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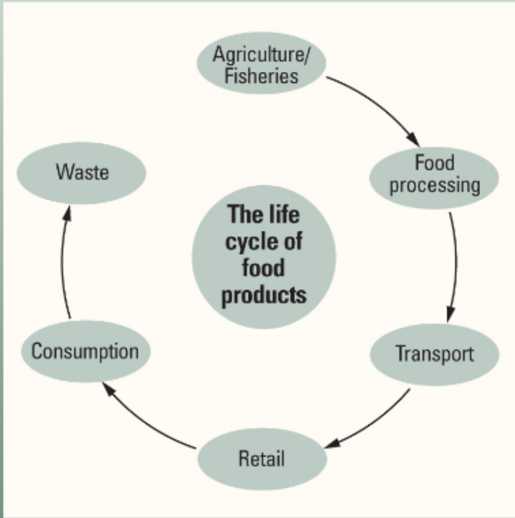

Environmental assessment and management in the food industry

Life Cycle Assessment and related approaches

Edited by U. Sonesson, J. Berlin and F. Ziegler

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Environmental assessment and management in the food industry

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Preface

Food production accounts for a significant share of the total impact of several important environmental categories, such as climate change, eutrophication and loss of biodiversity. Food production systems are also often complex and involve biological systems, meaning that they are difficult to control and measure. This means that assessing the environmental impact of food products and production systems is therefore a difficult task. Assessing the sustainability of food systems is a young and emerging research field that is facing rapidly growing attention, with emergent problems associated with climate change, biodiversity loss and water issues on one hand and food security on the other. This book gives an overview of the environmental impacts caused by food production, as well as detailed insights in methodologies and tools for assessment and improvements. It provides detailed insights, and should be useful for environmental managers in the food sector, policy makers and graduate- and post-graduate students in agriculture and food science. The first part shows how food production systems affect the environment, including the pressing issue of water use, an aspect where food production is the most important human activity (Chapter 2). The next part covers Life Cycle Assessment (LCA) methodology. After a general introduction (Chapter 3), details on LCA methodology are given, such as how methodological choices are made and their data requirements (Chapters 4 and 5). Thereafter, various product categories (e.g. animal products, seafood) are discussed (Chapters 6–8), where the reader will gain in-depth knowledge about important aspects specific to the product groups. The third part focuses mainly on how LCA and related approaches can be applied for different purposes. It covers production development (Chapter 9), land use and ecotoxicity (Chapter 10), as well as how LCA can be used together with economic tools to improve environmental and economic performance simultaneously (Chapter 11). Other areas covered are social aspects in LCA (Chapter 12) and Ecodesign (Chapter 13). The use of LCA and related tools for communication through 'footprints' is also elaborated on, both generally (Chapter 14) and specifically for carbon footprint (Chapter 15), and from a broader sustainability perspective (Chapter 16). The fourth and final part of the book covers more general issues such as how companies can implement and make use of environmental management systems (Chapter 17), environmental training within industry (Chapter 18) and eco-labelling (Chapter 19). It is our hope that this book will constitute a useful contribution to the process of transforming present food production systems into more sustainable ones.

Ulf Sonesson, Johanna Berlin and Friederike Ziegler

Part I

Environmental impacts of food production and processing

1

Improving nutrient management in agriculture to reduce eutrophication, acidification and climate change

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Abstract: The present organisation of our food production leads to significant alterations of the global nitrogen cycle and this is an important cause of increasing emissions of reactive nitrogen into ecosystems and the atmosphere. In this chapter, important environmental themes related to the mismanagement of the nitrogen cycle – eutrophication, acidification and climate change – are discussed. The prospect of a doubled consumption of animal products in 2050 globally means a giant challenge for stakeholders in the food chain to improve production and to find new innovative ideas in order to substantially reduce nutrient losses from the food chain.

Key words: reactive nitrogen, surplus phosphorous, livestock production, animal products.

1.1 Introduction

Eutrophication, acidification and climate change are impacts ranked high on the policy agenda, and today's food production has a profound influence on these environmental themes. Human alteration of the global nitrogen cycle, which is very much associated with food production, is an important cause for emissions of reactive nitrogen into ecosystems and the atmosphere. Through land clearing, production and use of fertilisers, increasing animal production accompanied by increasing manure production, and discharges of human waste, nitrogen has been mobilised at an unprecedented rate in the 20th century. The role of food production is unquestionable in this context and the task of feeding an increasing world population in the 21st century with less disturbance of the global nitrogen cycle is one of the greatest challenges for all stakeholders in the food chain.

Nitrogen is essential for everything that grows; one hundred years ago, before the introduction of the Haber-Bosch method to synthesise nitrous gas into ammonia, the nitrogen problem was regarded as a shortage problem in food production. Now, at the beginning of the 21st century, the problem is the opposite. There is overwhelming evidence of several serious environmental consequences due to excess nitrogen from human activities. This chapter describes the environmental impact of eutrophication, acidification and climate change, with a focus on the nitrogen issue which is a common problem almost exclusively linked to the food chain.

1.2 Eutrophication and acidification

1.2.1 Aquatic eutrophication

Aquatic eutrophication can be defined as nutrient enrichment of the aquatic environment. Excess input of nutrients increases the primary production of fast-growing algae such as phytoplankton, and as this algae biomass grows, the water becomes turbid. Slow-growing vascular plants (e.g. eelgrass) that are best adapted to low-nutrient environments decrease and the fish community shifts due to habitat changes (less light, changes in plant species). In tropical waters, nutrient enrichment stimulating production of macro-algae can lead to overgrowth and replacement of corals (Cloern, 2001). When driven to a far extent, nutrient enrichment of coastal stratified waters* can cause anaerobic conditions or low-oxygen conditions and result in significant bottom fauna mortality and losses of fish resources.

Generally, fresh waters (lakes, reservoirs, rivers) in temperate regions are phosphorus (P) limited, whereas nitrogen (N) is considered to be the primary limiting element in marine systems. This is however an oversimplification; it was early suggested that both N and P are important nutrients in estuaries and this has been confirmed with observations of P limitation during spring and N limitation during summer in coastal-near environments, such as the Gulf of Riga (Latvia), Roskilde Fjord (Denmark), Bay of Brest (France) and Delaware Bay (USA). Also, present understanding of the relative importance of N and P is strongly biased by the predominance of studies at temperate latitudes, since tropical marine systems seem to be more frequently P-limited (Cloern, 2001).

Increased human disturbance of the nitrogen and phosphorus cycle during the 20th century is the main cause for the eutrophication problem. Nitrogen fluxes in rivers in Europe and the US have increased significantly; movements of total dissolved N into most temperate-zone rivers in the North Atlantic Basin may have increased by as much as two to twenty-fold since pre-industrial times (Howarth *et al.*, 1996). The highest N increases have been found in

* Waters having sharp temperature gradients that prevent mixing of warm surface waters with cold bottom waters.

rivers in the North Sea region. Phosphorus loading to estuarine systems has increased two- to six-fold since 1900. When examined as a whole, existing nutrient records show a rapid change in the fertility of coastal ecosystems over the last half of the 20th century (Cloern, 2001).

Emissions of nitrogen to water from agriculture occur predominantly as nitrate leaching from the soils, but in severe cases it can also be in the form of discharged effluents from manure waste storages. The magnitude of soil leaching is determined by farming systems, type of soils and climate. High livestock density and crop rotations dominated by annual crops with high fertilising intensity and short growing season (e.g. potatoes) are examples of farming systems that are characterised by relatively high nitrate leaching, as opposed to crop rotations with a high degree of perennial crops, such as grasslands. Climate conditions with rainy and mild winters increase risks for leaching, and lighter sandy soils generally have higher nitrate leaching as compared to heavier clay soils.

During the transport of leached N in rivers, there are transformation processes leading to some of the emitted nitrate being removed by plant uptake and denitrification; these processes are referred to as 'nitrogen retention'. In Sweden, current average retention of nitrogen lost from arable land has been estimated at approximately 40% (Arheimer *et al.*, 1997). In other words, 40% of the leached N from arable land is disarmed in rivers and lakes mainly through the denitrification process transforming the nitrate into environmentally harmless nitrogen (N_2), and 60% reaches surrounding seas and water via river mouths as reactive N and is thereby potentially environmentally harmful. During the second half of the 19th century and first half of 20th century, lakes and wetlands were extensively drained to gain more arable land in many European countries and this has affected the net losses of N to surrounding seas since the landscape has lost some of its capacity to neutralize the emitted reactive nitrate and transform it into inert N_2. The rise in N concentration in rivers has often been connected to the increased input of fertiliser N. Studies show that there are also other mechanisms, notably draining of lakes and wetlands, which can be as important as the input of fertiliser N in affecting net losses to the surrounding seas from arable land (Hoffman *et al.*, 2000).

Since the 1980s there has been an increasingly efficient removal of P by sewage wastewater systems in the developed world and this has resulted in agriculture's contribution of diffuse P losses to aquatic environments becoming relatively more important to the eutrophication problem. Phosphorus is lost from arable land by soil erosion, surface runoff and leaching. One problem of today is that many agricultural soils have accumulated phosphorus in excess. In the first half of the 20th century it was seen as economically justifiable to add extra phosphorous when applying fertilisers in order to increase the soil P-content. But the adding of more phosphorus than crops remove and, moreover, application of farmyard manure that is often rich in P, has resulted in high accumulations of phosphorus in many soils. In Denmark, P-accumulation was 1400 kg ha^{-1} during the 20th century (Damgaard Poulsen and Holten Rubaek, 2005); in Sweden, soil-P accumulation between from ~1950 to 2000 was around 700 kg ha^{-1} as a nation average, and in regions with high animal density around 1000 kg ha^{-1} (Andersson *et al.*, 1998). The structural movement towards geographic concentration of livestock production has affected P accumulation in soils in regions with high animal density and this is bound to increase the risk of phosphorus losses to aquatic ecosystems.

1.2.2 Terrestrial eutrophication

Terrestrial eutrophication includes the effects of excess nutrients on plant functioning and species composition in natural or semi-natural terrestrial ecosystems. Under uninfluenced conditions, vegetation in natural ecosystems is mainly controlled by the limited availability of nitrogen. Atmospheric N deposition caused by human activities leads to increased loads of nitrogen and, from this, follows changes in structures and functions in N-limited ecosystems. For example, there is an increased competition from nitrogen adapted species at the expense of less adapted species and an altered tolerance towards diseases, drought, frost, etc. Based on measurements of precipitation in remote areas, annual wet deposition of inorganic N in unpolluted regions is estimated to be in the range 0.1 – 0.7 kg N ha^{-1} (Vitousek *et al.*, 1997). These background figures are less than 10% of the rates of wet deposition in the mid-western and eastern United States, and less than 1% of the rates in the most heavily affected areas of northern Europe.

Ammonia emissions largely derive from animal production, principally from livestock manure during housing, storing and spreading on the land. Volatilisation of ammonia from manure to the air represents a significant nitrogen loss from agriculture; of the nitrogen excreted by the livestock, as much as 20–40% can be lost as ammonia, depending on farming systems, feeding routines, application methods, etc. Application of synthetic N-fertilisers also induces ammonia losses, especially in the case when nitrogen is applied as urea. Model calculations indicate that ammonia in northern Europe is largely removed from the atmosphere by dry deposition at distances less than 1 km from the source or by wet deposition of ammonium at distances even larger than 1000 km from the source (Asman, 2001). Emissions of ammonia can therefore cause environmental damage both at local and regional level, but the most important influence of deposited ammonia N is the impact that this reactive nitrogen has on the nitrogen cycling in natural ecosystems and subsequent changes in ecological balances.

1.2.3 Acidification

The consequences of acidification (e.g. leaching of toxic aluminium, reduced forest and plant health, loss of aquatic life) have been observed in surface-and ground-waters, soil and vegetation during several decades. The major acidifying substances are oxides of nitrogen (NO_x) and of sulphur (SO_2), and ammonia, NH_3. Intense agricultural production leads to soil acidification due to plant growth and nutrient uptake in exchange of H^+. However, the acidifying pollutant of highest significance in food production (especially in animal production) is ammonia (NH_3), which is not acid in a chemical sense, but has a strong acidifying effect as a result of nitrification in the soil involving the conversion of ammonium into nitrate by micro-organisms: $NH_4^+ + 2\ O_2 \rightarrow NO_3^- + H_2O + 2\ H^+$. Depending on the state of the ecosystem where the ammonia is deposited, the acidifying impact varies. Up to a certain level, a forest can absorb deposited nitrogen, but above that level, excess nitrogen is leached. The forest soil is said to be nitrogen-saturated. In forests that are saturated with nitrogen, nitrification and leaching of base cations and nitrate are usually the most important mechanisms behind soil acidification. There is a close interaction between terrestrial eutrophication and acidification.

1.3 Climate change

The average global temperature has increased by 0.74 °C during the past 100 years (1906–2005) and during this period, the average Arctic temperature has increased by almost twice the global average. This global warming has led to a number of observed changes; for example, mountain glaciers and snow cover have declined on average in both hemispheres, global average sea levels have risen at an average rate of 1.8 mm per year over 1961 to 2003, long-term trends from 1900 to 2005 have been observed in precipitation amount over many large regions, more intense and longer droughts have been observed over wider areas since the 1970s, particularly in the tropics and subtropics, and widespread changes in extreme temperatures have been observed over the last 50 years. Cold days, cold nights and frost have become less frequent, while hot days, hot nights, and heat waves have become more frequent (IPCC, 2007).

Unlike for industrial and transport systems, carbon dioxide (CO_2) from fossil fuel use is the least important greenhouse gas (GHG) emitted from the food sector. Instead, it is biogenic emissions of methane (CH_4) and nitrous oxide (N_2O) that contribute mostly to food's carbon footprint*. Also, land-use related emissions of CO_2 can be an important source for emissions in food production's life cycle. Emissions of CH_4 and N_2O in agriculture contributed 10–12% of world total emissions in 2005, according to IPCC (Barker *et al.*,

* Carbon footprint is a term used, e.g. by British Standard and in ISO-working documents, to describe the amount of GHG emissions of a process or a product system to indicate their contribution to climate change. It also includes emissions of nitrous oxide, which are of special importance for agricultural products.

2007), while FAO has estimated that global livestock production make up 18% of total GHG emissions when land-use related CO_2-emissions also are included (Steinfeld *et al.*, 2006).

The atmospheric concentration of CH_4 has increased from ~715 ppb* pre-industrial to 1774 ppb in 2005, i.e. by close to 150%. The corresponding change of atmospheric N_2O is a rise from ~270 ppb to 319 ppb in 2005, an increase of around 18% since 1750. Present-day radiative forcing† from long-lived greenhouse gases is estimated at 2.63 W m^{-2} distributed as 1.66 W m^{-2} from CO_2, 0.48 W m^{-2} from CH_4, 0.33 W m^{-2} from halocarbons and 0.16 W m^{-2} from N_2O (Forster *et al.*, 2007). The relative contribution of CO_2, CH_4 and N_2O today to current radiative forcing is thus about 63, 18 and 6%, respectively.

1.3.1 Nitrous oxide

Anthropogen emissions of nitrous oxide are closely connected to man's interference with the nitrogen cycle, which has been emphasised in the second half of the 20th century as a result of the very strong increase of synthetic N fertiliser use in agriculture. Current anthropogenic emissions of N_2O are estimated at 6.7 million tonnes N yr^{-1}, of which 2.8 (1.7–4.8) originates from agriculture, mainly from denitrification and nitrification processes in the soil and also nitrogen transformations in manure (Denman *et al.*, 2007). Around 60% of global N_2O emissions can be allocated to agriculture (Barker *et al.*, 2007). Moreover, nitrous oxide is released in industrial N fertiliser production and this must be added as a part of the food chain's total emissions. Direct nitrous oxide emissions from agricultural soils are difficult to reduce significantly since N_2O production in soils is an unavoidable by-product from nitrogen transformations, implying that high nitrogen turnover per area triggers production of N_2O in the soil (to what extent depending on several factors, e.g. soil moisture and temperature). It is difficult to measure and monitor N_2O emissions from agricultural soils and therefore it is hard to foresee the reduction potential for mitigation actions. Present models used for calculating N_2O emissions from soil systems are simplified and static, and need to be developed. While uncertainties in estimates of fossil CO_2 are estimated to be low (2–4%) in developed countries emission accounts, corresponding numbers for N_2O are 30–230% (Rypdal and Winiwater, 2001).

Bearing in mind the insecurities in N_2O-estimates, it has been calculated that more efficient fertiliser application in crops could reduce N_2O emissions

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* ppb (part per billion) is the ratio of greenhouse gas molecules to the total number of molecules of dry air.

† Radiative forcing is a measure of the influence a factor has in altering the balance of incoming and outgoing energy in the Earth-atmosphere system, and is an index of the importance of the factor as a potential climate change mechanism. It is expressed in watts per square metre.

in soils by 25–30% (Beach *et al.*, 2008) while Smith *et al.* (2007) estimate the technical reduction potential of N_2O emissions from livestock at just around 5%. Reports of such low reduction potentials of N_2O emissions are discouraging when studying present and future trends in GHG emissions from agriculture. Globally, agricultural GHG emissions increased by 14% from 1990 to 2005, and N_2O increased by almost twice the rate seen for CH_4 emissions. According to FAO, agricultural N_2O emissions are forecast to increase by 35–60% up to 2030, due to increased N fertiliser use and increased animal manure production, and by about 50% by 2020 (relative to 1990) according to US-EPA (Smith *et al.*, 2007).

1.3.2 Methane and land-use related carbon dioxide

Important sources of CH_4 emissions from global food production are ruminants (enteric fermentation), rice cultivation and waste management, mostly slurry and landfills (see Table 1.1) (Denman *et al.*, 2007). There are large uncertainties in the global methane budget and different references give an estimate of yearly emissions in the range of 239 – 465 million tonnes CH_4 yr^{-1}, of which emissions from agriculture are responsible for approximately 50%. When also including methane emissions from food waste handling, around 60% of CH_4 emissions can be allocated to the food chain.

Emissions of fossil CO_2 are of minor importance in the food chain but CO_2 emissions from land-use change processes are closely connected to expanding food production. Today, more than 35% of the global land surface (corresponding to 4.6 – 5.1 billion hectares) are under cultivation or pasture as compared to only 790–920 million hectares (6–7% of total land) around 1750 (Forster *et al.*, 2007). The period 1850–1950 saw a rapid increase of agricultural area and carbon emissions from forest clearing, constituting about one-third of total anthropogenic CO_2 emissions in the period 1850–2005. In the last 50 years, there has been a stabilising or even decrease in cropland area in many regions but in the tropics, deforestation is still occurring rapidly. During the 1990s, it is estimated that tropical deforestation gave rise to CO_2 emissions in the order of 1.0–2.2 GtC yr^{-1}, comprising 14–25% of total anthropogenic carbon emission (Denman *et al.*, 2007).

Table 1.1 Overview of global anthropogen emissions of methane (Source: Denman *et al.*, 2007)

Anthropogenic source	Emission (Tg CH_4 yr^{-1})
Energy production (coal, gas)	82–104
Landfills and organic waste	35–69
Ruminants	76–92
Rice cultivation	31–112
Biomass burning	14–88
Total	238–465

1.4 Mismanagement of nutrients

1.4.1 Nitrogen

Without interference of humans, the nitrogen cycle is in equilibrium without serious accumulation of reactive nitrogen leading to negative impacts on ecosystems. During the 20th century, human activities have increasingly transformed inert N_2 in the atmosphere into reactive forms; human-driven conversion occurs primarily through four processes: industrial fixation of ammonia (80 million tonnes N (Mtonnes) yr^{-1}, biological fixation in leguminous crops (40 Mtonnes yr^{-1}), fossil fuel combustion (20 Mtonnes yr^{-1}) and biomass burning (10 Mtonnes yr^{-1}). This means that around 120 Mtonnes new reactive nitrogen (as synthetic N fertilizers and leguminous crops) goes into agriculture every year. If we compare this input with the human consumption of nitrogen in crops, dairy and meat products, corresponding to 17 Mtonnes N in 2005, we realize that the N-efficiency is very low (<20%) in today's food chain. Also, it has changed for the worst; the global nitrogen-use efficiency of cereals decreased from ~80% in 1960 to ~30% in 2000 (Erisman *et al*., 2008).

The increasing demand for food for a rapidly growing and more affluent world population is the major driving force behind the profound change of the nitrogen cycle. Since the 1960s, world population has doubled while the available calorie per head has increased by 25%. Worldwide, households now spend less income on their daily food than ever before, in the order of 10–15% in the OECD countries, as compared to 40% at the middle of the 20th century (Fresco, 2009). The flip side of the coin is the negative consequences this development has had for the global nitrogen cycle and, due to the complexity of nitrogen processes, this is poorly understood by many who work in the food chain. The emission of one nitrogen molecule can lead to a cascade of negative effects in the ecosystems. For example, emitted ammonia from manure can first give direct impact on the vegetation very close to the source, then contribute to acidification and/or eutrophication and/or pollution of surface water and/or coastal water on the regional scale, and finally contribute to the global problem of increasing greenhouse gases in the atmosphere through a transformation into N_2O.

Due to the large-scale anthropogenic pressure on the Earth system, a research group has recently suggested the need to establish nine planetary boundaries for estimating a safe operating space for humanity with respect to the functioning of the Earth system. They suggested that humanity has already transgressed three of these proposed planetary boundaries namely (i) climate change, (ii) biodiversity loss and (iii) changes to the global nitrogen cycle. It is striking to conclude that present food production has a profound impact on all these three crucial Earth system processes. When it comes to the disturbance of the nitrogen cycle, the research group propose that the simplest and most direct approach is to consider the human fixation of N_2 from the atmosphere as a giant valve that controls the massive flow of new reactive nitrogen into the Earth system, and they suggest that the boundary value initially to be set at approximately 25% of the current value, or to about 35 Mtonnes N yr^{-1} (Rockström *et al.*, 2009). Consequently, such a target means an enormous challenge for improving nitrogen efficiency in the entire food chain, from fertilisation and manure handling to the treatment of N in waste from food consumption.

So far, it has been common that abatement strategies have been focused on individual environmental issue, e.g. eutrophication. Due to the complex and interlinked characteristics of food production's nitrogen problem, it would be much more rational to work holistically with the management of nitrogen in the whole production chain, especially in the more complex life cycle of animal products where the nitrogen efficiency is generally low today, on a global level estimated slightly over 10% with great variation. For example, global N efficiency in pig production is estimated at around 20% (van der Hoek, 1998). Generally, pork and poultry production have higher N efficiency than ruminant production, but the ongoing development of monogastric animal production into landless or 'industrial' farming systems makes it very difficult to utilise the gained N efficiency in animal production in other parts of the production life cycle. According to FAO's definition, landless livestock production systems have less than 10% of the dry matter fed to animals produced at the farm. Instead, the feed is grown elsewhere and transported to the farms, sometimes as far away as from other continents. Today, around 50 and 70% of global pork and poultry meat, respectively, is produced in landless/industrial production systems. These systems are prevalent in areas with high population densities, in particular coastal areas in East Asia, Europe and North America, often connected to ocean ports for feed import (Steinfeld *et al.*, 2006). This concentrated production system leads to high manure production on limited areas, which makes it extremely difficult to recycle the nutrients efficiently in crop production. The geographical separation of the livestock production from fodder cultivation puts an increased demand on input of new nitrogen into the product life cycle, and the input is as synthetic N fertilisers into the feed cultivation. The manure is used badly, applied at too high rates due to a restricted area close to animal production or, in the worst case, deposited in the near environment. Although pigs and poultry have a relatively high feed efficiency and thereby relatively high N efficiency in production, the landless meat production system as a whole has low potentials to use the manure efficiently, which makes it necessary to continuously introduce new fertiliser N in crop cultivation. The concentration of manure in small areas, disables the benefits from a relatively high N-efficiency in monogastric meat production and this is a good example of the need for a holistic perspective when working with the nitrogen issue; its use, flows and emissions in the entire life cycle of meat, milk and eggs.

1.4.2 Phosphorus

Similarly to nitrogen, the phosphorus cycle is now also greatly influenced by human activities. Today the annual production of phosphate is around 53 million tonnes P_2O_5 (IFA, 2009), of which around 80% is used as fertilisers and another 5% in mineral feed (Steen, 1998). Production and consumption of food is thereby almost exclusively responsible for the phosphorus problem, not only leading to emissions of nutrifying pollutants but also to depletion of a non-renewable resource.

Phosphorus utilisation (i.e. uptake in meat, milk and eggs in relation to input with feed) in animal production varies between different livestock groups but is normally in the range of 15–40% (Damgaard Poulsen and Holton Rubaek, 2005). Consequently, the dominant part of phosphorous in the feed ends up in the manure, and therefore the ongoing trend on organising the world's pork and poultry production in landless production systems poses a long-term problem also for the phosphorous cycle. In areas close to concentrated livestock operations, there will be increasing soil P accumulation whereas the feed crops that are geographically separated for the animal farms must be fertilised with 'new' phosphorous from synthetic fertilisers. Present P distribution on global farmland must be improved and there are two important arguments for that today: the eutrophication problem and resource depletion. Such an improvement must start with a critical review of the effects of the present structure that is now developed in the world's livestock production system.

1.5 Future trends

At present, global animal production makes use of 70% of all agricultural land and 30% of the land surface of the planet. In several ways, this has a profound influence on the Earth's systems, perhaps most markedly on the global nitrogen cycle. Growing population and incomes, which shifts dietary preferences into more animal food, lead to an increasing demand for livestock products. Global meat consumption is projected to double from around 230 million tonnes in 2000 to 465 million tonnes in 2050, see Fig. 1.1. The lion's share of the growth is projected to take place in the developing countries, both due to increasing population and higher *per capita* intake; annual *per capita* consumption is projected to increase from 18 kg in 1990 to 37 kg in 2050. Despite this doubling, meat consumption *per capita* in the developing world will still be substantially lower than that in the developed countries, where it is projected to average at around 89 kg *per capita* in 2050 (Steinfeld *et al*., 2006).

Nearly two-thirds of meat production growth now takes place in Asia, especially China, where there has been a rapid increase in production and consumption of animal products, especially pork (Brighter Green, 2008). Large-scale livestock farms are increasingly important for Chinese pork

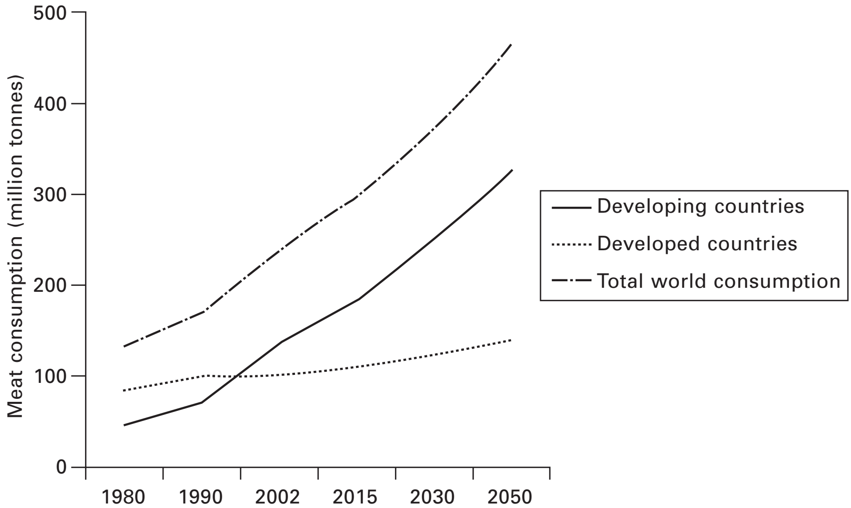

Fig. 1.1 Past and projected trends in consumption of meat (million tonnes) in developing and developed countries

production and are often located near cities near the coast. Manure is commonly discharged into the environment or stored in vast lagoons from which waste may spill or leak into nearby streams and groundwater supplies. The animal wastes are degrading seawater, threatening mangroves and coral reefs, and causing ‘red tides,’ or massive algae growth (FAO 2005). The development of industrial livestock farming systems without sensible nutrient use and cycling, has resulted in many environmental problems in the western world. It is urgent to prevent the same mistakes being repeated when animal production expands in the developing world.

Doubling global animal production to 2050 involves the task of cutting the environmental impact per unit of livestock production by 50% in order to main the level of damage at the present level. Already this is a very difficult mission, but it is still insufficient to reduce the environmental impact from food. For example, the suggested target of halting global warming at 2 °C implies that global GHG emissions should be reduced by 50% by 2050. From this, it follows that a doubled animal production would require that GHG emissions per product unit must be lowered by 75% in 2050. The giant magnitude of this challenge tells us that it will not only take existing solutions and new innovative ideas to improve food production, but also that consumers needs to be involved. After all, it is consumer demand that drives production, and an increasing awareness of the climate crisis and food consumption’s contribution to greenhouse gas emissions, along with growing concern about unhealthy diets, can change consumer preferences in the 21st century.

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Increasing the efficiency of water use in crop production

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Abstract: Agriculture is by far the largest user of water. Increasing the use efficiency of water is essential to sustainably provide food for humans and water for maintaining natural ecosystems. The production ecological approach presented in this chapter allows us to identify constraining factors in crop production that depress use efficiency of water and to determine intervention measures. Much of the additional water needed for world food production in 2050 can be obtained by improving agronomic practices, though expansion of agricultural land to capture rainwater will be inevitable.

Key words: production ecology, water for food and nature, agriculture, water scarcity, world food.

2.1 Introduction

Global dialogues about the looming water crisis have placed water scarcity problems high on the political and research agendas, because assessments of current water-related problems depict a depressing view and estimates of future demands for water suggest that billions of people will live under water-stressed conditions. Agriculture is by far the largest user of water, ranging from over 90% in various developing countries in the semi-arid regions, to some 50% in highly industrialized nations. On average, 70% of water withdrawal from natural systems is used for agriculture, 20% for industry and 10% for municipalities. Water withdrawn is not necessarily 'lost', but may be available for reuse, though generally of degraded quality. Any diversion of water from its natural course will affect ecosystems. Some rivers, for instance, do not even reach the sea anymore as their water, apart from evaporation and natural discharge to the subsurface, is completely withdrawn for human activities, creating ecological and environmental problems. Salt intrusion, water pollution, erosion, declining levels of groundwater and the drying up of lakes are phenomena that have become worse in recent decades due to the fierce competition for water. Water withdrawals remain necessary, however, for economic growth and food production in particular. It is essential, therefore, to make the most efficient use of water.

With agriculture being by far the largest user, largest gains are likely to come from this sector. The use efficiency in agriculture of water is relatively low, as a small fraction only of both irrigated water and rain water is ultimately used for transpiration, i.e. the actual physiological process in crop growth in which water is used for cooling the canopy that is heated by the incoming radiation. Large amounts of water could be saved as a means of resolving water scarcity problems, by raising the use-efficiency of water in agriculture in quantitative terms. However, gains in water use efficiency are not easy to achieve.

In this chapter, we first present and analyse the global balance of available fresh water for the production of food crops. Apart from food crops, most diets comprise animal products, which come from secondary production systems that rely on available plant products. We will simplify our analysis to plant production by integrating required plant production for animal products, such as meat. In this way we are able to assess current and future water use requirements. Then we introduce basic production ecological concepts that are essential for identifying realistic options for improving water productivity and concurrently water use efficiency. Subsequently, ways and means to enhance productivity will be elaborated, and finally, institutional arrangements to implement these options on various scales will be outlined.

2.2 Water scarcity: the global dimension

Estimates of future demands for water suggest that billions of people will live under water-stressed conditions. Many projections developed before 1980 showed near exponential increases in water requirement, but actual water withdrawals have been much lower, as has been analysed by Brown (2002). Projection studies made after 1980 have been adjusted to account for possible improvements in water productivity and forecast lower increases (Gleick, 2003). However, demands will still result in increased water withdrawal from natural systems. Current freshwater withdrawals from blue water sources approximate 4000 $km^3\ y^{-1}$, which is used for irrigation, industry and domestic purposes (Gleick, 2003), with 70% or an equivalent of about 2800 $km^3\ y^{-1}$ for food production. Oki and Kanae (2006) estimate 2660 $km^3\ y^{-1}$ of water to be withdrawn from fresh water sources for irrigation, whereas Shiklomanov (2000) estimates 1800 $km^3\ y^{-1}$. Rockström (2003) estimates 5000 $km^3\ y^{-1}$ of water is needed for rain-fed agriculture, and an additional 1800 $km^3\ y^{-1}$

for irrigation to meet food demand based on the current amount of caloric intake. This distinction in water supply for food production through irrigation or rainfall is essential for further considerations in search for improvement of water use efficiency.

The total amount of precipitation on terrestrial land equals 111 000 km^3 y^{-1} (Postel *et al.*, 1996), which is equivalent to an average of 8536 m^3 ha^{-1} or 854 mm and is in line with the 834 mm y^{-1} estimated by Rockström and colleagues (1999). The current arable land of 1402 million hectares therefore receives approximately 11 970 km^3 of rainfall, which is similar to the value of 11 600 km^3 y^{-1} stated by Oki and Kanae (2006).

We have developed an approach to estimate current and future water use in agriculture, based on food production statistics. Goudriaan and colleagues (2001) show cereal crops to account for 60% of global carbon fixation in agriculture, followed at a far distance by oil crops (including nuts) and sugar crops, with 9% each. Combined with a productivity rate, i.e. carbon fixation per area unit per year, which is at 87% of the global average fixation rate, cereals are a good representation of global food production. For assessing global food and water requirement and for calculation of global food production we therefore follow the grain-equivalent approach, which converts non-cereal food items into grain equivalents (WRR, 1995). Diets composed of various food items can then be converted to grain equivalents, which facilitates analyses of production and consumption of food.

Current annual global cereal production (2005) is 2239 million tonnes which, under the above assumptions, converts to a total food production of 3732 million tonnes per year. The global arable acreage derived from cereals can be estimated by correction for 60% (the proportion of food produced) and 87% (the relative carbon fixation rate relative to the global average) which reaches 685.6 million hectares (current area for cereals)/0.6/0.87 = 1313 million hectares. This estimated acreage is close to the current arable land of 1402 million hectares. For the 6.4 billion people on earth in 2005, a total amount of 1600 g cereals is available per day, equivalent to 584 kg y^{-1}. This amount converts to a diet in between a vegetarian and moderate diet (see WRR, 1995).

Rockström (2003) derived an empirical relation between water productivity and yield of cereals which reveals a natural logarithmic relation. The water requirements per tonne of cereals produced decreases drastically with yields increasing from 0.5 to some 2 t ha^{-1}, to gradually decline to stable equivalents of approximately 800 L per kg grain at yield levels exceeding 6–7 t ha^{-1}. By using this relationship, Rockström (2003; Fig. 2.1) estimates global water productivity for cereal crops at 1800 L kg^{-1} at a global average yield level of 2 t ha^{-1} for rainfed cereals.

Due to the non-linear nature of the relation, however, a distinction in yield classes is justified. Using the more-or-less socio-economically homogeneous regions as distinguished by the United Nations and further disaggregating Europe because of large differences in cereal yield between different agro-

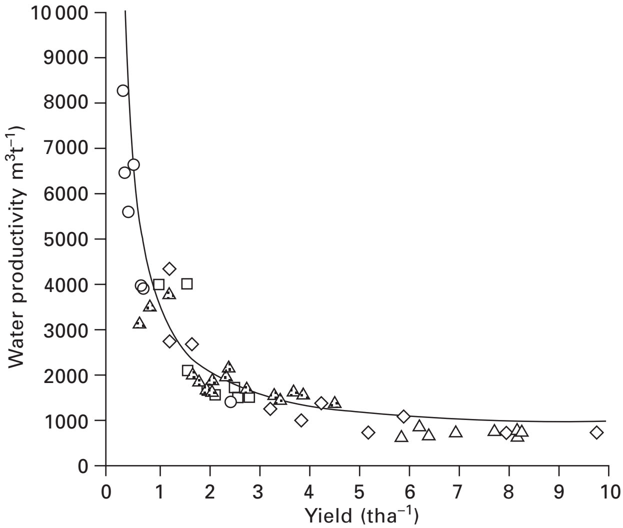

Fig. 2.1 Water productivity (m^3 t^{-1}) in relation to yield. Data points are derived from several crops and several publications. Solid line represents $WP = WP_{(800)} /(1\text{-}e^{[-bY]})$, with $WP_{(800)}$ = 800 m^3 t^{-1} and b = –0.3 (From: Rockström, 2003).

ecological zones, water productivity is more accurately calculated for each region (Table 2.1). Current actual cereal yields are used without distinction between rainfed or irrigated cultivation (FAOstat, 2006). The highest productivity of some 900 L kg^{-1} cereal is obtained in Western Europe at yield levels of 7 t ha^{-1}, and lowest values of 3500 L kg^{-1} at yields of 0.8 t ha^{-1} in Central Africa. Based on these disaggregated values, the global weighted average water productivity is estimated at 1300 L kg^{-1}, which is much lower than the estimate of Rockström *et al.* (1999, 2003).

Water needed for the total cereal production of 3732 million tonnes totals 4831 km^3, which is lower than other estimates that forecast water use to reach some 7000 km^3 (Rockström, 2003; Comprehensive Assessment of Water Management in Agriculture, 2007). As we have based our estimate of water requirements using actual yield data and water productivity at field level, we assume the required amount of 4831 to 7000 km^3 y^{-1} to be composed of both irrigation and precipitation volumes. With 1800 to some 2800 km^3 y^{-1} provided by irrigation water, some 3000 to 4000 km^3 y^{-1} of water for crop growth is obtained from rainwater. When we assume that the irrigation water has been collected on non-arable land areas, an estimated 25–33% of the total precipitation falling on arable land of 11 970 km^3 y^{-1} is used by crops. The remainder of the rainwater on cropland therefore does not contribute to crop production and is 'lost' for the crop through drainage, evaporation, leaching, run-off and off-season rains. These water related-components of agricultural production systems need to be looked into for improving water-use efficiency.

Table 2.1 Water requirements for the major global regions at 2005 levels of cereal production (acreage and cereal yield data from FAOstat-agriculture, 2006)

	Actual cereal acreage (10^6 ha)	Actual cereal yield (t ha^{-1})	Cereal volume (10^6 t)	Water productivity (L kg^{-1} = m^3 t^{-1})	Water total (km^3)
South America	36.5	3.3	120.7	1271	153.3
Central America	13.2	2.7	35.4	1448	51.3
Caribbean	0.9	2.1	2.0	1687	3.4
Northern America	73.4	5.7	416.9	978	407.7
Northern Africa	23.8	1.6	38.1	2097	80.0
Western Africa	41.9	1.0	41.0	3142	128.8
Central Africa	6.6	0.8	5.5	3579	19.9
Eastern Africa	24.0	1.3	30.6	2518	77.1
Southern Africa	5.1	3.0	15.2	1349	20.5
Oceania	19.7	2.1	40.8	1729	70.5
Southeast Asia	52.4	3.7	192.3	1199	230.6
Eastern Asia	87.5	5.2	451.4	1016	458.6
Southern Asia	139.6	2.5	347.0	1522	528.1
Western Asia	22.2	2.2	47.9	1676	80.4
Former USSR	77.9	2.0	156.1	1771	276.5
Former DDR	21.2	3.7	78.2	1197	93.6
Southern Europe	14.8	3.8	57.0	1169	66.6
Western Europe	17.6	6.9	120.4	917	110.4
Northern Europe	7.2	5.9	42.7	964	41.2
World	685.6	3.3	2239.2	1295	2898.7

2.3 Future demand for water and food

An equivalent production of 3732 million tonnes of cereals for 6.4 billion people in 2005 suggests a global average diet of 1600 g grain-equivalents per person per day, which requires 760 m^3 of water per person per year at an efficiency of 1300 L water kg^{-1} grain. Gleick (2003) estimated as high as 1700–1800 m^3 water for food production per person per year for North American diets exceeding 3200 kcal p^{-1} d^{-1}, and 600–900 m^3 for African and Asian diets of 2700 kcal p^{-1} y^{-1}. Rockström *et al.* (1999) arrived at 1200 m^3 p^{-1} y^{-1} based on water productivity and agricultural production, using lower water productivity values of 1800 L water kg^{-1} grain. They estimated a water requirement of 1300 m^3 p^{-1} y^{-1} for a desired diet of 3000 kcal p^{-1} d^{-1}.

Estimating future water demand using a global average food intake that assumes a decrease in intake in wealthier nations and an increase in poorer regions may lead to regional underestimates. As current caloric intake in wealthier nations is not likely to decrease, while the intake in developing nations should reach desired healthy amounts, global average is likely to reach higher values, assumed at 3100 kcal here.

Assuming an increase in the consumption of grain equivalents to 2000 g p^{-1} d^{-1} a total of 945 m^3 p^{-1} y^{-1} water would be required without improvement in water productivity or yield (Table 2.2). The requirement would decrease to 752 m^3 at an average cereal yield level of 5 t ha^{-1}. With 9 billion people on earth, likely to be reached in 2040/2050, total water for cereal production would reach 8500 km^3 without yield improvement and 6800 km^3 at yields of 5 t ha^{-1}. When we assume the amounts of water withdrawal for irrigation to remain within the range of 1800 km^3 y^{-1} (Shiklomanov, 2000) and 2660 km^3 y^{-1} (Oki and Kanae, 2006), a total of 4000–5500 km^3 should be provided by rainwater. Also, assuming the food to be grown on the same land area of 1.4 billion hectares, in order to refrain from further clearing of natural lands, some 33–45% of the rainwater on those lands should be used as evapotranspiration. When correcting for the seasonal effect, i.e. assuming that 60% of annual precipitation is received during the growing season, the efficiency of rainwater would have to increase to 55–77%. Alternatively, expansion of agricultural land will be needed when yields cannot be raised to the required levels on the current land areas and, equally important, to collect the rainwater.

Crops may have multiple cropping seasons depending on variety, thermal conditions and water availability. Under rainfed conditions in Europe, single cropping systems cover 93.0% of the arable area, limited double cropping systems cover 6.4%, and 0.6% for double cropping systems, resulting in 1.04 crop seasons in equivalent all over Europe. In sub-Saharan Africa, 42% of arable areas are used for single cropping systems, 48% for double cropping systems, and 10% for triple cropping systems (FAO and IIASA, 2000), which is equivalent to 1.56 cropping seasons all over Sub-Saharan Africa. Using multiple cropping systems is one of the agronomic measures to increase the effective length of the growing season and thereby the amount of rainwater that can be used for crop production. This does not necessarily mean that the water use efficiency is increased.

2.4 Improving water use efficiency in agriculture

Efficiency gains in the agricultural sector are essential to meet expected demand for water. The main question is whether these efficiency gains can be attained and realized in agriculture. The relations (Fig. 2.1) described by Rockström (2003) show that efficiency gains are possible through an empirical relation that reflects actual water to yield ratios. However, this relation does not reveal the underlying mechanisms, and cannot disclose opportunities as to how efficiency gains could be achieved, except through yield increase. For a sensible assessment of the potential gains in efficiency that can be achieved in agriculture, production ecological concepts have to be used that account for eco-physiological processes in crop growth (e.g. De Wit, 1992). Often, studies that analyse how use efficiencies in agriculture can be enhanced, consider production factors in isolation (e.g. Tilman *et al.*, 2002), while the

Table 2.2 Estimated water use for different diets at three population scenarios

Diet	Current WUE	Current WUE	WUE at $Y = 5\ t\ ha^{-1}$	WUE at $Y = 5\ t\ ha^{-1}$	Population 7.5 billion	Population 9.0 billion	Population 10.0 billion
Grain equivalents ($g\ p^{-1}\ d^{-1}$)	Water req. ($m^3\ p^{-1}\ d^{-1}$)	Water req. ($m^3\ p^{-1}\ y^{-1}$)	Water req. ($m^3\ p^{-1}\ d^{-1}$)	Water req. ($m^3\ p^{-1}\ y^{-1}$)	Water req. at WUE for $Y = 5\ t\ ha^{-1}$ ($km^3\ y^{-1}$)	Water req. at WUE for $Y = 5\ t\ ha^{-1}$ ($km^3\ y^{-1}$)	Water req. at WUE for $Y = 5\ t\ ha^{-1}$ ($km^3\ y^{-1}$)
1600	2.07	756	1.65	601	4510	5412	6014
1700	2.20	803	1.75	639	4792	5751	6390
1800	2.33	850	1.85	677	5074	6089	6766
1900	2.46	898	1.96	714	5356	6427	7141
2000	2.59	945	2.06	752	5638	6766	7517
2100	2.72	992	2.16	789	5920	7104	7893
2200	2.85	1039	2.27	827	6202	7442	8269

combined use of resources has been shown to generate many synergistic effects in raising agricultural productivity.

To systematically search for options to enhance the water use efficiency at crop level, we apply the production ecological approach (Fig. 2.2). It provides a systematic approach that relates the physiological and agronomic dimensions of plant growth. Crops realize their potential growth as is determined by their genetic characteristics and by climatic conditions (primarily temperature, radiation, CO_2 concentration and day length), when other production factors are optimally supplied, i.e. sufficient water and nutrient are available and crops are protected against pests and diseases. Crop growth is limited under insufficient availability of water or nutrients to meet their requirements, resulting in lower yields than optimal. Crop growth is further reduced because of pest, disease and weed infestation. In the following sections of this chapter we systematically describe essential crop and field processes.

2.4.1 Plant water use

Eco-physiological processes form the fundamentals to identify whether it is technically feasible to improve water use efficiencies. There is an almost linear relation between the rate of crop photosynthesis and transpiration because of the exchange of both CO_2 and H_2O through stomata (Fig. 2.3). The physiology of this gas exchange leads to a linear relation between crop photosynthesis and transpiration. However, over 99% of the water required by plants is used for cooling by transpiration. Consequently, with stomata wide open under favourable growth conditions, much water transpires and much carbon dioxide enters the leaves, favouring the photosynthetic process

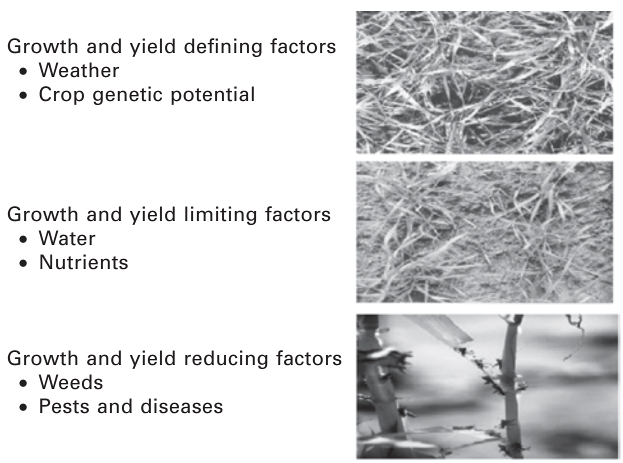

Fig. 2.2 The production ecological approach to systematically arrange production factors that affect plant growth and production (based on Rabbinge, 1993; Van Ittersum and Rabbinge, 1997).

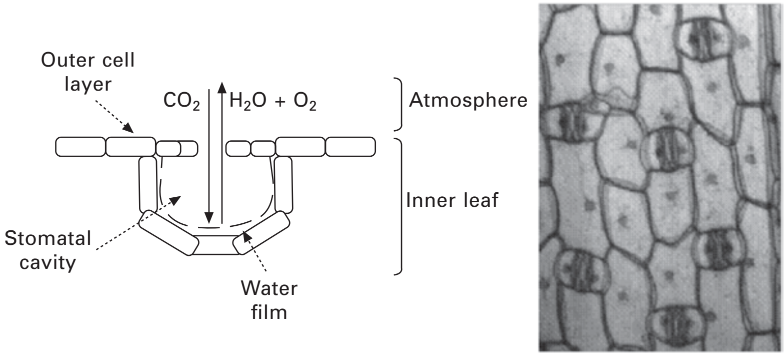

Fig. 2.3 The inflow of CO_2 and outflow of H_2O (schematically represented at the left) is controlled by stomata (right).

and plant production. At optimal rates, plants require approximately 250–300 litres of water for transpiration to produce 1 kg of organic material (Tanner and Sinclair, 1983). This linear relation is affected under extreme conditions, such as drought and low nitrogen contents of plant tissue. The leaf osmotic potential collapses during drought, suppressing photosynthetic capacity (Shimsi, 1970; Chapin III *et al.*, 1988) and because of the resulting closure of stomata (Brodribb and Holbrook, 2003). Associated high temperatures also reduce the rate of photosynthesis under these water stress conditions (Lösch, 1979; Bindraban, 1999). Under nitrogen limitation, the low leaf-chlorophyll content depresses photosynthetic capacity (Shimsi, 1970; Evans, 1983; Bindraban, 1999) while transpiration may remain unchanged, leading to an increased transpiration to photosynthesis ratio. Low nitrogen contents in plant tissue lead to an increase of abscisic acid which, in turn, induces stomatal closure (Chapin III *et al.*, 1988). Schematically, the relation between transpiration and photosynthesis appears as depicted in Fig. 2.4.

2.4.2 Water use at field scale

Cultivated lands lose water not only through transpiration by crops (Fig. 2.5), water evaporates from the bare soil because of heating by solar radiation. The larger the fraction of bare soil in crop cultivation, the larger the loss of this 'unproductive' water will be. Also, water runs off the field or percolates below the rooting zone (drainage) and is lost to the crop. The consequence of the plant physiological processes and the processes at the field scale is that the efficiency of water use can be improved by optimizing agronomic measures such as ensuring the availability of sufficient nutrients and water whenever the plant needs it, and gives the incentive to look in more detail at the physiology and agronomy of plant growth for identifying options to enhance the ratio between water use and growth.

The Production Ecological Approach as developed by De Wit (1992)

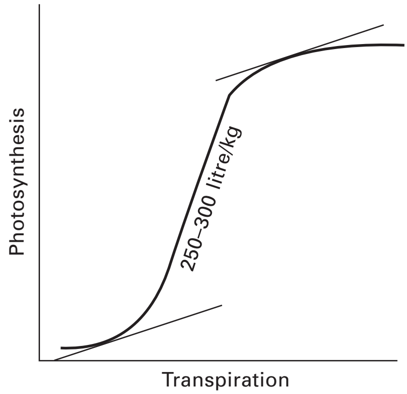

Fig. 2.4 The linear relation between transpiration and photosynthesis resulting in a constant transpiration efficiency over a wide range of growth conditions of 250–300 litres of water for the production of one kilogram of plant dry matter. Deviations occur under extreme conditions (see text).

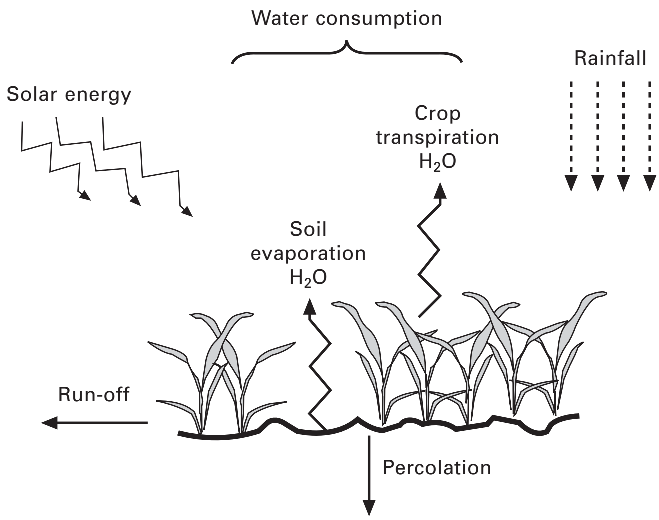

Fig. 2.5 Schematic representation of a water balance at field scale.

and colleagues (Van Ittersum and Rabbinge, 1997; Bindraban *et al.*, 2000) provides a systematic approach that relates the physiological and agronomical dimensions of plant growth (see also Fig. 2.2). The use efficiency of water under optimal growth conditions will be maximal. With insufficient water available to optimally transpire in order to cool its organs, plant growth will be limited and will decrease below the potential. Further growth limitation will occur with inadequate nutrient availability and when the crop experiences competition by weeds or it is attacked by pests and diseases. These production levels are schematically presented in Fig. 2.6, which reveals the decreasing use efficiency of water with worsening growth conditions.

In the field, the soil is exposed to radiation and the heat accelerates the evaporation process, thereby depleting soil water that is actually lost for productive crop transpiration. A range of agronomic measures can be taken to reduce this unproductive loss, such as mulching (providing an isolation layer of organic material that prevents the direct heating of the soil), zero tillage (which prevents direct opening and drying of wet soil surfaces and retains a resistant boundary layer with less favourable evaporation characteristics), and fast ground coverage by rapid closure of the crop canopy. Also, water management practices can be adjusted so as to limit evaporation loss, e.g. the timely delivery of (irrigation) water to the crop or spatially more precise allocation methods, such as drip irrigation.

Water that could be available to the crop, especially rainwater, can leave the field beyond the reach of plant roots through run-off, deep infiltration (drainage) and seepage (horizontal underground soil water movement). Water engineering measures such as contour ridges or other constructions to prevent run-off and increase water storage (on various scales), lining of canals, and installation of pipes are feasible options to increase water availability to the crop.

As a result of other limiting or reducing factors, crop yields can dramatically vary at similar rainwater levels, such as has been expounded by the findings of French and Schultz (1984a,b) for Mediterranean-type climates. Figure 2.7 reveals clear maximum yield levels at a certain level of rainfall, while most observations are scattered in a vertical line below this maximum level, due to limiting factors such as nitrogen and phosphorus availability and reducing factors such as diseases that limit yield more than water availability. In addition, yield reductions occur due to delayed time of sowing, weed infestation, and waterlogging.

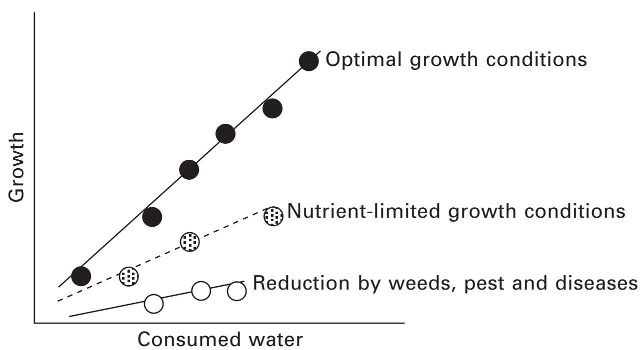

Fig. 2.6 The relation between water consumed by the plant and the amount of carbon dioxide fixed under different production conditions, revealing the decreasing use efficiency of water.

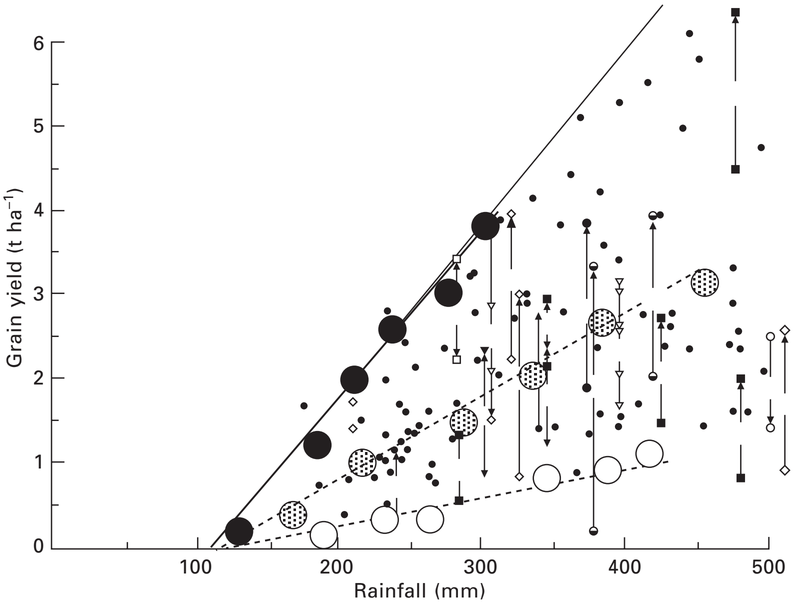

Fig. 2.7 The relation between wheat grain yield and seasonal rainfall for experimental sites and farmers' fields (from French and Schultz, 1984a). The lines and circles are an overlay of Fig. 2.6 to illustrate the applicability of the production ecological concept in explaining variation in yield.

Rockström and colleagues (2003) indeed present strong linear relations between water productivity and yield for field experiments. The largest improvement in water productivity and yield was obtained when combining supplementary irrigation with nitrogen fertilization, underlining the strong synergy between production factors. Fertilization application alone gave better improvements of water productivity than irrigation when dry spells were mild. However, crops would completely fail under heavy drought, with or without fertilization. These data indicate that full benefits of water (harvesting) for supplementary irrigation can be met only by simultaneously addressing soil-fertility management. This principle has been illustrated in Fig. 2.8. However, it should be realized that fertilizer application might also increase production risk under poor rainfall conditions, especially if these are severe enough to induce total crop failure. Additionally, plant growth might be too vigorous during the vegetative phase, using up all available water leading to a collapse in yield with failing rainfall during the reproductive phase (e.g. Fig. 2.9).

More recently, Sadras and Angus (2006) made a similar inventory that allows assessing in more detail the factors that cause low efficiencies of water use (Fig. 2.10). The straight line represents the maximum attainable water use efficiency for cereal crops. The intercept at approximately 60 mm represents evaporation. The slope of the line is equivalent to about 500 L

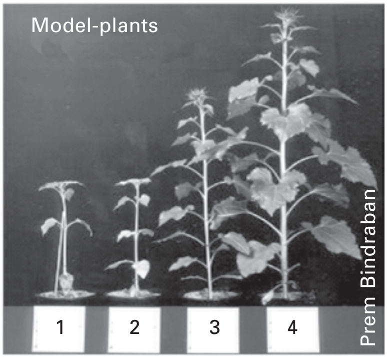

Fig. 2.8 The effect of water and nutrients on plant growth (Own experiments, P.S. Bindraban).

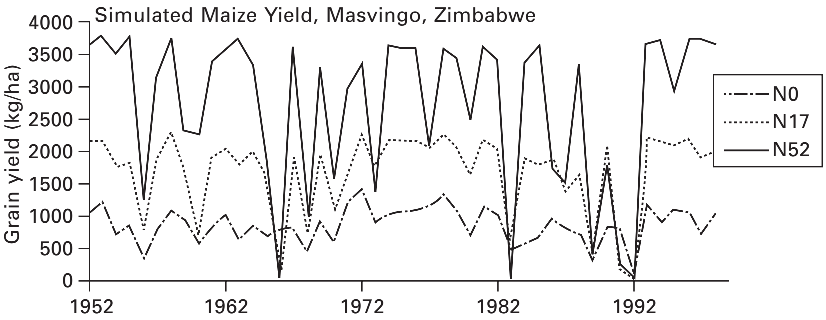

Fig. 2.9 Average yield increases with increasing fertilizer application (0,17,52 kg N ha^{-1}), but so does yield risk. Effective management of the variable rainwater is essential to reduce yield risk. Source: Twomlow *et al.*, 2008.

water per kg of grain, well in agreement with the maximum transpiration efficiency of 250–300 L kg^{-1} biomass of Fig. 2.5, as about half of the total crop biomass ends up in grains, i.e. a harvest index of 50%.

2.5 Future trends and options to increase water use efficiency

The eco-physiological processes described suggest strong interaction between production factors such as for water and nitrogen. Indeed, De Wit (1992) states that 'most production resources are used more efficiently with increasing yield levels'. Many field observations in arid and semi-arid regions indeed reveal that insufficient nutrients limited yield more than water availability, such as for eastern Africa (Smaling *et al.*, 1992; Breman *et al.*, 2001), sub-

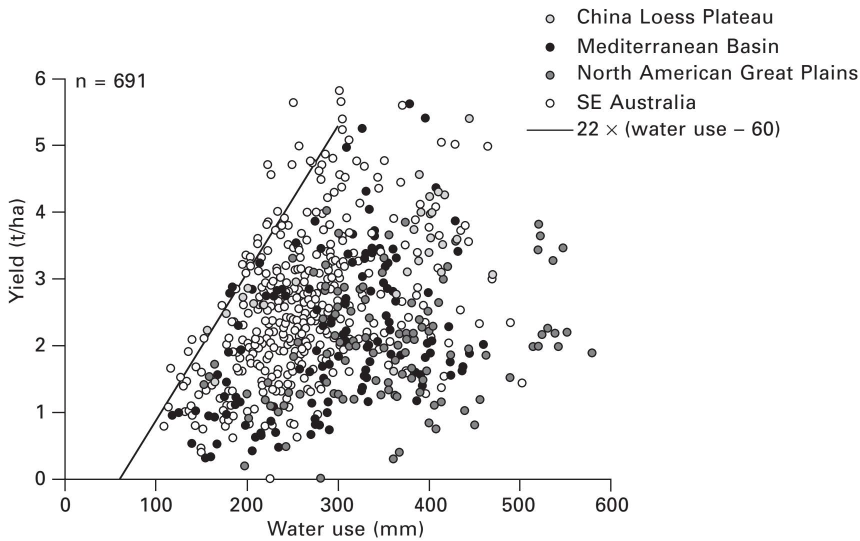

Fig. 2.10 Grain yield related to water use in crop field (Sadras and Angus, 2006). The straight line reflects the maximum attainable yield at the lowest water use and therefore represents the maximum attainable water use efficiency (here about 500 L kg^{-1}).

Saharan Africa (Rockström, 2001), southern India (Ahlawat and Rana, 1998) and western China (Li *et al.*, 2001).

The production ecological approach reveals great potential for increasing the use efficiency of water in agriculture. If losses of rainwater can be reduced and the physiological efficiency can be increased through optimized agronomic measures, including fertilization, much of the required increase in efficiency of rainwater use could be attained without expansion of the agricultural area.

Following the production ecological approach, Conijn and colleagues (presented in Bindraban *et al.*, 2009) showed that land productivity could be doubled or tripled in sub-Saharan Africa if rainwater was properly managed, soil nutrients precisely applied, weeds effectively controlled, and crops protected from pests and diseases. Yield increase would reduce the need for area expansion, while adverse environmental effects due to intensification could be contained within acceptable limits (Fig. 2.11). However, even such large productivity increases are unlikely to be able to supply the growing population with an adequate diet, making further area expansion of agriculture unavoidable.

Table 2.3 presents a summary of the options for increasing water use efficiencies under different agronomic conditions and which of the components of water use are tackled by these measures.

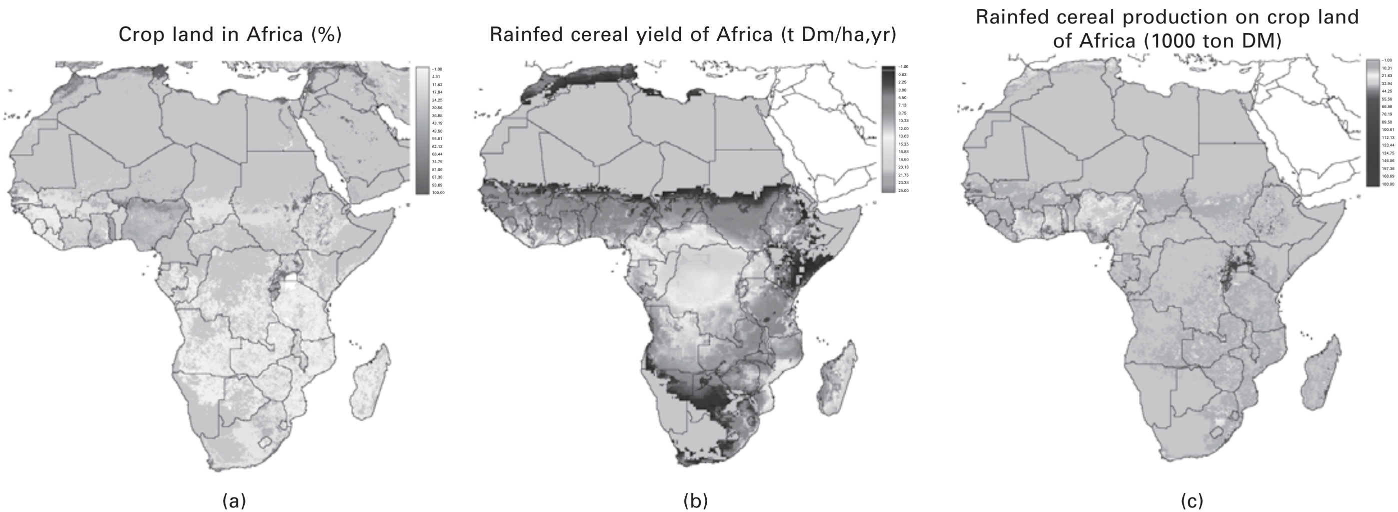

Fig. 2.11 Calculated ecological production potentials based on rainfed agriculture. A. Current distribution of agricultural land (from dark to light – decreasing fraction of grid is agricultural land). B. Maximum attainable biomass production under rainfed conditions on the entire continent. C. Production volumes on current agricultural lands (see Bindraban *et al.*, 2009).

Table 2.3 Measures to increase water use efficiencies per water use component(s)

Components of water used	Agronomic conditions	Growth/ Yield	Range in water use (litre/kg biomass)	Measures to improve the efficiency of water use
Transpiration	Optimum growth conditions – no water stress, sufficient nutrients, no competition by weeds or pests and diseases. Highly controlled/closed systems (e.g. greenhouses)	High	250–300	Breeding Improving transpiration/ photosynthesis mechanisms – virtually no gains likely (Tanner and Sinclair, 1983). Improving the proportion of the biomass that is allocated to edible portion (increase Harvest Index) – limited gains (Bindraban, 1997; Bennett, 2003).
Transpiration + Evaporation	Optimum growth conditions – no water stress, sufficient nutrients, no competition by weeds or pests and diseases. Open fields.	High – Medium	300–600	*Agronomic measures* Mulching, zero tillage, fast ground cover, timely delivery to crop, precise allocation methods (drip irrigation), etc. – numerous location-specific measures feasible.
Transpiration + Evaporation + Drainage/ deep infiltration	Good growth conditions – relative excessive water supply, nutrient limitations or competition by weeds or pests and diseases may occur. Open fields.	High – Medium – Low	500–800	*Water engineering and institutional measures* Lining of canals, installation of pipes to prevent water loss – various options available. Timely and adequate supply of water to minimize drainage loss – possible through improved institutional arrangements.
Transpiration + Evaporation + Drainage/ deep infiltration + Seepage, Run-off	Good to moderate growth conditions – excessive water supply, nutrient limitations or competition by weeds or pests and diseases likely to occur. Open fields.	Low (High)*	800–2000 (5000)*	*Agronomic and water measures* In addition to all measures mentioned above, contour ridges to reduce run-off; water storage constructions at various scales are feasible to increase water availability for the crop. Improvement of agronomic measures to ensure timely and adequate supply of

Table 2.3 Continued

Components of water used	Agronomic conditions	Growth/ Yield	Range in water use (litre/kg biomass)	Measures to improve the efficiency of water use
				nutrients, and suppression of weeds and disease infestations will enhance growth.

* For inundated rice cultivation, water use may be as high at 5000 litres per kg of rice, even when growth conditions are optimal.

2.6 Conclusions

The concern for water to limit global food production in the future and to hamper the development of other economic sectors should not be taken lightly. Much emphasis is currently placed on a better distribution of available water. This might indeed alleviate immediate pressures, especially within societies and between social groups because of a fairer distribution between stakeholders. However, these delicate balances may be difficult to sustain, because of the enormous increase in demand for water for food and other amenities to meet human needs. Realizing substantial gains in water use efficiency are the most effective way out.

In general, it is assumed that increase in food production should be achieved with a proportional increase in water demand by the agricultural sector, further depleting water from natural ecosystems. However, we have shown that a substantial gain can be achieved with currently available rainwater and water that is withdrawn from natural systems for irrigation. By enhancing the productivity of agriculture ('more crop per drop') we can diminish the demand for withdrawal of additional water and for converting natural lands into agricultural lands. The resulting increase in crop yield per area unit will also decrease the need for expansion of the agricultural area. However, as the analysis for the African continent shows (Fig. 2.11), it is not likely that all the increase in water and land productivity can be realized on the current agricultural land. Realizing the production potential will take decades, during which period the population will steadily increase, as well as the demand for even more food, because of improving income and increasing dietary requirements. Bindraban and colleagues (2008) showed that the rate of increase in yield of crops on the African continent is so low that it would take several decades to reach the production potentials calculated in Fig. 2.11. Therefore, expansion of the agricultural land will remain a necessity to obtain the required increase in production volume and to collect the additional water required. Both additional withdrawal of water from natural systems as well as the expansion of agricultural areas worldwide should be limited as much as possible.

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Part II

Life Cycle Assessment of food production and processing

3

Life Cycle Assessment (LCA) of food production and processing: An introduction

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Abstract: This chapter provides an introduction to life-cycle thinking, Life Cycle Assessment and Life Cycle Management. It gives a brief history of LCA development, describes LCA methodology, and discusses specific issues that arise in assessment of food systems. These include assessment of land occupation, soil quality, carbon storage, crop rotations, variability in agricultural practices and yields, consumer behaviour, sewage treatment, definition of the functional unit, and co-product allocation.

Key words: Life Cycle Assessment, food systems, Life Cycle Management, environmental impact.

3.1 Introduction

In the last few years, there has been unprecedented interest in the environmental impacts associated with food systems. This has been engendered by the results of studies such as the European Science and Technology Observatory (ESTO) project on the 'Environmental Impact of Products' (EIPRO) (Tukker *et al*., 2006). The final report for this project reviewed seven existing studies and presented the results of a separate environmental input–output study for final household consumption in the EU25 countries. It found that 'food and beverage consumption' accounted for 22–34% of total life-cycle impacts in all the studied environmental impact categories (apart from eutrophication, where it accounted for 60% of this impact) (Tukker *et al*., 2006, p. 105). Other recent work has confirmed the relative importance of food systems in contributing to environmental impacts. For example, Garnett (2008) calculated that the life-cycle impacts of food consumption in the UK contribute 19% of the UK's greenhouse gas (GHG) emissions, and the United Nations Food and Agriculture Organisation (Steinfeld *et al.*, 2006) calculated that livestock contribute 18% of global GHG emissions (both measured in carbon dioxide equivalents).

Looking forward, the global human population is estimated to grow by 34%, from 6.8 billion, today to 9.1 billion in 2050; the Food and Agriculture Organisation (FAO) of the United Nations predicts that a 70% increase in food production will be required to meet the needs of the human population in 2050 (FAO, 2009). Such a challenge can seem overwhelming. However, a number of alternative – or complementary – strategies can be envisaged, including initiatives to reduce food wastage, change diets, and develop more efficient and effective food distribution systems. Evaluation of these alternatives in order to inform decision-making for more sustainable food systems, requires systems-based thinking. Life-cycle thinking, Life Cycle Assessment (LCA) and Life Cycle Management (LCM) provide a conceptual framework, supporting analytical tools and management systems for evaluating and improving the environmental profile of alternative options from a systems perspective. Life-cycle thinking expands the focus of attention from specific production processes to include consideration of the impacts of products, activities and services 'from cradle to grave' i.e. from extraction of raw materials (such as coal, phosphorus and aluminium) through processing, manufacture, distribution, use and on to final waste management. LCA is an analytical tool for assessing these impacts, and LCM is a management system for delivering continuous improvement in the environmental and socio-economic aspects of products.

This chapter provides an introduction to LCA as a decision-support tool for the assessment of food systems. Section 3.2 outlines the historical development of LCA to meet the emerging demand for product-oriented environmental management systems. LCA methodology is described in Section 3.3, and Section 3.4 discusses the application of LCA to food systems. The chapter concludes with a summary of future trends in the use of LCA for assessment of food systems (Section 3.5), and a short list of resources for those interested in further information (Section 3.6).

3.2 History of Life Cycle Assessment (LCA)

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. 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.

The first international meetings for LCA researchers and practitioners were held under the auspices of the Society for Environmental Toxicology and Chemistry (SETAC) in 1990 and 1991 (Jensen and Postlethwaite, 2008). Throughout the 1990s, SETAC organised Working Groups and published reports on various aspects of LCA methodology and application, and the SETAC annual meetings became a forum for LCA researchers and practitioners to develop a common understanding of the purpose and practice of LCA.

The emergence of LCA was recognised by the International Standards Organisation (ISO) in the mid-1990s, and a series of four ISO LCA standards (ISO 14040 to 14043) were published between 1997 and 2000. During this time, a growing number of researchers had become interested in the use of LCA as a decision-support tool. Some questioned the emphasis on developing 'an objective scientific analytical tool,' arguing that LCA is rooted in a particular way of framing environmental issues* and that its use to support decision-making can undermine the arguments advanced by others using different – and equally valid – frames that are perceived as more subjective (see, for example, Bras-Klapwijk, 1997; Finnveden, 1997; Tukker, 1997).

The increasing interest in the application of LCA emerged as a new topic in the LCA literature around the end of the 1990s. It was called 'Life Cycle Management' and is described as (Remmen *et al*., 2007, p. 18):

> ... a product management system aiming to minimise environmental and socio-economic burdens associated with an organisation's product or product portfolio during its entire life cycle and value chain. LCM is making life-cycle thinking and product sustainability operational for businesses through the continuous improvements of product systems ...

LCM recognises the central importance of using a life-cycle perspective in product-oriented environmental management but also recognises the importance of focusing on how life-cycle thinking is integrated into decision-making processes.

In 2002 a collaborative partnership was launched between SETAC and the United Nations Environment Programme (UNEP): the Life Cycle Initiative. Its objectives are (UNEP and SETAC, 2009):

- Collect and disseminate information on successful applications of life-cycle thinking;
- Share knowledge about the interface between Life Cycle Assessment and other tools;
- Identify best practice indicators and communication strategies for Life Cycle Management;
- Provide a basis for capacity building;

*'Framing is a way of selecting, organizing, interpreting, and making sense of a complex reality to provide guideposts for knowing, analyzing, persuading, and acting' (Rein and Schön, 1993, p. 146).

- Expand the availability of sound LCA data and methods;
- Facilitate the use of life-cycle based information and methods.

Work is being carried forward through three work programmes with associated task forces: Life Cycle Management, Life Cycle Inventory, and Life Cycle Impact Assessment.

Over the last few years, interest in carbon footprinting – and the emerging interest in water footprinting – has led to a more widespread recognition of LCA's appropriateness in support of environmental management initiatives. The UK's PAS 2050 (BSI, 2008) is largely based on LCA methodology as defined in the ISO LCA standards. The draft ISO 14067 Carbon Footprint of Products standard is also likely to be based on the ISO LCA standards.

With respect to application of LCA to agri-food systems, the popularisation of the Food Miles concept around the middle of the 2000s led to greater interest in development of LCA-type tools and assessments. In early 2007, Tesco's, in the UK, announced that it intended to label all its products with carbon footprint information (Leahy, 2007), and retailers such as Marks & Spencer and Walmart have also indicated their commitment to reducing the carbon footprint throughout the supply chains of products sold through their retail outlets. Pre-dating this recent interest, the first 'International Conference on LCA in the Agri-Food Sector' was held in Brussels in 1996. It has been convened every two years or so since that time; the last one was in Zurich in November 2008 and attracted 160 participants from 32 countries (Gaillard and Nemecek, 2009).

3.3 LCA as a decision-support tool

LCA is a technique for assessing the environmental aspects and impacts of products, activities and services along the life cycle from extraction of raw materials, through processing, manufacturing, distribution, use, and on to final waste management. This is shown diagrammatically in Fig. 3.1. The unit processes under study (indicated by the individual boxes in Fig. 3.1) are collectively termed the 'product system'.

Four different phases can be defined when undertaking an LCA study:

- *Goal and Scope Definition*: The goal and scope of the LCA study are defined in relation to the intended application.
- *Inventory Analysis*: The inventory analysis involves the actual collection of data for individual unit processes and the calculation procedures. The result of this phase is a table which quantifies the relevant inputs and outputs of the product system.
- *Impact Assessment*: The impact assessment translates the results of the Inventory Analysis into environmental impacts (e.g. climate change,

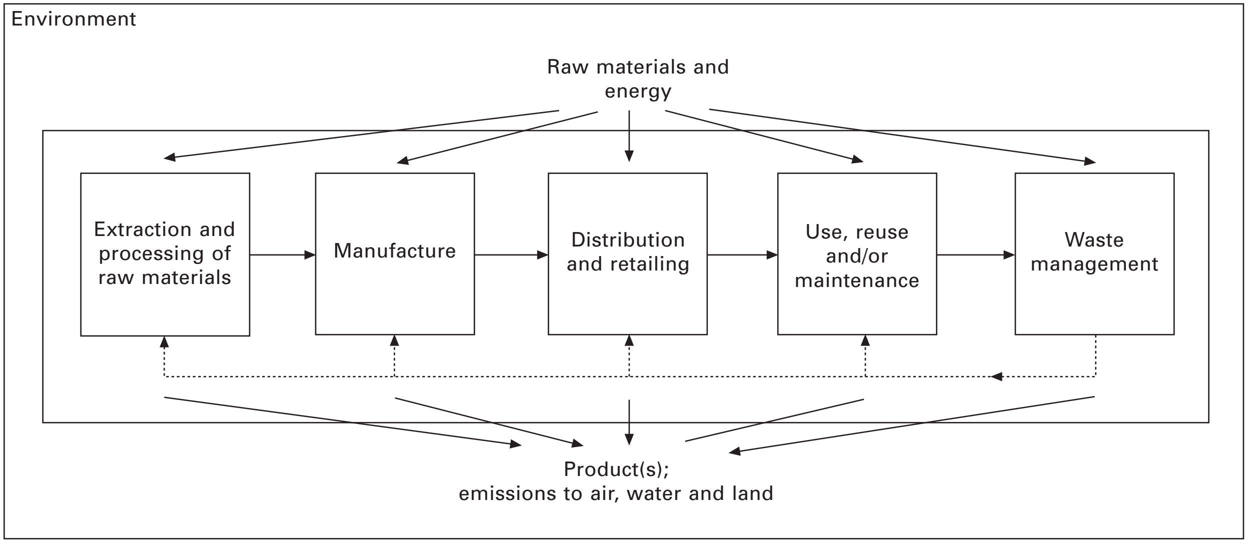

Fig. 3.1 Generic flow diagram for life-cycle thinking and LCA (Source: Hodgson *et al.*, 1997).

ozone depletion). The aim of this phase is to better understand the significance of the environmental impacts of the product system.

- *Interpretation*: At this phase, conclusions and recommendations for decision-makers are drawn from the Inventory Analysis and Impact Assessment.

These phases can be represented as shown in Fig. 3.2. 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.

Critical features of LCA that distinguish it from other environmental management approaches, include:

- Consideration of environmental aspects and impacts occurring along the life cycle of products and services, i.e. from raw material extraction through to final disposal.
- Assessment of more than one type of environmental impact. This distinguishes LCA from approaches such as carbon and water footprinting.
- Inclusion of environmental impacts in the assessment, irrespective of their geographical location and whether they occur in the past, present or future. This links with the commitment to inter- and intra-generational equity articulated in sustainable development agendas.
- Use of a functional unit that is defined in terms of the service delivered by the product system.
- The relative nature of LCA, which is expressed by defining environmental impacts relative to a reference unit (the functional unit) in a study.

A more fundamental, and controversial, feature of LCA concerns whether it is regarded as a type of systems analysis or more of an analytical method.

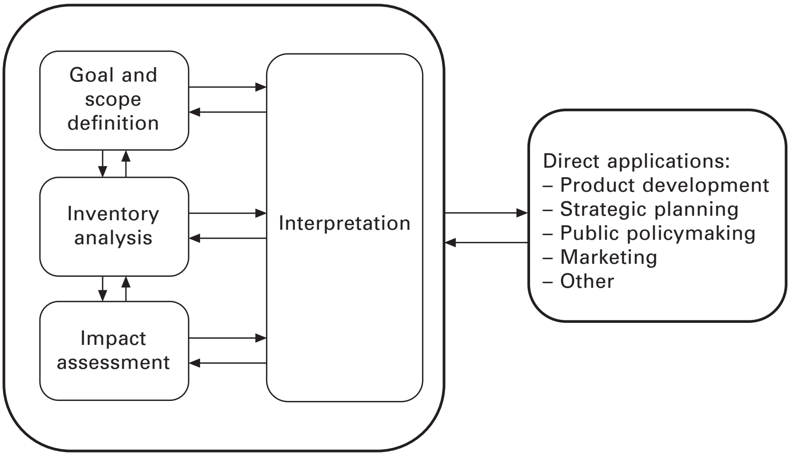

Fig. 3.2 The phases of Life Cycle Assessment (Source: ISO, 2006b).

In systems analysis, the focus is on the interactions between different constituent parts of the 'whole' as well as the constituent parts themselves, and on the emergent properties of this 'whole.' Analytical methods, on the contrary, focus on increasing understanding by breaking down the object of study into smaller parts and analysing each one separately (Baumann, 1995, p. 20–27). Thus, for example, the 'wetness' of water cannot be understood by analytical study of its constituent elements hydrogen and oxygen – but only in relation to water molecules and their behaviour *en masse* (Wilson, 1990, p. 23).

Applying systems thinking to LCA, the product system under study can be regarded as situated within a wider background system that is itself situated within 'the environment'; the service delivered by the product system under study is effectively the emergent property of this system. This is illustrated in Fig. 3.3 where the term 'Foreground System' includes the economic processes directly contributing to the product system, the term 'Background System' includes all other economic processes that contribute to the Foreground System (e.g. material and energy production and supply), and the 'Environment' is the wider world within which human economic activities take place.

The significance of this conceptualisation with respect to LCA is that, in addition to assessing the direct environmental impacts of Foreground System activities, associated changes in the environmental impacts of the Background System are equally relevant. For example, an electricity generating station may generate heat in addition to electricity; this heat is supplied to a district

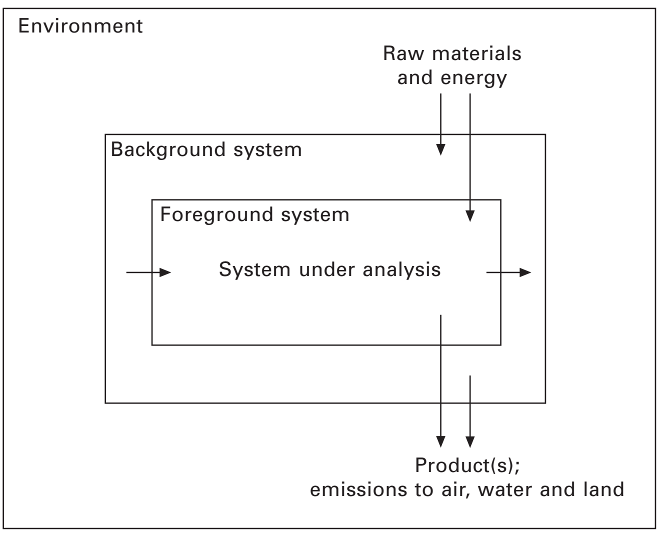

Fig. 3.3 The foreground and background systems in LCA (Source: Cowell, 1998, p. 15).

heating system, displacing heat from gas heaters used in residential homes. Let us assume that the purpose of the LCA in this particular case is to assess the environmental impacts of electricity generation. Using a systems perspective, the *avoided* environmental impacts of gas production and use are valid for inclusion in the study because they occur as a direct result of the Foreground System activities (i.e. electricity generation). However, from a more analytical perspective, it may be considered most appropriate to allocate the direct environmental impacts associated with electricity generation between the two co-products (electricity and heat), and omit consideration of the avoided impacts in the Background System. This aspect underlies the continuing debates over the legitimacy of attributional versus consequential modelling approaches, and system expansion versus allocation in LCA; they are considered further in Chapter 4 by Tillman.

3.4 Application of LCA to food systems

3.4.1 Overview

Many of the challenges associated with the use of LCA to assess food systems are the same as for industrial systems. Examples include data quality, communication of results, and use of input–output data and hybrid methods. However, other aspects are particularly relevant in the analysis of food systems, or are unique to food systems, including:

- Use of agricultural land:
    - Land occupation – two alternative agricultural systems producing the same output may have different land occupation requirements (usually assessed as m^2*days). An issue concerns whether the 'surplus' land in one system should be included by using system expansion to assess the services provided by alternative activities on that land.
    - Soil quality – factors contributing to soil quality include the presence of organic matter, water content, trace substances (such as nutrients and heavy metals), living organisms, and soil texture and structure (Cowell and Clift, 2000). The impacts of agricultural activities on these aspects of soil quality are rarely considered in agricultural LCAs.
    - Carbon storage in soils and standing biomass – a considerable amount of carbon is stored in agricultural soils and in trees, bushes and vines. Accounting for maintenance of, and changes in, soil and biomass carbon due to agricultural activities is a challenging methodological issue.
    - Crop rotations – land may be left fallow (with occasional mowing) between crops or used for green manure production, and the environmental impacts associated with these activities need to be

'allocated' in some way among the agricultural outputs over an entire crop rotation. Similarly, the environmental impacts associated with infrequent activities (such as applications of lime and compost) need to be allocated among the agricultural outputs over a crop rotation.

- Variability:
  - Yield variability between years – yields may vary significantly between years due to weather conditions and other factors. When the aim is to support decision-making leading to longer-term changes in agricultural production systems, this variability needs to be interpreted for decision-makers so that decisions are not based on atypical years.
  - Individual farmer practices – several studies have identified the important role of individual farmers' behaviours in determining the magnitude of environmental impacts associated with (some) food products. There are issues around how to use this information to drive improvements, and whether it should be communicated to consumers.
- Post-farm life-cycle stages:
  - Consumer behaviour around food consumption – some interesting interactions arise between portion sizes, packaging, and wastage of food in the home. The relevance of a study can be enhanced by explicitly addressing these interactions.
  - Sewage treatment – the 'waste management' stage of food products involves human excretion and subsequent sewage treatment. However, it is often omitted from LCAs of food products.
- Other modelling aspects:
  - Co-product allocation – many agricultural systems produce more than one economic output, and so accounting for the environmental impacts associated with any one output is an issue.
  - Definition of the functional unit – this can be expressed in terms of mass, a portion, a specified amount of protein or nutrients, or some other parameter.

In Sections 3.4.2 to 3.4.5, a brief overview is provided of these aspects. To illustrate the issues, examples are provided from work undertaken to support two recent carbon footprinting studies on kiwifruit and apples produced in New Zealand and consumed in the UK (Mithraratne *et al*., 2008; Hume *et al*., 2009). For both studies, ISO 14040 and 14044 standards were followed for the analysis, and their interpretation in the PAS 2050 specification was also taken into account. These studies were funded by the Ministry of Agriculture and Forestry (MAF) in New Zealand and two industry bodies (Zespri International for kiwifruit, and Pipfruit NZ for apples), and were undertaken by Landcare Research in association with AgriLINK, Plant and Food Research, and Massey University. The aim of the studies was to gain a better understanding of the carbon footprint for these two fruits using an LCA-based approach, identify improvement options, and work towards development of strategies for reducing the carbon footprint on a sector-wide basis. The kiwifruit study modelled the carbon footprint of green, gold and green organic kiwifruit based on data from 61 orchards. The apple study modelled the carbon footprint of Braeburn and Royal Gala apples produced on 59 orchards using integrated and organic production techniques. For both kiwifruit and apples, the studies modelled a baseline scenario where products are shipped to the UK and subsequently distributed to retail outlets by truck; the consumer is assumed to travel by car to and from the retail outlet, and subsequent consumption and sewage treatment is taken into account.

3.4.2 Use of agricultural land

Assessment of land use in LCA requires consideration of the occupation of land (measured in area*time), and changes and/or maintenance of a certain quality of land use (see, for example, Milà i Canals *et al*., 2007a, 2007b).

Occupation of land can be measured in a straightforward way by consideration of the area and time required for production of an agricultural output. However, an interesting issue arises when comparing alternative agricultural systems producing the same output but having different area*time values. In these cases, it has been suggested that the difference in area*time values should be taken into account in an LCA by accounting for alternative use of the 'surplus' land in one of the systems (Gaillard, 1996; Charles *et al*., 2006). In such cases, the choice of this alternative use of land can be a determining factor in the overall LCA results.

Assessment of land quality is more complicated. Relevant aspects include biodiversity, soil quality and the biotic production potential of the land (Milà i Canals *et al*., 2006). To date, most researchers have focused on development of methods for assessing soil quality (e.g. Cowell and Clift, 2000; Milà i Canals *et al*., 2007b) and biodiversity (e.g. Jeanneret *et al*., 2009; Koellner and Scholz, 2008; Schmidt, 2008). An underlying question concerns whether it is actually possible to define a generic assessment method for all types of land use in all parts of the world. A more pragmatic approach may be to focus on developing a process for selecting specific indicators of soil quality and biodiversity (such as soil compaction or soil organic matter for soil quality, and certain taxonomic groups for biodiversity), depending upon their relevance to the systems under analysis. Alternatively, these aspects could be considered in a qualitative way alongside the LCA results based on existing impact categories. These issues are discussed further in Chapter 6 by Nemecek and Gaillard.

