DEPARTMENT OF AGRICULTURE AND ECOLOGY
FACULTY OF LIFE SCIENCES
UNIVERSITY OF COPENHAGEN

ENVIRONMENTAL ASSESSMENT OF IMPORTED ORGANIC PRODUCTS

Focusing on orange juice from Brazil and soybeans from China

PhD thesis

Marie Trydeman Knudsen

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FACULTY OF AGRICULTURAL SCIENCES
AARHUS UNIVERSITY

’Not everything that can be counted counts and everything that counts can not be counted’.
A. Einstein

SUMMARY

The import of organic products to Denmark has seen a four to five fold increase during the last seven years, due to a growing demand for organic products. Globally, the market for organic food and drinks has also more than doubled during the same period. The global organic agricultural land has concurrently expanded from 26 to 35 million hectares, mostly due to major conversions in Argentina, China, India, Australia, USA and Brazil. While Europe and North America still makes up the major markets for organic food and drink, the extraordinary growth in the organic markets offers export opportunities for both developed and developing countries. The production and export of organic products in developing countries might offer both economic and environmental benefits, though this has mainly been investigated in Europe and North America. However, long-distance trade with organic products has also given rise to a debate on the sustainability of this development especially with regard to global warming. Consumers of organic products might ask themselves what are the environmental benefits of buying organic products from e.g. South America or Asia as compared to conventional production and is this outweighed by the long-distance trade?

The overall aim of the present PhD study was to assess the environmental impacts of selected imported organic products from developing countries based on impacts during both production (as compared to conventional) and during processing and transport to Denmark. Firstly, the global trade, development and overall environmental impacts of organic agriculture and food systems were investigated to form the foundation for the further analysis. Organic soybeans from China and orange juice from Brazil were selected as relevant case studies. Secondly, environmental life cycle assessments (LCA) were conducted for the case studies in China and Brazil focusing on comparing organic with conventional cultivation and on identifying environmental hotspots of the imported organic product during production, processing and transport to Denmark. Thirdly, LCA was evaluated as a tool for evaluating the environmental soundness of organic agricultural products and a new methodology on including of soil carbon sequestration in LCA was suggested.

The PhD study shows that the increasing globalization affects both conventional and organic agriculture and food systems. Major environmental problems with regard to agriculture and food systems are identified as global warming, nutrient enrichment and pollution of water resources, reductions in biodiversity and soil degradation. The growing global trade and development within organic food and farming systems holds a potential to offer economic and environmental benefits for developing countries, but at the same time holds a risk of increasing the environmental load from long-distance transport and of pushing organic food and agricultural systems toward the conventional farming model and thereby diminishing the environmental benefits of organic farming.

The case studies in China and Brazil show that the total greenhouse gas emissions associated with imported organic soybeans from China is 429 kg CO_2 eq. per tonne soybean, whereas imported organic orange juice from small-scale farmers in São Paulo, Brazil is 424 g CO_2 eq. per litre orange juice. Transport accounts for 50-60% of the total greenhouse gas emissions from the imported organic plant products. As a comparison, transport only account for 1-15% for imported meat products, since the greenhouse gas emissions per kg meat is a lot higher than for plant products. However, the actual contribution from transport is the same. The mode of transport is a determining factor in that truck transport has a much higher greenhouse gas emission per km than sea transport. Thus, sea transport from South America (reloaded to trucks in Rotterdam) is comparable to truck transport from Italy and France and lower than truck transport from Spain and sea transport from South Africa and China (also reloaded in Rotterdam) in terms of greenhouse gas emissions per kg imported product. It should be noted that greenhouse gas emissions from the agricultural production can vary (due to inputs, yields etc.) among countries and in some cases outweigh the emissions from transport.

Comparing organic and conventional small-scale production until farm gate in the case studies, greenhouse gas emissions per kg organic product is found to be 60-75% of a comparable conventional production in the case study area. The nutrient enrichment (eutrophication) per kg organic product is 38-82% of a comparable conventional production whereas land use per kg organic product is 10-13% higher. Higher crop diversity is found on small organic compared to small conventional orange farms in Brazil, which may have a positive effect on biodiversity along with the absence of pesticides and the interrow vegetation. No differences are found in biodiversity potential in the Chinese case study except the absence of pesticides. Comparing large and small organic orange farms in Brazil, greenhouse gas emissions, eutrophication potential and copper use per hectare is found to be significantly lower on organic small-scale than on large-scale organic orange plantations.

The shortcomings of LCA as a tool for evaluating environmental soundness of agricultural products, especially with regards to including biodiversity and soil carbon changes, are discussed. The studies in Brazil and China find that including estimated soil carbon changes widens the difference in greenhouse gas emissions per kg product between organic and conventional, but there is a need for methodological development on how to estimate and include this. A methodological approach to include soil carbon changes in LCA is suggested.

SAMMENDRAG

Importen af økologiske varer til Danmark er fire- til femdoblet indenfor de seneste syv år på grund af en øget efterspørgsel på økologiske produkter. Globalt set er markedet for økologiske føde- og drikkevarer også blevet mere end fordoblet i den samme periode. Samtidig er det totale areal der er dyrket økologisk i verden øget fra 26 til 35 millioner hektar, primært på grund af store omlægninger i Argentina, Kina, Indien, Australien, USA og Brasilien. Mens Europa og Nordamerika stadig udgør de største markeder for økologiske føde- og drikkevarer, giver den ekstraordinære vækst på det økologiske marked eksportmuligheder for både udviklede lande og udviklingslande. Produktionen og eksporten af økologiske produkter i udviklingslande kan give mulighed for økonomiske og miljømæssige fordele, hvilket dog primært er undersøgt i Europa og Nordamerika, men den lange transport af økologiske produkter har også rejst en debat om bæredygtigheden af denne udvikling specielt med henblik på global opvarmning. Forbrugere af økologiske produkter kan spørge sig selv om, hvilke miljømæssige fordele der er forbundet med at købe økologiske produkter fra f.eks. Sydamerika og Asien sammenlignet med konventionel produktion og om hvorvidt dette bliver opvejet af transporten til Danmark.

Det overordnede mål med ph.d.-studiet var at belyse miljøpåvirkningerne af udvalgte importerede økologiske produkter fra udviklingslande baseret på påvirkninger både under produktion (sammenlignet med konventionel) og under forarbejdning og transport til Danmark. Den globale handel, udvikling og overordnede miljøpåvirkninger ved økologiske landbrugs- og fødevaresystemer blev undersøgt for at danne grundlag for den videre analyse. Økologiske sojabønner fra Kina og økologisk appelsinjuice fra Brasilien blev udvalgt som relevante casestudier. Dernæst blev der gennemført miljømæssige livscyklusvurderinger for casestudierne i Kina og Brasilien, der fokuserede på at sammenligne økologisk med konventionel dyrkning samt at identificere miljømæssige hotspots af de importerede økologiske produkter under produktion, forarbejdning og transport til Danmark. Endelig blev livscyklusvurdering evalueret som redskab til at vurdere miljømæssig bæredygtighed af økologiske landbrugsprodukter og en ny metode til at inkludere kulstoflagring i jord i LCA blev udviklet og beskrevet.

