A</td>
<td>N/A</td>
<td>?</td>
<td></td>
</tr>
</tbody>
</table>


References


APPENDIX VI.1

INPUT AND OUTPUT DATA FOR CASE STUDY

Inputs other than use of machinery for Systems A, B and C are listed in Table 1 below. Use of machinery, and associated use of diesel, is listed in Tables 2 to 4 for Systems A, B and C respectively. Table 5 lists the outputs from each system. All the data are taken from Audsley et al. (1997).

All references in Appendix VI.1 are listed in the References section at the end of Chapter VI.
<table>
<thead>
<tr>
<th>Type of Input</th>
<th>Input</th>
<th>System A</th>
<th>Active Ingredients In Pesticides</th>
<th>System B</th>
<th>Active Ingredients In Pesticides</th>
<th>System C</th>
</tr>
</thead>
<tbody>
<tr>
<td>Seeds</td>
<td>Mercia seeds</td>
<td>185 kg</td>
<td>Fenpiclonil (50 g/litre); 0.036 kg</td>
<td>0.64 litres</td>
<td>Fenpiclonil (50 g/litre); 0.032 kg</td>
<td>200 kg</td>
</tr>
<tr>
<td></td>
<td>Galaxie seeds</td>
<td></td>
<td>Methiocarb (4%); 0.22 kg</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Variety mixture</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Phosphorus fertilisers</td>
<td>Triple superphosphate</td>
<td>130 kg</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Thomas meal</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>300 kg</td>
</tr>
<tr>
<td>Potassium fertiliser</td>
<td>Muriate of potash</td>
<td>100 kg</td>
<td></td>
<td></td>
<td></td>
<td>110 kg</td>
</tr>
<tr>
<td>Nitrogen fertilisers</td>
<td>Ammonium nitrate (34.5% N)</td>
<td>464 kg</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Ammonium nitrate (27.5% N)</td>
<td>350 kg</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Ammonium sulphate</td>
<td>190 kg</td>
<td></td>
<td></td>
<td></td>
<td>170 kg</td>
</tr>
<tr>
<td></td>
<td>Liquid urea</td>
<td>87 kg</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Other fertilisers</td>
<td>Farmyard manure</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lime</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Seed dressings</td>
<td>Dressing-Beret</td>
<td>0.72 litres</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Slug pellets (Draza)</td>
<td>5.50 kg</td>
<td></td>
<td></td>
<td></td>
<td>15,000 kg</td>
</tr>
<tr>
<td></td>
<td>Javelin Gold</td>
<td>5.00 litres</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Ambush C</td>
<td>0.25 litres</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>CCC700</td>
<td>2.30 litres</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Manganese sulphate</td>
<td>5.00 kg</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Starane</td>
<td>0.75 litres</td>
<td></td>
<td></td>
<td></td>
<td>50 m²</td>
</tr>
<tr>
<td></td>
<td>Tern</td>
<td>0.50 litres</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Punch C</td>
<td>0.40 litres</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Bravo</td>
<td>1.50 litres</td>
<td></td>
<td></td>
<td></td>
<td>3,000 kg</td>
</tr>
<tr>
<td></td>
<td>Terpal</td>
<td>1.00 litres</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Folicur</td>
<td>1.25 litres</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Aphox</td>
<td>0.14 kg</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Ioniz P</td>
<td>4.25 litres</td>
<td></td>
<td></td>
<td></td>
<td>Mecoprop-P (114 g/l); 0.485 kg</td>
</tr>
<tr>
<td>Type of Input</td>
<td>Input</td>
<td>System A</td>
<td>Active Ingredients In Pesticides</td>
<td>System B</td>
<td>Active Ingredients In Pesticides</td>
<td>System C</td>
</tr>
<tr>
<td>--------------</td>
<td>-------</td>
<td>----------</td>
<td>-------------------------------</td>
<td>----------</td>
<td>-------------------------------</td>
<td>----------</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Ioxynil (71 g/l): 0.302 kg</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Diflufenican (14 g/l): 0.06 kg</td>
<td></td>
</tr>
<tr>
<td>Calypso</td>
<td></td>
<td></td>
<td>2.50 litres</td>
<td></td>
<td>Chlorothalonil (TCPN) (300 g/l): 0.75 kg</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Hexaconazole (75 g/l): 0.188 kg</td>
<td></td>
</tr>
</tbody>
</table>

| Additional labour | Identification of limits of tolerance for fungal diseases | 3.0 hours |
|                  | Drying grain | 1.6 hours |
|                  | Manual weed control | 1.2 hours |

| Other | Net (6.84 m³/bale, 71 m³/kg) | 1.06 kg |
|       | Fuel oil | 55.1 kg |

|         |         |         |         |
|         |         |         |         |

|         |         |         |         |
|         |         |         |         |

Table continues...
<table>
<thead>
<tr>
<th>Timing</th>
<th>Activity</th>
<th>Hours of Labour</th>
<th>Input</th>
<th>Number of Units</th>
<th>Rate of Diesel Use (litres/hour)</th>
<th>Litres of Diesel</th>
</tr>
</thead>
<tbody>
<tr>
<td>Beginning</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>September</td>
<td>Primary cultivation</td>
<td>2.1</td>
<td>Four-furrow plough</td>
<td>1 ha</td>
<td>2.1 hours</td>
<td>37.8</td>
</tr>
<tr>
<td></td>
<td>Base fertilisation</td>
<td>0.4</td>
<td>Disk broadcaster (over 450 litres, 1.2m) (for TSP and</td>
<td>1 ha</td>
<td>0.4 hours</td>
<td>1.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>potash)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Tractor (4WD, 100kW)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Seedbed preparation</td>
<td>0.9</td>
<td>Rotary cultivator (4m)</td>
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<td>0.9 hours</td>
<td>1.2</td>
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<td></td>
<td>Twin wheels</td>
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<tr>
<td></td>
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<td></td>
<td>Tractor (4WD, 100kW)</td>
<td></td>
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</tr>
<tr>
<td>Middle March</td>
<td>Top fertilisation (ammonium nitrate)</td>
<td>0.4</td>
<td>Disk broadcaster (12m)</td>
<td>1 ha</td>
<td>0.4 hours</td>
<td>1.2</td>
</tr>
<tr>
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<td></td>
<td>Tractor (2WD, 50kW)</td>
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<tr>
<td>End April</td>
<td>Top fertilisation (ammonium nitrate)</td>
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<td>Disk broadcaster (12m)</td>
<td>1 ha</td>
<td>0.4 hours</td>
<td>1.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Tractor (2WD, 50kW)</td>
<td></td>
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<tr>
<td>Middle May</td>
<td>Top fertilisation (ammonium sulphate)</td>
<td>0.4</td>
<td>Disk broadcaster (12m)</td>
<td>1 ha</td>
<td>0.4 hours</td>
<td>1.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Tractor (2WD, 50kW)</td>
<td></td>
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</tr>
<tr>
<td>End June</td>
<td>Top fertilisation</td>
<td>0.4</td>
<td>Mounted crop sprayer, 1000 litres (15m) (for liquid</td>
<td>1 ha</td>
<td>0.4 hours</td>
<td>1.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>urea)</td>
<td></td>
<td></td>
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<td></td>
<td></td>
<td></td>
<td>Tractor (2WD, 50kW)</td>
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<tr>
<td>Beginning</td>
<td>Sowing</td>
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<td>Drilling machine (6m)</td>
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<td>0.75 hours</td>
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<td>October</td>
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<td>Tractor (4WD, 75kW)</td>
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<tr>
<td>Middle October</td>
<td>Slug pellets</td>
<td>0.3</td>
<td>Disk broadcaster (12m)</td>
<td>1 ha</td>
<td>0.34 hours</td>
<td>1.2</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td>Tractor (2WD, 50kW)</td>
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<tr>
<td>Beginning</td>
<td>Javelin Gold (Diflufenican and Isoproturon)</td>
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<td>Mounted crop sprayer, 1000 litres (15m)</td>
<td>1 ha</td>
<td>0.34 hours</td>
<td>1.3</td>
</tr>
<tr>
<td>November</td>
<td></td>
<td></td>
<td>Tractor (2WD, 50kW)</td>
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</tr>
<tr>
<td>Beginning</td>
<td>CCC700 (Chlormequat); Manganese sulphate</td>
<td>0.34</td>
<td>Mounted crop sprayer, 1000 litres (15m)</td>
<td>1 ha</td>
<td>0.34 hours</td>
<td>1.3</td>
</tr>
<tr>
<td>April</td>
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<td></td>
<td>Tractor (2WD, 50kW)</td>
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<tr>
<td>Middle April</td>
<td>CCC700 (Chlormequat); Starane (Fluroxypyr)</td>
<td>0.34</td>
<td>Mounted crop sprayer, 1000 litres (15m)</td>
<td>1 ha</td>
<td>0.34 hours</td>
<td>1.3</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Tractor (2WD, 50kW)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Timing</td>
<td>Activity</td>
<td>Hours of Labour</td>
<td>Input</td>
<td>Number of Units</td>
<td>Rate of Diesel Use (litres/hour)</td>
<td>Litres of Diesel</td>
</tr>
<tr>
<td>---------------------</td>
<td>--------------------------------------------------------------------------</td>
<td>----------------</td>
<td>----------------------------------------------------------------------</td>
<td>-----------------</td>
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</tr>
<tr>
<td>Middle April</td>
<td>Tern (Fenpropidin); Punch C (Flusilazole and Carbendazim); Bravo (Chlorothalonil)</td>
<td>0.34</td>
<td>Mounted crop sprayer (1000 litres) (15m) Tractor (2WD, 50kW)</td>
<td>1 ha</td>
<td>0.34 hours</td>
<td>3.8</td>
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<tr>
<td>Middle May</td>
<td>Tarpal (2-chloroethylphosphonic acid and Mepiquate chloride)</td>
<td>0.34</td>
<td>Mounted crop sprayer (1000 litres) (15m) Tractor (2WD, 50kW)</td>
<td>1 ha</td>
<td>0.34 hours</td>
<td>3.8</td>
</tr>
<tr>
<td></td>
<td>Folicur (Tebucanazole); Bravo (Chlorothalonil)</td>
<td>0.34</td>
<td>Mounted crop sprayer (1000 litres) (15m) Tractor (2WD, 50kW)</td>
<td>1 ha</td>
<td>0.34 hours</td>
<td>3.8</td>
</tr>
<tr>
<td>Middle June</td>
<td>Folicur (Tebucanazole); Aphox (Pirimicarb)</td>
<td>0.34</td>
<td>Mounted crop sprayer (1000 litres) (15m) Tractor (2WD, 50kW)</td>
<td>1 ha</td>
<td>0.34 hours</td>
<td>3.8</td>
</tr>
<tr>
<td>Beginning August</td>
<td>Harvesting</td>
<td>1.05</td>
<td>Combine harvester (5m) (150 kW) Tractor (2WD, 50kW)</td>
<td>1 ha</td>
<td>2.11 hours</td>
<td>35</td>
</tr>
<tr>
<td></td>
<td></td>
<td>2.11</td>
<td>Fourwheel trailer (8t) (two used) Tractor (2WD, 50 kW) (two used)</td>
<td>2.11 hours</td>
<td>2.5</td>
<td>5.27</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.44</td>
<td>Round baler Tractor (4WD, 100 kW)</td>
<td>11 bales</td>
<td>0.44 hours</td>
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<td></td>
<td></td>
<td>1.1</td>
<td>Frontloader Tractor (5WD, 100 kW)</td>
<td>1.1 hours</td>
<td>3</td>
<td>3.3</td>
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<td></td>
<td></td>
<td>0.3</td>
<td>Fourwheel trailer (8t) (two used) Tractor (2WD, 50 kW) (two used)</td>
<td>0.3 hours</td>
<td>2.46</td>
<td>0.74</td>
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<td><strong>TOTALS</strong></td>
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<td><strong>13.4</strong></td>
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<td></td>
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<td><strong>126.4</strong></td>
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Table 3. Use of Machinery In System B