An important question, given the current interest in climate change, concerns assessment of carbon storage in soil and in standing biomass on agricultural land. The UK's PAS 2050 (BSI, 2008) on carbon footprinting includes changes in the carbon content of soils due to direct land use change but excludes changes in existing agricultural systems (BSI, 2008, Sections 5.5

and 5.6). The reason given is that 'there is considerable uncertainty regarding the impact of different techniques in agricultural systems'; however, it will be considered further in future revisions of the specification (BSI, 2008, Section 5.6, Note 2). It is worth noting that changes in the carbon content of soil and standing biomass may be significant in LCAs of some food products. Box 1 gives an example from the NZ apple carbon footprinting study.

Box 1

Carbon storage in soils and standing biomass

For the apple study, limited data were available on orchard soil organic matter. Extrapolation from these data suggests that the carbon footprint (CF) of soil carbon lost following conversion from pastoral land use to apple orchards can be approximately the same as the CF for integrated orchard production, and half the value of the CF for organic orchard production (up to the orchard gate). These data are based on measurements of soil carbon losses in the top 0.3 metres over 12 years in the rows and alleys of an integrated and organic orchard, and assume Global Warming Potentials (GWPs) equivalent to those used for assessing GHG emissions from burning fossil fuels. If it is assumed that this same carbon loss should be extrapolated over 20 years rather than 12 years (i.e. the time period recommended in the PAS 2050 following a change of land use), then the CF for loss of soil carbon would be almost half of these values. Depending upon the assumptions made, then, the data suggest that the CF for a long-term loss of soil carbon can be potentially significant relative to the CF associated with other orchard activities. (Conversely, of course, the CF for a long-term increase in soil carbon will also be significant.) The data also suggest that different agricultural production techniques (such as integrated and organic production) may have significantly different CFs for changes in soil carbon over time, and also raise the question of whether maintenance of soil carbon should be included in the assessment. Methodological issues related to this topic are discussed in more detail in Müller-Wenk and Brandão (2010).

Likewise, a study of apple trees on integrated orchards found that their dry matter (DM) increased annually by 2.2 t-DM per ha over the first eight years (Palmer *et al.*, 2002). Assuming the carbon content of the woody tissue of apple trees to be 47% (Walton *et al.*, 1999), the avoided CF for this carbon storage is approximately equivalent to the CF for integrated orchard production (on a per ha*year basis). However, it should be noted that this comparison is relevant only if carbon storage continues for a long period of time (such as 100 years), and furthermore the total avoided CF would have to be 'shared' amongst all the harvested apples over the entire time period. In reality, orchard trees may be replaced every 15 years or so due to breeding of new cultivars, and so the results would need to be modified to represent this short storage time (and/or alternative use of the wood). Nevertheless, the data do suggest that carbon storage in the standing biomass of orchards is worth further consideration in carbon footprinting studies.

Source: Deurer *et al.*, 2009, Appendix 1.

Finally, crop rotations raise some interesting modelling issues in LCAs.

Some activities may take place less frequently than once per crop and yet have benefits for the different crops in a rotation. For example, compost may not be applied every year in kiwifruit and apple orchards and yet it may contribute to increased productivity of the orchards for several years after initial application. For many other crops, a single phosphate fertiliser or lime application may have benefits for several crops in a rotation – or a 'green manure' crop may be grown specifically to enhance the soil's productivity for future crops (Cowell and Clift, 2000). Regarding compost application in orchards specifically, this issue was overcome in the kiwifruit and apple studies by sampling a relatively large number of orchards in one year; the average application calculated from data for all the orchards was assumed to be representative of average practice. However, for studies that involve collection of data from a limited number of sites, care must be taken to include consideration of infrequent activities occurring either prior to or after the system under analysis (see, for example, van Zeijts *et al.*, 1999).

3.4.3 Variability

Yields of agricultural products may vary markedly between years as a result of weather conditions, disease and pest outbreaks, differences in management practices, and – in the case of orchards – the maturity of the vines, bushes or trees. An example is provided in Box 2 for kiwifruit. Thus, even if the environmental impacts associated with agricultural activities are the same on a per hectare basis across different years, the environmental impacts per kg agricultural output may still vary markedly, solely due the differences in yields between years – and some of these differences may be due to weather conditions that are outside the control of the farmer. In comparative studies between agricultural products originating in different geographical areas (and produced under variable weather conditions), then, it is important to be aware of yield variability and whether the data collected for a specific year are actually representative of average yields. If this information is to be used to support policymaking and/or support improvements in agricultural production over the longer term, it may be wise to collect data across more than one

> Box 2
>
> *Yield variability in New Zealand kiwifruit*
>
> Figure 3.4 shows the average yields of green, green organic and gold kiwifruit in New Zealand over six consecutive years (measured as 'tray equivalents' (TEs) which typically comprise about 33 kiwifruit). The difference between the lowest and highest value over the six years, measured as a percentage of the lowest value, is 31%, 54% and 59% for green, green organic and gold kiwifruit respectively. The variation in yield is due to differences in climatic conditions during critical stages of development, increases in planted areas with lower yields in early years, and improved productivity of orchards.

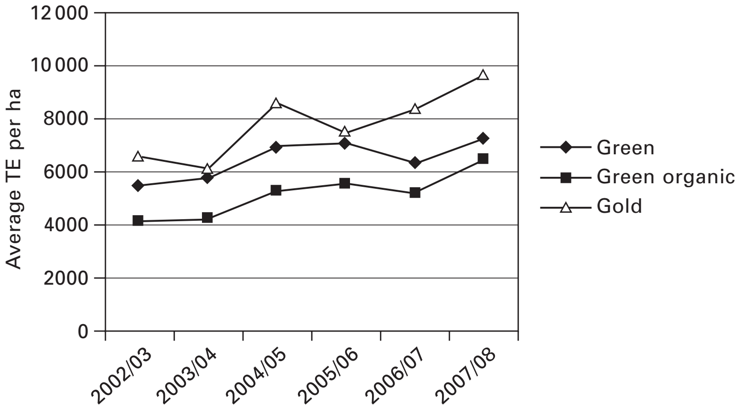

Fig. 3.4 Yield variability in New Zealand kiwifruit

year in order to calculate (more) representative values. For example, in the kiwifruit and apple studies, it was suggested that a system for benchmarking different orchards against a weighted average should present data as five-year rolling averages, where initial years are averaged until Year five is reached and there is then a rolling five-year average (McLaren *et al.*, 2008).

Another source of variability concerns individual farmer practices which may differ due to factors other than the specific topographical, soil and weather conditions (e.g Milà i Canals *et al.* 2006; Alig *et al.*, 2009). For example, in a sample of orchards in the apple study in one particular region, the average number of tractor passes for spraying pesticides for 2006/07 was 23.6 with a standard deviation of 5.1, and 21 with a standard deviation of 4.8, for Braeburn and Royal Gala respectively (Deurer *et al.*, 2009). The researchers hypothesised that those using a higher number of tractor passes could be less experienced and therefore more conservative in their management of pests and diseases, or could be more averse to risk. Recognising the variability between individual farmer practices, an industry-led consortium has established the Apple Futures Programme which encourages voluntary behaviour change with respect to use of pesticides; in addition to the reduced environmental impacts associated with using less pesticides, this programme may also result in fewer tractor passes for spraying pesticides (and reduced financial costs and environmental impacts from use of less diesel).

3.4.4 Post-farm life-cycle stages

The role of consumers in influencing the environmental impacts associated with the life cycle of food products is increasingly a focus of attention. Wastage of food in the home is particularly relevant; a UK study found that consumers throw away 22% of all food and drink purchased, and more than 80% of that could have been eaten (WRAP, 2009); that is to say, excluding inedible food, 18% of household food is wasted. From a life-cycle perspective, each kiwifruit or apple that is thrown away because it has gone soft, is modelled as additional environmental impacts associated with the life cycle of the kiwifruit or apple that is eaten; the additional environmental impacts are those associated with production, storage and distribution of the wasted fruit and its subsequent waste treatment. Thus, for example, effectively the production, storage and distribution of 1.2 kiwifruit or apples needs to be modelled for every kiwifruit or apple that is eaten, assuming that the WRAP wastage rates are accurate for kiwifruit and apples as well as other food products. Wastage is equally important at other life-cycle stages (see, for example, Berlin *et al.*, 2007; Berlin and Sonesson, 2008). It is obvious that initiatives to reduce food wastage at all life-cycle stages have to be among the highest priorities when considering improvement options.

Related to consumer behaviour, packaging of food products has been studied using LCA for many years. It is argued that appropriately sized food packages can reduce wastage of food. For example, the additional environmental impacts associated with packaging for providing 500 g butter to a restaurant in 50 small packages rather one large package, may be much smaller than the environmental impacts associated with the butter that would be wasted if it had to be cut from the one large package and leftovers thrown away (as discussed in Büsser and Jungbluth, 2009).

Consumers also influence the life cycle of food products through food preparation choices. For example, Büsser and Jungbluth (2009) undertook LCAs of five types of coffee; they concluded that, for the majority of environmental impacts, the most relevant life-cycle stages are coffee production, brewing of coffee, and milk production (in the case of white coffee). For brewing of coffee, the consumer can reduce environmental impacts through choice of an energy-efficient kettle rather than a coffee machine, switching off the coffee machine when not in use, and only heating a minimal amount of water in the kettle.

Finally, as Munoz *et al.* (2008) state, 'Human digestion and excretion remains the least studied life-cycle stage of food products.' Yet this life-cycle stage is the equivalent of the waste management stage for industrial products, and its omission from LCAs of food products may compromise the usefulness of such studies in identifying opportunities for environmental improvements. For example, Cowell and Clift (1997) modelled the flows of phosphorus through a conventional bread production system, demonstrating that the greatest loss of phosphorus from the system occurs through dispersion in the sewage effluent. Precipitating phosphorus from the sewage effluent into the sludge and subsequently using the sludge as a fertiliser (provided the phosphorus is in an available form) may displace application of phosphate fertilisers; this may be a more effective improvement option (from a resource depletion perspective) than attempts to reduce the use of phosphorus fertilisers in farming systems. Other studies have focused on the relevance of sewage treatment in assessing the eutrophication potential of food products; for example, Munoz *et al.* (2008) calculated that sewage treatment contributes 45% of the total eutrophication potential throughout the life cycle of broccoli.

3.4.5 Other modelling aspects

A number of interesting modelling issues arise when applying generic LCA methodology (as defined in the ISO 14040 series) to food products, and some have already been discussed in the preceding sections. This section provides an overview of two additional issues that are particularly relevant: definition of the functional unit, and co-product allocation.

According to ISO 14040 (Section 3.20), the functional unit is defined as 'the quantified performance of a product system for use as a reference unit.' It is an expression of the service provided by the product system, and may be interpreted in terms of:

- Functionality: for spreads such as butter and margarine, 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 (Weidema, 1993, p. 2).
- Nutritional value: relevant parameters may be calories, protein or vitamin content. Thus, for example, one might compare a specified quantity of protein delivered by X grams meat or Y grams soybeans.
- Portion size: smaller kiwifruit or apples may be perceived as equivalent (or not) to larger fruit in a fruit bowl because the consumer considers that one piece of fruit is an appropriate dessert portion.

Furthermore, the 'service provided' may vary depending upon the intended audience for the LCA study (Cowell, 1998). Thus, for example, in the kiwifruit study, the functional unit was defined as, 'a single-layer-tray equivalent quantity of kiwifruit eaten by the consumer.' This functional unit was chosen rather than a mass-based functional unit or number of kiwifruit because the primary audience for the study was the kiwifruit industry, and 'tray equivalents' are the conventional unit of analysis used throughout the kiwifruit industry. It was surmised that the results would be most meaningful to this audience when expressed in tray-equivalents. A more detailed discussion on functional units can be found in Tillman (Chapter 4).

The allocation issue arises when more than one economic output is produced from a single production system. According to the ISO 14040 series of LCA standards, 'decisions within an LCA are preferably based on natural science' (ISO 14040, Section 4.1.8.2), and a hierarchy of approaches should be followed in situations where allocation becomes an issue (ISO14044, Section 4.3.4.2). Step 1 in the hierarchy involves avoiding allocation by (a) dividing the unit process to be allocated into sub-processes relevant to the different co-products and collecting data separately for these sub-processes, and/or (b) expanding the system boundary to including the additional functions related to the co-products. Step 2 involves partitioning the inputs and outputs between the co-products in a way that reflects the underlying physical relationships between them, and Step 3 involves allocation on some other basis (e.g. economic value).

Choice of a particular allocation method is often required in LCAs involving agricultural production, as these systems characteristically produce more than one economic output. For example, cereal crops produce grain and straw, oilseed crops produce meal and oil, dairy cows produce milk and beef, and chickens produce eggs and chicken meat. In the kiwifruit and apple studies, allocation is an issue because each orchard produces different grades of fruit: kiwifruit are classified as export or domestic quality, and apples are graded as either export, domestic or process apples. The allocation question concerns how the inputs and outputs associated with orchard activities are allocated to these different fruit grades because they are co-products from one production system. Recognising that the primary function of kiwifruit and apple orchards in New Zealand is to produce export-grade fruit, it was decided to follow Step 1 of the hierarchy and apply system expansion to account for the other fruit grades. For domestic grade kiwifruit, it was assumed that they displace other fruit in the marketplace whose production and distribution have environmental impacts equivalent to kiwifruit production; the same approach was taken for domestic and process grade apples (Assumption 1). Effectively, this means that the inputs and outputs are allocated on a mass basis between the different co-products from the kiwifruit and apple orchards; however, it is important to note that the line of reasoning is entirely different from allocation on a mass basis. For example, it could alternatively be argued that a large proportion of process apples are processed into apple juice, and that the closest alternative to apple juice (from a consumer's perspective) is either orange juice (Assumption 2) or tapwater (Assumption 3). In these cases, system expansion to account for the process apples would require subtraction from the product system of the environmental impacts associated with production and distribution of either the displaced orange juice (Assumption 2) or the tapwater (Assumption 3). Depending upon which displaced product is chosen, the remaining environmental impacts that are associated with the export apples may be either larger (Assumption 3) or smaller (Assumption 2, depending upon the CF for oranges compared with apples) than those resulting from Assumption 1.

The example of process apples illustrates an interesting point about the use of system expansion: it can lead to quite different ways of perceiving a product system and improvement options. For the process apples, use of system expansion shifts the focus of attention to the products that are displaced by apple juice in the marketplace – and suggests that reduced environmental impacts can be realised by market positioning of apple juice to displace alternative products with greater environmental impacts.

3.5 Future trends

Considering this evidence and the trends, it is clear that life-cycle thinking is critical in the evaluation of alternative options for more sustainable food systems. And it is just as relevant for 'big picture' thinking about global strategies as for a detailed comparative analysis of Product A versus Product B. Some of the themes that are likely to become a focus of increasing attention include:

- Standardisation of life-cycle-based assessment methods for different product categories – Although the ISO 14040 and 14044 standards provide generic guidelines for undertaking LCA studies, more specific methodological guidelines are required for comparisons between alternative products within a single product category (e.g. apples, kiwifruit). These guidelines may be desirable for issues such as: inclusion or exclusion of particular unit processes, allocation method, and definition of the functional unit. Publication of Product Category Rules (PCRs) may be an appropriate approach for driving this standardisation for product categories; PCRs are defined as sets of specific rules, requirements and guidelines for developing environmental declarations that use quantified environmental data (ISO, 2006a, Section 3.3.5). However, recognition of the legitimacy of PCRs will depend upon development of an internationally recognised process for drafting, reviewing and ongoing management of the PCRs.
- Standardisation of life-cycle-based assessment methods for different impact assessment categories – Recent interest in carbon footprinting has resulted in two international initiatives to standardise carbon footprinting methodology: the draft ISO 14067 standard on carbon footprinting, and the Greenhouse Gas Protocol Initiative. Another ISO Working Group has recently started work on a standard for water footprinting. Rather than focusing on specific product categories, these initiatives focus on standardising methodology for a particular impact category.
- Communication of LCA results – As noted in Section 3.2, although much effort has been invested in development of LCA methodology, less emphasis has been placed on how LCA studies are used to support decision-making. However, this is changing with the increased interest in carbon labelling of products amongst retailers and consumers. Key issues to be resolved concern whether environmental declarations and labels communicate:
    - Absolute values for environmental impacts or values relative to the performance of a reference or 'average' product
    - Information on whether environmental impacts associated with the product have been reduced over a defined time period
    - One value that has been derived by weighting of the quantified results for different environmental impacts or several values that separately quantify the product's environmental impacts.

Of course, a related question concerns whether such information is actually likely to influence consumers' purchasing choices. Instead, it may be considered more appropriate to use this information in business-to-business communication; retailers may then implement 'choice editing' to ensure that products stocked on their shelves meet specified environmental standards (Sustainable Consumption Roundtable, 2006).

- Life Cycle Management
  As discussed in Section 3.2, Life Cycle Management (LCM) is the pragmatic application of life-cycle thinking to support decision-making. In other words, LCM is the process (or series of processes) that delivers the environmental (and associated socio-economic) benefits of applying life-cycle thinking. In future, there is likely to be a greater emphasis upon efficient implementation of LCM in companies and governmental organisations, and the role of LCA in this process.
- Sustainable consumption
  The emphasis in environmental management since the 1980s has been on development of more sustainable production systems. However, more recently there has been increased interest in how and why people choose to consume products and services – and the implications for development of more sustainable societies. This raises questions about the role of businesses, governments and individual consumers in fostering more sustainable patterns of consumption (see, for example, Sustainable Consumption Roundtable, 2006; World Business Council for Sustainable Development, 2008). Initiatives such as the United Nations' 10-year Framework of Programmes on Sustainable Consumption and Production (SCP) and the EU's SCP and Sustainable Industrial Policy Action Plan (Commission of the European Communities, 2008) mean that this new agenda is unlikely to disappear. LCA has a central role to play here in demonstrating the relevance (or not) of life-cycle stages under the control of different stakeholders along the life cycle of products, stimulating thinking about improvement options from a life-cycle perspective, and providing information to inform consumers' choices between alternative products.

It is clear, then, that life-cycle thinking, LCA and related techniques have a part to play in defining a sustainable future for the Earth's burgeoning human population. Challenges in use of LCA such as the lack of data, inherent variability in datasets, allocation methods, and definition of appropriate system boundaries will continue to be a subject of discussion. However, these challenges must be set in context against LCA's role in identifying environmental hotspots along the life cycle of products, revealing trade-offs between alternative improvement options with environmental impacts at different life-cycle stages in a product system, and comparing the environmental impacts of alternative ways of providing goods and services for society.

3.6 Sources of further information and advice

Publications

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Websites

The Food Climate Research Network (FCRN) provides a free e-newsletter on research, events, publications related to the climate change impacts of food systems, and hosts an extensive library with links to relevant publications: http://www.fcrn.org.uk/

Prior projects with relevant information on LCA and foods include:
http://www.lcafood.dk/
http://www-mat21.slu.se/eng/

The Life Cycle Initiative, supported by the United Nations Environment Programme (UNEP), and SETAC is coordinating international activities to support implementation of Life Cycle Management: http://lcinitiative.unep.fr/

The European Commission Joint Research Centre's Institute for Environment and Sustainability has a website on Life Cycle Thinking and Assessment: http://lct.jrc.ec.europa.eu/index_jrc

Events

The International Conference on LCA in the Agri-Food Sector takes place every two years. See:
http://www.lcafood2010.uniba.it/
http://www.agroscope.admin.ch/aktuell/02720/02722/03985/index.html?lang=de

The International Conference on Life Cycle Management takes place every two years. See:
http://www.lcm2009.org/
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Methodology for Life Cycle Assessment

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Abstract: Life Cycle Assessment (LCA) is the quantitative environmental assessment of a product over its entire life cycle, which includes raw material acquisition, production, transportation, use and disposal. This chapter provides an overview of the different applications of LCA and of the methodology. The more critical methodological issues and choices are discussed at some depth, after which examples are given of the ways in which methodological choices relate to application. The chapter is concluded by the presentation of a number of more recent trends in LCA methodology.

Key words: Life Cycle Assessment, LCA application, LCA methodology, goal and scope definition, inventory analysis, life cycle impact assessment.

4.1 Introduction

4.1.1 Thinking in life cycles

Put yourself into the position of a consumer, doing his or her weekly grocery shopping, choosing among the constantly increasing variety of food products now being offered. More or less intuitively, you will trade price for quality aspects. But you may also be concerned about the way the food was produced. For instance, were the apples produced locally or were they transported from halfway across the globe? Did the meat come from animals that were raised and slaughtered under conditions that you find acceptable? Were excessive amounts of toxic pesticides used to grow the vegetables, were water resources over-exploited and were the workers paid decently? These are all questions that reflect environmental or ethical concerns about our consumption choices, from a life-cycle perspective.

Consumers are not the only ones that may voice (and act on) such life-cycle concerns. Producers may also want to ensure that they use ingredients that were produced in an acceptable way, or that they source them from a supplier that can be trusted with regard to environmental and/or ethical concerns. And if they do, they are likely to want to tell their customers about it. Producers are also in the position to choose the ingredients, manner of production, packaging and marketing of a particular product; in other words, they develop the product, which can be done with varying levels of concern for the environment.

Producers and consumers are actors directly affecting the food chains. There are, however, other actors working to influence the food chains in a manner that demonstrates a growing recognition of life-cycle issues. First and foremost, one can think of legislators and other policy makers – one example related to agriculture if not also food, is the proposed European certification of biofuels (EC, 2008a). But sector organisations and non-governmental organisations (NGOs) are also undertaking life-cycle initiatives, such as the Marine Stewardship Council, which works globally to certify the sustainability of fisheries and fish products (MSC, 2009).

What we have discussed so far is the fact that people and organisations have increasingly started to *think* in terms of life cycles; where products come from, whether they are produced in a sustainable manner, whether they are safe, and what will happen with the waste they will ultimately become. We have also touched on the fact that life-cycle concerns may be *acted* upon and that product life cycles may be *managed*, and even *governed*. Paramount in the development of all these life-cycle approaches has been the development of *Life Cycle Assessment* (LCA), which is a quantitative method for the assessment of environmental impact related to products. This chapter is dedicated to describing LCA methodology and its relationship with the many possible applications for LCA.

4.1.2 Defining Life Cycle Assessment (LCA)

Let us start with defining LCA as the environmental assessment of a product from a cradle-to-grave perspective – everything from raw material acquisition to production, use and disposal. Inflows to the system (in terms of natural resources) and outflows (in terms of waste and pollutant emissions) are accounted for in a quantitative manner. Thus, a cradle-to-grave flow model, as depicted in the left-hand portion of Fig. 4.1, is one of the elements that define the methodology. The other element is the procedure according to which such a study is conducted (right-hand portion of Fig. 4.1). During *goal and scope definition*, what to study and how to do it is defined, while the cradle-to-grave flow model is constructed during *inventory analysis*. An *impact assessment* involves interpreting the physical flows of natural resources and pollutant emissions into metrics that are more related to environmental impact, and *interpretation* means to analyse the results and draw conclusions.

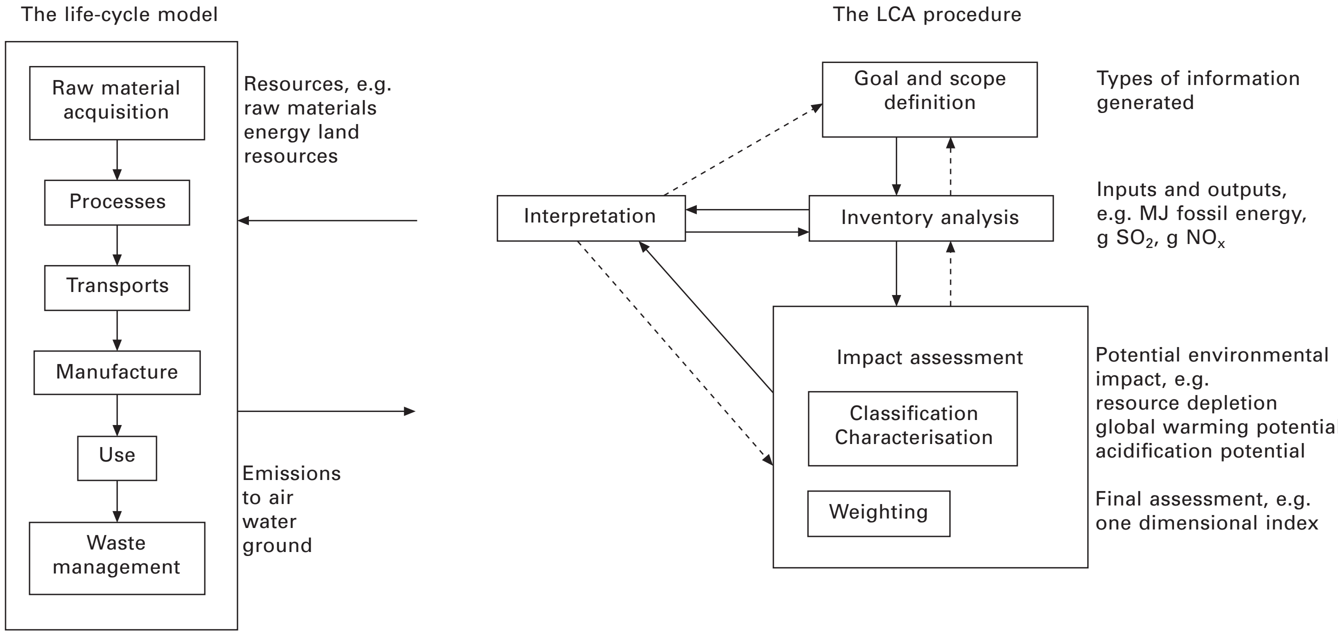

Fig. 4.1 The life-cycle model and the LCA procedure. In the *model*, boxes indicate physical processes and arrows flows of energy and matter whereas in the *procedure* the boxes indicate procedural steps and the arrows the order in which these are performed. Broken arrows indicate possible iterations. (Reproduced from Baumann and Tillman, 2004).

It is worth noting that the concept of a 'product' in the LCA context includes services, and that services may therefore also be subject to LCA. It is also worth noting that LCA, as we know it, is limited to *environmental* impact in terms of the use of natural resources, the impact on human health ('via' the environment) and the impact on the natural environment. It aspires to cover the full breadth of environmental problems. Social issues other than the impact on human health 'via' the environment, or ethical aspects such as animal welfare or child labour are not covered in the ordinary LCA concept, although they may of course be considered in life-cycle thinking, as in our initial examples. (Current efforts to include social aspects in LCA will be discussed in Section 4.5.) A key characteristic of LCA is its holistic nature – its aim to avoid problem-shifting between phases in the life cycle or between different environmental problems.

There are many different uses for LCA, including product development, support for product-related environmental policies, learning and the search for potential improvements, and communication. There are also many different ways in which an LCA study can be conducted or, in other words, there are many methodological choices to make, including the choice of system boundaries and methods to describe the environmental impact of emissions. That is why numerical results from different studies of the same product may very well differ, without any of them being 'wrong'. Although there are many different ways in which LCA studies can be conducted, these choices are not arbitrary or chosen at whim by the analyst. Instead, it should be realised that different ways of constructing models generate answers to different questions, and thus certain methodological choices are more or less well adapted to different applications. This chapter will provide an overview of the different applications of LCA (Section 4.2) and of the methodology (Section 4.3). The more critical methodological issues and choices are discussed in greater depth in Section 4.4. Section 4.5 will attempt to link methodology to applications, and Section 4.6 will present a number of more recent trends in LCA methodology.

LCA is a generic assessment methodology and is applicable to any type of product, including, but in no way limited to, food products. That is also the way in which it will be described in this chapter. The methodology issues raised by food products in particular will be pointed to, and reference will be made to other chapters in this book where these issues are discussed in greater detail.

4.1.3 LCA standards, guidelines and textbooks

Although what has been described as the first LCA study (a study on packaging for Coca-Cola) was carried out as early as 1969–70, it took until the early 1990s for massive methodology development and harmonisation efforts to take place. These efforts resulted in a series of ISO standards (ISO 14040–14043), published from 1997 onwards. This first set of standards has been subject to editorial revision and condensed into two new, currently valid documents, ISO 14040 (2006) and ISO 14044 (2006).

There are also a number of manuals and guidelines on LCA, many of which were issued before the first version of the standard, thus making important contributions to it. These include the SETAC Code of Practice (SETAC, 1993), and guidelines for environmental LCA from the Netherlands (CML/NOH, 1992), the Nordic countries (Nord, 1995), Denmark (EDIP, 1997) and the United States (US-EPA, 1993). The Dutch guidelines have been updated as an operational guide to the ISO standards (CML, 2002). Also the US guidelines have been up-dated (US-EPA, 2006). The most recent guideline is the *European ILCD Handbook* (2010). The guidelines are more detailed in their recommendations than the ISO standards, and many of them include data for impact assessment. Since several of the guidelines were written before or at the same time as the standards, terminology differs somewhat between them, and they can be difficult to read for those not well-versed in LCA terminology or those who have followed the scientific debate on LCA methodology over the years.

For LCA beginners, a textbook was published in 2004 (Baumann and Tillman, 2004), providing an orientation to LCA methodology and application, and providing exercises of differing levels of difficulty.

The description of LCA in this chapter will be based on these kinds of sources, in order to provide a baseline understanding of LCA. More recent trends in LCA methodology will be discussed in Section 4.5.

4.2 Application of LCA

LCA is very often presented as a *decision-making* instrument, the main aim of which is to underpin decisions. It is true that decision-making is an important application for LCA, but by no means is it the only one. Studies of LCA practices in companies have revealed that applications more related to *learning*, such as the search for potential improvements and risk management, are important application areas for LCA (Baumann, 1998; Rex, 2008). *Communication* is another important field of application, as revealed by, for example, the abundance of schemes for eco-labelling, environmental product declarations and, increasingly, less formalised communication of life-cycle messages.

4.2.1 Decision-making

Product development has been seen as the principal field of application of LCA since early on. One reason for this is that the focal point of LCA – the product – coincides with that of the product design process. Another is that since most aspects of the environmental performance of products are built into them during the design phase, product development is seen as decisive for achieving sustainability in industrial society. A holistic life-cycle perspective is thus a prerequisite (Baumann and Tillman, 2004).

Product development is an extensive and complex process consisting of several phases (e.g. planning, conceptual design, embodiment design and detailed design) and iterations between them. It is carried out by interdisciplinary project teams which, when it comes to the design of products for mass production in large-scale industry, may include hundreds or even thousands of members. The process is characterised by time pressures and the constant need for trade-offs between competing issues such as performance, shelf-life, aesthetics, the need for and ease of maintenance, production costs, production facilities, market constraints and legal requirements (Ulrich and Eppinger, 1995). Environmental issues need to be considered alongside all of these other issues.

The literature on eco-design stresses the importance of bringing environmental considerations into the process at an early stage, when there is a larger degree of freedom of design, and environmental concerns may have a larger influence. However, using ordinary LCA when no concrete design yet exists and data is scarce presents a problem. That is why a large number of simplified life-cycle approaches have been developed for product development purposes (see, for example, Tukker *et al*., 2000). These range from life-cycle-influenced matrices, software tools for simplified LCA with accompanying databases and LCA-derived proxies, and rules of thumb. Many of these simplified tools, however, depend on the existence of one or several full-scale LCA studies (Baumann and Tillman, 2004).

A less mature application area for LCA is that of *process development*, which focuses on the processes by which products are produced or disposed of, rather than the products themselves. This may be done with consideration for life-cycle implications of different process configurations. There are potential business drivers for cleaner production processes (e.g. if they are also more cost efficient). Policy drivers for cleaner production increasingly take a life-cycle perspective, as is evident in, for example, the EU's Sustainable Consumption and Production and Sustainable Industrial Policy Action Plan (EC, 2008b).

Particular to this field is the need to describe the focal parts of the life cycle (the processes to be developed) with a higher level of detail than what is possible in 'ordinary' LCA. Thus, in these approaches, LCA is often merged with other, more sophisticated modelling techniques, such as process simulation (e.g. Alexander *et al*., 2000; Azapagic *et al*., 2006; Bojarski *et al*., 2008) and discrete event simulation (e.g. Solding and Thollander, 2006; Löfgren, 2009). The processes in focus are modelled in a more advanced way, whereas the processes upstream and downstream are modelled with ordinary LCA methodology, often using data from a database. Optimisation techniques are sometimes used, and it is not uncommon that the more advanced models include not only environmental aspects, but technical and financial ones as well.

More advanced process modelling has been applied in the waste management field in particular. Integrated waste management models typically include a library of detailed models of different waste management processes, which may be combined in various ways. A recent overview of environmental models of waste management is provided in Finnveden *et al*. (2009).

Purchasing departments manage incoming material flows to organisations. Depending on the volumes being bought, purchasing activities can have considerable influence over suppliers and, more indirectly, over the market as a whole. Purchasing is therefore an activity with considerable potential for environmental improvement, for both public and private purchasing. These two realms, however, are different in other respects. Green public procurement is driven by policies and regulation, while incentives for green purchasing in private companies are more related to business issues such as market demands, cost saving potential, product design, business risk and company environmental policies (Baumann and Tillman, 2004).

Green purchasing is dependent on environmental information about products, which is where LCA comes in. LCA is one of the possible information tools used in supporting green purchasing. There are other tools, some of which are life-cycle-based (such as eco-labels and environmental product declarations), while others are less comprehensive (such as questionnaires and lists with banned substances) (Baumann and Tillman, 2004; Leire, 2009).

Life-cycle approaches are increasingly used in *environmental policy*. Examples from the 1990s include eco-labelling schemes and LCA studies underpinning regulation for recycling and producer responsibility. Later, life-cycle approaches have, in a broader sense, been central to several environmental policies. Examples from the European context include the directive on eco-design and policies for green public procurement. Several of the product-related policies are now being brought together and complemented, with aspirations of increased integration between policy instruments, under the Sustainable Consumption and Production and Sustainable Industrial Policy Action Plan (EC, 2008b). An example of a policy where LCA is being used in a very concrete and direct way, in terms of numerical requirement of life-cycle greenhouse gas emissions, is the proposed directive on the Promotion on the Use of Energy from Renewable Sources (EC, 2008a).

LCA can also be used as part of *technology assessment*, both to underpin policy decisions regarding emerging technologies and to support corporate strategies for new technologies. The role of LCA in technology assessment has been discussed by, e.g. Karlström (2004) and Hillman (2008).

4.2.2 Learning

The identification of environmental 'hot-spots' and the search for potential improvements were identified as important applications of LCA at a very early stage (see, for example, SETAC, 1993). These applications are not so much related to decision-making (although they may of course lead to that), as they are to learning – to getting to know the environmental strengths and weaknesses of a product from a life-cycle perspective. If the product has not been subject to LCA before, it is common to see LCA studies generate unexpected results (Baumann, 1998). Apart from physical changes to the product or the product chain, this learning may lead to procedural or organisational changes. For instance, the results from an LCA may be used to formulate rules of thumb for product development. Life-cycle knowledge about products also enable companies to answer questions from customers or NGOs regarding the environmental performance of their products, and hence better manage the risk for exposure (Rex, 2008).

4.2.3 Communication

It is probably fair to say that marketing has acted as a driver for the development of LCA methodology in general, and the standardisation of LCA in particular. In the early days of LCA, expectations were that LCA would make it possible to show that 'my product is better than yours' in a quantitative, scientific and objective manner. Such hopes were soon dashed, as environmental claims based on LCA were criticised on the grounds of ambiguity and gaps in methodology. A refined LCA methodology, as well as standardisation, was seen as the way forward. However, as the ISO standards were written to support LCA for many different purposes, they did not resolve all of the methodological obstacles to using LCA for the promotion of products. Instead, the standardisation of environmental product information for marketing purposes took other routes, such as eco-labelling and environmental product declarations (Baumann and Tillman, 2004) and, more recently, carbon footprinting. There are, however, many other ways to use LCA in marketing, such as communicating the fact that life-cycle approaches are used in a serious way or by simply communicating life-cycle information in a non-standardised format.

4.3 LCA methodology in short

As already mentioned, LCA is defined by the cradle-to-grave flow model and the LCA procedure depicted in Fig. 4.1. This section will provide an overview of what each phase in the LCA procedure encompasses. The following section will then discuss in greater detail some of the more critical methodology issues.

4.3.1 Goal and scope definition

When defining the goal of an LCA study, one states the reasons for carrying out the study and its context, such as the intended application and the intended audience. According to the ISO standard (ISO14044, 2006), it is also important to consider whether the results of a comparative study are to be made public, since the standard contains stricter rules for such studies.

It is often useful to formulate the purpose of an LCA study as a question, such as: What are the improvement possibilities in this product's life cycle? Is Product A preferable from an environmental perspective to Product B? What would be the environmental consequences of changing Raw Materials A and B to C and D?

When the goal and context of the study have been determined, its scope and hence the requirements of the modelling to be conducted must be decided. There is a long list of issues to be decided (see, for example, ISO14044, 2006 and Baumann and Tillman, 2004). Some of the more critical ones will be discussed in Section 4.4. These issues include:

- Decisions related to what to study: Which options to model, the function of the system, and the functional unit.
- Decisions related to the inventory analysis: System boundaries and allocation methods, time horizon and geographical settings, requirements on data and data quality, assumptions and limitations. Already at this early stage, it is useful to document the chosen system boundaries as a flow chart (which does not yet need to be very detailed).
- Decisions related to impact assessment: What types of impacts to include, methodology for impact assessment, and whether to weigh environmental impacts across different impact categories.
- Decisions related to the planning and reporting of the project: Whether to have a critical review conducted and, if so, of what type, and what type and format of report to make.

These are genuine *choices*, and thus are inherently value-laden. Ideally, all such choices should be made in the goal and scope definition phase, and the methodology used in the subsequent phases of the study should follow these choices. Although the methodological choices are value-laden, they are not arbitrary. As early as 1993, the SETAC Code of Practice stated that the choice of methodology should depend on the purpose of the study; this has since been emphasised many times over.

4.3.2 Inventory analysis

Life Cycle Inventory (LCI) analysis intends to build a systems model, in terms of a cradle-to-grave flow model, according to the requirements set out during the goal and scope definition. The result is an incomplete mass and energy balance for the system. It is incomplete in the sense that only the *environmentally relevant* flows are considered, which more or less include the use of resources and the emissions of substances that are considered harmful to the environment.

Briefly, the activities of the LCI analysis include the following:

(i) Construction of the flow model and documentation of it as a flowchart showing production processes, transports, use and waste management activities and the flows between them.
(ii) Data collection for all the activities in terms of:
- raw materials, including energy carriers,
- products, and
- solid waste and emissions to air and water.

(iii) Calculation of the amount of resource use and pollutant emission of the system in relation to the functional unit.

This is a rather iterative process, as is the whole LCA procedure. It is good advice to try and get an overview of the system, and to decide which parts are most important for its environmental performance, before going into too much detail. This means that the calculations may be done several times, first on a less detailed model, then on a more refined one. A first estimate, using estimated data and data that are easy to come by, but covering the entire system, helps the analyst determine which parts of the system are most important for its overall environmental performance, and hence where to concentrate the continued data collection efforts. Furthermore, during data collection the analyst learns more about the system, which allows him or her to add more detail to the model. A common beginners' mistake is to get stuck 'in one corner' of the model, and spend too much time collecting detailed data for that particular part, without knowing whether it is the one that matters the most.

Data collection is often the most time-consuming activity when conducting an LCA study, although the emergence of databases has made it easier. Nevertheless, data collection requires a degree of creativity and imagination, and many different types of data sources may have to be investigated. It is important to document the data as it is collected, especially in terms of its source and what system boundaries go with each set of data.

While inventory analysis may seem straightforward, it is usually complicated by the fact that many (if not most) technical processes produce more than one product, while LCA is interested only in the environmental impact related to *one* product. This presents a methodological problem, dealt with through the application of allocation procedures. We will discuss these in Section 4.4.

4.3.3 Impact assessment

Since many different types of natural resources are being used in industrialised society, and also since many different substances are released into the environment as pollutants, inventory results tend to consist of many different parameters; easily between 50 and 100 for a simple product, and sometimes more. The purpose of Life Cycle Impact Assessment (LCIA) is to indicate the environmental effects of the physical flows quantified in the inventory analysis. It is also designed to condense, or aggregate, the extensive information of the LCI results into fewer indicators that are easier to take in. Impact assessment is thus done in several subsequent steps, which also aggregate the information in a stepwise manner, as depicted in Fig. 4.2.

According to the ISO standard 14040 (2006), conducting an impact assessment is mandatory if a life-cycle study is to be called an LCA study. Without an impact assessment, an LCA study is referred to as an LCI study. Furthermore, there are certain phases of the LCIA that are mandatory according to the standard, namely those based on natural science. Other phases, which are more based on values, are optional (and even barred in LCA studies that are 'intended to be used in comparative assertions intended to be disclosed to the public').

When impact categories have been selected and it has been decided how to indicate them and what models to use for that process, a procedure called *classification* is conducted. This simply sorts the inventory parameters according to the type of environmental impact categories they contribute to. As an example, Fig. 4.2 shows three such impact categories. The inventory parameters shown to the left have been classified according to their contribution to the three impact categories shown: acidification, eutrophication and global warming. In the next step, *characterisation*, the relative contributions of the emissions and resource consumption to each type of environmental impact category is calculated. For example, all greenhouse gas emissions are aggregated into one indicator for global warming and all acidifying emissions into one indicator for acidification. Such calculations are based on scientific models of cause–effect chains in the natural system. Some of the characteristics of these models will be discussed in Section 4.4.

The numerous result parameters of an LCI may be aggregated into a

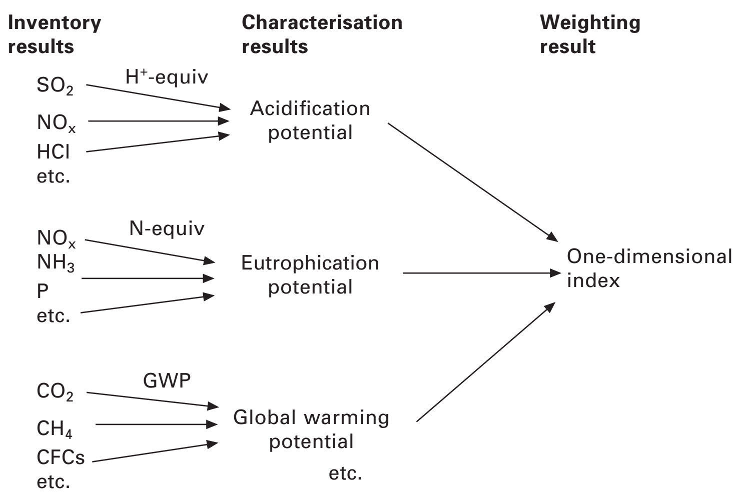

Fig. 4.2 Illustration of the stepwise aggregation of information in LCIA. (Reproduced from Baumann and Tillman, 2004.)

limited number (typically 7 to 15) of impact categories. These are sometimes aggregated even further through the use of formalised, quantitative weighting procedures. Such weighting across impact categories cannot be done based solely on natural science; it also relies on values. Weighting methods can be described as a 'yardstick' with which all environmental problems are measured. They are based on values and preferences regarding environmental issues expressed in society in various ways. For example, political environmental goals may be used to create a weighting system, or monetarisation methods borrowed from cost–benefit analysis may be used.

4.3.4 Interpretation

Interpretation is the last phase in the conduct of an LCA study. It includes the identification of the environmentally significant issues in the studied product life cycle. It also includes an evaluation of the reliability of the study, in terms of *completeness*, *sensitivity* to uncertainties in data and methodology, and *consistency* in relation to the goal and scope definition. Finally, this is the phase where conclusions are drawn, limitations are stated and recommendations made.

4.4 Critical methodology choices

As already mentioned, there is a long list of methodological choices to be made when conducting an LCA. While there is insufficient space in this chapter to discuss them all, some of the more critical ones have been selected for review. As an informed reader of an LCA report, one needs to be aware of these issues, and be prepared to question the choices made. One also needs to be aware that different methodological choices will result in different numerical results.

4.4.1 Functional unit

The functional unit is a central concept in LCA. It expresses the function of the product being studied and with it goes a reference flow, to which all other flows in the system are related or normalised. It is also the basis for comparison in comparative studies, which makes the choice of a functional unit a critical one.

For instance, when comparing packaging, such as one-litre plastic bottles to 250 mL aluminium cans, it is not meaningful, or perhaps 'fair', to compare the one litre packaging to the 250 mL packaging on a piece-by-piece basis. Instead, the LCA logic states that the function of the packaging is to deliver a certain volume of the packaged goods to the consumer. The functional unit should then be stated as a volume unit. The exact size of this is arbitrary – it could be one litre, 1000 litres or one gallon – but it must be the same for all compared alternatives.

The reference flow is defined once the functional unit has been decided. In our example, if we decide on the containment of one litre as a functional unit, the reference flow would be one plastic bottle and four aluminium cans, respectively. In addition, *where* in the flow chart this flow takes place also needs to be stated. In our example, it would be one litre *delivered to the consumer*.

Different products compared in LCA studies can basically fulfil the same function, but do so in qualitatively different manners. For instance, the barrier properties of the plastic bottles and the aluminium cans in our example may differ. One can also think about different types of flooring, such as PVC flooring and stone flooring, where the stone flooring may last two or three times as long as the PVC flooring. Qualitative differences in function between compared alternatives can sometimes be included in the functional unit, especially if they can be expressed in quantitative terms. A functional unit of $m^2 \times$ years for the flooring will account for their different lifetimes. Their different aesthetic properties, however, cannot be accounted for in the functional unit, but would simply have to be accepted.

The choice of functional unit is perhaps more critical for food products than many other types of products. What is the function of food? While it obviously delivers nutritional value, does it do so in terms of nutritional energy or content of proteins and vitamins? Or is the function about pleasure? Sometimes, as in the case of many 'light' products, it appears as though the value of food products is to deliver as much pleasure as possible with as little nutritional value as possible. In contrast, in situations without food abundance, nutritional value is the key function. Clearly, the function of food products is context-related. In LCA practice, simple metrics such as kilograms or litres of food seem to be the most commonly used functional units; this works so long as food products of similar type are being compared.

4.4.2 Attributional and consequential LCA: System boundaries, allocation and choice of data

As mentioned earlier, the 1993 SETAC Code of Practice had already recognised that methodological choices should be made with regard to the purpose of the study. In later efforts to sort out *how* they should depend on the purpose of the study, two fundamentally different types of LCA were distinguished: attributional and consequential. While these are the terms used in more recent literature, similar concepts such as accounting versus change-oriented have also been used.

The attributional type of LCA answers questions such as 'What environmental impact is associated with this product?' The consequential type of LCA compares the environmental consequences of alternative courses of action, answering questions such as 'What would happen if … ?'

Knowing what type of LCA is being undertaken helps when setting system boundaries and choosing the type of data to represent the system. Which type of methodology goes with which type of LCA is summarised in Table 4.1. The subsequent sections will take us through the details in the table.

Any LCA study needs to be framed in terms of time and geography. System boundaries in a more strict sense need to be drawn between the technical system modelled in the inventory and the natural system modelled in the impact assessment. The flows passing through this boundary are called elementary flows, and constitute the inventory results. Since different technical systems are connected, there is also a need to draw boundaries between the studied part of the technical system (the life cycle) and connected parts of the technical system (Tillman *et al.*, 1994).

Drawing the boundary between the technical system and the natural system is often straightforward. Inflows that have not been subject to any technical transformation but are directly drawn from nature (e.g. ore, crude oil) pass this boundary, as emissions to air and water at the points where they are no longer under human control. For agricultural and forestry activities, however, the boundary between the natural system and the technical system is less obvious and needs to be specified in greater detail. This is because while farmland and managed forests are indeed ecosystems, they are simultaneously subject to extensive technical manipulation. The same applies to land or water areas used for 'trapping' flowing energy resources, such as hydropower and wind power. How the system boundary is drawn with regard to land use affects not only inventory modelling, but also impact assessment of land use. This is further discussed by McLaren (Chapter 3, this book). In addition, for waste deposits it is not obvious where to draw the boundary; it has been suggested that they should be regarded as part of the technical system, and that their emissions should be accounted for as elementary flows, at least up to a certain point in time (Tillman *et al.*, 1994, Finnveden *et al.*, 1995).

As mentioned above, system boundaries also need to be drawn within the technical system, since the isolated product life cycles of LCA are model

Table 4.1 Methodological characteristics of attributional and consequential LCI models (adapted from Tillman, 2000)

Characteristic	Type of LCA Attributional	Consequential
System boundaries	Additivity Completeness	Parts of system affected
Allocation procedure	Reflecting causes of system Partitioning	Reflecting effects of change System expansion
Choice of data	Average	Marginal

constructs. In reality, industrial systems are networked in many different ways. For instance, many industrial processes produce more than one product, while LCA is usually interested in only one of them (*the multi-output problem*). Most types of waste treatment processes, for example, treat waste that consists of a variety of products, while LCA tries to isolate one of them (*the multi-input problem*). Finally, the same material may be used in several consecutive products, such as when a product is recycled into another product (*open loop recycling*). Together, these three cases constitute what, in LCA terminology, is called the allocation problem, the solution to which is completely different in the attributional and consequential approaches.

Attributional LCA strives to account for the environmental impact of a product life cycle in a manner that is as full and complete as possible, in the sense that all processes, from cradle to grave, are included, and as 'clean' as possible in the sense that only the processes strictly belonging to the cradle-to-grave system are included. Theoretically, if the results of LCA studies of all the products in the world were multiplied with production volumes and added together, the sum should equal the total environmental impact of the entire world. If any processes in the studied product life cycle are shared with the life cycles of other products (such as a multi-output process), their environmental impact is split (allocated) between the products.

On the other hand, consequential LCA is concerned with describing the environmental consequences of alternative courses of action. This means that processes not affected by the contemplated action need not necessarily be included in the system model, even if, strictly speaking, they belong to the cradle-to-grave life cycle. This also means that allocation is avoided. When the system includes processes with multiple functions, the model is expanded to accommodate the additional function, thus more fully describing the effect of the contemplated action.

An example of this would be the consideration of whether to install a heat pump in a dairy, to deliver heat to a district heating system. The system boundaries, according to a consequential LCA, would be outlined as in Fig. 4.3. The recovered heat would be followed up to the point where it is delivered to the district heating system, and the avoided environmental impact from producing the same amount of heat (for the district heating system) in some other way would be credited to the studied (dairy) system.

With an attributional approach, the avoided heat production would not be included. Instead, the dairy, which now has two products, milk *and* heat, would be rendered virtually mono-functional through the allocation of its environmental impact between the two products.

The example brings us to another difference between attributional and consequential LCA, that between using data describing how a system behaves on average or data describing how it behaves on the margin, as a response to changes in demand and supply.

What sort of heat production would be affected by the instalment of the heat pump? Let us assume that the district heating system is fed by a mix

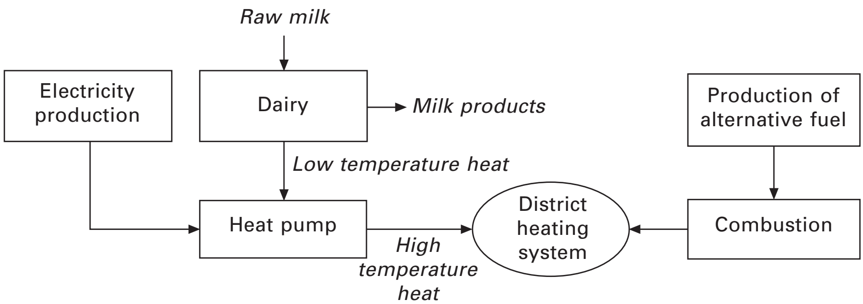

Fig. 4.3 Example of system expansion. The studied dairy is credited with the avoided production of heat elsewhere. The district heating system is not part of the modelled system, but the heat flows are followed up to a point where equal functions are delivered to the district heating system.

of energy sources, such as surplus industrial heat, biomass and natural gas. These would not be equally affected, but the most expensive source of energy, presumably the natural gas, would probably be replaced first. In consequential LCA, the system would be credited with such *marginal* heat production. In addition, all other affected flows would be modelled as affected on the margin. For example, the electricity used to operate the heat pump would be modelled as the marginal electricity supply. Use of data representing effects on the margin introduces an additional choice for the analyst, namely how do decide which technology is the affected one. It may be the most expensive one, as in our district heat example, but the marginal technology may also be decided by other factors, such as a customer's relationships with suppliers (it may be easier to buy more raw material from an established supplier than to find a new one).

In attributional LCA, data representing the average (as opposed to marginal) production is always used. In our example the electricity used to operate the heat pump would be modelled as supplied by the average electricity production mix.

The debate on how to deal with allocation problems, through system expansion or partitioning, has been long and heated and remains ongoing; Finnveden *et al.* (2009) provide a review of the different viewpoints. The ISO standard 14044 (2006) is of limited help, since it is not self-consistent. While, on the one hand, it acknowledges that system boundaries should be chosen to be consistent with the goal of the study, on the other it recommends an order of preference for how to handle allocation problems, without regard for the goal definition. The standard recommends that allocation should be avoided whenever possible, either through increasing the level of detail of the model, or by expanding the system. Allocation may be used only if this cannot be accomplished, and should preferably reflect physical causal relationships between inputs and outputs. Only as a last resort other relationships between the functions of the system, such as relative economic value, may be used as a basis for allocation.

4.4.3 Impact assessment

As already mentioned, the purpose of the impact assessment in LCA is to indicate the environmental effects of the physical flows quantified in the inventory analysis while aggregating the LCI information into fewer indicators. LCA practitioners use 'ready-made' impact assessment methods developed by experts. It is therefore important to be aware of their underlying logic, assumptions and limitations. An overview of these will be provided in this section.

The nature of life-cycle inventory (LCI) sets certain limitations to the way that life-cycle impact assessment (LCIA) can be done. LCI sums emissions and resource consumption occurring along the entire life cycle and the information about *where* and *when* the emissions and resource consumptions take place is usually not carried through to LCIA. Furthermore, the LCI results are provided in relation to the functional unit, thus representing only a very small portion of the total environmental load. Compared to a full-blown Environmental Risk Assessment, there is a significant amount of information missing, such as total emissions volumes, background pollution, and different sensitivity to pollution in different geographical locations and points in time. For this reason, LCIA can indicate only the *potential* contribution to actual impacts. The LCIA models seek to provide the best estimates for the potential impacts; it is worth being aware that the best estimate approach may imply a conflict with the precautionary principle, and that irreversible or other serious damages can be overlooked (Finnveden *et al.*, 2009).

A whole chain of events begins when a pollutant is released into the environment. For instance, when the pollutant is dispersed, it can react chemically, then reach an organism, which is then eaten by another organism, which in turn gets its reproduction disturbed, and so on. One important choice in designing and selecting impact assessment models is, therefore, how far to follow such cause–effect chains in the environment. In mid-point characterisation models, the indicators often represent an effect quite early in the chains. Typical indicators in this type of characterisation are potential contributions to global warming (CO_2 equivalents), acidification (SO_2 equivalents), eutrophication (PO_4 equivalents) and photo-chemical oxidant formation (ethylene equivalents). There are also indicators for toxicity and resource use. Results expressed as mid-point indicators may be evaluated as such, although they may be difficult to relate to. ['How many units of global warming potential are good for you?' is a highly relevant question that has, in fact, been asked by people confronted with the results of LCA studies.] For this reason, and perhaps also to aggregate the results even further, the cause–effect chains can be followed further, to indicators of impact on areas of protection (end-points), such as human health, biodiversity, ecosystem production capacity and abiotic resources. While end-point indicators are of higher relevance because they are easier to relate to, they also introduce more uncertainty into the models since more causal relationships are included.

According to the ISO standard 14044 (2006), classification and characterisation – whether of the mid-point or endpoint variety – are mandatory elements of impact assessment. They are also the elements based on natural science (although not devoid of estimates and judgments). The assessment may be taken further, through assessing the contributions to the environmental-impact categories on a single scale. This produces one single number as the result of the entire study. There are ready-made weighting methods for doing this, typically including the sub-steps of grouping, normalisation and weighting. Weighting has always been a controversial topic in LCA, since it cannot be based on natural science. Instead, methods from social science and economics are used to reveal preferences and societal judgements regarding the severity of different types of environmental impact. Several types of such 'value sources' exist, and they do not necessarily generate the same results. Borrowing methods from environmental economics, monetarisation has been used to create weighting methods in LCA. Another method is to use panels, to seek their advice on weighting different types of environmental impact against each other. A third method is using political goals and the distance to these targets in order to compare the relative severity of different environmental problems against each other.

4.5 Examples of different applications' demands on methodology

Throughout this chapter it has been stressed that methodology needs to be chosen in accordance with the purpose of the study. This section will further explore the relationship between application and methodology.

The choice between an attributional and a consequential LCA approach has been, and remains, subject to debate among LCA experts. Finnveden *et al.* (2009) provide a recent review of the different perspectives: In summary, most LCA experts seem to agree that LCAs intended to underpin decisions should be consequential, whereas attributional LCA can be used (or should be used, the positions on this differ) when there is no specific decision at hand. However, there are also those arguing that all LCAs are, in one way or another, intended for decision-making and thus should be consequential (see, for example, Weidema, 2003).

Bringing these arguments further, to LCA applications, those that include decision-making are best supported by consequential LCA – the design of policies, products or processes. For those applications more related to learning, on the other hand, consequential or attributional LCA may be used, depending on what author is followed. Perhaps it is fair to say, following Ekvall *et al.* (2005), that one may learn from both approaches.

Purchasing inherently involves decisions, and according to the logic described above, information intended to support it, such as eco-labels, carbon footprints and environmental product declarations, should be based on consequential LCA. However, such information schemes are usually based on LCAs of the attributional kind. Arguments for this, which are further detailed in Tillman (2000), are that the LCA methodology for such applications needs to be broadly accepted among those using the schemes, and that even though purchasing involves choices, the sender of information does not know what the alternative choices are. Moreover, for some of the information schemes (such as environmental product declarations) additivity is an important feature, and is only rendered by an attributional approach.

Impact assessment is another methodology feature that different applications place different demands on. Some applications need a very short, simplified message, such as a single number or, as in the case of eco-labels, an 'approved stamp'. This is when the users of the life-cycle information have limited environmental knowledge and/or are under time pressures. Product development is one such application, and the purchasing of daily commodities is another. It is no coincidence that most LCIA weighting methods were developed for product development purposes. In other cases there is more time to digest the results, the recipient may be more environmentally competent, and the value judgements built into weighting may be unacceptable. Such an example would be support for environmental policy.

4.6 Recent trends in LCA

LCA is currently evolving along several lines. Since it is such an ambitious method, there have always been efforts to simplify LCA, more recently in terms of carbon footprinting. A trend in the opposite direction has been the expansion of the aspirations of LCA by merging it with other types of tools and methodologies. Input–output LCA, where LCA is merged with macroeconomics, is one such example. Another is social LCA, which widens the scope of LCA to also include social impacts. In addition, the use of more refined process models in LCA models, as discussed in Section 4.2, belongs to this category. Finally, there are ongoing efforts to refine the way LCA does what it always has done. Better impact-assessment models are being developed, more databases are being set up, and so on. The following section will discuss some of these development trends. The chapter will then end by identifying other sources of information regarding current developments.

4.6.1 Carbon footprinting

Carbon footprinting and other similar concepts have gained significant momentum in recent years, clearly spurred by the urgency of the climatechange agenda. Carbon footprints are the accumulated life-cycle greenhouse gas emissions from entities such as nations, corporations, individuals or products. Of interest here is *product* carbon footprinting, which in all essence is LCA but with respect to greenhouse gas emissions only.

There have been many different initiatives related to product carbon footprinting, and for several reasons food products have been at the centre of these developments (see further Lillywhite, Chapter 14, this book). Since product carbon footprints are used for market communication, comparability between products is essential. That is why several standardisation projects have been launched. The British Standards Institution published a Publicly Available Specification in 2008 (PAS 2008) and The World Business Council for Sustainable Development recently issued a draft standard for the accounting and reporting of product life-cycle greenhouse emissions, for stakeholder review (WRI/WBCSD 2009). With regard to LCA methodology, both of these standards explicitly recommend an attributional LCA methodology.

4.6.2 Input–output LCA

Input–output tables are part of the national accounts. They state, in monetary terms, how much goods and services the different sectors of the national economy bought from each other. LCA researchers realised that this information could be used to estimate LCI data if combined with information on average physical resource consumption and environmental emissions for the sector. Since input–output tables and LCA were constructed for different purposes, however, this is not a perfect match, and several ways to hybridise them have been suggested. Suh and Huppes (2005) provide a review of the different approaches. The most influential of these is perhaps the integrated hybrid approach presented by Suh in 2004.

Not only can input–output LCA help *estimate* LCI data, it is also advocated as a way of forming a more *complete picture*. The argument here is that when conducting an ‘ordinary’ LCA, it is easy to miss certain processes, whereas the input–output tables do not miss anything. The completeness ambitions are very much in line with the accounting aspirations of the attributional type of LCA. In addition, other features of the input-output tables, such as their average data, align with the attributional LCA approach.

A third application of input–output LCA is that of *life-cycle studies of the consumption of products* in an entire economy – such as a country, a city or even the whole of the European Union – to underpin policies on sustainable consumption and sustainable product policies. Several such studies have been used to identify what type of consumption has the largest environmental impact. Tukker and Jansen (2006) provide a review of eleven such studies, and conclude that although the methodology between different studies differed extensively, they all point to the same product groups as the main contributors – namely food, housing and transportation.

4.6.3 Expanding the scope of LCA

The LCA community has long had the ambition of expanding the scope of LCA to cover not only environmental aspects, but also the other two 'pillars of sustainability' of social and economic aspects. To this end, efforts have been undertaken to develop methodologies similar to LCA in their cradle-to-grave approach, but instead accounting for social aspects and costs, respectively.

As regards cost, the concept of Life Cycle Costing (LCC) has existed since the 1960s; it has thus been tempting to believe that such an analysis could be used as an economic parallel to LCA. This has proven not to be the case, however, primarily because LCC determines cost from the perspective of a single decision-maker. Efforts to develop a life-cycle costing methodology that could parallel LCA were undertaken by many individual researchers, some of whom collaborated in a SETAC working group and developed Environmental Life-Cycle Costing and presented it in a recent report (Hunkeler *et al.*, 2008). Guidelines are currently under development. This methodology is intended to be applied in parallel with LCA.

The methodology for LCA accounting for social impacts (social LCA) proved to be an even greater challenge (for a review see Jørgensen *et al.*, 2008). Guidelines for social LCA have recently been published by the UNEP–SETAC life-cycle initiative (2009). Among the motives for developing such a methodology was making LCA more relevant to developing countries. The guidelines establish a framework for social LCA that aligns with the ISO LCA framework as far as possible, but it is clear that many challenges remain, particularly when it comes to impact assessment. For instance, it is not yet clear what impact categories to consider, and even less how to model the relationship between the data collected during inventory and the social effects in the selected categories. It is not surprising that the guideline document ends with presenting a research agenda.

4.7 Sources of further information and advice

A number of information sources for LCA beginners and those interested in established LCA methodology have been presented in Section 4.2. Further information can also be found on the following websites, which are also useful for those interested in following current developments:

- Since the early 1990s, the Society for Environmental Toxicology and Chemistry (SETAC) (http://www.setac.org) has been a main actor in the development and harmonisation of LCA methodology. SETAC arranges conferences and international working groups.
- Somewhat later, the United Nations Environment Programme (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC) launched their Life Cycle Initiative (http://lcinitiative.unep.fr) to enable

users around the world to put life-cycle thinking into practice.

- The European Platform on LCA (http://lct.jrc.ec.europa.eu/eplca) is a project of the European Commission working towards a European reference Life Cycle Database (ELCD) with European scope inventory data and an International reference Life Cycle Data system (ILCD) data network, providing a registry for quality-assured life-cycle inventory data. It is also producing guidelines on LCA.
- The US EPA maintains a comprehensive LCA portal http://www.epa.gov/nrmrl/lcaccess/index.html), which, among other things, provides guidelines and portals to LCI data sources and available LCA resources.

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5

Challenges relating to data and system delimitation in Life Cycle Assessments of food products

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Abstract: This chapter describes the major challenges relating to LCA data and system delimitation when carrying out LCAs on products from the food industry. Food is characterised as a product where emissions related to the feedstock (agriculture) and electricity in the use (and processing) stage can be identified as hotspots. Calculation of environmental impacts from agricultural products and electricity are highly sensitive to the applied approach for modelling; consequential versus attributional. Both consequential and attributional modelling are demonstrated in different situations, and pros and cons of the two approaches are discussed. Based on this, consequential modelling is proposed as the preferable option because it takes into account cause–effect relationships, it does not ignore the effects of co-products and it maintains mass- and substance-balance of processes. Since consequential modelling seeks to be more accurate, the risk of misleading decision support is minimised in consequential modelling. The disadvantage of consequential modelling is that no commonly accepted marginal supplies of products for different countries/regions exist. The same applies for attributional modelling, but national/regional market averages are more fixed and less sensitive to assumptions introduced by the individual LCA practitioner.

Key words: food industry, system delimitation, co-products, Life Cycle Assessment (LCA).

5.1 Introduction

Food is one of the most significant product groups regarding environmental impacts on climate and land use (Tukker *et al*., 2006; Weidema *et al*., 2005). LCAs of food product systems are characterised by being sensitive to the approach and assumptions relating to system delimitation; namely, the adoption of the consequential approach versus the attributional approach to modelling in the life-cycle inventory.

The difference between consequential and attributional modelling can very briefly be described in two differences: (i) consequential modelling includes the processes that are actually affected (sometimes termed 'marginal supply') as a consequence of the decision the LCA is aiming to support, and attributional modelling includes the market average supply, and (ii) in consequential modelling, co-product allocation is avoided by system expansion, and attributional modelling allocates by using allocation factors (Ekvall and Weidema, 2004).

Food product systems are characterised by having environmental hotspots related to the feedstock production in the agricultural sector and in the use stage where electricity is used for food cooling and preparation (Nielsen *et al*., 2005), and they are characterised by often having several by-products, i.e. many food processing processes are multiple-product output processes. The differences between actually affected processes and average supply, especially for agricultural products and electricity, are significant, and since food-product systems are often associated with by-products, the choice between the consequential and the attributional approach to modelling becomes specially significant for food products. Therefore, this is a major issue in this chapter.

5.2 System delimitation in agricultural Life Cycle Assessments (LCAs)

This section focuses on which processes should be included when defining the system boundaries for crop cultivation. Most agricultural LCAs performed to date include the current actual piece of land cultivated in order to produce a given reference flow, e.g. one hectare cultivation of agricultural land in Sweden in order to supply the Swedish food sector with crops. The inputs and outputs related to the cultivation of one hectare (ha) in one year (yr) using this modelling are illustrated in Fig. 5.1. This way of modelling is used in attributional modelling.

Striving towards cause–effect modelling, there are several problems related to attributional modelling:

- *Actual affected supply (marginal supply)*: If, for example, the Swedish food sector is not specifically demanding crops cultivated in Sweden, it is likely that suppliers outside Sweden are more competitive (because several suppliers provide crops to the regional/global market, and it would therefore be a coincidence if Sweden is the most competitive). Therefore, the Swedish agriculture may not be a good representative.
- *Land constraints*: If the available agricultural land is constrained, a

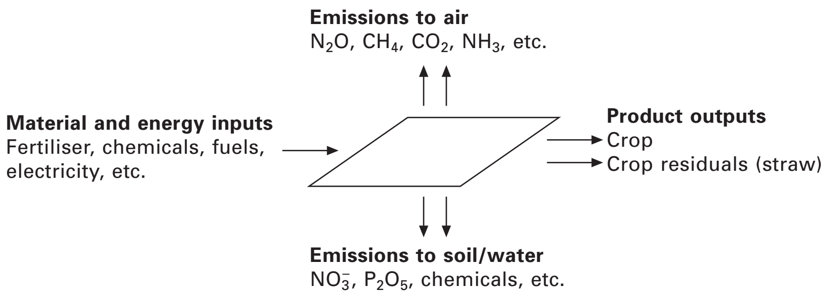

Fig. 5.1 Illustration of the included inputs and outputs related to the cultivation of 1 ha yr in attributional modelling.

change in the demand for a certain crop cultivated in Sweden will have its major effects outside Sweden.

- *Cultivated area versus intensification*: A change in agricultural production can be achieved either by a change in the cultivated area or a change in intensification (or a combination).
- *Emissions from natural land*: Even natural vegetation (here defined as non-productive land) is associated with emissions. Therefore, the decision to cultivate a piece of land in a certain period has the following resulting emissions during this period: emissions from cultivation minus emissions from natural vegetation.
- *Land transformation*: A change in cultivation of a piece of land in a certain period will cause a change in the pressure on natural land being transformed into agricultural land.

The following sub-sections address the abovementioned problems, which are not addressed in attributional modelling.

5.2.1 Actual affected supply (marginal supply)

In attributional modelling, the average supply is most often included, and sometimes the major used local supply is included. An example is vegetable oil, where most existing LCAs in the EU focus on rapeseed oil produced in the EU (Schmidt and Weidema, 2008). When rapeseed oil is specifically demanded, it is obviously desirable to include rapeseed in the study. However, in many cases several types of vegetable oils can be used for the same purpose (Schmidt and Weidema, 2008) and companies often shift between substitutable oils, depending on which oil is the cheapest. The term 'substitutable' refers to products that have the same obligatory product properties as well as fulfilling other requirements defined by buyer requirements within the market segment. The latter cannot be defined in general terms here because it is defined by the buyers in the market. For more information, see Weidema and Ekvall (2009). If the specific oil is not known or if the used oil depends on which oil is the cheapest, then the most likely oil to be used is the cheapest (which is also the most competitive). According to Schmidt and Weidema (2008), the cheapest oil until year 2000 was rapeseed oil. After 2000, palm oil has been the cheapest source of vegetable oil. Thus, within the market segment for which the oils are substitutable, palm oil is the most likely to be affected. From 1990 to 2006, the production of rapeseed oil in the EU increased from 2.7 mill tonne to 5.8 mill tonne. During the same period, the importation of palm oil increased from 1.9 mill tonne to 6.7 mill tonne (FAOSTAT, 2009). This also underpins that there has been a shift from rapeseed oil to palm oil as the major oil where the two are substitutable.

The procedure to identify the marginal supply is firstly to identify the relevant market segment. In the vegetable oil example that follows, this includes the identification of which oils belong to the substitutable oils that can potentially be used. If rapeseed oil is specifically demanded, then the market segment is simply just rapeseed oil. Secondly, it is to check if any of the commodities within the market segment are constrained. An example of a constrained vegetable oil is soybean oil, which is constrained by the demand for its co-product; soybean meal. i.e. It is the demand for soybean meal that determines the production volume of soybean oil mills, not the demand for soybean oil (Schmidt and Weidema, 2008). Thirdly, the most competitive supplier among the flexible supplies (non-constrained) within the market segment must be identified. This can be done either using price information, such as the FAOSTAT price database (FAOSTAT, 2009) or UN trade database (UN, 2009), or by using historical production statistics, or outlook statistics in order to identify which supplier that has, or is predicted to, increase its production volume most. Examples can be found in Schmidt (2007, p. 19). Alternatively, marginal suppliers can be identified using economic modelling and price elasticities, see Kloeverpris *et al.* (2008). It should be noted that significant uncertainties may be related to the identification of marginal suppliers of crops. In addition, the identification of the marginal supply is highly dependent on the temporal scales of the system as the capacity for expanding cultivation may be fully utilised in different regions over time, e.g. mid-term changes in demand may be produced by in a certain region, but this region is fully utilised (constrained) when it comes to long-term changes in demand. An example of the latter could be palm oil; there is a tendency that oil palm cultivation, which was previously found in Malaysia, is now at a larger scale found in Indonesia (Schmidt, 2007, p 19). This is partly due to the fact that land for further expansion is becoming scarce in Malaysia.

5.2.2 Land constraints

In some regions, especially in EU countries, there is no more land available for expansion of the agricultural area. In many EU countries, the agricultural area has been decreasing during the last decades (due to protection of natural land, increases in urban and built-up area, etc.). Of course, changes in prices of commodities may cause shifts between agricultural and forestry production systems, but in the end such changes will just export the effect outside the constrained region: see the following explanation. If cultivation of a certain crop is increased, this will take place at the expense of another crop or forest (Schmidt, 2008). This displaced crop will most likely be the crop related to the lowest margin. An example is rapeseed cultivated in Denmark. Agricultural land in Denmark has been slightly been decreasing in the last decades (FAOSTAT, 2009), thus the land can be said to be a constraint and thus, the cultivation of rapeseed in Denmark will take place at the expense of other crops. Schmidt (2008) has identified the marginal crop to be displaced as spring barley in Denmark. The product system now accounts for the demanded rapeseed minus the displaced spring barley. Thus, we need to compensate for the 'missing' barley. Schmidt (2008) identifies Canada as the marginal supplier of barley. The marginal supply of barley may also be from a mix of different regions, as described in Kloeverpris *et al*. (2008).

5.2.3 Cultivated area versus intensification

A change in the demand for a certain crop in a certain region can be met either by a change in the cultivated area, by a change in intensification, or by a combination of the two (Schmidt, 2008). The way the change in production will be achieved can be based on price information, i.e. which is the cheapest, it can be based on historical production statistics or outlook statistics, or it can be based on other market information such as regulation of agriculture, and demand for crops cultivated using a certain technology/practice, e.g. organic crops.

A change in intensification can be achieved in several ways, examples of parameters in play are fertiliser input, weed control, irrigation, soil treatment, seed improvements, introduction of genetically modified organisms, etc. However, a common way to vary yields is by changing the fertiliser input (Schmidt, 2008). The identification of the actual affected way of changing yields as a consequence of a change in demand is related to uncertainties.

5.2.4 Emissions from natural land

It is evident that natural land is also associated with emissions such as N_2O, CH_4, CO_2, and nitrogen and phosphate compounds. Some emissions may be negative (e.g. in the case of carbon sequestration). The decision to cultivate a piece of land in a certain period will cause the directly-induced emissions (as illustrated in Fig. 5.1), but at the same time the emissions from natural land are avoided. Thus, these should be subtracted from the direct emissions. Generally, emissions from natural land are not significant compared to the emissions from cultivation; examples can be found in Schmidt (2007, p. 216).

5.2.5 Land transformation

According to IPCC (2007), 17% of the GHG emissions in 2004 originated from land use change, mainly deforestation. Deforestation is caused mainly by the demand for timber and the demand for agricultural products (grassland for cattle and crops). Often land transformation is not included in LCAs. One reason for this is because it is not straightforward to establish a relationship between the occupation of land which is proportional with annual yields/ functional unit and the transformation of land. The problem is to identify the consequence of an additional year of land occupation. In attributional modelling, the most common way to include land transformation is to assume a certain number of functional units supported by the initial transformation, e.g. cultivation in 100 years. However, this is highly variable; some of the oldest cultivated fields are thousands of years old while some cleared land supports only a few years of cultivation (e.g. slash and burn). Therefore, this approach is likely to introduce some arbitrary and at the same time very important assumptions. In addition, this approach focuses on the number of functional units in the past and future to be allocated to an initial transformation rather than the consequences of occupying a piece of land in a certain period of time, which is the real interest of an LCA.

A more desirable way to solve the problem of assigning land transformation to the functional unit is to assume that all occupation of agricultural land contributes to the current pressure and thereby transformation of land. Then, based on the total global transformation (measured in ha) of land divided by the total global occupation of agricultural land (measured in ha yr), a global average expression of what the land transformation (ha) is per unit of occupation (ha yr) can be calculated.

Care should be taken when identifying the determining activity in transforming primary forests into degraded/secondary forests. When comparing the disappearance rate of primary forests with the rate of agricultural expansion in Brazil and Indonesia in FAO (2006) and FAOSTAT (2009), it appears, that between year 2000 and 2005, primary forests have been disappearing 3–4 times faster than the expansion of agricultural land. This indicates that logging is the activity that determines the rate of degradation of primary forests. However, other studies identify other determining activities, e.g. Wassenaar *et al*. (2007) and Morton *et al*. (2006) point to the need for pasture and cropland as the driving forces of clearing the forest in the Amazon region.

5.2.6 Pros and cons of consequential and attributional modelling in the agricultural stage

Attributional modelling can be illustrated by the system in Fig. 5.1, and consequential modelling can be illustrated by Fig. 5.1 plus the preceding sections on additional parameters to be included. A conceptual figure (decision tree) of this is presented in Schmidt (2008). The proposed way of modelling in the preceding sections are all related to considerations in consequential modelling. The inclusion of the proposed aspects of modelling contributes to increased accuracy in identifying the actual affected processes, making the LCAs more suitable for decision support. At the same time, it also adds uncertainties to the LCA results. According to Schmidt (2008), these uncertainties are, in particular, related to the identification of marginal crops and suppliers, prediction of how production is increased (area or yield) and modelling of emissions when increasing yields. In this respect it should be kept in mind that the same uncertainties apply to attributional modelling, because the introduced variable parameters do not disappear in reality as they are left out of the way to model a product system.

5.3 System delimitation in electricity LCAs

In attributional modelling, it is common to use national or regional grid mix to represent electricity. In consequential modelling, the marginal supply of electricity is identified and included. The reason for including the marginal supply of electricity is that this better represents what is affected if anything is changed throughout the product chain, i.e. to produce the product or not, to save energy or not, etc. An electricity system is often composed of a mix of several technologies. Some technologies will typically not react on a change in demand, because they are constrained. Examples are hydropower if the potential is fully utilised in the region, as in Norway and Sweden, wind power, to some extent nuclear power (which is typically determined by political decisions), and waste incineration (which is determined by the amount of waste) (Weidema, 2003). Other, technologies will typically not react because they are not the cheapest, such as fuel oil and biofuels (Weidema, 2003). Often, the most competitive and non-constrained technologies are coal- and natural gas-based electricity (Lund *et al.*, 2009). However, in some regions, marginal electricity may be almost fully based on renewables (Schmidt and Thrane, 2009).

There are two types of marginal electricity: the production marginal and the build marginal. The production marginal represents the short-term marginal generated within existing capacity as a consequence of a change in demand, and the build marginal is the long-term marginal representing the type of capacity that will be installed or phased out as a consequence of a long-term change in demand. The short-term marginal can be identified as the yearly average marginal using energy system models. Examples can be found in Lund *et al.* (2009) which analyse the Danish/Nordic grid. When identifying constrained technologies for the production marginal, typical constraints are reservoir water available for hydropower, wind for wind power, and if the capacity is fully utilised, nuclear. The identification of the long-term marginals can be based on energy plans, emissions reduction targets (such as Kyoto), and predicted trends in energy outlook (such as IEA, 2008). Examples of the latter can be found in Schmidt and Thrane (2009). Identification of the long-term marginal is related to significant uncertainties because it is highly sensitive to political targets and the implementation of these targets. Based on a review of several energy plans in Denmark, Mathiesen *et al*. (2009) demonstrates that there is no clear relation between targets/plans for the electricity system and the actually implemented technologies.

The typical aim of LCAs is to support long-term decisions, e.g. decisions regarding energy savings in food storage and preparation. Therefore, the applied marginal should also be the long-term marginal, e.g. 5–10 years. It should be noted that it is likely to identify significantly different energy systems depending on which of the following approaches are used: market average supply (used in attributional modelling), short-term marginal, or long-term marginal.

5.3.1 Pros and cons of consequential and attributional modelling of electricity

As described previously, identification of the marginal source of electricity may be related to significant uncertainties, especially when considering the long-term build marginal. These uncertainties are not present in attributional modelling, where national electricity mixes are typically applied. National electricity mixes are relatively constant over time, and good estimates of the development over time can be based on energy plans and targets.

However, the relevance of attributional modelling may be questioned. The average GHG emissions related to average electricity production are 0.052 kg CO_2e/kWh in Sweden and 0.764 kg CO_2e/kWh in Denmark (ecoinvent, 2007). The low GHG emission related to Swedish electricity is because of the high shares of hydropower and nuclear. If the food industry or the end-users in Sweden use more or less electricity, it is not likely that any of these two technologies will be affected. The potential for hydropower in Sweden is close to fully utilized, and nuclear plants are running at full load most of the time because of low marginal costs, and the construction of new nuclear plants is likely to be determined by political decisions rather than changes in the demand for electricity. The Danish and the Swedish grids are connected. Therefore, a change in demand for electricity in Sweden will be more likely to affect Danish electricity production, which is based mainly on coal and gas (Lund *et al*., 2009). Hence, in the Danish/Swedish example, it is likely that the actual emissions related to the use of electricity in Sweden and the reduction potentials are highly underestimated when using attributional modelling. Even if average supply of a larger geographical area is used, this will not add more cause–effect relationships into the modelling. Competitive issues in the Swedish/Danish case will be eliminated, but still the emissions related to the use of electricity will be over- or underestimated because the used electricity mix includes suppliers that are not affected.

The main problem of consequential modelling is that no commonly accepted marginal electricity for different countries and regions exists. The same applies for attributional modelling, but national electricity mixes are more fixed and less sensitive to assumptions introduced by the individual LCA practitioner than marginal electricity.

5.4 System delimitation and by-products in food LCAs

An allocation problem arises when having a multiple output process and only one of the products is used in the LCA. According to ISO 14044, the first option to solve the allocation problem is to avoid allocation by (i) subdividing the multiple output process (which is often not possible), or (ii) expanding the product system to include the functions related to the co-products, i.e. system expansion (also sometimes referred to as the avoided burden approach). The second option is to allocate by physical relationship, e.g. allocate transport services by measuring the service in units of tonne kilometre (tkm) or cubic metre kilometre (m^3km), depending on whether mass or volume is determining the load. A transport service can be perceived as a multiple-output process, because it provide the service to transport several different products. The third and last option is to allocate by other relationship, such as mass, economic value, energy content, etc. The difference between the second and the third option is that the second option reflects how the multiple output process is actually affected due to a change in demand, whereas the third option does not take into account how the process (and other related processes) are affected as a consequence of a change in demand of one of the multiple outputs.

5.4.1 Procedure to avoid allocation by system expansion

The general procedure to perform system expansion is described comprehensively in Weidema (2003), Ekvall and Weidema (2004), and Weidema and Ekvall (2009), and examples where the method is applied to food systems are given in Schmidt and Weidema (2008), Dalgaard *et al.* (2008), and Thrane (2006). The general procedure applies to any multiple output process: co-products, waste outputs, and recycling processes. It does not matter if the output is waste or a product. Recycling activities can be characterised as multiple-output processes because they supply (i) the service to treat waste and (ii) the recycled material which has the potential to substitute other/virgin materials. When performing system expansion to solve the allocation problem, the first step is to identify which one of the outputs is the determining product. The determining product is identified as the co-product that determines the production volume of that process. Note that this is not necessarily the co-product of interest to the LCA. Often, the determining co-product is also the product that is associated with the highest turnover; however, this is not always the case. Weidema and Ekvall (2009) provide a procedure to identify the determining co-product.

If the co-product of interest in the LCA is the determining product, then all emissions from the multiple-output process are included and the emissions from the displaced product are subtracted. An example is a rapeseed oil mill, where the determining product is the rapeseed oil and the dependent co-product is the rapeseed meal. The identification of the oil as determining is based on personal communication with market actors in the rapeseed oil industry. The identification of the oil as determining could also be done using the procedure provided in Weidema (2009). The rapeseed meal displaces the marginal supply of animal feed and the associated emissions are avoided. In some cases, intermediate processes have to be added before the dependent co-product can displace other products. Another example is when a process has an output of food waste which first has to be collected for treatment, digested in a biogas plant, and the biogas has to be combusted in a gas motor before the marginal supply of energy (electricity and heat) is avoided. The two examples are illustrated in Fig. 5.2.

If the co-product of interest in the LCA is the dependent product, then the multiple output process is not affected and it should not be included in the study. Instead the marginal supply of the product of interest should be included. An example is when a company uses soybean oil. Since the determining co-product from a soybean oil mill is the soybean meal (Weidema

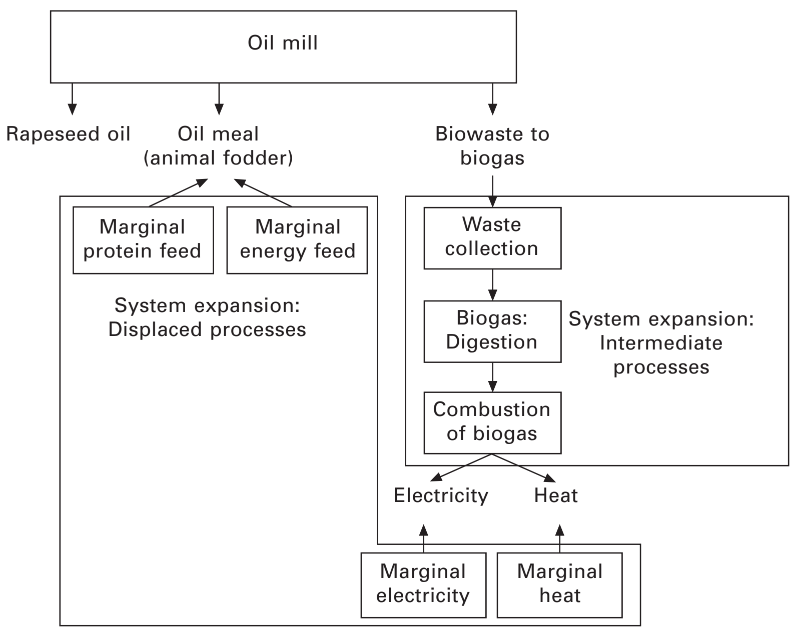

Fig. 5.2 Illustration of system expansion when modelling the consequences of rapeseed oil production. The oil meal can directly displace other products (protein feed and energy feed) and the bio-waste first has to undergo some intermediate processes before it can substitute other products (electricity and heat).

2003), the soybean oil is constrained by the demand for soybean meal. In this case the actual affected vegetable oil will be the marginal one, i.e. palm oil (Schmidt and Weidema, 2008). If a company uses soybean oil, there will simply just be less soybean oil available for other users in the market. Thus, in the example where a company uses soybean oil, the oil included in the study should be palm oil.

There may be special cases where system expansion is not possible because there is more than one product output without an alternative product route. Then there is no alternative production to subtract. An example of this is the supply of different meat products, e.g. tenderloin, fillet, and other meat from bovines. When there is more than one product output without an alternative product route, the prices of the co-products will adjust so that they have the same market trend. If this was not the case, the market would not be cleared. A change in demand for one of the co-products will affect the production volume of the multiple output process in proportion to its share in the gross margin of the co-product. This is equivalent to the result of an economic allocation of the multiple-output process.

An illustrative detailed example is a change in demand of 1 kg tenderloin from slaughterhouse. The slaughterhouse process per kilo product output is illustrated in Fig. 5.3. If 1 kg tenderloin is demanded, the slaughterhouse process will increase its total product volume of tenderloin by 8.4% of 1 kg bovine meat, i.e. 0.084 kg. Thus, the total product volume from the slaughterhouse will be 0.084 kg tenderloin, 0.277 kg fillet and 1.592 kg other bovine meat, i.e. total 1.953 kg meat. The emissions related to this are 37 kg CO_2e (19 kg CO_2e/kg * 1.953 kg). The 19 kg CO_2e/kg are specified in Fig. 5.3. This is exactly the same result that would be obtained if economic allocation was used. Since the production volume of the slaughterhouse is increased by only 0.084 kg tenderloin, two other effects will occur:

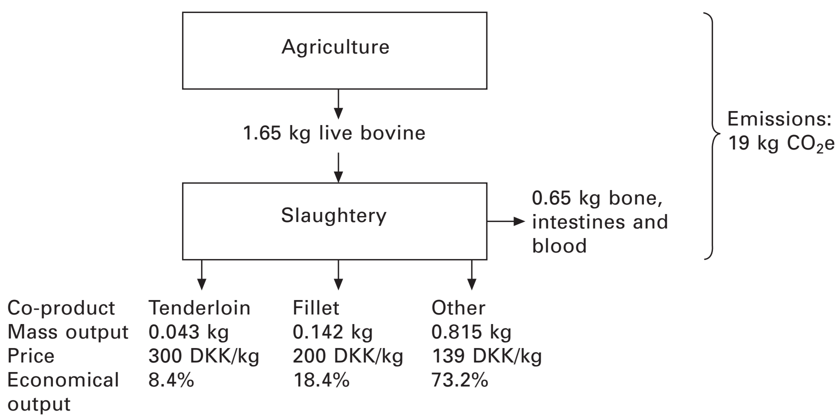

Fig. 5.3 Inputs and outputs per kg total product output of bovine meat from slaughterhouse. Figures are based on Nielsen *et al.* (2005).

(i) to provide the remaining demanded tenderloin, other users of tenderloin have to decrease their use by 0.916 kg (the 1 kg demanded minus the 0.084 kg actually additionally produced), and
(ii) since the 0.084 kg tenderloin will be co-produced with 0.277 kg fillet and 1.592 kg other bovine meat, other users will increase their use of fillet and other bovine meat correspondingly.

In most consequential LCAs, the latter two adjustments are assumed to level each other out, and hence they are not included (same result as economic allocation). The difference between the consequential and the attributional approach are that the consequential approach does not ignore that co-products exist and that the demand for one of the co-products may cause changes in the use phase of the different co-products. Because of the adjustments in the use phase, the consequential approach ensures that the mass balance is maintained.