Ph.d.-studiet viser, at den øgede globalisering påvirker både konventionelle og økologiske landbrugs- og fødevaresystemer. De primære miljømæssige problemstillinger i forbindelse med landbrugs- og fødevaresystemer er identificeret som global opvarmning, næringsstofberigelse og forurening af vandressourcer, reduktion i biodiversitet samt forringelse af jordressourcer. Den voksende globale handel og udvikling indenfor økologiske landbrugs- og fødevaresystemer kan potentielt bidrage med økonomiske og miljømæssige fordele for udviklingslande, samtidig er der imidlertid risiko for en øget miljøbelastning fra lang-distance transport og for at økologiske landbrugs- og fødevaresystemer bliver skubbet mod den konventionelle model og dermed mindsker de miljømæssige fordele ved økologisk jordbrug.

Casestudierne i Kina og Brasilien viser, at den totale drivhusgasemission, der er forbundet med importerede økologiske sojabønner fra Kina, er 429 kg CO_2 ækvivalenter per tons sojabønner, hvor importeret økologisk appelsinjuice fra småbønder i São Paulo, Brasilien giver 424 g CO_2 ækvivalenter per liter appelsinjuice. Transport udgør 50-60 % af den totale drivhusgasemission fra importerede økologiske planteprodukter. Til sammenligning udgør transport kun 1-15 % for importeret kød, da drivhusgasemissionerne per kg kød er langt større end for planteprodukter. Det reelle bidrag fra transport er dog det samme. Transportformen er en afgørende faktor, i det transport i lastbil er forbundet med langt større drivhusgasemissioner per km end skibstransport. Skibstransport fra Sydamerika (omlastet til lastbiler i Rotterdam) er sammenligneligt med lastbilstransport fra Italien og Frankrig og lavere end lastbiltransport fra Spanien og skibstransport fra Sydafrika og Kina (også omlastet i Rotterdam) med hensyn til drivhusgasemissioner per kg importeret produkt. Det skal bemærkes, at drivhusgasemissionerne fra landbrugsproduktionen kan variere fra land til land (afhængig af input, udbytte mv.) og i visse tilfælde opveje bidraget fra transport.

Ved sammenligning af økologiske og konventionelle smålandbrug i casestudierne indtil afgrøderne forlader gården, er drivhusgasemissionerne per kg økologisk afgrøde 60-75 % af en sammenlignelig konventionel produktion i caseområdet. Forurening med næringsstoffer (eutrofiering) per kg økologisk produkt udgør 38-82 % af forureningen med næringsstoffer ved en sammenlignelig konventionel produktion, hvorimod arealforbruget per kg økologisk produkt er 10-13 % højere. Der er fundet en højere afgrødediversitet på små økologiske sammenlignet med små konventionelle appelsinbedrifter, hvilket kan have en positiv effekt på biodiversitet. Ligeledes forventes fraværet af pesticider og en øget vegetation mellem trærækkerne at bidrage positivt til biodiversiteten. Der blev ikke fundet nogen forskelle i biodiversitetspotentialet i det kinesiske casestudie, bortset fra fraværet af pesticider. Ved sammenligning af økologiske små og store økologiske appelsinbedrifter er der fundet signifikant lavere drivhusgasemissioner, eutrofiering og kobberforbrug per hektar på små sammenlignet med store økologiske appelsinplantager.

Svaghederne ved LCA som et redskab til at evaluere miljømæssig forsvarlighed af økologiske produkter er evalueret, specielt med henblik på at inkludere biodiversitet og jordens kulstofændringer. Studierne i Brasilien og Kina viser, at forskellen i drivhusgasemissioner per kg økologisk og konventionelt produkt bliver forøget, når jordens kulstofændringer inkluderes i beregningerne, men der er behov for metodeudvikling med henblik på at estimere og inkludere dette. En metodemæssig tilgang til at estimere og inkludere jordens kulstofændringer i livscyklusvurderinger er blevet udviklet og beskrevet.

PREFACE AND ACKNOWLEDGEMENTS

This PhD study provides a contribution to the debate on the environmental implications of the growing global trade with organic products and organic versus conventional agricultural production.

The thesis is based on four papers. Paper I provides an overview of the trends and implications of the increasing globalisation of agricultural systems, which has also affected organic food systems. It was concluded that the environmental effects of the growing global trade with organic products needs to be addressed. A statistical overview of imports of organic products to Denmark was then conducted. This inventory was used to select the case studies. Paper II and III provides environmental assessments of two case studies of organic products being imported to Denmark; soybean from China and orange juice from Brazil, in a life cycle perspective. The need for developing the methodology with regard to inclusion of soil carbon changes in the life cycle assessments was identified, which is the point of departure for Paper IV.

I am very grateful to be allowed to do this work, which I have enjoyed very much.

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Marie Trydeman Knudsen
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PAPER II: KNUDSEN MT, YU-HUI Q, YAN L & HALBERG N (2010) ENVIRONMENTAL ASSESSMENT OF ORGANIC SOYBEAN (GLYCINE MAX.) IMPORTED FROM CHINA TO DENMARK: A CASE STUDY. JOURNAL OF CLEANER PRODUCTION 18: 1431-1439.

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LIST OF APPENDICES

1 INTRODUCTION

1.1 Globalisation in the organic food chain and environmental sustainability

Organic farming and food systems have seen a rapid development during the last decade towards increasing global trade with organic products. This development has given rise to a discussion of the sustainability in the organic food system also with regard to the organic principles. Paper I (Knudsen et al., 2006) provides an overview of the trends and implications of the increasing globalisation of the world's agriculture and food systems, which has also affected organic food systems. An update of the developments with organic agricultural areas and markets during the last four years is given in Box 1.1.

Globalisation of the organic food systems means that organic products are not necessarily consumed locally and that long-distance trade with organic products is a growing reality. Thus, the organic food system has over the past two decades been transformed from a loosely coordinated local network of producers and consumers to a globalised system of formally regulated trade which links socially and spatially distant sites of production and consumption (Raynolds, 2004). Although preferences for local organic food persist, Northern countries are generally increasing their organic imports.

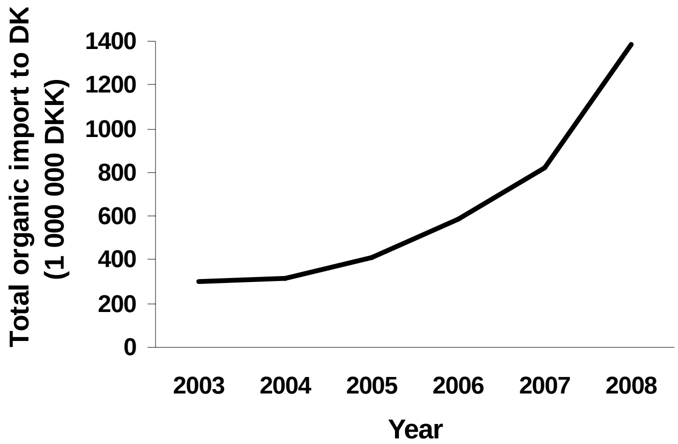


Figure 1.1 Total import of organic products to Denmark (StatBank Denmark, 2010)

As an example, the sales of organic products in Denmark have more than doubled by value from 2003 to 2008 (StatBank Denmark, 2010), while the import of organic products to Denmark has seen a 4 to 5 fold growth during the same period (Figure 1.1)