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<tr>
<th>Timing</th>
<th>Activity</th>
<th>Hours of Labour</th>
<th>Input</th>
<th>Number of Units</th>
<th>Rate of Diesel Use (litres/hour)</th>
<th>Litres of Diesel</th>
</tr>
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<tbody>
<tr>
<td>Middle September</td>
<td>Primary cultivation</td>
<td>3.8</td>
<td>Two-furrow plough Tractor (4WD, 50kW)</td>
<td>1 ha</td>
<td>3.8 hours</td>
<td>9</td>
</tr>
<tr>
<td></td>
<td>Base fertilisation (Thomas meal)</td>
<td>0.7</td>
<td>Disk broadcaster (up to 450 litres, 12m) (for TSP and potash) Tractor (2WD, 41 kW)</td>
<td>1 ha</td>
<td>0.7 hours</td>
<td>2.5</td>
</tr>
<tr>
<td></td>
<td>Base fertilisation (potash)</td>
<td>0.7</td>
<td>Disk broadcaster (up to 450 litres, 12m) (for TSP and potash) Tractor (2WD, 41 kW)</td>
<td>1 ha</td>
<td>0.7 hours</td>
<td>1.75</td>
</tr>
<tr>
<td>Middle October</td>
<td>Seedbed preparation</td>
<td>1.2</td>
<td>Rotary cultivator (3m) Tractor (4WD, 50kW)</td>
<td>1 ha</td>
<td>1.2 hours</td>
<td>17</td>
</tr>
<tr>
<td>Middle March</td>
<td>Top fertilisation (ammonium nitrate)</td>
<td>0.6</td>
<td>Disk broadcaster (up to 450 litres, 12m) Tractor (2WD, 41 kW)</td>
<td>1 ha</td>
<td>0.6 hours</td>
<td>2.5</td>
</tr>
<tr>
<td>Beginning May</td>
<td>Top fertilisation (ammonium nitrate)</td>
<td>0.6</td>
<td>Disk broadcaster (up to 450 litres, 12m) Tractor (2WD, 41 kW)</td>
<td>1 ha</td>
<td>0.6 hours</td>
<td>1.5</td>
</tr>
<tr>
<td>Beginning June</td>
<td>Top fertilisation (ammonium sulphate)</td>
<td>0.6</td>
<td>Disk broadcaster (up to 450 litres, 12m) Tractor (2WD, 41 kW)</td>
<td>1 ha</td>
<td>0.6 hours</td>
<td>1.5</td>
</tr>
<tr>
<td>Middle October</td>
<td>Sowing</td>
<td>1.1</td>
<td>Drilling machine (3m) Tractor (2WD, 41 kW)</td>
<td>1 ha</td>
<td>1.1 hours</td>
<td>4.0</td>
</tr>
<tr>
<td>Beginning April</td>
<td>Ioniz P (Isoproturon, Mecprop-P, Ioxynil, Diflufenican)</td>
<td>0.9</td>
<td>Mounted crop sprayer (600 litres) (15m) Tractor (2WD, 41 kW)</td>
<td>1 ha</td>
<td>0.9 hours</td>
<td>3.1</td>
</tr>
<tr>
<td>Middle June</td>
<td>Calypso (Chlorothalonil, Hexaconazole)</td>
<td>0.9</td>
<td>Mounted crop sprayer (600 litres) (12m) Tractor (2WD, 41 kW)</td>
<td>1 ha</td>
<td>0.9 hours</td>
<td>2.79</td>
</tr>
<tr>
<td>End July</td>
<td>Harvesting</td>
<td>0.86</td>
<td>Combine harvester (5m) (150 kW) Straw chopper</td>
<td>1 ha</td>
<td>1.0 hours</td>
<td>35</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Fourwheel trailer (8t) Tractor (4WD, 50 kW)</td>
<td>1.0 hours</td>
<td></td>
<td>29.93</td>
</tr>
<tr>
<td></td>
<td>Baling</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td></td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Bale carting</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td></td>
<td>-</td>
</tr>
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<td>TOTALS</td>
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<td>-</td>
<td>-</td>
<td>105.0</td>
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<tr>
<td>Timing</td>
<td>Activity</td>
<td>Hours of Labour</td>
<td>Input</td>
<td>Number of Units</td>
<td>Rate of Diesel Use (litres/hour)</td>
<td>Litres of Diesel</td>
</tr>
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<td>--------------------------------------------</td>
<td>----------------</td>
<td>---------------------------------</td>
<td>-----------------</td>
</tr>
<tr>
<td>Beginning September</td>
<td>Primary cultivation</td>
<td>3.8</td>
<td>Two-furrow plough</td>
<td>1 ha</td>
<td>3.8 hours</td>
<td>34.2</td>
</tr>
<tr>
<td></td>
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<td></td>
<td>Tractor (4WD, 50kW)</td>
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<td></td>
<td>Base fertilisation (manure)</td>
<td>1.95</td>
<td>Manure spreader (4.5t-5.5t)</td>
<td>3 cartloads</td>
<td>1.95 hours</td>
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<td>Tractor (4WD, 50 kW)</td>
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<td></td>
<td></td>
<td>1.3</td>
<td>Hydraulic loader</td>
<td>1.3 hours</td>
<td>2.5 hours</td>
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<td></td>
<td>Seedbed preparation</td>
<td>0.7</td>
<td>Harrow with spring teeth (3m)</td>
<td>1 ha</td>
<td>0.7 hours</td>
<td>5.25</td>
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<td></td>
<td>Tractor (4WD, 50kW)</td>
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</tr>
<tr>
<td>Middle March</td>
<td>Top fertilisation (slurry)</td>
<td>1.0</td>
<td>Slurry pump</td>
<td>1 hour</td>
<td>5</td>
<td>5</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td>Tractor (4WD, 50kW)</td>
<td>1 hour</td>
<td></td>
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<tr>
<td></td>
<td></td>
<td>1.5</td>
<td>Three-point reel (300m)</td>
<td>1.5 hours</td>
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<td></td>
<td></td>
<td>PVC hoses (100m)</td>
<td>1.5 hours</td>
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<td></td>
<td></td>
<td>Three-point spreader</td>
<td>1.5 hours</td>
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<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Tractor (2WD, 41 kW)</td>
<td>1.5 hours</td>
<td></td>
<td></td>
</tr>
<tr>
<td>End April</td>
<td>Top fertilisation (slurry)</td>
<td>1.0</td>
<td>Slurry pump</td>
<td>1 hour</td>
<td>5</td>
<td>5</td>
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<tr>
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<td></td>
<td>Tractor (4WD, 50kW)</td>
<td>1 hour</td>
<td></td>
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<td>1.5</td>
<td>Three-point reel (300m)</td>
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<td>PVC hoses (100m)</td>
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<td>Three-point spreader</td>
<td>1.5 hours</td>
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<td></td>
<td>Tractor (2WD, 41 kW)</td>
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<tr>
<td>Beginning October</td>
<td>Sowing</td>
<td>1.1</td>
<td>Drilling machine (3m)</td>
<td>1 ha</td>
<td>3.4</td>
<td>3.74</td>
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<td>Tractor (2WD, 41kW)</td>
<td>1.1 hours</td>
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</tr>
<tr>
<td></td>
<td></td>
<td>0.8</td>
<td>Clod breaking rollers (3m)</td>
<td>1 ha</td>
<td>3.4</td>
<td>2.7</td>
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<td></td>
<td>Tractor (2WD, 41 kW)</td>
<td>0.8 hours</td>
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<tr>
<td>Mechanical maintenance</td>
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<td>Spring line cult (6m)</td>
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</tr>
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<td></td>
<td></td>
<td>Tractor (2WD, 41 kW)</td>
<td>0.6 hours</td>
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<tr>
<td>Mechanical maintenance</td>
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<td>0.6</td>
<td>Spring line cult (6m)</td>
<td>1 ha</td>
<td>4.0</td>
<td>2.4</td>
</tr>
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<td></td>
<td></td>
<td>Tractor (2WD, 41 kW)</td>
<td>0.6 hours</td>
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</tr>
<tr>
<td>Timing</td>
<td>Activity</td>
<td>Hours of Labour</td>
<td>Input</td>
<td>Number of Units</td>
<td>Rate of Diesel Use (litres/hour)</td>
<td>Litres of Diesel</td>
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<td>-----------------</td>
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</tr>
<tr>
<td>Beginning August</td>
<td>Harvesting</td>
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<td>Combine harvester (5m) (150 kW)</td>
<td>1 ha</td>
<td>35</td>
<td>28.35</td>
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<td>Fourwheel trailer (8t) (125 kW)</td>
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<td>2.5</td>
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<td></td>
<td>Tractor (4WD, 50 kW)</td>
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<tr>
<td>Baling</td>
<td></td>
<td>1.3</td>
<td>Round bale press</td>
<td>26 bales</td>
<td>6.7</td>
<td>8.71</td>
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<td>Tractor (4WD, 50 kW)</td>
<td>1.30 hours</td>
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<tr>
<td>Bale carting</td>
<td></td>
<td>2.7</td>
<td>Frontloader</td>
<td>2.7 hours</td>
<td>2.9</td>
<td>7.75</td>
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<td>Fourwheel trailer</td>
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<td></td>
<td></td>
<td></td>
<td>Tractor (4WD, 50 kW)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>TOTALS</td>
<td></td>
<td>21.7</td>
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</tr>
</tbody>
</table>

N.B. An additional allowance is made for use of lime assuming 0.6 hours labour, machinery equivalent to a disk broadcaster (under 450 litres) and tractor (2WD, 50 kW), and 1.5 litres diesel.

Table 5. Outputs for Systems A, B and C

<table>
<thead>
<tr>
<th>Output</th>
<th>System A</th>
<th>System B</th>
<th>System C</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dried grain (15% humidity)</td>
<td>8,000 kg (12% protein)</td>
<td>6,000 kg (11% protein)</td>
<td>4,000 kg (12% protein)</td>
</tr>
<tr>
<td>Straw</td>
<td>5,000 kg of which</td>
<td>3,500 kg of which</td>
<td>5,000 kg of which</td>
</tr>
<tr>
<td></td>
<td>2,000 kg baled into 11 bales</td>
<td>0 kg baled</td>
<td>5,000 kg baled into 26 small bales</td>
</tr>
</tbody>
</table>
APPENDIX VI.2
DATA FOR CALCULATION OF ENVIRONMENTAL BURDENS

All references in Appendix VI.2 are listed in the References section of Chapter VI.

1. Production and Maintenance of Agricultural Machinery

For steel and rubber production, machines were assumed to be 100% steel except for self-propelled machines that were 95% steel and 5% rubber (See Section 2). Energy utilisation was calculated for each machine related to three aspects:

- Manufacture
  Energy utilisation was calculated on the basis of MJ required per kg of finished machine using data in Bowers (1992, based on data in Fluck and Baird, 1980, which itself is based on data in Doering et al., 1977). It was assumed to be 100% electricity.

- Repairs and maintenance
  Energy utilisation was calculated as a percentage of the energy requirements for machinery manufacture using data in Mughal (1994). The energy mix was taken as: 62% electricity, 27% fuel oil, 3% diesel and 8% natural gas (Audsley et al., 1997, p.38).

- Transport (from the factory gate to the farm)
  Default distances of 1000 km by rail and 200 km by road were used for the study.

Total energy utilisation was calculated for the lifetime of the machinery and then divided by the units of utilisation over the lifetime (in hours, hectares or bales). This gave a value for average energy utilisation for production and maintenance of each item of machinery per hour, hectare or bale (see Table 1).

2. Manufacture of Tyres

The total extracted energy used in manufacture of the tyres was taken as 23,446 MJ per tonne of tyres (Guelorget et al., 1993, p.17; “HEAT - Average European” used from the PEMS database).
For manufacture of materials used in tyres, it was assumed that tyres are 16% natural rubber, 25% polybutadiene, 13% steel, 25% carbon black and 21% other materials (Guelorget et al., 1993). The value for polybutadiene includes styrene-butadiene rubber (SBR) and butyl because data are not available on the input-output data for SBR and butyl. Since butadiene is a major ingredient for these substances, this is likely to be a reasonable assumption. No burdens were allocated to the natural rubber or other materials due to a lack of data. Therefore, the data in this analysis represent the input-output data for 63% of the tyre’s weight, plus the manufacturing energy consumption.