5.4.2 Pros and cons of consequential and attributional modelling related to allocation problems

The advantage of allocation as in attributional LCA is that it may be relatively easy to estimate some allocation factors and go on with the modelling. Then there is no need to identify alternative production routes (marginal supply) of the dependent co-products. However, there are significant uncertainties related to the determination of allocation factors. In the bovine meat example illustrated in Fig. 5.3, the emissions related to 1 kg tenderloin are 37 kg CO_2e if economic allocation is applied, but if mass allocation is applied, the emissions are only 19 kg CO_2e. The difference is almost a factor of two. It should also be noted that it is not straightforward to apply attributional modelling consistently throughout an LCA study. The only allocation method that applies to all types of multiple-output processes is economical allocation. Also the identification of average market supply for all included products in a product system is a major task.

The advantages of consequential modelling using system expansion are:

- The actual utilisation of dependent co-products is taken into account. An example is if the dependent co-product, e.g. rapeseed oil meal, was used for energy purposes in Country A and for animal feed purposes in Country B, it is obvious that the emissions related to marginal energy and marginal animal feed are likely not to be the same. Therefore, one of the two options for utilisation of the rapeseed oil meal will be preferable. If allocation is used instead of system expansion this difference will not be visible
- System expansion does maintain mass and substance balance for unit processes. If economic allocation is applied to tenderloin on the product system in Fig. 5.3, the allocated process for tenderloin would look like

– Outputs: 2.270 kg distributed on 1 kg tenderloin and 1.270 kg bone, intestines and blood. Inputs: 3.223 kg live bovine. Also, if mass allocation is applied and the different co-products and wastes have different material compositions, the substance balance will be ruined by any allocation.

- System delimitation does not ignore that co-products exist, and even though simplifying assumptions can be introduced, e.g. by using economic allocation for co-products which have no alternative production routes, the derived assumptions on changes in the use stage are kept visible.

As in the case of allocation, system expansion may be related to significant uncertainties when identifying the avoided production (marginal supplies). The uncertainties related to the identification of marginal supplies are described in Sections 5.2.6 and 5.3.1.

5.5 Future trends

The pros and cons related to consequential and attributional modelling in LCAs on food products have been discussed. Currently, very little support on which approach to use and how to do it is provided in standards and guidelines such as ISO 14040, ISO 14044, and PAS 2050. In addition there is no commonly accepted data basis and system delimitation to stick to.

The most obvious problem related to this is that different LCA studies are not comparable. Common standards and consensus would solve this problem. Ongoing initiatives such as the GHG-protocol by the World Resource Institute and World Business Council on Sustainable Development, as well as future ISO standards, may solve some of the problems.

Some very important and not well covered aspects in LCA of food products are social impacts (such as food security and competition with biofuels), indirect land use, toxicological effects of pesticides, effects related to GMO, and biodiversity effects from land occupation and transformation. Also, many attributional LCAs are lacking cause–effect considerations which can be achieved by using consequential modelling. It is evident that the methodologies on the above mentioned issues will develop in the future.

Based on the discussion of pros and cons of the approaches in the present chapter, consequential modelling is proposed as the preferable option because it takes into account cause–effect relationships, it does not ignore the effects of co-products, and it maintains mass- and substance-balance of processes, which is not the case for allocated processes. Thus, the risk of misleading decision support is minimised in consequential modelling. The disadvantage of consequential modelling is that no commonly accepted marginal supplies of products for different countries/regions exist. The methodology for identifying marginal suppliers is available in Weidema and Ekvall (2009), but no 'catalogue' of default marginal suppliers exists. The same applies for attributional modelling, but national/regional market averages are more fixed and less sensitive to assumptions introduced by the individual LCA practitioner. The current problem of the absence of a commonly accepted ‘catalogue’ of default marginal supplies of products for different countries/ regions may be solved in future databases such as ecoinvent, where the next version will include both a consequential and an attributional version (Weidema, 2009).

5.6 Sources of further information and advice

Further information and advice on consequential modelling can be found in Weidema and Ekvall (2009), and consequential modelling in agriculture is further described in Schmidt (2008).

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6

Challenges in assessing the environmental impacts of crop production and horticulture

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Abstract: LCA has been applied to assess the environmental impacts of crop production and horticulture for over 15 years. This chapter discusses the main challenges of this procedure. It gives guidance on how to consider the multifunctionality of agricultural production and how to draw the boundaries of the production system. The complexity and variability of the production systems and the environmental mechanisms underlying the environmental impacts give rise to a number of challenges that have to be met. From this analysis, recommendations are deducted for a solid analysis of the environmental impacts. The chapter ends with an outlook on expected future developments.

Key words: crop production, horticulture, system boundaries, functional unit, variability, data collection.

6.1 Introduction

The first Life Cycle Assessments (LCA) of crops and horticultural systems were carried out in the 1990s (Büchel, 1993; Gaillard and Hausheer, 1997; Jolliet and Crettaz, 1996). The development of the LCA methodology of crops was significantly boosted by research performed in that decade on the environmental impacts of biofuels (Kaltschmitt and Reinhardt, 1997; Wolfensberger and Dinkel, 1997). Sleeswijk *et al.* (1996) provided the first methodological guide for LCA in agriculture to complement the general LCA methodology proposed by CML (Heijungs *et al.*, 1992). Later, the comparison of approaches and data used when several research groups each calculated an LCA of a wheat crop resulted in further recommendations for the application of LCA in agriculture (Audsley *et al.*, 1997). Since these early beginnings, a large number of LCAs have been published in the fields of crop production and horticulture (e.g. Bennett *et al.*, 2004; Brentrup, 2003; Charles *et al.*, 2006; Jungbluth, 2000; Mattsson *et al.*, 2000; Mouron *et al.*, 2006a; Munoz *et al.*, 2007; Nemecek *et al.*, 2005; Piringer and Steinberg, 2006; Van Der Werf *et al.*, 2005) and life cycle inventories that were made available in databases of previous LCAs are also available (ecoinvent, Danish LCA food database, Swiss Agricultural Life Cycle Assessment Database SALCA).

This chapter aims at highlighting the main challenges faced when assessing the environmental impacts of crop production and horticulture. It is structured as follows:

- In Sections 6.2 and 6.3, we show how the different specificities of agriculture constitute challenges for LCA in agriculture.
- Section 6.4 discusses the implications of these challenges and gives recommendations for LCA methodology
- Section 6.5 gives some expected future trends.

6.2 Main challenges: (a) Defining agricultural systems

6.2.1 Defining the functional unit

Recently, the concept of multifunctional agriculture has become widely recognised. Agriculture is needed for the production of goods such as food, feed, fuels and fibres, implying that production is the most important function of agriculture, and its main goal. However, agriculture also has other goals and functions such as landscape maintenance, rural development, ensuring farmers' income, etc. In order to take into account these aspects of agriculture, we have two options: we can include such additional functions by means of allocation or system expansion in order to make systems comparable, or we can use several functional units to express the different functions of agriculture.

The first method uses highly complex models that are difficult to communicate to stakeholders. As shown by Hayashi *et al.* (2005), the use of multiple functional units is quite common in agricultural LCAs (see also Charles *et al.*, 2006). Nemecek *et al.* (2005) considered three functions leading to different functional units:

(i) *Productive function*: Agricultural activity aims at producing food, feed or biomass for other uses (bioenergy, renewable materials). The environmental goal of this type of analysis is to minimise the environmental impacts per product unit. The productive function mainly reflects the perspective of the consumers and is quantified by physical units, such as kg of product, MJ of digestible energy, or kg of raw protein.

(ii) *Land management function*: One of the goals of agricultural policy is to maintain the agricultural production in a given area. This is reflected by the land management function. The cultivation of a piece of land should be achieved by minimising the environmental impacts per unit of area and time, which means reducing the land use intensity in most cases. The land management function mainly reflects the willingness of the society to preserve land for agricultural production and is measured in hectares and years (ha^{-1} a^{-1}).

(iii) *Financial function*: From the farmer's perspective, income is the key motivation. The environmental goal of this type of analysis is to minimise the environmental impact per unit of income. Depending on the study, several indicators for income can be used, such as gross profit or gross margin, which are expressed in a currency unit (e.g. €, $).

Using the functional units hectare and year, gross energy harvested (in GJ) and gross margin 1 (in €), Nemecek *et al*. (2008) give examples of the application of different functional units for the three functions.

6.2.2 Defining the system boundaries

The delimitation of the temporal, spatial and process-related system boundaries of crop production is particularly tricky.

Temporal system boundaries

In order to define system boundaries, we need to start with the final product. We have to list the activities required to cultivate a crop and produce a harvestable product. It is obvious that every process between sowing, planting and harvest needs to be considered. Soil tillage and fertilisers applied before sowing or planting should also be included. But how should the processes between the harvesting of one crop and the sowing of the next be handled? 'Catch crops' are sown to reduce erosion and emissions such as nitrate leaching, both of which may occur during this interim period. If these catch crops are harvested for fodder, they can be considered another product system and therefore the related burdens are charged to the harvested fodder. If the biomass is not harvested, these burdens have to be assigned either to the previous crop or to the following crop, or be divided between both. Nemecek *et al*. (2005) chose the second option, i.e. all burdens occurring after the harvesting of a certain main crop are charged to the following crop.

Arable crops are usually grown in a crop rotation and the crops are not independent of each other. This fact needs to be considered. The soil status before and after growing a crop is not the same (Crozat and Fustec, 2004). Leaving crop residues in the field after harvest that contain nutrients and sometimes pests or pathogens, alters the growing conditions for subsequent crops. Factors such as different types of soil cultivation, competition for resources (space, light, nutrients, water) and weed management measures, change the weed flora and seed in the soil. In many cases, base fertilisation using P, K and other nutrients, as well as lime, is not carried out every year, but only after several years, since these interventions have medium to long-term impacts. All these processes influence the management of a crop (i.e. fertiliser and pesticide applications, soil cultivation) as well as the emissions during the growing season. Growing cereals after a legume crop allows for a reduction in the N fertilisation of the cereal, but can, at the same time, lead to a higher nitrate leaching risk. Growing cereals too frequently or in monoculture increases weed and pathogen problems; as a consequence, more pesticide application will be needed and/or the yields will be likely to diminish.

When taking such effects into account, we have two options. They can be included by system expansion, e.g. by adding credits for nutrients left after harvest or for a yield increase of the following crops, or we can consider the whole cropping system, which can also be seen as a kind of system expansion. The first procedure can become quite complicated, and hardly ever accounts for all changes in impact (for examples see Nemecek and Baumgartner, 2006; Nemecek *et al*., 2005). Not considering the crop rotation effects can lead to erroneous conclusions, as shown already by Nemecek *et al*. (2001). An early example of LCA of a complete crop rotation was presented by Alföldi *et al*. (1999). If we want to compare cropping and farming systems, it is clearly preferable to consider whole crop rotations, rather than trying to include all the effects of crop rotation using credits. However, this procedure is also more demanding in terms of data and resources.

Spatial boundaries

All processes occurring during the period considered in the field have to be included. This includes all human activities, all inputs, and the upstream processes required for their production, and the direct field emissions to air, soil and water. It can also include impacts of crop production on the surroundings, such as field borders or neighbouring fields. This is of particular relevance for impacts on biodiversity.

Process related boundaries

All relevant activities, inputs and processes required for the cultivation of a crop product need to be included in the system boundaries (Fig. 6.1). In contrast to many industrial processes, agricultural systems are open and therefore delimitation of the system, i.e. drawing the line between technosphere (agriculture) and ecosphere (nature), is difficult. Emissions into the air and into bodies of water (leaching, run-off) are considered as emissions into areas outside the agricultural system, and therefore can be classed as emissions into the environment. The agricultural soil is, in itself, a boundary issue. On the one hand, the agricultural soil is a production resource, an integral part of the production process, and is managed by soil cultivation, fertilisation

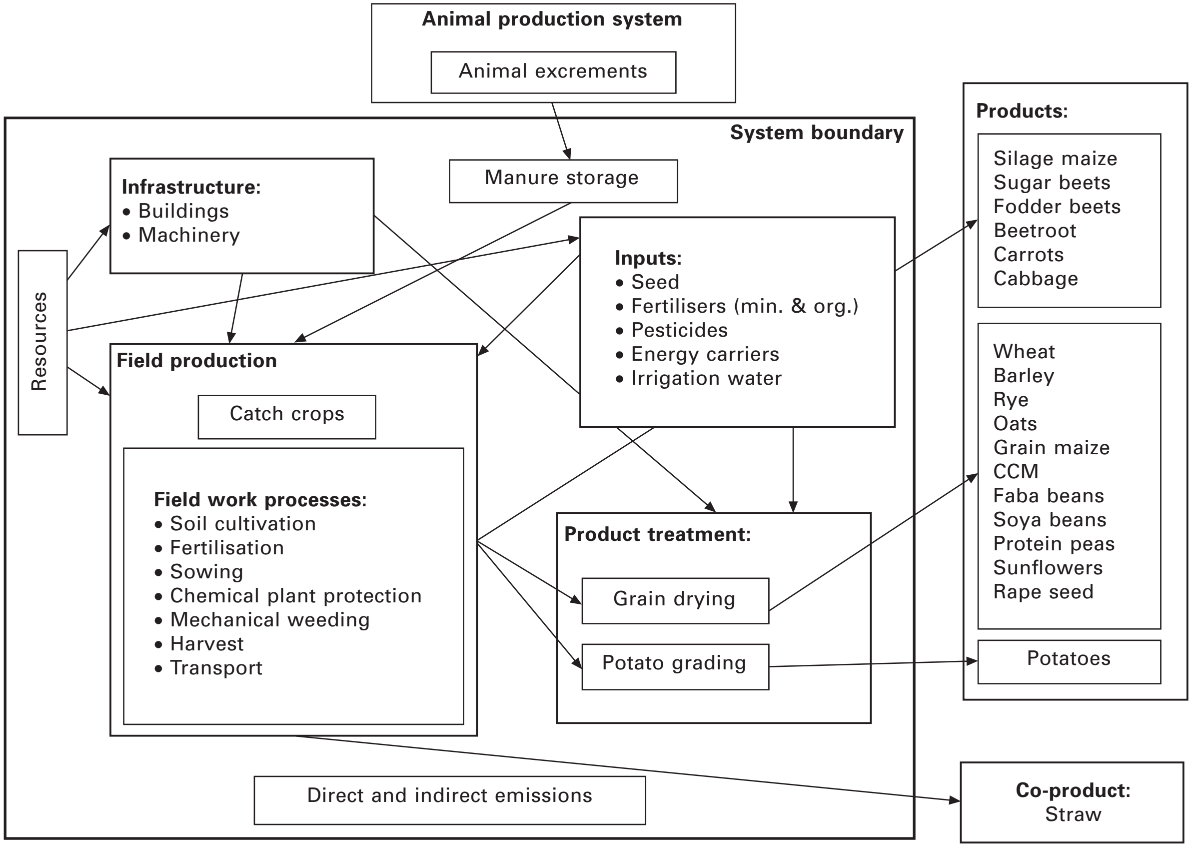

Fig. 6.1 Typical system delimitation for a crop product (after Nemecek *et al.*, 2005).

and pesticide application. On the other hand, the soil is a protection good, which needs to be preserved in high quality for future generations. Sleeswijk *et al.* (1996) have argued that the soil should be considered as part of the environment. Oberholzer *et al.* (2006) made a distinction between short-term (e.g. one season) and medium–long term impacts. For example, a pesticide application could have a significant short-term impact, but if no effect is observed after several years, then it is not considered a reduction of soil quality. In a crop LCA, the agricultural soil can be considered as part of the technosphere during the growing season and as part of the ecosphere after harvest.

A further question that is not treated uniformly in the various LCA studies is the inclusion or exclusion of infrastructure. Agriculture uses capital goods such as machinery, buildings and equipment. Due to the vegetation periods of some crops and weather/soil conditions, the rate of infrastructure use is generally low. Harvesting machines for example can be used, at best, during only a few weeks or months. The influence of infrastructure is therefore not negligible in agriculture (as shown by Frischknecht *et al.*, 2007). In general, infrastructure should be included in a LCA, unless evidence is provided which demonstrates that it has a negligible effect on the studied system and the impacts considered. Standard databases such as ecoinvent (ecoinvent Centre, 2007) contain datasets for some agricultural infrastructures that allow at least rough estimates. Frischknecht *et al.* (2007) showed that infrastructure contributes e.g. 20% to the fossil energy demand for crop products; the share is higher for organic products than for integrated or conventional ones. This difference is explained by the fact that mineral fertiliser manufacturing – in particular nitrogen fertilisers – is a main cause of energy demand in conventional and integrated agriculture, while it is not used in organic farming. The relative importance of infrastructure in organic agriculture is therefore higher. For some impact categories, the percentage of the overall impact that can be attributed to infrastructure is over 50%.

6.3 Main challenges: (b) Understanding agricultural systems

6.3.1 Understanding and modelling environmental mechanisms

As has already been pointed out, agricultural systems are quite open and are therefore dependent on the natural environment. Soil and climate have strong effects on the yield, management and emissions of a given crop. As it is impossible to measure emissions on a large scale, we must have recourse to modelling. The processes leading to emissions are often highly complex, as can be shown by the example of nitrous oxide. We therefore need to simplify the relationships between quantities of emissions and influencing factors, and to keep only the processes that are most relevant for the emissions. However, we must be careful not to over-simplify the environmental mechanisms. Despite huge efforts in LCA and agricultural system modelling during this last decade, this remains a big challenge.

The models used should be adapted to the goal of the study. There is not one 'correct' model to be applied in all LCA studies, but several models could be suitable according to the system investigated, the scope of the study (e.g. the region covered) and its goal. Ideal models for the estimation of direct field and farm emissions should reflect the underlying environmental mechanisms, be site and time dependent, consider the effect of soil and climate, appropriately include the effect of management and be applicable under a wide range of different situations. Such ideal models are unfortunately not available. Current models can explain, at best, only part of the variability. Particular attention needs to be paid to the degree of detail in the different models (Freiermuth Knuchel *et al.*, 2009). If, for example, the model for nitrate leaching considers the influence of no-till agriculture on nitrate leaching, but the model for nitrous oxide does not, any conclusions drawn from a comparison of ploughing and no-till agriculture will necessarily be limited. Some models reflect the non-linear nature of the processes, thus allowing for more realistic estimates. Particular attention has to be paid to this issue, which has implications for practical work. For example, applying 50 kg N/ha on two separate occasions does not necessarily lead to the same emissions as applying 100 kg N/ha all at once. Splitting the fertiliser application should lead to a reduced risk of nitrate leaching, while on the contrary, ammonia emissions increase if an amount of slurry is split into two applications. In the example given in Fig. 6.2, application of 20 m^3 of slurry leads to emissions of 8.7 kg NH_3, which corresponds to an emission rate of 63% related to the total ammonium content (TAN) in the slurry (calculated according to Menzi *et al.*, 1997). Doubling the amount of slurry to 40 m^3

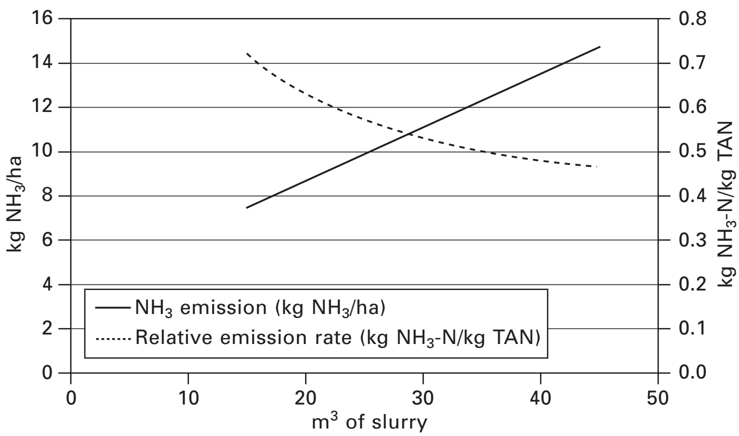

Fig. 6.2 Illustration of situation dependent emission rates for the case of ammonia emissions after slurry application. TAN = total ammonia nitrogen in the slurry.

results in an emission of 13.5 kg NH_3 (increase of 54% only); the related emission rate has dropped to 48%. Location is also an issue; if the data for such a comparison are collected at different sites, the simple arithmetic mean of the fertiliser rate will give misleading results.

It is best to assess some impacts in a qualitative manner. It remains a challenge to include such assessment in an LCA, which is of quantitative nature. However, this can be done, for example by quantifying certain areas as having a certain quality.

In crop production in general, and in horticulture in particular, the use of pesticides is an important environmental concern. Although the results are subject to a due amount of uncertainty, methods are available to assess the impacts of a large number of pesticide active ingredients on aquatic or terrestrial ecotoxicity or human health (Kägi *et al.*, 2008; Rosenbaum *et al.*, 2008). Despite this, we see LCA studies of agricultural systems ignoring the impact of pesticides. In future studies, the authors should either perform an assessment of the toxic impacts of pesticides, or provide evidence that pesticides do not play in important role in the studied system. Otherwise, such a study should be considered incomplete.

6.3.2 Assessing land use impacts

Several issues can be summarised under the general heading of 'land use'. We need to distinguish between direct and indirect impacts. Direct impacts are the impacts caused on site by cultivating a certain area of land. Indirect impacts are the impacts that are caused indirectly in other areas.

Direct impacts have consequences for soil quality, biodiversity and the landscape. Although it is widely recognised that there is a need to assess the impact of land use, a single generally accepted assessment method is still lacking (Milà i Canals *et al.*, 2007). The ecoinvent database, where the transformation of land and its occupation were systematically quantified and the different types of land use were classified based on the CORINE land use system (Bossard *et al.*, 2000), has significantly advanced progress in this area. Some impact assessment methods consider only the total area of land occupied (e.g. CML), while others weigh potential impacts on biodiversity (like EcoIndicator '99).

To assess direct impacts on agricultural land, two specific methods have been developed in the frame of the SALCA method (Swiss Agricultural Life Cycle Assessment): potential impacts on biodiversity (Jeanneret *et al.*, 2006) and on soil quality (Oberholzer *et al.*, 2006). These methods respectively estimate the impacts of various agricultural practices on eleven indicator organism groups for biodiversity and on nine indicators for physical, chemical and biological properties of the soil.

Most LCAs of crops do not include indirect land-use effects. In contrast to most industrial processes, emissions from land do not occur only in cultivated agricultural land; they are also observed in uncultivated areas (Gärtner *et* *al*., 2001). Some studies, therefore, compare the effect of the cultivation of a given crop to a reference state, for example extensive meadow (Wolfensberger and Dinkel, 1997). The authors compared the production of biofuel crops to an alternative land use, which was assumed to be extensively managed grassland. In attributional LCA, the indirect effects are generally not taken into account, or only strong impacts of land-use changes such as deforestation with the associated emissions (Jungbluth *et al*., 2007). In consequential LCA, efforts have been made to include indirect land-use impacts as well (Kloverpris *et al*., 2008).

6.3.3 Understanding and quantifying variability

Soil and climate conditions have a large amount of influence on emissions, resource use and yield. Consequently, crop management must be adapted to the pedo-climatic conditions (Fig. 6.3). Crop management is also dependent on the socio-economic structures and the traditions of the production region. Since these factors are highly variable, the environmental impacts vary even more. For many crops, the growing season is determined by availability of sunlight, water and temperature, at least for field production. However, some crops, in particular those cultivated in warmer regions, can be grown at different periods of the year, which is a further source of variability. The characterisation factors for the impact assessment may also depend on the pedo-climatic conditions. An example is the pH of the soil, which determines its sensitivity to acidification. Emission models should therefore account for the relevant regional, site-specific and even season-specific factors.

Although the number of farms is decreasing in most countries, the number

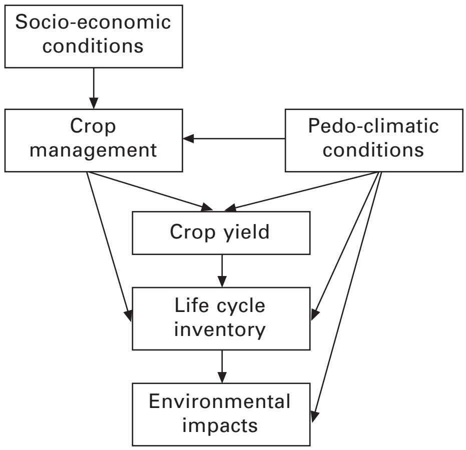

Fig. 6.3 Schematic representation of the relationships between pedo-climatic conditions, crop management, crop yield and the environmental impacts (per product unit).

of productive units is still relatively high in agriculture compared to production sites in many industrial sectors. The farmer is a key stakeholder, largely determining the environmental impacts by his management decisions. The variability between the environmental impacts of different farms is huge, even if their boundary conditions (region, farming system, farm type, etc.) appear to be similar (Alig *et al.*, 2008; Bockstaller *et al.*, 2006; Mouron *et al.*, 2006b; Rossier and Gaillard, 2001). Differences of a factor of five to ten can easily be observed, even within one region or farm type. The differences can be attributed to such variables as the farmer's education, his skills, the internal organisation of the farm, etc. We conclude from such a high variability that environmental impacts can be substantially reduced, on condition that the farmer is aware of the environmental impacts caused by his farm management (Alig *et al.*, 2008). The current high variability makes the collection and calculation of representative and accurate data tedious (see below).

Multivariate techniques are very useful in analysing the variability of the impacts and are helpful when making recommendations for the management (Nemecek and Gaillard, 2007). Studies for dairy farms (Rossier and Gaillard, 2001), conventional, integrated and organic farming systems (Nemecek *et al.*, 2005) and apple production (Mouron *et al.*, 2006a) showed that midpoint impact categories can be classified into three groups: *Resource management* encompasses the energy demand, the global warming potential and the ozone formation. *Nutrient management* is represented by the eutrophication and the acidification. The aquatic and terrestrial ecotoxicity, as well as the human toxicity, can be summarised by the *pollutant management*. The impact categories *biodiversity* and *soil quality* are influenced by all three of the abovementioned management axes and must therefore be dealt with separately. These five environmental areas cover the whole analysis, enabling a simplified communication to decision makers. The management axes are related to different management actions with different time horizons (from long-term to short-term decisions), such as use of machinery and the application of fertilisers and pesticides.

Practical examples show that the environmental impacts along these axes vary rather independently. Some farmers or systems perform well in all three dimensions; some are poor for all axes, while others have a low performance on one or two axes (Mouron *et al.*, 2006a; Rossier and Gaillard, 2001).

The analysis allows a quick assessment of the environmental impacts of each individual farm or system and the derivation of improvement measures. The case studies have also made it clear that there is not necessarily an environmental trade-off between the groups of impact categories: it is possible to have good results in all three dimensions. Mouron (2005) showed for apple orchards in Switzerland that a better eco-efficiency (lower environmental burdens per unit of economic return) was related to higher return. This indicates that for these systems there is not a contradiction between environmental and economic performance.

6.3.4 Collecting data

Relatively good data exist on the production volume and area of various crops, which can be retrieved from national statistics: EUROSTAT for Europe or FAOSTAT on a global scale. From these data, the average yields can be calculated, which are also reported in the above sources. There is not much data available on yield variability, which is in general very high. Detailed management data – as required in LCA studies – is largely missing. Farm Accountancy Data Networks (FADN) give good indications of the economic performance of farms in a given country, but they do not contain sufficient information on management. The best source of management data are large pilot farm surveys. If such surveys exist in the region being studied and the data are accessible, the LCA practitioner is in a lucky situation, but considerable time is needed to analyse such data for the purposes of an LCA. However, in many regions these surveys are not available and therefore, in general, other sources have to be used such as statistics, FADN, recommendations, documents from extension services, legislative norms, data from field experiments and expert knowledge in order to obtain representative data for the given region (Nemecek and Erzinger, 2005). Field experiments usually provide very detailed and precise data, but are normally not representative. Because the different sources may be partly contradictory, particular care has to be taken to ensure overall consistency of the dataset used in a study. To obtain reliable and representative data, large farm samples are required, since the inter-farm variability is huge (see Section 6.3.2). A modular extrapolation approach has been developed by Nemecek *et al.* (2009b). It enables us to extrapolate crop inventories and LCAs from existing inventories to all producing countries in the world. Global and multinational average impacts and their distribution can be calculated in this way. The method allows a significant resource-saving over the classical approach of defining detailed inventories for each considered situation.

The lack of standardised data collection tools and formats for agricultural management is a major obstacle for the development of widely usable LCA tools in agriculture.

6.3.5 Managing complexity

Crop production and horticultural systems are complex, as explained above. The high variability of pedo-climatic conditions, farming systems and management techniques, as well as individual differences caused by the farmer, can be assessed only by a large number of data points. This assessment cannot be carried out using standard LCA procedures; therefore, tools are needed for automated and standardised data collection and calculation procedures. Audsley and Williams (2008) presented a method to combine LCA with system models, which gives a high flexibility for the description of new systems. The agrarian model presented by Deimling *et al.* (2008) provides a generic approach to assessing different farming systems, crop types and growing locations. The SALCA (Swiss Agricultural Life Cycle Assessment) methodology (Gaillard and Nemecek, 2009) gives a general LCA framework for agriculture, consisting of a life cycle inventory database for agriculture, models to estimate direct field and farm emissions and life cycle impact assessment methods with special reference to agriculture (in particular for biodiversity and soil quality). This methodology also includes standardised and generic calculation tools for different crops (SALCAcrop), and farms (SALCAfarm) as well as an interpretation scheme. One instance of integration into a FADN network has also been realised (Nemecek *et al.*, 2009a). Care must be taken to build a modular structure in order to keep the whole system manageable. The construction of such standardised tools and their integration into existing models, databases and networks requires a lot of resources. However, it is the only way forward to achieve an integrated assessment, to manage complexity and to ensure a consistent, reliable and efficient assessment.

6.4 Implications and recommendations

Great progress has been made during the last fifteen years in the field of crop and horticultural LCA. However, considerable challenges still remain. In the following we discuss the implications of these challenges and give recommendations that will hopefully boost the development of agricultural LCA.

- The classical concept of a product LCA with a single function cannot adequately represent the nature of multifunctional agriculture. We therefore recommend considering different functional units to represent the land management, productive and financial functions, according to the goal and scope of the study. The suitability of product LCA has to be checked before commencing each evaluation. A product LCA is a very useful tool to compare the environmental impacts of products, but too often it is applied in situations where processes or systems are to be compared or analysed. In such cases, a process or system LCA would be more appropriate (Geier and Köpke, 2000; Nemecek *et al.*, 2005). This could be an LCA of a cropping system, a farm, or a whole region.
- We need a more standardised methodology. The differences between various studies are too great, making the results difficult to compare, for example as shown by Basset-Mens (2008) for milk production. If one wishes to perform a meta-analysis over several publications, the author of each study is often a highly influential factor, and can shape the study even more than the country, the intensity, or the farming system. However, harmonisation and standardisation should not inhibit further progress, since the consensus is often achieved at the lowest common denominator, which means a step back towards what has been achieved by the innovative research.

- Standardised data formats for LCA have been developed to facilitate data exchange between LCA practitioners and different databases (e.g. EcoSpold, ILCD). Tools for conversion between different formats have also been developed (Ciroth, 2007). However, there is currently no format to use when reporting and exchanging agricultural management data. Detailed data, such as information on fertilisers, pesticides, machinery use and feedstuff use, is a necessary basis for LCA databases, and needs international standardisation. This is a main obstacle for the collection and exchange of agricultural data between countries. In this respect, developments such as AgroXML (KTBL, 2009) or the SEAMLESS project (van Ittersum *et al.*, 2008) are welcome.
- Due to the large variability of agricultural systems, we need a large amount of observational data to better understand and analyse the underlying mechanisms. In order to make further progress, standard calculation tools and methods are needed that are able to describe a large number of situations efficiently. Data collection, LCA calculation and result analysis should be automated as far as possible in order to manage the large amount of data and also to allow non-specialists to use it. Past experience has shown that many processes and inputs related to crop and horticultural systems are similar; therefore it should be possible to define a generic crop system that can be parameterised for various situations. Standard tools such as SALCA (Gaillard and Nemecek, 2009) can make LCA calculation in agriculture faster and more standardised. A big challenge for the next decade will be the collection of more representative inventory data covering the large variety of situations (crops, cultivar, regional differences, and differences in farming systems). However, inventory data collection makes considerable demands upon resources, and is expensive. We feel that without a considerable investment in inventory data, progress of agricultural LCA will be hindered.
- The assessment of variability and uncertainty should become a standard procedure in LCA studies. The impact values should be accompanied by an indication of their variability and in a comparative study, a statistical test should be carried out to show the significance of each difference.
- LCA in agriculture is suffering from a lack of generally applicable and sufficiently reliable emission models. As exact measurements of specific emissions are rarely available, we need to model these processes. Most models are either too limited in scope or ignore important mechanisms. They should cover not only pedo-climatic conditions, but also agricultural management practices in sufficient detail.
- Many LCAs of agricultural systems are still published without including infrastructure such as machinery, buildings and equipment. In future, the infrastructure should be included as a standard, unless clear evidence is provided that, for the particular system and the impact considered, infrastructure plays a negligible role.
- Impacts on ecotoxicity are largely neglected in many LCAs, although

numerous studies have demonstrated the predominant role of pesticides in agriculture and observed strong impacts on aquatic and terrestrial ecosystems. Unless clear evidence is provided that pesticides do not play a role and ecotoxicity impacts are negligible, these impacts need to be included.

- Land use impacts of agriculture have long been acknowledged as important. This discussion has been accentuated in the last few years, as the focus has been directed to the strong impacts of deforestation on climate change and biodiversity. There is an urgent need for standardised methods of assessing the various direct and indirect land use impacts. Such methods require a solid scientific basis and robust data. The assessment of water resources should also become a standard in crop LCAs.
- The application of midpoint impact assessment methods is quite widespread in LCA of arable and horticultural crops (Hayashi *et al.*, 2005). Stakeholders interested in agricultural systems are interested in this level of detail; they do not generally need an aggregation to a single figure that is based on highly disputable value choices. Stakeholders from non-agricultural fields generally prefer simpler endpoint methods. We recommend using midpoint impact assessment methods as a default.

6.5 Future trends

We expect the following future trends to occur in the field of crop and horticultural LCA in the next decade:

- The recent food crisis has revealed the vulnerability of the food sector. The growing world population and increased demand for meat and biofuels will dramatically increase the need for crop products. Accentuated climate change will increase risks for agriculture and partly reduce the available production resources, in particular fertile soil and water. Producing enough biomass for all these needs will be a great challenge and lead to strong intensification of agricultural production. We have to take care that the environmental concerns are not totally neglected. In order to ensure this, tools are needed to develop more sustainable and resource-efficient production systems. In this context, we believe that LCA of agricultural systems will play a central role.
- The concerns about climate change will favour carbon footprints and environmental labelling of food products. On the one hand, it will boost research in this area and provide resources for further development. But this evolution is also risky: it puts science under a lot of pressure, since answers are needed quickly, which does not allow for a proper investigation of the food production systems. Moreover, the various stakeholders may try to influence the results, which can hamper the credibility of science.

- Direct and indirect land use impacts will be included in standard LCAs. A discussion has to be held as to how far the indirect land use impacts are to be considered. Only impacts that can be observed in reality and that are based on solid data should be included, in order to avoid speculative assessments. Impacts on water resources will be included as a standard, except in regions where water is not a limiting factor. In addition to land, the water resource will be the main factor limiting future agricultural production.
- The setup of inventories, as well as the impact assessment, will better take into account regional aspects. Coupling LCA to geographical information systems (GIS) opens new perspectives. Extrapolation tools will allow a fast estimate for unknown situations. Generic data with validity over a larger area or a group of products will become widespread. LCA tools will also be developed that are flexible and can cover a wide range of situations.
- The debate on attributional and consequential LCA will most likely continue for some time. Consequential LCA needs further development to be more widely used by LCA practitioners and stakeholders. The modelling of the consequences of decisions yet to be taken remains a big challenge.
- We expect LCA to be more frequently combined with economic models and system models at different levels (crop, farm, regional, national). This will offer new perspectives for the analysis and will make the evaluation more useful for decision making. For an integrated sustainability assessment, social aspects also need to be included. It is neither necessary nor desirable from our point of view to include all aspects in LCA. Rather, LCA should be combined with other tools for the assessment of socio-economic impacts.
- For a long time LCAs were calculated mainly in Europe. The last decade showed a strong increase of activities in North and South America, Asia and Oceania, which is encouraging. However, the activity in Eastern Europe and in Africa is exceptional and we hope that LCA will become a standard methodology also in these regions.

LCA of crop production and horticulture has experienced an exciting development in the past. The number of researchers and research groups has strongly increased during the last few years. We are confident that these past and present efforts will lead to more widespread use of LCA and to its application in practical decision making.

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7

Complexities in assessing the environmental impacts of livestock products

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Abstract: Livestock systems are complex in a number of aspects, which is why it is not an easy task to assess associated environmental impacts. Some of the most important aspects are: (i) the way or the degree by which the rearing of the farm animals is integrated with the production of crops, (ii) the impact of different housing and feeding conditions, (iii) the multi-process and multi-product nature of livestock production systems, (iv) the interactions between livestock and land use, and (v) the complexities in estimating emissions/leaching related to livestock production. This chapter illustrates these complexities and discusses how they can be dealt with.

Key words: livestock products, life cycle impact assessment, pig production, organic pig, landless system, land-based system, manure handling.

7.1 Introduction

Livestock products are amongst the types of food with the highest environmental loads. Furthermore, considering the life cycle stages for a livestock product from the farm to the fridge in the supermarket, it is clear that the primary production on the farm and the related inputs are responsible for the major part of the load of many impact categories (Dalgaard *et al.*, 2007; Nielsen *et al.*, 2003). In assessing the environmental impact of livestock products, it is thus important to have a good estimation of the impacts at the farm level.

Livestock systems show a huge diversity in the way or the degree by which the livestock production is integrated with crop production, in their reliance on inputs from distant places in the world, and in housing and feeding conditions. All mentioned aspects have a huge effect on emissions related to the production. Even when livestock rearing and fattening may take place independently from land use for crops, the production of manure is of great environmental importance and it will most often have to be applied on agricultural land. Thus the aspect related to crop production and land use needs to be dealt with in assessing livestock systems, besides the particular livestock aspects. In addition, emissions related to livestock production are strongly influenced by a number of environmental factors.

The aim of this chapter is to illustrate some of these complexities and to discuss how they can be dealt with. The normal steps in performing an LCA, e.g. goal and scope definition, inventory analysis, impact assessment and interpretation are presented, and the problematic methodological issues in assessing the environmental impacts of livestock systems are highlighted where relevant.

7.2 Complexities in assessing the environmental impacts of livestock systems

As an example, we perform an LCA comparing pork production in an indoor system versus an outdoor one in order to illustrate the main complexities. The indoor pig production system was defined in a recent report on the environmental impacts of meat and dairy products in Europe (Weidema *et al*., 2008). In particular, it is representative for EU pig production with high feed efficiency and optimal manure handling and utilization, assumed to cover the situation in Denmark, Germany, France and other north-western European countries. Similar intensive pig operations also can be found in, for example, Spain and Brazil. The outdoor system considered is an organic pig production system representing the most common one in practice today in Denmark, where the sow herd is kept on grassland with access to small huts for protection, and the fattening pigs are kept in indoor facilities. Relevant complementary information is available from Halberg *et al*. (2008).

7.2.1 Goal and scope definition

In our example, we aim at performing an LCA of pig meat that leaves the farm gate. As mentioned earlier, the net environmental impact at the farm gate is very important. This information can be relevant to both meat processors and farmers in decision making. Whereas meat processors are concerned with the question as to where to buy the pigs for processing in order to be able to produce a final product with low environmental impact, farmers consider how to optimize the resource use related to the pig production.

7.2.2 Functional unit

In conducting life cycle analyses for livestock systems, different product-related functional units can be chosen depending on the aim of the study.

They can be defined, for instance, as one kg of meat (live-weight, carcass, or edible), kg protein, kcal energy, a part of the animal, etc. If the aim is to assist producers (farmers and meat processors) to optimize the production process for resource savings and environmental improvements, 'one kg meat' is often used to display the LCA results. Alternatively, for supporting customers to decide their meal choice, 'one kg protein' or 'one kcal energy' would be the preferred functional unit to 'one kg meat'. In other words, choosing the appropriate functional unit is regarded as the first critical task when accomplishing an LCA to get the right focus. The functional unit used in our study is one kg live weight of the meat delivered from farms. This selection is satisfactory to capture the performance of different production systems for pig meat.

7.2.3 Different types of LCA

Considering the significance of decision-making for LCA methodology, a distinction has been recognized between the two types of LCA: attributional or consequential. The former seeks to cut the portion of the global environmental impact related to a particular product, and the later seeks to capture change in environmental impact as a consequence of a certain activity and thereby provides information on consequences of actions (Nielsen *et al*., 2003). In practical applications, the 'attributional' LCA uses average or supplier-specific data and applies allocation factors to deal with co-product allocation. On the other hand, the 'consequential' LCA typically uses marginal data and avoids allocation by using system expansion.

7.2.4 System boundary

After defining the functional unit, the next step is to give the definition of the system boundary to identify which unit processes are included in the LCA and which are not. The definition of system boundaries is important for designing an LCA and theoretically it should include as completely as possible all unit processes necessary for delivering the functional unit. System boundaries need to be specified in several dimensions, which are discussed in more detail in Baumann and Tillman (2004). Briefly reviewed, they are: (i) boundaries related to natural systems; (ii) geographical boundaries; (iii) time boundaries; (iv) boundaries within the technical system in relation to capital goods, personnel, etc; and (v) boundaries in relation to other products' life cycles.

For livestock production, the system boundaries typically start at the production of raw materials such as feed, fuels and electricity consumed at the farm, and end where the finished animals leave the farm. With regard to our example, Fig. 7.1 presents the system boundary chosen for the pork production chain until farm gate in indoor and outdoor systems. It appears that on the farm itself, and its upstream, there exist a number of component

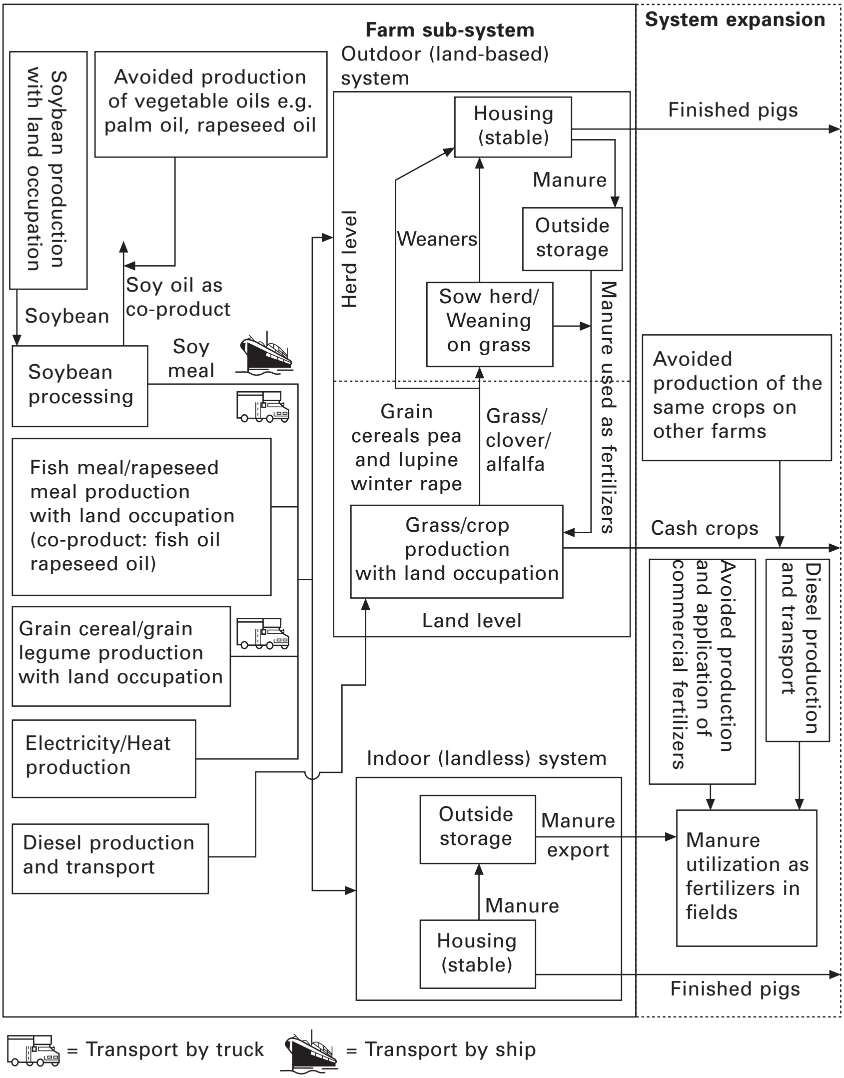

Fig. 7.1 Overview of the product chain of pig meat until farm gate in indoor and outdoor systems.

processes that contribute to the overall environmental impact. Some of these interact directly with the pig production (e.g. cereals used for feed) whereas others, as parts of the farm, interact to some extent and so they also need to be considered. For example, cash crops that are not used to feed the pigs in the system but are exported, will usually receive part of the pig manure and the question how to account for emissions from this manure is perhaps not easy to answer. Handling such complexity by expanding the system boundaries or disaggregating the given process into different sub-processes represents a challenging task for livestock impact assessment.

The livestock production often takes place at small, unique units compared to the later operation stages in the process chain, such as slaughtering and meat processing. This impacts on data collection and raises a question on the possibility of simplifying the system without losing validity of the LCA being used for supply chain inventory modelling. It has been shown that under certain circumstances, simplifications are possible for livestock production systems. If the animals in all stages of their life are kept under housing conditions, this livestock production *per se*, seems to take place independently from the land-based crop production. However, as mentioned before, the manure produced by indoor feedlots has to be used for land-based crops elsewhere. Examples can be intensive pig and poultry systems where the feed is based on globally traded feedstuffs. On the other hand, if the system includes grazing animals or the feed used is very site specific, this crop–livestock interaction becomes more evident. This will typically be the case in most beef, milk, and outdoor organic pig production systems.

In our example, we model the indoor pig production system as a landless livestock system and the organic one as a land-based system. This impacts highly on what types of data and what basic assumptions are needed. The differences between the two systems lie mainly in the origin of the feed (bought from outside or grown on farm), the opportunity for the pigs to access to grazing areas, and how impacts related to manure handling are modelled and assessed.

Various components can make up the feed for the pigs, such as grain cereals, grain legumes, grass silage, grazed grass, soy meal, fish meal and rapeseed meal. In conducting life cycle inventory for livestock products, the composition of imported feed often can be simplified to include only feed derived from those crops that are affected by an increased demand for feed (Dalgaard *et al.*, 2007). From Fig. 7.1, it is seen that feedstuffs for indoor pigs in the present example include primarily cereal and oilseed meal. For outdoor pigs, a certain amount of feed may come from grasslands.

The production of the three protein feed components, soy meal, rapeseed meal and fish meal, also results in a co-production of soy oil, rapeseed oil and fish oil, respectively, which raises the question how to divide the environmental impacts from the process between product and co-product. Practically, it is not straightforward to do so unless allocation or system expansion is used. The manure produced by the pigs, either in stables or on grass, is considered to be available for growing crops, saving the use of a certain amount of artificial N and P fertilizers, thus reducing environmental burdens associated with the manufacturing of these agrochemicals. Manure can also enhance carbon storage potential of the soil and thus help remove atmospheric carbon, if properly applied (FAO, 2001). So, the use of livestock manure on the one hand contributes to resource savings and GHG mitigation, but its adverse effects on the environment, on the other hand (resulting from ammonia emissions to air and nitrate and phosphate leaching to water) have to be considered. A detailed discussion regarding the issue of how to account for emissions/leaching from manure in an LCA of livestock production is presented in Sections 7.2.5 and 7.3.2.

Another assumption related to the example is that direct energy consumed by the two systems is for farm operation and for animal housing, derived from diesel fuel and electricity, respectively. Transport of soy meal and other imported feed, which consumes energy and creates emissions, can be modelled either as a separate unit process or integrated into 'feed import' unit process. Livestock production is also related to land use for feed production and assessing the impacts of potential land use change to meet increased demand for meat adds one more challenge. Figure 7.1 also presents input and output flows accounted for in the inventory for pig farming at herd, land level and farm gate.

7.2.5 System expansion and allocation

A critical methodological issue in conducting an LCA is how to distribute environmental loads between different products produced within the same system. It is a common picture that besides meat, a pig farm also produces other products, and in some cases not only one or two but a range of co-products. While manure is a co-product directly linked to pig production itself, others are not depending on the unique structure of the farm. Thus, in performing an LCA of livestock products, one needs to be very specific in how to handle complexities of co-products, i.e. allocation versus system expansion.

In principle, the ISO standard (ISO 14041, 1998) recommends three methods for handling co-products in LCA. The first priority is to avoid allocation, if possible, by using the expansion of system boundaries or dividing the unit process into sub-processes. If unable to do so, allocation for the system can be done in such a way that reflects the physical relationships (e.g. mass, volume, energy, etc.) between its different products. Finally, economic evaluation based on the contribution of each product to the economy is the essential option for allocation if both approaches mentioned above appear to be impracticable.

The choice between the two methods for distributing the environmental loads between product and co-products (allocation and system expansion) depends on which type of LCA is intended to be used: attributional or consequential. In most cases, LCA for agricultural and food products is seen as a support tool for decision making – what would happen if? – and in this respect, the consequential LCA is viewed as appropriate. This approach takes into account all processes that are affected by a change in the production of a product, e.g. milk in dairy systems. By an increase in the production of milk, the co-production of meat is also increased and therefore it is difficult to estimate the marginal environmental burden per functional unit of milk.

In order to avoid allocation, resource use and emissions associated with the co-product (i.e. the extra meat) are included in the milk production. As a way to compensate for this, the milk system is expanded to include the avoided production of meat elsewhere. The assumption behind this is that the increased production of meat, a co-product of milk, resulting from increased milk production, will reduce meat production in other systems. The choice of method to distribute the environmental loads between milk and meat thus has a decisive impact on LCA of milk production and it has been shown that system expansion is preferred to analyse the consequences of changes in future milk production (Cederberg and Stadig, 2003).

Another complexity for distribution of environmental loads in livestock products such as meat, is that the carcass is subdivided into different meat parts before selling to the consumer. These different parts have different qualities and prices. There has been, so far, no consensus on how to handle this complexity. Weidema (2003) has discussed possible ways of handling the issue. A major argument here is that it is the expensive parts of the carcass that determine if a production process takes place or not, since without these parts, such process will not be viable. Thus, it is suggested that economic allocation is a relevant procedure, at least when allocation is performed. Furthermore it is suggested – with the example of beef – that the parts of the meat that can only be sold as minced meat or as ingredients in other composite foods in fact will not be the driver in producing the beef and thus are by-products from the 'pure' beef production. If system expansion is used, these parts can be considered to replace other cheap meat products, such as pork and chicken. Following this route, the expensive parts of the beef carcass, e.g. tenderloin, fillet, top round and steaks, will have to pay a proportionally larger share of the environmental burden. There is, however, a need to elaborate on this issue in more detail and with more examples.

7.2.6 Data quality requirements

It is well recognized that the reliability of the results from LCA studies strongly depends on the extent to which data quality requirements are met. The following parameters, also regarded as different aspects of data quality (Baumann and Tillman, 2004; ISO 14044, 2006), are of importance.

- Time-related coverage, i.e. the age of data and the minimum time duration over which data should be collected;
- Geographical coverage, i.e. the geographical area from which data should be collected;
- Technology coverage, i.e. the specific technology or technology mix that is implemented;
- Precision, i.e. the variance of the data values;
- Completeness, i.e. the percentage of the locations reporting primary data for each data category in a unit process;

- Representativeness, i.e. the qualitative assessment of the degree to which the data set reflects the true population of interest;
- Consistency, i.e. the qualitative assessment of how uniformly the study methodology is applied to the various components of the analysis;
- Reproducibility, i.e. the qualitative assessment of the extent to which information about the methodology and data values allows an independent practitioner to reproduce the results reported in the study;
- Uncertainty of data, models, and assumptions used in the analysis.

7.3 Inventory analysis

7.3.1 Data collection

The second phase of LCA is inventory analysis, which involves data collection and calculation procedures to quantify the relevant inputs, outputs and emissions from a product system. Inputs comprise the use of resources (e.g. fossil fuels, minerals, water and land) and outputs are the products/co-products produced by the processes involved. Emissions include air releases, solid wastes and waterborne wastes. This phase can be very work-intensive and time-consuming compared with other phases of LCA, in particular for livestock production, due to its high complexity. Data on transport, extraction of raw materials, processing/production of commonly used products such as coal, diesel, natural gas, electricity, fertilizers, etc. can be obtained from LCA databases, e.g. ecoinvent (ecoinvent Centre, 2004). The ecoinvent database is available from various LCA software packages such as SimaPro, Gabi, EMIS, Regis and Umberto. For livestock production, the first task is collecting data on various inputs, outputs of products and co-products, and emissions. In dealing with time-consuming data collection from farms, most studies are based on a limited number of farms and thus are not statistically representative for the sector. The question of valid and representative data on agricultural production has been addressed by Dalgaard *et al*. (2006). The authors used farm account statistics to establish a national (in particular, Danish) agricultural model for estimating data on resource use and environmentally important emissions from farms. It should be pointed out that data for emissions from a certain production unit or a set of farms are often not available and thus need to be obtained, based on direct measurements or some modelling techniques. In practice, most emissions are estimated (rather than measured) by calculations using emission factors obtained from the literature, e.g. IPCC, but special care should be taken to ensure that the values are valid for the system under investigation. Data obtained in this way are considered adequate in many cases, and in certain cases even better than those obtained from direct measurements, which require considerable financial and time efforts. In fact, most farm emissions cannot, in real life, be measured with reasonable accuracy.

Table 7.1, which summarizes the life cycle inventory of pig meat produced in two production systems, serves as a good example to show what types of data need to be collected for a typical process. As shown in the table with the reference to Fig. 7.1, data collection is performed in accordance with the system boundary defined for the study. For the land-based outdoor pig system, in addition to input–output flows displayed at farm-gate level as a total picture of farm performance, it is often necessary to quantify the flows at herd and land level in order to obtain good estimates of on-farm emissions. In this case, manure application, as well as emissions related to it, is considered to be part of the farm model. In the landless system, manure is produced as a co-product, which is exported for use as fertilizers on other farms. The environmental consequences of manure export cannot be analysed without a number of assumptions, as detailed later.

Table 7.2 describes in detail how various emissions from the pig farm can be modelled using state-of-the-art methodology for estimating non-CO_2 greenhouse gas (methane, nitrous oxide) emissions, ammonia volatilization and nitrate and phosphate leaching. It is clear that modelling nitrogen emissions/leaching requires, in particular, detailed data on the N dynamics on the farm. For that purpose, it is necessary to analyse the nutrient balances at herd, land level and farm gate. Table 7.3 summarizes the results of the analysis. In addition, such analysis offers a means to identify hot spots in the production systems regarding nutrient losses. Note that the balance at land level for the outdoor pig system includes deposition and biological N fixation by legumes and grass-clover.

In current LCA studies on livestock products, the IPCC Guidelines are commonly used to obtain default factors to estimate methane and nitrous oxide emissions. Recently IPCC has published the 2006 IPCC Guidelines for National Greenhouse Gas Inventories, which are used for preparing national inventory estimates by all countries in the UN Framework Convention on Climate Change. It has been acknowledged by IPCC that there is often a high degree of uncertainty associated with these emission factors when applied to individual countries. In general, using country-specific emission factors is encouraged, since they provide more accurate estimates of emissions than using default factors. It is ‘good practice’ to develop country-specific emission factors for those activities that are found to be major sources of emissions for the country (‘key sources’). Otherwise, it is necessary to analyse the influence of uncertainty when using default assumptions. The Monte Carlo simulation tool in SimaPro, for example, can be used to handle uncertainty in LCA if appropriate values for variation on the parameter means can be established (Halberg *et al*., 2008).

As shown in Table 7.2, three major sources of NH_3 emissions from livestock production are manure management, crop cultivation and commercial fertilizer utilization. In manure management, further segregation into stable, storage, spreading and/or grazing is necessary in estimating emissions with a reasonable accuracy. Emission factors for ammonia vary according to soil

Table 7.1 Inventory for the two farm sub-systems of pig production: Inputs and outputs at herd, land level, and farm gate per 1000 kg live weight pigs

	Unit	Outdoor (Land-based)			Indoor (Landless)	
		Herd level	Land level	Farm gate	Farm gate	Manure export for fertilizer use[a]
Inputs	kg					
Feed						
Imported feed						
Grain cereals		1187		1187	2460	
Grain legumes (pea and lupine)		556		556		
Soy meal		24		24	540	
Fish meal		18		18	10	
Rapeseed meal		302		302		
Mineral feed (as phosphorous)					6	
Home-grown feed						
Grain cereals and legumes		1199				
Grass/alfalfa silage		874				
Grass for grazing		375				
Manure in stables (as nitrogen)			41.3			
Manure on grass (as nitrogen)			24.2			
Manure (total, as phosphorous)			11.4			
Direct land use for home-grown crops	m^2yr					
Grassland, permanent			399	399		
Cropland			4111	4111		
Direct (on-farm) energy use						
Electricity	kWh	199		199	195	
Heat	MJ	0		0	239	
Diesel (traction)	MJ		1246	1246		156
Transport of feed import	tkm					
By truck		123		123	582	
By ship		284		284	6480	

Outputs	kg					
Live pigs		1000		1000	1000	
Home-grown cereals and legumes			1199			
Home-grown forages			1249			
Cash crops			164	164		
Manure (as nitrogen)		65.5			40.0	24.0
Manure (as phosphorous)		11.4			10.1	6.1
Direct emissions[b]	kg					
CH_4				37.1	37.3	
N_2O				2.6	0.6	0.5
NH_3				24.1	11.6	2.5
NO_3				87.9		61.0
PO_4				0.4		0.4

[a]As shown in Fig. 7.1, the manure produced in the indoor system is exported for use as fertilisers on other farms. Diesel used for manure application is included since it would contribute to environmental burden (in this case, use of resource) of manure. According to environmental regulations in Europe, the substitution ratio between manure-N and fertilizer-N is 0.6, i.e. from 40 kg manure-N remaining after ammonia and nitrous oxide losses during housing and storage, about 24 kg nitrogen is available for crops when applied to fields. The substitution ratio for phosphorous in manure is assumed same as that for nitrogen.
[b]References as well as guidelines for estimating emissions are summarized in Table 7.2.

Table 7.2 Modelling of 'direct emissions' from the pig farm (related to the case study). Emissions from on-farm energy use are accounted for separately: they are calculated in SimaPro, using LCA food database (Nielsen *et al*., 2003)

Pollutant/stage	Emission factor	Reference/ guideline
CH_4 (kg.head^{-1})/		
Enteric fermentation	1.5 kg/head/year (default factor for swine in developed countries) × 145 days × (365 days/year)$^{-1}$	IPCC (2006)
Manure management	0.45 m^3 CH_4/kg VS (volatile solids) × 0.3 kg VS (default factor for swine in Europe)/head/day × 145 days × 0.67 kg/m^3 CH_4 × methane emission factor for typical manure management, e.g. 17% for slurry in-house storage for more than one month, 10% for slurry outside storage with natural crust cover and 1% for pasture (cool climatic condition)	
NH_3-N (kg)/		
On grass	0.23 × kg N surplus of the grazed area	Eriksen *et al.* (2002)
In stables	0.15 × kg manure-N deposited in stables	Andersen *et al.* (2001)
In storage	0.05 × kg manure-N collected in storage	
In field application	0.07 × kg manure-N applied to fields	
Crop cultivation	3 kg/ha × total ha grassland 5 kg/ha × total ha land for other crops	
Fertilizer use	0.03 × kg fertilizer-N used	
Direct N_2O-N (kg)/		IPCC (2006)
On grass	0.02 × kg manure-N on grass	
In stables	0.002 × kg manure-N deposited in stables	
In storage	0.005 × kg manure-N deposited in storage	
In field application	0.01 × kg manure-N applied to fields	
Crop residues	0.01 × kg N in crop residues	
Fertilizer use	0.01 × kg fertilizer-N applied	
Mineralization of soil organic matter	0.01 × kg soil N mineralization	
NO_3-N (kg)	kg (N surplus – N loss as NH_3, N_2O and N_2 – soil N immobilization)	Nutrient (N) balance
Indirect N_2O-N (kg)	0.01 × kg NH_3-N lost 0.0075 × kg NO_3-N leached	IPCC (2006)
PO_4-P (kg)	kg P surplus × 3% leaching	Nutrient (P) balance

type, climatic conditions and agricultural management practices. For the case study, it is fortunate that almost all emission factors for NH_3 derived for EU conditions (or at least, Danish conditions) are available.

In the equation to model NO_3 leaching, the variable 'soil N change' can be estimated from C-inputs from manure and crop residues and the current soil C/N, using a dynamic model namely C-tool (Halberg *et al*., 2008). The value of soil N change presented in Table 7.3 is the change predicted to occur after ten years. The figure of P leaching, 3% of P-farm gate balance, is based on the Danish average (Anon., 2003). The difference between P farm

Table 7.3 Nutrient balances at herd, land level and farm gate for pig farming systems (kg N, P/1000 kg live weight)

	Outdoor (Land-based)			Indoor (Landless)
	Herd level	Land level	Farm gate	Farm gate
Inputs (N)				
Imported cereals	44.5		44.5	38.7
Imported concentrates	18.3		18.3	38.2
Straw bedding	2.2			
Seeds		0.9	0.9	
Biological fixation		18.1	18.1	
Deposition		7.2	7.2	
Home-grown cereals and legumes	24.4			
Home-grown forages	9.6			
Grazing	3.4			
Manure in stables		41.0		
Manure on grass		24.2		
Total input	**102.4**	**91.4**	**89.0**	**76.0**
Outputs				
Home-grown cereals		24.4		
Home-grown forages		13.0		
Cash crops		5.2	5.2	
Live pigs	26.9		26.9	26.9
Straw		2.2		
Manure	65.2			
Fertilizer-N replacement				24.0
Total output	**92.1**	**44.8**	**32.1**	**50.9**
N balance (N surplus)	10.3	46.6	56.9	26.0
Losses				
Denitrification	0.9	5.4	6.3	
N_2O-N	0.4	1.3	1.7	0.7
N_2-N	0.5	4.1	4.6	
NH_3-N				
Stable and storage	9.4		9.4	9.5
Grazing		5.6	5.6	
Spreading and crops		4.9	4.9	2.1
Soil change (N immobilization)		10.8	10.8	
Potential leaching (NO_3-N)		19.9	19.9	13.7
Inputs (P)				
Imported cereals and concentrates	10.4		10.4	9.6
Mineral feed P				6.0
Straw beddings	0.5			
Seeds		0.2	0.2	
Home-grown cereals and legumes	4.5			

Table 7.3 Continued

	Outdoor (Land-based)			Indoor (Landless)
	Herd level	Land level	Farm gate	Farm gate
Home-grown forages	1.0			
Grazing	0.5			
Manure		11.4		
Total input	**16.9**	**11.6**	**10.6**	**15.6**
Outputs				
Home-grown cereals		4.5		
Home-grown forages		1.5		
Cash crops		1.0	1.0	
Live pigs	5.5		5.5	5.5
Straw		0.5		
Manure	11.4			
Fertilizer-P replacement				6.1
Total output	**16.9**	**7.5**	**6.5**	**11.6**
P balance (P surplus)	0	4.1	4.1	4.0
Potential leaching (PO_4-P)		0.12	0.12	012
P sorbed to soils		3.98	3.98	3.88

gate balance and P leached thus represents the portion of phosphate sorbed to soil particles. This sorption represents a potential loss to the environment if soil erosion or leaching occurs.

7.3.2 More about allocation and system expansion applied to the two pig production systems

The feed mixture for the pigs in the two systems is assumed to contain soy meal, rapeseed meal and fish meal, which are co-produced with soy oil, rapeseed oil and fish oil, respectively, as mentioned earlier. They offer very good examples of how co-product allocation is avoided by using system expansion. The production of soy meal from soybean results in the co-production of soy oil. By using system expansion to avoid allocation, the inputs and outputs are entirely ascribed to soybean meal, and the product system is expanded to include the avoided production of palm oil as marginal vegetable oil (Dalgaard *et al.*, 2008). In the same manner of argument, the production of fish oil as a co-product of fish meal is assumed to replace the production of palm oil. Demand for rapeseed meal, the second protein feed for pigs, would not affect the production of rapeseed meal itself since this production is determined by the demand for rapeseed oil. Instead, there is an increase in the production of soy meal and barley for animal feed as a response to a rising demand for rapeseed meal (Nielsen *et al.*, 2003). The cash crop produced in the outdoor system, handled by 'system expansion', is considered as an avoided product, the production of which is assumed to replace the comparable production process on another farm.

One issue that needs to be discussed more thoroughly is the utilization of manure as fertilizer in relation to allocation. In an integrated farming system such as the outdoor organic pig system, where manure is used for home-grown crops to feed the pigs, allocation is not necessary since this is a case of internal recycling and all emissions from the manure are allocated to the pigs. But when manure is used as fertilizer somewhere outside the pig farm, as is often the case for indoor pig production, then the issue of allocation of emissions from handling manure has to be considered. By using system expansion to avoid allocation, the manure produced is assumed to substitute a certain amount of artificial fertilizer and credits for the displaced fertilizer production are assigned to the environmental profile of manure application. One way to estimate how much fertilizer-N can be substituted by a certain amount of manure-N is to use assumptions about environmental regulations in Europe (i.e. via controlled application rate, tonnes manure-N/ha). For example, according to the environmental regulation set by Denmark, every 100 kg of N in pig manure applied to a crop should replace 60 kg N in fertilizers. The life cycle inventory data sheet for manure application to fields in the indoor pig system is thus basically constructed by inputting four variables: (i) the marginal inputs of traction energy and lubricant oil used for manure application, (ii) the avoided production of the corresponding amount of N and P fertilizers, (iii) the marginal emissions of N_2O, NH_3 to air, and (iv) the modelled leaching of nitrate and phosphate to water. Marginal energy use and marginal emissions are the extra energy use and extra emissions, respectively, when manure is substituted for fertilizers in fields.

7.4 Impact assessment

The process of impact assessment analyses the environmental burdens associated with the material and energy (input and output) flows determined in the inventory analysis phase.

7.4.1 Impact categories

For making environmental assessment of livestock production systems, five impact categories commonly considered are Global warming potential, Acidification potential, Eutrophication potential, Non-renewable energy use and Land use. Table 7.4 gives an overview of the potential impacts caused by common stressors, methods used for the assessment and equivalence factors to convert inventory results to environmental impact results. The table also briefly explains how livestock contributes to these impact categories. In particular, the magnitude of the sector's contribution to Global warming and Land use has been evaluated at 18% and 30%, respectively, by Steinfeld *et al*. (2006).

Table 7.4 Impact categories associated with stressors and how livestock production contributes to them

Impact categories	Common stressors	Equivalency factor	Method used for the assessment	How livestock production contributes to the impact category
Global warming potential (GWP), kg CO_2e	CO_2	1	EDIP (Wenzel *et al*., 1997) with update from (IPCC, 2007)	– N fertilizer production; on-farm fossil fuel (feed and livestock-related); deforestation; cultivated soils (tillage and liming); desertification of pasture; processing; transport
	CH_4	25		– enteric fermentation; manure management
	N_2O	298		– N fertilizer application; indirect fertilizer emission; leguminous feed cropping; manure management; manure application/deposition; indirect manure emission
Acidification potential (AP), g SO_2e	SO_2	1	EDIP (Wenzel *et al*., 1997)	– fertilizer production; coal-based electricity use; transportation
	NO_x	0.7		– fertilizer production; electricity use; transportation
	NH_3	1.88		– deposited and applied manure
Eutrophication potential (EP), g NO_3e	NO_3	1	EDIP (Wenzel *et al*., 1997)	– leaching from fertilizer application and manure discharge, run-off or application
	NO_x	1.35		– fertilizer production; electricity use; transportation
	NH_3	3.64		– deposited and applied manure
	PO_4	10.45		– leaching from fertilizer and manure discharge, run-off or application
Non-renewable energy use, MJ primary	Coal	18–29.3 MJ/kg	IMPACT 2002 + (Jolliet *et al*., 2003)	– fertilizer production; farming operation; feed crop processing; livestock processing; transportation
	Oil	41–45.8 MJ/kg		
	Natural gas	35–46.8 MJ/m^3		
Land use, m^2a	Occupation, arable	1 m^2a/m^2a	EDIP, LCAfood, simple (Wenzel *et al*., 1997)	– feed crop production; grazing

In relation to the method of handling co-products in LCA, 'system expansion' gives a coherent methodological alternative to allocation in which both the extra burdens (energy consumption and emissions from fields when manure is applied instead of fertilizer) and the benefits (e.g. avoided production of artificial fertilizers) are accounted for. As an illustration, Table 7.5 summarizes the inventory and impact assessment results of 1 kg manure-N applied to a field, assuming a substitution ratio of 0.6 kg fertilizer N per kg of manure N. The net environmental burdens associated with manure application are then attributed to the meat production.

Applying different assumed figures for the portion of manure N utilization in crops would lead to different results on the net environmental profile of manure application to fields. This gives rise to a sensitivity analysis to examine how a change in manure utilization rate would affect the result. In Fig. 7.2, the net environmental profiles of different assumed substitution rates resulting from the other three assumed figures for the fraction of manure N utilized in crops (0.7, 0.5 and 0.4) are compared to that from the baseline, which is set at 100 for AP and EP and –100 for GWP and non-renewable energy use. As shown, the higher the assumed fraction of manure utilized in the crop, the better the resulting net environmental profile of the manure application. The change has a larger effect on GWP than the other categories e.g. AP, EP and energy use. This is because of a large saving in GHG emissions from reduced emissions of N_2O, a greenhouse gas 298 times as potent as CO_2, when manure utilization rate is raised.

Table 7.5 Inventory and impact assessment results of 1 kg manure-N applied to field (indoor system)

	Unit	Amount	Global warming potential g CO_2e	Acidification potential g SO_2e	Eutrophication potential g NO_3e	Non-renewable energy use MJ primary
Substituted artificial fertilizers						
Nitrogen	kg	0.6	–5516	–18.9	–28.2	–29.9
Phosphorous	kg	0.15	–404	–615	–3.96	–5.94
Marginal energy use						
Traction	MJ	3.91	427	3.62	6.32	5.6
Lubricant oil	mL	9.6	6.2	0.084	0.0201	0.0921
Marginal emissions						
Ammonia	g	62.5		117.5	227.5	
Nitrous oxide	g	11.16	3325			
Modelled leaching						
Nitrate	kg	1.52			1523.4	
Phosphate	g	9.28			97	
Net environmental profile*			–2162	96	1822	–30.1

*A positive figure means a net contribution to the environmental impact potential, whereas a negative one implies a net reduction or 'savings'.

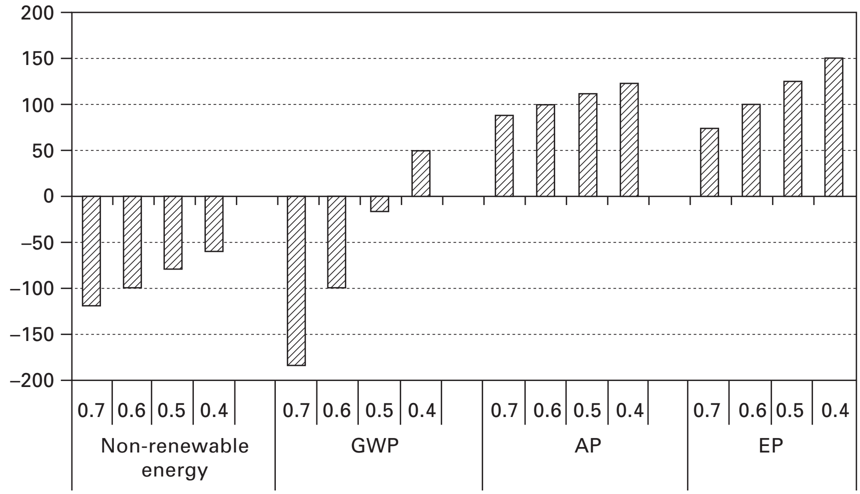

Fig. 7.2 Effect of changing assumed fraction of manure utilized in crops on the resulting net environmental profile of manure application.

7.4.2 The issue of land use and land use change

Apart from other conventional impact categories such as acidification, eutrophication, global warming and energy use, the issue of land use has recently received increasing attention, both from academic and policy-oriented audiences. In current LCA studies, this land use impact category is often quantified as how much surface area in combination with a certain time period is required to produce one unit of output (Lindeijer, 2000). This information is useful as a way to assess land use efficiency/productivity by making a comparison between different farming systems. However, the potential of land transformation and associated carbon emissions resulting from an increased demand for farm products should be considered further. It is a matter of concern that a growing demand for feed to support livestock production would impose increased pressure on land use globally (Stehfest *et al.*, 2009). Since land is a limited resource, any increase in demand would inevitably lead to land use change and this may decrease carbon stocks in vegetation and soils. According to FAONewsroom (2006), when emissions from land use are included, the livestock sector accounts for 9% of all carbon dioxide emissions derived from human-related activities.

The most important protein feed that livestock production more or less has to rely on is soy meal, which originates from Latin American countries such as Brazil, Argentina, Paraguay and Bolivia. These countries occupy a large share of the world's tropical forests. Expansion of the area under soybean cultivation would speed up deforestation and thus accelerate carbon emissions globally. The GWP of soy meal import reported by current LCA studies (Eriksson *et al.*, 2005; Casey and Holden, 2006; Williams *et al.*, 2006), however, does not include this potential increase in CO_2 emissions caused by deforestation. This is due to methodological problems in establishing a direct link between the surface area used and carbon emissions. In general, so far no agreement has been reached on an acceptable method to incorporate land use change impacts into livestock LCAs. Searchinger *et al*. (2008) have recently estimated GHG emissions in the production of corn-based ethanol in the US using a worldwide agricultural model to account for land use change. Their calculation assumed the loss of all carbon in vegetation and 25% of soil carbon following land conversion from forest to cropland. The conversion also sacrifices the ongoing carbon sequestration that would take place each year if forest is not cleared. Another assumption behind the estimate is that the land conversion costs would occur within 30 years. All background information available from the reference enables an estimate of the weighted average of carbon loss per hectare per year considering forest systems in Latin America. The figure amounts to 2 kg CO_2/m^2yr. It can be used to perform a sensitivity analysis to preliminarily quantify GHG emissions from livestock production, if potential land use change (LUC) associated with soy meal import is included.

With the help of the LCA software tool Simapro 7.1, the potential environmental impacts associated with the two pig production systems are quantified. According to the analysis, the GWP of the indoor pig system is about 1.1 times that of the outdoor one at 3.5 kg CO_2e versus 3.1 kg CO_2e/kg meat (live weight). When GHG emissions from land use change, due to an increase demand for soy meal, are factored in, the relative GWP of the indoor to the outdoor pig system is enlarged by a factor of two. This is explained by the fact that the indoor system uses a much higher amount of soy meal than the outdoor one, 540 versus 24 kg per 1000 kg pig meat (see Table 7.1).

The question is whether a substitution of locally-produced feed for imported soy meal would save GHG emissions from land use change. The substitution may induce a higher demand for local land to grow feed crops. The argument whether this extra land demand would eventually lead to land use change elsewhere so far has not yet been resolved.

7.5 Interpretation of results

In this phase, the results obtained are checked and evaluated for consistency with the goal and scope to ensure that the study is complete. Four steps that should be performed are (i) analysing results to identify significant issues, (ii) evaluating completeness, sensitivity and consistency, (iii) making conclusions and explaining limitations, and (iv) providing recommendations.

The first and also the most important step, 'analysing results to identify significant issues', is illustrated through the case study used in this chapter. For making such illustration, Table 7.6 and Fig. 7.3 are provided. Table 7.6 summarizes the comparative LCAs of the two pig production systems

Table 7.6 Comparative LCA of the two pig production systems per kg meat (live weight)

Impact category	Unit	Indoor	Outdoor
Global warming without LUC	kg CO_2e	3.5	3.1
Acidification	g SO_2e	43	58
Eutrophication	g NO_3e	255	230
Land occupation	m^2year	6.2	11.4
By soy meal		1.9	0.1
By other feeds		4.3	11.3
Non-renewable energy use	MJ primary	14.9	10.4

whereas Fig. 7.3 shows the impacts contributed from different components of the pig meat life cycle. It can be seen that the table showing the results *per se* does not provide information about on which part (or component) of the pig meat life cycle, the highest load of different environmental impacts occurs or the largest potential for improvements lies. On the other hand, further analysis on the environmental 'hot spots' as presented in Fig. 7.3 gives a clear picture of the environmental profile of the two systems and also identifies from where the relatively good or bad performance arises.

Taking the example on 'global warming impact potential', the breakdown analysis shows that 'feed import' as the dominant contributor for the indoor system (66%) makes GHG performance of the system worse than that of the outdoor one, despite the relatively low GWP from other chain components e.g. 'direct emissions' and 'direct energy use' on the farm.

7.6 Future trends

A major issue currently unresolved in LCA regarding livestock impacts is the land use. This is actualized by the increased demand for livestock products worldwide and the emerging demand for biofuels, which, in combination, put heavy pressure on the limited land resource. The occupation of land for livestock production is closely interrelated to impacts on biodiversity and, potentially, on emissions of CO_2 when land-use changes affect carbon stocks.

These impacts are related to global land use changes following the increased demand for food, feed and fuel, as well as to how the land occupied *per se* by the livestock is managed. The land issue, together with the hitherto almost entirely ignored impact on labour use, are elaborated in the following.

7.6.1 Land use changes

Recent reviews and discussions (Milà i Canals *et al.*, 2006, 2007) have addressed this issue, acknowledging that it is important to include land

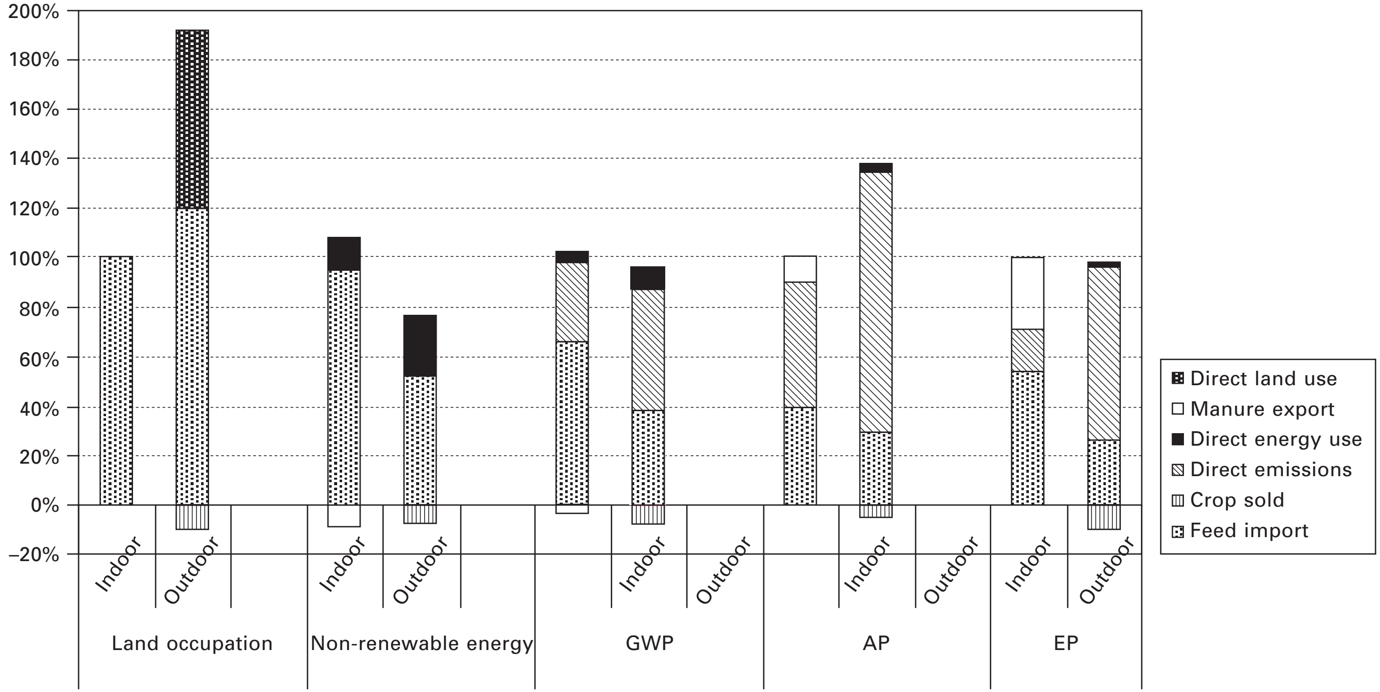

Fig. 7.3 Breakdown of the environmental impacts from the two pig production systems (impacts displayed in % relative to those for the indoor systems).

use impacts in LCA. However, a number of limitations and/or challenges have been raised regarding off-site effects (identification of marginal land use), allocation of initial transformation impacts, assumptions of natural relaxation, and knowledge about the site specific effects on soil quality and biodiversity.

While a link between the use of soy meal as protein feed in livestock production and land use change is quite obvious, based on what has been going on in Latin American rainforests (Nepstad *et al.*, 2006; WWF, 2004), the method used to quantify GHG emissions from the change has to be developed further. For this research question, better knowledge of the driving forces behind land use change is essential. In addition, it is a matter for argument whether land use change would occur elsewhere to support an increased production of local feed as a response to a boom in the demand for livestock products, e.g. meat, milk, eggs (Garnett, 2009).

Likewise, the livestock sector also poses a threat to global biodiversity, through its use of new land for expanded pasture and crop area. In relation to this particular subject matter, unfortunately, so far there have been no commonly accepted methods that reasonably translate the land use pressures into loss of biodiversity, though a great deal of effort has been undertaken to do so (Weidema and Lindeijer, 2001; Lindeijer, 2000; Mattsson *et al.*, 2000; Michelsen, 2008). Schmidt (2008) pointed out that most existing methods addressing the impacts are either too coarse regarding the differentiation between different land use types, or too narrow regarding spatial coverage. The generic characterization factors for local species diversity in Central Europe developed by Koellner and Scholz (2008), whilst useful for marginal land use decisions within this specific region, are recommended to be used just as a reference methodology for other regions, given that species diversity and the impact of land use on it can very much differ from region to region. Thus, there is an urgent need to work out further a coherent method for assessing the impacts of land use on biodiversity.

Given the fact that considerable efforts are being made to solve this issue, it will probably be typical to include the impacts of land use changes in the LCA of livestock products. This will greatly influence the magnitude of impact categories such as GWP and biodiversity of the products.

7.6.2 On-site land use impacts

The management of the land used *per se* by the livestock may also influence GWP and biodiversity impacts. Thus it is acknowledged that appropriate grazing by ruminant animals plays a role in climate change mitigation through the 'carbon sequestration', where carbon in the air is captured in the form of soil organic matter. Likewise, the use of animal manure may influence soil organic matter and thus GWP. Such impacts are often not yet included in LCA studies. However, the issue is being discussed at policy level in relation to land use policy measures, and appropriate tools to quantify such effects are presently subject to extensive research efforts. It can be foreseen that a solid base will be established to include these factors in LCA and this may change the balance in GWP among different livestock products.

7.6.3 Labour accounting in LCA studies

Soybean, an important source of animal feed, is considered a labour-intensive crop. The whole process from seeding to harvesting consumes about 700 man-hours/ha (Hsieh and Su, 1991). It is not surprising that nearly 50% of the production cost is for labour. The question whether to include labour-related flows in the system boundaries (and if yes, which method is to be used to quantify them) has already been discussed in the 1970s and 1980s in the context of agricultural energy balances. A good summary can be found in Fluck (1992). Unfortunately, methods developed so far are all controversial. Given the undeniable significance of labour inputs for labour-intensive processes, it is necessary to build up a generally acceptable accounting method to quantify labour.

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8

Challenges in assessing the environmental impacts of aquaculture and fisheries

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Abstract: Modern fisheries have developed into an industrial-scale activity during the last 60 years and today represent the only large-scale harvesting of a wild resource for food. Intensive aquaculture has had even less time to develop to an important food producing sector and continues to expand rapidly. The environmental debate around seafood, as well as consumer awareness programs, has focused on biological impacts on the marine ecosystem in terms of sustainable use of target stocks, by-catch and discard levels, as well as seafloor impacts of fishing. Additional aspects, such as the use of energy and formation of greenhouse gases caused by seafood production, have recently emerged and received increased attention. Results of seafood LCAs show that most resources, both biotic and abiotic, are used in the fishing phase, with key factors determining resource use being the condition of the stock and the fishing gear used. In aquaculture, a considerable part of the feed is crop-based; however, many of the species popular in developed countries require inputs of fish meal and fish oil and therefore depend on capture fisheries. Environmental debate about aquaculture has, in general, focused on the biological impacts taking place in the immediate surroundings of the farm facility. LCAs on the other hand have shown that feed production is the activity causing most environmental impact in the life cycle of farmed fish with impacts in areas much larger than the fish farm itself; therefore, key factors regarding the climate impact of farmed fish are the amount and composition of the feed used. The only post-landing activity that has been able to outcompete the climate impact of both industrial fishing and fish-farming is airfreight. Seafood LCAs can be used to improve the environmental performance of supply chains or to establish criteria for eco-labeling of the products. The integration of criteria regarding climate impact into existing eco-label programs is desirable to avoid certifying products causing high greenhouse gas emissions as sustainable seafood. A broader view on sustainability in certification schemes would make consumers more confident that their purchasing choices actually support more resource-efficient fisheries and aquaculture.

Key words: capture fisheries, aquaculture, stock, feed, by-catch, discard, seafloor impact, climate impact, greenhouse gas emissions.

8.1 Introduction

Capture fisheries have, over the last sixty years, seen considerable development from making use of certain fish stocks at small scale to a large-scale industrialized operation, highly advanced in localizing and utilizing the main part of the world's fishery resources. This has been achieved through technological development of fishing gear, equipment for navigation, and technology to localize fish stocks with high precision. An important explanation for the remarkable growth of capture fisheries is also found in the many types of economic subsidies that both directly and indirectly contribute to the use of marine resources beyond what is biologically and economically defendable. Marine capture fisheries have stabilized at annually landing around 80 million tonnes, a level that was reached in the late 1980s after continuous increase (all information below about the fisheries and aquaculture sector are from FAO, 2008). While there was room for some increase of catches in about 20% of the number of the world's fish stocks, 52% were fully exploited and 28% were overexploited, depleted or recovering from depletion in 2007. Hence 80% of the world's marine fish stocks were fully or overexploited that year. Most of the stocks of the top ten species, accounting for 30% of total catches, are found in this group and can therefore not be expected to increase, including popular food species such as Alaska pollock, Atlantic herring, yellowfin and skipjack tuna, as well as species that today are primarily used for feed purposes, such as anchoveta, blue whiting and jack mackerel. In fact, almost 40% of the fish caught in the sea is used for non-food purposes, if we assume that the fish used for non-food purposes comes from marine capture fisheries (i.e. not from aquaculture or inland fisheries). Non-food purposes are mainly production of fishmeal and fish oil for further use in animal rearing in agri- and aquaculture. In all, the decreases that are necessary in many fisheries more than outbalance the increases possible in others; therefore the limits of production of capture fisheries have been reached and exceeded. The alternative way to produce seafood, aquaculture, has seen impressive growth over a number of decades and continues to be the fastest growing animal food-producing sector in the world. The *per capita* supply from aquaculture increased from 0.7 kg in 1970 to 7.8 kg in 2006, at an average annual growth rate of 7%. Today, aquaculture and capture fisheries deliver equal amounts of food fish.

A substantial share of research on fisheries and their environmental impact has historically dealt with the direct biological impact on fish stocks as a result of the annual harvesting of a substantial part of their biomass. The more indirect effects on the surrounding marine ecosystem due to landing and discarding of fish, as well as seafloor impacts of fishing, have also received research attention (Jennings and Kaiser, 1998; Pauly *et al.*, 1998; Kaiser and de Groot, 2000; Myers and Worm, 2003) and these are components of what is commonly referred to as 'sustainable fisheries', a term used widely by the seafood industry and seafood eco-labeling organizations.

As regards aquaculture, the environmental debate has largely centered on the local eutrophication effects in the immediate vicinity of fish farming facilities, especially in Europe. Another research focus, particularly in North America, has been assessing the risks imposed by escaped farm fish on wild stocks of the same or closely-related species, in terms of disease and parasite transmission (Krkosek *et al.*, 2007), as well as inter-breeding of wild and farmed specimens.

While these highly relevant impacts have by no means decreased or become less important, additional aspects, primarily climate impact, have emerged and received increased attention worldwide recently.[1] Climate impact is caused by emissions of greenhouse gas emissions along the seafood supply chains from the production of fuel and feed used for fishing and fish farming, respectively, over processing, transportation, retail and consumption. It represents an additional aspect that needs to be taken into account to broaden the perspective on sustainability. If we do not, we take the risk of calling fisheries sustainable (thereby promoting them) that have high greenhouse gas emissions and are not at all sustainable from a climate perspective. In most cases, climate and biological aspects do not contradict each other and including climate impact in such cases will, as is shown later in this chapter, strengthen the conclusions on action that needs to be taken from a purely biological assessment.