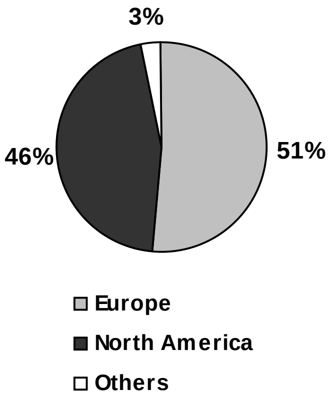
The extraordinary growth in the major organic markets in Europe and North America offers export opportunities to developing countries and can be seen as a development pathway with economic and environmental benefits for the farmers, regions and countries in the South (e.g. Twarog, 2006; FAO, 2001). The environmental benefits have until now mainly been investigated in the European and North American context (e.g. Mondelaers et al., 2009; Gomiero et al., 2008), and the growing global trade with organic products, which also implies long-distance trade has given rise to a debate on the sustainability of this development especially with regard to global warming and the carbon footprint of organic products (e.g. Soil Association, 2007; Rigby and Bown, 2003). In this debate, the long-distance transport resulting from global trade with organic products is said to be challenging the basic principles of organic agriculture (IFOAM, 2005). Organic agriculture is based on four principles of Health, Ecology, Fairness and Care. The Principle of Ecology states that '*Those to produce, process, trade or consume organic products should protect and benefit the common environment including landscapes, climate, habitats, biodiversity, air and water*'. It is further stated that '*Inputs should be reduced by reuse, recycling and efficient management of materials and energy*' (IFOAM, 2005). Long-distance transport is seen as challenging this principle of efficient use of energy that adds to global climate change. Furthermore, the principle of recycling nutrients is challenged by the global trade with organic products. In addition to the debate on environmental impacts it is also argued that the globalized market for organic products limits small-scale farmers in entering the market due to costs of certification (Barrett et al., 2002) and demands for large and stable quantities to be delivered (Kledal, 2009; Blanc, 2009), which at the same time tend to favour large-scale organic farms. This could be seen as challenging the Principle of Fairness (IFOAM, 2005). The push towards large-scale organic farming and possible 'conventionalisation' at the farm level might also affect the environmental sustainability of organic agricultural production (Darnhofer et al., 2010), with regard to e.g. greenhouse gas emissions, nutrient enrichment, and biodiversity.

This leads to the question of what kind of agricultural development urban consumers in the North and South support when choosing organic products and what are the environmental implications. How much does the long-distance transport mean for greenhouse gas emissions associated with the imported organic products – and how do the organic systems in the producing countries affect the environment compared to the conventional systems?

The environmental impacts need to be addressed both at the farm level and along the food product chain. Life cycle assessment (LCA) is a method for such integrated assessment of several environmental impacts (e.g. global warming and eutrophication potential). The strengths and weaknesses of LCA and how well the tool captures the environmental sustainability and differences between farming systems will be evaluated.

Box 1.1 Development of organic area and markets from 2003-2008.

The global markets for organic food and drink have grown remarkably during the last decade and global sales of organic products have more than doubled from 2003 to 2008 (Figure 1.2).



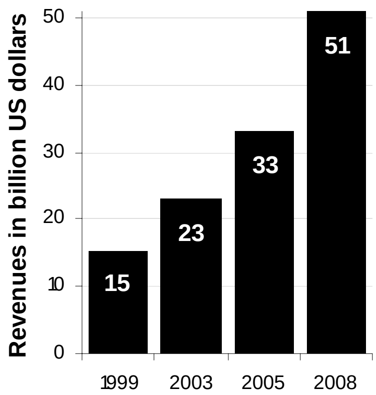


Figure 1.3 Global revenues by region 2008 (left) and global market growth for organic products (right) (Willer &Kilcher, 2010).

The major markets for organic food and drink are still Europe and North America, which account for 97 percent of global revenues (Figure 1.2). Interestingly, Danish consumers are the world's leading buyers of organic food with comprising more than four percent of total food and drink sales (Willer & Kilcher, 2010). The Asian market for organic food and drinks are also growing, especially in the more affluent countries, such as Japan, South Korea, Taiwan, Singapore and Hong Kong. Furthermore, domestic markets are developing in some major cities in Latin America (Willer & Kilcher, 2010). Major northern markets offer good prospects for suppliers of organic products and vast areas of agricultural land are located far away from the major markets. The organic agricultural land has increased from 25.5 million hectares in 2003 to 35.0 million hectares in 2008 (Willer & Kilcher, 2010), mostly due to increasing organic areas in Argentina, China, India, Australia, USA, Brazil and Spain (Willer & Yussefi, 2005). Oceania is the continent with the largest area of organically managed land (12 mill. ha), followed by Europe (8.2 mill. ha) and Latin America (8.1 mill ha) (Willer & Kilcher, 2010). Figure 1.3 presents the countries with the most organic agricultural land in 2008. More than 95% of the agricultural land in Australia is however managed as permanent grassland, which is also the case for parts of the land in Argentina.

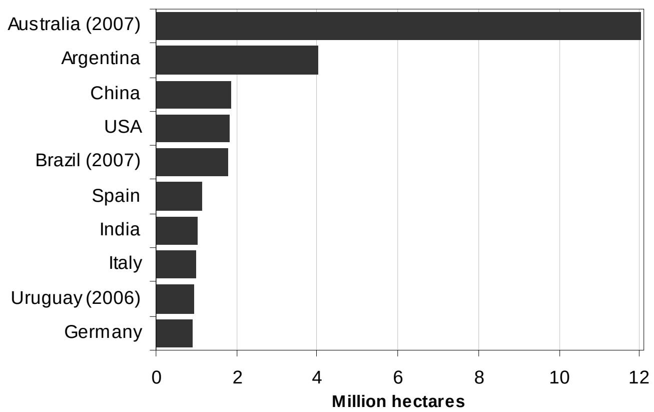


Figure 1.3 The ten countries with the most organic agricultural land in 2008 (Willer & Kilcher, 2010).

One-third of the world's organically managed land is located in developing countries, with Argentina, China and Brazil having the most organic land (Willer & Kilcher, 2010).

1.2 Aim of the project

The overall aim of the present PhD study is to assess the environmental impacts of selected imported organic products produced by smallholders in developing countries. The work is based on evaluation of the selected products and product chains using a product oriented environmental assessment approach, especially Life Cycle Assessment (LCA). Thus, impacts during both production (as compared to conventional) and during processing and transport to Denmark are evaluated.
The objectives are:

- To investigate the global trade, development and related environmental impacts of organic agriculture and food systems to form the basis for the further analysis. Specifically, the import of organic products to Denmark is identified.

- To assess the environmental profile in a food chains perspective of selected organic food products produced by smallholders and imported to Denmark.
  - To identify the critical points in the product chains contributing the most to the environmental impact and resource use and thereby evaluate the effect of long-distance transport.
  - To compare organic with conventional agricultural production at farm level.

- To assess the LCA methodology with regard to usefulness for environmental assessment of organic products.

1.3 Methodological approach and delimitations

The PhD project was based on three tasks:

Task 1: Overview of the development of organic food and farming systems to form the basis for the research questions and the base for selecting the case studies (Paper I). Establish an overview of the import of organic products to Denmark (unpublished working paper and section 2.1).

Task 2: Case studies in China and Brazil (Paper II and III)

Task 3: LCA as a tool for environmental assessment (Paper IV and conference paper in appendix 8.4)

Task 1 was based on a literature review of relevant literature of the development and challenges within organic agriculture and food systems. This also implied a mapping of the flow of imported organic products to Denmark based on information from Danish official statistics (StatBank Denmark, 2010), supermarkets, retailers and literature.