The inputs and outputs associated with production of the polybutadiene, steel, and carbon black were calculated using data from the PEMS database. For the carbon black, it was assumed that production of one tonne requires 1.7 tonnes feed oil ("Oil - North European" in the PEMS database), 1570 m³ natural gas ("Natural Gas - North Sea (APME) in the PEMS database), 220 kWh electricity ("Electricity - Average European" in the PEMS database), and 1465 kWh fuel ("HEAT - Average European" in the PEMS database) (Guelorget et al., 1993).

Data omissions included: burdens associated with disposal of tyres and processing “waste” from the tyre manufacturing plant, transportation of the different materials and final tyres to the machinery manufacturing plant, and production of natural rubber.

3. Production of Synthetic Fertilisers

The study accounted for:

- Energy used in production of different fertilisers (based on Kongshaug, 1992, and Bøckmann et al., 1990) (Table 2).
- Natural gas as feedstock for nitrogen fertilisers (Kongshaug, 1992) (Table 2).
- Process emissions for nitrogen fertilisers (Bøckmann et al., 1990) (Table 3) and triple superphosphate (TSP) production (derived from Hoogenkamp, 1992, and Hazewinkel, 1992) (Table 4).

No data were available on process emissions for potassium fertiliser production, and therefore these data were not included in the study.
Table 1. Material Composition and Energy Consumption In Production and Maintenance for Different Types of Farm Machinery

<table>
<thead>
<tr>
<th>Materials</th>
<th>Energy</th>
<th>Space</th>
<th>Units of use</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Weight (kg)</td>
<td>% steel</td>
<td>% tyres</td>
</tr>
<tr>
<td>Tractor: 2WD, 50kW</td>
<td>3,400</td>
<td>95</td>
<td>5</td>
</tr>
<tr>
<td>Tractor: 2WD, 41kW</td>
<td>2,300</td>
<td>95</td>
<td>5</td>
</tr>
<tr>
<td>Tractor: 4WD, 50kW</td>
<td>3,900</td>
<td>95</td>
<td>5</td>
</tr>
<tr>
<td>Tractor: 4WD, 75kW</td>
<td>4,700</td>
<td>95</td>
<td>5</td>
</tr>
<tr>
<td>Tractor: 4WD, 100kW</td>
<td>5,500</td>
<td>95</td>
<td>5</td>
</tr>
<tr>
<td>Plough: two-furrow plough</td>
<td>600</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Plough: four-furrow plough</td>
<td>1,300</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Drilling machine: 3m</td>
<td>550</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Drilling machine: 6m</td>
<td>1,200</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Mounted crop sprayer: under 450L (12m)</td>
<td>130</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Mounted crop sprayer: over 450L (12m)</td>
<td>280</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Cultivator: 600L (12m)</td>
<td>400</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Cultivator: 1000L (15m)</td>
<td>800</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Cultivator: rotary cultivator (3m)</td>
<td>1,000</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Cultivator: rotary cultivator (4m)</td>
<td>1,300</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Cultivator: (2.2m)</td>
<td>700</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Cultivator: spring tine cultivator (6m)</td>
<td>500</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Twin wheels</td>
<td>160</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Combine harvester (5m) 150kW</td>
<td>11,500</td>
<td>95</td>
<td>5</td>
</tr>
<tr>
<td>Fourwheel trailer (8t)</td>
<td>2,500</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Round baler</td>
<td>1,700</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Frontloader</td>
<td>400</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Straw chopper</td>
<td>500</td>
<td>100</td>
<td>-</td>
</tr>
</tbody>
</table>
Table 1 (continued). Material Composition and Energy Consumption in Production and Maintenance for Different Types of Farm Machinery

<table>
<thead>
<tr>
<th>Materials</th>
<th>Materials</th>
<th>Energy</th>
<th>Space</th>
<th>Units of use</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Weight (kg)</td>
<td>% steel</td>
<td>% tyres</td>
<td>% PVC</td>
</tr>
<tr>
<td>Harrow with spring teeth (3m)</td>
<td>650</td>
<td>100</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Manure spreader (4.5t-5.5t)</td>
<td>1400</td>
<td>100</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Hydraulic loader</td>
<td>1600</td>
<td>100</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Slurry pump</td>
<td>380</td>
<td>100</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Three-point reel (300m)</td>
<td>450</td>
<td>100</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>PVC hoses (100m)</td>
<td>200</td>
<td>-</td>
<td>100</td>
<td>-</td>
</tr>
<tr>
<td>Three-point spreader</td>
<td>110</td>
<td>100</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Clod-breaking rollers (3m)</td>
<td>700</td>
<td>100</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Round bale press</td>
<td>1700</td>
<td>100</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

¹ Energy used in manufacture and retail is taken from Table 10.3 in Bowers (1992); where machines are not specified in that table, the average value of 15.79 MJ/kg is used in this table.

² Energy used in repair and maintenance over the lifetime is taken from Table 10.5 in Bowers (1992); where machines are not specified in that table, the average value of 0.55 is used in this table. The values are the percentage of manufacturing energy used in repair and maintenance.
Table 2. Energy Used In Production of Fertilisers

<table>
<thead>
<tr>
<th>Fertiliser</th>
<th>Feedstock Energy (^a)</th>
<th>Process Energy (^a)</th>
<th>Total Energy</th>
<th>Energy source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Calcium ammonium nitrate (27.5% N)</td>
<td>30.5 MJ/kg N</td>
<td>15.1 MJ/kg N</td>
<td>45.6 MJ/kg N</td>
<td>Natural gas</td>
</tr>
<tr>
<td>Ammonium nitrate (34.5% N)</td>
<td>30.5 MJ/kg N</td>
<td>13.8 MJ/kg N</td>
<td>44.3 MJ/kg N</td>
<td>Natural gas</td>
</tr>
<tr>
<td>Ammonium sulphate (21% N)</td>
<td>30.5 MJ/kg N</td>
<td>14.5 MJ/kg N</td>
<td>45 MJ/kg N</td>
<td>Natural gas</td>
</tr>
<tr>
<td>Urea (46% N)</td>
<td>30.5 MJ/kg N</td>
<td>32.5 MJ/kg N</td>
<td>63 MJ/kg N</td>
<td>Natural gas</td>
</tr>
<tr>
<td>Thomas meal (17% P(_2)O(_5), 7% P)</td>
<td></td>
<td></td>
<td>9.6 MJ/kg P</td>
<td>Heavy fuel oil</td>
</tr>
<tr>
<td>Triple superphosphate (46% P(_2)O(_5), 20% P)</td>
<td></td>
<td></td>
<td>29.2 MJ/kg P</td>
<td>67% heavy fuel oil</td>
</tr>
<tr>
<td>Potash (60% K(_2)O, 50% K)</td>
<td></td>
<td></td>
<td>5 MJ/kg K</td>
<td>87% heavy fuel oil</td>
</tr>
</tbody>
</table>

\(^a\) Assumed to be total extracted energy.

Source: Audsley et al., 1997, p.31-32 (based on Kongshaug, 1992).

Table 3. Process Emissions From Production of Nitrogen Fertilisers

<table>
<thead>
<tr>
<th>Type of Emission</th>
<th>Emission to Air (kg/kg N)</th>
<th>Emission to Water (kg/kg N)</th>
</tr>
</thead>
<tbody>
<tr>
<td>CO(_2)</td>
<td>1.570</td>
<td></td>
</tr>
<tr>
<td>N(_2)O</td>
<td>0.016</td>
<td></td>
</tr>
<tr>
<td>NH(_3)</td>
<td>0.013</td>
<td>0.00022</td>
</tr>
<tr>
<td>NO(_3)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>NO(_2)</td>
<td>0.013</td>
<td></td>
</tr>
</tbody>
</table>

Source: Audsley et al., 1997, p.31 (based on Kongshaug, 1992).

Table 4. Process Emissions From Production of Triple Superphosphate

<table>
<thead>
<tr>
<th>Type of Emission</th>
<th>Emission to Air (g/kg P)</th>
<th>Emission to Water (g/kg P)</th>
</tr>
</thead>
<tbody>
<tr>
<td>As</td>
<td>0.0075</td>
<td></td>
</tr>
<tr>
<td>Cd</td>
<td>0.014</td>
<td></td>
</tr>
<tr>
<td>Cu</td>
<td>0.045</td>
<td></td>
</tr>
<tr>
<td>Cr</td>
<td>0.050</td>
<td></td>
</tr>
<tr>
<td>Dust</td>
<td>2.32</td>
<td></td>
</tr>
<tr>
<td>F</td>
<td>0.46</td>
<td>167</td>
</tr>
<tr>
<td>Gypsum</td>
<td>7500</td>
<td></td>
</tr>
<tr>
<td>Hg</td>
<td>0.0095</td>
<td></td>
</tr>
<tr>
<td>Ni</td>
<td>0.038</td>
<td></td>
</tr>
<tr>
<td>NO(_x)</td>
<td>5.43</td>
<td></td>
</tr>
<tr>
<td>P(_2)O(_5)</td>
<td>0.45</td>
<td>103 (assumed as PO(_4)(^3-))</td>
</tr>
<tr>
<td>Pb</td>
<td>0.043</td>
<td></td>
</tr>
<tr>
<td>SO(_2)</td>
<td>11.4</td>
<td></td>
</tr>
<tr>
<td>Zn</td>
<td>0.058</td>
<td></td>
</tr>
</tbody>
</table>

4. Production of Nitrogen-Fixing Crop

Figure 1 shows the model used for the nitrogen-fixing crop in the study. It was assumed that the total nitrogen yield of the crop was 128 kg/ha (Garrett et al., 1992)\(^1\), and there was an additional 5% area requirement for seed production. In addition, it was assumed that:

- Diesel consumption was 36.3 litres/ha
- The burdens associated with machinery and buildings were in the same proportion as for diesel consumption for this crop compared with System C (i.e. 36.3/107.2)
- The soil drew down 2 kg methane per hectare.
- 12,500 kg organic matter per hectare were added to the soil (5,000 kg by roots (Davies et al., 1993, p.198) and 7,500 kg above-ground vegetative matter (adapted from data in Nix, 1997, p.80, which quotes 11,100 kg dry matter for grazed grass and grass silage)).

Figure 1. Environmental Burdens for Cultivation of Nitrogen-Fixing Crop

\[ \begin{align*}
\text{N-fixing crop} & \rightarrow 202 \text{ kg} \\
\text{Balance of} & \rightarrow 43 \text{ kg NO}_3\text{-N} \\
\text{+12 kg N to soil} & \rightarrow 12 \text{ kg N}_2\text{-N} \\
& \rightarrow 7 \text{ kg NH}_3\text{-N} \\
& \rightarrow 0.11 \text{ kg NO}_3\text{-N} \\
& \rightarrow 0.01 \text{ kg NO}_2\text{-N}
\end{align*} \]

Crop contains 128 kg N/ha

Source: adapted from data in Garrett et al., 1992.

5. Production of Pesticides

For energy used in production of pesticides, Green (1987) lists the requirements for almost 40 active ingredients (AIs), and these values were used in the study. For AIs not listed in Green (1987) but used in the study, the average energy requirement for that chemical family was used (again calculated from the data in Green, 1987). When no data were available for the AI or its chemical family, the average value for all the AIs listed in Green (1987) was used (after Audsley et al., 1997, p.33). Data on energy use are presented in Table 5.

\(^1\) The total nitrogen content was used to estimate nitrogen available for the wheat crop, assuming that all the nitrogen becomes available over an extended period of time (steady state assumption).
For emissions during pesticide production, as a first approximation, the total annual release of different substances by pesticide producers in England and Wales was divided by the total annual production of pesticides in the UK to give the average emissions per tonne of active ingredient. The results indicated that these emissions were likely to be negligible compared with emissions of pesticides in the field (see Table 6). However, they were included in the study for comparative purposes.