In this chapter, an overview of the various types of environmental impact caused by seafood production will be given. Findings of seafood LCA studies are summarized. Examples of existing systems for eco-labeling of seafood products will be presented, along with proposals as to how these could include more environmental aspects than is currently the case. Finally, a concluding section gives an outlook that points to need for further research.

8.2 Overview of the biological impacts of fishing

8.2.1 Effects on target, by-catch and discard species

The most evident and direct environmental effect of fishing is the removal of biomass of the target species, as well as of by-catch species. A fishery can have one or several target species, which are the main purpose of the fishing activity. By-catch is the part of the catch, apart from the target species, that is also landed and sold, but is not the main driving force of the fishermen to go out fishing (Fig. 8.1). Sometimes, the by-catch in one fishery can be the target species of another fishery. For example, demersal fisheries along the Swedish west coast target both cod (*Gadus morhua*) and

[1]Climate impact, greenhouse gas emissions, global warming emissions and carbon footprint are all used as synonyms in this chapter, meaning the sum of emissions contributing to climate change weighted according to IPCC 2007.

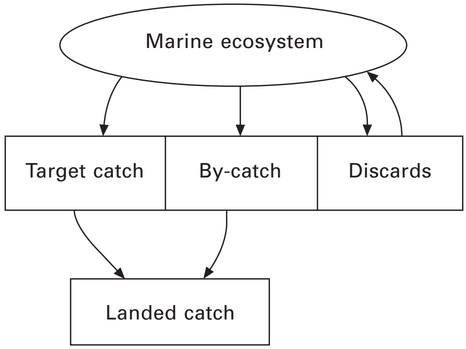

Fig. 8.1 Various components of the catch of fishermen: Target catch, by-catch and discard (illustration by Jürgen Asp).

Norway lobster (*Nephrops norvegicus*) and land by-catch species such as plaice (*Pleuronectes platessa*), saithe (*Pollachius virens*) and brown crab (*Cancer pagurus*). In the eel trap fishery in the same area, however, eel (*Anguilla anguilla*) is the target species and cod is the main by-catch species of which a part is landed.

The discard, in contrast, is not landed but is thrown overboard, and discards can consist both of undersized specimens of commercial species, of fish and evertebrate species with a current low economic value, of damaged fish and sometimes even of fully marketable commercial fish. The latter occurs when there are limiting quotas and fishermen can 'save' their quotas for a more valuable catch – a phenomenon called upgrading or high-grading. A large proportion of the discards do not survive the treatment of being fished, taken on-board, handled and then thrown back into the sea (Kelleher, 2005) and therefore discarding must be seen as a waste of limited resources. In the case of discarding undersized specimens of commercial species, it is not only a waste of biological but also of economic resources, since a part of the discards, if left in the sea, would have grown to commercial size.

The large scale removal of target and by-catch species by fishing the world's oceans has caused changes in the populations of these species. Very generally, we can say that intensive use of a fish species leads to markedly lower mean size, both in the catches and in experimental sampling of the fish stocks, size at a certain age, and lower genetic diversity in the population. How fast such changes occur, under what fishing pressure, and reversibility or non-reversibility, depends on the sensitivity of the species. Species with a high reproductive age, low growth rate and low fecundity (for example sharks and rays) are more at risk of being over-exploited than small, fast-growing species with a fast reproduction cycle, such as herring (*Clupea harengus*). For management purposes, the spawning biomass in relation to the fishing mortality and recruitment rates is evaluated for a stock in order to determine whether it is within so-called safe biological limits or not. This means that the spawning biomass is over a certain threshold level and the fishing mortality is under a defined level. If not, lower fishing mortality is recommended, resulting in lower quotas.

8.2.2 Ecosystem effects

Marine species do not live in isolation, but are tightly connected to each other in marine food webs. Therefore, a large-scale decrease of one species can indirectly impact on other species which are either prey, predators, competitors or even the prey of a prey species (Ramsay *et al.*, 1998; Anon., 2000; Pauly *et al.*, 2002).

There are theories that fishing down one species, such as cod in the Baltic Sea, has led to an increase in the main prey species which are herring and sprat (*Sprattus sprattus*) (Anon., 2000). Indirect effects can occur also further away in the food web. In the above mentioned example with cod and herring/sprat one could, for example, expect zooplankton abundance to decrease, since it is the main food item for herring and sprat, and phytoplankton to increase, since zooplankton are the most important phytoplankton grazers (Fig. 8.2). This hypothetical system is top-down-controlled (i.e. the decrease in cod abundance due to fishery affects lower trophic levels). However, such indirect effects are most often hard to follow and to connect to a single human activity as there are many other simultaneous ways that humans interfere with the marine

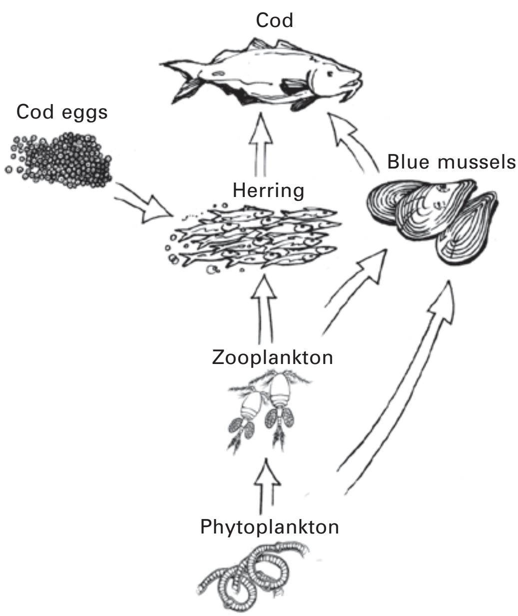

Fig. 8.2 An example of a marine food web from the Baltic Sea (illustration by Jürgen Asp).

environment, e.g by discharging nutrients into the oceans, causing local or regional eutrophication which can also affect phytoplankton abundance and growth. The latter represents a so called bottom-up-controlled system, i.e. where the change in nutrient concentrations causes changes in higher trophic levels. Additional factors can make the picture even more complex, such as the fact that sprat feeds on cod eggs, the survival of which is a bottleneck for the successful reproduction of cod in the Baltic. Marine ecosystems are generally so complex, with inter-connected food chains, natural variation in the abundance of species and variation in environmental conditions, that clear cascading effects are seldom observed (Pauly *et al*., 2002) and therefore it is in most cases difficult if not impossible to establish cause–effect chains. However, it is without doubt that fishing can affect entire food webs and that fishing down stocks and simplifying food webs leads to increasing vulnerability to natural factors such as environmental variation. Exploited stocks, hence, are more susceptible to environmental changes (Pauly *et al*., 2002). Severe over-fishing can cause complete ecosystem shifts. When for example cod, the former main predator in New England ground fisheries, was fished down to a minimum level and reproduction failed during a period (probably due to natural variation), other species such as the lower-value Arctic cod (*Boreogadus saida*) and a few species of rays and sharks, took its place and still, after almost 20 years of stopping the cod fishery completely, cod has not re-established its position as the main predator.

8.2.3 Seafloor effects of towed gear

Causing seafloor impact is inevitable when fishing for demersal species. Towed gear such as dredges and trawls, which are actively pulled along the seafloor by engine force in order to obtain the catch, affects a greater area than does passive gear such as gillnets, long-lines and traps, which are left in the sea for a period to fish (Jennings and Kaiser, 1998). A typical otter trawl used for cod trawling in Sweden normally consists of a nylon net, 50–100 m wide and several hundred meters long, connected with iron chains to two heavy iron otter boards (around 450 kg each) which make sure the trawl is dragged along or just above the seafloor (depending on the targeted species) and that the net is kept open (Fig. 8.3). Chains and ropes connect the otter boards to the fishing vessel. The trawl is pulled with a speed of 2–3 knots for maybe five to six, sometimes up to ten, hours and then hauled, emptied and set again.

The passage of fishing gear can cause physical, biological and chemical changes on and in the seafloor. Physical structures of biological origin such as corals, sponges or reef-building organisms are very sensitive to this type of disturbance. They can be crushed by the gear itself or, since they filter-feed on planktonic organisms, be damaged by the increased resuspension of sediment after the passage of a trawl. Too much sediment can clog their filtering organs and kill them. Patchy environments tend to become more

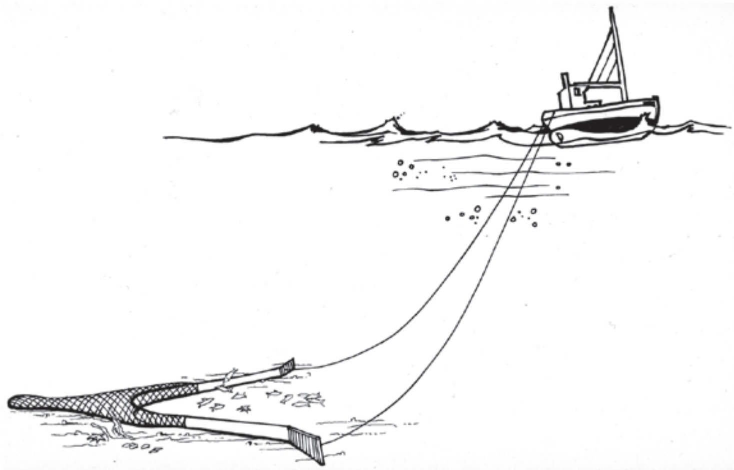

Fig. 8.3 A typical otter trawl, the most widely used fishing gear in industrialised fishing today.

uniform when such biogenic structures disappear and this reduced habitat complexity leads to reduced overall species diversity in fished areas (Thrush *et al.*, 1998; Kaiser and de Groot, 2000). Biologically, benthic organisms can be directly killed or injured by the passage of the gear. Dragged fishing gear causes turbation which exposes benthic organisms to predators together with the directly killed and injured organisms and in addition can bring back both nutrients and environmental toxins into the food web. This sudden availability of food favors scavenger species (certain fish, crayfish and evertebrate species such as molluscs and starfish), which have been reported to increase in abundance (Ramsay *et al.*, 1998; Demestre *et al.*, 2000), while long-lived, sessile and fragile marine species often decrease in abundance at trawled sites (Olsson and Nellbring, 1996; Hall, 1999; Anon., 2000; Bergman and van Santbrink, 2000). The biological impact of demersal trawling depends on the frequency of the activity and the habitat type impacted. The geographical distribution of fishing effort in different habitat types and thereby the intensity in fishery-related seafloor impact is rarely known (Auster and Langton, 1999; Jennings *et al.*, 2000). For regions where seafloor maps exist and fishing effort is reported with high geographical resolution, analysing these data in a Geographical Information Systems (GIS) could be valuable to quantify the overall fishing intensity in an area or in a specific habitat type (Nilsson and Ziegler, 2007).

8.2.4 Anti-fouling

The hull surface of fishing vessels is kept free from growth of marine organisms by applying anti-fouling paints in order to minimize friction during operation and thereby energy use for propulsion. Such paints contain toxic agent(s) that prevent organisms such as barnacles, algae and mussels from settling and growing on the hull. They are applied once or twice a year and the active substance slowly leaks out into the surrounding water, efficiently preventing fouling. Toxicity of these substances is broad and many marine organisms that are not targeted are affected. The earlier most common substance, tributyltin (TBT) was phased out globally in 2003 due to its documented negative impact in the marine environment. In many countries it has been replaced by, e.g. mixtures of copper and the herbicide Irgarol. However, in order to avoid these broadly toxic substances in the future, research is ongoing trying to find 'smarter', i.e. more specific, ways to target the species that cause the fouling problems or non-toxic ways of keeping the hull free from marine organisms. Making the hull surface unattractive for marine larvae to settle on is one possible way. Mechanical cleaning and docking in freshwater every now and then are traditional ways of getting rid of marine organisms on ship hulls.

8.3 Overview of the biological impacts of aquaculture

8.3.1 Intensive or extensive

There are numerous methods for aquatic farming. Shrimp farming can, for example, be done in basins on land or in coastal constructed ponds in former mangrove swamps, where shrimps are hatched and grown until they reach commercial sizes. All feed is added, excretion products removed and medicines used when needed. Such systems are called intensive farming systems (D'Souza and Colvalkar, 2001). The contrary are extensive-traditional farming systems, where wild shrimps are fished or trapped during high tide and then cultivated in natural ponds in mangrove areas where they are left to grow and reproduce without, or with very little, addition of feed. Extensively farmed shrimps feed on planktonic organisms just as wild shrimps do, and they are not treated for diseases or to increase growth. Organism densities are much lower in extensive systems and therefore the need for disease control is lower. There are also many stages in between intensive and extensive farming; more intensive normally means higher yields and more input of energy and chemicals and resulting emissions. Land use for tropical shrimp farming is described in the next section (8.3.2).

Mussel farming is often done in extensive systems by placing a rope in the water at the time when mussel larvae abundance is high, when the larvae will settle on any free surface such as the rope. The mussels are then left growing, feeding on the planktonic organisms passing with water currents, for a year or two until they are harvested. In seabed farming, another method used, small mussels are fished and then placed in suitable areas in high densities where they can be easily recollected with dredges after a growth period. Harvesting of mussels can often be done continuously and is restricted only by blooms of harmful algae which periodically can make mussels toxic.

Farming of finfish in industrialised countries is normally done in intensive farming systems based either on land, in coastal areas or lakes, which is described in the next section.

8.3.2 Land based or marine

Aquaculture is either done in artificial ponds or basins on land, when freshwater or seawater (depending on the species to be farmed and on the life-cycle phase of the species) is pumped to the facility, or in the sea where fish or crayfish are caged in some way. In the case of land-based units it is easier to keep track of discharge of residual water and its content of nutrients and other emissions, since there is a single wastewater tube. It is also easier to control spreading of infections and escapes. Sea-based systems require less energy for pumping water and building of ponds or basins, but are more difficult to control regarding nutrient emissions, disease spreading, and escapes, since the farming is practically done in the seawater, with only a net or cage separating the farmed fish from the wild stocks (in the case of farming domestic species).

Shrimp farming is done in ponds, either natural ones or constructed ponds in former mangrove areas. In the case of intensive farming where mangroves are cut down and artificial ponds constructed, land use will be a limiting environmental factor as the ponds can be used for only a couple of years, after which they are abandoned and new sites have to be occupied. Land use per yield is greater in extensive systems; on the other hand, the land can be used for a longer time and is not left as devastated as former intensive shrimp ponds. An additional problem is the salinization of soils (which, if not used for aquaculture, could have been used for agriculture) by the sea water pumped through aquaculture ponds and leaking out, especially as aquaculture is practised further and further away from the coast, e.g. in Thailand (R. Mungkung, pers.comm.).

8.3.3 Species cultured

The species to be cultivated is also an environmental issue, since the feed demand and sensitivity to disease and environmental settings differ between species and imply certain environmental characteristics of the farming activity. Feed is discussed in the following section (8.3.4). When non-domestic species are farmed, the environmental risks of such introductions are that new diseases or parasites might be brought in with the brood to be cultured. Escapees of foreign species, or of genetically distinct specimens of domestic species, can pose an ecological risk if they succeed in surviving in the wild and reproduce their genes. The introduction of new species or new properties in domestic species (or simply loss of genetic diversity) can change sensible ecological linkages of marine food webs such as competition, predation and disease/

parasitism. Escapees of domestic species such as salmon (*Salmo salar*) or rainbow trout (*Oncorhynchus mykiss*) can spread lice and diseases to wild salmon and trout stocks and this has been shown to be a threat to the health of the natural stocks of these species (Krkosek *et al.*, 2007). Diseases and parasites naturally spread more easily in fish farms due to the high densities of individuals in the cages.

8.3.4 Feed and eutrophication

Most fish species considered for aquaculture and that are domestic in northern, temperate oceans are carnivorous, whereas some fish species farmed in tropical countries are herbi- or omnivorous. When farming carnivorous fish species, about half of the feed is normally based on fish meal and oil, which in turn is based on capture fisheries of small pelagic fish species. These dedicated feed fisheries for small pelagic species have a bad environmental image among consumers and for this reason, and also because some pelagic stocks are considered to be overfished, the aquaculture industry is trying to decrease its dependency on fisheries for feed production. In the light of anticipated growth of the aquaculture sector, decreasing this dependency while maintaining the quality of the farmed fish, e.g. with regard to contents of omega-3 fatty acids, is urgently needed. Alternatives are, for example, soy-based protein, leading to other types of environmental impact.

Optimizing feed dosage and feeding technique is crucial in order to minimize loss of feed, which ends up as nutrient emissions from the fish farm. Excretion products from the animals cultured always do so and, in order to avoid local eutrophication problems around the production facility, wastewater treatment as well as proper localization of the marine farm (in areas with sufficient natural water exchange) is very important. Local eutrophication can otherwise lead to oxygen depletion and formation of hydrogen sulphide in the bottom-near water layers, altering the benthic community beneath and surrounding the aquaculture facility. It should also be mentioned that extensive farming, i.e. when no feed is added and the species is feeding on planktonic organisms, can contribute to decreased eutrophication by removing biomass from the water column. For example, mussel farming has been suggested as a method to improve water quality in a eutrophied area (Haamer, 1996) and as a measure to mitigate eutrophication.

8.3.5 Chemicals

The use of antibiotics in aquaculture is common although amounts used, e.g. in Norwegian salmon farming, have been reduced considerably. Antibiotics are not always broken down easily in nature and are potentially bioaccumulating. The spreading of anti-bacterial substances in nature also creates a risk of the development of resistant bacterial strains and of the spreading of resistance to other bacteria than those originally targeted.

Vaccines are today used on a routine basis to prevent the outbreak of some common diseases in salmon farms. To combat the problem with salmon lice, various chemicals are also used either as dipping treatments or as feed additives. The environmental effects of these chemicals are not fully known and, in addition, it seems that the problem is growing rather than decreasing due to developed resistance against treatments (FHL, 2010). To prevent settling and growth of marine organisms on the equipment used in aquaculture, these are often treated with anti-fouling agents such as copper (compare Section 8.2.4 on anti-fouling).

8.4 Energy use and carbon footprint of seafood supply chains

8.4.1 Energy use and greenhouse gas emissions in fishing

A number of factors affect energy consumption per kilo of fish landed. Two of the most significant are fishing gear and species biology. Of course, the latter is linked to the design of fishing gear, but it should be noted that schooling pelagic (mid-water) species, such as herring, offer better potential for large-scale, energy-efficient fishing compared with seafloor (demersal) fish or shellfish, which live less densely and close to the seabed. Thus, fishing that uses gear such as purse seines and pelagic trawls to catch pelagic species are often ranked as energy-efficient. In many cases, several fishing methods are deployed to catch a particular species, resulting frequently in major differences in terms of energy efficiency (see, for example, Thrane, 2006). Examples include Pacific salmon fisheries that deploy purse seines, trolling or gillnets and Norway lobster fisheries using seafloor trawls or creels (Fig. 8.4). Flatfish can be caught using gillnets, bottom trawls or beam trawls with beam trawling requiring 15 times as much fuel per kilo of flatfish landed compared to Danish seine (Thrane, 2006).

Energy efficiency depends, in part, on the fishing technique. The terms ‘active’ and ‘passive’ fishing methods are commonly used, with active meaning that fishing gear is actively pulled through the water or along the seafloor (as in trawling and dredging), while passive, or fixed gear, means that the gear is laid out and emptied one or two days later (such as gillnets, long lines, creels and pots). Occasionally, bait is used – normally from pelagic fisheries – to attract the target species to the fixed gear (such as creels, pots and long lines). Generally, fixed gear types are more energy-efficient (Thrane, 2004, 2006; Ziegler and Valentinsson, 2008; Ziegler *et al.*, 2009).

Although pelagic trawling is an active fishing method, it is one of the most energy efficient because fishing is done in the water column rather than along the seafloor. The fact that many pelagic target species are schooling fish also contributes to a lower fuel-per-catch ratio.

Fig. 8.4 Energy utilized in the production of one kilo of cooked Norway lobster (unshelled) from the catch to the consumer. Creel-fishing and conventional trawling represent alternative fishing methods. Source: Ziegler and Valentinsson (2008).

The stock situation is another key factor that can reduce fuel efficiency, even with the use of fixed gear. LPUE (Landings Per Unit of Effort) is a measure of the fish volume landed per unit of expended fishing time; a common unit is kilos landed per hour fished. Given the same fishery (i.e. a combination of target species, fishing gear, geographic area and nationality of fishermen and management system), a low-density fish stock means that more time is required to accumulate the same catch compared with fishing at a higher density. In other words, in addition to fishing method and species biology, the stock situation is a key factor in determining the energy efficiency of fisheries. It is difficult, if not impossible, to fish an over-exploited stock in an energy-efficient manner.

Figure 8.5 shows that, in the early 1980s, four times more cod per hour were caught in Swedish trawl fisheries in the Baltic compared with 2005. This means that back then it was necessary to expend only 25% of the time required to harvest a given catch, which heavily influences energy utilization per kilo of fish landed. Tyedmers (2004) noted that energy efficiency in many fisheries worldwide has declined in recent decades, despite parallel technological progress that made it easier to localize favorable fishing grounds. This is probably due to the increasing fishing pressure on many stocks during the same period (Tyedmers, 2004). The same conclusion was drawn in a recent analysis of a number of Norwegian fisheries between the years 2001–2004 (Schau *et al.*, 2009). There are indications that the steep increases in fuel prices after 2004 together with improved conditions of stocks have led to higher fuel efficiency in fisheries since then (Winther *et al.*, 2009). Hospido and Tyedmers (2005) also showed the potential for

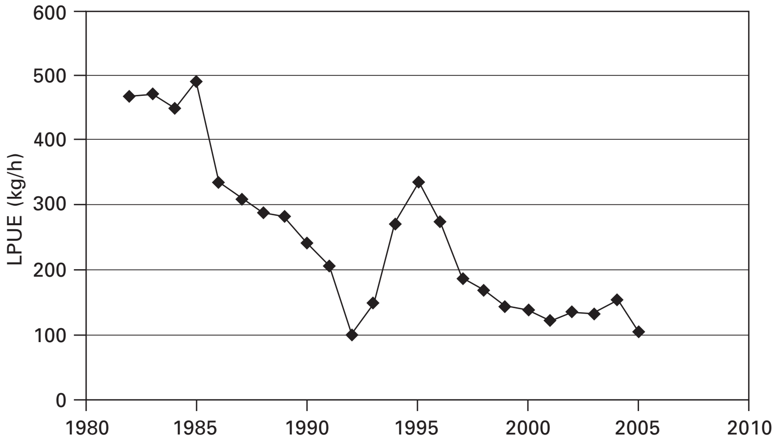

Fig. 8.5 Cod landings per trawl hour in Swedish trawl fisheries in the Baltic Sea, 1982–2005 (data from the Swedish Board of Fisheries).

decreased greenhouse gas emissions when a stock is rebuilt to a biologically sustainable level.

While the climate impact of fisheries is dominated considerably by emissions from onboard diesel combustion, the leakage of refrigerants from onboard cooling equipment (for cooling, freezing, ice-making, etc.) sometimes constitutes a considerable portion of total emissions. This is the case when the refrigerants used have a high climate impact and leakage rates are higher than minimal. As opposed to the progress made in onshore operations, fishing and shipping lag behind in the phasing out of freons with a high impact on the ozone layer. HCFCs – among the most ozone-depleting types of refrigerants (and which also have a substantial climate impact) – are still common onboard fishing vessels. This factor, combined with the fact that mobile cooling and freezing units, especially in the maritime environment, are subject to higher leakage than onshore fixed units, represents a substantial contribution of onboard refrigerants to the overall climate impact of wild-caught fish-based products. The phasing out of HCFCs in maritime applications was scheduled for completion by 2010, which means that the global fishing fleet is faced with major changes regarding this aspect.

One problem is that while the HFCs that offer an alternative are preferable in terms of ozone depletion, they have an even higher climate impact. Consequently, a transition from ‘synthetic’ to ‘natural’ refrigerants such as ammonia/carbon dioxide-based systems is preferable from the environmental viewpoint, and is readily feasible in terms of technology (SenterNOVEM, 2006; UNEP, 2000; NMR, 2000). An additional benefit for onboard freezing is that we can expect a reduction of 20–30% of the energy used for freezing which is generated by the diesel engine can be expected, since ammonia is a more effective cooling agent than is R22.

8.4.2 Energy use and greenhouse gas emissions in aquaculture

Energy consumption in aquaculture begins with the energy required for the production of feed and other inputs used in fish farming (such as net pens and anti-fouling agents). Farmed mussels require no feed input, as opposed to farmed fish, since they feed on planktonic organisms. Some fish (such as carp, tilapia, and pangasius) are omnivores and can grow without animal-based feed ingredients, which means they can be fed using agricultural products or residues. However, a small proportion of fish meal and oil is often added to the diets to enhance growth. Other species (such as cod, salmon, turbot, halibut, and rainbow trout) are predators that, being higher up in the food web, require some marine-based feed, e.g. a combination of fishmeal and fish oil. This marine feed derives either from by-products from fish processing or from targeted fishing for small pelagic species such as herring, sprat, sand eel, blue whiting and anchovy in various parts of the world.

It has been found that energy use for the production of marine-based ingredients is generally higher per kilo than for vegetable feed ingredients (Pelletier and Tyedmers, 2007). The same study also noted that feed represented more than 90% of the total energy utilization in the production chain from feed production to ready-to-eat salmon. Thus, the energy used directly in the form of electricity for pumping air and water in fish farming, and that used in cooling and freezing equipment, plus fuel for transport, represents a smaller share of the total energy utilized in getting a salmon to the consumer's plate (Troell *et al.*, 2004; Tyedmers *et al.*, 2007; Pelletier *et al.*, 2009). For the crop-based feed inputs, biogenic emissions formed in agriculture that are not related to energy use become more important. Emissions, particularly of nitrous oxide, are generated in the handling of manure and in the production process of fertilizers, but also as a result of microbial processes in the soil.

8.4.3 Environmental impact of post-landing phases

Looking at the entire production chain from fisheries to fish consumption, a common conclusion from all seafood LCA studies done in industrialized fisheries is that the fishing phase accounts for the greatest share of total energy utilization in the form of onboard fuel combustion during fishing (Thrane, 2004, 2006; Ziegler *et al.*, 2003; Ziegler and Valentinsson, 2008). This applies *also* when fishing is relatively energy efficient and the product is prepared and packaged in a relatively energy-intensive manner (such as marinated herring, Christensen and Ritter, 1997).

However, a much-discussed aspect is the significance of long-distance transport. In addition to the actual distance that the raw material or product is transported, there are several other major factors to be considered. One of these is the transport mode, meaning whether transport is by truck, train, ship or aircraft. Other factors include vehicle size, load capacity used and cooling requirements.

Seafood is a very special type of food in terms of transport. Fresh fish continues to be transported on ice in boxes and – due to odor and melt-water – it cannot be transported in combination with other food products, making its transportation inefficient from the energy viewpoint. Also, small sub-optimally loaded trucks are used. This means that the climate impact of transporting a kilo of fresh fish using a small fish truck can be of the same magnitude as transportation in frozen form on a fully loaded container vessel from Southern Africa to Sweden, despite the inclusion of the extra energy and refrigerant required to keep the product frozen.

Of course, the form of the fish product (fresh, frozen, smoked or preserved) considerably influences how transport to the consumer is undertaken. If the market requires fish from the other side of the globe, it must be air-freighted. The use of air transport gives rise to the only known example in which transport is of greater energy significance and overshadows the fishing operation from the climate viewpoint (Winther *et al*., 2009). Otherwise, the proportion of frozen seafood products total carbon footprint represented by transport is typically under 20%.

A large share of fish catches worldwide is conveyed in frozen form to Asian countries (mainly China) for filleting before being transported back as finished products. Thus, the trip that fish make from their initial landing (in Northern Norway, for example) before moving via Rotterdam to Qingdao, in China, and back again on the same route to a European consumer, resulting in a total distance of 42 000 km – more than the circumference of the planet – is literally a round-the-world trip. Also, the fish volume transported to China is almost 50% more than the final fillet volume transported from the country. So those who criticize imports of fish fillets from New Zealand (24 000 km), Chile or Vietnam due to long-distance transport should remember that a sizeable portion of the fish caught in local waters is processed in Asia.

A factor complicating this argument is that the fillet yield from filleting in China is higher than mechanical filleting of the same quality in Norway, for example. Thus, more fillets per kilo of fish are gained from the initial volume entering the processing plant. This difference means that a smaller fish volume needs to be caught to get a kilo of fillet to the consumer, which is important from a biological resource use perspective. Of course, the optimum situation would be to avoid transportation and fillet the fish close to the fishing operation and consumer – using manual or mechanical means to provide the maximum yield.

8.5 Eco-labeling of seafood

8.5.1 Eco-labeling of wild-caught seafood

A number of organizations impose criteria covering the eco-labeling of wild-caught seafood. The UK based Marine Stewardship Council (MSC) and the Swedish KRAV are two examples. A common feature of these bodies is that they use a third party in assessing whether or not a fishery meets the criteria. The three organizations also assess all criteria regarding the utilization of fish stocks, the by-catch volume and the effects on other aspects of the marine ecosystem associated with the management system. In brief, they ensure that fish stocks are used in a sustainable manner or are progressing in this direction; that fishing does not give rise to unacceptable impacts on the surrounding ecosystem; and that there is a functioning management system within which appropriate measures are taken if the stock situation deteriorates (Thrane *et al*., 2009). MSC certified fish is produced and sold around the globe, while KRAV has Sweden as its main market.

In addition to these certification systems, there are a number of consumer guides, including those designed by WWF in Europe (in Sweden and elsewhere) and Monterey Bay Aquarium's Seafood Watch, which primarily deals with seafood in the North American market. These players assess the stock situation, ecosystem effects on by-catch species and the seafloor. Using such data, fish species are categorized using a green, amber and red rating code. While they currently do not include energy use nor climate impact, these could certainly be included and are often correlated to stock status, seafloor impact and discard levels (Thrane, 2006; Ziegler, 2006).

8.5.2 Eco-labeling of farmed seafood

A number of eco-labeling organizations provide criteria for aquaculture. In the Swedish market, for example, there is KRAV/Debio (with rules drawn up in cooperation between the Norway-based Debio and the Sweden-based KRAV) and Naturland (Germany-based Association for Organic Agriculture). MSC does not currently cover aquaculture, which the Soil Association (UK) does. However, the WWF has recently started an initiative to form an Aquaculture Stewardship Council (ASC) that will certify seafood following the standards developed in a number of Aquaculture Dialogues. The criteria cover requirements regarding fish farm location, water quality, fish feed origin and dosage, fish density in cages and other aspects of fish welfare, medication and other use of chemicals in operations, measures to guard against cage damage, origin of fish for stocking, keeping of records, transport, and slaughter.

The consumer guides noted in the previous section (8.5.1) also assess farmed seafood; for example, in terms of feed composition and local eutrophication effects.

8.5.3 Improvement potential for eco-labels

KRAV (the Swedish certification system for organic production) since 2010 has operations-based criteria concerning climate impact of seafood products integrated into the existing rules for eco-labeling of wild-caught fish. 'Operations-based' means that they encompass requirements governing the production method, just as in the case of KRAV's rules for organic production.

For seafood products from capture fisheries, fishery rules entail that the stocks are sustainably fished; that fuel consumption in fishing is less than a certain level per kilo of fish landed; and that synthetic refrigerants are not used onboard.

In the case of farmed fish, the rules have not been introduced yet but are suggested to cover the amount of feed required per volume of fish produced and feed composition with regard to proportion of animal-derived ingredients. The fuel consumption for designated feed fisheries should be more than ten times lower than the limit for fish for direct consumption. This represents a way to integrate knowledge gained from Life-Cycle Assessment studies without introducing a new label and without confusing the consumer with carbon footprint figures on food packages.

8.6 Conclusions and future trends

A more widespread use of eco-labels promotes more sustainable forms of seafood production, especially when more dimensions of sustainability are integrated. Optimally, it gives the seafood consumer a clear message about which products are preferable from an environmental point of view (Thrane *et al.*, 2009). Life-Cycle Assessment provides a methodology to assess both initial performance and improvement along the way for some types of impact. If we fail to integrate the various dimensions of sustainability, we risk sending very mixed and confusing messages to consumers and also promoting products that do well with regard to the dimensions covered by the label but may be highly unsustainable from other perspectives (Thrane *et al.*, 2009; Pelletier and Tyedmers, 2008). A concrete example would be a dolphin-safe or carbon footprint label on a can of bluefin tuna because the fishery doesn't catch any dolphins or uses energy-efficient gear, not considering the severe condition that bluefin tuna stocks are in. However, the responsibility cannot be completely off-loaded on consumers (Jaquet *et al.*, 2009). All stakeholders in the seafood supply chain play a role in improving resource efficiency. Retailers, wholesalers and the processing industry need progressive fish sourcing policies and to provide incentives for suppliers to improve performance. Fishermen themselves can, of course, influence the environmental performance of fishing operations as it has already been shown that the fishing activity is responsible for all of the biological impacts and most of the climate impact occurring throughout the supply chain from capture fisheries to seafood products. The high variability in environmental performance between individual fishermen demonstrates a scope for improvement on the level of the individual fishing vessel (Winther *et al.*, 2009). However, fishermen are limited by the framework set out by the fisheries management system that tells how and where to fish, what to land, and how to report. It distributes the right to fish and limits catches in different ways and therefore plays a very important role in determining the environmental impact of fisheries. Much more focus is needed in evaluating the environmental impact of single measures within the management system and introducing climate impact as one of many factors to be taken into account in fisheries management in the future.

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Part III

Current and future applications of Life Cycle Assessment and related approaches

9

Towards sustainable industrial food production using Life Cycle Assessment approaches

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Abstract: The environmental impact of food products occur in all nodes of the supply chain; from production of inputs to agriculture, through farming, industry and retail to households. Impacts at one node often depend on activities at other nodes, and improvement options for food industry might often involve changes that result in large savings at other nodes rather than for the industry's own emissions. This must be considered when assessing food products, and Life Cycle Assessment (LCA) is a well established tool for such analyses. However, for novel processes or products, the LCA methodology needs to be adapted to the specific circumstances and questions posed. In this chapter, two case studies where LCA has been used are presented, exemplifying the value of adopting a food chain perspective. In addition, a methodological proposal for a working framework is given.

Key words: LCA, novel food processes, novel food products.

9.1 Introduction

Industrial development generally involves uncertainties about profitability and always involves a certain risk to the company. There is a broad range of considerations to be taken along the decision-making process and these considerations are often complex, with many parameters involved. Although the most important parameter for industrial decision makers obviously is the economic return in both short and long term, another aspect has become more important recently: the environmental consequences of processing development. This has also an economic aspect, as legislation is becoming more stringent, which means increased costs for less environmentally efficient production methods. In addition, the consumer interest is becoming more focused on the environment, hence environmentally-conscious production systems are important for producers.

Traditionally, the main environmental concern has been energy use, and sometimes water use, of the industry itself. However, with increasing consumer and societal interest, it has become more important to look at the consequences in a life-cycle perspective; no producer can afford to face the risk of increasing the environmental impacts of their activities. The up-side is that certain investments that would have been disregarded due, for instance, to increased internal energy use, can be shown to be the best option in a life-cycle (or food chain) perspective, since they facilitate large savings in other parts of the chain, thus leading to an overall decrease of the environmental impacts of the products. Life-cycle improvements in the food chain can be of very different kinds, ranging from reduced need for fertilisers in agriculture through energy savings in industry to reduced wastage at the retail or household level. The impacts of one node in the chain are often affected by other nodes, both upstream and downstream. So, the introduction of a new, more energy demanding, process that increases shelf-life of a food product will lead to reduced wastage at the retail stage, and in order to evaluate the environmental consequences of that change the reduced need for raw materials needs to be taken into account along with the increased energy use in industry and less need for waste management due to retail wastage.

Conclusively, in order to efficiently consider environmental aspects when developing new possible processes and products, a holistic life-cycle perspective is necessary. Life Cycle Assessment is a very suitable tool for this, but it needs to be applied considering the context: new products, new processes and future systems.

9.2 Case studies

As examples of the importance of having a systems perspective and also to show some different approaches and applications, two case studies are presented; dairy processing and meat rendering. In Sonesson (2009), two other case studies are presented, focusing more on product development, whereas the case studies in this chapter focus on novel processes.

9.2.1 LCA of changed dairy processing

In work presented by Berlin and co-workers (Berlin *et al.*, 2007, Berlin and Sonesson, 2008), results from a systematic analysis using LCA on dairy processing was presented. The research question asked was 'Is it possible to reduce the environmental impact from the dairy industry?' This question arose from results of several LCA studies of dairy products, showing that the agricultural phase was the absolutely dominating one in a life-cycle perspective, but still the dairy companies wanted to investigate other possibilities.

The study focused on the production management of a line for flavoured yoghurt, the rationale being that it involved a large number of products being produced on the same line, leading to a lot of product change-overs and cleaning operations, which had been found to have a great environmental impact for dairies in previous studies (Eide-Høgaas, 2002). The approach was based on LCA. The life cycle assessed starts with the activities at the agricultural level and ends at the factory gate when the packed yoghurt is leaving the dairy. The activities beyond that in the life cycle were similar for all studied alternative production plans studied and were therefore not included within the system boundary. Data for the background system; dairy farming, energy system, packaging production, water production and waste management was assumed to be constant per unit used, while the foreground system was modelled using actual data from the process line. The study performed compared different production planning schemes, i.e. in what order and in what amount to produce the weekly production mix, trying to minimise product shifts, since they involve energy-, detergent- and water-use, and also product losses. In Fig. 9.1, the systems boundaries are shown, and it is important to note that even though primary data and changes in the production system occur in only a small part of the system, still a life-cycle perspective is maintained.

The results showed that the fewer product changes there were, the smaller the environmental impact. The largest part of the improvement was due to decreased wastage, leading to a reduced need for milk produced at farms. In fact, for some environmental impact categories the improvement made possible by a changed production pattern was larger than the total environmental impact for the dairy company's total impact. For example, the reduced eutrophication impact due to less raw milk used was larger than the total eutrophying emissions from the dairy's part of the life-cycle impact. In Table 9.1, results from one of the two case studies on yoghurt are shown. The 'Reference A', 'Goal A' and 'Future A' refer to the three different scenarios assessed of the production sequence for producing the same amount of products in the very same production line during a week, just changing the product order for each product to be produced. The Reference A was a one week sequence used in the dairy when the study was performed. In the Goal A sequence, the product frequency of two times per week was used and in the scenario called Future, the product frequency was one to two times per week depending on shelf-life of the product. As can be seen in Table 9.1, production according to all sequences causes waste. Nevertheless, there is also obviously a great reduction potential in using a waste-minimised sequence. In fact the potential was classified to be so great by one of the dairies involved in the project that they changed their product sequence in line with Berlin and Sonesson's suggestions and were very satisfied with the result, both from a cost and from an environmental point of view.

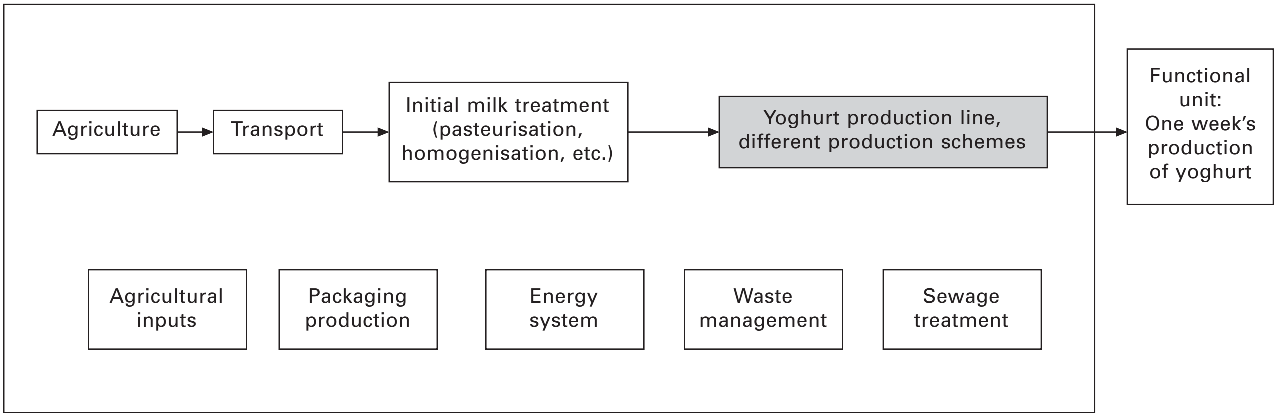

Fig. 9.1 Systems boundaries and functional unit for the yoghurt case study (Berlin and Sonesson, 2008). Shaded box is considered the core system; other boxes constitute the background system.

Table 9.1 The sequence-caused waste from the different sequence scenarios, with the waste attributed to its source (Berlin and Sonesson, 2008)

(kg)	Reference A	Goal A	Future A
Product change	6743	4776	3069
Daily cleaning	3010	2290	1775
Laboratory tests	486	408	264
Discarded containers	1476	1224	792
Total amount of waste	11 715	8698	5900

Translating the result in Table 9.1 to the environmental impact of the production of all the products produced in the sequence, gives an environmental reduction potential of 1.3% with Goal A compared to Reference A, and 2.6% with Future A. These figures may be considered low at first glance but, in fact, they correspond to a reduction of 33% of the direct global warming potential from the dairy and for the impact of eutrophication 1.3% corresponds to a reduction of 310% of the direct impact of the dairy. This is due to the low amount of emitted emissions with impact on eutrophication from the dairy.

The fact is, at least for dairy products, an individual actors' possibilities to induce environmental improvements often means that the actual improvement takes place somewhere else in the chain, which was clearly shown by Berlin *et al.* (2008). That study analysed all post-farm actors' possible improvement options and analysed them from a life-cycle perspective. The results showed that a reduced wastage was the most powerful option for all actors, but the actual improvement took place within agriculture. For the dairy industry, energy savings and improved logistics (two of the most common areas of focus for environmental work within food industries) were shown to be much less efficient from a life-cycle perspective. This conclusion would not have been possible without having a systems perspective and applying LCA in the analysis.

9.2.2 LCA of inclusion of new processing equipment in meat rendering

In this case study, new membrane filtration technology as an alternative treatment of a liquid flow in a meat rendering plant was analysed. The motivation was that the company sought to increase the economic return from their activities, and by introducing membranes, the product mix would change, and new, improved products could be obtained. However, the membrane treatment was expected to use more energy within the process, so the question was how the environmental impact would be affected. This study is reported by Davis *et al.* (2007), and a full description of the analysed process and the results are given there.

Three different process alternatives were analysed and compared with the current system, from technological, economical as well as environmental points of view. The alternatives involved changes in several flows and moreover, the product mix and product quality changed between alternatives. In all cases there were four inflows: Pig bones, cattle bones, pig process water and cattle process water. The products obtained were: Meat and bone meal (fertiliser), technical fat, bone chips (gelatine production), protein powder (ingredient for human foods), and calcium powder.

Alternative 1: By a combination of ultra-filtration (UF) and nano-filtration (NF), the pig process water was used to increase the amount and quality of the protein powder, and at the same time the amount of calcium powder decreased.

Alternative *2*: The same technology was used but the product mix was changed by filtering all of the pig process water. This involved a larger investment and a higher energy use, but gave even better quality protein powder, since only retentate from the NF was used for producing the protein powder (as opposed to the present system where a share of the evaporated pig bones is also used). The actual amount of protein powder was not increased.

Alternative 3: The production of protein powder was maximised by combining Alternatives 1 and 2, i.e. using all pig process water for filtration and also using pig bones for production of protein powder.

The basic functional unit was defined as a certain amount of pig and cattle bones received at the plant in one year. The logic for this is that the basic business for the company is to take care of slaughterhouse waste. At the same time the alternative processes obviously fulfilled different functions, both different amounts of different products, but also different quality and hence value. This must be considered in the environmental assessment. Therefore, and following the recommendations introduced above, it was decided to use the economic value of the total production as an alternative functional unit – this means the environmental impact per year divided by the yearly profit of all products produced by each alternative. The reason for using this approach instead of systems expansion was that it was considered impossible to identify what products would be replaced by the improved quality; it would simply increase the quality of the final products in a way that was not possible in any other way. Mass allocation was considered irrelevant due to the completely different nature of the products (very different dry matter content, different nutrient content, etc.).

The results generally showed small differences when compared with the basic functional unit, which could be said to be the traditional way of assessing new technology. The alternatives had slightly higher GHG emissions since membranes use more electricity but even more in that larger amounts of thermal energy was needed for spray drying to produce more protein powder (Fig. 9.2). The results looked similar for other impact categories, generally small increases except for eutrophication, where the alternative systems lead to a small decrease.

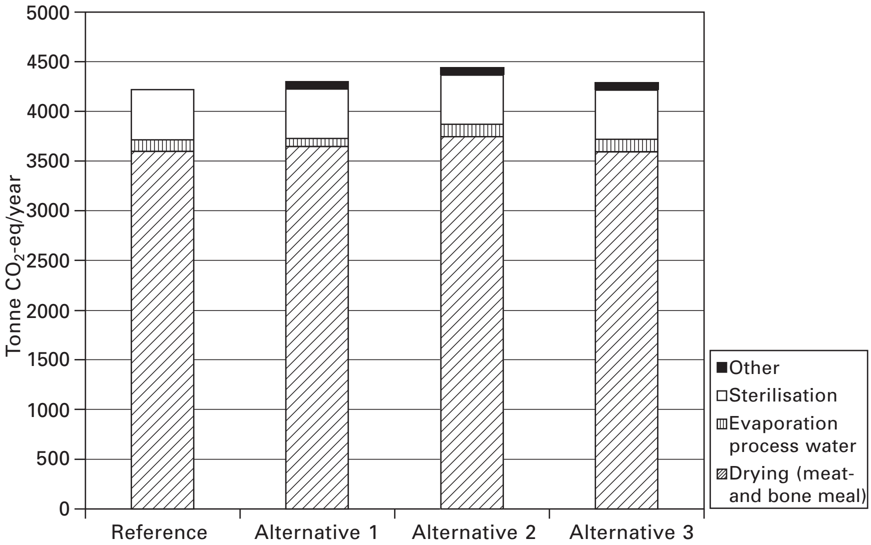

Fig. 9.2 Emissions of GHGs for the reference and the three alternatives. Total production per one year was the functional unit. Note that the emissions for UF and NF were negligible.

When the environmental impact was related to the economic performance instead, the picture shifted. Since the product mix from the alternative systems yielded higher prices, the environmental impacts for the alternatives became lower. In Fig. 9.3, the results for emissions of greenhouse gases (GHGs) per Swedish kronor (SEK) is presented. It is obvious that an increased intensity of processing, leading to more valuable products, decreased the environmental impacts in total.

The example shows the importance of identifying an appropriate set of functional units in order to understand the system, and also realise the differences in conclusions. This facilitates better-grounded discussions and decisions.

9.3 Methodological approach: proposal of a framework

LCA was initially developed to assess the environmental impact of available products or services, implying that the system under study was possible to investigate in detail (e.g. Baumann and Tillman, 2004). However, for novel processes and products that is not the case; there is no system from which to collect data. Moreover, in order to evaluate the environmental impact of new processes and products, a stringent methodology must be applied since there are so many choices to be made and so much uncertainty.

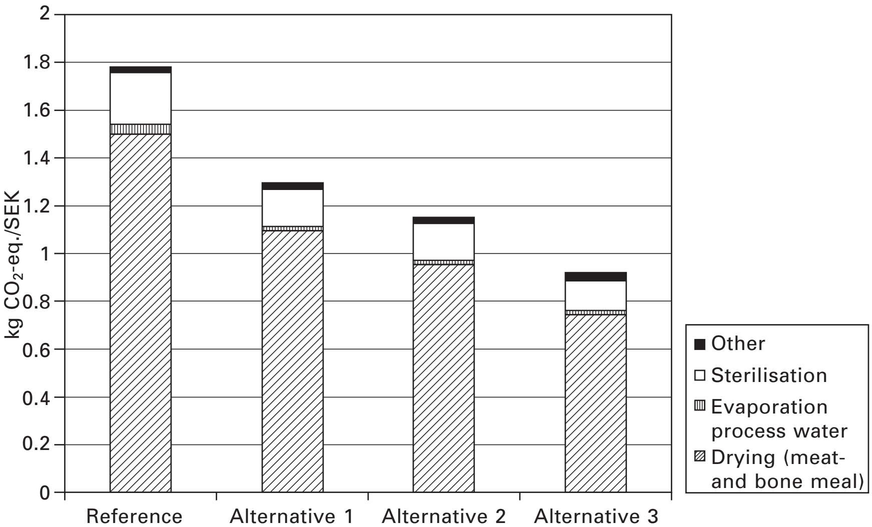

Fig. 9.3 Emissions of GHGs per SEK and year for the reference and the three alternative systems. Note that the emissions for UF and NF were negligible.

At SIK we have been working with a large number of projects where LCA has been applied to evaluate the environmental consequences of technological improvements in the food industry. The case studies presented above are two examples of that. Based on our experiences and insights, we have developed a proposed framework for such assessments, and this framework is presented below. (It is presented in detail in Hospido *et al*., 2010.) There are a number of specific aspects to consider when evaluating the environmental consequences of new processes and products; in our opinion the most important are:

- What type of LCA to apply
- What functional unit is the appropriate to use.
- How to deal with uncertainties in future systems
- How to define systems boundaries
- How to make data inventories.

Considerations and recommendations for each of these points are described below.

9.3.1 Type of LCA to apply

There are several choices to make on this issue and probably the most important are attributional versus consequential, and retrospective versus prospective (time perspective). For a more detailed description of the methodology please refer to Chapters 4 and 5 in this book. At first hand it might seem obvious to choose the consequential prospective approach since we are dealing with changes in processes or products. However, we argue that a prospective attributional approach is the most appropriate since we are generally interested in how the new process or product would perform in a future steady state rather then the dynamic impact when the new technology is applied.

9.3.2 Functional unit

The question of how to define the functional unit depends on the new process or product. If it is a completely new product, a 'common stand-alone' LCA can be performed and evaluated as such. However, if the novel product can be said to compete with existing similar products, or a new technology is introduced to produce similar products, the situation is slightly different. In such cases there is a need for comparisons and then the definition of the functional unit becomes critical. The challenge is to cover improved quality of the product. Quality attributes can be divided into categories – obligatory, positioning and market-irrelevant (Weidema, 2003). The first two need to be included in the functional unit, and if the positioning quality involves specific extra functions it can be considered as a co-product. A simple example is that vitamin C fortified juice provides the same function as ordinary juice plus vitamin C pills. In such cases a systems expansion is preferred to include different quality. Another option is to use economic values to cover differences in function: many food products are characterised more by the quality perception being a complex mix of consumer's expectations, etc, and hence the price paid is likely to be a more adequate approximation of differences in quality.

9.3.3 Uncertainties in future systems

When assessing new processes and products it is obvious that surrounding systems might change by the time of implementation. If the implementation probably is close in time, it is justifiable to use data on present systems, as today's energy mix, raw material production or waste management system. If the studied system can be said to be further in the future, possible changes in surrounding systems must be taken into account. This can be done by using information from published scenario work on, for example, energy systems and demography, and using this as a basis for data inventory on these systems.

9.3.4 Definition of systems boundaries

As for all LCA studies, relevant systems boundaries are critical. The system studied must be large enough to cover all relevant activities and at the same time not include parts that do not affect the results. In comparative studies, where the new product or process is being compared to the present one, it is often possible to leave certain parts of the chain out since they are identical for both systems. By doing this, the amount of work needed is reduced and still the difference between the systems can be quantified, thus answering the question whether it is environmentally beneficial or not to introduce the new process or product.

9.3.5 Data inventories

Generally it is recommended to use specific data on what is called the foreground system, i.e. the system in which the product under study is actually managed. For data on the background system, i.e. systems supporting the foreground system as energy supply, packaging and production, database and average sector data can be used.

A main characteristic of new systems is that no real production data exists; this is basically the definition of a new system. At the same time, data for the present production system exists and the system has often been fine-tuned and optimised for a certain time, so it is efficient. Data for novel systems can be obtained either from lab-scale experiments, or computer modelling, or a combination of the two. Lab-scale results may often deviate from actual data on the process when up-scaled to industrial production, as shown by Muñoz *et al.* (2006) (where the order of rank differed when using lab-scale data and industrial scale data on the same processes).

Our recommendation is to use a combination of lab-scale and computer modelling results and apply sensitivity analyses based on expert judgements on up-scaling phenomena.

9.4 Discussion

The need for a holistic perspective when evaluating the environmental impact of new processes and products is obvious. Looking at too narrow a system leads to large risks of sub-optimisation. By this we mean that solutions that score well in a narrow system might lead to increased environmental impact when looking at the entire food chain. The dairy case study exemplifies this in a good way; the improvements possible for the dairy company actually took place at the dairy farms, but since their production depended on how the dairy was run, they could be said to be within the domain of influence for the dairy. If only the direct impacts from the dairy industry would have been included, the conclusions would have been different. Another example is if the food producer uses a new technology that increases the energy use for the process (as in the case study on membranes) but this new process generates larger volumes or improved quality.

However, the situation is rarely simple; a combination of changed environmental impact, resource use, and production volume, makes the comparison difficult. By choosing system boundaries and input data arbitrarily, the results from the study can be difficult to interpret and this also makes comparison between products and processes problematic, which can lead to lost credibility among stakeholders of all kinds; the whole concept can be questioned. This calls for a well based methodology, where choices are made in a systematic and transparent way. It is not possible to state what choices should be made since the systems studied differ too much, as well as the questions posed. But a stringent framework can be used that increases the usefulness of using LCA in these cases. The membrane case-study gives an example of where two different functional units have been used to manage the complexity involved in the proposed new technology. This leads to slightly contradictory results, but by having these results, the conclusions made can be motivated in a logical way from the definition of the functional unit.

Data accessibility varies a lot, and when performing a study, large amounts of data are often needed. By combining LCA data from databases and literature for the background systems (not directly affected by changes analysed) and using primary data only for the foreground system (directly affected by changes analysed), the amount of work needed can be drastically reduced. Moreover, data on new processes are often not available at the industrial level; at best, pilot- or lab-scale data is at hand. This can be managed by a combination of modelling, literature data and expert judgements. Examples of this are given by Johansson *et al.* (2008), Östergren *et al.* (2007, 2008), and Berlin (2007). All this means that results should be interpreted with care and too strong conclusions cannot be made on small differences in results. This is a general point; the results are not detailed but give a good indication of the order of magnitude of differences between systems. One way of making the results more resilient is to apply sensitivity analyses of critical assumptions made, as for example energy mix and production efficiency. By doing this, possible uncertainties and variation can be quantified and managed in the discussion and decision-making.

9.5 Conclusions

- A systems perspective is crucial when assessing the environmental impact of new processes and products, and this needs a systematic tool and procedure. For this, LCA is a suitable tool.
- A systematic and transparent approach on how to apply LCA is needed.
- Results should be interpreted with care; data quality varies and methodological choices affect the results.
- Specific attention should be paid to the functional unit, time frame and data quality/accessibility.
- Transparency is extremely important for the credibility of a study and the study should be reported as thoroughly as possible.

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10

Addressing land use and ecotoxicological impacts in Life Cycle Assessments of food production technologies

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Abstract: Effects of land use and ecotoxicity are not commonly addressed in Life Cycle Assessment on agricultural food production, due to the expected high level of uncertainties in the impact assessment and a lack of available inventory data. This chapter provides an overview of the cause–effect pathways related to the release of toxic chemicals and physical land use practices caused by food production practices. It also discusses the background and application of several Life Cycle Impact Assessment methods that produce so-called Characterization Factors to quantify the environmental effects of the agricultural activity occurring along the cause–effect pathways. Particular attention is paid to advances in the data and modelling of ecotoxicological and land use impacts that resulted in the development of a consensus model to calculate characterization factors for aquatic ecotoxicity and several models to calculate characterization factors for land use. Finally, for both ecotoxicity and land use modelling, a number of uncertainties are discussed and several requirements for improvement are proposed.

Key words: land use, ecotoxicity, Characterization Factors, Life Cycle Impact Assessment, midpoint, endpoint, cause-effect pathway, physical land use changes, pesticides.

10.1 Introduction

Maintaining ecosystems is a great challenge for the agricultural sector. Compared to other economic activities, the agricultural sector has the drive to change the whole ecosystem it uses in order to optimize productivity or yield. This is achieved by four different routes: the application of toxic chemicals (herbicides and pesticides) to eliminate unwanted species, the application of fertilizers to change the nutrient level and acidity in the soil, the control of soil humidity, and physical activities to change the land. These activities have a multitude of wanted and unwanted consequences. Some of the unwanted consequences can be modelled, while others are very difficult to model and can only be monitored by using observational data. In this chapter we analyze the consequences of toxic chemicals to the ecosystem and the influences of physical changes, usually referred to as land use. The first effect is addressed with models while the second is addressed with observational data.

As the world population continues to grow from 6 billion in the year 2000 to 8.1–9.6 billion by 2050, ecosystem change is expected to prolong to meet the demands of food production (Sarukhán *et al.*, 2005). One way to meet these demands is by land transformation through deforestation and loss of grassland. Of the total terrestrial surface, 24% is taken by grassland and cropland (Sarukhán *et al.*, 2005), and by 2030 it is expected that areas will rise by a further 16% (OECD, 2008). However, the deforestation rate is slowing down by restoration and replanting initiatives, although still a net loss of 7.3 million hectares per year takes place (FAO, 2006). Another way to meet productivity demands is by increasing the yield through intensive use of pesticides. However, both land transformation and use of pesticides tend to reduce biodiversity. Therefore, tradeoffs between land use and ecotoxicity should be considered when analysing the environmental impacts of various food production technologies over their entire life cycle (Bengtsson *et al.*, 2005; Mansvelt *et al.*, 1995; De Boer, 2003). This can be done with help of Life Cycle Assessment (LCA).

Land use and ecotoxicity are important impact categories within the agricultural production step of LCAs on food products (De Boer, 2003; Mattsson *et al.*, 1998b), even more when intensive farming is compared to less intense farming (Mattsson, 1999; Williams *et al.*, 2006). In the Life Cycle Impact Assessment (LCIA) of agricultural food production it is important to include both the acre and quality of the land used, together with the pesticide application. Next to pesticides, metal pollution from energy use in food production and processing can also make a large contribution to ecotoxicity (Berlin, 2002; Milà i Canals *et al.*, 2006; Mouron *et al.*, 2006; Zabaniotou and Kassidi, 2003). When the effects of land use or ecotoxicity are considered, in several case studies simply the amounts of pesticides, or land occupied or transformed are taken as an indicator (Blonk, 2006; Williams *et al.*, 2008; Mattsson *et al.*, 1998a; Basset-Mens and van der Werf, 2005; Berlin, 2002; Cederberg and Mattsson, 2000; Stern *et al.*, 2005; Weidema *et al.*, 2008). Some specific LCA studies attempt to implement land use and ecotoxicity in a more complete way, however, each by applying a different methodology (Mattsson *et al.*, 2000; Brentrup *et al.*, 2004a; Blonk, 2006; Cordella *et al.*, 2008; Milà i Canals *et al.*, 2006; Mouron *et al.*, 2006). This indicates a high need for consensus and guidelines regarding the implementation of land use and ecotoxicity into LCAs for food production and processing.

This chapter provides an overview of methods to address land use and ecotoxicity in LCIAs of food production and processing. Hereby, guidelines on the application of methods in case studies are presented. Section 10.2 focuses on effects of land use and Section 10.3 on effects of ecotoxicity. Both sections outline the potential related impacts, how these are identified in LCAs and present available tools together with a number of case studies on food and food related products. For ecotoxicity, we focus on environmental impacts outside agricultural fields, as the direct effects of application of pesticides on agricultural soils are already included in land use indicators. Fields of further research are identified and summarized in Sections 10.5 and 10.6 our conclusions and some sources of further information and advice are given.

10.2 Life cycle impact methods for land occupation and transformation

10.2.1 Cause effect pathway

Land use refers to the occupation or transformation of a certain area for human activities, such as storing materials or waste and production of agricultural products or resources (Muller-Wenk, 1998). The physical consequences are multiple, such as fragmentation, direct loss in biodiversity, altered vegetation, soil degradation, changes in water regimes (discussed in Chapter 2) and differences in reflection capability of the earth surface (albedo). Figure 10.1 gives an initial insight into the complexity of the different consequences of land occupation and transformation, and how they are interrelated. Important effects of agricultural land occupation and transformation are reduced soil quality (Oldeman, 2000), and direct species loss (Mace *et al.*, 2005; Sarukhán *et al.*, 2005). Soil degradation results from compaction, erosion, salinization, and depletion of minerals, nutrients, and organic matter. The removal of natural vegetation and deforestation are the main causes of soil degradation (43%), resulting in the main erosion routes being water erosion (55%) and wind erosion (28%) (Oldeman, 2000). After long-term land clearance or extensive land occupation, it alters vegetation, water regulation, and agricultural productivity (Mantel and Engelen, 1997; Lal, 2001). The Food and Agriculture Organization (FAO) estimated a global loss of 5 to 7 million ha productive land every year due to soil degradation. In 2008, the land surface degraded was 24% (Bai *et al.*, 2008), 3100 million hectares, reflecting an area 3.2 times larger than Europe. Regarding terrestrial biodiversity, land transformation due to agricultural activities is currently considered the main driver for species loss (Sarukhán *et al.*, 2005; Clay, 2004; Mace *et al.*, 2005; Schleuning *et al.*, 2009). Furthermore, not all species

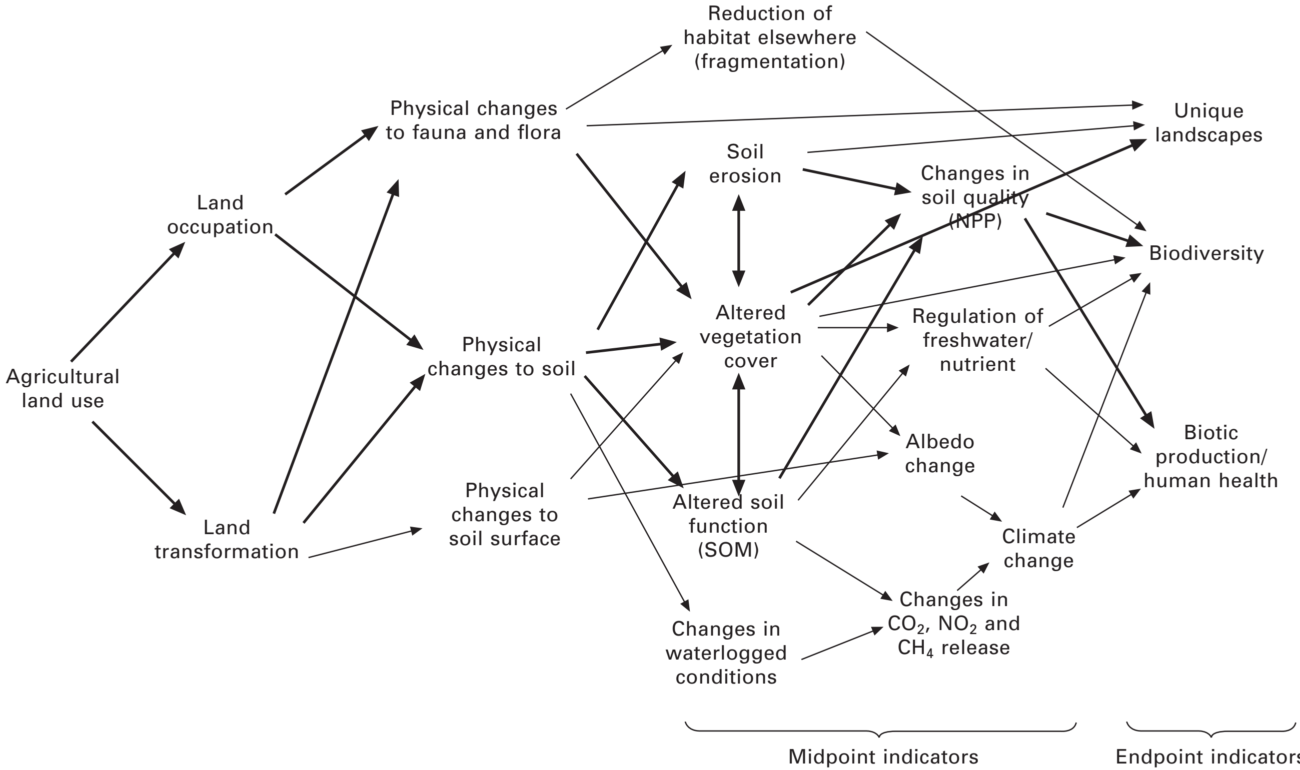

Fig. 10.1 Cause–effect pathway for land use (adapted from Guinée *et al*., 2006; Hauschild *et al*., 2008a). Midpoint indicators refer to indicators in the middle of the cause–effect pathway, while endpoint indicators refer to the actual damage resulting at the end of the cause–effect pathway. The thick arrows present the main pathways related to agricultural land use.

have the same value for the ecosystem. Some species have key functions (keystone species) and, by their disappearance or extinction, the loss of the function of the ecosystem may be disproportionally higher compared to the disappearance of other species (Benedek *et al.*, 2007).

10.2.2 Framework

Within the framework of LCIA, the effects of land use can be divided in three activities: transformation, occupation, and restoration of land (Fig. 10.2). All three activities can be combined, whereby occupation follows transformation and results in restoration. As a consequence of each activity, nature is modified in a way that is defined as damaging. The level of damage is measured against a chosen reference or baseline land that refers to the non-use of the area; for example the natural state of an area without human interactions (Milà i Canals *et al.*, 2007a).

The LCIA score for land occupation (IS_{occ}) can be expressed as:

$$IS_{occ} = CF_{occ,i} \cdot A_i \cdot t_i \qquad [10.1]$$

where $CF_{occ,i}$ is the Characterization Factor (CF) for land occupation with land use type i; and $A_i \cdot t_i$ the area occupied (m^2) multiplied with the time of occupation by land use type i (yr).

The LCIA score for land transformation or restoration ($IS_{trans/rest}$) is expressed as:

$$IS_{trans/rest} = CF_{trans/rest,i} \cdot A_i \qquad [10.2]$$

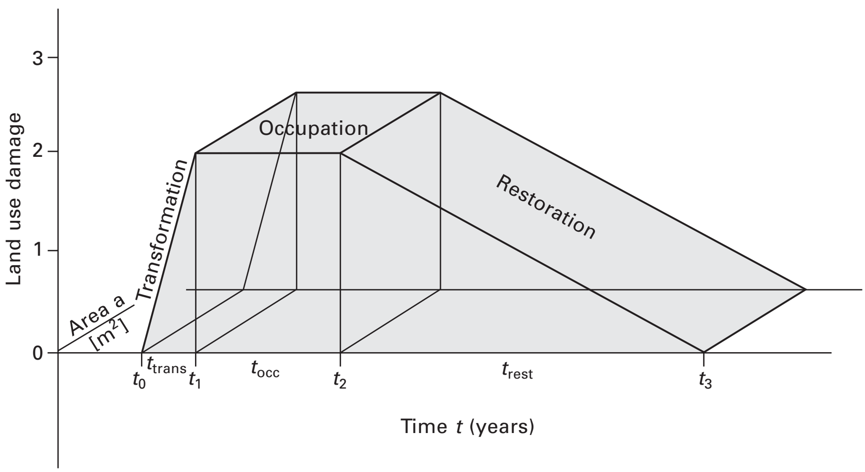

Fig. 10.2 Land use activity subdivided into transformation, occupation and restoration. The grey volume represents the total damage score of a land use activity and is the integral over area a and duration *t* of each land use activity. At which level the damage starts and ends is determined by the chosen baseline (Adapted from Koellner and Scholz, 2007).

where $CF_{trans/rest,i}$ is the CF for land transformation into land use type i or restoration to natural land; and A_i the area transformed or restored (m^2). The time needed for transformation or restoration is included in the CF and is usually based on an estimate of the transformation and the restoration time. The CF for occupying or transforming land surfaces quantify the physical consequences of the human activity, using one or more quality indicators chosen in the middle or at the end of the cause-effect pathway (see Fig. 10.1). Milà i Canals *et al*. (2007a) present a list of possible midpoint and endpoint quality indicators that cover direct and indirect effects of land occupation and transformation.

In the midpoint approach, quality indicators, such as soil pH, soil organic matter, and net primary production, are applied. Midpoint indicators are well-suited for the comparison of different land use activities, but do not provide the possibility to compare the environmental impact of land use with other terrestrial ecosystem related impacts, such as acidification or eutrophication. The midpoint CFs for land occupation ($CF_{occ,i}(midpoint)$) and land transformation ($CF_{trans/rest,i}(midpoint)$) can be expressed as:

$$CF_{occ,i}(midpoint) = Q_b - Q_i \qquad [10.3]$$

and

$$CF_{trans/rest,i}(midpoint) = (Q_o - Q_i) \cdot t_{trans/rest}/s_{i-o} \qquad [10.4]$$

with Q_i, Q_o and Q_b the midpoint quality indicator for land use type i, the original land use type o and the baseline land use type b, $t_{rest/trans}$ the time needed for transformation or restoration and s_{i-o} the slope factor to reflect that restoration appears gradually. Note that the original and baseline land use type can be the same and that not every midpoint method considers the impact from transformation or restoration.

The endpoint approach refers to quality indicators for species richness (biodiversity), loss of unique landscapes, and reduction in biotic production. The change in species richness, or biodiversity, is commonly used as endpoint indicator within LCA and allows us to integrate or compare the direct effects of land use with other environmental impacts (Muller-Wenk, 1998). The endpoint CF for land occupation ($CF_{occ,i}(endpoint)$) is expressed as the potentially disappeared fraction of species (PDF):

$$CF_{occ,i}\,(endpoint) = PDF_i = 1 - \frac{S_i}{S_b} \qquad [10.5]$$

with S_i and S_b the number of species at land use type i and the number of species at baseline land use type b. The endpoint CF for land transformation or restoration ($CF_{trans/rest,i}$ $(endpoint)$) is expressed as:

$$CF_{trans/rest,i}\,(endpoint) = \left(1 - \frac{S_i}{S_o}\right) \cdot t_{trans/rest}/s_{i-o} \qquad [10.6]$$

with S_o the number of species at original land use type o. The number of species depends on the size of the area, also defined as Species Area Relationship (SAR). The area size can be considered in the PDF calculations (Koellner and Scholz, 2008; Goedkoop *et al.*, 2008). The use of biodiversity as endpoint indicator covers only part of the cause–effect pathway of land use (Hauschild *et al.*, 2008a). For example, the loss of unique landscapes or the effects on human health through albedo climate regulation are not covered by this indicator (see Fig. 10.1).

10.2.3 Methods

Each method has its own quality indicators and thereby covers a certain part of the cause-effect pathway (Fig. 10.1). To be able to choose the preferred indicator we need to know what we want to preserve. Method developers can choose for indicators that reflect the naturalness of the land or for indicators that reflect the service of the land. In Fig. 10.3 we position the different methods to the extent that they focus on impacts on 'naturalness' or 'system service'. For several methods we use ovals to position them on the axes because they clearly focus in one direction, but it is vague to what extent they consider the other vision. Three main clusters can be identified: (i) methods that apply indicators that focus on the naturalness of the system, such as PDF, (ii) methods that focus on system services,

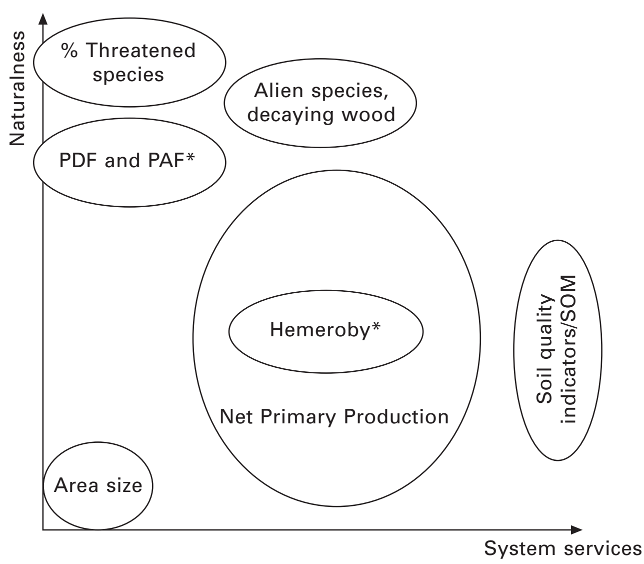

Fig. 10.3 Different models scaled on level of naturalness and system service. *Measures the degree of human interventions by indicators such as the share of species and physical and chemical soil features.

and (iii) the indicator 'area size' that does not have a focus on naturalness or system service.

Table 10.1 provides an overview of available land use methods. Case studies are introduced that apply the various methods, and their main conclusions regarding land use are presented. The simplest way of considering land use in the life cycle of food production is by simply adding up all land occupation and transformation area size (Heijungs *et al.*, 1992). This way of implementing land use is simple and robust, but lacks environmental relevance. Other methods give a more complete overview of the damage from land use by using multiple midpoint indicators (Bos and Wittstock, 2008; Baitz *et al.*, 1998; Oberholzer *et al.*, 2006; Mattsson *et al.*, 2000; Michelsen, 2007). The use of several quality indicators next to each other requires substantial data input that is not always present. Furthermore, it creates multiple results, which cannot be easily aggregated. Milà i Canals (2007b) argues that the quality indicator Soil Organic Matter (SOM) can be used as a single midpoint indicator for agricultural land use, and covers different impacts such as soil fertility, climate regulation and water regulation. However, several land use impacts are excluded by the SOM indicator, such as erosion, compaction and salination. Endpoint methods use the loss of species diversity, expressed as Potentially Disappeared Fraction of species (PDF), as an indicator to assess the physical effects of land use (Muller-Wenk, 1998; Schmidt, 2008; Lindeijer, 2000; Weidema and Lindeijer, 2001; Koellner, 2000; Goedkoop *et al.*, 2008; Koellner and Scholz, 2007, 2008; Itsubo, 2008).

Recently, eight different land use characterization methods with readily available CFs for land occupation and transformation were evaluated for the European Commission (Hauschild *et al.*, 2008a). The method of Milà i Canals (2007b), which uses Soil Organic Matter as a quality indicator, is identified as the best applicable midpoint approach, while ReCiPe2008 (Goedkoop *et al.*, 2008) is preferred as an endpoint approach. However, the most recent work of Koellner and Scholz (2008) contains new elements and data, such as the use of target species, that bring the method at least to the same level as ReCiPe2008.

10.2.4 Uncertainty

Several uncertainties arise within the application of land use methods. First, the applied quality indicators cover only part of the land use cause–effect pathway (Fig. 10.1). Most midpoint methods refer only to soil quality, while endpoint methods take into account only direct species loss. Several effects, such as fragmentation, the loss of unique landscapes, or albedo climate regulation, are not covered by these indicators.

Second, even though endpoint indicators allows us to aggregate several ecosystem effects, such as land use, ecotoxicity, and eutrophication, a certain risk in double counting environmental impacts occurs (Hauschild *et al.*, 2008a). The loss of species does not only reflect the consequences of land

Table 10.1 Land use models applied in LCIA, together with food production and processing case studies performed with the models. When no case studies on food products were available, others are introduced. *M = Midpoint, E = Endpoint, O = Occupation, TR = Transformation and Restoration

Model	Characteristics	M/E*	O/TR*	Implemented in	Region	Case studies	Main outcomes
Area size	Adds up land occupation and transformation separately (m^2) (Goedkoop *et al.*, 2008; Heijungs *et al.*, 1992)	M	O/TR	CML 92/2002 (Guinée *et al.*, 2002)	–	Pig production (Basset-Mens and van der Werf, 2005)	Organic pig production has highest land use score. Crop and feed production stage are dominant contributors to land use (>80%).
				ReCiPe 2008 (Goedkoop *et al.*, 2008)		Bread (Blonk, 2006)	Most land use takes place in the wheat production.
Soil quality indicators	Combination of five to seven indicators: Emission filtering; Physical and Chemical Filtration; Ecosystem stability and biodiversity; Erosion stability; Filter, buffer function for water; Groundwater availability/protection; Net Primary production; Water permeability; soil organic matter; soil structure. Soil pH, accumulation of heavy metals, high soil content of phosphorus and potassium (Baitz *et al.*, 1998; Bos and Wittstock 2008; Mattsson *et al.*, 1998)	M	O/TR		Not specified	Vegetable oil crops (Mattsson *et al.*, 2000)	The indicators erosion, SOM, soil structure, soil pH, P and K status and biodiversity provided a good picture of land use damage. The different indicators make it difficult to draw conclusions. The loss of SOM was the most serious threat for soybean. Soil compaction is a problem for rapeseed. Soybean and palm oil production give highest thread to biodiversity.
Hemeroby	Degree of human interventions (Natural Degradation potential,	M/E	O		Not specified	Wheat (Brentrup *et al.*, 2004b)	Level of NDP decreases in relation to higher yield from

Table 10.1 Continued

Model	Characteristics	M/E*	O/TR*	Implemented in	Region	Case studies	Main outcomes
	NDP) measured by the share of species, physical and chemical soil features and land use types (Brentrup *et al*., 2004a; Brentrup *et al*., 2004b)						fertilizer use. Applying more than 144 kg N/ha didn't affect the yield and thus NDP. Aggregation with other impacts was difficult.
General quality indicators and forest specific indicators	Biodiversity is measured by species richness, ecosystem scarcity and ecosystem vulnerability. Amount of decaying wood, areas set aside and introduction of alien tree species scaled from 0 to 3. Rotation time = restoration time (Michelsen, 2007)	M	O/TR		Not specified	Logging of spruce (Michelsen, 2007)	Quality indicators determined for taiga forest, and coastal forest. Logging coastal forest gives 40% more impact than logging taiga forest.
Soil organic matter (SOM)	(Milà i Canals *et al*., 2007)	M	O/TR		Not specified	New method, no case studies yet	
Biodiversity and Net Primary Production (NPP)	Transformation is not included yet, but can easily be incorporated in the existing method (Lindeijer, 2000)	M/E	O		Global	Construction materials (Lindeijer *et al*., 2002)	Case study on brick, stone and wood, to test the method. Wood can score favourable or unfavourable, depending on the chosen baseline.
	Biodiversity measured by species richness, ecosystem scarcity and ecosystem vulnerability. Exchange of	M/E	O/TR		Not specified	No case studies	

	chemical substances added as extra factor. Transformation time is included in the occupation factor, together with a slope factor $s_{i\text{-}o}$ (Weidema and Lindeijer, 2001)						
	Biodiversity measured as species loss due to local and regional effects, based on Japanese red species list. This is combined with net primary production data from the Chikugo model (Itsubo, 2008).	M/E/	O /TR	LIME (Itsubo and Inaba, 2003)	Japan	New method, no case studies yet	
Biodiversity	% of threatened plant species in a region. Land restoration is considered by adding a fixed restoration time of 30 years to the original occupation time (Muller-Wenk, 1998).	E	O/TR	Eco-indicator 99 (Goedkoop and Spriensma, 1999), IMPACT 2002+ (based on Eco-indicator 99) (Jolliet *et al*., 2003)	Central Europe	Beer production (Cordella *et al*., 2008)	EI 99 was used. Barley cultivation was main contributor for land use, although the uncertainties for this impact were high.
						Meat products (Blonk *et al*., 2007)	EI 99 was used. Farming and feed production stages are dominant contributors to land use. However, the effects of cheap grazing in natural areas should be considered and animal welfare is closely related to the area occupied. Both could not be analysed in a quantitative way and thus worst case results are presented.
	Potential disappeared fraction of vascular plant species					Meat and dairy products (Weidema	IMPACT2002 was used. The impact of land occupation is

Table 10.1 Continued

Model	Characteristics	M/E*	O/TR*	Implemented in	Region	Case studies	Main outcomes
	combined with fraction of land available (Koellner, 2000). Schmidt used this approach to develop characterization factors for Malaysian and Indonesian forest systems (Schmidt, 2008)					*et al.*, 2008)	dominant in the consumption of meat and dairy and contributes most for dairy and beef.
	Potential disappeared fraction of all vascular plant species, threatened plant species, mosses and molluscs. Considering species area relationship. Restoration/ transformation time is considered with the inclusion of a slope factor $s_{i\text{-}o}$ (Koellner and Scholz, 2007, 2008)	E	O	Ecological Scarcity 2009 (Frischknecht *et al.*, 2008)	Central Europe	Biofuels (Scharlemann and Laurance, 2008)	The benefits from ethanol production from sugarcane diminishes when the total environmental impact is considered, including biodiversity loss and soil erosion. For 50% of 26 biofuel crops the total environmental impact is higher than fossil fuels.
		E	O/TR	ReCiPe 2008 (Goedkoop *et al.*, 2008)	Central Europe + Great-Britain	New method, no case studies yet	

use but also other impacts caused by farming, such as the effects of pesticide or fertilizer use (Goedkoop and Spriensma, 1999).

Third, the calculation of land use CFs requires the choice of a baseline. The historical land use state or potential land use state after restoration can be chosen but does not consider land evolution (Milà i Canals *et al.*, 2007a). The average species richness of the region (Koellner, 2000), the maximum species richness (Weidema and Lindeijer, 2001), or another alternative system can also be considered as baseline (Milà i Canals *et al.*, 2007a).

Fourth, often occupation occurs in an area that is already in use. Therefore, land transformation and restoration are mostly excluded in LCA of products. However, including these two land use activities within the impact category 'land use' is mostly relevant when new conversions of natural land take place. For food production, this is the case within continents where agriculture still expands, such as soy bean production in South America.

Fifth, for each region the number of species differs. This makes the species-richness indicator region dependent, which significantly influences the results. While the work of Koellner and others (Koellner and Scholz, 2007, 2008; Koellner, 2000; Muller-Wenk, 1998) is developed for Central Europe, ReCiPe2008 (Goedkoop *et al.*, 2008) uses a combination of Swiss and British data, LIME is based on Japanese species (Itsubo, 2008), and Schmidt (2008) introduced Malaysian land use types. To make the LCIA of land use globally applicable, more region-specific CFs need to be developed.

Sixth, the species area relationship applied to calculate the endpoint CFs makes the calculations area dependent. Schmidt (2008) calculates CF for an area size of $100\,m^2$, Koellner and Scholz (2008) apply an area size of $1m^2$, and Goedkoop *et al.* (2008) an area of $10\,000\,m^2$. Which area size to use is not yet standardized.

Finally, by using the overall species loss as indicator, no differentiation between species that contribute more to the ecosystem than others (keystone species) is made. To differentiate between vulnerable species, the loss of target species can be considered as the species richness indicator. Both the LIME method (Itsubo, 2008) and the work of Koellner and Scholz (2008) give the possibility to follow this approach.

10.3 Life cycle impact methods for ecotoxicity

10.3.1 Cause–effect pathway

Ecotoxicity refers to the potential for biological, chemical, or physical stressors to affect ecosystems. The term was first outlined by Truhaut (1977), who defined it as 'the branch of toxicology concerned with the study of toxic effects, caused by natural or synthetic pollutants, to ecosystems, animals (including human beings), plants, and microbial communities'. Research within ecotoxicology is being used to set environmental regulations. Legal environmental quality criteria are set based on generic risk limits for toxic compounds for water, sediment and soil, derived in ecological and human risk assessments (Sijm *et al.*, 2002). Although the pollution peaks in surface waters in the 1970s have now largely subsided in the western world due to strict toxic-chemical regulations, the problem of pollutants is still with us today. Environmental quality standards are not being met at many sites (Posthuma *et al.*, 2008). In addition to well-identified spots, a diffuse pollution, defined as the chronic presence of mixtures of toxic chemicals over large surface areas, in concentrations exceeding generic and protective environmental quality standards for one or more compounds, covers vast areas of land, vast volumes of sediment, and many surface water bodies. The apparent magnitude of ecotoxicological effects (in terms of increased, diffuse contamination levels) creates major problems for policymakers, due to (i) concerns in the general public, (ii) an array of regulations that suggest a need to manage ecological stressor impacts and pollution risks, for example in relation to protected species, and (iii) uncertainty over the magnitude of local impacts (Kapo *et al.*, 2008; Posthuma *et al.*, 2008).

In intensive agricultural practice, pesticides can cause substantial impact on ecosystems. Previous research, focusing on mixture toxicity assessment, showed that there is a large variation among pesticides regarding their impact on freshwater ecosystems (De Zwart, 2005; Henning-de Jong *et al.*, 2008). It is therefore important for agricultural production processes to find and apply alternatives to some of the currently used pesticides.

Figure 10.4 shows the cause–effect pathway for ecotoxicity impacts. Emissions to air, vegetation, water, or soil will affect a variety of species. Starting from cold-blooded organisms, chemicals can be accumulated along the food chain. The whole ecosystem is affected by toxic compounds, which, in terms of biodiversity, can be expressed as the potentially affected fraction (PAF) or potentially disappeared fraction (PDF) of species. The ecotoxic effects that chemicals cause on the environment can be assessed up to the PAF or PDF, which is called the endpoint. All points earlier on the cause–effect pathway are referred to as midpoints.

10.3.2 Framework

The LCIA score for ecotoxicity of a chemical x in compartment j ($IS_{x,j}$) equals the emission of a chemical x to compartment i ($M_{x,i}$) multiplied by the CF:

$$IS_{x,j} = M_{x,i} \cdot CF_{x,i,j} \qquad [10.7]$$

where $CF_{x,i,j}$ is the ecotoxicological CF of chemical x emitted to compartment i and transported to compartment j (e.g. in $m^3.yr.kg^{-1}$). To estimate pesticide emissions from field application, models can be used, such as PestLCI (Birkved and Hauschild, 2006).

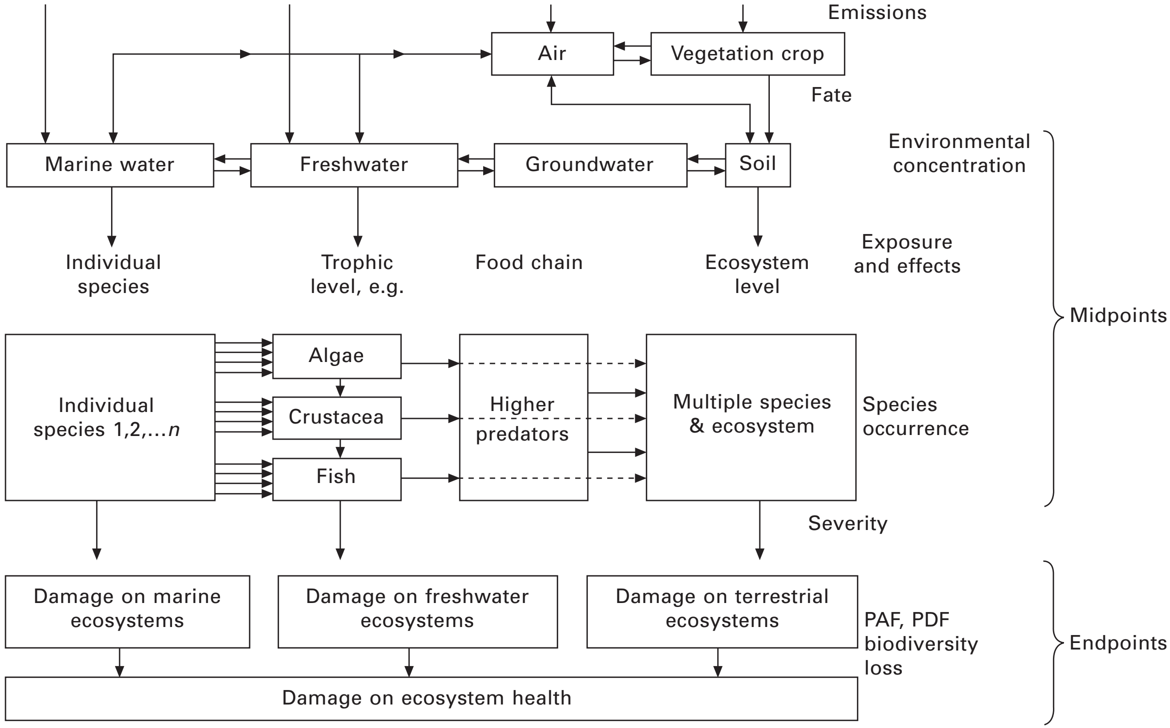

Fig. 10.4 Cause–effect pathway for ecotoxicity (adapted from Hauschild *et al.*, 2008a).

The CF accounts for the environmental persistence (fate), and ecotoxicity (effect) of a chemical:

$$CF_{x,i,j} = FF_{x,i,j} \cdot EF_{x,j} \qquad [10.8]$$

$FF_{x,i,j}$ represents the compartment-specific fate factor that accounts for the transport efficiency of substance x from compartment i to, and persistence in environment j (FF in yr), and $EF_{x,j}$ is the effect factor of chemical x in compartment j.