Task 2 was based on a case study and life cycle assessment approach. The case study approach was chosen to be able to follow a real, relevant and typical imported organic product and to be able to acquire in-depth and detailed information on the actual farming practices and product chains (Midmore et al., 2006). Focusing on two cases enabled a cross-case comparison for identification of common trends. The focus on case studies also implies that the results should be interpreted with caution with regard to generalization. It is outside the scope of the present thesis to generalise on the sustainability of production and export of all organic products, due to the selection of a few products and the diversity of the organic sector. The life cycle assessment approach was chosen as the most appropriate method to estimate the environmental effects of long-distance trade (Garnett, 2003) and to compare organic and conventional production at the farm level in an integrated assessment of several impact categories, despite weaknesses of the method with regard to certain impact categories such as biodiversity and land use (Reap et al., 2008a; Geyer et al., 2010a; Knudsen & Halberg, 2007). The impact categories global warming, eutrophication, acidification potential was assessed along with land use and non-renewable energy use. Aspects with regard to biodiversity were qualitatively described. The environmental impacts associated with production, processing and transport to Denmark were assessed. The impacts during further use of the product were not included. Social and economic aspects of sustainability were not included in the assessment.

Task 3 was connected to the use of LCA in the case studies in task 2. Thus, the strength and weaknesses of the LCA methodology with regards to organic agricultural products were evaluated on the basis of the case studies and literature. The suggested methodology was also based on challenges during the case study and the use of soil carbon modelling tools.

1.4 Outline of the thesis

The thesis is based on four papers. Paper I provides an overview of the trends and implications of the increasing globalisation of agricultural systems, which has also affected organic food systems. It was concluded that the environmental effects of the growing global trade with organic products needs to be addressed. Paper II and III provide environmental assessments of two case studies of organic products being imported to Denmark; soybean from China and orange juice from Brazil, in a life cycle perspective. The need for developing the methodology with regard to inclusion of soil carbon changes in the life cycle assessments was identified, which is the point of departure for Paper IV.

The PhD thesis is based on three main sections; introduction, methodology and results and discussion. The introduction presents the background and aim of the thesis. The section on methodology describes the selection and characteristics of the case studies and the environmental assessment methodology. Finally the results and discussion present a synthesis of three main discussions; 1) The environmental impacts of organic versus conventional products, 2) The relative importance of transport for greenhouse gas emissions per kg imported product, and 3) A discussion of LCA as a tool for organic agricultural products. The thesis is finalised by a conclusion and outlook.

2 METHODOLOGY

2.1 Selection and characteristics of the case studies

In order to answer the research questions, the countries and products in the case studies were chosen based on the following criteria: 1) Developing country with a large area and potential for organic farming in which the organic agricultural sector is rapidly developing with regard to domestic markets and farmed land, 2) organic products that are imported to Denmark and represent one of the main important organic exported products from the country, which are produced by small-scale organic farmers and 3) products that represent both food and fodder and processed and non-processed products.

2.1.1 Import of organic products to Denmark

In the Danish supermarkets you will find organic apples from Argentina, orange juice from Brazil and red pepper from Israel. Going to the farms, the Danish organic animals are fed with e.g. organic rape seed from Italy and soybeans from China (Figure 2.1).

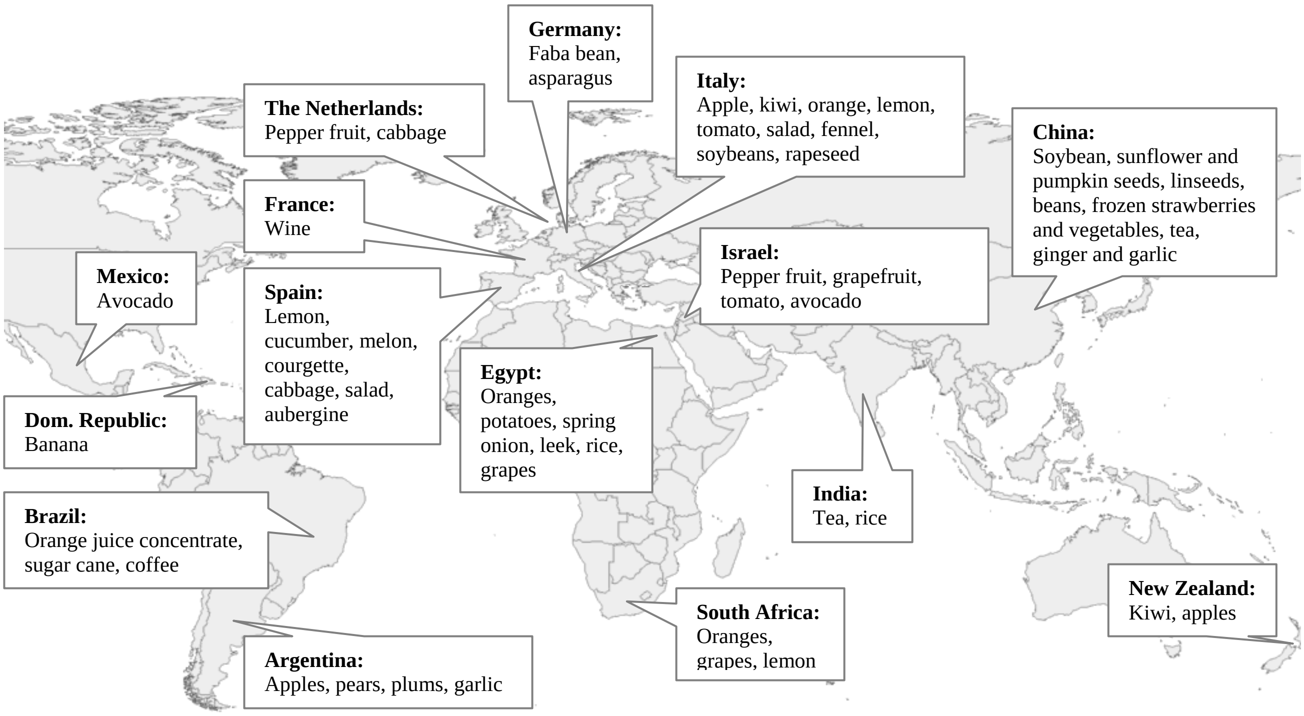


Figure 2.1 Import of selected organic food, drinks and fodder to Denmark in the period 2006-2010 (based on information from supermarkets, retailers and Danish official statistics (StatBank Denmark, 2010)).