Table 5. Energy Use for Production of Pesticide Active Ingredients

<table>
<thead>
<tr>
<th>Type of Pesticide</th>
<th>Active Ingredient</th>
<th>Indirect Energy (MJ/kg Active Ingredient)</th>
<th>Direct Energy (MJ/kg Active Ingredient)</th>
<th>Total (MJ/kg Active Ingredient)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Naphtha</td>
<td>Natural Gas</td>
<td>Coke</td>
</tr>
<tr>
<td>Growth regulator</td>
<td>Clormequat</td>
<td>61.1</td>
<td>42.3</td>
<td>1.6</td>
</tr>
<tr>
<td></td>
<td>Mepiquat chloride</td>
<td>61.1</td>
<td>42.3</td>
<td>1.6</td>
</tr>
<tr>
<td></td>
<td>2-chloroethyl-phosphonic acid</td>
<td>61.1</td>
<td>42.3</td>
<td>1.6</td>
</tr>
<tr>
<td>Fungicide</td>
<td>Carbendazim</td>
<td>86.7</td>
<td>71.2</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>Chlorothalonil</td>
<td>38.0</td>
<td>14.0</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>Fenpiclonil</td>
<td>37.9</td>
<td>37.6</td>
<td>2.8</td>
</tr>
<tr>
<td></td>
<td>Fenprodidin</td>
<td>37.9</td>
<td>37.6</td>
<td>2.8</td>
</tr>
<tr>
<td></td>
<td>Flusilazole</td>
<td>37.9</td>
<td>37.6</td>
<td>2.8</td>
</tr>
<tr>
<td></td>
<td>Hexaconazole</td>
<td>37.9</td>
<td>37.6</td>
<td>2.8</td>
</tr>
<tr>
<td></td>
<td>Tebuconazole</td>
<td>37.9</td>
<td>37.6</td>
<td>2.8</td>
</tr>
<tr>
<td>Herbicide</td>
<td>Diflufenican</td>
<td>88.1</td>
<td>52.2</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>Fluoroxypr</td>
<td>71.8</td>
<td>45.6</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>Ioxynil</td>
<td>71.8</td>
<td>45.6</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>Isoproturon</td>
<td>99.7</td>
<td>59.7</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>Mecoprop-P</td>
<td>56.1</td>
<td>26.7</td>
<td>0</td>
</tr>
<tr>
<td>Insecticide</td>
<td>Cypermethrin</td>
<td>89</td>
<td>71.2</td>
<td>0</td>
</tr>
<tr>
<td>Aphicide</td>
<td>Pirimicarb</td>
<td>54.8</td>
<td>50.2</td>
<td>9</td>
</tr>
<tr>
<td>Slug pellet</td>
<td>Methiocarb</td>
<td>54.8</td>
<td>50.2</td>
<td>9</td>
</tr>
</tbody>
</table>

a Assumed to be total extracted energy required for production of intermediates used in pesticide production.
b Assumed to be total extracted energy required for production of energy used in pesticide production.
c 2 MJ added for packaging (Green, 1987, p.169).
Table 6. Emissions From Pesticide Production In the UK

<table>
<thead>
<tr>
<th>Receiving Medium</th>
<th>Substance</th>
<th>Total Emissions Per Tonne Active Ingredient, 1994</th>
</tr>
</thead>
<tbody>
<tr>
<td>Air</td>
<td>Atrazine</td>
<td>0.0000017 g</td>
</tr>
<tr>
<td></td>
<td>Benzene</td>
<td>4.19 g</td>
</tr>
<tr>
<td></td>
<td>Biocides</td>
<td>0.0002604 g</td>
</tr>
<tr>
<td></td>
<td>Chlorides</td>
<td>8.68 g</td>
</tr>
<tr>
<td></td>
<td>Dichloride</td>
<td>1771.26 g</td>
</tr>
<tr>
<td></td>
<td>Hydrogen chloride</td>
<td>0.001041 g</td>
</tr>
<tr>
<td></td>
<td>Particulates</td>
<td>9.81 g</td>
</tr>
<tr>
<td></td>
<td>Pesticides</td>
<td>0.0008977 g</td>
</tr>
<tr>
<td></td>
<td>Simazine</td>
<td>0.0086843 g</td>
</tr>
<tr>
<td></td>
<td>Sulphur dioxide</td>
<td>2.60 g</td>
</tr>
<tr>
<td></td>
<td>VOCs</td>
<td>41.87 g</td>
</tr>
<tr>
<td>Water</td>
<td>Amitrole</td>
<td>0.40 g</td>
</tr>
<tr>
<td></td>
<td>Chlorpyrifos</td>
<td>0.001667 g</td>
</tr>
<tr>
<td></td>
<td>Diquat</td>
<td>1.56 g</td>
</tr>
<tr>
<td></td>
<td>Hexachlorocyclohexane</td>
<td>0.0004688 g</td>
</tr>
<tr>
<td></td>
<td>Mercury</td>
<td>0.0081616 g</td>
</tr>
<tr>
<td></td>
<td>Parquat</td>
<td>1.74 g</td>
</tr>
<tr>
<td></td>
<td>Pentachlorophenol</td>
<td>0.0001041 g</td>
</tr>
<tr>
<td></td>
<td>compounds</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Simazine</td>
<td>0.30 g</td>
</tr>
<tr>
<td>Land</td>
<td>Aqueous residues</td>
<td>10690.08 g</td>
</tr>
<tr>
<td></td>
<td>Oil and oil/solid</td>
<td>38.20 g</td>
</tr>
<tr>
<td></td>
<td>mixture</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Pentachlorophenol</td>
<td>7.12 g</td>
</tr>
<tr>
<td></td>
<td>compounds</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Pesticides</td>
<td>0.99 g</td>
</tr>
</tbody>
</table>


6. Use of Fertilisers

6.1 Fate of Nutrients

Outputs in the harvested crop were calculated based on the quantities of each nutrient in the harvested grain and straw (Table 7). For nitrogen emissions, the values for nitrate (NO$_3$-N) emissions to water were calculated using a computer crop/soil/fungicide simulation program (Audsley et al., 1997, p.43). They were assessed as: 31.0 kg NO$_3$-N per hectare for System A, 7.8 kg NO$_3$-N per hectare for System B, and 7.3 kg NO$_3$-N per hectare for System C. Nitrous oxide (N$_2$O-N) emissions were calculated using the values in Table 8 (based on data in Armstrong-Brown et al., 1994). Nitrogen oxide (NO$_x$-N) emissions were calculated as 10% of the N$_2$O-N emissions (Audsley et al., 1997, p.49). Ammonia (NH$_3$-N) emissions from synthetic fertilisers were taken from Asman (1992) (Table 9). Ammonia emissions (NH$_3$-N) from manure and slurry were taken as 50% of the applied ammonium nitrogen (after Menzi, 1995); it was assumed that the proportion of ammonium nitrogen in total nitrogen applied was 20% for farmyard manure and 50% for liquid manure (after Audsley et al., 1997, p.45).
For phosphorus, it was assumed that 1.0 kg/ha leached into surface water. For potassium, it was assumed that 37 kg/ha leached into surface water (see Appendix VI.3: average value used from Brady and Weil, 1996, p.478).

Table 7. Amounts of Nitrogen, Phosphorus and Potassium In Wheat Grain and Straw

<table>
<thead>
<tr>
<th>Term</th>
<th>Nitrogen</th>
<th>Phosphorus</th>
<th>Potassium</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wheat grain</td>
<td>17.6 kg/t for 11% protein grain</td>
<td>3.2 kg/t</td>
<td>4.3 kg/t</td>
</tr>
<tr>
<td>Wheat straw</td>
<td>5.0 kg/t</td>
<td>0.7 kg/t</td>
<td>8.3 kg/t</td>
</tr>
</tbody>
</table>

Source: Audsley et al., 1997, p.40 (and similar to values in Jollans, 1985, p.36).
Note: Values are for dry matter; it is assumed that harvested grain and straw have a 15% moisture content.

Table 8. Losses of N₂O as Percentage of Nitrogen Applied To Soil

<table>
<thead>
<tr>
<th>Type of Nitrogen Fertiliser</th>
<th>November - April (0-10°C)</th>
<th>May - October (10-20°C)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nitrate</td>
<td>1.7</td>
<td>1.1</td>
</tr>
<tr>
<td>Ammonium</td>
<td>0.4</td>
<td>0.5</td>
</tr>
<tr>
<td>Urea</td>
<td>0.8</td>
<td>3.0</td>
</tr>
<tr>
<td>Miscellaneous¹</td>
<td>1.05</td>
<td>0.8</td>
</tr>
</tbody>
</table>

Source: Audsley et al., 1997, p.44.
¹ Assumed to be half nitrate, half ammonium.

Table 9. Percentage Loss of Nitrogen as Ammonia for Different Synthetic Fertilisers

<table>
<thead>
<tr>
<th>Fertiliser</th>
<th>Percentage Loss of N as NH₃-N</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ammonium sulphate</td>
<td>8</td>
</tr>
<tr>
<td>Urea</td>
<td>15</td>
</tr>
<tr>
<td>Ammonium nitrate</td>
<td>2</td>
</tr>
<tr>
<td>Calcium ammonium nitrate</td>
<td>2</td>
</tr>
<tr>
<td>Ammonia, direct application</td>
<td>1</td>
</tr>
<tr>
<td>Nitrogen solutions</td>
<td>2.5</td>
</tr>
<tr>
<td>Other straight nitrogen</td>
<td>2.5</td>
</tr>
<tr>
<td>Total straight nitrogen</td>
<td>4</td>
</tr>
<tr>
<td>Ammonium phosphate</td>
<td>4</td>
</tr>
<tr>
<td>Other NP N</td>
<td>3</td>
</tr>
<tr>
<td>NK N</td>
<td>2</td>
</tr>
<tr>
<td>NPK N</td>
<td>4</td>
</tr>
<tr>
<td>Compound N</td>
<td>4</td>
</tr>
</tbody>
</table>


6.2 Fate of Heavy Metals

The following aspects were considered in modelling the fate of heavy metals:

- Quantities applied in synthetic fertilisers, Thomas meal, manure and lime
- Quantities added to the soil through atmospheric deposition (only considered as part of sensitivity analysis)
- Quantities lost in eroded soil
- Quantities harvested in the grain.

The balance of heavy metals were assumed to remain in the soil. Only a very small proportion would leach out, and so it was not considered necessary to model this fate pathway.

Data used to model the different aspects are presented below.

**Quantities Applied**

The estimated quantities in synthetic fertilisers, Thomas meal, manure and lime are shown in Table 10.