The environmental fate factor is defined as the change in the steady state concentration in an environmental compartment due to a change in emission (e.g. Huijbregts *et al.*, 2005):

$$FF_{x,i,j} = \frac{V \cdot dC_{x,j}}{dM_{x,i}} \qquad [10.9]$$

in which V is the volume of environment j (m^3), $dC_{x,j}$ is the change in the steady state dissolved concentration of substance x in environment j ($kg \cdot m^{-3}$), and $dM_{x,i}$ is the change in the emission of substance x to compartment i ($kg \cdot yr^{-1}$). Emission compartments commonly included in ecotoxic evaluations within LCIA are urban and rural air, freshwater, seawater, and agricultural and industrial soils. Environmental receptors generally identified are terrestrial, freshwater, and marine environments (Margni *et al.*, 2002; Rosenbaum *et al.*, 2008). FFs are generally calculated by means of 'evaluative' multimedia, multi-pathway fate and exposure models, such as CalTOX (McKone, 1993), IMPACT 2002 (Pennington *et al.*, 2005), and SimpleBox (Den Hollander *et al.*, 2004).

The effect factor is defined as the change in potentially affected fraction of species (PAF) due to a change in concentration in compartment j:

$$EF_x = \frac{\mathrm{d}PAF}{\mathrm{d}C_x} = \frac{\mathrm{d}PAF}{\mathrm{d}TU_x} \cdot \frac{\mathrm{d}TU_x}{\mathrm{d}C_x} = S_{PAF} \cdot \frac{1}{10^{\mu_x}} \qquad [10.10]$$

where EF_x represents the effect factor of substance x ($m^3 \cdot kg^{-1}$); and dTU is the change in toxic units. The PAF-value expresses stress on ecosystems due to the presence of a single chemical or a mixture of chemicals. A PAF reflects the fraction of all species that is expectedly exposed above a certain effect-related benchmark, such as the Effect Concentration for 50 percent of species (EC_{50}) (De Zwart and Posthuma, 2005). S_{PAF} is the slope factor of the potentially affected fraction of species.

Two main classes of methods are currently identified for the calculation of the slope factor S: (i) methods assuming linear concentration–response relationships, and (ii) methods accounting for the non-linearity in concentration–response relationships (Larsen and Hauschild, 2007; Pennington *et al.*, 2004; Van de Meent and Huijbregts, 2005). In the non-linear methods, S depends on the toxic mode of action of the chemical (Van de Meent and Huijbregts, 2005; Van Zelm *et al.*, 2007). Figure 10.5 shows the linear and non-linear approach to derive S_{PAF}.

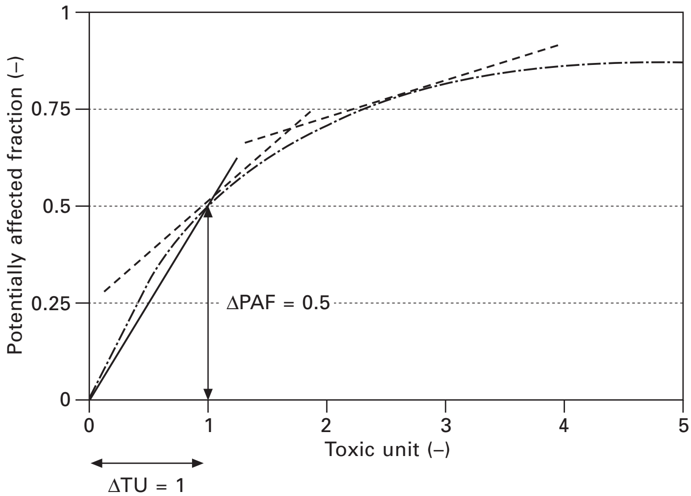

Fig. 10.5 The linear and non-linear approaches for deriving the slope factor for potentially affected species (S_{PAF}). The black line refers to the linear approach in which $\Delta PAF/\Delta TU$ is always 0.5, while the dotted lines refer to the non-linear concept (in which dPAF/dTU depends on the ambient TU of the chemical or mode of action under consideration).

The chemical-specific part of the effect factor equals $1/10^{\mu}$ and reflects the inherent toxicity of a chemical, defined as the inverse of the average toxicity of a chemical, which is the concentration of substance x, where 50 percent of the species is exposed above an acute or chronic toxic value ($kg \cdot m^{-3}$). μ_x is the average sensitivity of species to pesticide x ($g \cdot l^{-1}$), with sensitivity being expressed as an EC50 or another ecotoxicity test endpoint.

Midpoint indicators are referred to as ecotoxicity potentials and express the relative impacts of chemicals towards each other. The midpoint CFs can be used in comparison studies to understand which alternative(s) cause(s) most ecotoxicity. Different pesticide applications can, for example, be compared to see the environmentally best option to apply on a certain crop. Midpoints cannot, however, be used to compare different impact categories with each other. There is still a debate going on regarding the differentiation between midpoint and endpoint characterization for ecotoxicity and the best damage assessment to be applied (Larsen and Hauschild, 2007; Rosenbaum *et al.*, 2008). The PAF, based on EC50 data, may be regarded as the endpoint level. Posthuma and De Zwart (2006) showed for responses of fish species assemblages that the observed loss of species that can be ascribed to mixture toxicity closely matches the predicted risks based on EC50-data, at least in a relative sense (slope 1:1), and with a maximum observed fraction of lost species equal to the EC50-based ecotoxicity predictor variable. Due to these relationships, PAF as predictor parameter may have the diagnostic properties required to assess ecological conditions. Explicitly modelling further up to potentially disappeared fraction of species via a damage approach is possible as well (Larsen and Hauschild, 2007).

10.3.3 Methods

Ecotoxicity assessment models, namely BETR (MacLeod *et al.*, 2001), CalTox (McKone, 1993), EDIP (Tørsløv *et al.*, 2005), IMPACT2002 (Jolliet *et al.*, 2003), USES-LCA (Van Zelm *et al.*, 2009b), and Watson (Bachmann, 2006) all work with (part of) the framework mentioned previously. The Task Force on ecotoxicity and human toxicity impacts, established under the LCIA program of the United Nations Environment Program (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC), aimed at making recommendations for CFs for toxicity that incorporated broad scientific consensus. The method has to be simple enough to be used on a worldwide basis for a large number of substances. After a comprehensive comparison of the existing human and ecotoxicity characterization models mentioned above, the scientific consensus model USEtox was constructed. USEtox consists of a multi-media fate and exposure model and includes the linear method in the effect calculations (see Rosenbaum *et al.*, 2008). Figure 10.6 shows the compartment setup of USEtox. As USEtox results from a consensus building effort amongst related modellers, the underlying principles reflect common agreed recommendations from these experts. Moreover, the

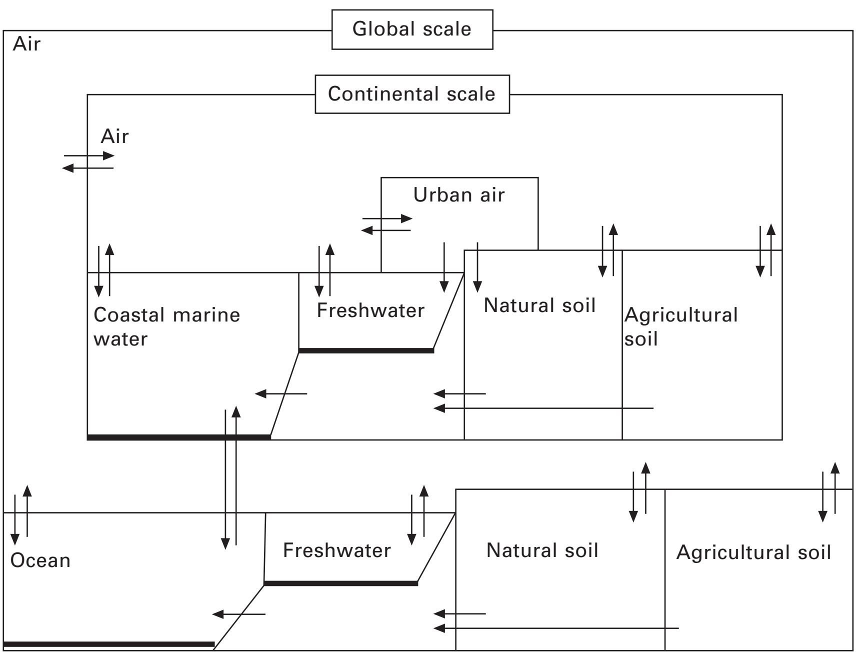

Fig. 10.6 Compartment setup of the USEtox consensus model (adapted from Rosenbaum *et al.*, 2008).

model addresses the freshwater part of the environment problem and includes the vital model elements in a scientifically up-to-date way. For example, for LCA comparative reasons average toxicity among species is taken as a basis and not the most sensitive species. Furthermore, chronic EC50 data are prioritized. USEtox can be considered an endpoint model as the factors express the potentially affected fraction of species integrated over time and volume per unit mass of a chemical emitted (PAF $m^3 \cdot day \cdot kg^{-1}$).

Table 10.2. provides an overview of ecotoxicity methods available, with their characteristics and in which methodology they are implemented. Furthermore, case studies are listed that apply the various methods, including their main conclusions regarding ecotoxicity.

10.3.4 Uncertainty

Previous LCA case studies show a relatively large uncertainty range for freshwater ecotoxicity, compared to other (nontoxic) impact categories (Geisler *et al*., 2005; Huijbregts *et al*., 2003). Geisler *et al*. (2005), in their study on plant protection products, state that before the freshwater ecotoxicity impact scores are used in decision support, measures to reduce uncertainty have to be taken. Uncertainty in ecotoxicological CFs is taken into account in various researches (Huijbregts *et al*., 2000; Payet, 2004; Van Zelm *et al*., 2009a, 2010). These researches show that the main uncertainty is related to the effect factor, and specifically to the availability of reliable species toxicity effect data and the choice of the slope factor. Concerning the latter point, it is considered debatable, whether the linear slope factor (S_{PAF}) of 0.5 is the best option to apply (Larsen and Hauschild, 2007). Van Zelm *et al*. (2009a) provided effect factors with non-linear slope factors for 397 pesticides with a focus on the toxic mode of action. From a conceptual point of view, the nonlinear slope factor can be preferred as it allows for addressing nonlinear concentration–response relationships. However, the nonlinear method is more complex than the linear method and has a high data demand. Therefore, more research in this area is still needed.

Midpoint indicators are calculated in all food case studies, except in a case study on beer by Cordella *et al*. (2008). There is a need to apply common agreed endpoint indicators that express the actual damage, such as the potentially affected fraction of species caused by the chemicals (Brentrup *et al*. 2004a). With the recent developments in endpoint models and ongoing research into the slope factor and damage indicators, application of endpoint CFs will become more common.

Exposure to higher predators due to bioaccumulation of chemicals along the food chain has not been addressed in LCA so far. As chemicals can accumulate in food chains, causing impacts on warm-blooded organisms that might be different from the impacts on cold-blooded organisms, there is uncertainty attached to the exclusion of bioaccumulation, and research in this area is needed as well.

Table 10.2 Ecotoxicity models applied in LCIA, together with food production and processing case studies performed with the models. *M = Midpoint, E = Endpoints

Model	Characteristics	M/E*	Region	Implemented in	Case studies	Main outcomes
CalTOX	Multi-media fate and exposure model. Linear effect method based on NOECs (McKone, 1993)	M	US	TRACI (Bare *et al*., 2002)		
EDIP	The fate assessment is simplified, i.e. no intermedia transfer processes are included. Linear effect method based on the geomean of trophic levels, applying chronic data. Spatially differentiated exposure factors for Europe. Marine compartment excluded. (Tørsløv *et al*., 2005)	M	World	EDIP 1997/2003 (Hauschild and Potting, 2005; Hauschild and Wenzel, 1998)	Apples (Milà i Canals *et al*., 2006)	Comparison of five orchards. Pesticides and metals in water included. Aquatic ecotoxicity dominated by emissions of heavy metals related to energy consumption
					Fish (Thrane, 2006)	Flat fish only. Marine ecotoxicity of biocides from anti-fouling agents. Fishing stage in life cycle dominant contributor to ecotoxicity.
IMPACT 2002	Multi-media fate and exposure model. Linear effect method based on EC50s. Uncertainty estimates and normalization factors included (Pennington *et al*., 2005)	M/E	Europe	IMPACT 2002+ (Jolliet *et al*., 2003)	Apples (Mouron *et al*., 2006)	Variability of impacts between fruit farms. Pesticides and heavy metals cause large impacts.
					Meat and dairy (Weidema *et al*., 2008)	Environmental impacts of consumption in EU-27. Copper emissions to soil from pig and dairy farming need to be reduced.
			Japan	Lime (Itsubo and Inaba, 2003)		

			Canada	LUCAS (Toffoletto *et al*., 2007)		
USES-LCA	Multi-media fate and exposure model. Linear effect method based on EC50s. Possibility to apply non-linear effect method for freshwater ecotox (Van Zelm *et al*., 2009b)	M/E	Europe	Eco-indicator 99 (Goedkoop and Spriensma, 1999)	Beer (Cordella *et al*., 2008)	Case study applying endpoint indicators. Ecotoxicity minor contribution.
					Tuna (Hospido and Tyedmers, 2005)	Marine ecotoxicity important impact category. Metals in diesel and anti-fouling paint are polluters.
					Fish (Thrane, 2006)	Eco-indicator 99 verified conclusions obtained with EDIP (see above)
				CML 2002 (Guinée *et al*., 2002)	Cane sugar (Ramjeawon, 2004)	Herbicide loss during cane cultivation sole contributor to aquatic toxicity.
					Industrial milk (Høgaas Eide, 2002)	Comparison of three Norwegian dairies. Emissions of heavy metals in waste management phase most important for ecotoxicity. Small dairy greatest environmental impact.
				ReCiPe 2008 (Goedkoop *et al*., 2008)	New method, no case studies yet	
BETR	Multi-media fate and exposure model. No effect part (MacLeod *et al*., 2001)	M	Europe/World			
Watson	Multi-media fate and exposure model. No effect part (Bachmann, 2006)	M	Europe			

Table 10.2 Continued

Model	Characteristics	M/E*	Region	Implemented in	Case studies	Main outcomes
USEtox	Multi-media fate and exposure model. Linear effect method based on (chronic) EC50s. Developed from the six above mentioned methods, this consensus model receives broad scientific agreement (Rosenbaum *et al.*, 2008)	M	World		New method, no case studies yet	

The main ecotoxicity pollutants in food production and processing are pesticides and metals, which have their own specific qualities and properties that lead to uncertainties in LCA modelling. In current LCIA ecotoxicity models, degradation of a chemical is taken into account by following the disappearance of the parent compound only. Many pesticides, however, are known to transform in the environment to degradation products that are also harmful to the environment, in some cases even more than their parent compounds (Fenner *et al.*, 2000; Gasser *et al.*, 2007). Van Zelm *et al.* (2010) quantified uncertainty attached to the exclusion of transformation products of a number of pesticides in freshwater ecotoxicological effect factors. They show that for several pesticides, transformation products cannot be disregarded as they can damage the aquatic environment to a large extent. The fate modelling of metals is still an unresolved issue and a source of large uncertainties (Rosenbaum *et al.*, 2008). Strandesen *et al.* (2007) developed a new concept to include speciation in the fate modelling of metals. They concluded that multi-species models need to be used to characterize the potential ecotoxicological impacts of metals, since the behaviour of metals cannot be addressed by a single-species model that assumes a fairly uniform behaviour of metals in very different model regions. This indeed, increases the need for spatially-differentiated fate and exposure modelling (Strandesen *et al.*, 2007).

10.4 Future trends

A number of future research trends are envisaged. First, endpoint indicators focusing on species disappearance allow us to aggregate land use and ecotoxicity effects with each other, but also with climate change, eutrophication and acidification. However, there is a risk in double counting environmental impacts because the CFs for land use are derived from empirical data (species counts) that can also include other environmental impacts.

Second, several specific methods are available to analyse land use effects. However, the regional dependency of land use, reduces the validity of applying these methods in local case studies. Within the existing land use methods, the following improvements are required:

- Investigate the sensitivity of land use CFs towards the choice of baseline, input parameters of the species area relationship, and the application of target species
- Define land use types and the inclusion of different land use practices in more detail
- Derive CFs for developing countries, as large food production takes place here (cassava, rice, palm oil)
- Improve insight into the influence of uncertainty in parameters and choices within the species area relationship of endpoint land use models
- Develop quality indicators that cover other parts of the cause–effect

pathway than are commonly considered, such as changes in unique landscapes

Finally, for the impacts of food production and processing on freshwater ecotoxicity, specific attention in further developments should be given to:

- Increased pesticide coverage
- Inclusion of transformation products for pesticides with harmful daughter products
- Modelling of metals in a more precise way
- Inclusion of parameter uncertainty in the estimates of the CFs
- Definition, and modelling up to, an endpoint level that receives consensus among researchers

10.5 Conclusions

This chapter presented an overview of method developments that allow the assessment of environmental impacts caused by land use and ecotoxicity. Over recent years, large improvements have been made to enhance the methods and their way of interpretation. Progress in defining recommended practice has also been made, particularly for aquatic ecotoxicity with the USEtox™ consensus model. However, further testing of the methods with case studies is necessary for both land use and ecotoxicity models, including the assessment of uncertainties in the estimates of the characterization factors. It is remarkable that only a few case studies were found that consider both impact categories (see Table 10.1 and Table 10.2). Especially for agricultural products, it is important to compare and aggregate land use and ecotoxicity effects, with special attention to avoid double counting of environmental impacts.

10.6 Sources of further information and advice

More information on land use in LCIA can be found in:

- Milà i Canals *et al.* (2007a), describing a framework for LCA of land use
- Milà i Canals (2007b), describing the Soil Organic Matter concept as midpoint indicator
- Koellner and Scholz (2007, 2008), addressing a state-of-the-art endpoint modelling method for land transformation and occupation
- Hauschild *et al.* (2008a), chapter 'Land use' (pp. 101–110), describing the evaluation and recommendation of land use models
- http://fr1.estis.net/sites/lciatf2/, describing Task Force 2 on natural resources and land use of the LCIA programme within the UNEP-

SETAC life cycle initiative. The task force focuses on improvements and consensus within land use characterization methods.

More information on ecotoxicity modelling in LCIA can be found in:

- Hauschild *et al*. (2008a), chapter 'Ecotoxicity' (pp. 94–101), describing evaluation and recommendation of ecotoxicity models
- Hauschild *et al*. (2008b), describing the consensus-building process of the USEtox consensus model
- Rosenbaum *et al*. (2008), describing the USEtox consensus model
- http://fr1.estis.net/sites/lciatf3/, describing Task Force 3 on toxicity impacts of the LCIA program within the UNEP-SETAC life cycle initiative. The task force focuses on improvements and consensus within human and ecotoxicity characterization methods.

More information on pesticide modelling in LCIA can be found in:

- Birkved and Hauschild (2006), describing how to estimate field emissions of pesticides
- Van Zelm *et al*. (2009a), addressing ecotoxicity endpoint modelling of pesticides in LCIA

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11

Combining Life Cycle Assessment of food products with economic tools

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Abstract: The economic counterpart of LCA, known as Environmental Life Cycle Costing (LCC), is of increasing concern for LCA practitioners. Just like LCA, LCC may concern food products. Yet, the literature provides few applications of LCC to food products and, more generally, to nondurable products; moreover, the methodologies adopted vary significantly within the available studies. Other examples of combined environmental–economic tools for the assessment of food products include applications of Input–Output Analysis along with Material Flows Analysis (MFA) and LCA. These combinations aim at studying the way materials and substances flow through the economy and applications in these fields are well-established ones. The main results achieved by such diverse combinations of tools are discussed here, especially those which are of managerial relevance. An effort will also be made to highlight the peculiarities that may be taken into account in future applications, when carrying out economic analysis concerning food products combined with environmental analysis.

Key words: Life Cycle Assessment, environmental accounting, food products, life cycle costing, economic and environmental assessment.

11.1 Introduction

The design and development of a measuring method for environmental sustainability in food production systems is a difficult task. As Gerbens-Leenes *et al*. (2003) put it, overall environmental implications of food production are still poorly understood, especially the interactions among given environmental strategies adopted in food production systems and their effects on other resources.

As Life Cycle Thinking became the prerequisite of any sound sustainability assessment (Klöpffer, 2003), the economic aspects have been widely recognized as forming one of the three pillars of sustainability. In this sense, economic tools can be combined with LCA – though not completely integrated – as a separate complementary analysis, within a toolbox or as a way of expanding it. The economic counterpart of LCA, known as Environmental Life Cycle Costing (LCC), is an example of this combination which is of increasing interest for LCA practitioners. Just as LCA, LCC may concern food products. Yet, the literature provides few applications of LCC to food products and, more generally, to nondurable products; moreover, the methodologies adopted vary significantly within the available studies. On the other hand, examples of expansion of LCA by means of combined environmental–economic analyses include applications of Input–Output Analysis along with Material Flows Analysis (MFA) and LCA. These combinations aim at studying the way materials and substances flow through the economy and applications in these fields are well-established ones.

The main results achieved by such diverse combinations of tools will be discussed here, especially those that are of managerial relevance. Furthermore, in this contribution, an effort will be made to highlight the peculiarities that may be taken into account, in future applications, while carrying out economic analysis concerning food products to be combined with environmental analysis.

11.2 Methods of combining Life Cycle Assessment of food products with economics tools

Economic tools can be combined with LCA in several ways, as it emerges from a wide series of applications, not only concerning food products. Generally speaking, economic tools can play two main roles in Life Cycle Management (LCM). On the one hand, they can provide ways of accounting for costs within the same boundaries and with reference to the same functional unit as in LCA (Hunkeler *et al.*, 2008). On the other hand, macroeconomic-oriented accounting tools such as Input–Output Tables, either in monetary or in physical terms – in the latter case leading to Material Flows Analysis (MFA) – aim at studying the way materials and substances flow through the economy. They can be either used in hybrid LCA to extend the system boundaries to include all the complex transactions that characterize the entire National Economy, thus significantly reducing cut-offs (Hendrickson *et al.*, 2006; Suh and Huppes, 2005); or they can be used to model the economy in its physical dimension, applying the mass balance principles, so that one can assess the origins of pollution problems and estimate the impacts of certain changes in the economic material management (Bauman *et al.*, 2000). Applications of such diverse combinations of tools to food products will be considered here. The main results achieved by such applications will be discussed, making an effort to highlight the methodological peculiarities that emerge while combining economic and environmental tools to this particular kind of nondurable product.

11.2.1 The microeconomic perspective: Applications of Life Cycle Costing to food products

Although LCC is not as standardized as LCA, there is a significant body of literature that addresses its conceptual framework and methodology – for an overview see Hunkeler *et al.* (2008). Thus, applications to food products, being just applications of more generalized concepts, might seem not to pose major methodological problems. There are, in fact, evidences that LCC is also being used as a decision support tool within LCA of food products (Roy *et al.*, 2009). Yet, the literature provides few applications of LCC to food products and, more generally, to nondurable products; moreover, the methodologies adopted vary significantly within the available studies.

LCC, in its original meaning, is basically a discounted cash flows analysis, which is widely applied when purchasing durable assets or equipments. LCA, on the other hand, can be applied to both durable and non-durable goods, equally applying its own general computational principles. From a theoretical perspective, this gives rise to consistency issues (Settanni, 2008) whereas from a practical perspective two alternative approaches can be found within literature: either applying traditional LCC which is carried out separately from LCA, or assessing in monetary terms the material flows resulting from the Life Cycle Inventory.

As far as food products are considered, applications of traditional LCC make sense only if an investment in some brand new food production plant is being evaluated. For example, Clark (1997) provides a technique for estimating alternative investments in food production plants based on their capital and operating costs, and their impact on profitability. In particular, typical components of a food plant to be taken into account are shown in Table 11.1.

The main aspect in this analysis is that both capital costs and operating costs are estimated on an annual basis and are then related to units of products by dividing by the expected quantities of final product. This can be summarized, as is usual in capital budgeting practice, as:

$$c = \frac{\sum_{t=-j}^{n} Cc_t \cdot (1+r)^{-t} + \sum_{t=1}^{n} Oc_t \cdot (1+r)^{-t}}{\sum_{t=1}^{n} q_t} \qquad [11.1]$$

where

c present value of unit product costs

Table 11.1 Typical components of a food plant (Source: Clark (1997))

Capital costs (plant's components)	
Raw material receiving and storage.	Food plants differ from other manufacturing plants due to the characteristics of food raw materials, that are: ○ often perishable, ○ usually variable in properties, ○ frequently contaminated because of their agricultural origin, ○ often seasonal in supply.
Packaging material receiving and storage	
Processing equipment and facilities.	Many food processes have in common elements such as mixing, forming, cooking and preserving. Cooking may involve direct or indirect heating, preserving may involve heating, freezing, chilling, chemical sterilization or irradiation.
Material handling	Conveying of food materials is a significant cost in most food plants because it can have a significant impact on material properties.
Packaging	It is necessary to deliver food products in relatively small consumer packages. Primary to tertiary packaging are to be taken into account.
Utilities	Food plants use all the usual factory utilities. Moreover, vacuum, refrigeration and sanitizing chemicals are common in food plants.
Environmental controls	Food plants generate relatively large quantities of liquid and solid wastes which normally are biodegradable but may be quite strong and may require special provisions for disposal. Air emissions control has not traditionally been a major concern for food plants, but is becoming more so.
Building	
Engineering and construction fees	
Contingency	Is intended to account for errors in the other elements of the estimate.
Operating costs	
Raw materials	
Packaging materials	
Energy	
Labour	
Depreciation	
Indirect costs	

Cc_t Capital costs in year t
Oc_t Operating costs in year t
q_t yield of finished food product in year t
r discount rate
j time period expressed in years, from the initial disbursement to the beginning of the plant's operations
n plant's economic life.

The outcome is the present value of capital and operating costs per unit mass of food product produced by a given plant. Even if the analysis is much more accurate than discussed above, usually the only relevant parameter linking cost estimation to the manufacturing system is the yield of finished product resulting from the consumption of raw materials. Although this may prove to be very useful for rapid preliminary cost estimations of food production plants on the basis of existing food factories' cost data and production capacities (Marouli and Maroulis, 2005; Montaner *et al.*, 1995), other stages concerning the food product, such as the agricultural and use ones, are not usually considered in the economic analysis. It can be noted, indeed, that the concept of 'life cycle' in such analysis is understood as a time-oriented one, being concerned with the economic life cycle of a plant (and more in general of some durable asset), whatever product it produces. LCA focuses, instead, on the 'physical' life cycle of a product, following its supply chain, from raw material extraction to final disposal. Yet this substantial difference in the meaning of life cycle within economic and environmental analysis seems not to pose major problems while developing applications. The prevailing practice, indeed, is to combine the unit cost figure obtained from the traditional LCC with some environmental impact indicator per functional unit – especially obtained from tools such as LCA. This solution is particularly evident in works such as those of Roy *et al.* (2006, 2007) concerning rice, where local parboiling processes have been evaluated considering both the investment in different boiler processes and the life-cycle inventory results concerning energy consumption and CO_2 emissions of such processes.

As a matter of consistency, however, the system boundaries should be the same for both the economic and the environmental analysis. On the one hand, if the economic analysis is focused on durable goods employed in food production, the same perspective will be adopted while carrying out the environmental analysis. The relevant life cycle should therefore range from the production of the asset to its operation and end-of-life. With reference to the production of rice, for example, Blengini and Busto (2009) consider also the indirect environmental burdens of capital goods within the LCA because the agricultural subsystem considered is characterized by a high degree of mechanization – whereas capital goods relevant to the post-agricultural phase are excluded, based on the assumption that the contribution of buildings and machinery used in the post-harvesting processes is virtually negligible. Another example of a study in the agro-food industry where durable assets are of concern for both LCA and cost analysis has been provided by a study in which an energy inventory of the must enrichment process has been developed (Tassielli and Notarnicola, 2008). It considers the phases of plant production, transport, operation, cleaning, components recycling at the end of life, waste disposal in landfill, together with assessment of the total energy consumption which takes into account the direct and indirect contributions. The potential energy savings result in cost savings that can therefore be estimated consistently with the scope and system boundaries of the analysis of physical flows.

On the other hand, if the 'physical' life cycle of the food product is of concern, as it is for LCA, the same should be for the economic analysis, namely LCC. The characteristic of such an approach is that of being more focused on the life-cycle inventory of the food product. The generalized formula has been outlined by Rebitzer and Nakamura (2008):

$$LCC = \sum_{(\text{life cycle phase } 1)}^{(\text{life cycle phase } n)} \sum_{(\text{process } 1)}^{(\text{process } i)} \left(\mu_i \times \sum_{(\text{cost element } 1)}^{(\text{cost element } p)} \sum_{(\text{flow } 1)}^{(\text{flow } q)} \text{amount}_q \times \text{cost}_p \right) \qquad [11.2]$$

where:

i process-specific variable
p cost category-specific variable
q process flow-specific variable, either input or output
μ process-scaling factor related to the product system
n life-cycle phase-specific variable.

From Eq. 11.2 it emerges that firstly the costs per unit process can be calculated by multiplying the costs per reference unit by the absolute amount of the process flow-specific variable; secondly, the result is multiplied by the amount of the different processes that is needed for the considered functional unit (scaling factor). Then the costs of all unit processes are aggregated for all life-cycle phases during the complete life-cycle time.

This approach has been applied to the comparison of conventional versus organic extra virgin olive oil (Ciroth *et al.*, 2008; Notarnicola *et al.*, 2004a). [This is one of the few applications of detailed life cycle inventory-based LCC to nondurable goods.] The study clearly takes into account the agricultural phase for the economic analysis, pointing out the major issue of a lower organic yield and its repercussions on both unit product cost and unit impact assessment. The importance of the agricultural phase due to production yields has been highlighted also for non-food products within the agro-industry sector, such as ethanol biofuel (Nguyen *et al.*, 2008; Hu *et al.*, 2004; Zhang *et al.*, 2003; Pimentel, 2003) – yet an in-depth analysis of this kind of commodity is outside the scope of this paper.

Finally, Krozer (2008) presented some applications of LCC to food products, focused on the environmental innovations along the product chain on the basis of data from life-cycle inventories. In particular, open-air cultivation of tomatoes has been compared with prospective greenhouse cultivation, considering the compliance cost due to intensive energy use and hazardous wastes. Moreover, the importance of fertilizers' emissions on fields has been highlighted, since fertilizers have been identified as being the main cost factors in the production of plant fats, which is also typical for many high-value edible food products.

One may notice that applying LCC to the agro-industry products not only allows one to identify the yield effect on unit costs, but also to expand the product cost structure in order to include subsidies, taxes, and potentially hidden or external environmental cost, as well as the additional environmental costs of transporting foods to retail outlets, and then to consumers' homes, and the cost of disposal of wastes (Pretty *et al.*, 2005). Fertilizers, in particular, are of great concern from both an environmental and an economic viewpoint. Most benefits of pesticides are based on the direct crop returns. Such assessments do not include the indirect environmental and economic costs associated with the recommended application of pesticides in crops, which must be estimated (Pimentel, 2005; Uri, 1997). The above mentioned cost elements, especially subsidies and the external costs, are expected to heavily affect the ranking of alternative options, unless the unlikely event that one specific option is found to be both environmentally sustainable and cost effective compared to the others.

11.2.2 The macroeconomic perspective: hybrid methods and material flow analysis

Macroeconomic-oriented accounting tools such as Input–Output Tables, either in monetary or in physical terms, aim at studying the way materials and substances flow through the economy. They can be either used in hybrid LCA of food products to extend the system boundaries to include all the complex transactions that characterize the entire economic system; or they can be used to reveal the importance of understanding the physical structure underlying any food production system. Applications of hybrid LCA using economic Input–Output Tables (IO-LCA) have been carried out for the improvement of the pasta life-cycle inventory (Notarnicola *et al.*, 2004b). It has been shown that the environmental profile of an agro-food product is strongly characterized by the use of products such as pesticides and fertilizers, whose production steps are quite weak in the LCA databases. For this reason, the use of IO-LCA has been recommended for the background processes, for which good quality data are not available, whereas the detailed LCA is to be used for the most important foreground processes' data. Indeed, the study demonstrated that, in general, the IO-LCA approach does not (or just partially) model the emissions coming from the use of chemicals such as methyl bromide, fertilizers and pesticides, though it allows one to reduce truncation errors due to neglecting upstream operations in process LCA.

Also, Input–Output Tables have been employed for the identification of the hot spots of a hamburger meal (Madsen and Effting, 2003). In this case, it has been pointed out that the agriculture sector is very heterogeneous, hence the average product from this sector is probably not representative of specific products such as potatoes, tomatoes and salad; this is relevant when comparing the contribution of processes. In order to carry out a reliable IO-LCA, sectoral disaggregation of the Input–Output Tables at the highest possible level is needed. For this reason, the practice of using data sets that are representative of different economies does not represent a limitation as far as these data offer the desired disaggregation level (Notarnicola *et al.*, 2004b).

The problem of sector aggregation is evident in another example of the use of input–output tables. A widespread analysis has been recently carried out within the European Union with the aim of ranking the impact of food products in the EU-25 economy in order to support integrated product policies (IPTS, 2006). From a methodological point of view, the model adapted the latest model developed for the United States highly disaggregated sectoral data to Europe, on the basis of the assumption that there are similar production processes in the US and Europe for most products. As a result, the study pointed out that products in the food and drink area are having the greatest impact, together with private transport and housing, causing 20 to 30% of the various environmental impacts of private consumption, considering the full food production and distribution chain 'from farm to fork'.

The above mentioned study pointed out that within food products, meat and meat products are the most impacting, followed by dairy products. More specifically, Weidema *et al.* (2008) assessed the total environmental impact from consumption of meat and dairy products in EU-27 by the use of hybrid life-cycle assessment. The study found out that there are clear differences among the various types of meat, with beef having larger environmental impacts than poultry and pork, and also having a monetarized environmental impact in terms of externalities amounting to 112% of its private costs.

Duchin (2005) described an integration of life-cycle assessment with a new input–output model of the world economy, to analyse the environmental and economic repercussions of alternative future diets. By using an Input–Output Table extended to include greater detail about agriculture and food production coming from life-cycle inventories, the study concluded that a global shift towards a Mediterranean-type or other plant-based diet could be expected to have a more favourable impact on the environment and on health. Yet, the environmentally beneficial impacts of adopting a plant-based diet could be more than offset by the upgrading of nutritionally deficient diets, especially in developing countries – though the outcomes will depend not only on dietary choices but also on changes in the current practice of food production.

Finally, another example of combining LCA and Input–Output Analysis for the study of food consumption chains is the economic extended MFA (Kytzia *et al*., 2004). It focuses more on the definition of a common system for both physical and economic dimensions of industrial systems, based on Input–Output Analysis, so that financial flows related to the physical structure can be included in the analysis. Starting from the assumption that food production in industrialized countries uses natural resources inefficiently, MFA focuses on a specific geographic area during a certain period of time, to investigate all major products, expressed as mass units per time period, used to produce and to distribute consumer food. Resource consumption can be described in terms of primary energy consumption and land use. The analysis of financial flows can be based instead on indicators such as material and other costs and the production volume of food and related products. A specific application of MFA assessed three scenarios in both physical (energy consumption and land use) and economic terms: the substitution of meat and milk with grain and vegetables, an overall change towards organic cultivation methods, and the adoption of Best Available Technologies in retailing and households for cooling devices. The study found that only a switch to full vegetarian diet results in significant efficiency gains, whereas all the other scenarios result in minor improvements.

11.3 Discussion and conclusion

In this paper, the methods for combining LCA of food products with economic tools have been briefly reviewed. As economic aspects are increasingly recognized as a pillar of sustainability, this combination is gaining more and more relevance.

It has been argued that economic tools can play two main roles in Life Cycle Management (LCM). On the one hand, they provide ways of accounting for costs within the same boundaries and with reference to the same functional unit as in LCA. On the other hand, macroeconomic-oriented accounting tools such as Input–Output Tables, in monetary or in physical terms, can be used either in hybrid LCA to extend the system boundaries, or they can be used to model the economy in its physical dimension, applying mass balance principles.

Examples of such methods of combining LCA and economic analysis have been provided with reference to food products. As far as the accounting for costs at the microeconomic level is concerned, it has been pointed out that there is little evidence of the application of combined LCC and LCA to food products. The LCC tool is mainly used when an investment in food production plants is being assessed. Furthermore, the approaches adopted when LCC is used within environmental management may vary significantly. Cost elements, especially subsidies and the external costs, are expected to heavily affect the ranking of alternative options, unless one specific option is found to be both environmentally sustainable and cost effective compared with the others.

As to the use of macroeconomic analysis in combination with LCA, it is important, on the one hand, to extend the system boundaries to include all the complex transactions that characterize the entire economic system. Such an approach has been used, even at the institutional level, to support integrated product policies. On the other hand, it can be used to analyse the effects of the choice of diet on the economic structure of the system – and possibly to carry out some combined analysis of material and financial flows to assess the economic effects of actions taken to improve the ecological performance of food production systems.

The combination of macroeconomic analysis and LCA may prove to be particularly useful since, compared to detailed life-cycle inventories, many models of entire economies employ a much smaller number of categories for representing production and consumption activities. As Duchin (2005) pointed out: 'The collaboration of input–output economists with life-cycle analysts makes it possible to handle systematically a moderate level of detail variable, and to represent the interdependency among variables while also respecting the physical constraints of the system'.

Further research should address the applications of LCC to food products, given the increasing interest for this tool at the methodological level, to develop an adequate approach to properly manage the peculiar aspects of agriculture and food products. Indeed, as Blengini and Busto (2009) remark, applying analytical environmental management tools to the agri-food chain is facilitated by modern and technological farming, which can be compared to the industrial systems; yet it involves a sequence of natural *and* industrial processes which cannot be controlled completely.

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12

Inclusion of social aspects in Life Cycle Assessment of food

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Abstract: This chapter focuses on how the social 'pillar' of sustainability might be more fully integrated into the Life Cycle Assessment framework in the specific context of food production systems. The chapter includes a discussion of why social impacts are relevant in the context of food production systems and also highlights several existing food system applications of social impacts using life-cycling thinking.

Key words: Life Cycle Assessment, food production, social impacts, social indicators.

12.1 Introduction

The global population is exploding. From 1950 to 2000, the population grew from 2.5 billion to just over six billion, a 247% increase. Projections suggest that while the growth rate itself is beginning to slow, the absolute size of the population will increase another 47% by 2050 (United Nations, 2004) and every single one of these individuals will eat.

The Food and Agriculture Organization (FAO) of the United Nations estimates that over the next 30 years, global food production will need to increase by 50% to meet the needs of the global population (Bruinsma, 2003, cited in Nonhebel, 2006), suggesting that food production must increase at a rate even greater than the existing population. This is likely due to increasingly diverse diets around the globe and a related increase in the consumption of both more food per person and more resource-intensive food products, such as meat and milk (Nonhebel, 2006), which require greater agricultural production per unit of food produced. One example, among many, is shown by Nonhebel (2004), who finds that 4 kg of wheat feed are required to produce 1 kg of pork.

These projections, coupled with a growing recognition that current levels of production, whether on land or at sea, are responsible for a wide range of environmental impacts (e.g. deforestation, loss of biodiversity, and greenhouse gas emissions) and have contributed to an increased demand for methods to assess 'best practices' for food production systems. This increasing demand is coming not only from policymakers and non-governmental organizations, but also from food producers and retailers focused on using sustainable business practices. These best practices no longer necessarily refer simply to economic efficiency, but rather, frequently focus on economic, social and environmental considerations, sometimes referred to as the 'triple bottom line'.

While there is lack of consensus on how sustainability is rigorously defined, the majority of broadly recognized sustainability concepts are based on the United Nations Brundlandt Commission (WCED, 1987) seminal definition and typically include two key principles: (i) balancing economic development with social well-being and environmental protection; and (ii) balancing the distribution of costs and benefits between present and future generations.

Klöpffer (2003) argues that any sound sustainability assessment, be it food production or otherwise, must use life-cycle thinking as a prerequisite. Life Cycle Assessment (LCA) is the only internationally standardized cradle-to-grave assessment methodology (ISO 14040 and 14044, 2006a and 2006b) for assessing environmental impacts; therefore, it seems potentially redundant to create a new sustainability assessment methodology if LCA can be adequately adapted or modified to include social (and economic) considerations. To this end, the United Nations Environment Programme (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC) are jointly supporting an International Life Cycle Partnership, also known as the Life Cycle Initiative. In 2004, a project on Social Life Cycle Assessment began as part of this initiative.

This chapter will first consider how the social 'pillar' of sustainability might be more fully integrated into the Life Cycle Assessment (LCA) framework in the specific context of food production systems. The remainder of the chapter will focus on (i) why inclusion of social impacts is of importance when assessing the sustainability of food production systems; (ii) methodological considerations when assessing social impacts; (iii) existing applications of life-cycle thinking in the context of food production systems and (iv) future trends.

12.2 Including social aspects in Life Cycle Assessment (LCA) of food

According to the International Labour Organization, the agriculture sector is the second largest contributor to global employment, accounting for an estimated 35% of jobs globally, many of which are found in developing countries (ILO, 2008). From a food system perspective, agriculture is typically only the first step in a much larger production chain, which also includes additional steps, such as processing, transportation, wholesale, and retail.

In addition, employment in food production is frequently found to be both low paying and physically dangerous, particularly for wage workers. A 1996 report by the International Labour Organization, *Wage Workers in Agriculture: Conditions of Employment and Work* (ILO, 1996) concluded that nearly half of those employed in agricultural production globally work for wages and workers in rural agriculture tend to be worse off than their non-agriculturally employed counterparts, particularly in Asia and Africa. The report also found that agricultural workers face higher accident and fatality rates than workers in other industrial sectors. Dangerous working conditions are found not only in rural developing countries, but also in developed countries such as the United States. The United States Bureau of Labor Statistics (Christie, 2007) reported that, in 2006, the top ten occupations with the highest fatality rates included fishermen (1), farmers/ranchers, (6) and agriculture workers (10).

In addition to the impacts of food production on those employed within the sector, this chapter will discuss three global trends or shifts in food production where social impacts merit additional consideration. The first is the transition from traditional to industrial agriculture and food processing. In developed countries, this transition has already occurred, leading to substantial increases in yields and efficiency, yet it also came with changes, some of them social. For example, in 1900, agriculture accounted for approximately 40% of employment in France, Germany and the United States; however, by 1990, agricultural employment was down to 6%, 3% and 3% respectively (Lindsay, 2003). Many developing countries still use traditional agricultural practices, but look to industrial agricultural methods to alleviate potentially economic and social problems, such as food scarcity.

Similar trends have been seen in capture fisheries, with much of this decline occurring in industrialized countries. In Japan and Norway the number of fishers employed has decreased by 61 and 42%, respectively, since 1970, while all industrialized countries globally have seen an estimated average decline of 24% from 1990–2006 (FAO, 2009). These declines are likely to be a result of increased operational efficiencies, in turn requiring fewer individuals per boat (FAO, 2009).

As evidenced in these examples, such transitions are not necessarily trade-off free. It may be possible to use data from the experience of developed countries, which have already made this transition, to assess the potential social impacts likely to result in developing countries and globally from such a shift.

Another major trend seen in food production systems is a shift in both consumption and trade from bulk foods to processed or value-added foods. Urbanization and rising incomes, among other factors, are likely catalysts of this trend, as diets diversify because of increased access and the cost of preparing food rises for consumers (Dyson, 1999). According to the FAO (2005), processed agricultural products not only are increasing, but they now account for almost half of agriculture trade at the global level.

Directly related is the final trend of note, the relatively recent increase in global trade of food products, which in many countries, particularly developing ones, has contributed to a transformation of agricultural and food markets (FAO, 2005). Of particular importance is the rapid growth of supermarkets in developing countries, which, according to the FAO (2005), are frequently owned by multi-national companies located in Europe, North America and Japan. While this may provide positive benefits for downstream consumers, such as lower prices, reduced seasonality and increased food safety, there are also negative social impacts. For example, in addition to out-competing small local retailers, the purchasing practices of these global chains frequently have significant implications for small-scale farmers through (often strict) quantity and quality requirements, such as sorting and grading of produce, documenting farming practices and meeting certification standards (FAO, 2005).

These trends also highlight that, while social impacts may most easily be associated with personnel or employees, potential impacts on other groups of individuals (e.g. stakeholders) may also require consideration. Examples of this include the effect of increased consumption of processed foods on consumers, the impact of large supermarkets on local communities, and the role of genetically-modified-seed suppliers in industrial agriculture.

12.3 Methods of including social aspects in LCA of food

A number of voluntary or self-imposed standards for social performance currently exist, from Fair Trade Certified or the Ethical Trading Initiative to industry level corporate social responsibility initiatives. More formal methods to evaluate and quantify social impacts also exist, such as Social Impact Assessment (SIA) (e.g. Barrow, 2000), but such methods do not always include life-cycle thinking. According to Vanclay (2002):

> SIA is the process of analyzing (predicting, evaluating and reflecting) and managing the intended and unintended consequences on the human environment of planned interventions (policies, programs, plans, projects) and any social change processes invoked by these interventions so as to bring about a more sustainable and equitable biophysical and human environment.

It is interesting to note that SETAC guidelines (Consoli *et al*., 1993) recommend a 'social welfare' impact category for all detailed LCAs, yet there is limited literature on how such metrics might be developed. One reason for this may be that many social impacts appear to be more heavily influenced by value judgments rather than by absolute standards, which typically dominate the biophysical component of any LCA. Furthermore, as noted by Dreyer *et al.* (2006), the primary driver of a product's environmental impacts is typically the production process, whereas social impacts may be dependent on individual company behavior and management of operations.

How then does one go about including social impacts in a Life Cycle Assessment for food, or any LCA for that matter? The following subsections will discuss several key considerations for doing so.

12.3.1 Defining key social indicators

Both Dreyer *et al.* (2006) and Kruse *et al.* (2009) suggest the use of a top-down and bottom-up approach in the development of social impact categories and indicators in the context of a life-cycle framework. In a top-down approach, broadly recognized societal values typically serve as a starting point. Common references for these values include any number of international conventions, agreements and guidelines, such as the International Labour Organization (ILO), United Nations Global Compact, The Universal Declaration of Human Rights, Corporate Social Responsibility (CSR) Europe, and the Global Reporting Initiative (GRI). For example, from 1930–99, eight ILO Conventions have identified fundamental rights for human beings in the workplace that fall into four categories: freedom of association and collective bargaining; abolition of forced labor; equality; and elimination of child labor.

This type of approach is also consistent with recommendations from the International Standards Organization (ISO) for Environmental Life Cycle Impact Assessment (LCIA) methods, which note 'the impact categories, category indicators and characterization models should be internationally accepted, i.e. based on an international agreement or approved by a competent international body' (ISO, 2006b). In the context of food production systems, indicators identified using a top down approach are likely to be similar, if not identical, to those used in other production sectors, although both Dreyer *et al.* (2006) and Kruse *et al.* (2009) note the importance of measuring social impacts using a country- or region-specific context. This can potentially be done using different criteria for different regions (e.g. developed versus developing countries), or by accounting for differences in the measurement methods themselves (e.g. define a 'fair wage' relative to the living wage for the region or country in which a company operates).

A bottom-up approach, in contrast, uses stakeholder or company level input as the starting point. Social impacts may vary between sectors/industries, and a bottom-up approach allows indicators to potentially address sector/industry-specific considerations, both in terms of who potentially affected stakeholders might be and how they might be affected (Kruse *et al.*, 2009; Dreyer *et al.*, 2006).

In order to have a common set of indicators of performance that relate to broadly recognized values (as identified by global initiatives), a top-down approach seems to be the logical starting point; however, a second tier of bottom-up indicators may serve as a complement, allowing for more detailed assessment of industry or region-specific issues or impacts.

In addition to Dreyer *et al*. (2006) and Kruse *et al*. (2009), there are several other publications of relevance to the development of social indicators in the context of Life Cycle Assessment. While some of the following publications focus on primary methodologies other than LCA, the suggested social impact categories and indicators may still serve as a relevant starting point.

Brent and Labuschagne (2006), in their development of a Social Impact Indicator (SII) calculation procedure, outline a social sustainability criteria framework, describe impact categories and measurement units, and conduct a case study using the SII calculation in the context of the South African process industry. Similarly, Schmidt (2003) presents a provisional selection of social indicators in the context of work on a case study within the chemical company BASF, and additionally describes the social goal each indicator represents, a possible measurement unit, a general baseline standard and the life-cycle phase of relevance. Indicators of particular relevance to food production systems that have been used in existing studies are also described in more detail in section 12.4 of this chapter.

12.3.2 Integrated versus non-integrated approaches

As mentioned previously, one potential difficulty with including social impacts in a Life Cycle Assessment is their relationship to the product. In order to trace the impact through the entire life cycle of a product, the impact must be related to the production process; however, social impacts in many cases appear to be more company specific than process specific. In the LCA context, analyzing impacts at a company level versus a process level, as done in the environmental LCA, means that the relationship between social impacts and a specific product or service may no longer be clearly defined (Dreyer *et al*., 2006).

Dreyer *et al*. (2006) in *A Framework for Social Life Cycle Impact Assessment* focuses their discussion primarily on the relationship between an individual company's conduct and the social impacts relevant to stakeholders. They suggest a two-layer framework with both an obligatory and optional set of impact categories, where the obligatory impact category reflects minimum expectations for a company based on broadly recognized social values and the optional impact category focuses on impacts of particular relevance to the company.

Another option is a multi-faceted approach, as suggested by Kruse *et al*. (2009), in which some indicators are directly related to the process (i.e. additive indicators), others are not additive but can be described using a consistent metric at each step in the chain (i.e. descriptive general indicators), and others are described only at a single step in the chain (i.e. descriptive specific). Four criteria are used to determine an indicator's categorization:

relationship to the functional unit, measurement method, applicability and comparability (Kruse *et al.*, 2009).

One need in the development of indicators is consistency in their selection. It is important not to allow data availability (or lack thereof) and/or industry accountability to affect the choice of appropriate indicators (Dreyer *et al.*, 2006). At the same time, however, Kruse *et al.* (2009) suggest that 'data needed to accurately measure/describe each indicator should either currently exist or can be readily collected with justifiable expenditure and effort.' On this same topic, Makishi *et al.* (2006) note:

> On the one hand, the selection of social indicators to provide a social profile of a product, a process or a system should follow a series of criteria such as impartiality and relevance in order to measure the promotion of employment, the improvement of living and working conditions, safety at work, etc. On the other hand, in social LCA other aspects need to be taken into consideration: product/process-relatedness, system boundaries, cut-off criteria, etc. If only the indicators that follow both criteria are selected, relevant social problems may be left aside. If all the socially relevant problems are taken into consideration, they might not fit into the LCA frame and therefore the three aspects of the life cycle sustainability analysis – environment, economy and society – cannot be represented together.

Finally, in the context of food production systems, regardless of the general social impact category/indicator development approach used, it may be appropriate to consider differentiating the primary stage(s) of production, such as farming, fishing or processing, and the rest of the production chain, as suggested by Kruse *et al.* (2009). This is suggested as a means to focus the analysis where the impacts (and possibilities to achieve useful results) are the greatest.

First, in the case of food production, one could argue that the primary stages of production are where the impacts (biophysical or social) are more tightly connected to the production of a specific project. Second, social data appear to be generally more available and relatable to the process/product in these early stages (Kruse *et al.*, 2009) and impacts may also be more important at these early stages. For example, wages or working conditions in a retail store selling apples are not likely to be as relatable to a unit of apples (given the large number of products typically sold in retails stores) as are the wages or working conditions on an apple farm.

12.3.3 Developing baseline standards

Another need for social indicators in the context of LCA, as for the biophysical indicators, is the development of baseline standards against which the indicators can be tracked and measured. Unlike the majority of biophysical indicators considered in Life Cycle Assessments, in the case of social indicators, such a standard may not be clearly defined, and furthermore, may be subjective. For example, when considering contribution to acidification, greenhouse gas emissions or carbon dioxide emissions, it is fairly obvious that the baseline standard for all three indicators would be zero.

Some social indicators have an equally clear baseline standard. For example, it is generally agreed that there should be zero forced labor and that the fewer number of accidents and fatalities in the workplace, the better. Other social indicators are more difficult, such as fair wages or worker benefits.

With respect to establishing a baseline or benchmark, it is also important to distinguish between absolute and relative values. For example, the indicator for worker safety is one for which a generally agreed upon absolute standard across all food production systems exists; that is, zero worker deaths/accidents. Another indicator which potentially can be measured using an absolute standard is 'fair wage', where the standard in all cases requires that the average wage of workers is at least equal to the 'living wage' for the region/country in which they work; however, in this case, an absolute wage value would not suffice as the wage necessary to attain the same standard of living is not equal across regions/countries. Instead, an absolute value specific to each region/country would perhaps be more appropriate.

In some cases, however, the establishment of an absolute standard is difficult, if not impossible. For these indicators, the use of a relative standard can still provide a meaningful measure of the indicator relative to itself or to similar fisheries over time. As an example, consider an indicator designed to measure the percentage of personal income derived from a fishery. It may not be appropriate to create an absolute standard (e.g. all fisheries should and/or could provide the same level of personal income, all fishermen should derive 100% of their personal income from a single fishery), but it is reasonable to assess performance relative to benchmarks. For example, if fishermen, on average, derive 75% of their personal annual income from a fishery at one point in time, then a trend away from a measure of 75% in the future could be an indication of changes in the socioeconomic structure of the fishery, although additional information would likely be necessary to assess whether this trend was of a positive or negative nature.

12.4 Applications

While the application of life-cycle based social indicators in the context of food production systems is fairly limited, this chapter will highlight three case studies of their application. This section does not represent an exhaustive review of all studies on the topic, but rather, attempts to highlight several different applications of social indicators for food systems in the context of Life Cycle Assessment or through the use of life-cycle thinking.

12.4.1 Life-cycle-based sustainability indicators for assessment of the US food system

In their publication *Life Cycle Based Sustainability Indicators for Assessment of the US Food System*, Heller and Keoleian (2000) develop and apply a number of social indicators (in addition to environmental and economic indicators) across life-cycle stages of the United States food system, including origin of resource, agricultural growing and production, food processing, packaging and distribution, preparation and consumption, and disposal. Their study does not conduct a Life Cycle Assessment, but rather, uses life-cycle thinking to conduct a national level food system assessment in an effort to generally describe its overall sustainability (i.e. environmental, economic and social).

Indicators are not tied to a product, process or even a company, but rather, relevant indicators are simply described at each point in the food system life cycle (see Table 12.1) and then the authors assess trends in these indicators, when possible, across time. At a minimum, this study is a good example of the types of indicators (both general as well as industry/production system specific) and one that stakeholder groups may want to consider when conducting a Life Cycle Assessment of food production.

Table 12.1 also highlights potential difficulties associated with integrating social impacts into a Life Cycle Assessment framework, namely, that both impacts and stakeholders may vary across the stages of the life cycle.

12.4.2 Social impacts of Danish fish products

Thrane (2004) assessed the environmental impacts from Danish fish products, but also included several social impacts. The study primarily focused on the fishing, processing and wholesale stages of the product life cycle, but also measured impacts at the landing and auction, transport, retail and use stages. A MECO (Materials, Energy, Chemicals and Other aspects) analysis was used as a structured way to collect data that then fed into a Life Cycle Assessment. In the MECO component of the study, health and safety impacts, if possible, are described at each point in the chain through the use of indicators describing the number of accidents, the types of accidents and the number of fatalities

In addition to a more traditional LCA, which focused on six flow related impacts, Thrane also conducted a qualitative LCA that covered the social impacts including occupational health and safety, and non-work-related noise, odor, accidents, and visual aspects. With respect to occupational health and safety, the primary stages of production (i.e. fishing and processing) were found to be the ‘hot spots’ in terms of social impacts as described by accidents and fatalities. Noise, odor, accidents, and visual aspects, all non-work related, were found to have the largest impact in the transport stage of the life cycle, having social impacts at a local level near transport routes.

Table 12.2 shows the results from the qualitative LCA for the social

Table 12.1 Summary of social indicators (adapted from Heller and Keoleian, 2000)

Life cycle stage	Stakeholders	Social indicator
Origin of (genetic) resource – seed production, animal breeding	– Farmers – Breeders – Seed companies	– diversity in seed purchasing and seed collecting options – degree of cross–species manipulation – average age of farmers
Agriculture, growing and production	– Farm operators – Farm workers – Ag. Industry – Ag. Schools – Government – Animals	– diversity and structure of industry, size of farms, # of farms per capita – hours of labor/yield and/income – avg. farm wages vs. other professions – # of legal laborers on farms, ratio of migrant workers to local laborers – % workers with health benefits – # of active agrarian community organizations – % of ag. schools that offer sustainable ag. programs – # animals/unit, time animals spend outdoors
Food processing, packaging and distribution	– Food processors – Packaging providers – Wholesalers – Retailers	– quality of life and worker satisfaction in food processing industry – nutritional value of food product – food safety
Preparation and consumption	– Consumers – Food service – Nutritionists/health professionals	– rates of malnutrition – rates of obesity – health costs from diet related disease/ conditions – balance of average diet – % of products with consumer labels – degree of consumer literacy re: food system consequences – time for food preparation
End of life	– Consumers – Waste managers – Food recovery & gleaning orgs.	– ratio of (edible) food wasted vs. donated to food gatherers

impacts considered in this study. For more detail on the other indicators considered, see Thrane (2004).

12.4.3 Social impacts of salmon production systems

Kruse *et al*. (2009) not only propose a methodological framework for using social and economic indicators as a complement to Life Cycle Assessment, but also describe how the indicators might be used in the context of salmon production systems. Salmon was chosen as an example of an international super commodity. Indicators are broken into three categories: additive,

Table 12.2 Summary of social indicators (adapted from Thrane, 2004)

		Welfare Human		
		Human toxicity	Occupational health and safety	Noise and accidents
Fishery	D	(+++)	+++	(+)
	S	(+++)	+++	(+)
	P	(+)	+++	(+)
Landing	D	~0	~0	~0
	S	~0	~0	~0
	P	~0	~0	~0
Processing	D	+	+++	+
	S	+	+++	+
	P	+	+++	+
Wholesale	D	~0	+	~0
	S	~0	+	~0
	P	~0	+	~0
Transport	D	+	No data	++
	S	+		++
	P	+		++
Retail	D	~0	No data	~0
	S	~0		~0
	P	~0		~0
Consumer	D	++	No data	+++
	S	++		+++
	P	++		+++

Impact potential compared to other life cycle stages: (+++) Large, (++) Medium, (+) Small, (~0) Very Small
Species: (D) Demersal fish, (S) Shellfish, (P) Pelagic fish

descriptive general and descriptive specific. While the additive indicators described in the study focus primarily on economic impacts, there are several that can indirectly account for social impacts in a way that allows the impact to be related to the functional unit. One example is working hours, which additionally can be broken down further by gender specific or migrant labor (e.g. working hours for men versus women or migrant versus non-migrant labor). Another important additive indicator, which has also been measured in other studies (e.g. Ellingsen in Mattsson and Ziegler, 2004; Thrane, 2004), is worker safety, measured as the number of fatalities or accidents per functional unit of the product.

Descriptive general indicators listed in this study tend to focus on globally recognized values for workers and working conditions such as living wage, worker benefits, and right to organize. While these indicators do not relate to the functional unit, they should be, in theory, measurable at each point in the life cycle of a product. Table 12.3 shows the complete list of descriptive general social indicators included in the article.

Descriptive specific indicators tend to describe more bottom-up, industry-specific impacts and may not be relevant at all points in the chain, but are still complementary to an LCA. Kruse *et al*. (2009) use the example of the potential impact of pesticide use on workers – which may be of relevance for certain types of food production systems (e.g. coffee) but not for others (e.g. fishing) and furthermore is unlikely to be relevant to other steps in the chain such as distribution or retail. For a complete list of descriptive specific indicators, see Table 12.3.

12.5 Future trends

As discussed previously, there are difficulties with measuring social impacts in the context of Life Cycle Assessment that do not exist for the majority of environmental/biophysical impacts currently considered in the context of LCA. Under future trends, there are two particular issues that merit discussion if social indicators are to be practically implementable and have wide applicability. The first is the issue of data. As Schmidt (2003) points out, 'Whereas ecologically relevant in- and outputs related to one product unit (e.g. 1 kg or 1 MJ) can be found in special Life Cycle Assessment databases, there have so far been no corresponding databases for social aspects.'

Certain social indicator data may be difficult, if not impossible, to collect. A possibility for filling data gaps may be the use of average data, as suggested by Weidema (2005); however, this may not be appropriate for all social indicators. Kruse *et al*. (2009) provide the following example to highlight this issue, 'To produce salmon feed, one can estimate with a high degree of certainty the amount of energy and materials that are needed to produce

Table 12.3 Summary of social indicators (adapted from Kruse *et al*., 2009)

Descriptive General	Descriptive Specific
Fair wage	Contribution to income
Employment benefits	Fair price
Hours worked per week	Access
Forced labor	Latent quota
Discrimination/gender	Owner-Operator
Right to organize	Adjacency
Age distribution of workers	Compliance
Minimum age of workers	
Access to bathroom	
Access to potable water	
Industry concentration	
Distance traveled	

one functional unit, but the same cannot necessarily be said for working conditions at all of the feed production sites in the system.'

A future trend, therefore, may be an increasing demand for reporting of social impact data, whether through voluntary reporting by individuals, companies or productions systems, or as a requirement based on regulation or standards. Many companies are already doing this through corporate social responsibility initiatives and reporting, although the lack of a set of common measures of performance results in each company choosing how and what it reports. One example is the international coffee company, Starbucks. According to their own report, they use the Global Reporting Initiative's (GRI) G3 Sustainability Reporting Guidelines to inform their CSR reporting on a variety of indicators across categories such as products (e.g. fair trade), society, environment, workplace and diversity (Starbucks, 2007).

This relates directly to the other trend of note, and that is the very strong role Life Cycle Assessment has the potential to play in assessing the sustainability of major trends both in food production and more generally, when considering broad scale issues such as climate change. Being able to assess trade-offs not only between different social impacts, but also between the three pillars of sustainability, will be of increasing importance as the world looks for ways to address these types of globally relevant issues.

The idea of assessing one's 'carbon footprint' is a good example of a trend that considers only one facet of sustainability. If we are trading off reduced carbon (and presumably decreased environmental impacts) for increased social impacts, then this needs to be considered. Life Cycle Assessment offers this potential.

As Kruse *et al.* (2009) note, '… trade-offs between the different pillars of sustainability must be addressed in the interpretation of the results. For example, how should machine labor, which results in varying levels of carbon dioxide emissions contributing to global warming, be handled in comparison to manual labor, with varying levels of working conditions?'

While the methods for assessing social impacts are still in their nascent phases, both the need for and potential benefits of being able to assess the social impacts of food production systems using a methodology that already lends itself to assessing environmental impacts suggest that this is an area of research that should continue to be developed and applied.

12.6 Sources of further information and advice

In an effort to facilitate effective practice of life-cycle thinking, the United Nations Environment Programme (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC) partnered to create the Life Cycle Initiative. One work area interest group under the Life Cycle Initiative focuses on Life Cycle Approaches Methodology, which includes a project on Social Life Cycle Assessment. According to their website, the goals of this project are:

- Goal 1: Provide the Life Cycle Community with a list of indicators for a social LCA.
- Goal 2: Disseminate the practicalities on how to develop a social LCA.
- Goal 3: Provide the Life Cycle Community with results of case studies of a social LCA.
- Goal 4: Extend the methodology to a triple bottom line tool.

More information on the status of the working group can be found on their website: http://fr1.estis.net/builder/includes/page.asp?site=lcinit&page_id=D45A9A8F-10FA-4501-BF93-FB9AEAE4D163

The *Human Rights Compliance Assessment (HRCA) 'Quick check'* (2004), published by the Danish Institute for Human Rights, describes a number of possible indicators related to human rights. This publication can be downloaded at http://www.humanrightsbusiness.org/pdf_files/Quick%20Check%20English%20.pdf.

In addition, the *International Journal of Life Cycle Assessment* has published a number of journal articles over the last several years, many of them already referenced in this chapter.

Other websites of interest:

- Australian life cycle assessment society – http://www.alcas.asn.au/events/roundtables
- Ecotrust salmon project – http://www.ecotrust.org/lca

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13

Ecodesign of food products

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Abstract: This chapter describes the characteristics of ecodesign, a tool for integration of environmental considerations in product development processes, and its relevance for the food sector. Similarities and differences between ecodesign and LCA are highlighted, and the chapter provides an example of how a specific ecodesign tool (based on eight design principles) can be applied to food products. Perspectives and future trends are addressed in the last section, followed by an overview of information sources that can be useful for companies that want to engage in ecodesign. The main target group of the chapter is the food industry, broadly speaking, but it can also be relevant for distributers, retailers, catering centres and private households.

Key words: ecodesign, product development, Life Cycle Assessment (LCA), environmental improvements.

13.1 Introduction

Food products are generally characterized by significant environmental burdens over their life cycle. Particular concerns are the contribution to global warming and land use from primary production, which reduces biodiversity and changes natural habitats such as the rainforests in the Amazonas and South East Asia. Food production also contributes, notably to a range of other environmental problems such as water depletion, nutrient enrichment, toxicity, animal welfare and occupational health and safety (Garnett, 2008; Steinfeld *et al.*, 2006; Rosegrant *et al.*, 2002; Angervall *et al.*, 2008; LRF, 2002; Thrane, 2004; ILO, nd). Life Cycle Assessment (LCA) studies show that for most food products, the majority of the environmental impacts occur in the first stage of the life cycle, such as agriculture or fishery (Angervall *et al.*, 2008). Food processing seldom represents the largest environmental burden, but many choices are made here that affect the environmental performance of other product stages. This could be choices in relation to product development/product design or choices about selection of raw materials, suppliers, logistics, consumer information, etc. Ecodesign is one way to ensure that considerations about the environment (or sustainability) becomes an integrated part of these decisions, and therefore it is an obvious approach to promote the development of more environmentally friendly (and sustainable) food products from a life-cycle perspective (Tischner *et al.*, 2000; Remmen and Münster, 2003).

13.2 What is ecodesign?

Ecodesign, sometimes referred to as Design for Environment, is defined by the International Organization of Standardization (ISO) as: 'the integration of environmental aspects into product design and development' (ISO/TR 14062, 2002). Hence, the purpose of ecodesign is to prevent or 'design out' adverse environmental impacts throughout the life cycle of products or service systems. In this chapter, we have chosen to follow the ISO definition and to focus on 'environmental' aspects, but there is obviously no reason to exclude broader aspects of sustainability (see also 'future trends' in the end of this chapter). Ecodesign takes a point of departure in the product development process (PDP), and it is obviously possible to implement ecodesign through stand-alone projects, where product developers work in isolation. It is more rewarding, however, to develop an ecodesign strategy and to engage more departments in the work. For example, the environmental department can help the product developers to focus on the important issues, to avoid sub-optimization, and to obtain documentation for the improvements. It is also highly relevant to involve the purchasing and marketing departments when dealing with supplier and customer issues. Hence, to use the full potential of ecodesign, it is necessary to have an interdisciplinary approach to product development. Product development can take place at all stage of the life cycle, but assuming that it mainly takes place at the processing stage, ecodesign can be illustrated as shown in Fig. 13.1.

Historically, ecodesign has been applied mainly by producers of electronic products, cars, buildings and materials. For food products, most examples deal with packaging – perhaps because packaging involves 'designers' while the food products often are developed by people with a background in, for example, chemistry or microbiology (Tischner, 2009). Another reason could be that the environmental importance of packaging simply has been overestimated compared to the food itself. The environmental debate within the food sector has, to a large extend, been dominated by a focus on single issues such as methane emissions from cows, nutrient enrichment caused by fertilizers, pesticides in the groundwater, and overexploitation of

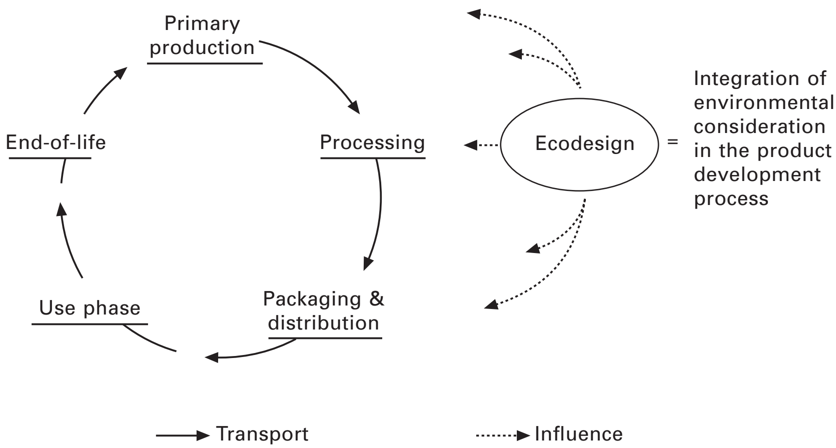

Fig. 13.1 Schematic overview of the life cycle of food products (left) and illustration of how ecodesign applied at the processing stage (right) can, and ideally should, influence all stages of the product life cycle.

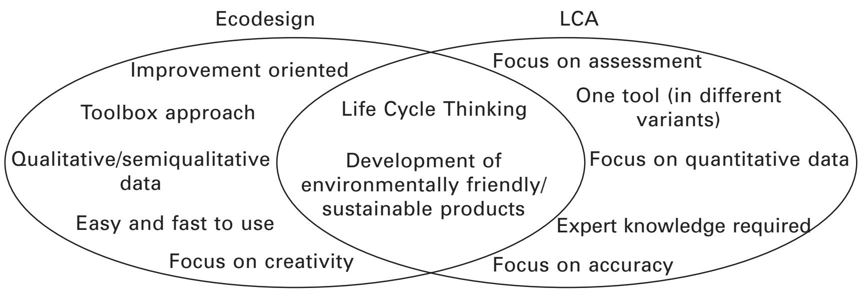

Fig. 13.2 Differences and similarities between the ecodesign and LCA approaches (Thrane and Eagan, 2007).

seafood resources. But ecodesign is one way of addressing environmental improvements in a more holistic and proactive way – by considering preventive solutions of the products' life cycle, already at the product development phase. It appears however, that an increasing interest in ecodesign for food products is emerging – and the subject was recently addressed by Baldwin and Wilberforce (2009).

13.2.1 Ecodesign versus LCA

Ecodesign and LCA have many similarities. Both tools are related to the ultimate goal of developing more environmentally friendly (or sustainable) products, and both tools are based on life-cycle thinking. Still, there are considerable differences – see Fig. 13.2. In contrast to LCA, which is primarily an 'assessment' tool, ecodesign is more focused on creative processes aimed at generating ideas for improvements. This does not mean that ecodesign ignores 'assessments' or that LCA ignores 'improvements'. It implies that ecodesign deals with improvements more directly and explicitly. Acknowledging that there are many variants of LCA (e.g. conceptual, screening and detailed LCA) and many complex modelling choices, ecodesign involve a larger set of different tools (a toolbox). The ecodesign toolbox includes LCA and other less complex assessment tools, as well as a range of improvement tools suited to different phases of the product development processes; see also Thrane and Eagan (2007). Qualitative or semi-quantitative data are widely applied in ecodesign, while most LCAs seek quantification, accuracy, completeness and consistency that require expert knowledge. Despite the differences, ecodesign and LCA both strive towards the same goal, namely more environmentally friendly (or sustainable) products from a life-cycle perspective. But the perspective is different. A product developer will typically think of the product life cycle as the phases from idea generation to detailed design, prototype, market launch and market life cycle – while the environmental expert focus on the stages from primary production, to processing, use and waste handling in the end-of-life stage; see Fig. 13.3.

One of the challenges of ecodesign is to make product developers aware of the LCA perspective, while making the environmental department aware

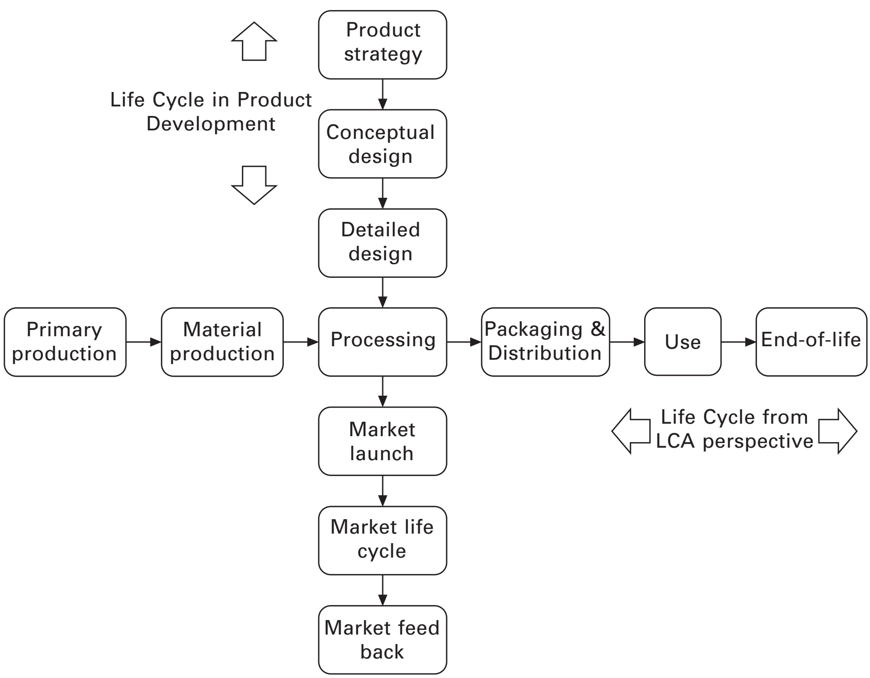

Fig. 13.3 Two perspectives on product life cycle. Vertical boxes illustrate the life cycle from a product development perspective and horizontal boxes are life cycle from LCA perspective.

of the life cycle in the product development processes. It is necessary to include both perspectives.

13.2.2 Relevance of ecodesign in the food sector

A food product such as yoghurt may not appear to be an evident target for ecodesign – yoghourt is yoghurt, and yoghurt is normally made from cow milk, end of story. But, just think about the various types of yoghurt that exist: there is organic versus conventional, yoghurt from cows that are grazing all year round versus cows that are kept in the stables (loose or fixed). There is yoghurt made from soy or different types of cows with different diets, and yoghurts with different taste, fat content, packaging type, shelf life, bacteria culture, etc. Hence, there are many choices in the agriculture stage, during the production at the dairy and for packaging, transport, etc. The literature provide numerous examples of food packaging based on recycled materials, or made of materials that can be reused, recycled or even consumed. Admittedly, improvements of packaging can deliver some benefits and may represent an important signal to consumers. But it is typically the food product inside the packaging that represents the largest impact potential (Angervall *et al.*, 2008). Food and packaging should therefore be analysed and optimised as an integrated unit, as part of ecodesign in the food sector.

13.3 The spiderweb approach

There exist a significant number of tools that are directly (or indirectly) associated with ecodesign. Tischner *et al.* (2000) refers to four categories:

(i) Tools for improvement
(ii) Tools for assessment
(iii) Tools for prioritisation
(iv) Tools for meeting other criteria related to costs, product quality, etc.

This chapter will only focus on the first two – because they are most directly concerned with the essentials of ecodesign, while the two latter are applied in all types of product design processes. One of the tools that can be applied in several phases of the PDP and which can be used for both improvements and assessments (at least to some extent), is the Lifecycle Design Strategies (LiDS) wheel, developed as part of the Dutch Promise Manual in the early 1990s (Brezet *et al.*, 1994). The LiDS wheel is a spiderweb diagram with eight axes (representing eight environmental principles). The first environmental principle (Principle zero) mentions the importance of rethinking the product and its function/-s. This is followed by seven principles that address the life-cycle stages, from raw materials to the use and end-of-life stage. The scale on each axis goes from no compliance (in the centre) to high compliance at the periphery. Hence, the wheel can also be used for simple assessments and comparisons of new and old product designs, but it should be acknowledged that the graphical representation in a spiderweb diagram is distorting because the diagram is not linear and because the visual importance of a score (on one axis) depends on the score on the previous axis. Most existing versions of the wheel, address products more generally, and recommend design choices such as a modular structure, design for disassembly, or timeless design. Such guidelines makes little or no sense in relation to food products, and the LiDS wheel has therefore been modified to complement food products in this chapter; see Fig. 13.4. As it appears in Fig. 13.4, each axis has a heading (bold) which represents a specific ecodesign principle and a list of specific recommendations (referred to as ecodesign criteria). The criteria are meant only for inspiration, and more product- or company-specific versions should ideally be developed by the user.

13.3.1 Rethink the product and its functions (Principle 0)

The first and most important environmental principle is to rethink the product and identify the services it provides. The function of food is to deliver energy and nutrition, but it is also a source of enjoyment, it should be tasteful, etc. Tara Garnett expresses it this way:

> 'Food is a basic physiological need but it is also one of the glues that bind families together. At the highest 'self actualisation' level, food and drink are bound up in the rituals and traditions of the world's major religions' (Garnett, 2008).

As an example, the function of food at a hospital is (or should be) to make patients healthy and reduce their recovery time. Here, it should ideally be sought to reduce the environmental impacts per function rather than per kg of food purchased. In this perspective it would be an environmental improvement to serve healthier food – even if the environmental footprint remains unchanged. The challenge could also be to develop food product service systems (PSS), which reduces environmental impacts on a system level; see also Manzini and Vezzoli (2002).

The focus on function and service requires the eco-designer to think 'out of the box' and sparks creativity. An example of a Danish food company that has been successful in thinking along these lines is Aarstiderne who produce organic vegetables, but also has developed a concept whereby the products are delivered directly to the consumers in boxes, with a selection of vegetables and fruit of the season (hence reducing the need for heated greenhouses and/or long transports), together with recipes for inspiration (which may increase the function while reducing the food waste). Recently, the product range has been expanded to include fish, meat, bread, wine and complete meal packages. Furthermore, the company engages in educational activities for children about sustainable food production (Aarstiderne, 2008).

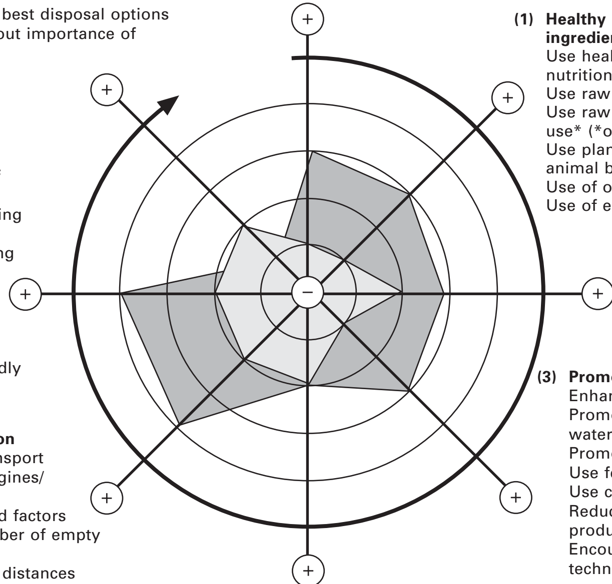

Fig. 13.4 Modified version of the LiDS Wheel (referred to as the 'spiderweb tool' in this chapter) relevant for ecodesign of food products, inspired by Brezet *et al*. (1994) and Remmen and Münster (2003).

13.3.2 Healthy and environmentally friendly raw materials and ingredients (Principle 1)

There are numerous environmental impacts to consider for raw materials and ingredients besides GHG emissions, but the carbon footprint is also often a good proxy for other impacts (Weidema *et al*., 2008). A general rule of thumb is therefore to use raw materials and ingredients with a low carbon footprint. It usually reduces GHG emissions to substitute vegetables for meat-based (or animal based) raw materials and ingredients, or to substitute chicken, pork or fish, for meat with a high carbon footprint such as beef. This could be an option for a producer of sausages, meatballs, or ready-made dinners. There is obviously also a quality or function aspect to consider (all consumers may not appreciate sausages with a high content of potato starch), but it is beyond the scope of this chapter to discuss this further.

It is also worth mentioning that some vegetables, such as cucumbers and tomatoes, can have a significant carbon footprint (nearly on the same level as meat), especially if they are produced outside the season in fossil-fuel heated greenhouses. Field-grown vegetables and root fruits generally have a low carbon footprint (Dall and Toft, 1996). To give an impression of the differences, Swedish studies have shown that GHG emissions are similar for: 1 kg of greenhouse tomatoes from Sweden (based on fossil fuel), 4 kg of field grown tomatoes from Spain (including transport to Scandinavia), and 40 kg of carrots from Sweden (Angervall *et al*., 2008). This is a difference of a factor of forty and this underpins the importance of distinguishing between vegetables, and to consider seasonal changes.

Another important option is make use of otherwise wasted raw materials and ingredients. In a seafood context, this could be to use fish or shell-fish that are otherwise discarded (thrown overboard) because they have too small an economical value, or to use ‘industrial fish’ for human food instead of reducing it to animal food. Spillage of food in the industry is, to a large extent, reused as raw material within the production or as animal feed. Nevertheless, there are still improvement potentials and situations where food ‘waste’ can be either further reduced or used for better purposes (i.e. better from an environmental and/or economical point of view). Upgrading of food waste to new products is investigated in the EU project REPRO. One of the studies in the project was to upgrade red cabbage trimmings to novel products (Krewer, 2008). The use of enzymes, which is discussed in relation to environmental Principle 3, is playing an important role in this regard. Generally, it is a question of selecting healthy and environmentally-friendly materials and ingredients and to develop demands to suppliers based on such considerations. This could also be demands to environmentally certified raw materials, to the extent they exist. Or to use suppliers that are certified according to an environmental management standard, such as ISO 14001 or EMAS. As shown by this book, there exist many reports and studies about environmental impacts of raw materials for food products. More and more information is also becoming available in LCA databases and on homepages such as: http://www.fcrn.org.uk, http://www.lcafood.dk and www.sik.se.

Finally it is worth stressing that the selection of raw materials and ingredients should also be made with concern for the shelf-life of the products, to reduce food waste in the following stages of the life cycle, not least the use stage.

13.3.3 Efficient use of raw materials and ingredients (Principle 2)

It is essential to use raw materials as efficiently as possible – especially in the food sector. Large amounts of materials are being lost in the food supply chain. As an example, fruit and vegetables representing a value of around 8 million Euro, are destroyed each year in transport from producer to retail in Europe. And in Africa it is believed that less than 50% of the produced food products reach the consumers (Jönson, 2008). Smaller, but still significant, losses occur in the processing stage. In Berlin and Sonesson (2008), the environmental improvements made by sequencing cultured dairy products were analysed. It was concluded that just by optimising the sequencing, as much as 29% or possibly up to half of the product waste could be avoided. Similar studies have been performed on other types of food production, e.g. sausage and juice (e.g. Ingvarsson and Johansson, 2006; Johansson *et al.*, 2008), also showing significant improvement potentials. Hence, there are possibilities for industries to reduce their waste, and thereby reduce the use of raw materials, resulting in a lower environmental burden.

Concerning the use stage, a recent UK study estimates that households alone (use stage) waste 1/3 of all the food that is purchased. The study suggests that more than 60% of this is avoidable food waste if managed better (Ventour, 2008). Reductions of food waste in the last stages of the life cycle can have a considerable effect, because less food needs to be produced, processed, distributed and transported per unit of function. Product developers can address this through the selection of raw materials and ingredients (see Principle 1), and also through changes in packaging and consumer information, which is discussed in relation to Principle 4.

13.3.4 Promotion of cleaner production processes (Principle 3)

Promotion of cleaner production practices during food processing (and during primary production) also needs to be addressed as part of ecodesign. Cleaner production aims at reducing amounts of materials, water, energy, and chemicals used at the processing stage, while reducing the emissions to air, water and soil. Generally, cleaner production may include:

- Better housekeeping practices
- Reuse and recycling (of raw materials, auxiliaries, water and energy)
- Substitution of materials, chemicals or energy sources

- Process optimization
- Technological change and innovation.

Good housekeeping practices often have a short pay-back time, as they can be implemented with little or no investments. In the food industry, special attention should be paid to cleaning practices, which consume significant amounts of water and energy, while producing wastewater. Studies from the fish processing industry in Denmark show that simple registration of water consumption and better planned cleaning procedures can have a large effect in terms of reduced water and energy use, and also generate less food waste due to the reduced contact between fish and water (Thrane, *et al.*, 2009). Reuse and recycling will not be further elaborated here, but numerous examples exist in the literature; see e.g. Thrane, *et al.* (2009). Substitution of materials and ingredients was addressed in relation to environmental Principle 1, but for cleaner production it is also worth considering the substitution of energy sources. Carbon footprint can be reduced considerably by switching from oil to gas or renewable energy sources. This is relatively simple for heat energy. It is more complex when it comes to electricity. Some companies address this by purchasing 'green' electricity through special contracts with electricity suppliers, or by other initiatives such as investments in windmills, carbon credits, etc. It should be stressed, however, that the current market for 'green' electricity is characterized by being relatively new and unregulated. A number of false claims are therefore being made. As an example, it is not especially green to purchase hydropower-generated electricity from a country where all the available hydropower resources already are fully utilized.

Concerning process optimization, there are many options for improvement, depending on the company. Examples could be switching from batch operation to continuous operation, to install more modern production equipment, and better production planning (as explained in the example about sequencing in relation to Principle 2). More innovative solutions may imply the use of cutting edge technologies and catalysts such as enzymes provided by greentech companies such as Danisco or Novozymes. LCA studies reveal that the food industry can save significant amounts of energy, materials and chemicals, and even increase yields, by using enzymes as a catalyst in food processing or for cleaning processes. Enzymes make it possible for chemical processes to take place at lower temperatures, while reducing the need for chemicals. Studies shows that the customers reduce the CO_2e emission with 100 kg CO_2e for each kg of enzymes they use (Nielsen *et al.*, 2007; Nielsen and Hoier, 2008).

The processing stage may not represent the largest environmental burden from a life-cycle perspective, but there are also improvement potentials here. More importantly, reductions of food waste at the processing stage will influence all upstream processes and thereby have a great improvement potential from a life-cycle perspective. The United Nations Environmental Programme (UNEP) has developed tools and guidelines for cleaner production for the processing of fruit and vegetables, dairies, meat, seafood, beer and wine, bread, sugar, vegetable oil, etc. Information sources/links are provided at the end of this chapter.

13.3.5 Optimisation of packaging (Principle 4)

Packaging seen in isolation does not typically present significant environmental impacts (compared to the food it protects), but is relevant to address by ecodesign because:

- Product developers have a significant influence on the type of packaging
- Some types of packaging has a substantial impact potential, e.g. aluminium and glass
- Packaging has a large influence on the amount of food waste.

At first, it seems evident that less packaging, and packaging made of eco-friendly materials, is always the better choice; but it is not that simple. Packaging plays an important role to reduce food spillage in the entire product chain. LCA studies have shown that small reductions of food spillage can justify significant amounts of packaging (Thiesen *et al*., 2007; Løkke and Thrane, 2008; Weidema *et al*., 2008). Thiesen *et al*. (2007) include a comparative LCA of two Danish yellow cheeses (250 gram each) in different kinds of packaging. The two cheeses fulfil the same function regarding taste and quantity, but one cheese is a 'convenience product' which comes in 'dish cover' packaging (or cheese dish) that is easy to open and close (ideal for storing). The other comes in a conventional flow-pack. There is significantly more packaging involved in the convenience product, but the cheese rind is removed at the dairy (44 g out of 294 g) where it is reused (for processed cheese). Besides, no additional packaging is needed at the use stage. The study therefore concludes that the convenience cheese represents a smaller contribution to global warming (Thiesen *et al*., 2007; Andersen *et al*., 2005). This example illustrates that less packaging is not always better, and that food and packaging should be analysed together, not separately.

According to Ventour (2008), limited durability is one of the key drivers of food spillage at the consumer stage in the UK. Packaging is therefore necessary in most cases – and the packaging should ideally ensure a high degree of protection of the product's quality while increasing the shelf life as much as possible. As illustrated by the cheese example, packaging should also ensure a minimum of product waste in the use phase. Convenient packaging and packaging that is easy to empty are ways to obtain this. We have all experienced difficulties emptying a yoghurt container or a bottle of ketchup. This is annoying, but also means that significant amounts of additional food must be produced. In Berlin *et al*. (2008), the loss of yoghurt at the consumer stage is analysed. The yoghurt left in the package after emptying ranged from 3.4% to 8.5%, suggesting a significant improvement potential. The size and shape of the packaging should also match the consumer needs and allow for efficient transport and storing. There are many aspects to consider, and the environmentally ideal packaging will:

- Provide a high degree of protection to the product during transport
- Give the product a long shelf life/durability
- Be convenient to use and easy to empty
- Be possible to compress after use (takes up less space in the waste bin)
- Be made of environmentally friendly (and healthy) materials
- Be reusable or recyclable
- Represent a low material usage
- Have a shape that is efficient for transport and storing
- Offer a size that matches the consumer needs.

Finally, it is also an option to use the packaging to convey messages to the consumer about environmental aspects and eco-friendly use of the product, but this is elaborated in relation to environmental Principle 6.