Interestingly, the rapidly growing imports to Denmark consist mainly of fruit and vegetables, sugar, rice and cereals, whereas the export from Denmark is mainly animal based products. The same pattern is seen in the UK, where 82% of the organic fruit and vegetable sales were met by import in 1999, while only 5% of the organic meat sales were imported (Barret et al., 2002). This can be partly explained by the short season for fruit and vegetables in the North. In Denmark, both organic imports and exports have been growing during the last seven years, but the imports grew from being comparable to twice as high as the exports during that period (StatBank Denmark, 2010). The main share of the imports comes from Europe, followed by South America and Asia. However, it should be noted that the numbers from StatBank Denmark (2010) do not always indicate the country of origin, since the last country from which the product enters Denmark statistically is recorded as the import country. E.g. Chinese products imported to Germany and sold again to Denmark will be classified as imported from Germany and not from China. The share of imports coming

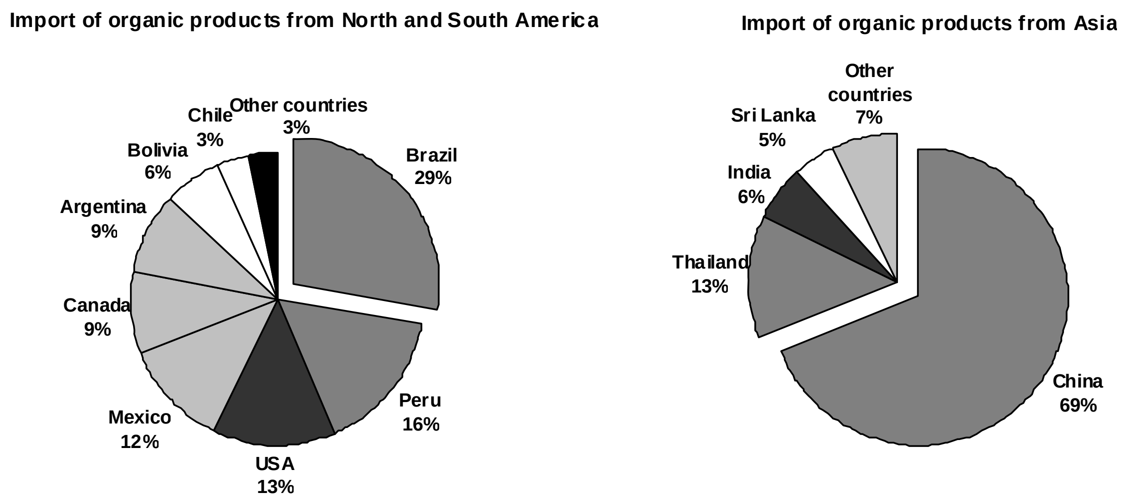
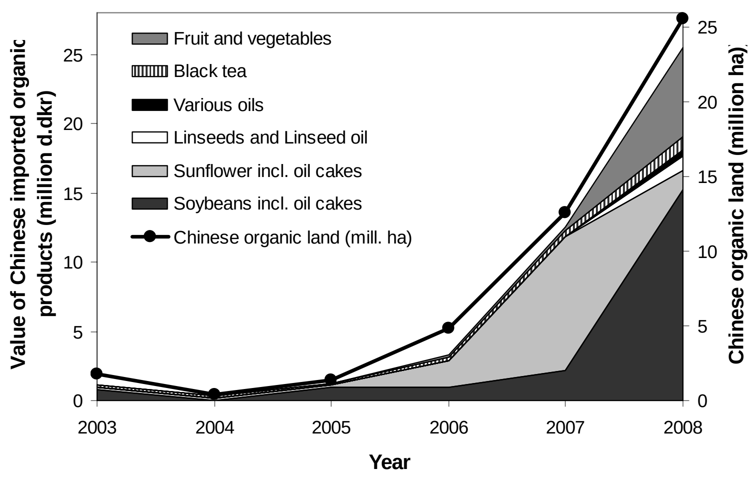


Figure 2.2. Import of organic products from North and South America and Asia to Denmark in 2007 (StatBank Denmark, 2010).

from Europe is therefore overestimated. The organic imports from North and South America and Asia are presented in Figure 2.2. As shown in Figure 2.2, Brazil is a dominating actor in Denmark's organic imports from North and South America (approx. 30%) in monetary terms. With regard to Denmark's import of organic products from Asia, China is the main actor representing the highest value of imported organic products from Asia (approx. 70%). Brazil and especially China have both seen major expansion in organic agricultural land during the last seven years and are in the top five of countries with the most organic land (Box 1.1). At the same time, both countries have a rapidly developing organic market and sector (Willer & Kilcher, 2010) On the basis of this information, Brazil and China were chosen as appropriate countries for the case studies.

2.1.2 Soybeans from China

China is by far the country in Asia with the most land under organic management (1.85 million hectares), followed by India (1.02 million hectares) and Kazakhstan (0.09 million hectares) (Willer & Kilcher, 2010). Figure 2.3 presents the main imported organic products from China to Denmark in the period from 2003-2008 along with the expansion in Chinese land under organic management.



Please note that products from China that are imported via another country will not be classified as imported from China.

The category 'Fruit and vegetables' are primarily frozen strawberries and vegetables.

Figure 2.3. Imported organic products from China registrered by StatBank Denmark (pers. com. Agnete S. Nilsson) and the land under organic management in China (Willer & Yussefi, 2005; 2006; 2007; Willer et al., 2008; Willer & Kilcher, 2009; 2010).

Soybeans and sunflower products are the major imported organic products from China, which are registered in StatBank Denmark (2010) until 2008, where a considerable import of frozen fruit (strawberries) and vegetables are registered (Figure 2.3). Likewise, Sheng et al. (2009) reported that the main organic export products from China were grains, beans, fruit and vegetables, accounting for 90% of the organic exports. Based on this information, soybean was chosen as a relevant product for the case study.

Soybeans were one of the first domesticated food crops and were actually first cultivated in China around 6000 years ago. Nowadays, soybeans are primarily cultivated to provide edible oil and high-protein animal feed (Clay, 2004). The organic soybeans imported to Denmark are primarily for livestock feed (pers. comm., Henrik Kløve, DLG and Agnete S. Nilsson, StatBank Denmark). However, soybeans imported to Europe are also used for human consumption to produce dairy substitute products.

Organic soybeans, sunflower and other seeds and beans are generally produced in the Northern provinces of China such as Jilin, Heilongjiang, Inner Mongolia and Liaoning whereas the frozen vegetables primarily are produced in the Shandong province South of Beijing (Kledal et al., 2007). The three North-eastern provinces of Heilongjiang, Jilin and Liaoning are considered as the major organic agricultural production base, holding 35% of the number of certified organic products and 38% of the organic enterprises, due to natural conditions that are favourable to organic farming (Sheng et al., 2009).

China’s organic products are mainly exported to Europe, North America, Japan and Australia and the export value of organic products has increased rapidly during the last five years (Sheng et al., 2009). The primary driver of Chinese organic food and farming is trade and export; even though Chinese domestic markets for organic products are developing (Yin et al., 2010). While organic food sold in domestic Chinese supermarkets are mainly certified by local certifiers that comply with China National Organic Product Standard, the organic export products are also certified by international certification agencies such as ECOCERT, BCS and Soil Association some of which that have already established their own branch offices in major Chinese cities (Sheng et al., 2009).

2.1.3 Orange juice from Brazil

A glass of orange juice is a part of a complete breakfast for many Danish consumers, which also accounts for one of the six pieces of fruit and vegetables that every Dane are recommended to eat daily. The consumption of organic orange juice has more than tripled during the last five years (StatBank Denmark, 2010). Generally, a rapidly growing consumption of orange juice has been reported from the UK (Garnett, 2006).

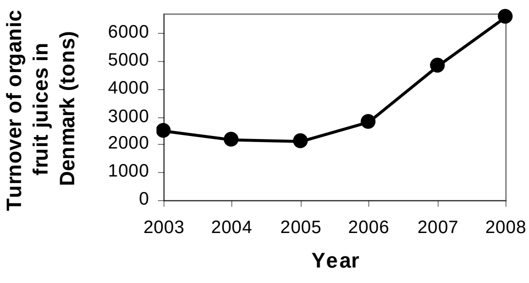


Figure 2.4. The turnover of organic fruit juice in Denmark (tons) from 2003 to 2008 (StatBank Denmark, 2010).