<table>
<thead>
<tr>
<th>Heavy Metal</th>
<th>Triple Superphosphate</th>
<th>Potash Nitrate</th>
<th>Ammonium Nitrate</th>
<th>Calcium Ammonium Nitrate</th>
<th>Ammonium Sulphate</th>
<th>Urea</th>
<th>Thomas Meal</th>
<th>Farmland Manure (dry matter basis)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Arsenic</td>
<td>1.3</td>
<td>0.43</td>
<td>0.43</td>
<td>0.41</td>
<td>0.4</td>
<td>0.6</td>
<td>0.03</td>
<td>0.03</td>
</tr>
<tr>
<td>Cadmium</td>
<td>52</td>
<td>0.06</td>
<td>0.05</td>
<td>0.05</td>
<td>0.05</td>
<td>0.25</td>
<td>3.0</td>
<td>0.03</td>
</tr>
<tr>
<td>Chromium</td>
<td>261</td>
<td>2</td>
<td>4</td>
<td>2</td>
<td>2</td>
<td>2</td>
<td>59</td>
<td>16.0</td>
</tr>
<tr>
<td>Cobalt</td>
<td>2</td>
<td>2</td>
<td>5</td>
<td>2</td>
<td>2</td>
<td>6</td>
<td>4</td>
<td>0.5</td>
</tr>
<tr>
<td>Copper</td>
<td>45</td>
<td>5</td>
<td>7</td>
<td>4</td>
<td>6</td>
<td>40</td>
<td>26</td>
<td>18.75</td>
</tr>
<tr>
<td>Lead</td>
<td>3.5</td>
<td>5.5</td>
<td>1.9</td>
<td>1.1</td>
<td>1.1</td>
<td>12</td>
<td>0.4</td>
<td>19.44</td>
</tr>
<tr>
<td>Mercury</td>
<td>0.022</td>
<td>0.01</td>
<td>0.023</td>
<td>0.023</td>
<td>0.01</td>
<td>0.013</td>
<td>0.007</td>
<td>0.001</td>
</tr>
<tr>
<td>Molybdenum</td>
<td>3.2</td>
<td>0.25</td>
<td>0.25</td>
<td>0.25</td>
<td>0.25</td>
<td>0.25</td>
<td>7.8</td>
<td>0.16</td>
</tr>
<tr>
<td>Nickel</td>
<td>44</td>
<td>2.1</td>
<td>13</td>
<td>1.8</td>
<td>2</td>
<td>2</td>
<td>35</td>
<td>23.16</td>
</tr>
<tr>
<td>Zinc</td>
<td>299</td>
<td>46</td>
<td>50</td>
<td>30</td>
<td>44</td>
<td>68</td>
<td>116.71</td>
<td>92.79</td>
</tr>
<tr>
<td>Vanadium</td>
<td>2</td>
<td>2</td>
<td>6140</td>
<td></td>
<td></td>
<td></td>
<td>20.0</td>
<td></td>
</tr>
<tr>
<td>Fluorine</td>
<td>17000</td>
<td>7</td>
<td>136</td>
<td>136</td>
<td>18</td>
<td>5</td>
<td>250</td>
<td></td>
</tr>
<tr>
<td>Selenium</td>
<td>2.8</td>
<td>0.25</td>
<td>0.25</td>
<td>0.25</td>
<td>0.25</td>
<td>0.25</td>
<td>0.27</td>
<td>0.011</td>
</tr>
<tr>
<td>Tin</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>3</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Thallium</td>
<td>0.25</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.25</td>
<td>-</td>
<td></td>
</tr>
</tbody>
</table>

Source: BUWAL, 1991, for all fertilisers except farmyard manure; Alloway (1990, p.35) for farmyard manure and lime. For farmyard manure and lime, the maximum value from the range for each metal was multiplied by the TSP value for that metal divided by the maximum for phosphate fertilisers given in Alloway (1990, p.35). The purpose of this calculation was to make the different datasets more comparable.

Note: the values for TSP and the nitrate fertilisers are of the same order of magnitude as cited in Alloway (1990, p.35). The exceptions are vanadium for TSP (2-1,600 mg/kg), mercury in nitrate fertilisers (0.3-2.9 mg/kg), and selenium (0 mg/kg) (Alloway, 1990, p.35).

Some values cited in the literature for quantities deposited from the atmosphere are given in Table 11. The values used in the study are also given in this table.
Table 11. Atmospheric Deposition of Heavy Metals (g/ha/year)

<table>
<thead>
<tr>
<th>Heavy Metal</th>
<th>Alloway (1990, p.37, 227)</th>
<th>White, 1987, p.168</th>
<th>Values Used In Study</th>
</tr>
</thead>
<tbody>
<tr>
<td>Arsenic</td>
<td>8 – 55</td>
<td>-</td>
<td>32</td>
</tr>
<tr>
<td>Cadmium</td>
<td>&lt;100</td>
<td>0.8 (minimum)</td>
<td>0.05c</td>
</tr>
<tr>
<td>Chromium</td>
<td>21 – 88</td>
<td>-</td>
<td>55</td>
</tr>
<tr>
<td>Cobalt</td>
<td>-</td>
<td>0.3 (minimum)</td>
<td>0.3</td>
</tr>
<tr>
<td>Copper</td>
<td>98 – 480</td>
<td>536 (median)</td>
<td>289</td>
</tr>
<tr>
<td>Lead</td>
<td>160 – 450</td>
<td>189 (median)</td>
<td>305</td>
</tr>
<tr>
<td>Mercury</td>
<td>0.2</td>
<td>-</td>
<td>0.2</td>
</tr>
<tr>
<td>Molybdenum</td>
<td>-</td>
<td>&lt;0.3</td>
<td>-</td>
</tr>
<tr>
<td>Nickel</td>
<td>35 – 110</td>
<td>39 (median)</td>
<td>73</td>
</tr>
<tr>
<td>Zinc</td>
<td>490 – 1,200</td>
<td>19 (median)</td>
<td>845</td>
</tr>
<tr>
<td>Vanadium</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Fluorine</td>
<td>-</td>
<td>-</td>
<td>183d</td>
</tr>
<tr>
<td>Selenium</td>
<td>2.2 – 6.5</td>
<td>-</td>
<td>4.4</td>
</tr>
<tr>
<td>Tin</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

a For UK in non-urban locations (based on Cawse, 1978).
b For Europe and North America (based on Sposito and Page, 1984).
c Calculated from world annual atmospheric emissions of 2,551 tonnes (Jackson and MacGillivray, 1995, p.A.27) and total world surface area of 5 x 10^8 km^2.
d Deposition rate of 100 ug/m^2/day adopted as the ambient air standard for gaseous F^- by South Carolina Pollution Control Authority (Low and Bloom, 1988). Half that value used here.

Quantities Leaving System In Eroded Soil

Assuming an even distribution of heavy metals throughout the furrow slice, the percentage of added heavy metals leaving the system in eroded soil is equal to the percentage of the total soil in the furrow slice that is lost in erosion.

Quantities Leaving System In Harvested Crops

Data on the concentration of heavy metals in plants and the soil are given in Table 12. These data were used to calculate the average mass of heavy metals in the soil and plant material per hectare. These values were then used to calculate the partitioning of heavy metals between the soil and harvested grain. The final percentage of each heavy metal harvested in the grain for each system under analysis (per hectare) was calculated as:

Total quantity of heavy metal applied in system

Multiplied by the appropriate percentage value for heavy metals in the harvested crop shown in Table 12

Multiplied by the yield in tonnes per hectare, divided by 6.5 tonnes per hectare.
This method was used because it gives a very rough estimate of the fate of heavy metals, accounting for differences between heavy metals in uptake by plants and harvested yield differences between the systems under analysis. More accurate modelling is outside the scope of this thesis, and indeed is an area where there are very limited data for undertaking such an exercise.

### Table 12. Concentrations of Heavy Metals In Soils and Plants

<table>
<thead>
<tr>
<th>Heavy Metal</th>
<th>Normal Range in Soils (mg/kg dry matter)</th>
<th>Normal Range in Plants (mg/kg dry matter)</th>
<th>Average Mass in Soil Per Hectare (kg/ha)</th>
<th>Average Mass in Plants Per Hectare (g/ha)</th>
<th>Percentage of Total Heavy Metal (Soil + Plants) in Plants</th>
</tr>
</thead>
<tbody>
<tr>
<td>Arsenic</td>
<td>0.1 - 40</td>
<td>0.02 - 7</td>
<td>38.2</td>
<td>22.8</td>
<td>0.060 %</td>
</tr>
<tr>
<td>Cadmium</td>
<td>0.01 - 2.0</td>
<td>0.1 - 2.4</td>
<td>1.9</td>
<td>8.1</td>
<td>0.425 %</td>
</tr>
<tr>
<td>Chromium</td>
<td>5 - 1,500</td>
<td>0.03 - 14</td>
<td>1,434</td>
<td>45.6</td>
<td>0.003 %</td>
</tr>
<tr>
<td>Cobalt</td>
<td>0.5 - 65</td>
<td>0.02 - 1</td>
<td>62.4</td>
<td>6.6</td>
<td>0.011 %</td>
</tr>
<tr>
<td>Copper</td>
<td>2 - 250</td>
<td>5 - 20</td>
<td>240</td>
<td>81.3</td>
<td>0.034 %</td>
</tr>
<tr>
<td>Lead</td>
<td>2 - 300</td>
<td>0.2 - 20</td>
<td>288</td>
<td>65.7</td>
<td>0.023 %</td>
</tr>
<tr>
<td>Mercury</td>
<td>0.01 - 0.5</td>
<td>0.005 - 0.17</td>
<td>0.5</td>
<td>0.6</td>
<td>0.120 %</td>
</tr>
<tr>
<td>Molybdenum</td>
<td>0.1 - 40</td>
<td>0.03 - 5</td>
<td>38.2</td>
<td>16.3</td>
<td>0.043 %</td>
</tr>
<tr>
<td>Nickel</td>
<td>2 - 750</td>
<td>0.02 - 5</td>
<td>716</td>
<td>16.3</td>
<td>0.002 %</td>
</tr>
<tr>
<td>Zinc</td>
<td>1 - 900</td>
<td>1 - 400</td>
<td>858.2</td>
<td>1,300</td>
<td>0.151 %</td>
</tr>
<tr>
<td>Vanadium</td>
<td>3 - 500</td>
<td>0.001 - 1.5</td>
<td>479</td>
<td>4.9</td>
<td>0.001 %</td>
</tr>
<tr>
<td>Fluorine</td>
<td>200 - 300b*</td>
<td>&lt;10 - 16b*</td>
<td>476.3</td>
<td>68.3</td>
<td>0.014 %</td>
</tr>
<tr>
<td>Selenium</td>
<td>0.1 - 5</td>
<td>0.001 - 2</td>
<td>4.9</td>
<td>6.5</td>
<td>0.132 %</td>
</tr>
<tr>
<td>Tin</td>
<td>1 - 200</td>
<td>0.2 - 6.8</td>
<td>191.5</td>
<td>22.8</td>
<td>0.012 %</td>
</tr>
</tbody>
</table>


* Data from Elrashidi and Lindsay, 1987.

b Data for lichens from Perkins and Millar, 1987a, 1987b.

Note: Average mass in soil calculated on basis of 1,905 tonnes dry matter in soil per hectare (for furrow slice), and average mass in plants calculated on basis of 6,500 kg dry matter per hectare (Brady et al., 1996, p.715).

### 7. Soil Compaction

Table 13 lists the relevant data used to calculate the soil compaction value in each system. Using the method developed in Chapter V, Section 4.6, the results are calculated as shown in Table 14.
### Table 13. Calculation of Soil Compaction