13.3.6 Effective distribution system (Principle 5)

Although the environmental burden of food transport is sometimes overstated, transport can be important. The most important aspect is not necessarily the distance – but is more likely to be the mode of transport. The GHG emissions for moving one tonne one kilometre are about:

- Ten times smaller for a transoceanic freight ship compared to a large truck (>32 tonne),
- Ten times smaller for a large truck compared to a small truck or delivery van (>3.5 tonne),
- And ten times smaller for a delivery van or airfreight compared to a car if it is assumed that the car transports 20 kg of groceries at a time (Ecoinvent, 2007).

The difference between the freight ship and the car (that transports 20 kg groceries) is roughly a factor of 1000. This is the reason why shopping by car has a significant environmental burden despite the limited food mileage involved. Efficient delivery services could be one way to address this. Airfreight is on the same level as a small delivery van, and is typically used to transport food products with a low shelf life and/or high value, such as fresh fruit, vegetables, and seafood. For transport it can be recommended to:

- Use efficient and low emission modes of transport
- Plan logistics to promote high load factors (and reduce empty returns)
- Plan logistics to avoid small trucks and vans (opposed to large trucks, trains or ship)
- Plan logistics to reduce transport distances
- Encourage eco-driving.

There can obviously be tradeoffs, and slow transport can, in some cases, increase product loss due to the limited product durability. And the use of large (instead of small) trucks can result in lower load factors. So the challenge is to obtain both at the same time; for example, by better transport planning and by coordinating transports with other companies if possible.

13.3.7 Environmentally sound use of product (Principle 6)

Environmental friendly food products should be designed to reduce impacts in the use stage as well. The following recommendations mainly address private households, but they are also, to some extent, applicable to other users such as restaurants and catering centres. Several of the criteria that have been mentioned for other environmental principles (Principles 0 to 5) address the use stage indirectly, e.g. packaging that reduces waste through correct size, convenience and increased durability. But the use stage can also be addressed directly by environmental information – through messages and dialogue with consumers. Homepages, commercials, and information on packaging can be used for this purpose. Environmental messages could attend to:

- The importance of reducing food waste
- Correct disposal of the packaging
- Eco-friendly food preparation/cooking methods
- Eco-friendly food recipes
- Importance of avoiding shopping by car if possible
- How to assess if the product is too old (or just older than the suggest 'best before' date)
- Information about environmental improvements made by the producer – why they are made, and why the product or the packaging maybe has changed as a consequence (e.g. thinner slices of meat of cheese to reduce consumption)
- Environmental issues more generally, and how the consumer can make a difference.

Regarding the food product itself, it is also possible to develop food products that reduce the need for cold storing or heating during food preparation, such as parboiled rice. A more futuristic solution is to use indicators that display whether or not the product is too old, thus avoiding a waste of product just because the 'best before' date has been passed.

13.3.8 Optimisation of end-of-life stage (Principle 7)

The end-of-life stage concerns the waste and the waste handling. Product developers can influence this stage by design choices that reduce product waste, or labelling that informs about correct waste disposal, as described earlier. A simple illustration is that of a milk carton which can be folded together after use; one way of reducing the environmental impacts from transport at the end-of-life stage. Other options are to use packaging that is easy to re-use or recycle, or where the energy content can be recovered in the waste incineration with least possible emissions. Incentives for recycling can be provided by putting a deposit on the bottle or can. There exist different waste handling systems in different countries and regions, and producers should ideally take this into consideration as well. Apart from an efficient reuse system for beer bottles and cans, most of the household waste in Denmark is burned in incinerators where the energy is recovered and used to produce electricity as well as heat. The incineration process is highly efficient and filters make sure that the emissions of dioxin are insignificant. In third world countries, waste handling systems are sometimes non-existent and the waste is either burned in the streets, or at local dumpsites, or deposited at uncontrolled landfills. Hence, ecodesign principles should ideally take these site-specific aspects into account.

13.4 Perspectives

The previous section has focused on the spiderweb approach, but there are other tools as well. The following will include some examples of other tools, a few words about prioritization and tradeoffs, as well as an outlook that addresses the future of ecodesign.

13.4.1 Other ecodesign tools

Apart from the spiderweb tool, which can be used for both improvements and assessments (i.e. visual representation of the performance of one product compared to another product), there is a range of other tools in the 'ecodesign toolbox'. This includes tools that address improvements, assessments or both – and which are suited for different phases of the PDP, from product strategy, to conceptual design, and detailed design. An excellent overview is provided by Tischner *et al.* (2000). Generally, it advised to use simple tools in the beginning of the PDP, because little knowledge exists about the final product anyway. However, it is also here (in the beginning) that the largest changes can be made, because the scope of design alternatives is largest; see Fig. 13.5.

An example of simple rules of thumb, that are highly relevant at the beginning of the PDP, is the 5xR strategy:

- Rethink (e.g. the product and its functions)
- Reduce (e.g. the use of energy and materials)
- Replace (e.g. hazardous chemicals and energy intensive materials)
- Reuse (e.g. the product)
- Recycle and recover (e.g. materials and energy content).

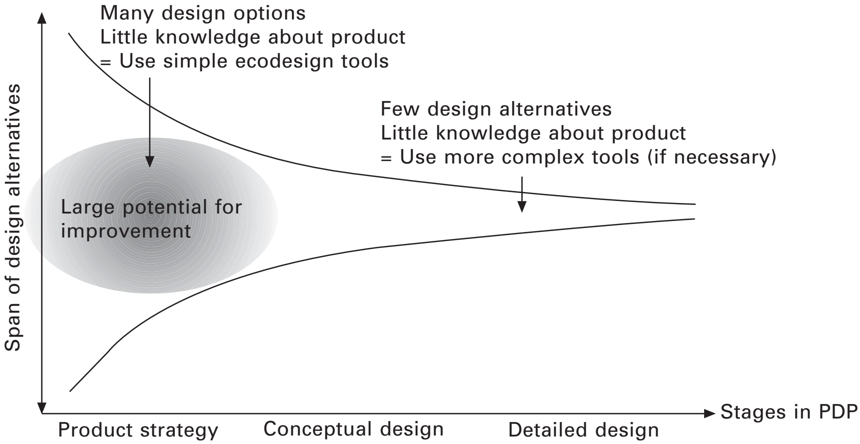

Fig. 13.5 The range of design alternatives in different stages of the product development process (PDP), inspired by Behrendt *et al*. (1997).

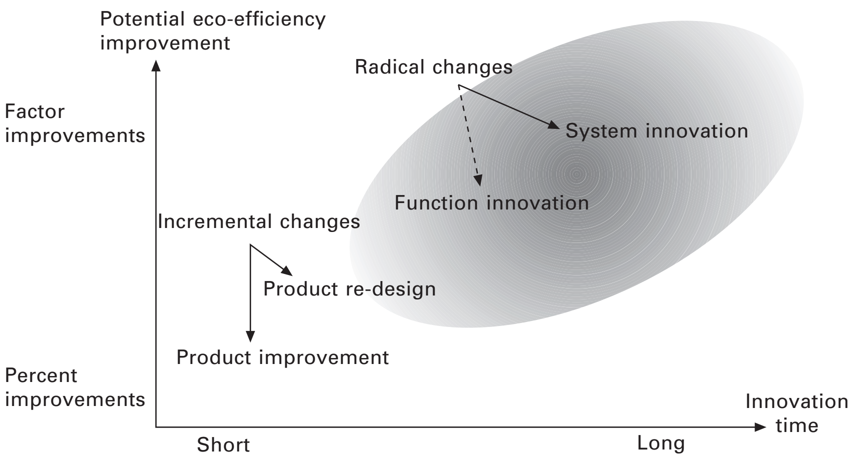

Fig. 13.6 Indicative relationship between eco-efficiency improvement and types of product development as a function of time, inspired by Tischner *et al*. (2000).

The first rule 'Rethink' is particularly important as it has the largest potential for generating radically new ideas – similar to environmental Principle 0 in the spiderweb model related to the product and its functions. But rethinking is important in all areas. One of the more experienced consultants within ecodesign in Europe explicitly stresses the importance of 'stupid' questions in their ecodesign guidelines (PRé Consultants, 2009). Radical changes have the largest potential for improving the eco-efficiency (environmental impact per function), as opposed to small product improvements (or re-design) where the basic technology remains unchanged; see Fig. 13.6. Radical changes could be *in vitro* meat (laboratory-grown meat or cultured meat), nano food, food service systems, or system innovation where changes occur at the society level (e.g. changes to 100% organic agriculture in an entire country). It is important to stress, however, that function innovation and system innovation do not necessarily lead to significant environmental improvements.

Tischner *et al.* (2000) mention another tool called 'Morphological boxes', where a design problem is broken down into a number of parameters that can be varied individually. The parameters for a drinking container could be form/shape, materials, size, handling, surface, colour, aesthetics, and durability. Each parameter can then have a number of variations. For example, the parameter 'form' could be cylindrical, conical, rectangular, triangular, spherical or bowl-like. Tischner *et al.* (2000) also describe other simple improvement tools such as 'brain storming' and 'brain writing'. More complex tools include 'Bionic', which is design that is inspired by nature – typically animals and plants. Results of such an approach could be enzymes (as explained in relation to Principle 3) or different types of genetically modified food.

Tools that are aimed more at environmental assessment could be the product summary matrix, which is a 5×5 matrix where life-cycle stages are arranged in the rows while environmental concerns are represented by different columns; see Table 13.1. To estimate the total environmental impact, a one integer number (between zero and four) is assigned to each of the matrix elements. Zero represents the highest and four the lowest estimated impact. These cell scores are generated by answering questions relevant to the cell in the matrix. The overall impact (product rating) is calculated as the sum for the matrix element values, where the maximum is 100 points, representing the absolutely best alternative. The method is relevant for comparative assessments, where a new product is compared to a reference, but it can also be used for a single product assessment and comparisons between different life-cycle stages (hot-spot assessment). The product summary matrix is a simple tool that is mainly relevant at the beginning of the PDP, but more advanced tools, relevant for later stages, include screening LCA, and detailed

Table 13.1 Product summary matrix (Source: Graedel and Allenby, 1995)

Life cycle stage	Environmental concerns					
	Materials	Energy use	Solid waste	Liquid waste	Air emissions	Total
Pre-manufacturing						
Manufacturing						
Packaging and transport						
Product use						
Refurbishment, recycling, disposal						
Total						Max 100 points

LCA. A large number of other assessment tools are described in Tischner *et al*. (2000). What is more, a large number of simple calculators for Carbon Footprint are becoming available, even online. Most of them are not accurate or useful for documentation purposes, but for companies who do not have the resources to engage in detailed LCA, the more serious versions of these calculators can be used to get an initial estimate at the beginning of the PDP.

13.4.2 Prioritisation and trade-offs

The spiderweb tool mentioned a large number of improvement options that can be addressed by the designer or in collaboration with the environmental and other departments. But how is it possible to prioritize and what should be the focus? This is obviously entirely up to each single company, but a simple rule of thumb suggests focusing on what is important in terms of environmental impacts and which are easy to influence (practically, economically, etc.) at the same time. This is illustrated by the upper left square in Fig. 13.7.

Tradeoffs (or problem shifting) are another challenge that makes prioritisation difficult in some cases. Tradeoffs can be situations where environmental improvement in one area increases the environmental impacts in another area, or have a negative impact on other product performance parameters such as price, quality, taste, smell, or appearance. LCA can be helpful to avoid environmental tradeoffs, but still has limitations concerning other aspects. There exist tools that can assist in prioritisation between all types of design criteria, such as the 'house of quality'. Examples are provided in Tischner *et al*. (2000). However, it is always up to the decision-maker in the end.

13.5 Future trends

Ecodesign focuses on the environmental aspects, but the social dimensions have received increased attention during the last decade, thus pushing design

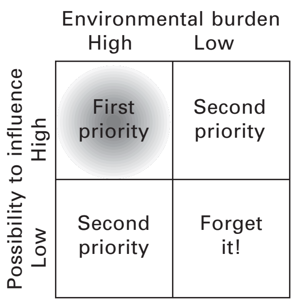

Fig. 13.7 Decision matrix for ecodesign, inspired by Remmen and Münster (2003).

for 'sustainability' and Corporate Social Responsibility (CSR). Design for Sustainability (or D4S) has not been the focus of the present chapter, but as it appears from the list of information sources provided at the end of the chapter, it is likely to take over and become the new 'buzz' word instead of ecodesign (UNEP, 2006, 2007). There is not much difference in approach, as such, but while ecodesign mainly address the environmental aspects, D4S also encompass social aspects and occupational health and safety in the entire product chain. Hence, the future scope of both LCA and ecodesign will be expanded not only to cover the entire life cycle of products but also all three pillars of sustainability (environmental, social and economical aspects). Ideally this will contribute to avoid burden shifting, not only between life-cycle stages, but also between, for example, environmental and social problems. However, it will also expose dilemmas in some cases, such as the choice between avoiding airfreight of food products from third world countries (which has a positive environmental effect) and continuing the air freight which probably has positive social and economical impacts on the exporting country. In the present chapter, it has been emphasized that ecodesign is about generating improvements and new ideas – often by the use of qualitative tools and approaches. This may appear to be decoupled from detailed 'assessment' tools such as LCA, but the tools should be seen as complementary. Assessments are necessary to guide decisions and ecodesign is necessary to spark creativity for innovation.

13.6 Sources of further information and advice

There are few information sources that specifically address ecodesign in relation to food. But there are many relevant information sources regarding ecodesign (and design for sustainability) and how it can be applied more generally. Also, there are relevant information sources that address cleaner production within the food sector that are relevant to ecodesign of food products.

Literature sources about ecodesign:

- http://www.d4s-de.org/manual/d4stotalmanual.pdf (A well illustrated guide to ecodesign, but with no particular focus on food)
- http://www.unep.fr/shared/publications/cdrom/DTIx0889xPA/ (CD rom about environmental management and ecodesign from United Nations Environmental Programme, available in many languages)
- http://www.o2.org/index.php (A list of relevant literature sources)
- http://vbn.aau.dk/fbspretrieve/115301/abstractfil.pdf (A good introduction to life cycle thinking and management from the Danish EPA, which also addresses ecodesign)
- http://www.unep.fr/scp/design/pdf/pss-imp-7.pdf (A short introduction to Product Service Systems (PSS), with good examples from the food sector as well)

Homepages about ecodesign:

- http://www.unep.fr/scp/design/
- http://www.econcept.org/
- http://www.unep.fr/scp/design/pss.htm
- http://ecoinnovationlab.com/
- http://www.ce.cmu.edu/GreenDesign/
- http://www.cfd.rmit.edu.au/
- http://www.cfsd.org.uk/
- http://www.pre.nl/ecodesign/default.htm
- www.O2.org

Homepages about cleaner production in the food sector:

- http://www.unep.fr/scp/cp/
- http://www.cleanerproduction.com/Directory/sectors/subsectors/FoodProc.html

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14

Footprinting methods for assessment of the environmental impacts of food production and processing

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Abstract: This chapter introduces a number of different, but related accounting methods that can be used to assess the environmental impact of food production and processing; the methods are grouped under the banner of footprinting. Footprinting has only a short history, but from the original introduction of ecological footprinting in 1995, the methodology has expanded to include water footprinting, carbon footprinting and latterly environmental and nitrogen footprinting. The various methods share a common approach but include many different indicators to illustrate the environmental impact of production. Although not necessarily compatible they are complimentary. This chapter examines each method and uses examples to show how they can be used within the food sector.

Key words: footprint, ecological, water, greenhouse gas, carbon, environment, nitrogen.

14.1 Introduction

This chapter introduces a number of different, but related, accounting methods that can be used to assess the environmental impact of food production and processing. The methods are grouped under the banner of footprinting and are related to Life Cycle Assessment, which is the subject of other chapters in this book. This chapter introduces the concept of footprinting, explains the principles that support its development and use, demonstrate how footprints are interpreted and considers what the future holds for this type of analysis. It starts with a description of ecological footprinting before moving on to discuss water and carbon footprinting and concludes by examining the recent developments of environmental and nitrogen footprinting.

Food security and sustainable agricultural production have become increasingly important topics within the last two years. With the global population forecast to grow to nine billion by 2050 and an increasing demand for foods requiring high-energy for their production (e.g. beef), feeding the world has never been such an important issue as it is today. Add in concerns over the impact of climate change and you have a situation where policymakers are thinking seriously about food supply and whether the projected increased demand can be met without degrading our agricultural lands to a point where they become unproductive. One approach would be to make agriculture truly sustainable, to the extent that the inputs required for production are minimised as far as possible and do not draw down the earth's capital but leave resources for future generations to use as well.

What is sustainable agriculture? There are, of course, many answers to that question depending on the region of the world, the availability of resources such as seed, water and fertiliser, and the market demand for food. A full answer is beyond the scope of this chapter; however, whatever the system, one thing is clear: to assess whether a system is sustainable or not, a baseline must be established, against which changes can be monitored over a given time period. This chapter addresses that situation by discussing how production systems can be assessed and quantified and the results used to assess whether one system is more sustainable than another.

The term 'environmental accounting' is used in this chapter to describe, in any of their many guises, the methods currently available for assessing the environmental impact of producing agricultural commodities. There are multiple methods, both in use and under development, that use different inputs and that report their results in different ways. Life Cycle Assessment has been used in commercial manufacturing for a number of years but the concept is not well recognised by the general public – whilst footprinting, although a relative newcomer, has quickly gained acceptance in some parts of the world, even if it is not understood, by the general readership.

14.2 Footprinting, comparison and contrast to Life Cycle Assessment (LCA)

Life Cycle Assessment (LCA) is an established scientific approach (ISO 14040: 2006a), which identifies all the different life cycle stages that are required to manufacture a product, produces an inventory of all the inputs, outputs and wastes that are part of that life cycle, and then quantifies them. This is often referred to as the 'cradle-to-grave' approach or sometimes 'seed to shelf' or 'farm to fork' when the product is food. The inventory approach, which accounts for every stage within a production process, will result in a very detailed analysis and has been used within manufacturing industry for a number of years.

However, the complex and variable nature of primary food production can result in the inventory holding large and discrete amounts of data which can subsequently make distilling the output into a manageable form a difficult task. LCAs are an excellent tool for investigating a production process or for comparing products that use the same functional unit, but they can be unwieldy when comparing different products and the outputs can sometimes be difficult to put into the context of the overall analysis, making interpretation difficult. An LCA inventory normally contains other, potentially useful, environmental information; for example, eutrophication potential, acidification potential, pesticide use and water use. This information is often presented in parallel with the main process results but it is sometimes difficult to link it directly and to show the relationship that exists between inputs and impact.

Footprinting can use the same boundary and inventory approach as LCA but presents the data in a different way. Data is transformed to use the same reporting unit and the results are aggregated into a single value. The appeal of footprinting lies in its ability to convey quite large amounts of (aggregated) information is an easy to understand manner. Although the use of ecological footprinting has remained restricted to academic and policy areas, water footprinting has become more accepted, and carbon footprinting has found applications over wide and diverse areas. Carbon is currently the main focus of footprinting techniques; the method has the ability to be used for multiple purposes and as a method of aggregation it has few peers. However, as long as they share a common dominator, there are multiple indicators available that can be incorporated into a footprint; this is demonstrated using an environmental footprint.

14.3 An introduction to ecological footprinting

Modern environmental accounting can be said to have started with the publication of *Our Ecological Footprint* (Wackernagel and Rees, 1996) in which the authors introduced the concept of the ‘ecological footprint’. They state that ‘ecological footprint analysis is an accounting tool that enables us to estimate the resource consumption and waste assimilation requirements of a defined population or economy in terms of a corresponding productive land area’. The ecological footprint calculates the bioproductive area required to supply the resources needed to supply any given consumption. The resources assessed by ecological footprinting come in three main forms:

- Bioproductive area (the land required to grow the commodity);
- Material flows (the physical resources required to grow the commodity, e.g. infrastructure (buildings and concrete), machinery, seed, fertilizers, pesticides and packaging;
- Energy flows (the energy needed to grow the commodity – direct, e.g. diesel; indirect, e.g. embodied energy of fertilizer).

The land area required to support these resources is calculated using a number of different criteria. Direct and indirect energy are assessed using their CO_2 emissions as these can be directly linked to biophysical area (e.g. land area required by green plants to sequester the CO_2). To relate CO_2 to an area it is assumed that newly planted forest area is used to absorb the CO_2 emissions. The world average CO_2 absorption rate per hectare of world average forest is used and converted to global hectares (gha). Alternative approaches have been tried, but this has proved to be the most conservative approach. It is important to note that the ecological footprint excludes a number of inputs, which include: the share of CO_2 absorbed through the oceans, other emissions that contain carbon, such as methane (CH_4), other greenhouse gases like nitrous oxide (N_2O), and water.

One of the criticisms levelled against ecological footprinting is the complexity and lack of transparency in the conversion of resources to bioproductive area. Different conversions factors are required to convert the many different types of consumption (energy, built environment, raw materials, waste) to a land area. This complexity is confounded by the many different practitioners involved in the discipline; one result is that it is difficult to compare the results of different analyses. Despite these difficulties, ecological footprinting has been used to explore environmental impact and sustainable development at different scales ranging from individual crops to whole countries.

Ecological footprinting methodology, support and promotion are provided by the Global Footprint Network which was established in 2003.

14.3.1 The use of ecological footprinting at country level

Where the method has found approval is in assessing the ecological impact of cities, regions and countries. This is well illustrated in WWF's annual review of global ecological status; the Living Planet Reports (WWF, 2008) which rank the ecological footprints of individual nations. This series of reports calculates the total productive area of the earth, its cropland, grazing land, forest, and fishing grounds, and divides it by the global population to allocate each individual, his or her share of the world's resources. In 2005, the total supply of productive area, the biocapacity, was assessed at 13.6 billion gha, or 2.1 gha per person. Of course, not everyone gets their fair share of global resources. In 2005, the top two countries as far as resource use was concerned were the United Arab Emirates and the United States of America, each of whose citizens appropriated over 9 gha per person, while the citizens of Afghanistan and Malawi made do with under 0.5 gha per person.

The philosophy underpinning this approach is that every citizen should have access to an equal share of the earth's biocapacity in order to support themselves; however, biocapacity is a world average value which does not distinguish between productive areas and less productive areas. For example, while both the UAE and the USA have equally high resource use, their situations are very different. Citizens of the UAE occupy a small and resource-poor (with the exception of oil) land area while citizens of the USA occupy a large, resource-rich land area. Although both countries have similar ecological footprints, the USA is the more sustainable system.

The fact that that a small, resource-poor country has the same ecological footprint as a large resource-rich one suggests that other factors are more important in determining footprint size – WWF's ranking of all countries by footprint size reveals that it is a country's economic power that allows it to use more than its 'fair share' of global resources. This 'buying power' can be translated into a 'guilt factor', to be used to persuade rich countries to reduce their resource use and share of global resources. This simplistic approach has been widely accepted as a term of impact by the general public. Although 'the footprint' approach used by the WWF is not transparent and its structure and calculation are complicated and hidden, it has successfully introduced the concept of 'footprint' and 'impact' to the general population, and it is the forerunner of all other footprinting methods. However, despite the best efforts of the Global Footprint Network to promote it, use of ecological footprinting has remained restricted to certain sectors. This is unfortunate since the method, despite its complexity, has a lot to offer a world looking for a sustainable future.

14.3.2 The use of ecological footprinting at food and crop level

The ecological footprint was not conceived to assess agricultural production but the method can be modified for this purpose and can be used to calculate the bioproductive area required to produce an area (or weight) of an agricultural commodity. Lillywhite *et al.* (2007) used the technique on selected arable and horticultural crops; and reported global hectare (gha) values between 0.08 gha/t for carrot and 0.80 gha/t for winter wheat, but found that the definition of cropland within the methodology was too imprecise to allow the differences between similar conventionally grown crops to be fully explained. A comparison between two Italian wines by Niccolucci *et al.* (2008), one conventional and one organic, was more successful and showed that conventional wine had double the ecological footprint of its organic equivalent, 2.19 gha/t and 1.12 gha/t respectively. The results from these two studies illustrate that the ecological footprint can be successfully used to assess the differences between agricultural products; however, the issue is whether the results can be interpreted and understood by a wider audience. The lack of take-up in the last 15 years suggests that the results are too abstract for most people to understand and, as a consequence, the method is rarely used at crop and food level because other methods, principally Life Cycle Assessment and, increasingly, carbon footprinting, are seen as better alternatives.

14.3.3 The use of ecological footprinting for food consumption

Where ecological footprinting has value, and is increasingly used, is where the results are expressed on a *per capita* basis, as this allows comparisons to be made between human populations and for the results to be assessed against regional or global bioproductive areas. In 2001, a study calculated the ecological footprint of food consumption in south-west England to be 1.63 gha per person; this was split 1.25 gha for animal-based and 0.37 for plant-based food (Chambers *et al.*, 2005). Collins and Fairchild (2007) undertook a similar study for Cardiff and reported that food and drink consumption was responsible for an ecological footprint of 1.33 gha per person; the allocation to animal and plant based materials was 0.82 and 0.51 respectively. Canadian food consumption was examined by Mackenzie *et al.* (2008) who reported an average ecological footprint of 2.13 gha, which varied between 2.06 and 2.24 depending on household income. In Australia, the average resident in the State of Victoria has a food ecological footprint of 1.90 gha per person (EPA Victoria, 2008). All these results should be treated with some caution since the analysis is often based on household expenditure surveys and requires a number of assumptions to be made that may not hold true in real life. Nonetheless, the ability to compare ecological footprints across different populations and regions can provide good insights into where the greatest impacts occur and may suggest where reductions could be made in the future in the drive for sustainable food production and consumption. The Canadian study points out the obvious, although often neglected, conclusion that the size of a *per capita* ecological footprint is heavily influenced by income – the more you earn, the more you can potentially consume.

14.3.4 The influence of ecological footprinting

Although ecological footprinting is used widely to assess resource use and has proved to have real value in measuring progress towards sustainability, its greatest impact probably lies elsewhere. The use of ecological footprinting has remained a niche activity undertaken by academics and consultants, but the concept of the 'footprint' has proved attractive to many diverse groups of people and organisations and the term is now universally recognised as a 'measure' of impact. Whether the approach is qualitative or quantitative, commercial organisations, NGOs and the media have adopted the footprint as a simple way of portraying the impact of many aspects of modern life and living. Footprint has come to mean impact. The success of the footprint may be attributed to the fact that it is easy to visualise, and for that the originators of the ecological footprint should be congratulated. Non-scientific use often makes the footprint a unitless measure, but even then it may still manage to portray the extent of impact.

14.4 Water footprinting

Water footprinting has its origins in virtual, or embedded, water and it was introduced by Allan (1993, 1994, 1998a,b). He proposed that virtual water is the water required to produce a product from start to finish, and introduced the concept as an attempt to understand how international trade affects the global flow of water. Allan's idea was that water-poor countries could import their food from water-rich counties, thus saving their own scarce water resources for drinking and sanitation, rather than agriculture. However, international trade figures have subsequently revealed that trade in food products is driven by economic and availability factors rather than water resources. However, the value of knowing the volume of water required to produce a product was recognised by Hoekstra and Hung (2002), who subsequently developed the idea of the volumetric water footprint. Allan, with his colleague, Chapagain, and The Water Footprint Network, have been responsible for the growing popularity of the water footprint.

Virtual water is the amount of water required to produce a product, from start to finish and is a mainly neglected and hidden component of production. Virtual water is normally divided into three categories for classification: blue, green and grey. Blue water is the water contained in rivers and lakes and is the water processed by the water companies to supply public and commercial demand; this water is used in the food processing industry. Green water is the water supplied through rainfall and contained in soils; the majority of agricultural production is based on this water. The concept of grey water is slightly more confusing and its definition can change with perspective: traditionally grey water is wastewater, from domestic or commercial sources, which is mildly contaminated with detergents or other pollutants but which can still be discharged to the public sewer, e.g. bath water or dish washing water. However, a second definition exists within water footprinting which defines grey water as the volume of water required to dilute contaminated water so that it reaches the same quality as ambient water resources. This concept suffers because contaminants are present in different concentrations (and are especially difficult to measure in field conditions) and ambient water standards vary from place to place and from country to country. This concept of grey water is not universally accepted and many researchers avoid the issue by reporting only green and blue water.

Agricultural production uses large amounts of water. Chapagain and Hoekstra (2004) calculate that in the Netherlands it requires 619 000 litres of water to produce a tonne of wheat and 11 681 000 litres to produce a tonne of beef. A recent report (WWF–UK, 2008) suggests that imported food and fibre account for 62% of the UK's total water consumption.

The very large volumes of water required for some agricultural products have led to concerns being expressed over how the methodology has been developed. The water used during the agricultural stage is based on crop evapotranspiration, with values extracted from regional and global computer models. Although this approach has the simplicity required for undertaking numerous calculations, evapotranspiration values vary widely depending on crop location and any assumptions made on production methods. For example, winter wheat can be both a rain-fed and irrigated crop, depending on location. The Water Footprint of Nations report (Chapagain and Hoekstra, 2004) prepared volumetric water footprints for 210 countries and the results for wheat ranged between 465 (Slovakia) and 18 070 litres/kg (Somalia). Hotter and drier countries had larger water footprints which reveal a great deal about the relative evapotranspiration of the countries and irrigation use but very little about the amount of water required to grow a crop of wheat. Wheat in the UK required 501 litre/kg.

Zwart and Bastiaanssen (2004) reviewed crop water productivity values for 28 wheat crops across five continents and reported green water consumptions of between 588 and 1667 litres/kg, with the greatest frequency in the range of 909 to 1111 litres/kg. No European crops were considered in their review and it is unclear which, if any, of the crops were irrigated with blue water.

The global crop water model (GCWM) developed by Siebert and Doll (2010) was used to calculate virtual water contents for 30 crops. The value for wheat was 1469 litres/kg and was split between 1113 litres/kg green water and 356 litres/kg blue water. The relatively high blue water content is due to the assumption that 37% of wheat crops are irrigated. Hanasaki *et al*. (2010) modelled the volumetric water footprint of some major crops and reported values for wheat between 366 (France) and 1359 litres/kg (USA).

Despite some methodological issues, the volumetric water footprint is a useful metric for quantifying the amounts of water required to produce crops and prepare products at a global scale, and it is useful for benchmarking water use and promoting water efficiency. However, its weakness is that it requires a written summary to discuss the impact of water consumption at its place of use. Intuitively, it is obvious that it is better to grow wheat in Slovakia rather than Somalia, but what is required is a way to incorporate that information into the metric itself rather than relying on a qualitative interpretation.

These concerns and the reliance on an evapotranspiration parameter has led some researchers to suggest that green water is a function of land use and that it should be excluded from the water footprint. This line of thought has prompted the introduction of the stress-weighted water footprint (SWWF) which is a development of the volumetric water footprint and an attempt to include an impact assessment related to local water resources (Ridoutt and Pfister, 2010). A water stress index factor between 0.01 and 1 is applied to water consumption at every stage within the life cycle of the product or process under investigation (Pfister *et al*., 2009). The normalised volumes are then summed for every life-cycle stage and the result is an SWWF. This approach assesses the individual impacts at different stages within the life cycle and is consistent with carbon footprinting and Life Cycle Assessment. Green water is excluded from the calculations since it is assumed to be a function of land use. This exclusion means that the resultant water footprints are considerably smaller than volumetric water footprints but may be a better assessment of the actual impact. This approach, with its use of impact factors for the different components, is similar to current carbon footprinting methodology (PAS2050, 2008) and may indicate that some of the current footprinting techniques are becoming compatible with Life Cycle Assessment.

Food processing, whether simply washing prior to sale or more complicated preparation, uses large quantities of water. This water, once used, can be immediately discharged to the sewerage system, may require cleaning or diluting before discharge, or may be reused a number of times before discharge. These different procedures, especially when considered together with the multiple definitions of grey water, ensure that calculating the volumetric water footprint of the processing stage is difficult. Currently there is no unified approach and researchers have adopted various approaches to overcome the challenge within this area.

In summary, the water footprint, both volumetric and stress-weighted, has a lot to offer the study of resource use and environmental impact. Ecological footprinting and water footprinting have been developed in parallel but, as yet, there has been no crossover. Ecological footprint does not consider the impact of water since it is difficult to express water use in terms of global hectares and although neither volumetric nor SWWFs are compatible with ecological footprinting, they do address a gap in environmental impact research. Hoekstra (2009) compared ecological and volumetric water footprint analysis and concluded that 'the two concepts are to be regarded as complementary in the sustainability debate'. Water footprints are useful on their own but future developments are likely to use them together with other environmental impact measures. The recent development of the SWWF and its similarities with carbon footprint methodology pave the way for a new era of environmental accounting.

In the same way that the carbon footprint is a large component of the ecological footprint (Global Footprint Network, 2010), the water footprint could be extended to include the CO_2e associated with the production and supply of water. All blue water has a carbon footprint. At some point in the supply chain, it has generated a carbon burden, whether it has been supplied by a water company, captured and stored on a farm, or abstracted from a river or ground source. All of these sources have carbon dioxide emissions associated with construction, cleaning, pumping, and application. The carbon footprint of embedded water can be of special interest, e.g. in produce imported from areas of Spain, where irrigation water can be supplied from high-energy desalination plants, or can have environmental and sustainable development implications, e.g. from Morocco, where water demand for horticultural irrigation has lowered the water table to the detriment of future supplies.

14.5 Carbon footprinting

Carbon footprinting has a short but energetic history. It evolved from the establishment of the Intergovernmental Panel on Climate Change (IPCC) who collated and interpreted information on the emissions of greenhouse gases and their global warming potential (GWP). Originally a carbon footprint referred only to emissions of carbon dioxide, but now the term is applied to a normalized summation of all greenhouse gases. Although the IPCC recognize sixty or more greenhouse gases, within food and agriculture only three gases are commonly assessed: carbon dioxide, methane and nitrous oxide (IPCC; 2006). A carbon footprint can also be referred to a carbon dioxide equivalent (CO_2e). The rise in popularity of carbon footprinting can be illustrated by the number of peer-reviewed academic papers that have been published since 2000. In the six years between 2000 and 2006 there were none, yet in 2009 there were 91 and the term 'carbon footprint' is now familiar to scientists and non-scientists alike.

The calculation of a carbon footprint requires two key pieces of information: the emission of the greenhouse gas associated with the product or process under investigation and its global warming potential. The weight of emissions is 'normalised' (made equivalent to CO_2 by multiplying it by its global warming potential) and all the normalised values are then summed to give the carbon footprint. The global warming potential of the individual gases, which is based on the radiative forcing of a tonne of the gas over 100 years, are $CO_2 = 1$, $CH_4 = 25$ and $N_2O = 298$. So, for example, a product emitting 1 kg each of CO_2, CH_4 and N_2O would have a carbon footprint (CO_2e) of 324 kg.

Carbon footprinting of a product or process was formalised by the publication of ISO 14064 (ISO, 2006b) 'Specification with guidance at organization level for quantification and reporting of greenhouse gases' and PAS2050 'Specification for the assessment of the life cycle greenhouse gas emissions of goods and services' (PAS 2050; 2008).

14.5.1 Carbon footprinting in food production

The popularity and acceptance of carbon footprinting may be explained by the fact that most people either accept, or can make the connection between, greenhouse gas emissions, global warming and climate change; and food is an ideal candidate for carbon footprinting since all three major greenhouse gases are emitted during its production – carbon dioxide emissions arise from the use of energy (diesel, electricity, gas); methane is a product of enteric fermentation in ruminants, and additional nitrous oxide is emitted through the use of high nitrogen fertilizers applied to land. The carbon footprint is now an accepted component of all Life Cycle Assessments and ecological footprints undertaken within the food and agricultural sectors.

The use to which carbon footprints are put varies considerably. At its simplest, the analysis can be used to identify the biggest greenhouse gas contributor with the life cycle of a product. This approach was adopted in the early stages of footprinting, when researchers were still coming to terms with the methodology, and will prove valuable as the producers seek to reduce the greenhouse gas emissions associated with food production and at the same time attempt to increase production to meet an increasing global population.

A second use is to compare product life cycles with the aim of identifying the most carbon efficient method or place of production. This approach can be divided into two categories: comparing different production systems in the same location, for example conventional versus organic production, or comparing the same production systems in two locations, for example, Europe versus New Zealand. Both uses seek to identify the most efficient production systems in terms of greenhouse gas emissions but the analyses are often confounded by other factors, for example, quality and freshness of product.

The conventional versus organic debate is interesting since the inbuilt differences between the two systems do not allow a clear answer to emerge. Currently, yields from organic systems are smaller compared to those from conventional systems and this difference makes most comparisons invalid since analysis based on a 'per yield' basis will normally be biased in favour of the conventional system and analysis on a 'per area' basis will favour the organic system. This was shown ten years ago when Cederburg and Mattsson (2000) reported that conventional milk production had a higher carbon footprint compared to organic, although the results, using energy-corrected milk as a functional unit, were not quite as different as may have been expected. More recent work by Lillywhite (2009) on winter wheat and potatoes has confirmed that, on an area basis, organic crops have a lower carbon footprint but that results are reversed on a yield basis; although for potatoes, where organic production often requires extra field cultivations and plant nutrients, such as farm yard manures, the advantage that it enjoys is again not as great as some people would believe.

Carbon footprinting analysis has proved popular in continuing discussions as to whether domestic production is better than imported produce, and how the balance between the two varies with location and season. The driver for many of these discussions has been whether the cost and environmental impact of transport (the 'air miles' debate) is offset against product storage and freshness. This area of research has illustrated the complexity of global food chains and that there is no right or wrong answer, just different viewpoints. The debate is well illustrated by a number of reports that assess production in New Zealand and Europe, and compare the results. Saunders *et al*. (2006) compared New Zealand and UK production of various agricultural commodities and reported that, even allowing for transportation from New Zealand to the UK, milk, apples and lamb had a lower carbon footprint than their UK counterparts. Williams *et al*. (2009) agreed that lamb produced in New Zealand and imported into the UK had a lower carbon footprint compared to UK produced lamb but suggested that the advantage was not as great as Saunders had shown. In contrast, Wiltshire *et al*. (2009) showed that UK lowland lamb had a smaller carbon footprint than its New Zealand equivalent. These different results are not necessarily reflective of the different researchers involved or their interpretation of methodologies, but are influenced by the multiple production systems that are commonplace and the difficulties involved in obtaining robust and comparative data sets. Although carbon footprinting is a valuable tool, it should be remembered that, unlike financial accounting, environmental accounting is not an exact process and that biological systems are inherently variable.

14.5.2 Carbon labelling

One of the biggest impacts of carbon footprinting has been the introduction of carbon labelling on food (and non-food items). A carbon label reports the carbon footprint on a per product basis. An example is Walker's crisps in the UK which reports that a 34.5g bag of salted crisps is responsible for 80g CO_2e. The drive to include carbon labels on food products has come from central governments who wish to reduce greenhouse gas emissions, and from the desire of major food processors and retailers to be seen to be 'green' and reduce their environmental impact.

In 2007, the UK supermarket group Tesco, announced that they would develop labelling for all their products displaying their carbon footprint; this is an enormous task. To achieve this goal, they have committed serious financial resources to undertake research into every one of their products. To date, ten products have carbon labels. Other retailers, being slightly nervous of the investment in time and money, are waiting to see what happens before committing themselves to similar schemes. To date, this is the only labelling scheme that relies on carbon footprinting.

Other, short lived, labelling schemes have appeared. Stickers with aeroplanes were briefly used to identify food that had been imported into the UK by air, but LCA and footprinting studies have subsequently shown that air transport contributes little to the overall carbon footprint and that, depending on the time of the year, imports by air from countries with production under ambient conditions could prove advantageous to the carbon footprint.

14.6 Environmental footprinting

The discussion so far on ecological, water and carbon footprinting has concentrated on their individual applications and highlighted that very little crossover exists. Although sustainability, water and carbon are very important topics within their own right, there exists a demand for an approach which can assess multiple indicators. Life Cycle Assessment fits that criteria but cannot quantify multiple indicators into a unified single-value result, which is the strength of footprinting.

The term 'environmental footprint' has appeared in a number of academic papers and grey literature in the last few years, but the term has no actual definition. Both The Advisory Committee on Releases to the Environment (ACRE, 2007) and the Agriculture and Environment Biotechnology Commission (AEBC, 2003) have used the term to describe the overall impact on the environment but without attaching any quantifiable indicators. Likewise, many commercial organisations (SABMiller, 2009) use the term, along with carbon footprint, to describe their overall environmental impact in terms of carbon dioxide, carbon dioxide equivalents or even their waste production.

In theory, there is no problem with aggregating multiple indicators and reporting a single value; after all, that is the approach used with carbon footprinting. Any number of indicators could be included in an analysis as long as they share a common denominator and reporting unit. However, in practice, a balance needs to be reached between making the assessment as wide as possible whilst still retaining a good level of detail. Lillywhite (2007, 2008, 2009) proposed an environmental footprint to include the carbon footprint along with indicators describing pesticide toxicity, eutrophication, acidification and water. The footprint is calculated on an area basis but can be reported by either product unit or by area. This approach possesses the breadth of the LCA but not its awkwardness in reporting. In summary, it may be viewed as complimentary to an LCA or as an extension to a carbon footprint. The area of environmental accounting is one that is developing quickly and the question of whether the environmental footprint will prove to be a useful addition to the current suite of methodologies is still open.

14.7 Nitrogen footprinting

Although global concern is currently focused on carbon, many researchers and academics consider nitrogen to have an equally negative impact on the environment. The nitrogen gas in the atmosphere is inert but many man-made reactive forms cause environmental damage: nitrous oxide is a greenhouse gas, nitrate from application of fertilisers can lead to eutrophication and ammonia from animal manures and fertilisers, and nitrogen oxides from the combustion of fossil fuels causes acidification. The success of the carbon footprint has shown the footprint concept to be a powerful communication tool and some researchers are hoping the same approach can be used to illustrate the problems of nitrogen. James Galloway at the University of Virginia is currently developing methodology for calculating nitrogen footprints in an attempt to highlight the effect that reactive nitrogen has on the global environment. His approach would mimic carbon footprinting and normalise all the nitrogen compounds with adverse environmental effects into a single value. This approach will pose many problems because individual nitrogen compounds have different origins and impact different areas of the environment, so development of any method will be difficult but it may be that it is an area where footprinting will develop in the future.

14.8 Non-environment footprinting

Footprinting is not confined to the physical sciences. The concept of aggregating data with a common unit has been extended to the economic and social sciences. Lobley *et al.* (2005) and Lillywhite *et al.* (2008) used multiple indicators to explore the differences between organic and non-organic farming systems, and between different farming systems and crops.

14.9 Future trends

Footprinting was introduced only twelve years ago, yet in this short period the term has entered common usage. This is a potential problem. So quick has been its uptake that its definition may have been lost in the process. Ecological footprinting has been developed by its innovator and The Footprint Network into a methodology with standards that has been used with impact by certain organisations, notably WWF. Carbon footprinting, now under PAS2050 specification, has become a recognised methodology; although implementation is still in its infancy. Water footprinting is also finding useful applications although it is still under development. Environmental and nitrogen footprinting are new concepts and it is still too early to say whether they will find an audience within environmental research and policy use. However, this type of methodology has proven capable of portraying a complicated subject in a relatively easy to understand manner and it likely that it is here to stay and that development will continue to expand its use into new and unknown areas.

Researchers will no doubt find new areas of science to which footprinting techniques can be applied but it may be time to stand back a step and consider where footprinting is going. Footprinting is an applied science and needs to supply answers to policy questions. Carbon footprinting in undoubtedly the analysis of choice at the moment and is likely to remain so, due to continuing global pressure to reduce emissions of greenhouse gases. However, the fixation on greenhouse gases may be neglecting water and nitrogen, which large parts of the scientific community consider to be as important as carbon. Given the popularity of the footprint concept, it would be beneficial for those research areas if these footprinting techniques (and others) are not swamped in the rush to concentrate of carbon.

14.10 Sources of further information and advice

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Wackernagel M and Rees W. 1996. *Our Ecological Footprint*. New Society Publishers, Gabriola, Canada.

A large amount of information is available on the internet. The following sites are useful starting points:

British Standards Institution (PAS2050) – www.bsigroup.com

Global Footprint Network – www.footprintnetwork.org

Water Footprint Network – www.waterfootprint.org

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15

Carbon footprinting and carbon labelling of food products

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Abstract: Carbon footprints estimate the total amount of greenhouse gases emitted during the production, processing and retailing of consumer goods. The aim is to identify major sources of emissions in supply chains to inform relevant stakeholders so that actions can be taken to reduce emissions. Carbon footprints can also be communicated to consumers via carbon labels. This chapter first describes the principles of carbon footprinting and labelling, and presents some examples of carbon footprints of food products. It then discusses problems in calculating carbon footprints and finally speculates on future developments of the methodology and its application.

Key words: greenhouse gas emissions, food, PAS 2050, carbon footprinting.

15.1 Introduction

Concern about climate change has stimulated interest in estimating the total amount of greenhouse gases (GHGs) emitted during the production, processing and retailing of many consumer goods, including food products. Carbon accounting and labelling for products are new instruments of supply chain management that analyse and present information on the emission of GHGs from products and services. A carbon footprint is the final summary of GHGs emitted from the system under analysis, which can be any defined system, e.g. a farm, an entire food supply chain including consumption and waste disposal, or parts thereof. Carbon footprints are expressed in terms of carbon dioxide equivalents per functional unit (CO_2e), where the word 'equivalents' indicates that the global warming potentials of the different GHGs have been normalised relevant to that of CO_2. The functional unit is the item of production that is being analysed, e.g. a litre of milk.

The overall purpose of a carbon footprint is to identify major sources of emissions in supply chains to inform policy makers, businesses and other stakeholders so that actions can be taken to reduce emissions. Carbon footprints can also be communicated to consumers via carbon labels in an attempt to change consumption behaviour.

The purpose of this chapter is to:

(i) describe the principles of carbon footprinting and labelling;
(ii) present some examples of carbon footprints of food products;
(iii) discuss the problems of calculating carbon footprints for food products; and
(iv) speculate on possible future developments of the methodology and application of carbon accounting.

15.2 Principles of carbon analysis and carbon labelling of food products

15.2.1 Background

The framework for carbon footprinting is provided by life cycle thinking and methods for Life Cycle Assessment (LCA). However, the needs of supply chain carbon footprints are not fully met by either the existing standards for LCA, as prescribed by the International Organisation for Standardisation (ISO), or standards for company greenhouse gas accounting such as the GHG Protocol developed by the World Resources Institute (WRI). Additional principles and techniques that address essential aspects of carbon footprinting need to be developed and established for carbon accounting. Although internationally agreed standards on carbon footprinting have not yet become operational, a lot of companies are keen to calculate and communicate the carbon footprint of their products to their customers, many of whom are increasingly interested in the climate change impact of their consumption (Carbon Trust, 2008a; Bolwig and Gibbon, 2009). This is why new methods for carbon footprint calculations are currently being developed by various organisations, businesses and governments around the world. While these responses to climate change are to be welcomed, the requirements of the different stakeholders can lead to inconsistency in different carbon accounting schemes, which in turn provides limited confidence in results and comparability of studies.

15.2.2 Current development of methodologies

At least 16 different methodologies for calculating the carbon footprint of products have been developed since 2007 or are still under development (Brenton *et al.*, 2009b). Countries in which standards are being developed include the UK, Germany, France, Switzerland, Sweden, New Zealand, the USA, Japan, Korea and Thailand. Both the International Organisation for

Standardisation (ISO) and the World Resources Institute (WRI) together with the World Business Council for Sustainable Development (WBCSD) are working on the development of international standards for carbon footprinting. The schemes vary greatly in approach and methodology and are mainly established by governments and businesses, especially supermarket chains. Some of their methodologies are publicly available and provide users with detailed advice on how to undertake a carbon footprinting exercise, while others have been developed but the detailed methodology is confidential. Most schemes, however, are still under development at the time of writing. Although there are several schemes that are already being implemented at the time of writing, the British PAS 2050 (BSI, 2008; Sinden, 2009) is currently the only finalised carbon footprinting methodology that has detailed calculation methods in the public domain. For this reason the PAS 2050 methodology is the only one that is presented in some more detail here.

PAS 2050 is built upon the existing ISO 14040/44 standards for LCA, which are further clarified, adapted and specified. A PAS 2050-compliant carbon footprint can be calculated either as a business-to-consumer assessment which includes the full life cycle of a product ('cradle-to-grave'); or as a business-to-business assessment, which includes all upstream GHG emissions up to the arrival of a product as an input to a new business or organisation ('cradle-to-gate'). PAS 2050 accounts for emissions of all GHGs including CO_2, N_2O, CH_4 and families of gases such as hydrofluorocarbons (HFCs) and perfluorocarbons (PFCs). Each gas is converted into a CO_2 equivalent value, determined by its ability to trap heat in the atmosphere over a 100-year period relative to CO_2, known also as its Global Warming Potential (GWP).

PAS 2050 specifies rules for identifying the system boundary and data quality rules for secondary data in order to standardise the process and leave fewer decisions up to the analyst. GHG emissions from energy use, combustion processes, chemical reactions, refrigerant losses and other fugitive gases, operations, service provision and delivery, land use change, livestock, other agricultural processes and waste have to be included in the assessment. The unit of analysis should be the unit in which the product is actually consumed by the end user. Direct land use change emissions where the land use change occurred on or after 1 January 1990 should be calculated according to the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006). One-twentieth (5%) of the total emissions arising from the land use change have to be included in the GHG emissions of the products concerned in each year over the 20 years following the change in land use. Non-CO_2 emissions from livestock, their manure and soils are to be calculated in accordance with the highest tier approach set out in IPCC guidelines or the highest tier approach employed by the country in which the emissions arise. Any changes in the carbon content of soils are excluded (either emissions or sequestration) other than those from direct land use change, due to the considerable uncertainty in their assessment. Capital goods are also not included in the PAS 2050 methodology, although this category might be of significance for agricultural products, where agricultural machinery has been shown to have an impact (Weidema *et al.*, 1995). In addition, PAS 2050 makes provisions for carbon storage in products and delayed emissions from the use phase and final disposal of a product. Emissions related to human energy inputs, transport of consumers to and from shops, transport of employees to and from work and animals providing transport services are excluded from the carbon footprint calculation. PAS 2050 was reviewed and updated in 2010.

15.2.3 Carbon labelling

There are two very different approaches to communicate the results of a carbon footprint calculation to the public. In the first approach, precise figures are calculated and communicated to consumers on labels. The second approach attempts to guide the consumer to more climate-friendly products without giving precise figures. PAS 2050 is an example of the former, and compliant carbon footprints can be communicated via a carbon label awarded by the Carbon Trust. This label indicates the greenhouse gas effect of a product by giving a single figure of CO_2e per functional unit of analysis, e.g. per kg of food product or litre of drink. A company that prints this carbon label on a product commits to an emission reduction across the whole of that particular supply chain within two years. If it fails to achieve this it will lose the right to use the label. This is seen as an encouragement for all businesses to reduce the carbon footprints of their products. It also means that any product can receive the carbon label. The UK Carbon Label Company was set up in 2007 to ensure that measurements are comparable across products within a product category, and to allow valid comparisons between products.

Food products with published carbon footprints developed according to the PAS 2050 guidelines include Walkers crisps (80 g CO_2e per 35 g bag), Tesco potatoes (e.g. 160 g CO_2e per 250 g serving of King Edwards), Tesco orange juice (e.g. 240 g CO_2e per 250 mL serving from a 1 litre carton) and Kingsmill sliced white and wholemeal bread (1.3 kg g CO_2e per 800 g loaf). Other companies have chosen to benefit from a PAS 2050 compliant GHG analysis of their supply chains, identifying emissions and cost-saving opportunities, and catering to their consumers' demands for information, but not to apply for the carbon label. An example is Innocent who declare the carbon footprint of their fruit smoothies on their website. They also communicate emissions reductions that have been achieved on their website, e.g. the carbon footprint of a 250 mL carton of cranberry and raspberry smoothies was 258 g CO_2e in March 2007, 217 g CO_2e in December 2007 and 206 g CO_2e in December 2008 (www.innocentdrinks.co.uk, accessed 22 September 2009).

Walkers crisps have recently been awarded the carbon reduction label for another two years due to the emissions savings that have been made since the first assessment (7%). The first assessment was carried out in 2007 using the draft version of PAS 2050 and resulted in 75 g CO_2e per 35 g bag. Due to changes in the calculations as the methodology moved from its draft to its published version, the carbon footprint was recalculated in 2009 as 85 g CO_2e per 35 g bag. Thus, although emissions were reduced, the figure printed on the bags has actually increased from 75 to 80 g CO_2e per bag. This might lead to some confusion amongst interested consumers and is one problem attached to the continued development of the methodology, which has been applied and the results communicated by many businesses keen to interact with their consumers before the methodology has matured.

In contrast to printing actual figures of CO_2e on products, the Swiss supermarket Migros or the German Blauer Engel approaches award a label only to the most climate-friendly products within a product group. In these schemes there are no precise figures on the product label, but rather the fact that the product carries the label informs the customer about its relatively better environmental performance. This is expected to stimulate competition for the label, encourage companies to innovate and achieve emissions reductions and to guide the consumer to more climate-friendly products. The possibility of including carbon footprinting within the European Ecolabel is being considered as part of a revision of the label in 2009. The revised Ecolabel might also be available to the categories of food and drink that so far have been excluded.

15.3 Examples of some food products for which carbon footprints have been published

In this section, some examples of food products for which carbon footprints have been published will be used to illustrate the method, its results and potential application. These products cover the categories fresh produce, meat, dairy and processed foods. For agricultural products, the main GHG contributing to the carbon footprint often is not CO_2, but rather N_2O, mainly related to the use of nitrogen fertilisers, and/or CH_4, especially on livestock farms (Flessa *et al.*, 2002; Williams *et al.*, 2006; Edwards-Jones *et al.*, 2009a; Hillier *et al.*, 2009). The section on meat also illustrates three different approaches to carbon accounting which relate to the method of data collection and calculation, each of them with advantages and disadvantages and serving different purposes.

15.3.1 Fresh fruit

Strawberries from Spain

In the context of the German Product Carbon Footprinting (PCF) Pilot Project, a case study was conducted to analyse the carbon footprint of strawberries produced in Spain and consumed in Germany (PCF Project, 2009a). The aims of the PCF initiative are to test and evaluate the practical application of the current and evolving methodologies for carbon footprinting (based on ISO norms for LCA) by working with several companies to calculate product carbon footprints; to give recommendations for the further development of methodologies based on the findings from the case studies; and to discuss how to best present carbon footprinting results in the form of labels. The project is run by the Institute for Applied Ecology (Öko-Institut), Potsdam Institute for Climate Impact Research (PIK), the think/do tank THEMA1 and WWF.

Strawberries were chosen as a case study as an example of fresh produce that is transported over long distances to be sold out of season in the importing country. These 'food miles', i.e. the distance that food has travelled from farm to consumer, are the reason why fresh products that are imported from far away countries have been criticised as having a worse environmental impact than more locally-produced food (Norberg-Hodge *et al.*, 2002; Morgan *et al.*, 2006).

The Spanish case study strawberries were produced adjacent to an important nature conservation area. In order to protect this habitat and the species it supports, measures were implemented to save water, reduce pesticide applications and rewet drained areas. The functional unit for the analysis was a 500 g punnet, as packed on farm, and all life cycle stages up to consumption and waste disposal were included. The distance from the farm in Spain to the distribution centre in Germany was 2224 km and the journey was conducted in 40 t refrigerated trucks with a 38% load; the return trips were used to transport other goods. All distribution centres and shops used renewable energy to refrigerate the strawberries and thus no emissions from refrigeration were considered in the carbon footprint of these supply chain steps. For the transport from the central to regional distribution centres, a worst case scenario based on the longest possible distance within Germany was assumed. Wastage of strawberries during retail due to their perishable nature amounted to about 4%. Consumer shopping trips were assumed to be by private diesel car over a distance of 5 km each way and the total shopping to amount to 20 kg, including one 500 g punnet of strawberries. Because strawberries are usually consumed quickly after purchase without long storage or further processing, the use phase was not considered in the analysis. Primary data on inputs used on the farms and transport from Spain to Germany were supplied by the company owning the farms. There were no co-products from the farm so that no allocation of emissions was necessary. The calculations were conducted using the software Umberto.

Table 15.1 shows the results for the different life cycle stages. The farming stage had the largest share in total emissions (41%), followed by transportation from Spain to Germany (32%) and consumer shopping trips (15%). Looking more closely at the farming stage (Table 15.2), four main emission sources can be identified: the production of the PET punnets; the

Table 15.1 Greenhouse gas emissions from the different life cycle stages of a 500 g strawberry punnet produced in Spain and consumed in Germany in g CO_2e and percent of total emissions (Adapted from PCF Project, 2009a)

	g CO_2e	% of total emissions
Raw materials	0.8	0.2
Production on farm	182.2	41.2
Distribution	139.8	31.6
Consumer shopping trip	65.4	14.8
Use phase	0	0
Waste disposal	53.8	12.2
Total	**442.0**	**100**

Table 15.2 Greenhouse gas emissions from the on-farm production stage of strawberries in Spain in g CO_2e/500 g punnet and percent of total emissions from production (Adapted from PCF Project, 2009a)

	g CO_2e	% of total emissions
Production of PET punnets	65.4	35.9
Production of PE sheets	61.0	33.5
Production of pesticides	31.0	17.0
Electricity use in packhouse	13.9	7.6
Production and use of fertilisers	8.3	4.6
Energy use on farm	1.0	0.5
Transport of PET punnets	1.0	0.5
Transport steps on farm	0.7	0.4
Total	**182.3**	**100**

production of plastics used in the field and for polytunnels; the production of pesticides; and electricity use in the packhouse in Spain. Recycling and re-use of PET punnets and packaging materials was calculated to allow an offset of 2.5 g CO_2e/500 g punnet or 0.6% of the total life cycle carbon footprint. The GHG with the greatest contribution was CO_2.

These results help identify opportunities for emissions reductions that can be achieved by both the producer and the consumer. Alternatives for packaging materials should be developed and tested to reduce the impact of PET and PE plastics used. Renewable energy on the Spanish farms and in the packhouses, the optimisation of pesticide and fertiliser applications, as well as improved logistics during transportation, also have the potential to significantly reduce overall GHG emissions from the production and distribution of the strawberries. Another emissions hotspot was the consumer shopping trips. Communication of the results and awareness raising might contribute to a reduction of these emissions if consumers increasingly walked to shops, or used public transport or bikes. Retailers could support this by increasing the number of small shops closer to the people so that consumers do not always have to travel to large out of town shopping centres.

The results of a modelled LCA yielded similar results for UK grown strawberries (0.73 kg CO_2e/kg strawberries) from farm to retail distribution centre (Williams *et al*., 2008). Spanish strawberries delivered to the UK were modelled to have a higher carbon footprint of 1.3 kg CO_2e/kg strawberries up to the UK retail distribution centre. This LCA also highlighted the greater use of water in Spain, where strawberry production used about 1.6 times more water than in the UK. In an area of naturally low levels of rainfall, this is an environmental concern which should not be overlooked (Williams *et al*., 2008).

Pineapples from Mauritius

As part of a research project that asked the question whether current carbon footprinting methodologies disadvantage developing countries, a case study was conducted on fresh pineapples imported to Europe from Mauritius (Brenton *et al*., 2009b). This is an example of a tropical commodity that is not produced commercially in Europe and is not being replaced by out of season glasshouse production on a commercial scale.

Primary data were collected by visiting one case study farm in Mauritius in May 2009. Data collection was carried out by means of questionnaire-style interviews with the agricultural managers and wider industry contacts. The carbon footprint of products can be of commercial sensitivity, and in order to protect the business that participated in the project we are providing only very limited information on the farm that was visited. Over 50% of its pineapples (variety Queen Victoria) are exported to Europe. Most of the work on the farm is done manually, including harvesting, planting and the application of fertilisers and plant hormones. Soil preparation is done using small machines in most years, but deep ploughing using a caterpillar becomes necessary every few years. Harvested pineapples are transported to the airport to be air freighted to their markets in Europe. The average weight of export quality fruits was 500 g. No land use change had occurred for several decades.

The carbon footprint was calculated according to PAS 2050 and included all emissions from cultivation, processing and transport to European export destinations. Primary data collected from the farm was used as much as possible. Emission factors were extracted from the Ecoinvent database (Althaus *et al*., 2007; Nemecek *et al*., 2007; Spielmann *et al*., 2007), Carbon Trust (2008b) and the International Energy Agency (2007). No allocation of emissions between pineapples of export quality and non-export quality was made; however, the latter are sold in local markets and could be regarded as a by-product. Transport to the airport was non-refrigerated. It should be noted that only one farm was analysed and, as such, the results do not represent average or statistically valid figures for Mauritian pineapples.

The carbon footprint of pineapples delivered to the airport was 0.23 kg CO_2e/kg of pineapples (Table 15.3). The use of diesel on the farm had the greatest impact on the farm-gate carbon footprint (26%), followed by the manufacture of N:P:K fertiliser (24%) and urea (15%), as well as nitrous

Table 15.3 Percentage of greenhouse gas (GHG) emissions per kg of fresh pineapples produced on a single farm on Mauritius, delivered to the airport. Also shown is the total carbon footprint in kg CO_2e/kg fresh fruit (Adapted from Brenton *et al*., 2009b)

Inputs and processes	% contribution to total GHG emissions
Diesel usage	26.2
N:P:K fertiliser (17:8:25) production	23.9
N_2O from N fertiliser	16.4
Urea production	15.1
Plastic production (for mulching)	6.9
TSP fertiliser production	4.8
Electricity usage	2.8
Herbicide production	1.8
Potassium sulphate production	1.4
Ripener production	0.6
Total kg CO_2e/kg of pineapple	**0.23**

oxide emissions following nitrogen applications to soil (16%). Trucking of pineapples to the airport amounted to 0.02 kg CO_2e/kg of pineapples. Transportation by air to Europe (10 000 km) added another 10.8 kg CO_2e/kg of pineapples, highlighting the great impact that this mode of transportation over large distances can have, bringing the total carbon footprint up to delivery in Europe to 11 kg CO_2e/kg of pineapples.

At the farm gate, emissions from the cultivation of pineapples were very low and compared well with other fruit, e.g. the previously mentioned study on Spanish strawberries, or oranges produced in Spain (between 0.23 and 0.28 kg CO_2e/kg at the farm gate for different production scenarios; Sanjuán *et al*., 2005). However, because pineapples in this case study were air freighted to their European destination due to their short shelf-life and logistics demands on a small island like Mauritius, their final carbon footprint increased to 11 kg CO_2e/kg. This carbon footprint is similar to that of other fresh produce that is air freighted from Africa, e.g. green beans (11 CO_2e/kg up to consumption) (Milà i Canals *et al*., 2008). Fruit that can be shipped at low to medium temperatures, in contrast, can be much less GHG intensive, e.g. oranges or bananas (Garnett, 2006). The production of highly perishable fruit and vegetables in heated and lighted glasshouses in European countries can have similar levels of GHG emissions to tropical produce that is air freighted to Europe, e.g. conventional and organic UK glasshouse tomatoes have a carbon footprint of 9.1 and 17.5 kg CO_2e/kg respectively at the farm gate (Williams *et al*., 2006), although other authors have calculated lower figures. Hospido *et al*. (2009) compared lettuce grown in the field in Spain with UK glasshouse lettuce, both consumed in the UK. The results showed the advantages of importation from Spain (0.4–0.5 kg CO_2e/kg lettuce at the UK retail distribution centre) over the lettuce grown in protected systems in the UK winter (1.5–3.7 kg CO_2e/kg lettuce). Other produce with large energy consumption during protected cultivation, leading to a high carbon footprint, includes peppers and cucumbers (Jones, 2006). It is also worth noting that the carbon footprint of meat produced and consumed in Europe can be as high or even higher; for example, see next section for figures on lamb produced in the UK and Ireland.

15.3.2 Lamb

The carbon footprint of lamb has been calculated by several authors in different countries. The results range from 10.0 kg CO_2e/kg live weight in Ireland (Casey and Holden, 2005a) and 12.9 kg CO_2e/kg live weight in the UK (Edwards-Jones *et al.*, 2009a) to 10.1–17.5 kg CO_2e/kg dead weight in the UK, depending on the farming system and based on a killing out percentage of 47% (Williams *et al.*, 2006). Figure 15.1 shows the contribution of different inputs and processes to the carbon footprint per hectare of farmland for an example farm in the UK uplands which produces

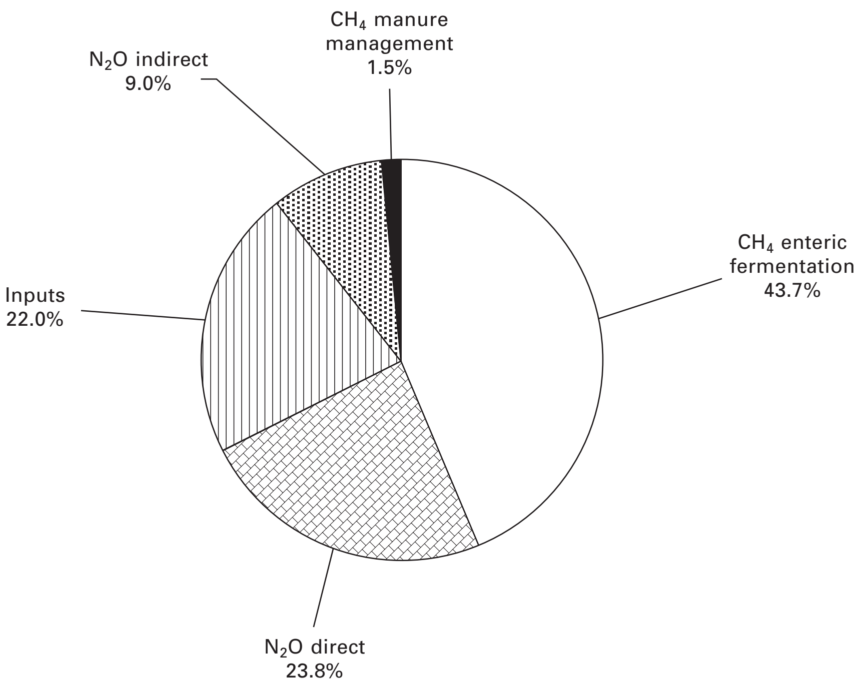

Fig. 15.1 Percentage contribution to total GHG emissions in kg CO_2e per hectare of farm inputs, direct and indirect N_2O emissions from soil and manure management, and CH_4 emissions through enteric fermentation and from manure management on a case study farm in the UK. Indirect emissions of N_2O arise after leaching and volatilisation. Inputs include the use of energy on the farm, as well as emissions embodied in farming inputs such as fertilisers, pesticides and concentrate feed. Source: Edwards-Jones *et al.* (2009a).

lamb and beef. Methane emissions from enteric fermentation dominated the carbon footprint, followed by direct N_2O emissions from soils as a result of nitrogen fertiliser applications. These results highlight the importance of best practice fertiliser applications which can help reduce N_2O emissions from soils, whereas not much can currently be done to reduce emissions from enteric fermentation. Measures such as using renewable energy on the farm can help reduce emissions, but these emission savings will be small in comparison to the emissions from enteric fermentation.

The three studies on lamb mentioned above illustrate three different approaches to carbon footprinting analyses: Edwards-Jones *et al.* (2009a) used real farm data collected from case study farms to calculate empirical carbon footprints; Casey and Holden (2005a) used data on the average sheep farming system in Ireland and used these aggregated figures to calculate a regional average carbon footprint; and Williams *et al.* (2006) modelled various farming systems using a top-down approach to identify potential differences between systems.

Models are based on theoretical considerations, rather than on data collected from farms. For example, Olesen *et al.* (2006) quantified imports of fertilisers and feed as a function of the desired milk production. These system models have not been developed to represent variation between regions and farms in terms of inputs, processes and outputs, and they do not take account of inter-annual fluctuations in yield due to edaphic and biological reasons. They do, however, allow the identification of more general trends and conclusions on e.g. emissions hotspots.

Aggregated LCAs and carbon footprints for food produce are based on real farm data collected from a large sample of farms, e.g. national statistical data. Another example for this approach is Thomassen *et al.* (2009), where environmental and economic indicators for Dutch dairy farms were calculated using Farm Accountancy Data Network figures for 119 specialised dairy farms. This approach is based on data from real farms and, as such, includes the variability between individual businesses. The results of these analyses are more statistically representative of a larger geographical area, and allow general conclusions on best-practice management or mitigation measures.

The third approach, using so-called empirical footprints, are those studies where detailed farm-specific data are collected from one or more farms or other actors in the supply chain, and then used to construct the carbon footprint for a number of individual farms or businesses.

Each of these methods has advantages and disadvantages and serves different needs. Modelling and aggregated analyses enable an efficient and powerful way of repeating and extending the analysis or assessing the impact of changes made in the system. A disadvantage of these approaches is that they may not represent the diversity and messiness of real agricultural systems. Important sources of uncertainty for agricultural products are the variability of inputs between years which leads to a variation in the carbon footprint from year to year (e.g. due to different weather patterns) and between farms (e.g. due to different soil types, management practices, yields) which will not be reflected in the results of modelling and aggregated analyses. The empirical approach is a more applied approach that is used by, for example, businesses, to calculate the climate change impact of their products, using methodologies such as PAS 2050. In these cases, the modelled and aggregated approaches are not relevant because the analysis should be based on the particular suppliers of a company, and empirical data specifically from these suppliers needs to be collected and used for the analyses. Another advantage of the empirical approach is that recommendations can be tailor-made for farmers, based on their individual farm management. In contrast, modelled and aggregated carbon footprints may be more useful for larger scale policy and decision making.

15.3.3 Milk

GHG emissions from milk production have been analysed in several studies, e.g. Thomassen *et al.* (2008, 2009) in the Netherlands, Casey and Holden (2005b,c) in Ireland, Cederberg and Flysjö (2004) in Sweden, Williams *et al.* (2006) in the UK, Vergé *et al.* (2007) in Canada and Olesen *et al.* (2006) and Weiske *et al.* (2006) on a European scale.

Casey and Holden (2005c) analysed two functional units for conventional and organic dairy farms in Ireland: per hectare farmland and per litre milk produced. A general trend observed was that total GHG emissions per litre output decreased with increasing farming intensity. However, it was also possible for milk from extensive farms in agri-environment schemes to have a similar carbon footprint to conventional dairies if they had animals with high annual outputs feeding on grass from a comparatively large area. The authors concluded that a move towards fewer cows producing more milk at lower stocking rates has the potential to reduce GHG emissions from dairy farming at the national scale.

An as yet unpublished study of eleven dairy farms in the UK conducted by the authors of this chapter highlighted the positive relationship between milk yield per dairy cow and the carbon footprint per litre of milk (Fig. 15.2). This is in accordance with the results of a modelling study that found a slight but significant reduction of GHG emissions per litre of milk resulting from an increase in the proportion of high yielding vs. low yielding breeds of cows (Williams *et al.*, 2006). However, increases in yield to 10 000 L could be disadvantageous due to the high feeding intensity associated with it (Foster *et al.*, 2007). The second most important predictor of the carbon footprint per litre of milk was the annual milk yield per dairy cow and the amount of nitrogen applied to soils. Other factors that have an impact on total GHG emissions include the time of year when calves are born, the proportion of forage maize in the diet of the cows, the amount of clover in their diet, and conventional or organic farming methods (Foster *et al.*, 2007). The results of these studies indicate that potential emissions reductions could

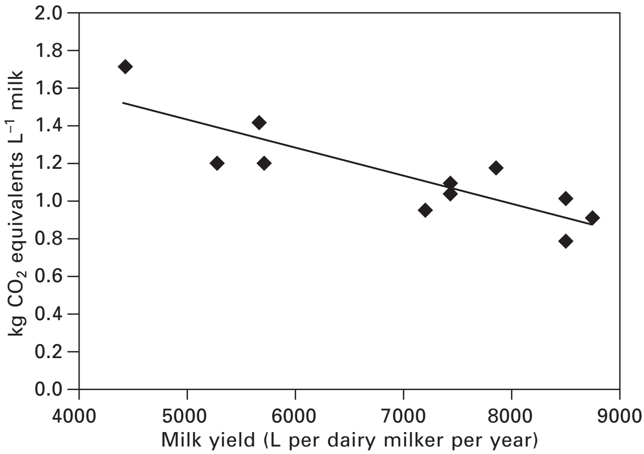

Fig. 15.2 Relationship between the average carbon footprint (kg CO_2e L^{-1} milk) and the average milk yield (L milk per dairy milker per year), $p = 0.001$. Source: Plassmann and Edwards-Jones, unpublished.

be achieved by encouraging the use of breeds and crosses with annual milk yields between 7000 and 9000 litres, although further research is needed to determine the optimum milk yield before the carbon footprint increases again due to high feeding intensities necessary. Reductions in nitrogen additions can be realised through following best practice recommendations and avoiding over-application. These measures are expected to have the greatest impact on the carbon footprint of milk. There is not currently much scope to reduce emissions from the main contributor, CH_4 from enteric fermentation, although research is ongoing into feed supplements that might reduce CH_4 production in the rumen (Waghorn *et al.,* 2008; Wood *et al.,* 2009). Direct inputs, i.e. the use of fuels and electricity on the farm, had a very low share of the total carbon footprint and were not significant predictors of the GHG emissions per litre of milk, and this is why measures such as the use of renewable energy on the farm have only a very limited potential to lead to a significant reduction of emissions.

Looking at the post-farm-gate supply chain of milk, reducing waste was identified as the most important GHG reduction measure for both dairies and consumers, while retailers can achieve the greatest reductions by decreasing energy use (or changing to renewable energy) for cold storage and display (Berlin *et al.*, 2008). The importance of consumer behaviour on total life cycle emissions from food products is also evident from the next section on processed foods.

15.3.4 Processed foods

Another case study of the PCF Project (PCF Project, 2009b) analysed the carbon footprint of convenience food, namely a frozen ready-meal made from pasta, wild salmon and carrots in a white creamy sauce containing crème fraîche and dill. The life cycle stages included in the calculations were: fishing of wild salmon in Alaska, production of the vegetables, production of the dairy products; transportation of all raw materials, their storage, processing and packaging; inputs at the plant including cleaning products and disinfectants; distribution of the final product to retail; all consumption stages (consumer shopping trips, storage, food preparation and washing up); and waste disposal including recycling where applicable. The functional unit was defined as a 500 g pack of frozen product as sold. Emissions from the production of machinery and buildings were not included, with the exception of electricity where emissions from the manufacture of the necessary infrastructure were included in the calculation. The distance from the processing plant to the central distribution centre was 420 km. Modelling of the consumer shopping trip was the same as for the strawberries (Section 15.3.1), while the use phase was based on a survey of 85 participants and measurements made during cooking experiments. Processes included were storage (28 days) in an A+ freezer with a total volume of 100 litres; cooking in a saucepan with lid and added milk; washing up in a full dishwasher (energy efficiency B), using 20 litres of water; and lighting using energy-efficient light bulbs. The complexity of these assumptions highlights the difficulties in modelling average consumer behaviour. Indeed, this is the reason why some carbon footprinting methodologies and analyses (e.g. Edwards-Jones *et al.*, 2009b) do not include this life cycle stage in the calculations.

The total carbon footprint of this frozen pasta product of 1.4 kg CO_2 per 500 g pack was dominated by the raw ingredients (750 g CO_2), including their packaging and transport to the processing plant as well as the final product packaging, consumption (370 g CO_2) and processing (240 g CO_2). Amongst the raw ingredients, dairy products accounted for 14% by weight but contributed 73% of emissions (Table 15.4). During the use phase, the addition of milk while cooking had the largest contribution to GHG emissions from this stage (Table 15.5). These figures highlight the GHG intensity of milk and dairy products. In a sensitivity analysis, different consumer behaviours were modelled. Driving to the shops was an important emissions source which is obviously dependent on the size and type of car used. Preparing

Table 15.4 Percentage of different raw ingredients in total frozen ready-meal by weight, g CO_2e per 500 g pack of final product and percentage of the ingredients in total greenhouse gas emissions (Adapted from PCF Project, 2009b)

Ingredient	% contribution to final product by weight	g CO_2e per functional unit	% of GHG emissions per functional unit
Wild salmon	12	66	10
Dairy products	14	486	73
Pasta	30	63	10
Vegetables	39	42	6
Others	5	7	1

Table 15.5 Emissions from the consumer use phase in g CO_2e per 500 g pack of a frozen ready-meal. Storage was assumed to last for 28 days (Adapted from PCF Project, 2009b)

Processes during the use phase	g CO_2e per functional unit
Storage in freezer	71
Lighting/heating	13
Cooking	85
Addition of milk	135
Washing up	62
Washing up liquid	3
Water and waste water	1

the product in a microwave instead of on a cooker can reduce emissions, which is not due to differences in electricity use between the two cooking methods, but due to less milk being added when using a microwave. Emissions increase eight-fold if the duration of home storage is increased from 28 to 182 days.

15.4 Difficulties and uncertainties in calculating carbon footprints

15.4.1 Differences between methods

The results of carbon footprinting calculations depend heavily on the methodology used, as well as the choice of database and emission factors. For example, some methodologies, such as PAS 2050, include emissions resulting from land use change, i.e. the conversion of natural habitat to agricultural land, in the product carbon footprint. This can have a major impact on the results for many products, especially from tropical and developing countries. However, if land use change emissions are not included in the calculations, then the comparison of products from different countries might yield very different results (Brenton *et al.*, 2009b). The same is true for other methodological differences, e.g. the approach to allocation between co-products (economic vs. mass allocation), the inclusion or exclusion of capital inputs or loss of soil carbon through agricultural practices. Depending on the choices made during the analysis, the results may vary tremendously. This leads to a very low comparability between different published studies, and using the results of different studies to compare products for commercial advantage, policy decisions, decisions on where to source produce from, etc., should be ardently avoided.

15.4.2 Variability of input data

Empirical carbon footprints are usually related to one particular farming year. However, farming inputs may vary greatly between years due to, for example, weather conditions, pests and diseases and prices of inputs such as fertilisers.

Depending on the year of study, the resulting carbon footprint will vary, along with the intensity of input use and the yield gained. A problem arises if inputs are not used every year (e.g. lime). Empirical carbon footprints represent a snapshot in time and include only inputs used in the particular year of study. However, this approach may allow analysts to manipulate the choice of farming years towards those with lower input use, leading to a lower carbon footprint and a possible commercial advantage. The further development of carbon footprinting methodologies could consider requiring the allocation of emissions from regular, but not yearly, used inputs over the full cycle.

15.4.3 Variation in published emission factors

Carbon footprinting analyses require the use of emission factors to calculate the GHG emissions resulting from an activity or embodied in farming inputs. However, there are considerable differences in emission factors published in the scientific literature and contained in LCA databases. Usually, studies consider only one emission factor in their analyses, e.g. the emission factor contained in an underlying database when using LCA software. The approach taken in Edwards-Jones *et al*. (2009a) was to calculate ranges of carbon footprints using a variety of published emission factors to define minimum to maximum ranges. The results clearly indicated the level of uncertainty that is associated with carbon footprint calculations and emphasised the need for standardisation not only of methodology but also of relevant emission factors to enable comparison between different studies. It also illustrates that single figures given as the result of carbon footprint calculations can be misleading by masking the uncertainties associated with the calculations. This is a problem that is very relevant to carbon labelling approaches that communicate single, precise figures without revealing the underlying uncertainty. It is also important for policy and decision makers who need to be aware of the uncertainties attached to carbon footprinting results for carbon accounting to be a meaningful tool, or for companies making decisions on future suppliers or supply chain structures based on calculated GHG savings that might result from changing current practices.

15.4.4 Carbon footprint vs. other environmental impacts

Farms with extensive production systems can have high carbon footprints per unit output produced (e.g. Edwards-Jones *et al*., 2009a). For example, on livestock farms, this could be due to low stocking rates and a low production efficiency in relation to the inputs used, resulting in high GHG emissions per functional unit. This highlights the potential conflict between carbon efficiencies and other environmental objectives such as biodiversity or reduced pesticide use. Further work is needed to identify systems which can produce food in a carbon efficient manner, while simultaneously maintaining other elements of the environment. Extensive farms with high carbon footprints per unit output might have low GHG emissions per hectare of farmland (Haas *et al*., 2001) which can be important where the intensity of land use and landscape issues are being considered alongside the product-level carbon footprint. From a climate change mitigation point of view, a justifiable intensification of farming systems that produce low carbon footprint foods while at the same time releasing less suitable land to nature conservation and biodiversity protection might be advisable.

15.4.5 Data collection and analysis

The collection of suitably accurate primary data may not be a problem for European and other developed countries. However, considerable time and resources are needed to collect data that enable the calculation of accurate carbon footprints of food products from developing countries, where data on production methods and inputs may be less reliably documented. This can have an impact on the calculations as more assumptions will have to be made by the analyst. In general, it is advisable to visit the farms and collect first hand information in the country of production; however, for most commercial studies this will not be possible and assumptions that have to be made will reduce the accuracy and reliability of the results. Another problem relating to developing countries is that most detailed LCA databases containing emission factors have been developed in industrialised countries for these countries' conditions. Due to a lack of country-specific emission factors in many developing countries, analysts have to use emission factors from these databases, which is unlikely to properly reflect the situation in the developing country and might under- or over-estimate actual emissions. The implications this might have for trade from the developing to the developed world are unclear.

15.4.6 Emissions from land use change

The inclusion of GHG emissions related to the conversion of natural or semi-natural habitats to agriculture in some carbon footprinting methodologies such as PAS 2050 can have a major impact on the final results. Land use change is more likely to occur in developing countries, where much natural land still remains and agriculture is expanding, than in Europe and other developed areas, where most land use change happened long ago. It is estimated that total emissions from all sectors in Least Developed Countries (LDCs) contribute only 5% to global GHG emissions, but emissions from land use change in these countries contribute 20% of global land use change emissions (Funder *et al*., 2009). Land use change and forestry account for over 74% of total GHG emissions in LDCs (world: 18%), highlighting the importance of land use change for the carbon footprints of products from these countries, but also the mitigation potential if further economic development follows sustainable and low carbon options.

Where land use change occurs in tropical countries, these values are likely to dominate the footprint, and so their inclusion (or not) in any footprinting methodology will have a major impact on the final results of that methodology (Plassmann *et al.*, 2010). In addition, there are large uncertainties associated with the calculation of emissions related to land use change. Even though the IPCC (2006) provide detailed guidance on how to calculate direct land use change emissions, there remains significant room for error and manipulation in these calculations. Of particular worry are the large-scale aggregated descriptions of different forest types in different countries, and the uncertainty surrounding their carbon content. PAS 2050 also requires a worst-case scenario approach where the country of origin of a product is not known to the company using it as an input, i.e. emissions from land use change have to be assumed to be equal to those arising from deforestation in Malaysia. This approach is expected to stimulate more accurate data collection; however, where this is not possible, the worst-case scenario will lead to a potentially very large over-estimation of the carbon footprint.

15.4.7 'Local food' and local emissions

The concept of 'local' food is appealing to many consumers. One of the purported advantages of local food relates to reduced GHG emissions from the food chain, a claim that was based on concern over emissions from transporting food over large distances. While this may result in increased GHG emissions, a growing number of studies has now shown that the distance that food has travelled from farm to consumer, so-called 'food miles', are not a good indicator of the overall sustainability of food, or of food carbon footprints (Edwards-Jones *et al.*, 2008). As a result, recent debates have focused on the overall emissions from the entire food supply chain (e.g. Williams *et al.*, 2006; Hospido *et al.*, 2009). In addition, difficulties remain in defining what actually constitutes local food. Some 22% of consumers who responded to an Institute of Grocery Distribution (IGD) survey expected local food to have been produced within 30 miles of where they live, while others extended their notion of 'local' to country limits (e.g. England, Scotland, or Britain as a whole) (IGD, 2006). The majority of respondents, however, considered food 'local' if it was produced in the county of consumption.

When it comes to GHG emissions, another difficulty for defining local food arises when looking at the international nature of modern supply chains. For example, so-called local bread may be baked in a small village bakery in the UK, but the flour may come from grain grown in Canada. Similarly, although local ice cream may be made on a dairy farm using milk from the farm's cows, many of the inputs to the dairy farm may come from outside the locality, and it is very unlikely that any of the major inputs to an English dairy farm would be made in the same county or region, e.g. tractors, fertilisers, diesel and concentrate feed. Indeed, it is highly likely that many of these would be made either outside the UK and/or be made from raw materials derived from outside the UK. These sorts of issues raise questions about what makes local food 'local'.

The analysis presented in Plassmann and Edwards-Jones (2009) illustrates how geographically 'local' GHG emissions from food production really are, by decoupling the place of production of the food from the locality of origin of the various inputs that are used on the farm during production. The basic concept is that if a high proportion of inputs and of GHG emissions relating to the manufacture and use of these inputs are derived close to the source of production, then that system may be considered 'local'. In the converse situation, a high proportion of the inputs to a farm, and the GHG emissions related to their production, may be made on a different continent to the farm itself. In this case the food system may not be classified as 'local'.

For two case study farms, empirical carbon footprints were calculated and the most likely origin of the raw materials for the production of farm inputs identified. On-farm emissions of N_2O and CH_4 were considered truly local. By combining the origin of the inputs with details on relative GHG emissions, it was possible to highlight the spatial location of emissions at a global scale. The results showed that less than 5% of GHG emissions related to the manufacture and use of inputs (e.g. fertilisers, diesel) could be considered local (i.e. within 50 km of the farms). When on-farm emissions from soils and livestock were included, more than 50% of all emissions were local. The inclusion of emissions related to land use change in South America for the production of soya beans contained in animal feed increased the amount of non-local emissions for both case study farms. These results cast serious doubts on the validity of claiming that any food is truly local.

15.4.8 Ethical considerations

Carbon footprinting and labelling are new instruments of supply chain management and, in some cases, of regulation that may affect trade from developing counties (Brenton *et al.*, 2009a). Developing countries have little influence on the methodological development and implementation of these instruments in industrialised countries (Brenton *et al.*, 2009a); at the same time, they tend to have a set of characteristics that make their economies particularly susceptible to the introduction of carbon accounting and/or labelling of food items in more developed countries. These characteristics include: often great distances to their markets and thus a high dependence on long distance transport, often by plane if the produce is perishable; low access to high-volume, energy-efficient shipping systems; variable yields which will lead to higher carbon footprints per unit product in low-yielding years. Land use change, including deforestation and the conversion of grasslands to cropland, is concentrated in tropical and developing countries expanding their agriculture, whereas the opposite trend is observed in Europe and North America where land clearance occurred many decades and centuries ago and current overproduction may even lead to set-aside. This means that developing countries will be much harder hit by a requirement to include land use change emissions in carbon footprints than industrialised countries. At the same time, no carbon credit can be claimed under current methodologies for the carbon that is stored in perennial cropping systems such as coffee, cocoa or tea. All these considerations lead to concern that developing countries might suffer from a reduction in export opportunities if carbon footprinting and labelling in Europe and other important export destinations gain in importance. This might have a great impact on a large number of farmers in some of the poorest countries in the world; for example, it has been estimated that 1–1.5 million African livelihoods depend on fruit and vegetable trade with the UK alone (McGregor and Vorley, 2006).

If we are serious about reducing climate change, then land use change emissions and GHG emissions from flying food over large distances should be included in carbon footprints. However, sustainable development and lifestyles should also include social and ethical considerations, e.g. about reducing third-world poverty. If carbon footprints from low income countries are greater than from industrialised countries, then the importing countries should support scientific research and technological developments to improve the situation rather than just reducing the amount of product sold with high carbon footprints. GHG emissions urgently need to be reduced, but further research is needed to enable more accurate assessments of GHG emissions in developing countries, e.g. into the potential carbon benefits through sequestration and the definition of GHG emission factors relevant to local conditions. Low carbon techniques for long distance transport need to be developed, e.g. increasing sea freight to reduce air freight. Discussions about schemes which will compensate developing countries for avoided deforestation are ongoing, and the Clean Development Mechanism involves investment in sustainable development projects in developing countries that reduce GHG emissions.

15.4.9 Issues not covered by current carbon footprinting methodologies

Current carbon footprinting methodologies such as PAS 2050 ignore changes in the carbon content of soils due to farming practices. These can be considerable, but they are highly variable and are surrounded by large uncertainties, and due to a lack of precise and region-specific data they are not normally included in carbon footprints. However, a full system analysis of GHGs in agricultural production would consider these stocks and flows (Edwards-Jones *et al.*, 2008). Similarly, carbon sequestered by perennial crops such as coffee, cocoa and tea in above- and below-ground biomass is usually not included in the analyses. The reason for the current exclusion of above- and below-ground carbon sequestration is probably due to a lack of precise and region-specific data, but it is expected that its inclusion could have a significant impact on the final carbon footprint for a variety of products from agro-forestry systems.

Some carbon footprinting analyses consider only a small part of the complete food system, most often the farming stage. There is an urgent need for more empirical studies that cover the complete food system from production to waste disposal for a variety of products, and include the emissions to and from natural parts of the ecosystem such as plants and soils. Only by taking such a holistic analysis can we obtain a thorough understanding of the impacts of food production systems on the climate.