The major part of the conventional orange juice comes from Brazil (pers. comm. Carina Jensen, Rynkeby Foods). Brazil dominates the market of frozen concentrated orange juice (FCOJ) with a share of more than 80 percent in total world trade and most of the FCOJ exports originate from the State of São Paulo (Neves, 2008). Until 2009, the major part of the orange juice sold in Denmark also came from Brazil, but the organic orange industry encountered problems and the organic orange juice concentrate is at the moment bought in Mexico instead (pers. comm. Carina Jensen, Rynkeby Foods, 2010). The organic orange juice concentrate bought in Brazil (pers. comm. Carina Jensen, Rynkeby Foods, 2010) is not visible in the data recorded by StatBank Denmark (2010) since the orange juice concentrate was imported through another country and as such does not appear as imported from Brazil.

However, orange juice concentrate along with cane sugar were until 2008 the two major organic products imported from Brazil to Denmark (pers. comm. Agnete S. Nilsson, StatBank Denmark, 2010). As a comparison, Barret et al. (2002) also reported that 70% of the tonnage of imported organic products from Brazil to the UK was citrus products (concentrated orange juice and oranges) with the remaining being cashew nuts and cane sugar. Most of the Brazilian orange juice originates from the State of São Paulo (Neves, 2008), which is also the production area for most of the organic orange juice that was sold in Denmark (per. comm. Carina Jensen, Rynkeby Foods, 2010). The main organic products produced in the State of São Paulo and destined for the export market are sugar, orange and coffee, while the production of organic vegetables are mainly for the domestic markets (de Abreu et al., 2009; Blanc, 2009). Based on the above-mentioned information, organic orange juice from Brazil was in 2006 chosen as a relevant product for the case study. The orange production in the case study area in São Paulo is further described in Paper III.

Most of the organic farms in the State of São Paulo are small- to medium-sized with the export markets being the primary driver for the certified organic sector (de Abreu et al., 2009, Blanc, 2009). Around 70% of the total certified organic production (sugar, orange, coffee etc.) in Brazil are exported to mainly Europe, North America and Japan (Blanc, 2009). However, there is a strong Brazilian organic movement with the point of departure in agroecology that has promoted organic farming especially with regard to social issues (Blanc, 2009). The Brazilian government introduced in 2003 legislation on organic farming to regulate organic farming and also as a social project oriented towards small-scale agriculture (Blanc, 2009). Due to this focus on small-scale farms versus large-scale farms, the size of the farms were included as a factor in the analysis of organic orange production for juice in Brazil (Paper III). Despite the national legislation in Brazil, internationally accredited organic certification is still demanded for the export market.

2.2 Environmental assessment methodology

In order to choose an appropriate method to assess the environmental impacts related to imported organic products, the major relevant environmental impacts related to food and farming systems should first of all be identified. Secondly, the most appropriate method should be selected.

2.2.1 Environmental impacts and indicators in food and agriculture

The environmental impacts related to global agriculture and food systems are discussed in Paper I (Knudsen et al., 2006). The major drivers for the environmental pressure on ecosystems are identified as deforestation, increased use of energy, reactive nitrogen and pesticides and over-exploitation of land resources in terms of soil degradation. The major environmental impacts are identified as global warming, pollution such as eutrophication and related to the use of pesticides and antibiotics, reductions in biodiversity and loss of land resources by soil erosion and degradation (Knudsen et al., 2006). In accordance with this, UN (2010) identifies the overall main stresses on ecosystems as climate change, interference with nitrogen cycle, water use and biodiversity loss, which is also in general agreement with the OECD (2004) key environmental indicators. Likewise, Rockström et al. (2009) suggest that humanity has transgressed three planetary boundaries: for climate change, rate of biodiversity loss and changes to the global nitrogen cycle. In the Principles of Organic Agriculture, the same environmental issues are mentioned to be protected including climate, landscapes, habitats, biodiversity, air, water and soils (IFOAM, 2005).

The above-mentioned environmental impacts are more or less related to agriculture and food systems and the relation between agriculture and food systems and three of the main environmental issues are given below.

Climate change is a result of emissions from several sectors in society. Baumert et al. (2005) estimated the global flow of greenhouse gas emissions by sector, activity and associated emitted gas (CO_2, N_2O and CH_4) in 2000 and found that agriculture accounted for 13.5% and transportation accounted likewise for 13.5% of total greenhouse gas emissions globally. More than one third of the emissions from agriculture are soil emissions (mainly N_2O) and one third is methane (CH_4) emissions primarily from ruminant's enteric fermentation. The remaining third comes from rice production, biomass burning and other (Baumert et al., 2005). Food and drinks, transport and housing are the three main contributors to the greenhouse gas emissions for EU countries. Food production and consumption contribute around 22-31% of the total GHG emissions for EU countries (Foster et al., 2006 based on Tukker et al., 2006), whereas transport contributes around 15% with private cars as a major contributor (Tukker et al., 2006). According to EEA (2010), Denmark had in 2008 a greenhouse gas emission of around 12 t CO_2 per capita, compared to the EU range of 5-15 t CO_2 per capita (except Luxembourg having an emission of almost 26 CO_2 per capita). Climate change as a single environmental impact category has gained considerable focus also due to the United Nations Climate Change Conference in 2009 in Copenhagen (COP15).

Pollution with nutrients and other substances are more directly linked to the agricultural practices, as further described and discussed in Paper I. The main driver of the human modification of the N cycle is industrial fixation of atmospheric N_2 to ammonia (~80 Mt N per year) followed by agricultural N_2 fixation via cultivation of leguminous crops (~40 Mt N per year), fossil fuel combustion (~20 Mt N per year) and biomass burning (~10 Mt N per year) (Rockström et al., 2009). The reactive N pollutes waterways and coastal zones and ads to local and global atmospheric pollution (Rockström et al., 2009). Eutrophication-associated dead coastal zones have been reported from both Brazil and China (Diaz & Rosenberg, 2008). According to OECD (2008), pollution of rivers, lakes, and aquifers exceeding recommended limits for drinking water remains a problem and excess levels of nitrate, phosphorus or pesticides were found in more than one out of 10 monitoring sites in 13 OECD countries. OECD (2008) furthermore stated that it is costly to treat pesticide- and nutrient-contaminated water to bring it up to drinking standards. Furthermore, contamination of coastal waters remains a major problem in most regions as nutrients cause rapid growth of algae and damage marine life. Pesticide use and nitrogen balance surplus have decreased in e.g. Denmark (which has also reduced greenhouse gas emissions) since 1990, while it has increased in other parts of the world (OECD, 2008).

Biodiversity loss is interlinked with both climate change and pollution with nutrient and pesticides, which are both causing a pressure on biodiversity and ecosystem services. Biodiversity is here considered as being the diversity of species, genes and ecosystems. Despite the extra focus given to biodiversity in 2010, which is the UN's International Year of Biodiversity, it does not change the fact that the global 2002 target on a significant reduction in biodiversity loss in 2010 has not been met. Biodiversity loss is caused by several pressures. The Secretariat of the Convention on Biological Diversity (2010) have identified five main pressures on biodiversity; 1) Habitat loss and degradation, such as conversion of wild land to agriculture, drainage and aquaculture, 2) Climate change, such as loss of arctic sea ice and pressure on ocean acidification, 3) Excessive nutrient load and other forms of pollution, such as pollution from nutrients and other sources as a threat to terrestrial and inland water and coastal ecosystems, 4) Over-exploitation and unsustainable use, such as overexploitation of marine fish stocks, 5) Invasive alien species. It is apparent that agriculture plays an important role in the pressures on biodiversity.