<table>
<thead>
<tr>
<th>System</th>
<th>Activity</th>
<th>Mass of Machinery (kg)</th>
<th>Time for Operations (hours)</th>
<th>Soil Compaction Indicator Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>System A</td>
<td>Primary cultivation: plough (4 furrow)</td>
<td>6,800</td>
<td>2.1</td>
<td>14.28</td>
</tr>
<tr>
<td></td>
<td>Seedbed preparation: rotary cultivator</td>
<td>6,960</td>
<td>0.9</td>
<td>6.26</td>
</tr>
<tr>
<td></td>
<td>Base fertilisation: disk broadcaster</td>
<td>3,680</td>
<td>0.4</td>
<td>1.47</td>
</tr>
<tr>
<td></td>
<td>Top fertilisation: disk broadcaster</td>
<td>3,680</td>
<td>1.2</td>
<td>4.42</td>
</tr>
<tr>
<td></td>
<td>Top fertilisation: crop sprayer (for liquid urea)</td>
<td>4,200</td>
<td>0.4</td>
<td>1.68</td>
</tr>
<tr>
<td></td>
<td>Sowing seed: drilling machine</td>
<td>5,900</td>
<td>0.75</td>
<td>4.43</td>
</tr>
<tr>
<td></td>
<td>Pesticide application: crop sprayer</td>
<td>4,200</td>
<td>2.4</td>
<td>10.08</td>
</tr>
<tr>
<td></td>
<td>Pesticide application: disk broadcaster (for slug pellets)</td>
<td>3,680</td>
<td>0.3</td>
<td>1.10</td>
</tr>
<tr>
<td></td>
<td>Harvesting: combine harvester</td>
<td>11,500</td>
<td>1.1</td>
<td>12.65</td>
</tr>
<tr>
<td></td>
<td>Harvesting: trailer</td>
<td>5,900+8,000/2^a</td>
<td>2.1</td>
<td>20.79</td>
</tr>
<tr>
<td>System B</td>
<td>Primary cultivation: plough (2 furrow)</td>
<td>4,500</td>
<td>3.8</td>
<td>17.10</td>
</tr>
<tr>
<td></td>
<td>Seedbed preparation: rotary cultivator</td>
<td>4,900</td>
<td>1.2</td>
<td>5.88</td>
</tr>
<tr>
<td></td>
<td>Base fertilisation: disk broadcaster</td>
<td>2,430</td>
<td>1.4</td>
<td>3.40</td>
</tr>
<tr>
<td></td>
<td>Top fertilisation: disk broadcaster</td>
<td>2,430</td>
<td>1.8</td>
<td>4.37</td>
</tr>
<tr>
<td></td>
<td>Sowing: drilling machine</td>
<td>2,850</td>
<td>1.1</td>
<td>3.14</td>
</tr>
<tr>
<td></td>
<td>Pesticide application: crop sprayer</td>
<td>2,700</td>
<td>1.8</td>
<td>4.86</td>
</tr>
<tr>
<td></td>
<td>Harvesting: combine harvester</td>
<td>12,000</td>
<td>0.9</td>
<td>10.8</td>
</tr>
<tr>
<td></td>
<td>Harvesting: trailer</td>
<td>6,400+6,000/2^a</td>
<td>1.0</td>
<td>9.4</td>
</tr>
<tr>
<td>System C</td>
<td>Primary cultivation: plough (2 furrow)</td>
<td>4,500</td>
<td>3.8</td>
<td>17.1</td>
</tr>
<tr>
<td></td>
<td>Seedbed preparation: harrow with spring teeth</td>
<td>4,550</td>
<td>0.7</td>
<td>3.19</td>
</tr>
<tr>
<td></td>
<td>Base fertilisation: manure spreader</td>
<td>5,300+7,500^b</td>
<td>1.95</td>
<td>24.96</td>
</tr>
<tr>
<td></td>
<td>Top fertilisation: slurry spreading</td>
<td>3,060</td>
<td>3.00</td>
<td>9.18</td>
</tr>
<tr>
<td></td>
<td>Sowing: clod breaking</td>
<td>3,000</td>
<td>0.8</td>
<td>2.40</td>
</tr>
<tr>
<td></td>
<td>Sowing: drilling</td>
<td>2,850</td>
<td>1.1</td>
<td>3.14</td>
</tr>
<tr>
<td></td>
<td>Maintenance: spring tine cultivator</td>
<td>2,800</td>
<td>1.2</td>
<td>3.36</td>
</tr>
<tr>
<td></td>
<td>Harvesting: combine harvester</td>
<td>11,500</td>
<td>0.8</td>
<td>9.2</td>
</tr>
<tr>
<td></td>
<td>Harvesting: trailer</td>
<td>6,400+4,000/2^a</td>
<td>1.0</td>
<td>8.4</td>
</tr>
</tbody>
</table>

^a Total weight of grain halved to account for change in load as grain is harvested.

^b Total weight of manure halved to account for manure spread during base fertilisation.

### Table 14. Calculation of Soil Compaction Indicator Value (Per Functional Unit)

<table>
<thead>
<tr>
<th>Soil Compaction Indicator Value Per Hectare Wheat Production</th>
<th>Soil Compaction Indicator Value Per Functional Unit</th>
<th>N-Fixing Crop</th>
<th>Green Manure^a</th>
<th>Soil Compaction Indicator Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>System A 77.16</td>
<td>9.9</td>
<td>-</td>
<td>-</td>
<td>9.9</td>
</tr>
<tr>
<td>System B 58.95</td>
<td>11.0</td>
<td>-</td>
<td>1.2</td>
<td>12.2</td>
</tr>
<tr>
<td>System C 80.93</td>
<td>21.3</td>
<td>4.3</td>
<td>-</td>
<td>25.6</td>
</tr>
</tbody>
</table>

^a Assuming soil compaction is in the same proportion as for diesel consumption in this crop compared with System C (i.e. 36.3/107.2).
8. Organic Matter In the Soil

Total quantities of organic matter (measured as dry matter) entering the soil from each system are shown in Table 15. The dry matter in farmyard manure and liquid manure were not included in the analysis for System C because the assessment method used to account for manure (Chapter VI, Section 4.2.2) meant that this source of organic matter was not relevant for inclusion.

Table 15. Sources of Organic Matter (kg dry matter per functional unit)

<table>
<thead>
<tr>
<th>Source of Organic Matter</th>
<th>System A</th>
<th>System B</th>
<th>System C</th>
</tr>
</thead>
<tbody>
<tr>
<td>Straw incorporation</td>
<td>326 kg</td>
<td>556 kg</td>
<td>-</td>
</tr>
<tr>
<td>Allowance for allocation on the basis of physical causality (see Chapter VI, Section 4.2.1)</td>
<td>218 kg</td>
<td>-</td>
<td>1,119 kg</td>
</tr>
<tr>
<td>Allowance for N-fixing crop or green manure(^a)</td>
<td>-</td>
<td>500 kg</td>
<td>1,750 kg</td>
</tr>
<tr>
<td>Roots and stubble(^b)</td>
<td>1,000 kg</td>
<td>1,090 kg</td>
<td>1,000 kg</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td>1,544 kg</td>
<td>2,146 kg</td>
<td>3,869 kg</td>
</tr>
</tbody>
</table>

Note: All values are dry matter (assuming straw is 85% dry matter).
\(^a\) For System C, the organic matter in manure is not included because of the allocation method used to assess use of manure (see Chapter VI, Section 4.2.2). Instead, the organic matter added by cultivation of the N-fixing crop or green manure is assessed (see Appendix VI.2, Section 4).
\(^b\) Assuming 2,500 kg/ha dry matter in roots of winter cereals (Davies et al., 1993, p.198), 4,000 kg/ha above-ground dry matter (Audsley, pers.comm.), and that this is correlated with an average grain yield of 6,500 kg. In other words, each 1,000 kg grain yield is associated with 385 kg dry matter in roots and 615 kg above-ground dry matter that is subsequently incorporated into the soil. [N.B. This assumes that the grain : vegetative dry matter ratio is constant between the three systems. This seems a reasonable assumption for agricultural systems which aim to maximise grain yield rather than vegetative dry matter production from any one plant (i.e. the ratio is unlikely to change by a large amount between different cultivation systems).]

References

References are listed in the References section of Chapter VI.
APPENDIX VI.3
MODELLING THE CYCLING OF NUTRIENTS
IN AGRICULTURAL SYSTEMS

1. Introduction

A general format for the flow of nutrients into and out of agricultural soils can be described, as shown in Figure 1. Inputs of nutrients occur through incorporation of organic matter, use of synthetic fertilisers, weathering of the rock substrate, and atmospheric deposition (either as particles or dissolved in rain). Outputs of nutrients occur in harvested crops, leached substances, via eroded soil and as gaseous emissions. In the soil itself, the nutrients may be in available form, meaning that they can be taken up by the crop, or immobilised in forms that cannot be utilised by the crop.

In Sections 2, 3 and 4, I discuss modelling of these inputs and outputs for nitrogen, phosphorus and potassium. In Section 5, I discuss the rates of release of available forms of these nutrients in the soil.

Figure 1. Inputs and Outputs of Nutrients In Agricultural Soil

Organic matter → Unavailable nutrient → Harvested crop
Synthetic fertilisers → Unavailable nutrient → Leaching
Mineral weathering → Available nutrient → Soil erosion
Atmospheric deposition → Available nutrient → Gaseous emissions
2. Modelling Nitrogen Flows In the Soil

*Availability of nitrogen*

Pimentel *et al.* (1995) state that a tonne of fertile agricultural topsoil typically contains 1-6 kg nitrogen, while Parkinson (1995, p.104-6) gives a value of 0.88-2.19 kg nitrogen. Brady and Weil (1996, p.403) give values of 0.2-5.0 kg nitrogen (average 1.5 kg) per tonne soil.

Between 95 and 99% of nitrogen occurs in organic forms (Brady and Weil, 1996, p.404). However, plants require nitrogen in the form of inorganic ammonium ($\text{NH}_4^+$) or nitrate ($\text{NO}_3^-$) ions, and organic nitrogen is converted into these forms by the action of soil organisms. Once present as these ions, the nitrogen is easily leached (although a small proportion of the ammonium is trapped in relatively unavailable form in the mineral colloids). Therefore, the challenge in agricultural production is to make ammonium and nitrate ions available when they are required by the crop, so as to minimise losses by leaching.

*Organic matter*

The nitrogen content of manures ranges from 6.0 kg N/tonne for cattle manure (25% dry matter) to 29 kg N/tonne for broiler/turkey litter (60% dry matter). The nitrogen content of slurries ranges from 2.3 kg N/m$^3$ for beef cattle slurry to 5.0 kg N/m$^3$ for pig slurry (MAFF, 1994, p.8). MAFF recommends that the amount of total nitrogen in organic manures spread on land should not exceed 250 kg N/ha/year (MAFF, 1994, p.11).

Another important organic source of soil nitrogen is **biological nitrogen fixation**. This is undertaken by a range of micro-organisms including species of bacteria, actinomycetes and blue-green algae. Some are only found in association with higher plants (in symbiotic relationships) while others are free-living. The amounts of nitrogen fixed vary between different organisms; clover and beans in association with the bacteria Rhizobium fix 100-150 kg N/ha/year and 30-50 kg N/ha/year respectively. Alders (*Alnus* spp.) in association with the actinomycete *Frankia* fix 50-150 kg N/ha/year. Nonsymbiotic organisms such as the cyanobacteria fix 10-50 kg N/ha/year, and other bacteria fix 5-10 kg N/ha/year (Brady and Weil, 1996, p.423).
**Synthetic fertilisers**

Typical application rates are 0-225 kg N/ha/year for cereals, 0-190 kg N/ha/year for oilseed rape, 0-240 kg N/ha/year for potatoes, and 0-420 kg N/ha/year for grass$^2$ (MAFF, 1994).

**Mineral weathering**

Weathering of minerals makes a negligible contribution to the soil nitrogen each year because most rocks contain very little nitrogen.

**Atmospheric deposition**

Rainfall adds 1-25 kg N/ha/year nitrogen to the soil (Brady and Weil, 1996, p.428).

**Harvested crop**

Examples of nitrogen removal in harvested crops are: 120 kg N/ha for cereals (including straw), 200 kg N/ha for sugar beet, 180 kg N/ha for potatoes, and 160 kg N/ha for grass silage (Parkinson, 1995, p.101).

**Leaching**

Brady and Weil (1996, p.412) state that annual loss of nitrogen (as nitrate) by leaching from well managed agricultural land is 5 to 10% of the nitrogen applied. This implies losses in the range 0 to 25 kg N/ha/year nitrogen for arable crops.

**Soil erosion**

Since eroded soil typically contains three times more nutrients than soil left in the field (Pimentel *et al.*, 1995), the mass of nitrogen in eroded soil can be calculated as 0.6-15.0 kg (average 4.5 kg) N/tonne eroded topsoil (derived from data in Brady and Weil, 1996, p.403).

---

$^2$ The highest value is for four cuts of grass silage where nitrogen is applied for each cut.
Gaseous emissions

Organic matter in the soil is broken down by organisms to ammonium ions. This process is called **mineralisation** (or **ammonification**), and its rate is determined by factors such as the temperature, moisture and aeration of the soil (Brady and Weil, 1996, p.405). Conditions favouring mineralisation are consistently high temperatures, a fluctuating moisture content and good aeration of the soil. Once formed, the ammonium may be converted to ammonia gas (NH₃) according to the reaction:

\[
\text{NH}_4^+ + \text{OH}^- \rightarrow \text{NH}_3 + \text{H}_2\text{O}
\]

The reaction is driven to the right (i.e. more ammonia gas is formed) by high pH, high temperature, and low moisture conditions. For example, it is favoured when manure is left on the surface of the soil and allowed to dry out; incorporation of the manure into the top few centimetres of soil can reduce ammonia losses by 25 to 75% compared with leaving it on the surface of the soil (Brady and Weil, 1996, p.407).