15.5 Future trends

15.5.1 Development of international standards

The World Resources Institute (WRI) and World Business Council for Sustainable Development (WBCSD) are currently developing two new standards for product and corporate value chain GHG accounting and reporting. These institutions have previously developed the most widely used standard for the measurement and management of GHG emissions at the company level (GHG Protocol). The development of these new standards is a multi-stakeholder, consensus-based process involving businesses, policymakers, NGOs, academics and other experts and stakeholders from around the world. The first draft guidelines were available in the autumn of 2009, pilot testing of the draft guidelines takes place in 2010 and the final guidelines will be complete in December 2010. It is likely that a WRI/WBCSD standard for product accounting will have widespread international uptake.

The International Organisation for Standardisation (ISO) has also started to develop a new international standard for carbon footprinting of products and services (ISO 14067), which is expected to be completed in 2011. The standard will be in two parts, quantification and communication.

The European Commission is working on developing an authoritative basis to ensure quality and coherence for Life Cycle Assessment and carbon footprinting tools in order to increase comparability of results between studies and decrease the current dependence on expertise provided by a small number of consultants, databases and contractors. The International Reference Life Cycle Data System (ILCD) Handbook and Database (European Union, 2010) are intended to improve consistency and quality assurance of life cycle inventory data, as well as the robustness of LCA studies. This is also relevant to carbon footprinting which is based on life cycle thinking.

However, even after the development of internationally agreed standards, there may still remain demand for different accounting methodologies. These could include more specific requirements that cannot be agreed internationally, leaving scope for a range of standards at the national or business level. Different schemes might also emerge and establish as a result of differing views on how to conduct the measurement of GHG emissions or through differing strategies on how to communicate these measurements to consumers.

15.5.2 Possible future developments

Carbon footprinting will probably remain an important tool by which businesses and others can understand the GHG emissions from their products and services. The exact method used to estimate the carbon footprint does not matter so much when the results are used internally. However, if they are to be communicated outside the business, i.e. to government or consumers, then comparability of methods is essential.

A convergence of methods will most likely come about through legislative means, as there is little incentive for major retailers to agree a convergence of the methods they apply to their supply chains. Indeed, the recent history of retailing initiatives has highlighted the role of differentiation, rather than convergence.

To date, carbon footprinting of food has been driven by initiatives taken by retailers, such as Tesco in the UK. This trend is likely to continue in the immediate future and one critical factor determining the development of carbon labels will be the reaction of consumers to the labels. As yet, it is unknown if consumers will preferentially purchase goods with lower carbon footprints. It is also unknown how consumers will trade off price differences in substitutable goods with differences in carbon label, i.e. will they select lower emitting goods even if the price is greater? A consideration of the impact of other labels on food in the UK (e.g. relating to animal welfare) suggests that they have very little impact on the purchasing behaviour of most consumers. Against this background, the most likely scenario in the immediate future is that carbon footprints will become a necessary step in gaining market access, where retailers will require suppliers to declare the carbon footprint of their product before the retailer will agree to stock that item. The need for retailers to have simplicity in their systems will clash with the important, but rather academic aspects of carbon footprinting that relate to variability and uncertainty, and until there is a radical change in the use of carbon labels to guide consumer decisions, it is probable that there will be continued divergence in the approaches taken by commercial and academic users of carbon footprints.

15.6 Sources of further information and advice

For updates on international standardisation of methods, the reader is referred to the websites of the International Organisation for Standardisation (ISO, www.iso.org) and the World Resources Institute (WRI) and World Business Council for Sustainable Development (WBCSD) (www.wri.org). The British Carbon Trust website can be found at www.carbontrust.co.uk. Information on the European Union's activities related to carbon footprinting and LCA is available at http://lca.jrc.ec.europa.eu.

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16

Sustainability indicators for the food supply chain

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Abstract: The chapter provides an overview of sustainability performance evaluations of a supply chain. It introduces a new methodology for sustainability assessment of food supply chains, and demonstrates it by using an example of potato supply chains in the UK. The framework identifies indicators within three dimensions of sustainability (economic, social and environmental) and applies them to stages of agriculture, food processing, wholesale, retail and catering. The framework assigns importance ratings (determined with help from an expert) for sustainability indicators using the Analytic Network Process. The chapter discusses possible applications of the framework and further trends in sustainability benchmarking.

Key words: sustainability indicators, food supply chain, sustainability performance, benchmarking.

16.1 Introduction

Recognising and considering sustainability implications beyond the organisation and across supply chains, including wider lifecycle influences of products and processes, is becoming an important element of corporate social responsibility (CSR). Stakeholders are expecting corporate responsibility to go beyond product quality and extend to areas of labour standards, health and safety, environmental sustainability, non-financial reporting, procurement, supplier relations, product lifecycle effects and environmental practices (Bakker and Nijhof, 2002; Waddock and Bodwell, 2004; Teuscher *et al*., 2006; Welford and Frost, 2006).

Nearly all Fortune Global 250 companies have established supply chain codes of conduct and the majority report on their supply chain relations. For example retailers have been working especially hard at building supply-chain compliance with various social and environmental standards and codes. Management of and reporting on supply chain risks and implications is now seen as an appropriate business response to growing demands for greater responsibility and transparency (KPMG International, 2008; Waddock and Bodwell, 2004; Teuscher *et al*., 2006).

Major retailers and brand manufacturers are often seen as focal companies within their respective supply chains and are increasingly held responsible for the environmental and social performance of their suppliers and products. They are forced to restructure supply-chain performance in relation to mounting sustainability concerns (Hughes, 2001; Welford and Frost, 2006; Seuring and Müller, 2008). If these focal companies are to assume their extended responsibility and are prepared to demonstrate accountability for sustainability implications of their operations and engage in effective management of sustainability issues, they need to measure and benchmark sustainability performance of their supply chains. However, current methodologies and frameworks for evaluation and benchmarking of sustainable supply-chain performance are not well represented in literature (Hervani *et al*., 2005). To address this gap we propose a framework to help organisations and policy makers measure sustainability performance of supply chains. The focus is on food supply chains, where sustainability issues are very prominent and sustainability performance is important for the modern food production and consumption system.

Following Stevens (1989), the food supply chain is seen as a sequence of stages that represent economic activities through which resources, materials and information flow downstream and upstream for the production of food products and services for ultimate consumption by consumers. The food supply chain is also a network of organisations, often integrated businesses encompassing several stages of production and distribution (Fine *et al*., 1996). In this chapter, we adopt a definition of the food supply chain that comprises the following stages: agricultural production, food processing, food wholesaling, food retailing and food catering; an approach utilised by the United Kingdom Department for Environment, Food and Rural Affairs (DEFRA, 2006).

Environmental, social and ethical concerns and growing negative impacts of globalised food supply chains have contributed to increased interest in evaluation of sustainability performance within product lifecycles from 'farm to folk' and the assessment of sustainability impacts of food supply chain, companies and individual food products (Marsden *et al*., 1999; Courville, 2003; Weatherell *et al*., 2003; Ilbery and Maye, 2005; Maloni and Brown, 2006; Matos and Hall, 2007). The operations of food supply chains are seen to have sustainability implications for the economy, health, development, communities and the natural environment (Marsden *et al*., 1999; Hinrichs and Lyson, 2008; Roth *et al*., 2008).

Food organisations and businesses are developing various approaches to sustainability, promoting alternative food supply-chain models and marketing specific agricultural/craft products or individual places/regions through labelling and accreditation schemes (Ilbery and Maye, 2007; Holt and Watson, 2008). Many focal companies in food supply chains (such as large supermarket retailers and brand food manufacturers and caterers) demonstrate ethical concerns through adopting and reporting on ethical and labour codes of conduct, or labelling of products that regulate social, environmental and ethical issues within their supply chains (e.g. Tesco Ethical Trading Code). In order to make sense of these schemes – for organisations to manage their food supply chains more sustainably, and for consumers to build trust in these supply chains, there is a demand for tools to help organisations and their stakeholders to audit, assess and benchmark sustainability performance of supply chains.

This chapter aims to demonstrate how the sustainability measurement framework can be applied to a food supply chain and proposes a methodology for assessing 'triple bottom line' performance of supply-chain stages using the Analytical Network Process (ANP). First, the chapter reviews principles of sustainability measurement and benchmarking and their applications in the supply chain context. Second, it presents a framework for sustainability assessment of the food supply chain and demonstrates the new methodology using 2002 data for the potato supply chain in the United Kingdom (UK). Finally, the chapter discusses future trends on sustainability indicators in the food sector and includes recommendations for further sources of advice on the subject of sustainability measurement and benchmarking of supply chains.

16.2 Sustainability indicators and sustainability benchmarking in the supply chain

Assessing sustainability performance of supply chains is not as advanced as traditional evaluation of financial, inventory, and general operations and business performance measurement. Most of the work on assessment of sustainability performance has been focused on environmental performance or a single link (or stage) in a value chain (e.g. Veleva *et al.*, 2003). Corporate environmental management systems (EMS) can be used as a tool for internal benchmarking of environmental performance (Matthews, 2003), but EMS frameworks (such as ISO 14001) require adjustment to enable effective benchmarking beyond internal operations of an organisation. Economic input–output life-cycle analysis (EIO-LCA) may also perform high level benchmarking (Matthews and Lave, 2003) and could be used by individual firms (or plants) to gauge their performance *vis-à-vis* other firms (or plants) within their own or a related industry.

Some companies, such as Sony and Philips, have tried to evaluate and benchmark environmental performance of their products (Boks and Stevels, 2003), and the results of such benchmarking can help change product and process design practices as part of environmental improvement. Generally, benchmarking is an evaluation of organisational products, services and processes in relation to best practice. This activity is devoted to improving organisational performance, quality and competitive advantage (Camp, 1995; McNair and Leibfried, 1995; Zairi and Youssef, 1995, 1996; Sarkis, 2001a; Manning *et al.*, 2008). Benchmarking could be successfully applied for purposes of sustainability evaluation and improvement.

Several tools have been developed for execution of benchmarking at various levels (either single process within a link or an entire supply chain) such as: flowcharts, cause-and-effect diagrams, radar/spider charts, and Z charts (Camp, 1995), the European Foundation for Quality Management (EFQM) business excellence model, the balanced scorecard, service quality (SERVQUAL) framework, gap analysis, the Analytic Hierarchy Process (AHP), scatter diagrams (Min and Galle, 1996; Ahmed and Rafiq, 1998), computational geometry (Talluri and Sarkis, 2001), data envelopment analysis (DEA) (Zhu, 2002), combination of dependency analysis approach and software tool (TETRAD) with DEA (Reiner and Hofmann, 2006) and the Operational Competitiveness Ratings Analysis (OCRA) (Jayanthi *et al.*, 1999; Oral, 1993; Parkan, 1994).

Sustainable development indicators are widely used in industry and are popular with private and public bodies at various levels. Developed frameworks for analysis of sustainability parameters in a supply chain usually cover economic and environmental dimensions (e.g. Faruk *et al.*, 2001) and to a lesser extent incorporate the three dimensions of sustainability (economic, environmental and social), as pointed out by Seuring and Müller (2008) in their review of sustainable supply-chain management frameworks. However, the three dimensions of sustainability have seen some integration into supply-chain analysis for a number of years (New, 1997; Kärnä and Heiskanen, 1998; Sarkis, 2001b).

There is a growing demand for methodologies and tools to implement performance analysis across supply chains for benchmarking purposes (Hervani *et al.*, 2005). Yet, some challenges arise from the difficulty of measuring performance across organisations, for example due to non-standardised data. Other challenges arise from the difficulty of tying performance results to one particular party in a multi-tiered supply chain. Finally, measuring sustainability performance itself raises challenges.

16.2.1 Triple bottom line benchmarking

The major trends sustainable indicator creation have been: the construction of aggregate indices (such as ecological footprint and environmental sustainability index); formation of headline indicators; and the emergence of goal-oriented indicators such as Millennium Development Goals Indicators. Significant work has been completed on development and application of sustainability indicators (Bell and Morse, 1999; Pintér *et al.*, 2005). Many sustainability indicators target country or firm level of analysis.

Sustainability indicators may take on a number of perspectives, sometimes depending on the definition of sustainability. One such definition and indicator categorisation is the triple bottom line. The triple bottom line accounting of business operations refers to the assessment of corporate implications for 'planet, people and profit'; it has received a lot of consideration within business and industry (Elkington, 1997). Triple bottom line accounting aims to measure and balance economic, social and environmental aspects of organisational performance. The concept extends from the sustainable development debate as it captures three dimensions of sustainability. It has been widely applied to reporting practices within the industry and is promoted by voluntary initiatives such as the Global Reporting Initiative and AA1000 Assurance Standard. Many organisations now use the triple bottom line as a basis of their sustainability reports (Kolk, 2004; KPMG International, 2008).

There is extensive literature on assessment of sustainability impacts of food production, concentrating on effects of single or several stages of the food supply chain, although not many analyse the entire extent of the food supply chain from agricultural production to retail. The studies assign various boundaries of assessment (supply chain, production system, country or region) and focus on different units of assessments (single food commodity or food product, production system, or several food products) (Faist *et al.*, 2001; Courville, 2003; Biffaward, 2005; Collins and Fairchild, 2007; Van Hauwermeiren *et al.*, 2007). With reference to food supply chains, the focus of many sustainability assessments has been traditionally on agricultural production (McNeeley and Scherr, 2003; Filson, 2004); however, there are many assessment frameworks that incorporate stages of food processing, food retailing and transportation (Heller and Keoleian, 2003; Green and Foster, 2005).

Various approaches have been introduced to measure sustainability of food supply chains, selecting multiple levels of analysis including regional, industrial, and firm levels. Some specific sustainability assessment frameworks developed for the food sector include:

- lifecycle assessment (LCA) of environmental impacts of food products (Andersson, 2000; Hagelaar and van der Vorst, 2002);
- lifecycle related approach to sustainability impacts (Heller and Keoleian, 2003);
- farm economic costing (Pretty *et al.*, 2005);
- food miles (Garnett, 2003; AEA Technology Environment, 2005);
- energy accounting in product lifecycle (Dutilh and Kramer, 2000; Carlsson-Kanayama *et al.*, 2003);

- material flow and energy use of food products (Faist *et al*., 2001);
- economically extended material flow analysis (Kytzia *et al*., 2004);
- ecological footprints (Gerbens-Leenes *et al*., 2002; Collins and Fairchild, 2007);
- mass balance of food sectors (Linstead and Ekins, 2001; Biffaward, 2005); and
- farm sustainability indicators (OECD, 2001).

In the United Kingdom, public bodies have produced several sustainability measures and guidelines for the food supply chain (MAFF, 1999, 2000; DEFRA, 2002a, 2002b, 2005, 2006), and the private sector has also made attempts to measure its sustainability impacts (FDF, 2002; J Sainsbury Plc, 2005; Marks and Spencer, 2005; Tesco, 2005; Unilever, 2005).

In summary, there has been an emergent set of investigations related to benchmarking and performance measurement of sustainability. Most of the research is oriented toward individual firms or processes rather than toward analysis of the entire supply chain. The efforts to measure supply-chain performance have primarily centred on economic performance such as efficiency, whilst attempts to measure sustainability mostly assess firm- or product-level performance with a strong emphasis on environmental performance. There is a significant need to measure sustainability across the supply chain incorporating economic, social and environmental performances; however, methodologies for incorporating stakeholder aspects and additional sustainability dimensions are rare. In the next section, we describe a methodology to undertake a complete assessment of the food supply chain using sustainability indicators and applying it to a product sector level, rather than a firm level. This enables comparison of stages in the food supply chain and could be applied further to benchmark food supply chains between each other.

16.3 Sustainability indicators for the food supply chain

This section outlines a methodology for assessing sustainability performance within a supply chain utilising data for the potato supply chain in the UK. We propose to use data for general industrial level analysis (that can be applied to commodities or products such as potatoes or flowers or other general agricultural products such as beef, chicken, etc.). Although strategic information can be obtained from product-level measurement and benchmarking (Wever *et al*., 2007), we use a higher level perspective for our analysis. We aim to compare stages in the food supply chain to identify problem areas, and inform and improve cooperation in the food sector for enhanced sustainability performance.

Firstly, the assessment aims to reflect the current food supply chain by including stages of agriculture, food processing, food wholesaling, food retailing and food catering, and secondly, it aims to assess the complete triple bottom line and measures the effects of the supply chain operations on three dimensions: economic, social and environmental.

Our proposed methodological framework for sustainability benchmarking of the supply chain consists of four major stages:

(i) Identification of sustainability indicators (see Section 16.3.1).
(ii) Raw data gathering and data transformation using performance rescaling (Section 16.3.2).
(iii) Data gathering and adjustment using ANP (Section 16.4.1).
(iv) Sensitivity analysis of results (Section 16.4.2).

16.3.1 Identification sustainability indicators

The proposed sustainability indicators were identified on the basis of sustainable development objectives and principles that are applicable for the food sector. Specifically, the indicators were developed on the basis of objectives for sustainable development, outlined by the United Nations Commission for Sustainable Development (UNCSD, 1998) for business and industry, and those stated in Agenda 21 (UN, 1992) that could be applied for business and industry operations. UNCSD (1998) recognised that sustainable industrial policy and responsible entrepreneurship are at the heart of sustainable development. Industry, including the food industry, can contribute to a variety of interrelated economic, social and environmental objectives for sustainable development including the: (i) promotion of economic growth and encouragement of an open, competitive economy (economic objectives); (ii) creation of productive employment, gender equality, improvement of labour standards, increased access to education and health care (social objectives); and (iii) protection of natural environment and improvement of environmental performance (environmental objectives).

Then, appropriate criteria for measuring the progress towards these objectives were selected, followed by a final choice of indicators (see Table 16.1). Selected indicators are deliberately generic as they could be applied to various food products and compared between the stages in the supply chain. Chosen indicators enable assessment of sustainability objectives at a national level. For example, the sequence for selection of an indicator within the economic dimension could be demonstrated as follows. Economic objective of sustainable development such as promotion of economic growth could be measured by productivity within an industry at a national level. A specific indicator is selected then to measure productivity such as Gross Value Added per workforce, data for which are readily available with statistical services. Although initially, more than 50 indicators were drawn for the assessment of the food system (Yakovleva and Flynn, 2004); the number of indicators was reduced, accommodating the data collection process based on secondary sources (research reports, market reports and statistical data). Only nine indicators were selected for assessment of five stages of the supply chain, three indicators per each dimension of sustainability, amounting to 45 units of measurement (Yakovleva, 2007) (Table 16.1).

16.3.2 Data gathering and data rescaling

The second stage of the proposed methodological framework includes the collection of raw data for calculation of chosen indicators. The data were collected for the potato supply chain in the UK for 2002 from DEFRA and Office for National Statistics (see Table 16.2). Potatoes represent an important product for the UK domestic production and consumption; this product penetrates various stages in the food supply chain including fresh and processed production routes (see Fig. 16.1).

This stage of our methodological framework also involves rescaling and normalisation of data to enable analysis and comparison of the data for various stages in the supply chain. Indicators were allocated scores on a scale of 1 and 6 using linear interpolation. '0' stands for no available information,

Table 16.1 Identification of sustainability indicators

Sustainable development objective	Measurement criteria	Sustainability indicator
Economic dimension		
• Promotion of economic growth	○ Productivity	▪ Indicator 1: GVA per workforce, £ (A)
• Encouragement of open and competitive economy	○ Diversity and structure of the industry	▪ Indicator 2: Share of large enterprises, % (B)
• Changing consumption pattern	○ Reducing transportation of imported products	▪ Indicator 3: Import dependency, % (C)
Social dimension		
• Creation of productive employment	○ Employment volumes	▪ Indicator 4: Number of employees per enterprise (D)
	○ Quality of employment	▪ Indicator 5: Average wages per person per year, £ (E)
• Achieving equality	○ Gender balance at workplace	▪ Indicator 6: Female vs. male employment, % (F)
Environmental dimension		
• Reduction in resource use	○ Energy consumption	▪ Indicator 7: Purchase of energy for own consumption per enterprise, £ (G)
	○ Water consumption	▪ Indicator 8: Purchase of water for own consumption per enterprise, £ (H)
• Protection of natural environment	○ Waste disposal	▪ Indicator 9: Cost of sewage and waste disposal per enterprise, £ (I)

Table 16.2 Sustainability indicators for the potato supply chain in the UK (data for 2002) (Adapted from Yakovleva, 2007). Note: This work contains statistical data from ONS which is Crown copyright and reproduced with the permission of the controller of HMSO and Queen's Printer for Scotland. The use of the ONS statistical data in this work does not imply the endorsement of the ONS in relation to the interpretation or analysis of the statistical data

Stage of the food supply chain/Dimension of sustainability/Indicators				
Agricultural production	Units	Potato	Agriculture	Total UK economy
Economic indicators				
Number of enterprises		4581	142 840	1 619 195
Total output	£'000	544 000	15 508 000	1 948 458 000
Total output	'000 tonnes	6663	n/a	n/a
Output per enterprise	£'000	118	108	1203
Output per enterprise	'000 tonnes	1.45	n/a	n/a
GVA	£'000	n/a	7 137 000	926 275 000
Labour productivity (GVA per workforce)	£	n/a	12 976	35 600
Large enterprises	%	16%[1]	14%[2]	2%[3]
Imported products vs. domestic	%	9%	38%	n/a
Social indicators				
Total employment, average per year	people	n/a	550 000	26 000 000
Employee per enterprise	people	n/a	3.8	16.1
Average gross wages per employee (min)	£ per year	n/a	15 735[4]/3 467[5]	21 685
Male vs. female employment full time labour	%	n/a	n/a	63%
Environmental indicators				
Purchase of energy for own consumption per enterprise	£'000	n/a	n/a	n/a
Purchase of water for own consumption per enterprise	£'000	n/a	n/a	n/a
Cost of sewage and waste disposal per enterprise	£'000	n/a	n/a	n/a
Food processing	Units	Potatoes	Food & drink manufacturing	Total UK industry
Economic indicators				
Number of enterprises		60	7535	164 366
Total output	£'000	1 400 000	67 576 000	531 081 000
Total output	'000 tonnes	1940	n/a	n/a
Output per enterprise	£'000	23 333	896	3238
Output per enterprise	'000 tonnes	32.33	n/a	n/a
GVA	£'000	585 000	19 643 000	179 061 000
Labour productivity (GVA per workforce)	£	53 182	40 252	45 160
Large enterprises, turnover £5m+	%	27%	15%	7%

Table 16.2 Continued

Food processing	Units	Potatoes	Food & drink manufacturing	Total UK industry
Imported products vs. domestic	%	7%	15%	26%
Social indicators				
Total employment, average per year	people	11 000	488 000	3 965 000
Employee per enterprise	people	183.33	64.76	24.1
Average gross wages per employee	£ per year	19 273	18 193	20 635
Male vs. female employment full time labour	%	62%	70%	63%
Environmental indicators				
Purchase of energy for own consumption per enterprise	£'000	1535	634	484
Purchase of water for own consumption per enterprise	£'000	208	67	27
Cost of sewage and waste disposal per enterprise	£'000	299	133	43

Food wholesaling	Units	Potatoes	Agri-food wholesale	Total UK wholesale
Economic indicators				
Number of enterprises		880	17 218	113 812
Total output	£'000	2 245 700	70 032 000	388 989 000
Output per enterprise	£'000	2552	4067	3412
GVA	£'000	349 400	7 678 000	52 643 000
Labour productivity (GVA per workforce)	£	47 216	34 124	42 834
Large enterprises, turnover £5m+	%	13%	7%	7%
Imported products vs. domestic	%	21%	38%	n/a
Social indicators				
Total employment, average per year	people	7400	225 000	1 229 000
Employee per enterprise	people	8.4	13.1	10.8
Average gross wages per employee	£ per year	13 888	16 876	19 129
Male vs. female employment full time labour	%	71%	73%	73%
Environmental indicators				
Purchase of energy for own consumption per enterprise	£'000	75	21	161
Purchase of water for own consumption per enterprise	£'000	5	1	8
Cost of sewage and waste disposal per enterprise	£'000	18	3	16

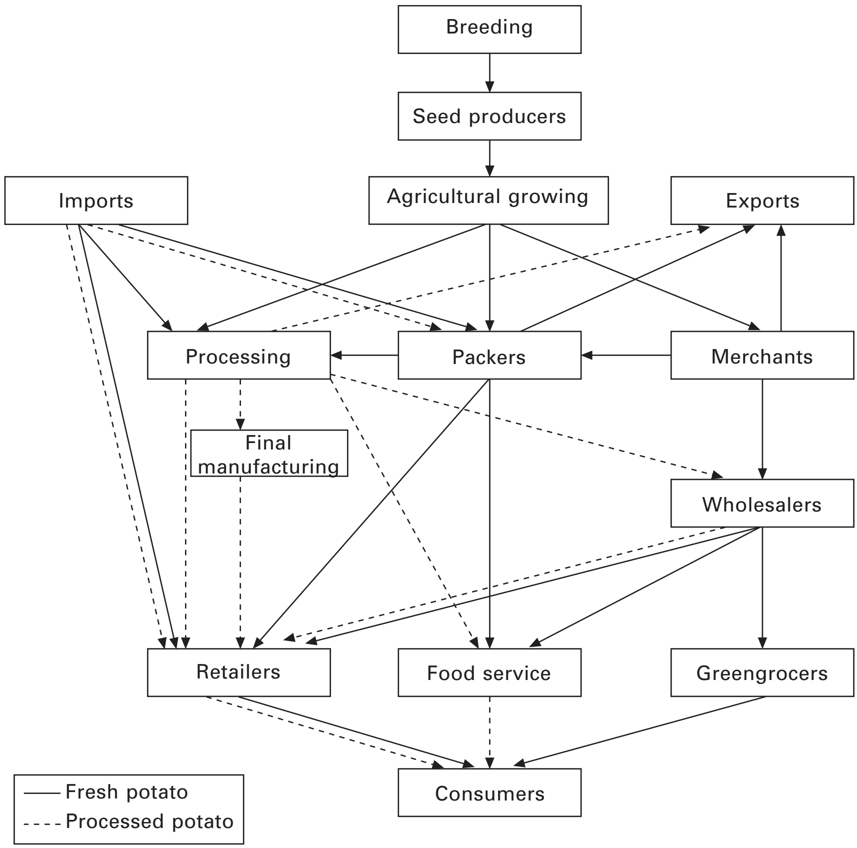
Table 16.2 Continued

Food retailing	Units	Potatoes	Food and drink retail	Total UK retail
Economic indicators				
Number of enterprises		1400	66 703	207 513
Total output	£'000	3 415 000	71 000 000	265 211 000
Total output	'000 tonnes	3338	n/a	n/a
Output per enterprise	£'000	2439	1064	1275
Output per enterprise	'000 tonnes	2.38	n/a	n/a
GVA	£'000	86 800	17 510 000	53 185 000
Labour productivity (GVA per workforce)	£	12 765	13 820	17 285
Large enterprises, turnover £5m+	%	0.2%	1%	1%
Imported products vs. domestic	%	21%	38%	n/a
Social indicators				
Total employment, average per year	people	6800	1 267 000	3 077 000
Employee per enterprise	people	4.9	18.9	14.8
Average gross wages per employee	£ per year	4840	7812	8798
Male vs. female employment full time labour	%	54%	54%	50%
Environmental indicators				
Purchase of energy for own consumption per enterprise	£'000	13	477	173
Purchase of water for own consumption per enterprise	£'000	1	32	13
Cost of sewage and waste disposal per enterprise	£'000	2	28	12
Food catering (non-residential)	**Units**	**Potatoes**	**Non-residential catering**	**Total UK economy**
Economic indicators				
Number of enterprises		8500	107 739	1 619 195
Total output	£'000	700 000	46 436 000	1 948 458 000
Total output	'000 tonnes	3141	n/a	n/a
Output per enterprise	£'000	82	431	1203
Output per enterprise	'000 tonnes	0.36	n/a	n/a
GVA	£'000	324 000	18 002 000	926 275 000
Labour productivity (GVA per workforce)	£	12 226	12 221	32 200
Large enterprises, turnover £5m+	%	1%	1%	2%
Imported products vs. domestic	%	21%	38%	n/a
Social indicators				
Total employment, average per year	people	26 500	1 473 000	26 000 000

Table 16.2 Continued

Food catering (non-residential)	Units	Potatoes	Non-residential catering	Total UK economy
Employee per enterprise	people	3.1	13.7	16.1
Average gross wages per employee	£ per year	6327	6327	21 685
Male vs. female employment full time labour	%	49%	49%	63%
Environmental indicators				
Purchase of energy for own consumption per enterprise	£'000	124	124	n/a
Purchase of water for own consumption per enterprise	£'000	22	22	n/a
Cost of sewage and waste disposal per enterprise	£'000	15	15	n/a
Total food supply chain	**Units**	**Potatoes**	**Food and drink**	**Total UK economy**
Economic				
Number of enterprises		15 421	342 035	1 619 195
Total output	£'000	8 304 700	270 552 000	1 948 458 000
Total output	'000 tonnes	6479	n/a	n/a
GVA	£'000	1 345 200	69 950 000	926 275 000
Labour productivity (GVA per workforce)	£	26 019	17 474	32 200
Large enterprises	%	11%	7%	2%
Imported products vs. domestic	%	16%	30%	n/a
Social				
Total employment, average per year	people	51 700	4 003 000	26 000 000
Average gross wages per employee	£ per year	8866	9842	21 685
Male vs. female employment full time labour	%	59%	61%	63%
Environmental				
Purchase of energy for own consumption per enterprise	£'000	437	314	n/a
Purchase of water for own consumption per enterprise	£'000	59	30	n/a
Cost of sewage and waste disposal per enterprise	£'000	83	45	n/a

[1]Potato holdings with 20 ha of land and over.
[2]Agricultural holdings with 100 ha of land and over (data from DEFRA (2003), *Agriculture in the United Kingdom 2002).*
[3]Enterprises with a turnover of more than £5m.
[4]Average wages per person per year, full-time labour.
[5]Average wages per person per year, gross wages in agriculture divided by total employment in agriculture in 2002.



Note:

- Breeding – the process of developing new varieties of potatoes;
- Seed selection – the stage where potato seeds are selected and improved for better potato production. Potato seeds are produced and later supplied to the farms.
- Agricultural growing – the stage where potatoes grow from seed to the stage of their harvesting. Potatoes are gathered and then transported to the distribution or processing stage.
- Imports – potatoes and potato products brought from abroad.
- Exports – potato products send to foreign countries for trade.
- Merchants – are engaged in exports and imports, supply for processing, packing and wholesale of potatoes at the stages of distribution.
- Packing – the stage when potatoes are cleaned, graded, weighed, packed and priced and later supplied to retailers. This stage refers to either primary processing or commonly distribution stages of the supply chain.
- Processing – the stage of value adding, such as peeling, pre-cooking, cooking, seasoning, preparation of various products.
- Final manufacturing – the stage for value adding leading to chilled production, where potatoes are used as ingredients for the preparation of soups, ready meals, salads, etc.
- Wholesale – the stage at which wholesalers acquire potatoes and potato products and distribute them amongst retailers and market outlets.
- Green grocery sale – the stage of retail through greengrocers, who are supplied by the wholesalers.
- Retail – includes supermarkets and other outlets, except for greengrocers.
- Food service – includes fast food service, restaurants, takeaways, work canteens, etc.
- Consumption – refers to household consumption of potatoes and potato products, including purchasing, storing, cooking, consuming and disposing of food.

Fig. 16.1 Potato supply chain in the United Kingdom.

score '1' reflects low benefit to sustainability and score '6' represents a high level of sustainability benefit. The scale for each indicator was developed based on general notions of a maximum desirable sustainability benefit or value and a minimum unacceptable or undesirable sustainability value. The indicator score ranges are defined in Table 16.3. The actual scores for each supply chain stage and food type are reported in Table 16.4.

If applied to a firm level, score '6' can represent sustainability targets at a firm level and within public policy context, score '6' can represent sustainability objectives or policy targets. Thus, the proposed assessment framework can be applied to monitor sustainability performance of supply chains over time either at a national level or at a firm level using policy goals or corporate sustainability targets. The framework can be used to make relative comparisons between various commodities, but most importantly can be applied to make relative comparisons between various models of supply chain configuration and methods of production (e.g. organic, slow food and conventional, etc) for same product or products produced by different supply chains (companies or retailers). If applied to a company level, the

Table 16.3 Scoring sustainability indicators (Adapted from Yakovleva, 2007)

Indicators *Mark*	0 n/a	1 Very poor	2 Poor	3 Fair	4 Average	5 Good	6 Excellent
Productivity (GVA per workforce, thousand pounds)	n/a	0	12.0	24.0	36.0	48.0	60
Market concentration (% of large enterprises)	n/a	40	32.0	24.0	16.0	8.0	0
Trade importance (import dependency, %)	n/a	100	80.0	60.0	40.0	20.0	0
Employment (employees per enterprise, number of people)	n/a	0	4.0	8.0	12.0	16.0	20
Wages (average gross wages per employee per annum, thousand pounds)	n/a	0	5.4	10.8	16.2	21.6	27
Gender balance (male vs. female employment full time labour, %)	n/a	100	90.0	80.0	70.0	60.0	50
Energy use (purchase of energy for own consumption per enterprise, thousand pounds)	n/a	1000	800.0	600.0	400.0	200.0	0
Water use (purchase of water for own consumption per enterprise, thousand pounds)	n/a	80	64.0	48.0	32.0	16.0	0
Waste (cost of sewage and waste disposal per enterprise, thousand pounds)	n/a	100	80.0	60.0	40.0	20.0	0

Note: 0 – information not available, 1 – lowest score, 6 – highest score

Table 16.4 Indicator scores for each stage of the potato supply chain (Adapted from Yakovleva, 2009)

	Indicators								
	Economic			Social			Environmental		
Supply chain stage	A	B	C	D	E	F	G	H	I
Agriculture									
Potato	0.00	4.00	5.55	0.00	0.00	0.00	0.00	0.00	0.00
Benchmark: Food production	2.08	4.25	4.10	1.95	1.64	0.00	0.00	0.00	0.00
Food processing									
Potato	5.43	2.63	5.65	6.00	4.57	4.80	1.00	1.00	1.00
Benchmark: Food and drink processing	4.35	4.13	5.25	6.00	4.37	4.00	2.83	1.81	1.00
Food wholesale									
Potato	4.93	4.38	4.95	3.10	3.57	3.90	5.63	5.69	5.10
Benchmark: Agro-food wholesale	3.84	5.13	4.10	4.28	4.13	3.70	5.90	5.94	5.85
Food retail									
Potato	2.06	5.98	4.95	2.23	1.90	5.60	5.94	5.94	5.90
Benchmark: Food and drink retail	2.15	5.88	4.10	5.73	2.45	5.60	3.62	4.00	4.00
Food catering									
Potato	2.02	5.88	4.95	1.78	2.17	6.00	5.38	4.63	5.25
Benchmark: Non-residential catering	2.02	5.88	4.10	4.42	2.17	6.00	5.38	4.63	5.25

Note: A = Labour productivity (GVA per workforce); B = Large enterprises, turnover £5m+; C = Imported products vs. domestic; D = Employees per enterprise; E = Average gross wages per employee; F = Male vs. female employment full time labour; G = Purchase of energy for own consumption per enterprise; H = Purchase of water for own consumption per enterprise; I = Cost of sewage and waste disposal per enterprise.

development benchmarking framework could assist consumers to evaluate sustainability performance of equivalent product lines.

16.4 Application of analytical network processing (ANP) to sustainability scores

16.4.1 Adjustment of sustainability scores using ANP

The next stage of our methodological framework is the most intricate. The values in Table 16.4 represent adjusted scores based on ranges as defined in Table 16.3. This rough estimate may not be adequate as it does not consider the relative importance of each of these factors with respect to each other, nor does it consider the interrelationships amongst various factors and indicators. To further this methodology we introduce a weighting scheme based on multi-attribute rating technique, ANP, to more accurately represent the performance of these actual supply chains.

ANP is a generalised form of the multi-criteria decision making technique, the Analytical Hierarchy Process (AHP) (Saaty, 1980). ANP offers a solution to scoring methods (Sarkis and Sundarraj, 2000). In the context of sustainability, the complexity of evaluating sustainability and assigning scores arises from multiple relationships and interlinkages amongst the sustainability factors within and between the sustainability dimensions (Sarkis, 2003). ANP modelling is a method that can incorporate interdependencies amongst factors and indicators included in the sustainability evaluation through utilisation of pairwise comparisons made by decision makers. The pairwise comparisons used as the inputs to ANP can allow sustainability evaluators to integrate the perception of relative importance amongst sustainability factors or parameters. ANP can structure the sustainability factors in a hierarchical (or network) relationship and thus help evaluators to assign weights for sustainability factors in the performance evaluation exercise (following Dou and Sarkis, 2008).

For this sustainability assessment, a general ANP model is constructed (illustrated in Fig. 16.2) that considers the relationships and interrelationships amongst a variety of sustainability factors such as:

(i) Interrelationships amongst the general sustainability factors or sustainability dimensions (external interdependency). For these relationships we can argue that economic factors are influenced by both social and environmental factors; and the social factors are influenced

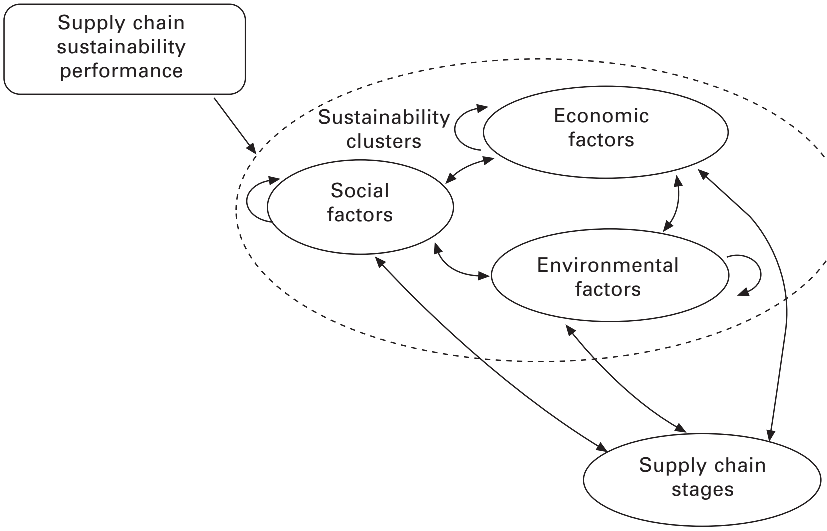


Fig. 16.2 A high-level schematic of the ANP network decision model for evaluating a supply chain's performance.

by the environmental and economic factors, etc. These relationships are shown in Fig. 16.2 by the double-arrowed lines that go between the clusters of factors.

(ii) Within each sustainability dimension, there is an internal interdependency between sustainability factors or indicators. For example, for environmental factors there are influencing relationships amongst factors of Water Consumption, Energy Consumption and Waste Generation (similar to the interdependencies of the general sustainability factors). We can evaluate these interdependencies and they are represented by the 'looped' arcs on each of the general sustainability factors.

(iii) In the hierarchical structure, the relative importance of the three general clusters (sustainability dimensions) influences the overall objective (sustainability performance evaluation of the supply chain), which is the goal of this model. This relationship is represented by the arrow from the objective to the overall cluster. Relative importance weights will also be determined for these general clusters.

(iv) There are also relative importance weights for each of the sustainability factors within their respective sustainability dimensions. These are not shown on the high level diagram but appear in the initial supermatrix (see Table 16.5) in the last nine rows of the supermatrix underneath columns labelled 'Env', 'Social' and 'Eco'.

(v) There are hierarchical representations of the supply chain stages' influence on each of the general sustainability dimensions and the influence of each of the specific sustainability factors on each of the supply-chain stages. These relationships are represented by the double-arrowed lines between the supply chain stages and sustainability factors.

For this study, we determine relative importance weights partly using opinions of an expert with an in-depth knowledge of the potato supply chain in the UK and partly using our opinions as an illustrative example. It is important to mention that the view of experts on sustainability issues in the supply chain is significant in determining the relative importance weights, which affects the final scores for the selected indicators and the overall index. Therefore, we selected a knowledgeable specialist with a substantial experience on sustainability aspects of the potato supply chain. As part of the weight evaluation process, a questionnaire was developed. An excerpt from the full questionnaire is shown in Table 16.6. All questions in the questionnaire are formulated as pairwise comparisons and are used to construct pairwise comparison matrices. These pairwise comparison matrices are used to determine the relative weights for the factors that are compared.

Pairwise comparison questions (105) are used to fully acquire the information for the three clusters of sustainability factors, each with three sub-factors, for the five stages of the food supply chain. For example, with respect to the first level of interrelationships in the ANP mode, the following three questions were posed:

Table 16.5 Initial supermatrix for ANP network decision model

	Obj	Env	Social	Eco	Agri	Proc	Whole	Retail	Cater	EnCon	WatCon	Waste	Employ	Wages	Gender	LabProd	Markcon	ImpDep
Obj	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Env	0.177	**0.500**	**0.084**	**0.375**	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Social	0.304	**0.084**	**0.500**	**0.125**	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Eco	0.519	**0.417**	**0.417**	**0.500**	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Agri	0.000	0.535	0.233	0.078	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Proc	0.000	0.264	0.342	0.302	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Whole	0.000	0.035	0.041	0.033	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Retail	0.000	0.134	0.218	0.466	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Cater	0.000	0.032	0.166	0.121	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
EnCon	0.000	**0.785**	0.000	0.000	0.567	0.319	0.778	0.333	0.333	**0.500**	**0.250**	**0.375**	0.000	0.000	0.000	0.000	0.000	0.000
WatCon	0.000	**0.066**	0.000	0.000	0.323	0.460	0.111	0.333	0.333	**0.417**	**0.500**	**0.125**	0.000	0.000	0.000	0.000	0.000	0.000
Waste	0.000	**0.149**	0.000	0.000	0.110	0.221	0.111	0.333	0.333	**0.084**	**0.250**	**0.500**	0.000	0.000	0.000	0.000	0.000	0.000
Employ	0.000	0.000	**0.761**	0.000	0.715	0.460	0.742	0.633	0.701	0.000	0.000	0.000	**0.500**	**0.375**	**0.375**	0.000	0.000	0.000
Wages	0.000	0.000	**0.191**	0.000	0.218	0.221	0.203	0.304	0.204	0.000	0.000	0.000	**0.375**	**0.500**	**0.125**	0.000	0.000	0.000
Gender	0.000	0.000	**0.048**	0.000	0.067	0.319	0.055	0.063	0.095	0.000	0.000	0.000	**0.125**	**0.125**	**0.500**	0.000	0.000	0.000
LabProd	0.000	0.000	0.000	**0.701**	0.701	0.429	0.685	0.685	0.726	0.000	0.000	0.000	0.000	0.000	0.000	**0.500**	**0.417**	**0.084**
Markcon	0.000	0.000	0.000	**0.097**	0.202	0.143	0.234	0.234	0.198	0.000	0.000	0.000	0.000	0.000	0.000	**0.125**	**0.500**	**0.417**
ImpDep	0.000	0.000	0.000	**0.202**	0.097	0.429	0.080	0.080	0.076	0.000	0.000	0.000	0.000	0.000	0.000	**0.375**	**0.084**	**0.500**

Note: In grey – weights determined by potato supply chain expert, in bold – weights determined by the authors.

Table 16.6 Extract from the questionnaire on comparative importance of sustainability indicators in the food supply chain

On the scale of one to nine please rate the significance of one issue over the other issue. Please mark with X one of the nine boxes provided for each answer.

No.	Questions	Extremely less important	Very much less important	Less important	Slightly less important	Equal	Slightly more important	More important	Very much more important	Extremely more important
		1	2	3	4	5	6	7	8	9
1	*In terms of SUSTAINABILITY OF THE FOOD SUPPLY CHAIN*									
A	How significant are environmental factors when compared to economic factors?									
B	How significant are environmental factors when compared to social factors?									
C	How significant are social factors when compared to economic factors?									
2	*In terms of their ENVIRONMENTAL IMPACT*									
A	How much more important are agricultural activities when compared to food processing activities?									
B	How much more important are agricultural activities when compared to food wholesale activities?									
C	How much more important are agricultural activities when compared to food retail activities?									
D	How much more important are agricultural activities compared to food catering?									

Table 16.6 Continued

No.	Questions	Extremely less important	Very much less important	Less important	Slightly less important	Equal	Slightly more important	More important	Very much more important	Extremely more important
		1	2	3	4	5	6	7	8	9
E	How much more important are food processing activities when compared to food wholesale activities?									
F	How much more important are food processing activities when compared to food retail activities?									
G	How much more important are food processing activities when compared to food catering activities?									
H	How much more important are food wholesale activities when compared to food retail activities?									
I	How much more important are food wholesale activities when compared to food catering activities?									
J	How much more important are food retails activities when compared to food catering activities?									
3	*In terms of their SOCIAL IMPACT*									
A	How much more important are agricultural activities when compared to food processing activities?									

B	How much more important are agricultural activities when compared to food wholesale activities?									
C	How much more important are agricultural activities when compared to food retail activities?									
D	How much more important are agricultural activities compared to food catering?									
E	How much more important are food processing activities when compared to food wholesale activities?									
F	How much more important are food processing activities when compared to food retail activities?									
G	How much more important are food processing activities when compared to food catering activities?									
H	How much more important are food wholesale activities when compared to food retail activities?									
I	How much more important are food wholesale activities when compared to food catering activities?									
J	How much more important food retails activities when compared to food catering activities?									

- How much more important is the influence of social factors on economic factors when compared to environmental factors in the food supply chain?
- How much more important is the influence of economic factors on environmental factors when compared to social factors in the food supply chain?
- How much more important is the influence of environmental factors on social factors when compared to economic factors in the food supply chain?

The responses were represented on a 1–9 Likert-type scale with a '1' response representing the 1/9 value for standard AHP, meaning extremely less important, and a '9' response meaning extremely more important. Table 16.5 reports the importance ratings derived from the responses of a potato supply chain expert (highlighted in grey are the weights determined by the potato expert and in bold are the weights determined by the authors).

Using these numbers as inputs, ANP determines the relative importance weights of each of the factors. The relative importance weights are calculated from each set of pairwise comparisons. An example pairwise comparison matrix comparing the relative importance of each of the sustainability factor groups, environmental, social, and economic, on the overall benchmarking exercise is shown in Table 16.7. The results of this pairwise comparison matrix show that economic factors (0.519) represent the greatest importance on the supply chain performance on sustainability by this decision maker. The relative importance is followed by social factors (0.304), then by environmental factors (0.177).

Each of these relative importance weights computed by a pairwise comparison matrix is then used to populate the initial supermatrix. The supermatrix is used to generate the final weightings after all the interdependencies, and relationships amongst the factors are integrated. The results of the example pairwise comparison matrix from Table 16.7 are shown as a vector of three weights in the first column of Table 16.5, under the 'obj' heading. After completing populating the supermatrix, we then have to make it 'column stochastic'. That is, the supermatrix is computed by normalising the summation of all the weights in a column to a sum of 1. The next step is to arrive at a convergent (stable) set of weights. One way of arriving at a convergent set of weights is to raise the matrix to a sufficiently high power where the

Table 16.7 Pairwise comparisons and ratings of general sustainability clusters on the overall objective

Cluster	*Environmental*	*Social*	*Economic*	Importance rating
Environmental	1	1	1/5	0.177
Social	1	1	1	0.304
Economic	5	1	1	0.519

scores are no longer changing to a specified number of decimal places. For our example, we stopped when the weights stabilised to the 10^{-4} power.

The final converged ANP scores for the potato supply chain are displayed in the converged supermatrix in Table 16.8. Highlighted in bold in the grey area are the global weights for each of the sustainability factors (indicators) that sum to 1. Final sustainability indicators are computed by weighting the indicator scores reported in Table 16.4 by the global ratings of Table 16.8 for each stage in the potato supply chain (see Table 16.9).

16.4.2 Sensitivity analysis

As a final stage of the proposed supply-chain sustainability indicator framework, a sensitivity analysis can be performed to evaluate the robustness of the obtained weights. To evaluate the sensitivity of the final values or relative influence weights of the various sustainability factors, a simple perturbation approach may be applied. That is, one vector of weights within a supermatrix (usually an influential vector such as the overall sustainability dimension weights) can be selected. The perturbations may occur by changing the weight structure of the vector. Many approaches may be used. One extreme approach is to give all the weight within a vector of weights a given factor and then calculate the converged weights of the supermatrix. This process then can be repeated for each factor within a vector. For example, initially we give all the weight 1.000 to the economics factor from the three major sustainability grouping factors and determine the final scores. Then we can see what happens to these final scores when we shift the full weighting to the environmental factor, and so on. An alternative mechanism is to change the weights over a range of 0 to 1 for a given factor in a vector, while the relative importance ratio of the other factors remains constant. The process will require recalculation of the converged supermatrix for each point within that range.

After determining the relative importance of the sustainability factors (indicators), the hierarchy of sustainability factors according to their weights in descending order is as follows: (i) market concentration; (ii) labour productivity; (iii) employment; (iv) import dependency; (v) wages; (vi) energy use; (vii) water use; (viii) waste; (ix) employment gender ratio. According to the opinion of the potato expert, the economic dimension of sustainability has a larger weight (0.5191) than the social (0.304) and environmental (0.177) dimensions.

Since sustainability factors for each stage have the same weights, we can compare the sustainability performance according to these factors between the stages in the supply chain. According to the final (weighted) sustainability scores, considering that we have no complete data for the stage of agricultural production, the stage of food wholesaling scored the highest in terms of sustainability performance with a sustainability index of 4.6, followed by the stage of food retailing (index of 4.3) and the stage of food

Table 16.8 Converged supermatrix for ANP network decision model

	Obj	Env	Social	Eco	Agri	Proc	Whole	Retail	Cater	EnCon	WatCon	Waste	Employ	Wages	Gender	LabProd	Markcon	ImpDep
Obj	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Env	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Social	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Eco	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Agri	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Proc	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Whole	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Retail	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Cater	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
EnCon	**0.105**	0.217	0.072	0.086	0.125	0.125	0.125	0.125	0.125	0.375	0.375	0.375	0.000	0.000	0.000	0.000	0.000	0.000
WatCon	**0.105**	0.217	0.072	0.086	0.125	0.125	0.125	0.125	0.125	0.375	0.375	0.375	0.000	0.000	0.000	0.000	0.000	0.000
Waste	**0.070**	0.145	0.048	0.057	0.083	0.083	0.083	0.083	0.083	0.250	0.250	0.250	0.000	0.000	0.000	0.000	0.000	0.000
Employ	**0.131**	0.079	0.245	0.081	0.143	0.143	0.143	0.143	0.143	0.000	0.000	0.000	0.429	0.429	0.429	0.000	0.000	0.000
Wages	**0.113**	0.068	0.212	0.070	0.124	0.124	0.124	0.124	0.124	0.000	0.000	0.000	0.371	0.371	0.371	0.000	0.000	0.000
Gender	**0.061**	0.037	0.114	0.038	0.067	0.067	0.067	0.067	0.067	0.000	0.000	0.000	0.200	0.200	0.200	0.000	0.000	0.000
LabProd	**0.142**	0.081	0.081	0.198	0.114	0.114	0.113	0.114	0.114	0.000	0.000	0.000	0.000	0.000	0.000	0.341	0.341	0.341
Markcon	**0.144**	0.082	0.082	0.201	0.115	0.115	0.115	0.115	0.115	0.000	0.000	0.000	0.000	0.000	0.000	0.346	0.346	0.346
ImpDep	**0.130**	0.075	0.075	0.182	0.104	0.104	0.104	0.104	0.104	0.000	0.000	0.000	0.000	0.000	0.000	0.313	0.313	0.313

Note: In grey – global weights for each of the sustainability indicators that sum to 1.

Table 16.9 Weighted sustainability scores for each stage in the potato supply chain

Indicator/Stage	Agriculture	Processing	Wholesale	Retail	Catering
EnCon	0	0.105	0.591	0.624	0.565
WatCon	0	0.105	0.597	0.624	0.486
Waste	0	0.070	0.357	0.413	0.368
Employ	0	0.786	0.406	0.292	0.233
Wages	0	0.516	0.403	0.215	0.245
Gender	0	0.293	0.238	0.342	0.366
LabProd	0	0.771	0.700	0.293	0.287
Markcon	0.576	0.379	0.631	0.861	0.847
ImpDep	0.722	0.735	0.644	0.644	0.644
Total	1.298	3.759	4.567	4.306	4.040

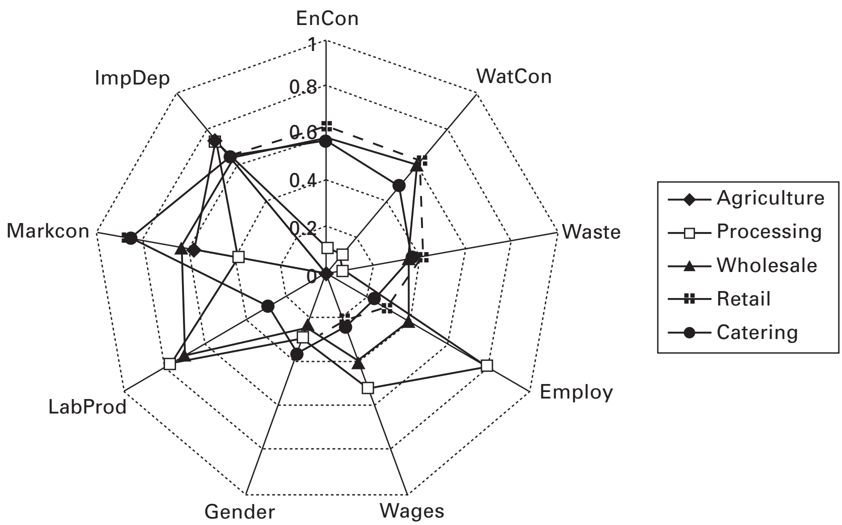


Fig. 16.3 Weighted sustainability factors for the potato supply chain.

catering (index of 4.0) (see Table 16.7). The higher the score (maximum of 6), the better the stage is performing in terms of sustainability within the three dimensions economic, social and environmental as determined by the range of scores in Table 16.3. The final scores for each supply chain stage are illustrated in a spider diagram (see Fig. 16.3). This method includes the interrelationships between the sustainability dimensions and sustainability factors (chosen sustainability criteria) within their respective sustainability dimensions. An advantage of this scoring and weighting scheme is that we can arrive at a single sustainability index score for each stage and compare the stages between each other. Policy makers or supply chain managers seeking to improve performance should see what aspects of a particular food supply chain stage make it more sustainable.

The overall sustainability index of the potato supply chain is 3.594, and is an arithmetic mean of five indices for the potato supply chain stages. As the stage indices already reflect the interrelationships between stages and sustainability factors, there is no need for weighting supply chain stages when computing the overall supply chain index. For further applications of the proposed assessment method, the calculation of an overall sustainability index for the entire food supply chain could be useful for benchmarking different food supply chains or production models.

The method uses statistical data for the food supply chain, in combination with expert opinion, to construct an overall index of sustainability. In this chapter we utilised the opinion of a potato expert together with the authors' opinion; however, for further application of the method, the opinion of several experts on particular supply chains could be utilised. Since we constructed and ranged indicators between 1 and 6, where score '6' is the desirable sustainability performance, we can say that the closer the overall sustainability score to score '6', the closer is the supply chain stage to conforming to set sustainability objectives or targets within the three dimensions of sustainability.

16.5 Future trends

Potential users of the framework may wish to consult stakeholders when selecting sustainability indicators for the assessment, and consult them on what would be the desirable sustainability values before ranging the indicators from 1 to 6. Furthermore, potential users (such as policy makers and individual organisations) may set the maximum scores as planned targets for sustainability performance (either policy targets or individual corporate performance targets) and use the framework to measure supply chain performance over time or between product lines. The higher the score, the closer the supply chain overall is to achieving sustainability targets or maximum set desirable sustainability values within three dimensions: economic, social and environmental. The framework can be used to make relative comparisons between various commodities, but most importantly can be applied for comparison of various configurations of the supply chain. In this study we used three dimensions of sustainability; however, more themes or dimensions could be utilised for the development of sustainability indicators.

Reporting on supply chain relations in the food sector has increased; large supermarket chains now publish sections on supply-chain operations in their sustainability or corporate social responsibility reports, and place similar information on corporate websites (see for example, Tesco's policy on Responsible Buying and Selling on Tesco's corporate website and CSR report). Monitoring, measuring and reporting on sustainability effects of supply chains will be growing as the demand for regulation of supply chain relations is increasing.

Since supply-chain relations are now seen within the merit of sustainability, CSR and corporate citizenship, various concepts will be applied to the formulation of supply-chain relations and their monitoring. We have applied the triple bottom line concept to measuring sustainability performance in the supply chain. Other concepts for evaluation of performance in the supply chain could be applied that may cover more aspects of sustainability or CSR, such as ethical dimensions, organisational effectiveness, human rights, animal welfare and so on. Since the use of ethical, social and environmental labelling is growing, there will be an increasing need for consumers to find their way through these claims.

The development of sustainability indicators needs to take into account the relative importance of sustainability measures and trade-offs between sustainability dimensions or individual sustainability factors. Moreover, since various groups perceive sustainability differently, it is important to involve stakeholders in developing sustainability measures for the supply chain, their importance, ranges and metrics.

16.6 Sources of further information and advice

16.6.1 Assessments of environmental and social impacts of food production and distribution

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16.6.2 Sustainability reporting standards

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Part IV

Environmental management in the food industry

17

Establishing an environmental management system in the food industry

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Abstract: This chapter is about controlling and improving environmental performance at the site or business unit level, and in particular about using formal management systems to do that. Drawing largely on experience in the UK, it begins by looking at the incentives for allocating resources to environmental management before describing the steps involved in putting an environmental management system in place. The benefits and drawbacks of various approaches are discussed and the pros and cons of external audit and certification to standards such as ISO14001 and EMAS are considered. Some factors for successful implementation of an environmental management system in a food business are identified.

Key words: environmental management, food industry, environmental management systems, EMS.

17.1 Introduction

Drawing on several connected definitions in ISO 14001 (ISO, 2004), a working definition of an environmental management system (EMS) can be established:

> *An environmental management system comprises inter-related elements – encompassing organizational structure, planning activities, responsibilities, practices, procedures, processes and resources – used to establish policy and achieve objectives relating to the interactions between an organisation's activities, products, and/or services and the environment.*

The environment, according to ISO 14001, constitutes '*the surroundings in which the organisation operates, including air, water, land, natural* *resources, flora, fauna, humans and their interrelation*'; these surroundings extend from '*within an organization to global systems*'.

Seen in this way, any structured set of endeavours a business undertakes to control and reduce the effects on the environment of its own activities, or of all or parts of the value chain in which it operates, can be construed as an environmental management system. (We use the term value chain rather than supply chain here to recognise that some businesses can influence not just their suppliers but their customers too.)

The earlier chapters of this book focus on Life Cycle Assessment (LCA). LCA evaluates the environmental effects of the operation of entire production–consumption chains, usually chains for individual products. Life-cycle management (LCM) is a term often used to describe structured endeavours aimed at reducing the environmental impacts of these production–consumption chains based on the outcomes of LCA(s). Environmental management systems (EMS) are, on the other hand, structured endeavours aimed at reducing the environmental impacts associated with the operation of particular organisations or individual operating sites. Environmental management systems are normally based on the outcomes of an environmental audit (or environmental review) of the site's or organisation's operations (see Section 17.4.2 of this chapter).

Despite these differences of focus, there are many ways in which Life Cycle Assessment and life-cycle management can usefully be used in the context of environmental review and environmental management systems (and *vice versa*). Thus, environmental management systems might be useful mechanisms to deliver objectives within life-cycle management programmes. For very large businesses in the food sector (or indeed in any sector) that have a significant degree of control over operations throughout their value chain, LCA represents a powerful tool that can be used to shape environmental management efforts at the corporate level (see, for example, Taylor and Postlethwaite, 1996). But the EMS – the subject of this chapter – is most commonly encountered as an organisational arrangement applying to a single site or a business unit encompassing several sites carrying out similar activities. Such management systems often adopt the general format set out in ISO 14001, and it is with these systems that this chapter is primarily concerned. As a result of their scope, and perhaps also of the knowledge base of the individuals implementing them, the role of LCA in these systems is often small, even non-existent.

17.2 Drivers for and benefits of implementing environmental management systems (EMS) in the food industry

In many food-sector businesses (and indeed in many businesses in other sectors) systematic control of the business' interactions with the environment comes relatively late to the list of management tasks. The forces pushing 'Environment' onto that list can usefully be split into four categories:

- direct regulatory pressure
- internal pressure for broader risk control (itself often a result of increasing scrutiny of the environment by public bodies and stakeholders)
- pressure to improve the efficiency with which resources are used
- direct pressure from stakeholders other than regulators

Considering these in a little more detail, it is evident that they have been strengthening for some time and many look set to strengthen further.

17.2.1 Direct regulatory pressure

In Europe, food businesses have had to handle regulations relating to individual environmental issues for many years, but this could often be done in a rather piecemeal way. To provide some examples from the UK, compliance with waste management legislation introduced in the Environmental Protection Act of 1991 chiefly involved record-keeping to establish an audit trail in case of need, while responding to the Packaging Directive (European Directive 94/62/EC) (ED, 1994) was, and remains, largely a data-collection exercise for many businesses, which can be carried out by one or a few people. But the introduction of the UK's Climate Change Levy (2000) provided some new impetus for structured energy management – an essential component of any EMS. These regulations imposed a tax on energy use in commerce and industry, for which a rebate could be obtained by operators in energy-intensive sectors if efficiency objectives were met. The EU Emissions Trading Scheme (EUETS, European Directive 2003/87/EC) (ED, 2003) then captured some larger, more energy-intensive food processors; although not the strongest driver for change in its early form, this provided extra impetus for structured energy management.

The implementation of European Directive 96/61/EC (ED, 1996), introducing Integrated Pollution Prevention and Control, brought the need for 'whole-site' environmental control to many food processors for the first time and also to larger, intensive pig and poultry farms. With regulation under IPPC covering releases to all media and the need to demonstrate the application of BAT (Best Available Techniques), sites cannot realistically operate under IPPC without a formal EMS of some kind. Although the smallest food manufacturing sites are excluded from this regulatory regime, as successful businesses expand, new entrants to the regime appear all the time.

On farms other than the intensive units mentioned above, the need to comply with environmental protection legislation has encouraged or necessitated the development of procedures for handling wastes and other polluting materials such as slurry, spent sheep dip, packaging and other non-biogenic wastes; but in most cases there appears to be no requirement for these to be consolidated in an environmental management system, in the sense of the definition advanced at the beginning of this chapter.

17.2.2 Internal pressure for broader risk control

The need to control environment-related risks has perhaps been the biggest driver for the implementation of environmental management systems in businesses not captured by environmental permitting regimes. This need grew (in the UK at least) in the 1990s, as the penalties for causing pollution became larger with the introduction or clarification of anti-pollution legislation (one element in the implementation of the 'Polluter Pays' principle). Whole milk, for example, has a high chemical oxygen demand (220 000 mg/L), so if it enters streams or rivers, it reacts with dissolved oxygen, effectively causing other life in the same water body to suffocate. Both dairies and other food processors have been fined over pollution incidents involving milk and milk products (cream and butter have even higher COD levels than milk). Offending businesses are normally required to pay the costs of clean-up and environmental rehabilitation, on top of the fine; these costs are often significantly greater than the value of the fine itself.

Many environment-related risks are of course related to direct costs rather than the potential costs of incidents: as waste management has become more closely regulated, so the cost of waste disposal has risen. In the UK, tax is a large component of the total costs of waste disposal to landfill (the 'default' destination for mixed wastes). Before the 2006 budget, the standard rate of landfill tax was £21 per tonne; landfill tax in early 2009 stood at £32/t and increased to £48/t in April 2010. Effluent treatment costs rose, too, in the UK, as regulation of the water industry tightened and the standards imposed on effluent discharge quality rose to meet regulatory requirements, e.g. from the European Urban Wastewater Treatment Directive (1991).

17.2.3 Pressure for increased resource efficiency

In any industrial activity, raw materials are transformed into products, using energy. Water is often needed, and ancillary activities (which consume materials and energy for the provision of services such as cleaning or compressed air needed in the production facility) are often essential. The pursuit of resource efficiency is simply about reducing the resources used to produce a unit of product. Since all inputs cost money, this brings opportunities to cut operating costs – always a key issue for suppliers to major retailers, and a concern to any established enterprise. Resource efficiency (or 'waste minimisation') programmes have delivered significant benefits to firms in the food sector over the course of some years (see Fig. 17.1 for examples). In many cases, introduction of an environmental management system provides the framework needed for the structured pursuit of this kind of opportunity.

HJ Heinz, at a site in the North West of England, developed a means of removing oil and grease contamination from steam condensate emerging from canned food cooker/sterilisers. Condensate recovery was improved and savings of £50 000 per annum were made (source: Atkins, 1994).

Taypack Potatoes, a potato packing and distribution company based in Scotland introduced a number of *resource efficiency measures*. It now *recycles* 95% of its waste; has reduced power consumption by 30%; and using *satellite tracking* technology to manage its fleet of potato collection lorries more efficiently saved £100 000 per year on fuel (source: Netregs, nd).

Natures Way Foods, a supplier of salads, reduced costs by £65 000 per year through segregating wastes and diverting them from disposal to recycling, adopting re-usable transit packaging. Auditing wastes prompted the company to find ways to increase overall product yields (source: Envirowise CS620, nd).

One EuGeos client producing ready-meals reduced absolute water consumption by 20% and specific water use (water consumption per kg product) by 35% over 6 months following introduction of internal water monitoring and low-cost water-efficiency measures.

Fig. 17.1 Some examples of waste minimisation in the food sector.

17.2.4 Non-regulatory stakeholder pressure

What some call 'private regulation' (the imposition of standards by major customers – large food retailers in the case of food processors and producers) is reported to be a growing incentive for control of environmental risks in the food sector. The strength and coverage of this force inevitably reflect the priorities of these customers rather than those of the business itself; these priorities, in turn, reflect a range of stakeholder interests. Voluntary agreements between sectors (with retailers often in the vanguard of action and measurement) and governments are a formal manifestation of efforts to address a range of stakeholder concerns through pursuit of agreed environmental performance targets. These often require businesses further back in retailers' supply chains to act, so that retailers can meet their own targets – for instance on issues such as packaging reduction. The Courtauld Commitment (WRAP, nd) and the Food Industry Sustainability Strategy (DEFRA, 2006) in the UK provide examples of voluntary agreements at the sector level.

Eco-labelling and other environmental certification schemes represent another form of private regulation; in this case, one into which businesses enter directly, rather than at their customers' instigation. Not all eco-labels require environmental management systems, but the LEAF (Linking Environment and Farming) Marque scheme, a UK environmental certification and labelling scheme for farms, does require participants to implement a formal EMS.

ISO 14001 and the European Eco-Management and Audit Scheme (EMAS) are, of course, environmental certification schemes dedicated to environmental management systems, and we return to them later. But other good-practice accreditation schemes in the food industry encourage participants to implement some aspects of an EMS. So both the Seafish (UK Sea Fisheries Authority) organisation's Responsible Fisheries Scheme (see http://rfs.seafish.org/about), in which fishing vessels can gain certification for implementing good operating practice, and the Global Standard for Food operated by the British Retail Consortium (BRC) require some environmental controls by operators.

These different types of private regulation appear still to be growing and, although as they grow some consolidation occurs, it is likely that their importance in driving environmental improvement along the food chain will increase too.

17.3 Food industry context

17.3.1 Food industry diversity

To discuss the implementation of any management system in food industry businesses in general, is challenging, given the diversity of those businesses. At one end of the scale are businesses operating large continuous process facilities such as grain spirit distilleries, flour mills or liquid-milk dairies, while at the other are small farms and foodservice outlets run by single individuals or families. In between is an array of enterprises of differing sizes; some making one product and some making one hundred; some largely mechanised and some oriented towards manual operations. Particular considerations apply to the implementation of environmental management on farms. Farming is, after all, an activity in which humans manipulate the environment to produce food in a controlled manner – management of the environment perhaps, rather than environmental management in the sense of this chapter. Farming can have unintended or undesired consequences for the wider environment, and these can be controlled systematically following the principles described here. But for advice on how to apply these in the farming context, the reader is directed to the more specialised sources of advice listed at the end of this chapter (Section 17.8).

17.3.2 Management systems in the food industry

To introduce an environmental management system effectively into any business, it is necessary to take account of existing management systems, both formal and informal. Complete integration of these systems is not necessary, but if environmental management is to become part of everyday practice, its design needs to take account of that practice. Almost all food businesses will have existing formal systems of some kind, at least to ensure food safety and in many cases to ensure product quality too.

Food safety systems

HACCP (Hazard Analysis Critical Control Point) has become very widespread as an approach to managing food safety. The UK Food Standards Agency (FSA) introduces HACCP thus:

> *'[HACCP] focuses on identifying the 'critical points' in a process where food safety problems (or "hazards") could arise and putting steps in place to prevent things going wrong. This is sometimes referred to as "controlling hazards". Keeping records is also an important part of HACCP systems'* (FSA, 2006.)

There are many parallels between the process of setting up a HACCP system and establishing an EMS – an environmental review fulfils a similar role to a HACCP hazard analysis; critical control points and significant environmental aspects are similarly situated as foci for action; and performance monitoring, record-keeping, corrective action and review are important elements in both HACCP systems and EMS.

Quality management systems (QMS)

It is well-known that in the context of quality systems, 'quality' means fitness for purpose rather than excellence, that purpose in this context is synonymous with customer requirements, and that customer requirements can be translated into specifications. Quality systems, i.e. management systems aimed at ensuring that products meet requirements as set out in specifications, are also widespread in food businesses. These are nearly always built on the 'plan-do-check-act' philosophy (PDCA, made popular by Deming) that also underpins both HACCP systems for food safety and environmental management systems.

In many cases, quality systems meet the requirements of ISO 9001 (ISO, 2008), which broadly means that they:

- include a quality policy
- identify a senior management representative with overall responsibility for quality
- define responsibilities for other staff whose work affects the quality of products and identify a management representative to co-ordinate the quality management programme
- ensure that appropriate resources are allocated to meeting quality objectives and running the QMS
- have certain aspects and procedures documented, including the control of documents
- incorporate monitoring and control measures
- contain provisions for internal audits and management review.

These basic requirements are echoed in ISO 14001 (ISO, 2004), the environmental management systems standard, and thus are echoed in EMAS, the EU's Eco-Management and Audit Scheme, since it requires an ISO 14001-compatible EMS. The PDCA philosophy (where P includes 'assess the current situation') is thus common to the most widespread forms of food safety system, quality system and environmental management system.

17.4 Implementing an EMS

17.4.1 Introducing an EMS: widening the scope of existing systems

It can be seen from the previous discussion that one attractive route to implementation of an environmental management system (EMS) in a food business is to extend the scope of existing systems. There are a number of factors favouring this approach:

- It is more likely that policies and programmes relating to different business issues form a coherent whole and are not mutually contradictory if they all sit within one framework.
- Document control is easier if all key documents are together.
- At a very practical level, process operators need to have just one reference document telling them how to perform a task (whether it is called a procedure or a work instruction), so integration here is almost essential.
- If the existing systems are indeed used as a management tool (businesses, even well-run businesses, do exist in which formal management systems serve essentially as decoration, used primarily to meet the expectations of visitors), then 'environment' is less likely to encounter total neglect once the initial impetus for environmental management has been removed and the first flush of enthusiasm has passed.

But there are some disadvantages too, of which the chief one is perhaps 'dilution'. This can hinder management of both the longer-established issues and of the new, environmental ones. In other words, bringing a number of new issues into one management system can lead to the emphasis on critical issues such as food safety being reduced, or to the environmental issues being 'crowded out' if they have to compete with more immediate concerns.

A compromise seems to work well in many cases: thus integration of environment as far as possible into documentation generated for other systems always seems better than creation of a stand-alone set of dedicated environmental system documents. But creating dedicated space for structured discussion of environmental performance and initiatives – often a monthly meeting – seems to keep environment in view better than making it the last agenda item for every general management meeting. Inevitably, different companies find different approaches work best.

The actual scope of an EMS may include:

- Pollution risks which might cause immediate breaches of legislation, local nuisance or harm, and/or longer-term damage to the environment – particularly soils and groundwater.
- Control of permitted emissions to air, water and land (regulated and un-regulated).
- The handling and fate of solid and liquid wastes produced by the business.

- The types of fuels used, sources of water used by the business and the efficiency with which these are used.
- The nature and origins of raw materials – not just those used in products but also packaging materials, materials used for cleaning, and ancillary materials such as lubricants, water treatment chemicals, etc.

Introducing such a system to a business that has no environmental programme in place, implies changing the scope of management attention, as shown in Fig. 17.2. Clearly, with such a potentially broad new agenda, implementing

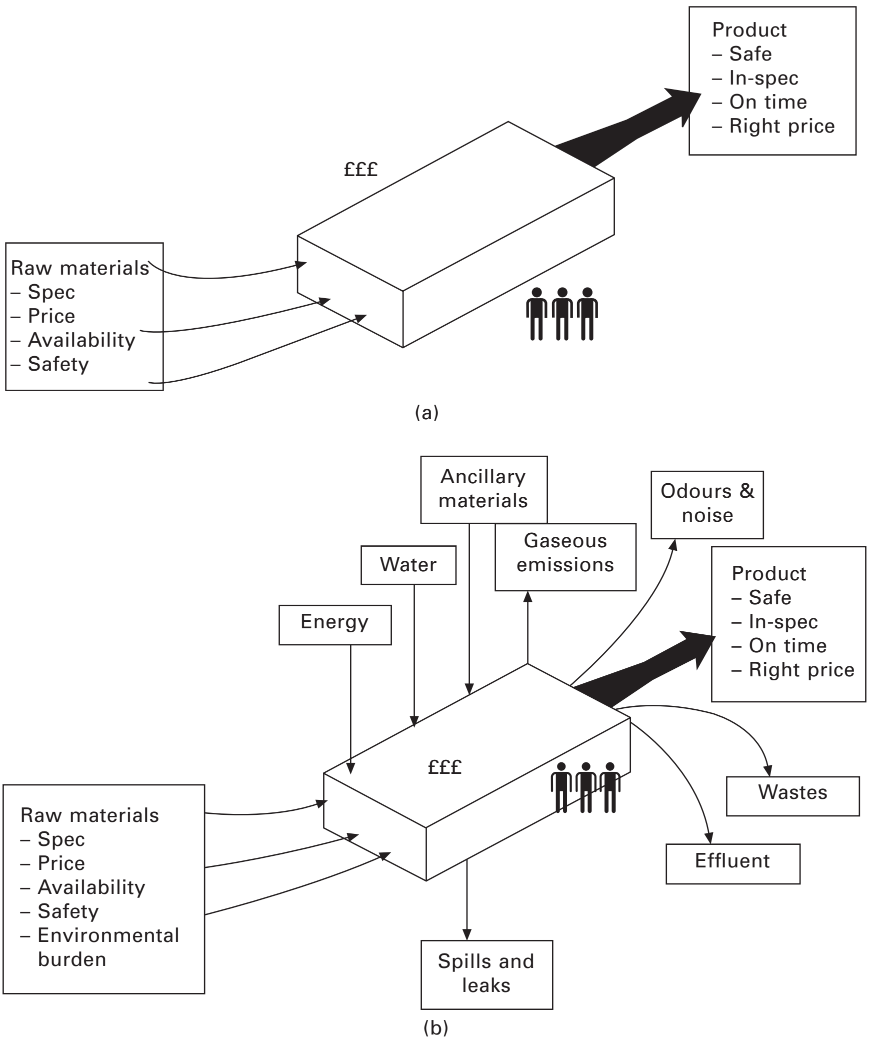

Fig. 17.2 Changing priorities on introduction of an environmental management system

an EMS requires prioritisation among these various issues, and this is discussed in the next section.

17.4.2 Environmental review

The 'environmental review' or initial environmental audit, is the entry point to managing the environmental issues that are most relevant to an individual business. It is useful to distinguish two stages to this process:

(i) identifying relevant issues and linking them to particular activities within the business (termed 'environmental aspects' in ISO 14001) through cause–effect relationships;
(ii) establishing the relative significance to the business of the various issues and of the different activities driving each one.

In systems certified to ISO 14001, the second stage needs to include consideration of the extent to which each issue is under the control of the business itself; it is part of the management process and must be documented. The coverage of the environmental review required by EMAS is stipulated in its governing Regulation, and includes not just consideration of environmental aspects and applicable environmental regulations, but also the assessment of significance, a review of existing, relevant management practices, and an evaluation of previous environmental incidents.

But even allowing for the formats imposed by EMAS and ISO 14001 on businesses interested in ultimately obtaining third-party certification of the EMS (of which more in Section 17.4.4), this initial environmental review can take different forms. Four methods are worth considering for food businesses:

(i) A review that adopts the format of a hazard analysis under HACCP. In this case, the auditor (who can be a member of staff or an external specialist environmental auditor) follows the process through and identifies environmental 'hazards'. Once the prioritisation stage has been done, Critical Environmental Control Points can be established in an analogous way to CCPs in the HACCP system. This approach is likely to lead eventually to an EMS that is very risk-oriented, and very site-centred. To be comprehensive, the boundaries of the 'process' need to be drawn widely – storage of ancillary materials needs to be within the 'process', for example.
(ii) A review that starts from a list of environmental themes or issues, such as that in Section 17.4.1, eliminates those that are not relevant to the business or individual parts of it, identifies cause–effect relationships for those that are relevant, and measures those that will enable Stage (ii) – prioritisation – to be carried out. A review adopting this format should be comprehensive. For larger sites and businesses, it makes sense to consider the whole entity as a collection of smaller units when conducting the review: the division can be department-based or activity-

based, or a combination of the two. When doing this, however, it is important not to miss activities that cut across department boundaries or serve many purposes within the overall process: compressed-air supply and machinery maintenance in general are good examples of environmentally-relevant activities that are easily missed.

(iii) A review that begins with compilation of an inventory of inputs (materials, energy and water) and outputs (wastes, aqueous discharges, emissions to air and/or land), connecting them to unit processes or activities within a facility. A separate assessment of the risks associated with abnormal operations (occasional events) and accidents is an essential second element in this form of review. The ranking of events in the risk assessment and benchmarking of the emissions against industry best-practice and/or environmental quality standards then provides the basis for the prioritisation stage. This is very close to the format of environmental assessment required by the UK's environmental regulator (the Environment Agency) for facilities applying for permits under the IPPC regime.

(iv) Life Cycle Assessment provides another form of environmental review that can be used as a starting point for an EMS. As noted at the very beginning of this chapter, LCA is more useful for this in some contexts than in others, with very large businesses or businesses with a great deal of sourcing flexibility the most likely to benefit from its application for EMS development. Unilever carried out an overall assessment of this form in the 1990s (Taylor and Postlethwaite, 1996) which allowed the major impacts of the business to be identified and further work to be prioritised.

A fine-grained LCA with a great deal of internal process detail and data would be needed to enable prioritisation of issues within a food-processing site, and even then some further risk assessment would likely be necessary. Normalisation of impact assessment results from the LCA could be used to inform the prioritisation activity.

Where LCA is used as the entry point to environmental management, the resulting system is likely to be very product-oriented. In some businesses, a combination of high-level LCA and site-based HACCP-analogous review(s) offers considerable promise. But risk assessment is a more familiar tool than LCA to many managers in the food industry, so adopting one of the more risk-oriented approaches to environmental review described above often not only makes local business sense but also allows the new territory of environmental issues to be explored with a familiar map-making tool.

Much guidance has been produced (some examples are noted in Section 17.8.3) about prioritising the environmental impacts and risks – and the activities that generate them – identified in an environmental review. In ISO 14001, this process is not strictly part of an environmental review at all, but the subject of a procedure within the management system (assessment of significance, in ISO 14001 parlance) which at least ensures that it is reconsidered periodically.

There are so many different ways of prioritising environmental issues and environmentally-relevant activities ('environmental aspects') that work well in businesses of different types that it would be unhelpful to recommend one alone. Scoring systems, rating by committee and decision-trees all work. Within the prioritisation process, it is wise to consider how much control the firm has over each aspect. It is also important that the process awards high priority to aspects that are critical to legal compliance. The most important criteria for choosing a prioritisation method are that it should be as simple as possible, understood by as many people as possible, and reproducible. It should be an aid to getting things done, not a swamp that hinders progress. In many food businesses, if the review has been sufficiently thorough, the priorities for action will be relatively obvious – as long as it is remembered that 'actions' can be aimed at maintenance as well as change.

Overall then, the environmental review assesses performance and risk: weaknesses are brought out in the open and areas with the greatest scope for improvement identified.

17.4.3 Next steps: process development or action?

The next steps to be taken in implementing an EMS also depend on the nature of the business. One approach is to follow the layout of ISO 14001 (ISO, 2004), which provides more detail on the 'Plan' phase of the PDCA cycle. This suggests that the next moves are to set environmental objectives and targets and to establish a structure for environmental management that reaches down through the organisational hierarchy. Once that structure is in place, delivery of the objectives and shorter-term targets can be negotiated by those within it, along with the resources for their delivery – i.e. the 'Act' phase begins. In fact, the structure of ISO 14001 implies that the compilation of an environmental policy should precede both the review and these further planning activities. But since the standard requires the policy to be '*appropriate to the nature, scale and environmental impacts*' of the organisation, writing the policy before a review has been carried out is likely to result in its very early revision. In reality, it makes sense to compile an environmental policy in parallel to setting environmental objectives.

In smaller, flatter and/or nimbler organisations, it is possible, and usually desirable, to abbreviate this process. Broad objectives are first established for the priorities identified in the review, for example: 'improve energy efficiency', 'reduce risks of land pollution from drains', etc. Practical measures that could contribute to the achievement of these objectives are then identified and the more promising ones pursued. This might be done by a dedicated environmental team, or through existing business management processes. For example, in some companies, the very fact that energy intensity is being monitored ensures that it is on the agenda of regular management meetings and therefore that efforts will be made to improve – no designation of a specific 'energy efficiency representative' or negotiation of this year's energy efficiency target is needed.

The danger of following the first route is that the environmental management process is developed so far that it gets in the way of environmental action. The danger of the second is that too little process is developed to ensure that all significant issues continue to be covered over an extended period. But these pitfalls can be avoided if companies:

(i) Look at other formal management systems and their use before implementing an EMS, consider what the 'management process' underlying those is, and design the process elements of the EMS to follow that. This involves asking two questions: 'How do these systems work, in this business?' and also 'How do we use these systems?'
(ii) Evaluate existing formal procedures, consider their relevance to the environmental priorities identified in the review, and amend relevant procedures so that they are aligned with those priorities. So if one environmental priority in a food processing facility is to reduce loadings in aqueous effluent prior to its treatment, it is essential that cleaning procedures specify that wastes should be swept up or scooped off the floor before any wet cleaning commences. At this level, as has already been noted, the merging of management systems is essential.
(iii) Commit to an environmental 'management review' of the type described in ISO 14001 at least once per year. This should revisit the priorities identified in the initial environmental review, consider how they are being addressed, and evaluate progress or its absence. It should also set some direction for the period ahead.

In addition, every environmental management system must have a strand that keeps abreast of, and informs decision-makers in the business about, environmental regulations. This is, of course, one element that distinguishes an EMS from a QMS.

17.4.4 Training

A separate chapter of this book is devoted to training, so no in-depth coverage is provided here. Training is, however, an essential element of any environmental management system: there is no point writing procedures if people are not trained so that they are in use; it is impossible to inform other managers about environmental regulation without some training in the workings of that regulation; and so on. The training needs of different people within a business with respect to the environment soon become apparent: most need some basic awareness of the company's position and of certain fundamental rules, like where to put wastes of different kinds. Others need specific training related to their normal functions; for example, storekeepers may need training in ways of handling chemicals that minimise the likelihood of spills. It is normal to train some individuals to respond to minor incidents such as oil spills. Those maintaining the environmental management system, or leading environmental initiatives, usually need more specialist training.

17.4.5 Environmental controls

One role that formal procedures play in an EMS is ensuring that 'the way things are done' complies with regulatory and non-regulatory requirements. If formal procedures (or work instructions) do effectively describe how people must carry out certain tasks in the workplace, then procedures for what could be termed 'compliance-critical' activities need to be designed to prevent environmental accidents and offences, to minimise the risk of incidents, and to avoid operating practices that cause nuisance or offence to neighbours. 'Compliance-critical activities' encompass waste handling, effluent plant operation, the operation of processes that generate direct emissions to air (including odorous emissions which are widespread in food processing operations), handling of cleaning chemicals in areas drained by surface-water drains, and so on.

Process controls built or programmed into equipment contribute to the overall set of environmental controls, as does site infrastructure. Thus high-level alarms in tanks at plants handling milk (or other polluting liquids), preferably linked to automatic shut-offs on filling valves, are important to preventing spillages caused by overfilling. But equally, the design of site drainage systems can make the difference between staff being able to contain spilt material somewhere on site – which leads to some subsequent local inconvenience – and that material running straight into the nearest river or stream – which means an offence has been committed. Maintenance of infrastructure, environmentally-relevant equipment and environmentally-significant process control systems are important. At sites with formally-structured plant maintenance programmes, these and the EMS need to interconnect – even if they do not actually overlap. At the least, if a site has an engineering manager, he or she needs to be engaged in the key elements of the EMS – initial review, establishment of controls and improvement programmes, and management reviews.

Performance targets can be set that allow the effectiveness of environmental controls to be monitored. Some relevant performance indicators are noted in Section 17.4.8.

17.4.6 Incident preparedness (environmental crises)

Making preparations to deal with the environmental consequences of accidents is another essential part of any EMS. Once again, in most businesses this requires an extension of the scope of existing preparations, for example those covering fires and floods. To do this, an existing plan, centred around accidents with implications for human health, needs to be extended to cover:

- the environmental consequences of accidents already within its scope. For example on a site with a large ammonia refrigeration plant, preparations to tackle a major ammonia leak should take into account the potential effects of ammonia on environmental receptors beyond the boundary fence.
- new accidents with primarily environmental consequences. Spills of chemicals or liquid foodstuffs into surface drains fall into this category.

Preparations need to encompass training and informing staff, obtaining and maintaining appropriate short-term response capacity (spill kits are a common element in this), and ensuring that crisis management reference documents point crisis managers to the relevant external bodies (environmental regulators, suction tanker hire firms, etc.).

17.4.7 Improvement programmes

Improvement programmes should be at the heart of an environmental management system. Indeed, they are almost its *raison d'être*, since a management system that yields no improvement cannot justify its own existence. ISO 14001 and EMAS both require the implementing organisation to have a formal environmental policy that includes a commitment to continual improvement; so for management systems compliant with these standards, improvement programmes are necessary to deliver that commitment.

Although the wording of the EMAS Regulation is that objectives and targets shall be established '*at each relevant function and level within the organisation*', in practice environmental improvement programmes are often organised in an issue- or topic-specific way. So it is common to build into an EMS one or more elements of an energy-efficiency programme, a water-efficiency programme, a pollution-risk reduction programme, a waste-reduction programme, and so on. Which ones will depend on which issues were identified as priorities in the environmental review: given the failure of measures aimed at preventing biodiversity loss in Europe, a programme aimed at supporting biodiversity might well be a major element within any farm's environmental management system.

Each improvement programme will likely cover a number of environmental aspects. Here again, if the review was thorough, some priority aspects will be apparent from the outset. But more detailed investigations are often needed before the activities can be spotted where most potential for change exists. For example, the exact drivers of water use are often poorly-understood in food processing sites: cleaning and steam generation are known to account for a large proportion of consumption in most sites, but metering at end-use locations or even at entry points to sub-systems is surprisingly rare. In such cases, a topic-specific audit carried out over a short time period (which could be one week or three months) provides vital information. External specialists represent a useful resource for conducting such an exercise on larger sites: in these circumstances, it is necessary to form a small internal team to co-ordinate the exercise and make use of its results. In a small food-service outlet on the other hand, a detailed water audit might just involve reading the water meter two or three times per day – to establish the amounts of water used in major cleaning activities, for example.

The biggest drivers of water use or energy use are not necessarily those that offer the most potential cost-effective improvement. Sometimes, the major aspects are so obvious that they are well-managed already, so it is important not to overlook opportunities to make big changes in activities of medium importance.

For waste, although reduction at source is usually the preferred improvement path (waste is, after all, a material that has been paid for but is not being sold as product), improving waste management by reducing landfill and increasing recycling or recovery has an important part to play. The options available for recovering food waste were severely curtailed (at least in the UK) following the BSE scare, but improved composting technology and the development of anaerobic digestion facilities have improved prospects.

Just as described for environmental control, in situations where formal procedures (or work instructions) do effectively describe how people do certain tasks in the workplace, they can play an important part in the implementation of performance improvement programmes. If the way that a task is carried out is important to the success of an improvement programme, then the relevant procedure needs to be checked – and amended if necessary. Naturally, any amendments need to be accompanied by suitable training.

The results of LCAs can be valuable in environmental improvement programmes. One improvement pathway open to a business or site is to try to reduce the environmental 'intensity' of the raw materials it uses. The results of LCAs for different raw materials, which may be published as 'environmental product declarations' (EPDs), provide sound information for the selection of raw materials on environmental grounds.