Water use remains an overall stress on ecosystems, but it has not gained much attention in the present thesis since none of the studied imported organic products were irrigated or grown in regions with a critical shortage of water.

The aim of assessing the environmental impacts related to imported organic products is based on an overall striving towards environmental sustainability (World Commission on Environment and Development, 1987; Dalgaard et al., 2006) of our food consumption pattern (Smith, 2008; Wallgren & Höjer, 2009). The environmental impacts and indicators are exploring the environmental aspects of sustainability and a multidisciplinary approach is needed (Hadorn et al., 2006) to capture a broad picture of the environmental impacts related to imported organic agricultural products. Thus, the environmental assessment method should be able to operate in a food supply chain context and deal with multiple environmental impacts.

2.2.2 Selection of environmental assessment methodology

Several methods are available for the evaluation of the environmental impacts of agriculture, such as Life Cycle Assessment (LCA), Ecological Footprint and Emergy Analysis (van der Werf et al., 2007; Thomassen & de Boer, 2005; Halberg et al., 2005; Finnveden & Moberg, 2005; van der Werf & Petit, 2002). The choice of methodology is important since the results not only depend on the characteristics of the analysed system, but also on the methodology used (van der Werf et al., 2007). However, the aim of the analysis is crucial to the choice of methodology. Since the present PhD thesis is aiming at evaluating multiple environmental impacts of imported organic products, including greenhouse gas emissions from long-distance transport, it requires a certain methodology to handle this challenge. Furthermore, the methods should be able to express environmental impacts both per unit products and per unit area, which were the case for four out of five environmental assessment methods evaluated by van der Werf et al. (2007) (including LCA and Ecological Footprint). Halberg et al. (2005) showed that only two (LCA and Ecological Footprint) out of six evaluated environmental assessment tools takes greenhouse gas emissions into account. The Ecological Footprint methodology calculates the areas (land or water ecosystem) required to produce the resources used and absorb the waste generated from a studied object (Wackernagel & Rees, 1996). The studied object could in principle be any type of object, but the methodology has mainly been used on regions, nations, project (Finnveden & Moberg, 2005) or at farm level (van der Werf et al., 2007; Thomassen & de Boer, 2005; Halberg et al., 2005). However, the Ecological Footprint methodology cannot be used to express the greenhouse gas emissions directly per unit product, which is also the case for Emergy Analysis. Emergy Analysis describes the accumulated energy associated with the total inputs needed of energy, materials, information and labour using emergy equivalents (Finnveden & Moberg, 2005). Moreover, these methods aggregates all resource use and environmental impacts into one indicator, which makes it difficult to interpret differences between food systems, farms etc. and thus to learn from the results (whether as a producer or a consumer). Based on this review, LCA was chosen as the most appropriate method for the given objective of the study despite weaknesses of the method. Garnett (2003) also concluded, when exploring the relationship between food, transport and CO_2, that an LCA approach was needed. The British carbon footprinting guideline, PAS 2050 (2008), is also based on LCA.

2.2.3 Life Cycle Assessment (LCA)

The LCA methodology is a tool to assess the environmental impacts from a product through the product chain. Recently, attempts have been made to include social or economic aspects into LCA, but traditionally the main focus has been on environmental aspects. The aim of including several environmental impacts and focusing on the product chain is to avoid problem-shifting e.g. from one life cycle phase to the other, from one region to the other or from one environmental problem to another (Finnveden et al., 2009).

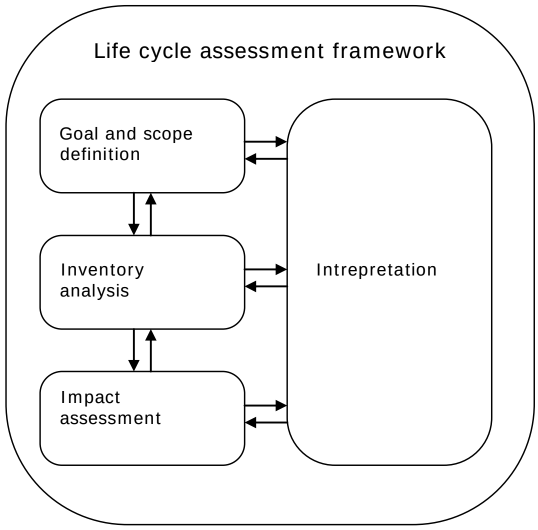


Figure 2.5. The four phases in the life cycle assessment (LCA) methodology.

LCA is an invaluable tool for assessing greenhouse gas emissions related to a product, but it is also a very data intensive and time consuming tool and there are still some challenges especially with regard to land use that needs to be addressed (Finnveden et al., 2009). These will be further discussed in section 4.3.

The LCA methodology consists of four phases as presented in Figure 2.5. Each phase of the LCA implies several methodological choices that may affect the final results differently even though the ISO guidelines are followed (Finnveden et al., 2009). In the following, the main aim of each phase is mentioned along with major methodological choices within each phase.

Goal and scope definition
This important phase includes the purpose of the study (incl. definition of functional unit), scope definition (incl. choice of system boundaries and procedure for co-product allocation), methodology (incl. choice of impact categories and key assumptions) and data collection (strategy for data collection and data quality).

The *functional unit* are related to the studied object. As the LCA is a product-oriented methodology focusing at the product chain, the environmental impacts are normally expressed per unit product, such as mass, energy or protein (Roy et al., 2009). However, the environmental impacts can also be expressed per unit area and the question whether the environmental impacts should be expressed per unit product or unit area has given rise to debate (van der Werf et al, 2007). Some aspects are more related to the local or regional level (such as land use in terms of biodiversity, soil and eutrophication), while other are more focused on the global level (such as energy use and greenhouse gas emissions) (Halberg et al., 2005). Haas et al. (2000) have argued that expression per unit area is more appropriate for local /regional impacts (such as eutrophication), whereas for global impacts (e.g. climate change) impacts should be expressed per unit product. De Koeijer et al. (2002) prefer expression of impacts per unit area to take the carrying capacity of the environment into account. Van der Werf et al. (2007) recommend the expression of impacts both per unit area and per unit product. The LCA's performed in the present PhD thesis focused on impact per mass unit, but impacts per area were also given in the LCA of orange juice (Paper III). If organic yields are lower, the difference between environmental impacts of organic and conventional products are normally reduced when estimating the impacts per mass unit compared to unit area.

The *system boundaries* for an agricultural product may either be limited to the farm gate or through processing to regional distribution centre or even follow the product to the end user and disposal. The end user stage with regard to food may differ with regard to how the food are handled in consumers kitchens (e.g.

boiled, roasted, fried or raw). Several studies of agricultural products end at farm gate (e.g. Thomassen et al., 2008; Halberg et al., 2010; Casey & Holden, 2006), while other studies end at regional distribution centre (Hospido et al., 2009; Williams et al., 2008; Saunders et al., 2006). In the present PhD thesis the imported organic products are followed through the product chain until the fodder company or retail distribution centre. For the comparison of organic and conventional products, the results at farm gate are given as recommended by Hospido et al. (2010).