Some of the ammonium is converted to nitrate ions in a process called **nitrification**. This is a two-step process where ammonium is converted first to nitrite (NO₂⁻) and then to nitrate ions, the first step being mediated by a specific group of autotrophic³ bacteria called *Nitrosomonas* and the second step by another group of bacteria called *Nitrobacter*. Conditions that favour nitrification include availability of ammonium ions, good soil aeration, moderate moisture content, and high temperatures (20 to 30°C) (Brady and Weil, 1996, p.408-9). Small amounts of nitrous oxide (N₂O) may be formed during nitrification through the decomposition of unstable intermediates (Royal Commission, 1996, p.36).

Once formed, nitrate ions may be converted by the action of organisms in the soil into gases, in a series of reactions collectively referred to as **denitrification** (Brady and Weil, 1996, p.413):

\[
2\text{NO}_3^- \rightarrow 2\text{NO}_2^- \rightarrow 2\text{NO} \rightarrow \text{N}_2\text{O} \rightarrow \text{N}_2
\]

³ Autotrophic organisms are defined as obtaining all their carbon requirements from carbon dioxide or carbonates, and energy by oxidising inorganic elements/compounds or from radiant energy. In contrast, heterotrophic organisms can only obtain energy by decomposing organic compounds.
The proportions of the three types of gas (NO, N₂O and N₂) formed depend upon factors such as the pH, temperature, degree of oxygen depletion, and concentration of nitrate and nitrite ions available. Loss of nitric oxide (NO) is generally small compared with the other two gases. Formation of nitrous oxide (N₂O) is favoured if the concentrations of nitrate and nitrite ions are high, the supply of oxygen is not too low, and the pH is low.

These processes are summarised in Figure 2. The total loss of nitrogen as gaseous emissions from well-drained soils is generally no more than 5 to 15 kg N/ha/year. However, where drainage is restricted and large amounts of fertiliser are applied, losses of 30 to 60 kg N/ha/year have been observed (Brady and Weil, 1996, p.415).

Figure 2. Simplified Representation of the Nitrogen Cycle In Soil

<table>
<thead>
<tr>
<th>Organic matter</th>
<th>NH₃</th>
<th>N₂O</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mineralisation</td>
<td>NH₄⁺</td>
<td>NO₂⁻</td>
</tr>
<tr>
<td>Nitrification</td>
<td></td>
<td>Nitrification</td>
</tr>
<tr>
<td>Denitrification</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

3. Modelling Phosphorus Flows In the Soil

Availability of phosphorus

Pimentel et al. (1995) state that a tonne of fertile agricultural topsoil typically contains 1-3 kg phosphorus, while Parkinson (1995, p.104-6) gives a value of 0.22-1.10 kg phosphorus per tonne soil. Brady and Weil (1996, p.21, 403, 447, 478) give values of 0.1-0.9 kg phosphorus (average 0.5 kg).

Phosphorus in the soil occurs in organic and inorganic forms; according to the soil type, between 20 and 80% of the phosphorus in mineral soils occurs in organic forms (Brady and Weil, 1996, p.456). Plants absorb phosphorus as inorganic phosphate ions (HPO₄²⁻ and H₂PO₄⁻) in soil solution, and also take up some soluble organic phosphorus compounds. However, phosphorus in these forms amounts to no more than 0.01% of the total phosphorus in the soil (Brady and Weil, 1996, p.456). The remainder occurs as compounds with low to very low solubility in the mineral fraction of the soil and
in organic matter. When soluble phosphorus is added to soil containing low overall concentrations of phosphorus, it is quickly converted through chemical reactions to fixed (i.e. unavailable) forms. Alternatively, the phosphorus may be assimilated by micro-organisms and plants, and become incorporated into organic matter where it is also unavailable to the crop (until the organic matter is decomposed). As a result, only a relatively small fraction of the added phosphorus will be used by the crop in the year of application (10 to 15%) (Brady and Weil, 1996, p.447). Furthermore, over a period of days, months and years the fixed phosphorus is converted into forms that are less and less soluble. Many agricultural soils, therefore, have built up relatively high contents of fixed phosphorus due to the addition of phosphorus over a number of years. These soils can be described as saturated, and in fact only moderate applications of phosphorus are required on an annual basis once soils have reached this state, because the phosphorus-fixing capacity of the soil has largely been satisfied (Brady and Weil, 1996, p.447, 471).

**Organic matter**

The phosphorus content of manures ranges from 1.5 kg P/tonne for cattle manure (25% dry matter) to 10.9 kg P/tonne for broiler/turkey litter (60% dry matter). The phosphorus content of slurries ranges from 0.5 kg P/m³ for dairy and beef cattle slurry to 1.3 kg P/m³ for pig slurry (MAFF, 1994, p.8). Application rates may be up to 42 tonnes manure/ha/year or up to 109 m³ slurry/ha/year (based on the MAFF recommendation that the amount of total nitrogen in organic manures spread on land should not exceed 250 kg N/ha/year (MAFF, 1994, p.11)). This is equivalent to a phosphorus input of up to 63 kg P/ha/year from manure and 142 kg P/ha/year from slurry.

**Synthetic fertilisers**

Typical annual rates of application of phosphorus are 0-52 kg P/ha/year for cereals, 0-44 kg P/ha/year for oilseed rape, 0-153 kg P/ha/year for potatoes, and 0-66 kg P/ha/year for grass\(^4\) (MAFF, 1994).

**Mineral weathering**

Weathering of mineral rocks adds 0.1-0.5 kg P/ha/year (Brady and Weil, 1996, p.522).

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\(^4\) The highest value for four cuts of silage where phosphorus is applied to the first two cuts.
Atmospheric deposition

Deposition of phosphorus adsorbed on dust particles is in the range 0.05-0.5 kg P/ha/year (Brady and Weil, 1996, p.456). Rainfall deposition of phosphorus is insignificant (Parkinson, 1995, p.106).

Harvested crop

Examples of phosphorus removal in harvested crops are: 22 kg P/ha for cereals (including straw), 20 kg P/ha for sugar beet, 22 kg P/ha for potatoes, and 17 kg P/ha for grass silage (Parkinson, 1995, p.101).

Leaching

Brady and Weil (1996, p.456) state that annual loss of phosphorus by leaching is in the range 0.01-3.0 kg P/ha/year.

Soil erosion

Since eroded soil typically contains three times more nutrients than soil left in the field (Pimentel et al., 1995), the mass of phosphorus in eroded soil can be calculated as 0.3-2.7 kg (average 1.5 kg) P/tonne eroded topsoil.

Gaseous emissions

There are no gaseous emissions of phosphorus.

4. Modelling Potassium Flows In the Soil

Availability of Potassium

Pimentel et al. (1995) state that a tonne of fertile agricultural topsoil typically contains 2-30 kg potassium, while Parkinson (1995, p.104-6) gives a value of >4.39 kg potassium (assuming that value is quoted for the topsoil). Brady and Weil (1996, p.21, 403, 447, 478) give values of 15.9-25.0 kg potassium (average 17.4 kg) per tonne soil.
Potassium is found in the soil as an integral part of minerals, on soil colloids, and in soil solution. It is only available to plants as potassium ions in soil solution and on the soil colloids (as “exchangeable” potassium). Potassium in these forms amounts to just 1 to 2% of the total potassium in the soil (Brady and Weil, 1996, p.477, 481). Plants take up the potassium ions most easily from soil solution, but some plants can take up ions on soil colloids and even in less available forms. However, there is continual movement of the potassium ions between these different forms, although it occurs relatively slowly between the nonexchangeable and available forms. Nevertheless, on soils with substantial “reserves” of potassium, it is often necessary to apply only maintenance quantities of potassium because the crop can draw on these reserves.

**Organic matter**

The potassium content of manures ranges from 4 kg K/tonne for pig manure (25% dry matter) to 15 kg K/tonne for broiler/turkey litter (60% dry matter). The potassium content of slurries ranges from 2.2 kg K/m³ for beef cattle to 2.9 kg K/m³ for dairy cattle (MAFF, 1994, p.8). Application rates may be up to 42 tonnes manure/ha/year or up to 109 m³ slurry/ha/year (based on the MAFF recommendation that the amount of total nitrogen in organic manures spread on land should not exceed 250 kg N/ha/year (MAFF, 1994, p.11)). This is equivalent to a potassium input of up to 630 kg K/ha/year from manure and 316 kg K/ha/year from slurry.

**Synthetic fertilisers**

Typical application rates of potassium are 0-125 kg K/ha/year for cereals, 0-83 kg K/ha/year for oilseed rape, 0-291 kg M/ha/year for potatoes, and 0-349 kg K/ha/year for grass (MAFF, 1994).

**Mineral weathering**

Weathering of mineral rocks adds 5-20 kg K/ha/year potassium (Brady and Weil, 1996, p.522).

**Atmospheric deposition**

Rainfall adds 1-10 kg K/ha/year potassium to the soil (Parkinson, 1995, p.106).

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5 Soil colloids are the very small particles in soil with (usually) negatively charged surfaces that attract and adsorb cations. They are called micelles (Brady and Weil, 1996, p.241-2).
**Harvested crop**

Examples of potassium removal in harvested crops are: 58 kg K/ha for cereals (including straw), 199 kg K/ha for sugar beet and potatoes, and 133 kg K/ha for grass silage (Parkinson, 1995, p.101).

**Leaching**

Brady and Weil (1996, p.478) state that potassium leached from agricultural land receiving moderate rates of fertiliser is 25-50 kg K/ha/year.

**Soil erosion**

Since eroded soil typically contains three times more nutrients than soil left in the field (Pimentel *et al.*, 1995), the mass of potassium in eroded soil can be calculated as 48-75 kg (average 52 kg) K/tonne eroded topsoil (derived from data in Brady and Weil, 1996, p.21, 478).

**Gaseous emissions**

There are no gaseous emissions of potassium.

5. Availability of Nutrients Added To Soil

In agricultural systems, nutrients can only be used by plants when they are in available forms. As discussed above, available forms of nitrogen are NH$_4^+$ and NO$_3^-$; the most common available forms of phosphorus are HPO$_4^{2-}$ and H$_2$PO$_4^-$; and the available form of potassium is K$^+$.

**Nitrogen** added to the soil in various forms is either taken up by the crop, becomes incorporated into organic matter, or is lost from the soil through leaching, soil erosion or as gaseous emissions (as discussed above). Over a period of time, organic matter containing this nitrogen is broken down and the nitrogen released in available forms for the use of agricultural crops. Therefore, it is reasonable to assume that all the nitrogen applied in one year will be taken up eventually by a subsequent crop or lost from the soil.

---

*The highest value for four cuts of silage where potassium is applied for each cut.*
The same assumption cannot necessarily be made about phosphorus and potassium. As discussed above, much of the phosphorus added to the soil is relatively quickly converted into insoluble forms that, over time, are changed into more and more insoluble forms. However, once the phosphorus-fixing capacity of the soil has been reached, phosphorus needs only to be added at the rate at which it is removed at harvest (Brady and Weil, 1996, p.447). In a similar way, on soils that have built up substantial reserves of potassium, only maintenance applications are necessary to avoid depletion of the potassium reserve in the soil. Therefore, it can be assumed that any phosphorus and potassium added to these soils, and not used by the crop under analysis, will be available for future crops (and leaching and loss in eroded soil). This is likely to be the situation for agricultural production in much of Europe, since soils with relatively high levels of phosphorus and potassium tend to be found where the agricultural land has received regular inputs of fertilisers over a number of years.

References

References are listed in the References section of Chapter VI.
As outlined in Chapter II, there is a recognised hierarchy of methods for dealing with allocation issues in LCA. In this Appendix, I describe how the hierarchy can be applied in this study for co-production of wheat grain and straw (Section 1), and use of manure (Section 2).

1. Co-Production of Wheat Grain and Straw

The following sections describe how the different allocation approaches can be used in assessing the burdens of grain production from the wheat production system.

1.1 Avoiding Allocation By System Extension

This method relies on extending the system boundary to account for the fate of the straw, and subtracting the burdens arising from an alternative way of delivering the same service as provided by the straw.