17.4.8 Monitoring and review

Without some overall monitoring element, and without some periodic review of the effectiveness and suitability of arrangements, environmental management is not 'systematic'. Monitoring within the EMS can take various forms: monitoring of performance indicators is key to deciding whether improvement programmes are achieving the desired effect. Absolute and relative performance indicators should both be used. Common indicators used in many food processing businesses are:

- total energy use and specific energy use (energy use per unit of production or unit of input material)
- energy-associated emissions in carbon dioxide equivalents

- total water consumption and specific water consumption (differentiated by source type if a mixture of mains water, groundwater and/or surface water is used)
- overall material yield
- packaging used per unit of product
- refrigerant release
- incident and complaint numbers
- cleaning chemical volume
- waste volumes and proportions of waste sent to disposal, recovery or recycling.

More specific performance indicators may be needed to monitor the progress of individual improvement programmes: changes in overall water use do not reveal whether the amount of water used for cleaning has been reduced. If a site water improvement programme includes an objective to achieve the latter, then arrangements must be made to measure and record the relevant water flows. Indicators of this kind, and practical measures to collect the relevant data, should be established as part of the improvement programme itself.

Of course some of the indicators listed above, such as yield, have relevance beyond environmental management and are often in use as business performance indicators before the EMS comes into being. Most of these indicators are most suitable for use at site or business-unit level, and capture relatively local effects. But recording energy-associated greenhouse gas emissions provides some measure of the more remote (in organisational rather than geographic terms) impacts of power generation, and the influence on these of local changes such as switches between fuel types. In the language of the Greenhouse Gas Protocol (WRI/WBCSD, 2004) this is covering Scope 2 as well as Scope 1 emissions. Capturing the more remote effects of switching raw materials (i.e. effects within the GHG Protocol's Scope 3, whether relating to GHG emissions or to other types of environmental impact) involves reference to LCA or EPD information, or information about specific practices in upstream organisations. A food processor might, for instance, use the proportion of ingredients produced on LEAF-certified farms as a measure of its progress in promoting biodiversity at the primary production stage of its product chain, or assess the relative merits of different cleaning chemicals using EPDs covering their production and the treatment of post-use residuals in typical wastewater treatment plants.

If product life-cycle assessments are subsequently carried out by the business operating the EMS or its agents, performance measurement undertaken within an EMS can provide valuable data. This is particularly so in multi-product sites when it becomes necessary to link a proportion of site inputs to a single product (or category of product).

The next section of this chapter deals with a second dimension of monitoring – that of monitoring the 'performance' of the EMS itself. Before turning to that, one final but important element of any EMS should be mentioned – what EMAS and ISO 14001 refer to as 'management review'.

This element must be present if any EMS is to have continuing relevance over time. Without checking that the management system is appropriate in form for the organisation and delivering results, there is a strong danger that it will fall into disuse through sheer irrelevance. Looking at the period of time since the last similar event, the management review should consider the following questions:

- Have we achieved what we set out to achieve through applying the EMS? *Performance monitoring summaries or records are needed for this.*
- What do we want to achieve next through applying the EMS?
- Is the environmental review still a review of *this* organisation?
- *This is a question of checking that the scope of the review still represents the organisation.*
- Are the elements of the EMS appropriate to the organisation as it is now and to what we want to achieve?
  The results of management system audits are useful in answering this.

17.4.9 Corrective action

Corrective action is needed at two places in the EMS. The first is as part of – or immediate follow up to – the response to incidents. If incidents happen, it is important that steps are taken to reduce the likelihood of recurrence. Food businesses normally have a corrective action mechanism to respond to food safety or product quality issues and it makes sense to use the same mechanism to follow up environmental incidents. It is simply necessary to make sure that the latter will get to the starting gate of the corrective action mechanism.

The second place for corrective action is after the management review. If the organisation has changed significantly, it will be necessary to revise the environmental review or conduct a new one. Even if the scope is still correct, it may be useful to amend the review to incorporate further evidence that has emerged as improvement programmes have been implemented. For example, if a detailed water monitoring programme has been undertaken as part of a water-efficiency project, the results of that monitoring could be used to update the environmental review. Similarly, if the management review concludes that the elements of the EMS are no longer all needed, then corrective action is required to remove the redundant ones and introduce relevant new ones.

17.5 Auditing environmental management systems

The effectiveness of any management system is monitored by an audit process. It is the detailed component of the 'Check' phase of the PDCA cycle and informs the management review. The function of audit is the same whether the management system covers environment, food safety, quality or anything else. Readers are directed elsewhere for general information about auditing as a skill (ISO, 2002 – currently being reviewed). Auditing environmental management systems requires some knowledge of environmental issues. Many courses are available to provide auditors accustomed to auditing non-environmental systems with the relevant environmental knowledge: the website of the UK Institute of Environmental Management and Assessment lists providers in the UK and other countries (www.iema.net). It is important to distinguish between an internal audit process and an external audit process because the two have different purposes.

17.5.1 Internal audit

The purpose of an internal audit is to check that the management system is working, in the sense that it is doing what it sets out to do. It is at this point that the environmental policy shows its real value. In general, environmental policies make bland reading and, since EMAS and ISO 14001 impose specific requirements on their content, provide little basis for differentiating organisations from each other. However, an environmental policy does (or should) set out the overall objectives of the EMS: it therefore provides the frame of reference for the internal audit.

The internal audit process can be carried out by auditors from within the organisation, or by external auditors hired for the purpose. In large organisations with extensive management systems, it can be conducted as a planned sequence of local audits, spanning the entire cycle between management reviews. In smaller organisations with slim management systems, it can be a single exercise carried out just before the management review. In the first case, it is important that the reporting of local audit results connects to the organisation's corrective action mechanism, so that failings can be rectified promptly.

17.5.2 External audit

External audit has a different purpose: external audit involves checking that the EMS meets a set of externally-established criteria for environmental management systems. The two most common sets of criteria are ISO 14001 (ISO, 2004) and the European Commission's Eco-Management and Audit Scheme (EMAS), but others exist. Criteria with particular relevance to smaller organisations were developed by the British Standards Institute and others as BS 8555 (BSI, 2003) and in Wales as the Green Dragon Environmental Standard™, while the Chemical Industries Association's (CIA's) Responsible Care programme had its own criteria for EMS.

On the definition implied by the previous paragraph, external audits can also be undertaken by individuals within the organisation or by specialists from outside. Certification audits are a particular form of external audit in which suitably-accredited auditors from outside bodies check that the management system meets specified, published criteria and issue a certificate to attest to that fact. A thorough certification audit checks that both the system meets the relevant external criteria and that it is working as it is intended to. Many food businesses will be familiar with certification audits in the context of schemes to assure food hygiene or good food manufacturing practice; for example, the British Retail Consortium's 'Global Standard for Food'.

17.5.3 EMS certification: cost or benefit?

EMAS and ISO 14001 provide valuable guidance on the design of environmental management systems. It is possible to implement an ISO 14001 system without having it certified by an external body, however. Certification has certain potential benefits:

- Access to some sales, since certain customers require suppliers to be certified to ISO 14001 or EMAS or an equivalent.
- Marketing advantage in some situations; for example, tenders in which extra credit is given to certified tendering organisations.
- The certifier's visit provides a deadline by which the EMS must be given attention. This 'deadline incentive' is particularly important as a means of setting priorities for management attention in some businesses.
- The certifier's visit also provides an external expert's perspective on the organisation's environmental management effort, which can be helpful in developing it further. Outside specialists conducting internal audits or non-certification external audits can provide this too.
- Reduction in the level of scrutiny the business receives from regulators in some jurisdictions. In the UK, operating a certified EMS normally leads to reduced charges under the environmental permitting scheme. The reduction will not normally cover the costs of the certification process, however.

Set against these potential benefits are the costs of the certification process – both certifying bodies' fees and staff time within the audited organisation. So before pursuing certification, any business should satisfy itself that, in the context and markets in which it operates, the benefits are indeed likely to outweigh the costs.

17.6 Success factors

The preceding sections have provided an overview of the process of implementing an EMS in a food business. Observing businesses in different sectors over time reveals management systems in all states of repair, some of them in use, some only evidenced by files gathering dust on a shelf. What then are the success factors for getting the benefits out of an environmental management system – making the effort of implementation worthwhile? The following seem particularly important, even if they are not the only factors:

- Management enthusiasm. More than just expressing commitment by signing off a policy document, senior management must believe that spending time on environmental matters is worthwhile.
- Staff engagement. By the same token, a sufficient proportion of influential members of the workforce must participate in environmental initiatives and controls for them to succeed: a system on its own achieves nothing, it is simply a tool to help people achieve things.
- A fit-for-purpose management system. Since a management system is a tool, it needs to be appropriate to the job in hand. A management system that is designed to fit the way a business already works has more chance of being used than one that is designed for a different context. This means that implementing an EMS in a business where the 'way we work' doesn't look much like the ISO 9001/ISO 14001 model does need more flexibility on the part of the system's designer.

17.7 Conclusions

This chapter has considered the implementation of environmental control and improvement programmes at the site or business unit level. Links have been noted between environmental management at this level and LCAs. LCAs can be useful as precursors to environmental management systems and as sources of information for improvement programmes within an EMS. The operation of an EMS within a business can, in turn, enable the provision of higher-quality data to LCAs of that business's products.

But another important conclusion of observing environmental management systems in practical use is that implementing an EMS does not necessarily lead to continued improvement in environmental performance, or ensure an unblemished record of compliance with environmental law. This is a specific case of a more general observation – that since management systems 'only' help managers manage better, while they enable companies to be better than they are already, they don't transform bad companies into good ones.

Despite that caveat, it seems likely that the pressure to implement environmental management systems, or to cover environmental issues within more general management systems that encompass a number of themes, will not lessen soon. If regulatory pressure does not increase, the demands of labelling schemes are likely to push more and more businesses towards implementation of some kind of EMS. If, as seems quite possible given current interest in mandatory EPD schemes, those labelling schemes are LCA-based, then data collection may well become a primary function of the environmental management system of the future.

17.8 Sources of further information and advice

17.8.1 Resource efficiency in the food sector

BREF notes: Reference documents on Best Available Techniques (BATs) for pollution prevention, prepared by the European IPPC Bureau (under the EC Joint Research Centre) http://eippcb.jrc.es/reference/fdm.html (last accessed 5 January 2010)

UNEP cleaner production guides in fish and meat processing, dairies and breweries (and other guides) http://www.unep.fr/scp/cp/publications/ (last accessed 5 January 2010)

Envirowise: UK business support service offering free advice on resource efficiency http://www.envirowise.gov.uk/uk/Sectors/Food-and-drink.html (last accessed 5 January 2010)

17.8.2 Environmental management on farms

UK Department of the Environment (DEFRA)'s Environmental Stewardship programme: Funding for farmers and land managers for environmental management, administered by Natural England http://www.naturalengland.org.uk/ourwork/farming/funding/es/default.aspx (last accessed 5 January 2010)

DEFRA's Code of Good Agricultural Practice (CoGAP): Guidance for farmers and others on pollution prevention and natural resource protection http://www.defra.gov.uk/foodfarm/landmanage/cogap/index.htm (last accessed 5 January 2010)

UK Environment Agency Pollution Prevention Guidelines: Guidance on legal responsibilities and good environmental practice for business and the public http://www.environment-agency.gov.uk/business/topics/pollution/39083.aspx (last accessed 5 January 2010)

17.8.3 EMS schemes and guidance

EMAS: The EU Eco-Management and Audit Scheme is a management tool for organisations to evaluate, manage, improve and report their environmental performance http://ec.europa.eu/environment/emas/index_en.htm (last accessed 5 January 2010)

Pilot 'EMASEasy' application for agriculture and food production http://www.life-emasfarming.org/ingles/index.htm (last accessed 5 January 2010)

The Green Dragon Environmental Standard is a stepped methodology for environmental management (five levels of achievement), mainly used in Wales and administered by Groundwork Wales http://www.groundworkinwales.org.uk/greendragon/index.html (last accessed 5 January 2010)

BS 8555 STEMS (Steps to Environmental Management Systems): A six-phase scheme for environmental management, designed by the British Standards Institution (BSI) http://www.bsigroup.com/en/Assessment-and-certification-services/management-systems/Standards-and-Schemes/BS-8555-STEMS/(last accessed 5 January 2010)

IEMA Acorn Scheme: A flexible EMS implementation and accreditation scheme based on BS 8555, promoted by the IEMA (the UK Institute of Environmental Management and Assessment) http://www.iema.net/ems/acorn_scheme (last accessed 5 January 2010)

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18

Environmental training for the food industry

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Abstract: This chapter presents a view on the current international research in workplace learning. The chapter is introduced by quoting the requirements on employee participation and training in the European regulation on environmental management and audit systems, and outlining the value of employee participation and training in the development, implementation and operation of an environmental management system. The different training needs are analysed for the various departments in a food company, and the concept of the learning organisation is introduced. The chapter covers both the external training situation and workplace learning. The maintenance of a learning organisation is treated, focusing especially on the problems related to personnel turnover, and the crucial role of management. Finally, some current and future trends in workplace learning are outlined, especially regarding training evaluation.

Key words: workplace learning, environmental management system, training needs, learning organisation, training evaluation.

18.1 Introduction

Little research has been done with particular reference to environmental training, and even less with reference to such training in the food industry. However, the chapter seeks to relate the general state-of-the-art in workplace learning to the specific field of environmental training, drawing upon the author's experience from 20 years as an environmental consultant to the food industry.

18.2 The value and nature of environmental training

The objective of environmental training for the food industry is to *build competence* to understand how the industry and its products impact on the environment and how these impacts can be continuously reduced through application of an environmental management system (EMS). In the food industry, the EMS will typically be integrated with a quality management system and a Hazard Analysis Critical Control Point (HACCP) food safety management system, and the training may therefore integrate aspects from all of these systems.

Decisions that affect the environment are made at many different levels in a company, and environmental training should therefore not be reserved for the personnel with explicit responsibility for environmental issues. A successful environmental management system relies on the positive forces of responsibility and creativity of all employees.

Employee participation is a requirement in an environmental management system registered according to the European EMAS II regulation (Annex I-B(4)): '*employees shall be involved in the process aimed at continually improving the organisation's environmental performance*', with the following specific references to training (Annex I-A(4)(2)): '*The organisation shall identify training needs. It shall require that all personnel whose work may create a significant impact upon the environment, have received appropriate training. It shall establish and maintain procedures to make its employees or members at each relevant function and level aware of:*

(a) *the importance of conformance with the environmental policy and procedures and with the requirements of the environmental management system;*
(b) *the significant environmental impacts, actual or potential, of their work activities and the environmental benefits of improved personal performance;*
(c) *their roles and responsibilities in achieving conformance with the environmental policy and procedures and with the requirements of the environmental management system, including emergency preparedness and response requirements;*
(d) *the potential consequences of departure from specified operating procedures.*'

Beyond these formal requirements, the real challenge of environmental training is to ensure that environmental management becomes and remains a productive force and a continuous source of innovation, rather than another burden on top of other daily procedures.

The value of employee participation lies in:

- Alertness to important causes of inefficient use of inputs or emissions, that may otherwise go unnoticed. As inputs are also directly linked to costs (and emissions signals a possible wasted input, sometimes even costly to handle), such alertness is of direct economic benefit, besides having merit for the environmental performance. There are numerous examples of employees, in their mapping of material and energy flows within their unit, discovering electrical appliances running without apparent purpose, 'hidden' water losses, and even a 'forgotten' heater in a cooling tunnel.

- Preparedness to accept changes when new procedures have to be implemented as part of the environmental management system or to improve environmental performance.
- Spreading responsibility for the environment to all employees who take operational decisions; the best guarantee that problems are minimised and eventually entirely prevented.
- Alertness to opportunities for reaping benefits through communicating the improvements in environmental performance already achieved.
- A more stable workforce that takes more pride in their work and acts as ambassadors for their company in the community.
- Embedding the investment in environmental training inside the company rather than letting the investment leak to external consultants.

It should be clear from the above that learning the procedures of the environmental management system is the least part of environmental training. The major part of environmental training lies in developing a commitment to continuous investigation of the structures and activities constituting the food production chain and its interaction with the environment, and a competence to respond to the result of this investigation.

Such training must be interactive, investigating and challenging the participants' attitudes to the environment and their understanding of their roles in relation to their work, the product they produce, and their opportunities in affecting its environmental impact.

Such training is best performed in groups, composed of participants from different parts of the company, preferably representing all activities from when the raw materials enter into the company until the product reaches its customer. It is important that both middle and top management also take part in the training. More specific tasks may be solved as homework, or in more homogeneous groups in each department.

18.3 Training needs and improving environmental understanding throughout the organisation

An understanding of what the environment is, and what environmental performance is and how it is influenced, may differ widely between the different departments and job functions in a food company. The environmental training should take into account such differences in perception, in order to appear meaningful to the personnel. Pre-conceived perceptions may be challenged and altered as a part of the training, but the personnel must feel that their perception is taken serious and investigated, in order to maintain and stimulate their active involvement.

The *purchasing department* will typically see environmental issues in terms of differences between suppliers and how environmental performance may become part of the purchase requirements. It may be less obvious that a specific choice of raw material supply may also have environmental consequences in the production and later stages of the value chain. Training should give the purchasers the ability to raise environmental issues with the suppliers, but also an understanding of the possible environmental consequences in the production and later stages.

The *production and logistics departments* will typically focus on their own emissions and raw material and energy use. Energy is typically the most important factor, especially when dealing with dried, heated or cooled products. Work environment may be a cause of concern that some employees may regard as part of the environmental work, and solutions to environmental issues may at times conflict with concerns for the working environment. Such positions need to be taken seriously, and integrated into the training. Product losses may also be an area that is in focus in the production and which, like energy use, has important direct economic consequences. Often it is found that the causes of product losses are not fully known, and are often blamed on suppliers or other factors outside the influence of the production and logistics departments. This may, in fact, make it even more important to include such issues in the training.

Sales and marketing departments typically see environmental issues as a complicating factor in sales, if at all relevant. Only in a few cases are environment issues seen as an opportunity in the food industry. Thus, a starting point for the training should be the options for turning 'environment' into a benefit for the sales. From this starting point, the sales and marketing may become important advisers for the overall development of the environmental policy of the company. An environmentally better product does not improve the environment unless it is sold, thus replacing less environmentally friendly alternatives. Therefore, increase in competitiveness and market shares must be part of any environmental business strategy. Sales and marketing departments need motivation and awareness of the tools available for handling environmental issues raised by customers, and for communicating the environmental work already done by the company.

Administrative functions, such as *personnel, legal, and financial administration* may likewise each have their view on environmental issues. Often, they see their role as less directly linked to the environmental performance of the company. However, a closer examination may reveal that signals coming from the administration play a major role in shaping the views of the entire organisation. The training for these groups should therefore focus on understanding the possible role they may play in the shaping of an environmentally-conscious company. It could include such issues as recruitment and training plans (for the personnel administration), environmental management accounting and total cost accounting (for the financial administration), and inclusion of environmental issues in the formal relations to suppliers, sub-contractors, licensees and local authorities (for the legal administration).

Top management has a large influence on the long-term environmental impacts of a food company. Location of a new plant, or expanding or closing of an old one, a decision on a merger, entering a new market, deploying a new process, or developing a new product, are all examples of decisions that can be taken with or without a view to the environment. The building of an environmental management system can be completely compromised if decisions at the strategic level are not judged on their environmental merits and communicated to the rest of the organisation with this in mind. Training for top management should include the use of long term planning and scenario techniques to include environmental issues in strategic decisions.

The individual needs of each personnel group should not blur the need for joint training events *across departments*, which serve to create a common understanding, sharing of views and integration of solutions. Especially, the involvement of top management in joint training events is essential for signalling commitment to the implementation and maintenance of the environmental management system.

18.4 The concept of the learning organisation: thinking in systems

The concept of the learning organisation was coined by Senge (1990), who outlined what he named *five disciplines* of a learning organisation: personal mastery, mental models, shared vision, team learning and systems thinking. Senge stated that these five disciplines, in combination, comprise a critical mass to build a learning organisation. The principles have been followed successfully in many small and large companies, also in the food industry.

The discipline of *personal mastery* is the individual foundation of learning. It is the will to learn, the desire to learn, or as Senge puts it: ‘approaching one’s life as creative work.’ A company that wishes to support learning needs to support the individual employees in their search for what is important to themselves as individuals. The motivation for learning needs to be founded deeply in each employee. This also implies openness towards new ideas and serious feedback to the employees who show initiative. Personal mastery cannot be forced. It cannot be taught. It is basically achieved by creating an atmosphere of openness, where ‘it is safe for people to create visions, where enquiry and commitment to the truth are the norm, and where challenging the status quo is expected’ (Senge, 1990). While personal mastery focuses on the individual, *empowerment* is a term that covers the same basic meaning, but is used more often in relation to groups.

When establishing environmental data monitoring procedures as part of an environmental management system, it is important that the data collection procedures and the data formats do not alienate employees, but are flexible enough to encompass their own perception of importance.

Understanding that individuals have different learning styles (see e.g. Felder, 1996) implies that training must be designed to accommodate such differences. Self-directed learning is one way to ensure that the individual learning styles are respected (Fisher, 1995). In a study involving 67 employees in the fish industry, Straka (1997) concluded that self-directed learning is, in itself, correlated with a feeling of competence, success and efficiency – although it is unclear whether this feeling is a prerequisite for the training or a result of the training. The most productive interpretation is probably to see this as a positively reinforcing loop.

The discipline of *mental models* is the skill to surface, investigate, challenge and modify the underlying 'pictures' that each individual has of a particular issue. We may all think that we know what 'the environment' is, but in fact some will think of personal health, others of nature, others again on pollution, each giving different weight and perspective to the issue. To create a learning organisation, it is essential to surface such different concepts that may else be a barrier to training and a hidden cause of conflict. A traditional business reaction to environmental issues has been 'we don't have any problems here'. This is a mental model that is certain to hamper implementation of environmental management and create conflicts in the training situation. Less extreme versions of this mental model are those placing the responsibility elsewhere, as the slaughterhouse that received the information that the main environmental impacts in the product chain were in agriculture, with a sigh of relief: 'Then we don't have to do anything.' However, most often, mental models are more or less subconsciously shaping the way we view the world and, as 'hidden agendas', they shape our attitude to new initiatives. Awareness of mental models is the key to ensure that new procedures and the environmental management system itself is shaped to fit the current understanding among all involved. By examining and sharing mental models, they develop and improve. Improving mental models *is* learning in essence.

The discipline of *shared vision* builds on the two previous disciplines, emerging from personal visions and sharing of mental models. Commitment to a shared vision is the key to liberate the creative forces in the employees to the benefit of the whole company. Like personal mastery, commitment cannot be taught. It must be *grown* from openness and feedback. Building a shared vision in the area of environmental work may be difficult if the company is not already a learning organisation. Small steps may be needed, starting with issues where commitment can be more easily achieved, such as, for example, the work environment. And even there, commitment to a shared vision cannot be forced. It requires freedom of choice.

The discipline of *team learning* is the skills of productive dialogue and discussion: to suspend mental models, seeing every colleague as a contributor, facilitating dialogue, and balancing dialogue and discussion.

The discipline of *systems thinking* is the core discipline of Senge's concept and gave the title to his book: *The Fifth Discipline* (Senge, 1990). Systems are essentially composed of positive and negative feedback loops, with or without time delays, which reinforce or balance each other. Business and its environment are systems that we can only truly understand and manage when we discover the 'hidden' rules of systems dynamics, and how these govern our actions. What we learn from systems thinking is that things are often not as simple as they may appear, and that sustainable solutions, both for the business and for the environment, often requires that we embark on a long, tough haul rather than jumping to quick, but temporary, solutions.

Without an understanding of systems dynamics, solving a problem may involve shifting the problem to somewhere else, as when we 'solved' environmental problems by building longer effluent pipes and higher chimneys in the 1960ies, or flue gas filters and wastewater treatment plants in the 1970ies. The realisation that those were only temporary solutions came slowly, evolving into current cleaner-production practices. It appears now as if we are repeating the problem, shifting at the level of product life cycles, with the application of so-called life-cycle based eco-design guidelines with a narrow focus on individual products, rather than targeting the broader consumption context in which these products are embedded.

Systems thinking reveal such feedback loops, where our own solutions may turn into later problems, e.g. through technological lock-in. For example, an early focus on biological cleaning of wastewater may make it more difficult to apply more fundamental, far-reaching and cheaper waste-preventive and water-saving solutions later on, since the biological treatment plants are dimensioned to a certain minimum inflow of organic matter in the wastewater. Cleaner production solutions are more often based on an understanding of the fundamental problems in a system.

An easy solution may turn out to be less of a solution in the long term. The pressure to introduce lines of ecological ('organic') food has led many a food company to accept reduced efficiency, resulting from the many stops and cleanings that are needed to run small ecological quantities in between the conventional batches. The environmental effects of the reduced efficiency (increased energy use, more product waste), which are caused by the ecological food, are seldom weighed against the environmental benefits that the ecological foods may involve. A systems view might have prevented such inefficient solutions, for example by running the ecological production on separate smaller production units that better fit the smaller quantities.

Systems thinking may also teach us to avoid 'death by data' – the situation where we focus on collecting environmental data from all over the production and the product life cycles, without the necessary understanding of which data are essential and which are just adding to the confusion.

Systems thinking may teach us to focus where the bottleneck is. Realising that the bottleneck in their bakery operation was the packaging machine, Interbakery obtained overall energy efficiency gains of 25%, simply by increasing output through additional packing capacity while keeping the bakery energy consumption constant. The improvement in energy efficiency was obtained as a consequence of the efficiency improvement, not as a consequence of focusing on the most obvious options, namely to reduce the energy consumption directly, e.g. through better insulation of the ovens. Such improvements in product flow and capacity utilisation are often possible and often have larger environmental consequences than the more obvious specific improvements that affect only a smaller part of the product chain.

Understanding the value chain – also known as the product life cycle – is also a fruitful application of systems thinking. Realising that freezing and consumer reheating of their bakery products was a major cause for their products', poor life-cycle performance, led Interbakery to invest in controlled-atmosphere packaging – a strategic decision to start phasing out frozen products from those areas where the consumer preferences could be shifted to the non-frozen alternative.

Using value chain thinking – and environmental product life-cycle assessments – without taking a true systems perspective may lead to wrong focus, i.e. focusing on what appears as immediate big issues rather than on what *determines* the overall long-term impact of the chain. For many food products, the agricultural part of the life cycle may be identified as a main source of environmental impacts. Improving efficiency in dairy farming through the use of growth stimulating agents therefore, at first sight, appears a good candidate for an environmental improvement (Capper *et al*., 2008). However, when considering the system implications, i.e. increased feed demand and less meat output per litre milk, and that the missing meat will be produced by less efficient meat cattle, the overall result of the 'efficiency improvement' is an *increase* in environmental impact (Weidema *et al*., 2008).

18.5 Barriers to effective training

Senge (1990) points to a number of behavioural patterns, so-called 'learning disabilities', that constitute barriers to implementing a learning organisation, and certainly apply to environmental training too:

- Inability to see beyond your own job description and see the meaning in learning new skills ('What does my job have to do with the environment?' 'We never had to care about that before').
- Reacting aggressively or evasively to challenges, rather than calmly analysing your own contribution to the problem and what you can do about it ('Why don't you make them do something about it?' 'You'll just have to communicate our position better').

- Focusing on specific short-term events, rather than slow, long-term processes ('We don't need an environmental management system, we already took our precautions when that problem came up last year').
- Focusing on experience, when the problems are in fact too large or long-term to experience, but rather call for logical analysis and reflection ('Let us now try this and then wait and see if that doesn't solve the problem').
- Believing or presuming that you know the answer instead of questioning your own prejudices ('We all learned that at school, you don't need to go over it again').

Opposition to systems thinking may be found among those who can benefit from a simple solution. As an example, the managers at Rose Meat were strongly opposed to building a costly washing unit for recyclable crates, which were becoming a requirement from the most important retail chains. Although a life-cycle assessment could show the environmental advantages of recyclable crates over cardboard packaging, they avoided the issue by hiring a consultant who could provide an inconclusive life-cycle study by using outdated and therefore more uncertain environmental data. Five years later, Rose Meat anyway had to build a much more costly separate cleaning unit for recyclable crates, which by then had become an industry standard.

Lack of time for reflection is one of the most important barriers to workplace learning. Lack of time may be especially problematic for the production departments where it can be difficult to take time off from the line. It is essential to consider how all employees are given free time and credit for participating in training events and for implementing the new skills in their job situation.

Motivation to participate in training depends on the initial attitude towards the environmental issues and environmental work. When motivation is not obviously present, the trainer should consider linking the subject to other subjects that are higher on the agenda for the employees in question, i.e. finding *synergies* between environmental work and the other success criteria for the personnel group in question. It may be issues such as work environment, quality performance, or reward systems – not to mention economic performance. It may also be important to stress the element of personal competence development.

Environmental issues may, for some employees, appear a very complicated topic, and initial training may therefore need to emphasise simple tools for identification and prioritisation of environmental aspects. It is important to convey a sense of success by drawing attention to solutions that can be easily implemented, so-called 'low-hanging fruits', even when these may be of minor importance in the overall company context. Cleaning without water is an example of such a successful concept that is easy to grasp and introduce, and can be then be referred to when motivation is needed in dealing with the larger, more complicated, and less visible aspects of environmental concern.

Delegating *responsibility* is an essential key to creating motivation. Both Ellström (1996) and Beckett and Hager (2000) point to the relationship between learning and the possibility to make judgements in the work process: The more complicated the task, and the more possibilities the employees have to control its solution, the higher the quality of learning. In a study including also dairy workers, Larsson *et al*. (1986) concluded that the employees expect training to be related to changes in the workplace organisation.

In the food industry, the traditional workplace organisation often constitutes a barrier to workplace learning. In a study of slaughterhouse workers, Jørgensen (1999) points to the limited options for communication and the inflexible nature of the work as a barrier to experimentation and innovation.

18.6 Learning in the supply chain

Placing suppliers and customers together in training situations may at first seem difficult, due to their opposed roles in the context of trade negotiations. However, experience shows that the parties soon see their mutual advantage from the interaction. By looking beyond the company gates, the competitiveness of the entire value chain can be enhanced, which in the long run will benefit all the involved parties.

Product chains seem to be natural starting points for co-operation, since products already constitute the common ground for the economic exchange between the companies. Their mutual interest, the product, will give the training situation a natural focus. A problem that the customer used to blame on the supplier may suddenly be investigated in an atmosphere of mutual curiosity – often resulting in surprising leaps in mutual understanding and problem solving.

When the suppliers and customers are placed together, and preferably from several steps of the value chain, they can no longer avoid tackling their mutual problems: to increase efficiency in the whole chain. No one can lean back and say that it is not their problem – because as soon as they do, it implies that the problem is owned by one of the other parties present.

Training events may be held on 'neutral ground', but as soon as confidentiality issues can be avoided, it improves the learning if the training can be held in-house at one or more of the companies involved. To see directly how the product is respectively produced and used increases the mutual ability to clear up misconceptions and suggest improvements in the interaction between the parties in the supply chain. The objective should be to feel at home, not only in your own company but in the value chain. In this context, it is important to note that it may not always be the immediate suppliers that are affected by changes made at, or required by, the customers. The environmental performance of the supplier is relevant only when the supplier is both willing *and able* to change the production in response to a demand from the supplier. Due to long-term production constraints at the local suppliers of fertiliser and fodder protein, the main environmental impact of additional outputs of most West European food products will actually be caused by suppliers of fertilisers in East Europe and suppliers of soy protein in South America. Finding ways to include these suppliers in the chain dialogue is the real challenge, which cannot be substituted by involvement of the immediate or current suppliers.

18.7 The external training situation: shared vision and the 'learning lab'

The choice of external training versus on-the-job (workplace) training may be governed by practical considerations such as training opportunities, timing, transport and costs. However, from a pedagogical perspective, the most important difference lies in the level of abstraction that typically accompanies an external training situation as opposed to workplace learning.

The external training situation is preferable when the purpose of the training is to provoke openness towards new ideas, to initiate new ways of social behaviour, or to place the everyday workplace in a new perspective. The external training situation loosens the feeling of control – both that of being under control and that of being in control. It allows the participants to voice opinions or observations that they may not venture in their normal job surroundings, and to accept other opinions for investigation, that they would have immediately rejected if they had been presented 'at home'. These characteristics of the external training situation make it ideal for building shared vision. Bold new ideas, such as a completely new way of distributing the product or servicing the customers, can be put forward in the open course atmosphere, without running the risk of being immediately reminded of all the practical obstacles. Criticism can be considered calmly and responded to positively. New alliances can be tested and the way the idea is presented to the decision makers can be improved.

The learning lab is a more formalised form of experimenting with new ideas. A small model world is constructed, typically first on paper, in which the different parts of the idea are detailed and the interactions can be explored. *Role-play* may be a way to make the model come alive and test the psychological mechanisms that may be involved in its implementation. Computer models allow a more advanced form for experimenting with a model world. Several software packages have been designed for this particular purpose; for example, '*Vensim*' from Ventana Systems Inc. (www.vensim.com), 'ithink' from isee systems (www.iseesystems.com), and 'Studio' from Powersim (www.powersim.com).

Some ideas need to be tested in a more production-like environment. Most technical schools and some larger companies have their own pilot production environments that allow full-scale experimenting without having to place the entire production at risk. But it may also be possible to isolate the testing to a specific product line where it can show its merits.

Allowing new ideas to be tested in practice, either in training situations, in computer simulations or in pilot plants, is the ultimate touchstone for a learning organisation. Employees are encouraged by seeing their ideas being taken seriously, and even a failure is a success, both because of what may be learned from it, and also because it emphasises the spirit of experimentation. In this context, it is obviously important that no one is blamed for a failure – on the contrary, the unlucky employees should be praised for their courage to experiment so daringly. Having made a mistake is punishment enough in itself.

18.8 On-the-job training

Compared to external training, workplace training provides better opportunities for integrating direct experience and real-life experimentation into the training situation. Furthermore, it becomes possible to embed the training results immediately in the work situation. Beside these pedagogical advantages, workplace training can more easily be adjusted to the pace of the individual employee and to the pace of the work situation, e.g. utilising periods of downtime.

Lave and Wenger (1991) suggested that effective workplace learning should be seen as a planned process of increased involvement. The idea is that the unskilled should continuously be provided with tasks that are at an adequate level of challenge, to provide a skill that can be learned with a not too overwhelming effort, while the learner is continuously part of the larger context in which the value and necessity of the skill is clearly visible. Thereby, the unskilled will move, step by step, from the periphery of the workplace to full participation. What Lave and Wenger point out is that this process does not happen automatically, but needs to be planned.

However, by focusing on the specific context of the workplace, the training runs the risk of being limited to a transfer of 'know how', rather than a less context-specific 'know what' that allows the employee to translate the know how into other contexts and apply the acquired understanding there, and 'know why' that allows the employee to motivate or question the rationale behind the know how. For example, the 'know how' of how cleaning without water is implemented in one specific situation should be translated into the more abstract 'know what' that mixing materials (here waste and water) reduces the solution space, a knowledge that can be transferred to, e.g., separation of solid waste into different recyclable fractions. And the 'know why' is reached when the employee can see the limits where the principle of separation no longer makes sense, e.g. when the effort does not match the (environmental) value of keeping the waste fraction separate. Such 'robust transfer' (Billet and Rose, 1997) of know how into know what and know why, requires theoretical reflection to be a planned part of the workplace training. This theoretical reflection is best ensured by creating situations where the learner is required to explain the reasons for particular procedures, e.g. to a supervisor or trainer, or as a trainer for others. Marienau (1999) and Boud (2000) argue that self-assessment may also be an adequate way to create such situations of reflection.

Summarising these understandings, Billet (1999) outlined a model for the design of workplace curricula, composed of four themes:

(i) the provision of a learning pathway from the periphery to full participation,
(ii) insight into the total production, so that one's own work can be seen as a part of the overall process,
(iii) direct support or guidance from experts that 'force' the learner into potential situations of learning, and
(iv) indirect support and guidance from colleagues and support by the physical workplace situation.

The fundamental implication is that the training must be systematically planned as part of the work processes.

18.9 Internet-based training

The main difference between internet-based training and off-line training is its additional options for interaction over long distances and for automated interaction. In relation to course planning, internet-based training gives the possibility of access to highly-skilled teachers, who would not otherwise be accessible for personal interaction. It can also potentially reduce the cost of training by substituting live teachers with automated interaction. In relation to training content, it is often seen that internet training, like much early computer-based training, is just a transfer of a traditional face-to-face course with paper-based handouts into the electronic media. The real interactive options of the media are seldom exploited to any significant degree.

Nevertheless, more creative use of the interactive options does appear. Creating learning groups across physical barriers is one important option. While some physical meetings may be required to create the right atmosphere in a learning group, the continued training may well be performed via internet interaction. Internet-based training may include a forum for group exercises and discussions among the employees. It may also include a chat-room, where teachers may be available at specified hours.

Creating learning groups with participants from a physically segregated supply chain, is an option to benefit from supply-chain training without prohibitive costs. Unique to internet-based training is the option to create training sessions that are based on real-time interaction between employees at very different locations and job situations. For example, employees may contribute in real time on real cases of product development, e.g. to design the most environmentally benign version of a specific product and carrying it through all design phases including consumer tests and negotiations with stakeholders and critical company board members. The training sessions may be designed as a competition between different teams, thus providing a case of 'Learning by Play.' Such training sessions provide opportunities for proving the value of the training in a real-life context without the pressure of a real-life work situation.

Another example is an environmental management module, offered as on-the-job training for groups of employees with different job functions in the same company or product chain. The module may feature group exercises in which the employees play the same roles as in their normal job situation, and learn to understand their own role in the larger company and/or product chain context, as well as the role, cultures, tools and language of the other participants. The module should be centred on the understanding that all actors share the same objective, in the form of the product output, but have different viewpoints, cultures and languages with which to address it (Weidema, 2001). The group exercises engage the employees in assisting each other to improve their mutual performance, identifying bottlenecks, removing friction, and adjusting procedures to optimize the value chain of the product they have in common.

18.10 Maintaining a learning organisation

When the environmental management system has become routine, there can be a tendency to professionalise the environmental work, i.e. to let a few selected persons take over the responsibility. Although this may be an appropriate allocation of resources, it carries with it the danger of stagnation of the environmental management system, since it may discourage employees from addressing further environmental issues. In this phase, it is important to maintain a learning environment that allows everyone to contribute to the further development.

When hiring new personnel, special attention is needed to introduce the newcomers to the spirit that has been obtained during implementation of the environmental management system. Specific introduction sessions may be required, where the focus should not be the 'bringing in line' of the newcomers, but rather to alert them to the openness with which these subjects are tackled. At the same time, special attention should be given to take immediate advantage of the possible expertise or experience that the newcomers bring along, also noting that new eyes sometimes see things that go unnoticed by those who have been in the same position for years.

Special care is needed when hiring senior personnel, that they are introduced to the history and spirit of the management system, since they may otherwise quickly disrupt the investment that has been made in both procedures and employee training. It is crucial that new senior personnel show respect for the environmental competence that has been built up and for the individual employees that represent this intellectual capital. The alternative will typically be disappointment and apathy on behalf of the employees. In fact, senior personnel and top management play the most crucial role in maintaining the learning organisation. It is from here that the spirit is originating and it is from here that the spirit is maintained. In the words of Nanda (1988): 'perhaps changes in attitude among top managers are key to the skill development of supervisors'. Only when managers show that they care about the environment and care about the learning organisation, is it possible to convince the employees that this is truly the company spirit. Thus, continuous attention and training is needed at the management level to keep the spirit of the learning organisation alive.

Active reward systems can be a good way to provide a continuous signal that environment matters and that continuous alertness and improvement is appreciated.

18.11 Future trends

The type of participatory training that has been the focus of this chapter is still not commonplace, but is gaining ground in workplace learning. In professional training, e.g. at technical schools, the lecture form is unfortunately still prevalent. Thus, the current trend is the spreading of the concepts outlined in this chapter, while future trends are mainly seen in the increased use of computer-based training, notably with the aid of system dynamics software and internet-based training, as mentioned in Sections 18.7 and 18.9.

Other current or future trends are:

(i) Teaching training skills to domain experts and other employees. Employees take over the responsibility for training, as a natural next step to participatory training. This does not mean that external expertise becomes superfluous, but that the local adaptation becomes more important. Local employees quickly gain an advantage over the external trainer in better understanding the local context, so that the role of the external expert rather becomes that of providing inspiration and material support to the local trainers. This also implies that development of training skills in the local employees, design of the training situations, and alerting local trainers to new trends in environmental management, become important responsibilities for the external trainer.

(ii) Less scheduled classroom teaching: more customised, individualised just-in-time training. As employees gain confidence through participation, they require more individualised training. Individualised development plans means less of classroom training and more customised training.

Impatience will also grow, increasing the need for just-in-time training, at the expense of scheduled courses.

(iii) Improving the measurement of the value of training. As importance of training increases, management will want to know that it is effective.

Kirkpatrick (1979) has provided one of the most widely referenced models for *training evaluation*, listing fours levels: reaction, learning, transfer and results. Today, most training sessions are evaluated only on the first level: on the participants' evaluation of the training, e.g. on a scoring form. The more formal evaluation of what the participants actually learned, through tests before and after the training session, is less common in workplace learning, but is gaining ground. The two last levels of evaluation are more difficult and are thus not widely used. However, the trend is clearly that also these forms of evaluation are on the increase.

Kirkpatrick's third level of evaluation is *transfer of learning*, as measured through pre- and post-training assessment of behavioural changes, e.g. whether employees after training react differently to situations where they become aware of wasteful practices. Transfer of learning measurements may include participants' own assessment, which may even contain an important element of learning in itself (Boud, 2000), as well as the assessment by subordinates, peers and superiors. The post-training measurement should not be performed earlier than three months after the training, in order that the participants have had an opportunity to practise what they have learned.

Kirkpatrick's fourth level of training evaluation is *results evaluation*, measured, e.g. in terms of improved productivity, reduced energy use, reduced waste or scrap, reduced water use, reduced biological content in waste water, reduced discard, improved safety record, reduced days of illness, improved compliance with regulations, number of employee suggestions, number of new ideas implemented, reduced turnover in the workforce, improved qualifications of job applicants, reduced downtime, reduced need for supervision, improved customer satisfaction (fewer complaints), and eventually increased market share. Results evaluation is not possible without the use of control groups, due to the many disturbing influences that may also affect the mentioned measurement criteria. Some additional effects of training may be less measurable, such as the ability to predict and avoid future problems.

18.12 Sources of further information and advice

Good reviews of current research in workplace learning can be found in Boud (1998), Boud and Garrick (1999), and Brown and Brown (2006). The concept of participatory learning developed by Lave and Wenger (1991) is still one of the fundamental sources of inspiration for the design of workplace training. Gerber (1998) provides a list of eleven ways to learning, which can serve as more specific inspiration for designing workplace learning. *Workplace Learning Today* provides a daily update on the field. Current research is published in e.g. *Adult Education Quarterly, Human Resource Development Quarterly, International Journal of Lifelong Education, Journal of Environmental Education, Studies in Continuing Education, Studies in the Education of Adults* and *Outlines: Critical Social Studies*.

Peter Senge's *The Fifth Discipline* is still an indispensable introduction to the concept of the learning organisation, and one of the few works that still stands out more than a decade after its publication. Other important works from this school are Argyris (1990) and Argyris and Schön (1978, 1996). Further viewpoints on implementation and exercises 'from the field' can be found in Senge *et al*. (1994) and ideas how to maintain a learning organisation can be found in Senge *et al*. (1999). Current research is published in, e.g. *The Learning Organization: An International Journal* and *Reflections: The SoL Journal on Knowledge, Learning and Change*.

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19

Eco-labelling of agricultural food products

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Abstract: This chapter discusses eco-labelling of food products from different types of agriculture, together with the underlying certification systems. Three types of agricultural production are on various points compared with conventional agricultural practice: organic farming, integrated agriculture and regional products. For comparison reasons, three related eco-labelling schemes are added that are relevant for agricultural products: industrial eco-labelling, sustainability certification of natural resources and Fair Trade labelling. Thereafter the effectiveness of the schemes is discussed. The chapter ends with a discussion of some perspectives, particularly of the ever increasing number of labelling schemes.

Key words: eco-labelling, environmental certification, sustainability certification, organic farming, integrated agriculture, regional products, sustainable agriculture.

19.1 Introduction

Eco-labelling, or in more official wording 'environmental labelling', concerns the attachment of environmental information to products in order to influence market behaviour. This can be based on an independent quality assurance process (called certification) of the underlying agricultural practice and possibly also of other relevant stages in the value chain, using strict procedures and criteria. The focus of this chapter will be on the labelling of agricultural products and, where relevant, attention will be paid to related certification activities.

Eco-labelling is a private, market-based instrument, complementary to public policy instruments. It can provide information about characteristics of the product itself, such as its nutritional value, taste and possible contaminants.

It also can provide information about the method of agricultural production, for instance related to pesticide use to or labour conditions on the farm, and about Production and Processing Methods (PPMs) of the products. These PPMs need not be reflected in the characteristics of the products themselves but are usually still very relevant in eco-labelling of food products. The information provided on the food products can influence market decisions in two different ways: it can aim at the final consumers (business-to-consumer information) or it can aim at the companies in the agro-production chain (business-to-business in formation). This has implications for the type of information that can best be provided.

As a private instrument, eco-labelling has a number of advantages over public policy instruments (see van Amstel *et al.*, 2008). As a form of 'self regulation' it has low costs for governments and therefore, from the standpoint of governments, it has a high efficiency. Further, it is, in principle, flexible, as adaptations in the requirements can be made without cumbersome political decision processes. This holds in principle because larger eco-labelling schemes do, of course, have their own bureaucracy. And thirdly, as it is a voluntary instrument, there is a fundamental willingness of the producers to comply with the requirements.

These opportunities also show from the different types of results that can be obtained by the application of eco-labelling. Following the production–consumption chain, these include: changes in agricultural practice, directly in the certified farms themselves, but also indirectly by acting as a catalyst for changing current agricultural practice; support of income for small and medium processing enterprises (SMEs) due to price premiums of eco-labelled products; support of the image of down-stream companies, particularly retailers and specialized shops; and, in various ways, potentially better food products for the consumers.

On the other hand, there are a number of limitations connected with the practice of eco-labelling when compared with public policy instruments. There is the overall problem that it is a voluntary instrument that is additional to binding policy regulations, which by itself leads to a more complex situation. More in particular, for the consumers there is the problem of quality assurance; how reliable are the voluntary agreements that constitute the basis of the schemes? See Nilsson *et al.* (2003) for a critical review of this issue. And there is the problem of the often substantially higher prices compared with products from conventional agriculture. Retailers particularly meet the rather practical problem of scarcity of space on their shelves. And both retailers and leading manufacturing companies meet the problem that the use of eco-labelling may be at variance with their market philosophy; providing some of their products with a quality label may suggest that the rest of their products are not of sufficient high quality. There can also be differing interest between these groups of stakeholders. For instance, the main pressure to shift to free-range eggs in the Netherlands came from consumer groups, followed by the retailers, and only after ample time by part of the poultry sector. More in general, there is the limitation that eco-labels may act as barriers to trade in the framework of the WTO. This is particularly the case if the labelling schemes are used as a requirement for public procurement policy.

There are quite different approaches in food product labelling. On the one hand this variety of approaches will surely have contributed to the success of eco-labelling. But the large number of labels also has led to a confusing situation in the market. In fact we can observe a proliferation of eco-labels on food products with more than fifty types of labels in Europe alone (Ilbery and Maye, 2007).

The general question of this chapter is: how can eco-labelling of food products, in its different forms and different roles and with its different market strategies, contribute to a more sustainable agriculture? The following aspects will be discussed. The chapter will start with an overview of three main approaches of eco-labelling of food products, and of three related types of labelling (Section 19.2). Thereafter, in Section 19.3, for these different approaches the following three aspects will be discussed: the scope of the envisaged impacts, the chosen market strategy and the way the agricultural value chain is taken into account. Then information on the effectiveness of food product labelling will be presented (Section 19.4). And finally a number of perspectives will be discussed; for instance, the differences between private and public use of the labelling schemes and on the increasing proliferation of product labels (Section 19.5). The focus is on the European situation, with the Netherlands as focal country, but also with added information from other parts of the world.

19.2 Main approaches

A wealth of labels has been developed for food products, and the number is still increasing; these labels, in part, can be used independently of each other, and can also support each other, but in part they can also be competing. The focus in this section will be on labelling and certification systems that have global relevance.

Three main approaches of food labelling will be distinguished here, which are distinguished on a basis of the type of agricultural practice used; these are – organic farming, integrated agriculture and regional products. In addition, there are three approaches that have not (or not specifically) been developed for food products, but that are increasingly used for this product category. These are industrial eco-labelling, sustainability certification of natural resources and Fair Trade. These six approaches will be briefly described below.

19.2.1 Organic farming

The historic basis of 'alternative' farming practices lies in 'organic farming'. This movement had a start in the 1930s, with pioneers such as Albert Howard in the UK (as a developer of composting methods), Rudolf Steiner in Switzerland (as the founder of biodynamic farming), and Jerome I. Rodale in the US (who founded a number of magazines in this field). Organic farming got off the ground as a response to the use of industrial fertilizers, and more so since the 1940s to the use of synthetic pesticides. All forms of organic farming focus on agricultural practice of the farm as a whole. The aims of organic farming are to enhance biological diversity within the whole system; to increase soil biological activity; to maintain long-term soil fertility; to recycle wastes of plant and animal origin in order to return nutrients to the soil, thus minimizing the use of non-renewable resources; to rely on renewable resources in locally organized agricultural systems; and to promote the healthy use of soil, water and air as well as minimize all forms of pollution thereto that may result from agricultural practices. This is translated into a number of specific requirements, including: no use of industrial fertilizers, no use of synthetic pesticides, no use of feed additives, no use of genetically modified (GM) crops, a none too strict crop rotation, and mechanical weed control. These strict and well recognizable requirements are all defined at the level of management activities; they are thought to result in a living soil ecosystem with high soil fertility, a good water quality, high biodiversity and a harmonious landscape. The farmers should be acting as part of an encompassing natural system.

Although there are basic standards of the International Federation of Organic Agriculture Movements (IFOAM), national systems still do differ. For instance, in the US and in the UK there are formal requirements for the minimum area of natural habitats, whereas in other countries such as the Netherlands there are no formal requirements for that purpose, and Manhoudt and De Snoo (2003) report that in this country organic arable farms have only slightly more non-productive land than conventional arable farms (3.1% compared with 2.1%). In contrast, for bio-dynamic farming in the US a minimum of 10% of the productive farm area should be set aside as natural habitat.

In this context it is of importance that international guidelines for organic agriculture be developed. This task has been taken up by the above mentioned IFOAM, but is also included in the Codex Alimentarius, an international forum of 176 countries founded by FAO and WHO, which has as its main task to develop standards, guidelines and related texts on food products (Codex Alimentarius Commission, 2007). This Codex has, in mutual interaction with IFOAM, just issued a third edition of guidelines for the production, processing, labelling and marketing of organically produced foods. The above mentioned aims are part of this document. The main drivers for organic agriculture are concerned farmers, followed by consumers focusing on human health and on environmental aspects.

19.2.2 Integrated agriculture

The second approach concerns 'integrated agriculture', a form of agricultural practice that aims to balance environmental and economic interests. By optimizing between these interests, it is more flexible than organic farming. For instance, the use of synthetic pesticides and industrial fertilizers is minimized, but not totally excluded. Within integrated farming, one can observe a very positive attitude towards high-tech farm practice. Whilst the main driver for organic farming concerns farmer and consumer concerns, the main driver for integrated farming is industry. It can best be described as a searching strategy rather than a set of prescriptions for the agricultural practice. Present integrated farming can be the common practice of tomorrow. As a consequence there is no overall organization, nor one encompassing global label.

Still, it is a very important approach, which up to now mainly has developed in the form of the development of separate environmentally-friendly production techniques, each with its own label. In particular, one can put under this umbrella: integrated pest management (IPM), with the aim to minimize the use of synthetic pesticides (as a balance between often still rather abundant use in conventional agriculture and a strict exclusion as in organic agriculture); integrated crop management (ICM), including both risk prevention by resistant races, and biological pest control; glass houses that aim at a carbon-neutral performance; precision farming, including the use of GPS for the spraying of pesticides, or automatic individual recognition of cattle; conservation agriculture, with a focus on limited use of soil tillage techniques; and recently also developments in sustainable aquaculture based on a people-planet-profit (PPP) paradigm (see also Section 19.2.5).

Integrated agriculture has strong roots in both the US and in Europe. For instance, integrated pest management has been developed in California and in rice fields in South-east Asia; conservation agriculture is being developed under the umbrella of the FAO; and carbon-neutral glass houses are particularly promoted in the Netherlands. But this form of agricultural practice is also strong in China, with farming of labelled 'green food' occupying 8% of all cropland, in contrast to 3% occupied by organic farming (Paull, 2008).

The integrated agriculture approach has been taken up by a number of industries and retailers in the form of upstream contracts with their suppliers. Two main European examples can be mentioned. Firstly there is the Sustainable Agricultural Initiative set up by the companies Unilever, Nestlé and Danone. Secondly there is EurepGAP, which is a common standard for farm management practice, created in the 1990s by several European supermarket chains including Sainsbury and Tesco in the UK, Migros in Switzerland, Delhaize in Belgium and Albert Hein in the Netherlands, together with their major suppliers. Recently this standard has achieved a global reach under the name of GLOBALGAP. There are also private labelling schemes that can be put under the heading of integrated agriculture, such as the Utz Certified label for coffee, which sets less strict ecological requirements than organic coffee and less strict social requirements than Fair Trade coffee (see below).

Integrated agriculture is essentially driven by technological innovation, as is also shown by its strong basis in industry. It can have a potentially large impact by achieving relatively small changes over large surface areas.

19.2.3 Regional products

The third main approach of farming practice concerns regional products. The first main distinguishing characteristic is that the products concerned originate from a well definable region. In addition, many quality aspects can be linked to this. There is a general focus on gastronomy and, in line with this, on the use of traditional techniques and related employment. It has therefore strong links with the so-called Slow Food Movement, originating in Italy but now with a spread over fifty countries. But regional products can also be related to landscape and biodiversity conservation in the given region; in Europe this is, for instance, supported by the ENVIREG programme of the EU.

Consumer perception of regional products is mainly determined by food quality, the locality of production, the vitality of rural areas, small transport distances, freshness, and animal well being (Roininen *et al.*, 2006). But in fact the concept behind 'regional products' and its use in practice is not clear and can lead to confusion. An example concerns the Belgian Flandria quality label for fresh fruit and vegetables, which has been used for ten years for more than fifty fruit and vegetable categories (Verbeke *et al.*, 2008). Buyers of Flandria tomatoes appear to have a stronger belief in the healthiness of tomatoes in general, than of the specific qualities of the certified tomatoes. The highest association both for buyers and non-buyers is with 'Belgian origin'. 'Better quality' and 'strict production control' also play a role, but are less dominant.

Although attention for the environment is increasingly connected with regional products, this need not be the case. For instance, there is no specific attention to the environment with the traditional use of calves' stomachs for the fermentation of cheese, with champagne from the French Champagne district, or with Parma ham (Prosciutto di Parma) from the Italian province of Parma.

It is deemed important that for regional products, a clearer basis should be found in the EU regulation on Protected Denomination of Origin (PDO), which offers opportunities to come to a stricter concept and which is starting to incorporate requirements on the environment and on sustainable development (Sanz Cañada and Vazquez, 2005). The main driver lies in regional farmer organizations that aim to enhance employment by promoting traditional production techniques.

19.2.4 Industrial eco-labelling

Industrial eco-labelling traditionally is concerned with the environmental impacts associated with products from different sectors of industry. It has not been specifically developed for agricultural production and food products, in its rational approach resembling integrated agriculture but with two differences. The first difference, of course, concerns the focus on industrial rather than on food products. The second difference concerns the broadening of the focus on the whole life cycle of the products and not only on the phase of agricultural practice. Thus, it is connected with the ISO 14020 series on eco-labelling and also often makes use of Life Cycle Assessment (LCA) for quantification of impacts over the life-cycle (see also Section 19.3.3).

Examples of industrial eco-labelling include the Blue Angel in Germany, the Swan in Scandinavian countries, and the EU eco-labelling scheme. In principle, this approach can also be used for food products, but up to now this has not often been the case. A forerunner in this context is the Dutch industrial eco-labelling scheme, which is now also being used for food products, under the name 'Agromilieukeur'. The main drivers of industrial eco-labelling lie in the manufacturing industry, but these are now going beyond their traditional field.

A recent development in industrial eco-labelling concerns the so-called carbon footprint. This label presents the total greenhouse gas production of the given product over its whole life cycle. Regarding food, particularly the UK-based retailer Tesco is active with this approach, having shifted from the earlier food miles to the more encompassing carbon labelling.

19.2.5 Sustainability certification of natural resources

The sustainability certification of natural resources is strongly gaining momentum world-wide. It started with the certification of timber, later on other resources followed, such as fish, palm oil and biomass for energy production, i.e. food products are now included. A first main characteristic is a so-called *People-Planet-Profit* (PPP) approach (Elkington, 1997). In contrast to a traditional focus just on ecological requirements, this approach gives equal weight to social requirements, such as labour conditions, income and land use rights ('people'); ecological requirements, in particular biodiversity, environmental quality and climate ('planet'); and economic profitability requirements, particularly also including long-term profitability ('profit'). It mostly includes attention to the *Chain-of-Custody*, in terms of requirements aiming at avoiding undue mixing of certified and non-certified products in the course of the value chain and mainly involves international, that is, world-wide schemes. A consistent structure of requirements consisting of Principles, Criteria and Indicators is the core of each of the systems. It mainly aims to be the basis for purchasing behaviour of consumers, retailers and manufacturing companies, but it increasingly also is becoming a basis for governmental procurement policies.

Examples of this approach are the following. For timber, the main schemes concern those of the Forest Stewardship Council (FSC) and the Programme for the Endorsement of Forest Certification schemes (PEFC). The first of these is an international system on its own, the latter a so-called meta-system, under which an increasing number of existing national schemes are being endorsed, including those of a number of European countries, the US and Canada, and a small number of developing countries. For fisheries there is the worldwide scheme of the Marine Stewardship Council (MSC). The label connected with this certification scheme is the most predominant of these labels on food products. A more recent development concerns the scheme from the Round Table for Sustainable Palm Oil (RSPO) and the Round Table for Responsible Soy (RTRS), both of which aim at the use as food products, and the first of these also on the use as bio-fuel. And there are schemes in development that focus on biomass for energy applications in general; these are the Round Table for Sustainable biomass (RSB), CEN/TC 383 of the European Committee for Standardization (CEN) and the International Organization for Standardization (ISO).

A last interesting feature of this approach concerns the combined input of both industry and NGOs. In the development of both FSC and MSC, WWF played a core role; in all the schemes mentioned, a crucial issue concerns the representation of NGOs on the governing bodies. NGOs are therefore, next to a number of pro-active industries in the given areas, main drivers for this certification approach. The balance can differ from one system to another; whilst FSC had its origin in an initiative from the NGO side, PEFC originated as a response to FSC and had its roots in the timber sector.

19.2.6 Fair Trade

In addition to the five above approaches, which have a main focus on environmental impacts, there is the Fair Trade label. This approach primarily focuses on the social and economic aspects of agricultural management but it also deals with the management of the farm as a whole, just as in organic agriculture. The key requirement concerns the achievement of a fair price for the producer in international trade; that is, a price which bears a relationship to the costs of production and is not simply determined by international market relationships. A well known example concerns Max Havelaar, originally a Dutch coffee importer, but now a global organization which also includes other product types. Often ecological requirements are also being dealt with in Fair Trade, as part of the label or by combination with another label; for instance, Raynolds (2002) reports that 30% of Fair Trade coffee schemes are combined with an organic label. Another example concerns olive oil from Canaan Fair Trade, the largest oil exporter of Palestine, which is also organic.

Food products that are often found with a Fair Trade label in retail shops include coffee, tea, cacao and processed cacao products, bananas, oranges, tangerines, mangos, grapes, pineapples, oil, rice and wine. Main drivers concern development NGOs and specialized Fair Trade shops with their customers.

19.2.7 Conclusion about labelling approaches

Concluding this section, one sees a wealth of activities and initiatives, with different characteristics and different market potentials. But at the same time one can observe that this is increasingly becoming a confusing situation, even for single food products. In fact, eco-label like quality assurance schemes, are presently flooding the market. For instance, Nilsson *et al*. (2003) recently compared the use of 58 different eco-labelling initiatives for food products in Europe. A comparable diversity is found in the US. An example of a confusing situation concerns coffee, for which there are a number of different labels with a varying but partly overlapping meaning: Organic coffee, Fair Trade coffee, the Smithsonian shade grown and bird friendly coffee, coffee from the Rain Forest Alliance, and the Amsterdam based Utz Certified coffee; a rich, but confusing diversity.

A first and essential step to reduce spurious eco-label development is to set requirements on independent control of the labels. This is well in place for organic farming, sustainability certification of natural resources and for fair trade. For the other labelling systems, such a control is far from complete.

In the next sections we will compare the six main approaches mentioned above in four crucial respects: the scope of envisaged impacts, the market strategies, the role of the value chain and the effectiveness. It should be recognized that the given distinction in six main approaches which are internationally relevant constitutes a simplification, while in practice many borderline cases are apparent.

19.3 Comparison

In this section, the different approaches distinguished in Section 19.2, will be compared in three aspects: the scope of the envisaged impacts, the market strategies and the way the value chain is taken into account.

19.3.1 Scope of the envisaged impacts

As will already be clear from the above description, there are substantial differences regarding the scope of the impacts of agricultural practice that are included in the different types of eco-labels. Overall, the main aspects include human health together with product taste, biodiversity and environmental issues, and economic/social/cultural issues. These main issues receive, however, a different amount of attention in the different types of labels. Summarizing the above section, we observe the following main focuses in the different approaches.

For organic farming there is a main focus on human health (by intrinsic quality of the product and by the absence of residues) plus product quality, and on biodiversity plus environment. Farmer income is also an important social issue. For integrated agriculture, the main focus is on biodiversity plus environmental issues, but explicitly balanced against the cost price of the products. For regional products there is a main focus on gastronomic aspects, combined with the use of traditional techniques and local knowledge, and related local employment. In addition the perceived high impact of long distance transport can also play a role. Industrial eco-labelling has a main focus on environmental issues only, generally not including biodiversity. Sustainability certification, in general, gives equal attention to environmental and to social and economic issues. This can be seen as a quite balanced approach, in line with the global impetus of the PPP strategy (see Section 19.2.5). Finally, Fair Trade puts the social issues (in particular the farmer's income in developing countries) in the core position, with additional attention to biodiversity and environmental issues.

For the consumer, the distinctions between the different approaches are often less clear. In particular there appears to be quite some overlap in the preferences for fair trade and organic products (see Loureiro and Lotade, 2005). Thus one may wonder whether a combined approach would be possible of the two types of requirements. In practice this is already happening in the field where one can observe that by one farmer both types of requirements are met. This is particularly relevant for developing countries; less so for industrialized countries where, in general, social requirements have sufficiently been laid down in legislation. A formal combination of the two types of schemes is not so easy, however, given the fact that they involve quite different organizations. It should be noted that sustainability certification schemes which are organized according to the People-Planet-Profit approach do bring the different aspects together, be it at a somewhat lower level (see Section 19.2.5).

19.3.2 Market strategies

The above approaches clearly differ with respect to their envisaged market strategies and market share. Three main strategies can be distinguished: legal compliance, beyond legal compliance and niche market.

In fact the whole market should meet legal requirements, both in ecological and social aspects. Meeting legal requirements concerns the products from conventional agriculture that are traded without any label or specific agreement between market parties. It should be noted that the borderlines between the first and the second level can be different in different countries. In Germany, for instance, the requirements regarding pesticide use are stricter than in most other EU countries, thus approaching what is called 'integrated pest management' in these other countries. It also is a dynamic process: what is deemed beyond legal compliance today, may be legal compliance tomorrow.

'Beyond legal compliance' is the domain of Corporate Social Responsibility (CSR) and best practice approaches. Specifically for food products, this is the field of 'Good Agricultural Practice' (GAP), as for instance promoted by the already mentioned EurepGAP or GLOBALGAP programme. This particularly involves agreements between these pro-active retailers with their suppliers on practices in the field of integrated farming (e.g. integrated pest management). In addition, it may involve products with regional products sold as specialities.

Niche markets are aiming at specific high-level labels including, in particular, organic farming, (part of) regional products, Fair Trade products, and labelled products such as for Utz coffee, MSC fish or RSPO palm oil. These are, in various ways, the front-runners, which aim at the top of the market. The products are generally sold in specialized shops, such as eco-shops and fair trade shops. But increasingly, certified products are also sold in supermarkets. In the UK, 70% of organic products is sold by three major retailers; in Denmark 64% by two major retailers (Wier *et al*., 2008). Also, in the Netherlands supermarkets have a larger market share for organic products than specialized shops.

A core question in the estimation of the potential market shares of the different strategies concerns the willingness of consumers to pay higher prices. In fact, for beyond-legal-compliance products, no price premiums are being paid; rather these are sold in retail shops that have higher price levels compared with discount retailers. But price premiums do have to be paid for products for niche markets. For assessing what premiums consumers are willing to pay, most often the Willingness-to-Pay (WtP) methodology is used, based on consumer enquiries. According to this method, consumers appear, in general, to be willing to pay some 5–10% higher prices for high-quality certified products. For instance, 70% of EU consumers appeared to be prepared to pay a 10% premium for Fair Trade bananas (Loureiro and Lotade, 2005). The outcome of this method can be doubted, however, due to the occurrence of socially correct responses, which need not predict actual consumer behaviour. Empirical studies are rare. Interestingly, Bjørner *et al*. (2004) performed a statistical analysis of actual changes in purchasing behaviour in a supermarket after introduction of a Nordic Swan eco-label on different products. For toilet paper, the consumers appeared to be willing to pay an additional price of 13–18%, for detergents they found a comparable willingness, but for other products they could not establish an effect; see also Kollert and Lagan (2007) for an overview of the Willingness-to-Pay price premiums for certified timber.

Looking at the six approaches, we can observe that they all set themselves a higher aim than just legal compliance. Although the line is not sharp, we can distinguish two groups. The first group, integrated agriculture, industrial eco-labelling and sustainability certification of natural resources, have their focus at the level of beyond-legal-compliance; they aim in different ways to cover the whole market of tomorrow. Organic farming, regional products and fair trade aim at niche markets; they principally serve different special groups of customers.

For organic farming, there is a specific reason why it will not be able to cover the full market – its productivity is lower than that of current modern agriculture, particularly because of the (near) absence of fertilizer use. Nutrients can never be fully recycled. Nitrogen supply can be warranted by nitrogen-binding leguminous species such as clover; phosphate, however, has to be supplied from external sources. Exchanges between farms, as with the import of animal feed, cannot solve this problem. Apart from guano manure, blast furnaces, and probably increasing extraction from sewage or from algae or mussels, fertilizers from phosphate rock are essential for the world food supply.

19.3.3 Agricultural practice and the value chain

All schemes that deal with agricultural food products include the impacts of agricultural practice. The question is how they deal with the further processes in the value chain, either upstream, such as the production of fertilizers or animal feed, or downstream, as in particular food processing, packaging, transport and retailing. We can distinguish three main options to deal with these effects. Firstly, there generally is an administrative control of these processes against undue mixing of certified with uncertified products. This can be achieved if the whole chain is in the hands of the given organization. This is true for organic food products (for instance certified by the international SKAL foundation), and also for Fair Trade products. Generally, this control is performed by administrative requirements for all stakeholders in the value chain on the administration of their purchases and sales. The requirements for this can be specified in a 'Chain of Custody' (CoC) standard that has to be applied by accredited auditors of the conformity of certified products, an approach which is common with sustainability certification of natural resources. Such a control needs not fully exclude all mixing of certified and non-certified products. A 'mass balance' approach can accept such mixing, if the 'percentage out' is equal to the percentage in', such as is the case for green electricity. This for instance also holds for timber in FSC and palm oil in RSPO certification.

Secondly, in addition to a Chain-of-Custody approach, impacts of processes along the value chain can be included in qualitative terms in the assessment of the products themselves. Information on the quality of these processes then constitutes part of the technical content of the label or of accompanying information. This is generally done in terms of pass/fail criteria. For instance, organic farming sets the requirement of 'no use of GM crops', or 'only use of sustainable soy' in the feed of the cattle. Or it can be done in requirements such as 'rather food products from nearby sources than from far-away sources', as is being implemented by a retailer such as Sainsbury in the UK.

The last example also leads us to the third option, implying that the impacts of processes along the value chain are included in quantitative terms in the assessment of the product itself. Then the travelling distance of the food product is expressed in 'food-miles', presented on labels attached to the boxes in the shops (see also Section 19.2.4). Quantification of impacts along the value chain (or life-cycle) can also be performed in a more sophisticated way by using Life Cycle Assessment (LCA). This tool usually includes the impacts of all relevant processes upstream of the farm, those of farm practice itself, and those of downstream stream processes in relation to farm practice. This is in particular performed in industrial eco-labelling as discussed in Section 19.2.4, and is a prerequisite for the calculation of the carbon footprint of products. A typical result of such LCA calculations is that the carbon footprint (and also the environmental impacts in total) up to the gate of the farm appear to be far larger than the footprint of the subsequent transportation of the products (see for instance Roy *et al*., 2009).

A few remarks will be made here on the use of LCA for food products. This tool essentially focuses on products over their whole life cycle, from cradle to grave. It therefore has quite another focus than the functioning of a farm as a whole producing various products (for a clarification of the difference between these two complementary viewpoints, see Udo de Haes and de Snoo, 1996, 1997). The extension of the use of LCA to food products is step-by-step increasing. The methodological basis of this tool is laid down in the ISO 14040 series. A main requirement for LCA is that all impacts are quantified in relationship to a so-called *functional unit* of the product at stake. This implies a well defined *amount* of the function of the product, enabling a quantitative comparison of the impacts of different products which provide the same function. For instance, the question can be asked what amount of fossil energy over the whole life-cycle is connected with a kilogram of tomatoes on the Dutch market, comparing tomatoes grown in glasshouses in the Netherlands or grown in free air in Spain but transported to the Netherlands.

This way of quantification has consequences for the type of issues that can be included in the analysis. Well fitting are impacts that have an input and/or output character (such as energy and hazardous substances), which in addition are of global relevance. Impacts on soil fertility and soil erosion, impacts on the water household and impacts on biodiversity do not (or hardly) meet these requirements; neither do requirements defined in terms of management measures (Udo de Haes, 2006). It should be noted that data for quantitative analysis, as for instance those included in the recent report 'Environmental Performance of Agriculture at a Glance' of the OECD (2008), cannot be quantified in relation to a functional unit and are generally not applicable in an LCA study. Eco-labelling therefore is not just an application of LCA; it rather is a tool which *can* or *may* apply LCA on specific points, but needs most of its information from other sources.

A very fruitful approach is to combine sustainability certification with LCA. In sustainability certification, requirements can be included in terms of management measures regarding the stage of agricultural (or marine) production; in the LCA study, impacts can be included that can be quantified over the whole life-cycle in relationship to a product-based functional unit. An example of such a combination concerns the labelling of fish products by the Swedish eco-label KRAV (Thrane *et al*., 2009). In this study it is argued that the MSC label specifies requirements that aim to avoid overexploitation of the fish resources as well as damage to biodiversity (amongst others, addressing by-catch); that is, there is a focus on issues which, for methodological reasons, cannot (presently) be dealt with by LCA. On the other hand, other aspects included in the KRAV criteria, such as the use of anti-fouling agents on boats, energy use during the fishing stage and after landing, and waste handling during the processes after landing, are activities included in an LCA study, but left out by, e.g. the MSC labelling. Using this combined approach, up until July 2009 eleven Swedish and Norwegian fisheries have been certified.

In overview, industrial eco-labelling is the only approach that includes LCA for quantitative analysis of the products over the whole value chain. Qualitative requirements of processes along the value chain are used by organic farming, integrated agriculture, regional products and fair trade, together with a Chain-of-Custody approach, preventing undue mixing during the trade chain. Only a Chain-of-Custody approach is found in sustainability certification of natural resources.

19.4 Effectiveness

This section deals with the effectiveness of food labelling approaches. We distinguish three levels to analyse this: the role of different ways in which the information can be provided; the market share of products; and factual changes in the quality of the product or of the environment. Clearly, the presented effects cannot be seen as consequences of just the labels by themselves. It is a mixture of both supply and demand factors, which together produce the result, and in which the label plays a crucial role.

19.4.1 Type of information

A precondition for effective labelling of environmental issues is that the given food must have a good quality. Thus a study in the US showed that new wine production areas, such as Colorado, cannot improve their reputation by using eco-labelling if their products are perceived by the consumers to be of low quality (Loureio, 2003). It should be added here however, that for many food products the organic label rather indicates an asset than a risk for the quality. A next question then is what type of information is the most effective for influencing purchase behaviour. A main distinction can be made between: just a label, a label together with information, or just information.

Most probably, a label can work well by itself if it is well known, such as is the case with organic produce, Fair Trade or MSC. But even for these labels additional information may be helpful. For instance, the German Blue Angel, the oldest industrial eco-labelling scheme, includes in its label information such as: '… because it is energy saving'. A study done by Hoogland *et al*. (2007) concerning 370 customers of a supermarket in Amsterdam, showed that an organic label together with information on requirements regarding animal welfare scored better than just an organic label. Clearly, the respondents did not realize that the organic label already included the additional requirements on animal welfare. It is questionable, however, whether the provision of additional information also will work for food products. Perhaps it is better that the content of the labels is explained in other media such as newsletters.

'Just information' is probably not appropriate as an instrument for business-to-consumer information. This is illustrated by a EU-wide study performed by Sleenhoff and Osseweijer (2008) on the purchase behaviour of consumers in retailer shops. From a comparison between interviews and a check of purchase receipts, it was concluded that consumers who were said to reject GM-food, still appeared to have bought GM-vegetable oil. Unlike the conclusion from the authors that consumers in reality do not mind GM-food, we suggest that this rather supports the idea that written information on food products is not observed well by consumers. For companies this may be different. A comparison can be made with Environmental Product Declarations (EPDs in Type III labelling according to ISO 14025), which are developed as a business-to-business tool for industrial eco-labelling and consist only of a written fact sheet. One can expect that for food products this may well be the same. With an EPD-like format, more complex information can be given about the product; for instance, in terms of a fixed set of quality indicators.

In the case of a label, what type of information is most effective? Should it be only positive information, as is generally the case, or should it also include negative information? In a computer-based experiment Grankvist *et al*. (2004) investigated the effectiveness of the use of green (= better than average), yellow (= average) and red eco-labels (= worse than average). Consumers with strong environmental interest equally responded on positive and negative labels, while consumers with intermediate interest appeared to be more affected by the red label. An example in practice is provided by Sainsbury, a retailer in the UK with green–yellow–red labels for the quality of food products.

19.4.2 Changes in market share

The second level that can show the effectiveness of eco-labels concerns the market share of the food products. Although more telling, unfortunately only few data are available which provide this information. Most information is available about the niche market of organic farming (http://www.agriholland.nl/dossiers/bioland/home.html#omvang). In the EU, the total area under organic cultivation amounted to 4.3% in 2006, showing a 10% increase per year. Over the whole of Europe, also including non-EU-member states, the area was 8% in that year. There are large differences, however, between the member states: Austria and Switzerland lead with 13 and 12%, showing still a small increase. The Netherlands is lagging behind with only 2%, even showing a small yearly decrease.

Industrialized countries in other parts of the world, such as California and Australia, show comparable figures. Interestingly, the largest amount of organic food products is produced in Australia, with China second (Paull, 2008). In the industrialized world, about 10% for different types of certification together seems to be the potential market share. But also in less industrialized parts of the world organic products can achieve substantial shares of the market. In China, in total 34 million ha (that is 28% of the total of 122 million ha of agricultural land) is used for eco-labelled food products, of which about three percentage points are organic, nine percentage points are Green Food (in fact integrated agriculture) and 16 percentage points are hazard-free food (beyond compliance) (Paull, 2008). There can be large differences between types of food products, however. In a study from Xu *et al*. (2008), it was shown that in the capital town of Yunnan province in western China, consumers' choice of organic vegetables amounted to 5%, but of organic meat to 28%. One may wonder why in a country like China, with such a large pressure on food production, organic agriculture takes such a relatively large share. A hypothesis is that also the lack in the availability of P-fertilizer plays a role there, a phenomenon which can also be observed in a number of developing countries.

The same range of data also emerges if one looks at single products. For instance, the different types of certified coffee together make up for 4% of the world market, with a green bean import of 8% in the US (http://www.coffeehabitat.com/2008/07/what-is-the-market-share-of-certified-coffees.html). For certified bananas, the share in Europe and the US ranges between 1.5 and 2%, but there is a strong yearly growth. In the Netherlands, fair trade bananas have a market share of 5%, in Switzerland of 10% (http://www.fao.org/DOCREP/MEETING/X1149E.html).

Comparable figures are also found for with the sustainability certification of natural resources. In 2005, the total amount of timber certified by FSC and PEFC together (including the Canadian CSA and the US based system SFI) amounted to a market share of 6% of all timber traded world wide (Udo de Haes *et al*., 2008). Of palm oil, world wide about 1.5 million tonnes of the world total of about 40 million tonnes, that is about 4%, is now certified by RSPO. Both types of certification are rapidly growing. An exception at the positive side concerns fisheries certified by MSC. The 70 fisheries engaged in the MSC programme record annual catches of over 4 million tonnes of seafood. They represent 42% of the world's wild salmon catch, 40% of the world's prime whitefish catch, and 18% of the world's lobster catch for human consumption (see www.msc.org). These certification schemes of natural resources can have a strong growing tendency due to active procurement policies of governments (see Section 19.5). An indication of this is given by the case of dolphin-friendly tuna, labelled by supermarkets, which conquered the full market both in the US and in Europe.

19.4.3 Changes in food and in the environment

The ultimate test of effectiveness concerns changes in the food and in the environment itself, connected with the use of a label. As has been stressed by van Amstel *et al*. (2008), not much data are available here, thus pointing at a general need for research on this aspect. The final proof lies in comparative studies, which compare different types of agricultural practice on end final effects.

Some data are available on the taste of products and of pesticide residues in/on products. Although no formal research is known to us, a recent Dutch radio programme presented its findings that top cooks do prefer organic meat and vegetables because of their superior taste (www.meatandmeal.nl, 15 December 2008).

Pesticide residues in food are frequently monitored in many countries. In the EU, for example, national monitoring data are gathered by the EFSA (European Food Safety Authority) and reported regularly (http://ec.europa.eu/food/fvo/specialreports/pesticides). For some commodities such as aubergines, bananas, cauliflower, grapes, orange juice, peas, sweet peppers and wheat, there are also EU co-ordinated monitoring programmes. From the national EU monitoring programmes of products from conventional agriculture in the latest report of EFSA over 2006, no residues could be detected in 54% of the samples, while a further 42% of the samples contained residues that were below or equal to the maximum residue limits (MRL) laid down in EU or national legislation. In 4.4% of the samples, residues above the MRL were found (4.7% for fresh products and 0.9% for processed products). EFSA reported a slight decrease over the years. In these reports, no information is available about residues in organic food or in food from integrated agriculture or regional production. However, since no artificial pesticides are allowed in organic farming we can assume that residues of such pesticides are absent in the corresponding food products. So here, one can assume that organic farming indeed achieves better than conventional agriculture. To a lesser degree this may probably also be true for products from integrated agriculture.

But not always do the results follow expectations. For instance, Vicini *et al*. (2008) compared organic milk, recombinant bovine somatropin (rbST) free milk, and conventional milk on their quality regarding antibiotics and bacterial counts, on the rbST concentration, and on the nutritional value (fat, proteins, solid non-fat) in 334 samples in 48 US states. They found only minor differences in all the parameters investigated and concluded that all milk on the market was wholesome.

Research on biodiversity in different types of farms is rare but is in progress. In the review study of Hole *et al*. (2005), a comparison was made at the European level between organic and conventional farming. The authors showed that organic farms had greater food availability for birds, in terms of both invertebrates and diversity of plant species. In part, this was due to greater abundance of non-crop habitats on these farms. A recent study in an intensive arable landscape in the Netherlands showed that organic farms had higher densities of skylarks and lapwings than conventional farms. The most important causing factor concerned the type of the cultivated crops (Kragten and de Snoo, 2008). Interestingly, a follow-up study (Kragten *et al*., 2008) showed that on organic farms the recruitment of ground breeding birds such as the lapwing (*Vanellus vanellus*) is lower than on conventional farms, because of nest losses due to mechanical weed control. In order to avoid these losses, organic farms should include specific nest-protecting measures in their practice.

Concluding this section, we argue that food labels can, in general, be improved by briefly indicating their scope; that in industrialized countries leading environmental and social labels together have a potential market share of about 10%; that organic products in the Netherlands have a better taste; that organic food products have EU-wide lower pesticide residues than products from conventional agriculture, and that research on the final effects of certified food products on biodiversity is rare but is in progress.

19.5 Future trends

In Section 19.1, eco-labelling was introduced as a private instrument, thus contrasting with public policy instruments. Although this is essentially true, governments do play an increasingly important role. Therefore, it is good to take this as a starting point for sketching future perspectives. Firstly, governments can set minimum standards for eco-labelling. A clear need for this can arise with the labelling of health aspects. There is no need for this if there is good health inspection, but that is not the case everywhere. For instance, in Spain the information that is to be included in fish labelling is ruled by the government (Asension and Montero, 2008). Comparably, in many countries there are governmental limits for pesticide residues in and on food, which explicitly have to be observed on labels. Where this is not the case, as for instance in Thailand (Roitner-Schobesberger *et al*., 2008), a confusing abundance of safe food labels can arise, which can also unduly compete with confident labels such as those for organic products. As described in Section 19.3.1, the semi-government organizations FAO and WHO play a role in defining general requirements for organic farming in their Codex Alimentarius. The Protected Denomination of Origin (PDO), which falls under EU regulation, forms a sound basis for regional products.

Governments also have to play a role by scrutinizing the market against undue claims. However, the main task is here in the hands of the systems themselves, by organizing independent control.

Secondly, a government can take the initiative for setting up a labelling scheme and can also support this with subsidies. For instance, Higgins *et al.* (2008) describe how the Australian government has developed a national framework for certification and labelling of agricultural products, based on the environmental management system (EMS) of ISO 14001. They exemplify this with a description of the positive functioning of this system for beef products, which is also subsidized by the Australian government. Another example concerns the initiative of, and subsidy by, the Dutch government for certification of sustainable biomass under the European Committee for Standardisation (CEN). The role as initiator and subsidizer seems to relate particularly to natural resources as presented in Section 19.2.5, and so far less to the certification and labelling of food products.

Thirdly, and most importantly, governments can play a role through their procurement policy. This is now strongly in development for sustainable natural resources, such as timber and biomass. Particularly in a number of European countries, including the UK, Germany, the Netherlands, Denmark and Sweden, public procurement policies for these products are in place or are being developed. For food products, the initiatives are as yet lagging behind. In the Netherlands, the central government has announced full sustainable procurement by national governmental agencies by 2010, including food products. As was already indicated in the introduction, such a transition from voluntary to enforced labelling can however have considerable consequences in view of the WTO (see Cheftel, 2005). In the context of the WTO requirements by governments regarding the procurement of sustainable timber or sustainable biomass can become a barrier to trade. This can be avoided if the given requirements are based either on characteristics of the products themselves, or on issues that are laid down in international treaties (such as on biodiversity in Convention on Biological Diversity). In practice, the boundary between what is and what is not acceptable by the WTO, has not yet been defined. In particular, social requirements that are not directly connected with the production process, such as attention for land use rights and the provision of social infrastructure, are regarded as barriers to trade and are seen as not acceptable for governmental procurement. This discussion has not yet been settled (see for instance Hobbs and Kerr, 2006), but for food products one may well expect that the same issues are being raised.

An illustrative example lies in the case of dolphin-friendly tuna. The catching of tuna generally led to a high by-catch of dolphins, because both dolphin and tuna species feed on the same shoals of fish. Due to high consumer pressure, the US government phased out the import of dolphin-unfriendly tuna. This government policy was sued under the WTO by tuna-producing countries, in particular Mexico. Thereupon, US retailers took over the initiative by putting a private label on the tuna tins, which then conquered the whole market. As a consequence, the tuna fisheries in Mexico underwent great losses, the other side of the coin (see also Teisl *et al.*, 2002).

Despite this limitation, we argue that governmental support of standard development together with governmental procurement policy can play a very important role in consistent further development of the labelling tool. In particular, we assume that only through such a link with governmental policy can one achieve a limitation of the confusing proliferation of label development. We will now further explore the desirability and feasibility of a general sustainability certification scheme for food products, comparable to the development of sustainability certification of fish products, of timber, and of biomass (see Section 19.2.5). The aim of this is not a niche market but a beyond-compliance level that should, in principle, be able cover a major part of the market. It aims to establish a higher base level, and may well go in hand with labels for specialized niche markets, such as organic farming, regional products or fair trade.

If we set such an aim we must realize that, in comparison with the other schemes for sustainability certification of natural resources, there are limitations of such an approach for food products. These particularly pertain to the following two issues. Firstly, the agricultural food market is more diverse than the timber market or the biomass market. All types of biomass for energy do produce ethanol or biodiesel, for a global market with a globally comparable price level. In contrast, food products show a variety ranging from wheat to steak, with highly differing production techniques and price levels. It will not be easy to define general requirements for this large variety of products. Secondly, there is not so much of a unifying external threat which urges a unified approach. Sustainable timber and biomass are needed to fight the conversion of the dwindling natural forests; and sustainable fisheries share the global threat of the depletion of fish stocks and of marine biodiversity.

Still, good health and protection of biodiversity and the environment make it worthwhile to investigate the feasibility of such a general certification scheme for agricultural products. The main characteristics may well be the following: The aim should be to cover all types of agricultural products from all types of farming (small and large, low and high intensity, individual and in cooperatives, domestic and imported products); to follow a beyond-compliance strategy, in line with the principles of Good Agricultural Practice of the EU, essentially following a People-Planet-Profit approach (see Section 19.2.5); to be organized under the authority of an official certification organization such as CEN or ISO; and to be initiated by a combined action of governments, companies and NGOs. Just as CEN TC 383 is now working on 'biomass for energy applications', the present suggestion concerns certification of 'biomass for food applications', perhaps as an extension of the present CEN committee on biomass for energy, thus including both food and non-food agricultural products.

From a methodological point of view, the focus may well be on the development of a meta-standard; that is, a standard for the certification of existing certification systems. This meta-standard should consist of a number of principles and underlying criteria and indicators, covering main aspects such as: nutrition, use of pesticides and antibiotics, water and soil protection, protection of biodiversity, animal welfare, sound health and safety conditions, land use rights, and fair prices. For each of these main aspects, criteria should be developed for the different product groups. The focus should be on the agricultural production stage, but also include directly connected upstream and downstream processes.

And last of all, a distinction must be made between the standards with full PPP coverage, and governmental procurement policy. Governments may well support the development of PPP-wide standards, but in their own procurement policy they should restrict themselves to requirements that are WTO compatible (or that may become WTO compatible, because also that is not carved in stone).

19.6 Conclusions

Eco-labelling of food products, together with the underlying certification systems, is principally a voluntary instrument, in contrast to public policy. A number of different approaches are compared, having their origin in agricultural practice (organic farming, integrated agriculture and regional production) or in other fields (industrial labelling, sustainability certification of natural resources and fair trade). In the different labeling schemes there is a varying focus with respect to human health effects, biodiversity, environmental and social issues. The fullest scope is found in labels which comply with a so-called people-planet-profit (or PPP) approach.

Eco-labels deal in different ways with impacts in the value chain that are upstream or downstream from agricultural practice. A first step concerns a control against undue mixing of certified and non-certified products. A next step is that impacts of upstream or downstream processes are included in the label in a qualitative way: 'no GM feed for the cattle', 'no child labour in the food processing'. The most elaborate is a quantitative inclusion of impacts along the value chain in the content of the label. The main tool for this is LCA, which particularly can deal with input–output like types of impact.

Different types of labelling schemes can have different marketing strategies. In addition to conventional agriculture, which has to meet legal requirements, there are two main strategies: 'beyond-legal-compliance', which essentially aims to – in due time – take over the full market; and niche markets, which essentially aim to serve only a small specific part of the market. The effectiveness of the labels is discussed at three different levels: (i) the effectiveness of the labels in influencing procurement behaviour, either of individual consumers or of companies; (ii) changes in market share; and (iii) changes in the quality of the food products or in the environment themselves. The role of governments is discussed in supporting eco-labelling and underlying certification approaches. This role can involve the setting of minimum requirements; support by providing subsidies and other types of support; and finally by establishing procurement policies. Particularly in this last role, governments can take up a strong influential role in the effectiveness of eco-labelling.

A suggestion is made for the development of a meta-standard for sustainability certification of agricultural products, as complementary to present developments in sustainability certification of biomass for energy applications. The focus should be on requirements regarding agricultural production, where relevant, supplemented with LCA on quantifiable impacts over the whole value chain.

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