An important question concerns the chosen fate of the straw: this may vary from use as bedding for livestock, to incorporation into animal feed, to incineration for energy recovery. For example, it may be assumed that the straw is burned to generate electricity for distribution via the UK national electricity grid, and the electricity displaces marginal base-load capacity which is provided by gas-fired plant. The “avoided burdens” for straw incineration are then those associated with generating an equivalent quantity of electricity by gas-fired plant. This gives the total burdens associated with grain production as:

\[
\text{Total burdens for wheat production to harvest} + \text{burdens for incineration of straw to produce electricity} - \text{Avoided burdens for equivalent electricity generation using gas-fired plant.}
\]

This is shown graphically in Figure 1.
Comments on methodology: realistic results for current systems depend upon accurate choice of extended systems (assumed to be straw incineration in this case) and displaced systems (gas-fired generating capacity in this case).

Figure 1. System Extension for Co-Production of Grain and Straw

1.2 Allocation On Basis of Physical Causality (Marginal Allocation)

This approach involves measuring the effects on the burdens of changing the output of one co-product (P) by a small amount (an incremental change) while holding the other co-product (Q) constant. Any changes in the burdens of the system are allocated to co-product P in proportion to the amount they change in relation to this incremental increase or decrease of P. In other words, if co-product P is increased by 1 kg while co-product Q is held constant and this leads to an increase of 2 kg CO₂ emissions, then the CO₂ emissions associated with co-product P are 2 kg CO₂ per 1 kg of co-product P. This process is then repeated for the other co-product, in other words co-product Q is increased by 1 kg while co-product P is held constant, and the resulting changes in burdens are allocated to co-product Q.

The marginal approach is only valid for systems that are unconstrained and homogeneous, and can be linearised within the range of variation relevant for the system under analysis. These conditions are met for this case study, although the range of variation has not been stated explicitly. Therefore, it is possible to use the physical causality approach. The only readily-apparent method of changing the

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7 For example, nitrate emissions may increase approximately linearly with increased applications of nitrogen fertiliser up to a certain quantity per hectare but will then increase exponentially. Therefore, the range of
grain: straw ratio in this system involves incorporating the straw into the soil\textsuperscript{8}, and this involves additional processes (straw chopping and deep ploughing). Burdens arising from these additional processes must be allowed for in the system model used to describe the marginal changes. Therefore, the analysis becomes a comparison between the current system and two different states of this system as shown in Figure 2. Differences in the burdens associated with a change from state X to state Y in Figure 2 may arise due to changes in:

- Use of machinery: straw chopping and deep ploughing to incorporate the straw
- Use of fertiliser: incorporation of straw releases nutrients in the longer term that substitute for fertilisers
- The soil's quality: addition of organic matter may have a number of effects, including effects on productivity (as discussed in Chapter V, Section 4.3).
- Yields of subsequent crops: some research suggests that straw incorporation may decrease the yields of subsequent crops.

Figure 2. Graph To Show Physical Causality for Wheat Production System

Differences in the burdens associated with a change from state X to state Z in Figure 2 may arise due to changes in the factors listed above plus an increase in all the existing burdens for wheat production in the system in state X.

\textsuperscript{8} Use of the physical causality approach depends upon the assumption that a change in the system to a different state does not lead to generation of new co-products. Therefore, it would not be possible to change the grain: straw ratio by incinerating the straw for energy recovery because this would produce energy as a co-product from the system. It would be possible to change the grain: straw ratio by burning the straw without energy recovery. But in this latter case, the straw would be a waste product rather than a co-product and so the issue of allocation among co-products would not be valid.
Therefore, using this method, the total burdens associated with grain production are:

- Total burdens for wheat production to harvest
- Plus burdens for straw chopping and deep ploughing to incorporate straw
- Plus burdens from addition of organic matter
- Minus avoided burdens from decreased use of fertiliser
- Plus burdens for additional production of subsequent crops.

**Comments on methodology:** results depend upon the chosen “states” for comparison in the system under analysis. In this case study, variation in the states due to straw incorporation seems to be the only possible way of varying the grain: straw ratio.

### 1.3 Allocation On Basis of Physical Parameters

Some options for allocation on this basis are given in Table 1. Again, it is necessary to make an assumption about the subsequent fate of the straw in order to define an appropriate physical parameter. However, even if the fate of the straw is known it is difficult to apply the method in this case because the grain and straw often have different purposes in the socio-economic system. In this study, the grain is used for bread production and so its protein content is relevant. The straw may be used as animal bedding, in animal feed or burned for energy generation. Relevant parameters for animal bedding are not obvious. Relevant parameters for animal feed may be digestible protein content (DCP) or metabolisable energy (ME). A relevant parameter for energy generation may be heat content. Therefore, a choice of one parameter as the basis for allocation between grain and straw is not obvious.

**Comments on methodology:** realistic results depend upon choice of an appropriate physical parameter. The choice depends upon assumptions about subsequent uses of the co-products, and may not be possible if the co-products have different uses.

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9 One could imagine another case study in which both the grain and straw are used in animal feed: in this case, the DCP or ME may be relevant parameters to use for both co-products.
1.4 Allocation On Basis of Economic Relationships

For allocation on the basis of economic relationships, the value of each co-product must be assessed prior to any further processing. In this case, it is the value of grain prior to drying and the value of straw prior to baling; the prices are taken as 600 ecu per tonne breadmaking grain and 24 ecu per tonne straw (Audsley et al., 1997, p.20).

Comments on methodology: results depend upon the prices of breadmaking grain and straw which may vary from year to year. Thus, although the results reflect short-term incentives for production, they are not useful for longer-term strategic decision-making where prices may be quite different (and especially in the agricultural sector with changes to the EU’s Common Agricultural Policy).

2. Use of Manure

The following sections describe how the different allocation approaches can be used in assessing the burdens of use of manure in System C.

2.1 Avoiding Allocation By System Extension: the “Avoided Burdens” Approach

In this case, application of system extension is facilitated by use of the Foreground-Background concept (as introduced in Chapter II, Section 3). The Foreground System (FS) comprises System C, and manure is taken from the Background System (BS) into the FS for use in System C. This is illustrated in Figure 3.
In order to use this method, an assumption must be made about the fate of the manure in the BS if it is not used in the FS. If the manure would be a waste product in the BS, the burdens associated with its use in the FS are those arising from:

- Processing and transportation for use in the FS
- \textit{Plus} use in the FS
- \textit{Minus} the Avoided Burdens from waste management in the BS.

Alternatively, if the manure would be used as a fertiliser in the BS, a further assumption is required before calculating the most appropriate burdens. This concerns whether the farm in System C can use manure from a non-organic farm or whether it is restricted to use of manure from its own and other organic farms. If the farm can use manure from a non-organic farm, then the burdens associated with use of manure in System C are those arising from:

- Processing and transportation for use in the FS
- \textit{Plus} use in the FS
- \textit{Minus} the Avoided Burdens associated with use of the manure in the BS
- \textit{Plus} additional burdens associated with production and use of equivalent quantities of nutrients in chemical fertilisers in the BS.
Although this sounds complex, in fact in most cases it can be assumed that use of manure in the FS and BS is associated with the same burdens. Therefore the total burdens associated with use of manure in the FS are those arising from:

\[ \text{Processing and transportation for use in the FS} \]
\[ \text{Plus additional burdens associated with production and use of equivalent quantities of nutrients in chemical fertilisers in the BS.} \]

Alternatively, if the farm must use manure from another organic source (and this is the case in Switzerland), a different scenario must be used to account for use of manure in the FS. In this case, use of manure in the FS will result in a need for more organic manure in the BS. However, this cannot be provided by producing more livestock because they would require additional organic manure for their production - hence, the allocation problem would not be solved. Instead, alternative sources must be found for the nutrients provided by the manure. Here, I focus on the N content of the manure on the assumption that it is the main valuable part of the manure in an organic system. A possible source of N is “green manure”; for example, grass-clover crops that fix atmospheric nitrogen. In this case, the burdens associated with use of manure in the FS are those arising from:

\[ \text{Processing and transportation for use in the FS} \]
\[ \text{Plus use in the FS} \]
\[ \text{Minus the Avoided Burdens associated with use of the manure in the BS} \]
\[ \text{Plus additional burdens associated with production and use of an equivalent quantity of nitrogen in an N-fixing crop in the BS.} \]

Again, assuming that use of manure in the FS and BS results in the same burdens, those associated with its use in the FS are those arising from:

\[ \text{Processing and transportation for use in the FS} \]
\[ \text{Plus additional burdens associated with production and use of an equivalent quantity of N in an N-fixing crop in the BS.} \]

Comments on methodology: realistic results for current systems depend upon accurate choice of extended systems, as illustrated above in deciding upon a source of nutrients in the BS to replace the manure used in the FS.
2.2 Allocation On Basis of Physical Causality (Marginal Allocation)

For allocation on the basis of physical causality, again it is necessary to focus on the N content of the manure. At high levels of N-content in the livestock fodder, a marginal increase in the N-content of the fodder may lead to a marginal increase in the N-content of the manure without increasing the output of other livestock products (assuming there is no other change in the feed ration). Following this logic, the burdens associated with the N-content of the manure can be regarded as those associated with production of an equivalent quantity of N in the livestock fodder.

Comments on the methodology: see comments in Section 1.2.

2.3 Allocation On Basis of Physical Parameters

For allocation by composition, a common component of livestock products must be identified in order to allocate the burdens of production among the different co-products (one of which is manure). One might assume that the common component of livestock products is their N-content (since the value of products such as meat and milk is related to their protein content, and the N-content of these products is directly proportional to their protein content). Using this method, the burdens of rearing livestock are allocated among the different co-products on the basis of their N-content. It should be noted that in order to operationalise this approach, it must be assumed that the upstream source of N is an N-fixing crop (otherwise the approach leads to the same allocation problem, i.e. the upstream source of N for the system). However, the approach is also likely to raise further allocation problems in calculating burdens associated with rearing livestock. This is because many feedstuffs for livestock are the co-products of other food production systems (oilseed cake and meal, sugar beet pulp, molasses, etc.). Overall, then, this approach seems unsatisfactory since it is likely to raise more allocation problems than it solves, it is questionable whether the N-content of other co-products is really representative of their value, and also it cannot account for co-products such as hide that do not share the “common component.”

Comments on methodology: see comments in Section 1.3.

2.4 Allocation On Basis of Economic Relationships

Using allocation by economic value, the burdens of rearing livestock are allocated among the different co-products on the basis of their economic values. As with allocation by composition, this
method is likely to raise more allocation problems than it solves due to the need to calculate burdens associated with rearing livestock. Therefore, it seems an unsatisfactory approach.

Comments on methodology: see comments in Section 1.4.

References

References are listed in the References section of Chapter VI.
# Appendix VI.5

## Ecotoxicity and Human Toxicity Factors for Pesticides

<table>
<thead>
<tr>
<th>Type of Substance</th>
<th>Substance</th>
<th>Aquatic Ecotoxicity Factor</th>
<th>Terrestrial Ecotoxicity Factor</th>
<th>Human Toxicity Factor: Air</th>
<th>Human Toxicity Factor: Water</th>
<th>Human Toxicity Factor: Soil to Food</th>
<th>Human Toxicity Factor for Applied Pesticides</th>
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Source: Audsley et al., 1997, p.79,83.

* For missing data for particular pesticides, the average values have been used for that group of pesticides. Although this is a simplification, it is used on the basis that a first approximation is better than no assessment.

b Pirimicarb values used for methiocarb where factors are missing.

c Other than the data omissions shown here, no Toxicity factors were available for molybdenum, selenium, thallium or vanadium.

**References**

References are listed in the References section of Chapter VI.
APPENDIX VI.6
CONTRIBUTION OF DIFFERENT SUBSTANCES TO IMPACT ASSESSMENT CATEGORIES

The table below lists the 26 substances that make a contribution of 10% or more to at least one of the IA categories in one or more of the systems.

Table 1. Different Substances Contributing To Impact Assessment Categories (>10% Contribution)

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<th>Type of Emission</th>
<th>Substance</th>
<th>IA Category</th>
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<td>CO₂</td>
<td>Global warming</td>
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<td>NH₃</td>
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<td>Eutrophication</td>
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</tr>
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