According to the ISO guidelines on LCA (14040 and 14044), the *allocation procedure* (where the unit process cannot be divided into subprocesses) may either be 1) expanding the systems to include the additional functions related to the co-products or 2) allocation according to e.g. mass, economy or energy. In the present PhD thesis, the system was expanded with regard to e.g. orange residues from the orange juice concentrate factory. Furthermore, a system expansion approach was used with regard to handling the manure (described in Paper II).

The environmental *impact categories* concerning the effect on climate change (global warming potential), water and air pollution (eutrophication, acidification) are commonly used in agricultural LCA's, while effects on e.g. biodiversity or ecotoxicity due to pesticides are only rarely assessed, due to methodological difficulties (e.g. Thomassen et al., 2008; Halberg et al., 2010) (as discussed further in section 3.3). In the present PhD thesis, the impact categories global warming potential, eutrophication, acidification and non-renewable energy use are included quantitatively, while pesticide toxicity and impacts on biodiversity are estimated qualitatively. Since the objective of the PhD study was focused on environmental impacts, social and economic impact categories were not included. Furthermore, the implementation of these socio-economic impacts in LCA is still in its infancy (Griesshammer et al., 2006, Jørgensen et al., 2008).

In the present thesis, the *data collection* was based on site-specific information from farmers and processors in the countries concerned, literature and relevant databases.

Inventory analysis

The life cycle inventory is the most time consuming and work intensive phase, since it includes data collection and treatment of data (Roy et al., 2009). Site-specific data are needed from the production and processing stage, while other data on such as electricity and transport can be found in databases. Furthermore, emissions need to be estimated. In the present thesis, emissions were estimated based on IPCC guidelines (IPCC, 2006), nutrient balances and literature on site specific data. Emissions, inputs and outputs were presented (in Paper II and III) on a yearly basis. Such static annual time perspectives are commonly used in LCA. However, emissions related to one year's activities are not always restricted to the same year, which is the case for soil carbon emissions. Implementation of a more dynamic time perspective in LCA is still in its infancy (Levasseur et al., 2010). The choice of whether the time perspective can be static or should be dynamic depends on the objectives and the studied objects. An annual static time perspective was used in Paper II and III, while a more dynamic time perspective was used for Paper IV, due to the inclusion of soil carbon changes. Another challenge is to obtain representative data based on several farms as a foundation for the analysis. In the present PhD thesis, the aim of assessing organic smallholders entering the global market (which in itself implies challenges for the smallholders) and the infancy of the organic production and markets, the number of farms found for the relevant products has been very restricted. In the case of orange juice in Paper III, this implies that any statistical differences are hard to obtain. Nevertheless, the data on organic orange juice represents the actual organic orange production from small- and large-scale farms in Brazil.

Impact assessment

The impact assessment implies characterization where the emissions are assigned to the relevant impact categories, converted into the main unit used in the impact category concerned and aggregated with other relevant emissions within the same impact category. This process is described in section 2.2 in Paper III (since it was submitted to a journal not specifically focused on LCA).

In the present PhD thesis, the result from each environmental impact category is presented individually (characterized results), which is required according to the ISO guidelines on LCA (14040 and 14044). A further valuation of the results that aggregates the individual scores into a single number is not required by the ISO guideline, but it is useful for communicating a simple message to the public and decision-makers. However, a single number also implies further uncertainty of the results due to simplifications, hidden assumptions and implicit value judgement (van Passel et al., 2007) and can lead to differing results (Daniel et al., 2004).

Interpretation
The fourth phase implies an evaluation of the results, including sensitivity analysis and conclusions. In the present PhD thesis, uncertainty is handled mainly by using sensitivity analysis. Furthermore, the statistical variation in results due to variation between farming practices are presented in Paper III.

3 RESULTS AND DISCUSSION

3.1 Organic versus conventional – what are the environmental impacts?

A number of studies have evaluated the general environmental impacts of organic versus conventional products and farming systems, mainly in the European or North American context. Paper I provides a short general overview of the environmental impacts of organic versus conventional food and farming systems. Since Paper I was published, a number of studies comparing environmental impacts of organic versus conventional food and farming systems have been published, but the conclusions have not changed markedly (e.g. Mondelaers et al., 2009; Gomiero et al., 2008). Furthermore, studies specifically in the Australian (Wood et al, 2006) and Canadian (Lynch, 2009) context have been published.

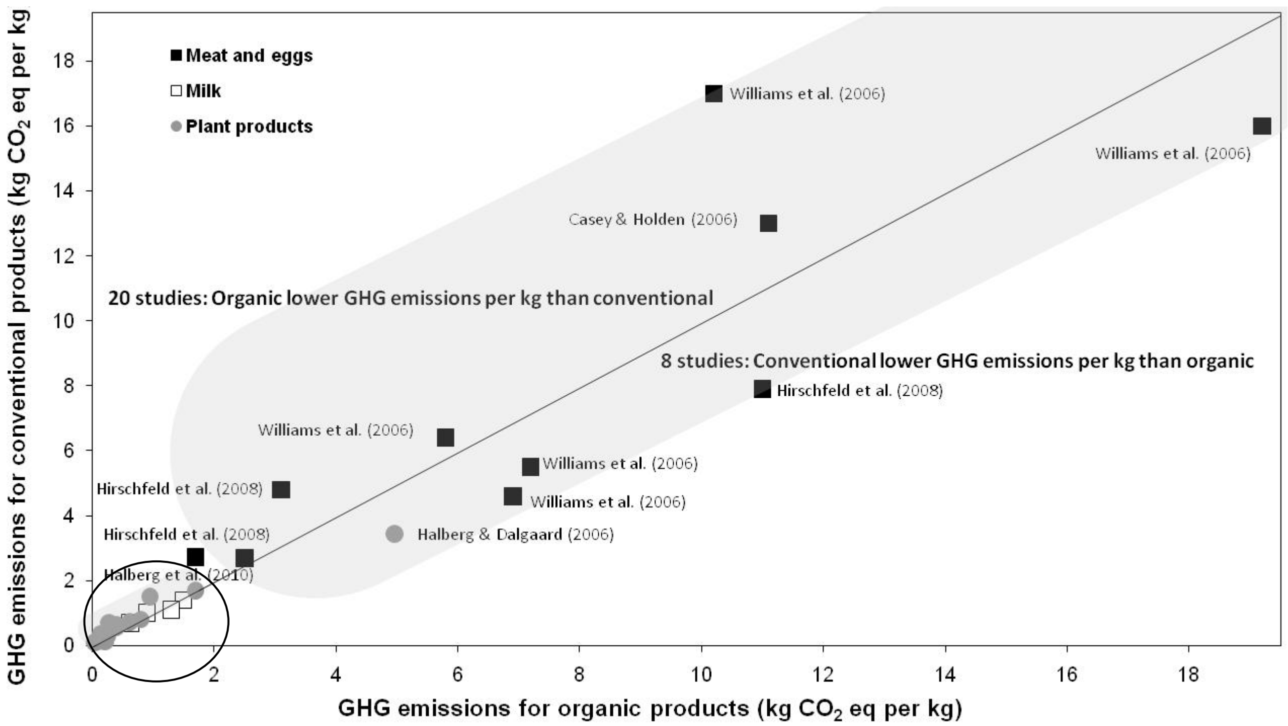
Overall, the conclusions are that soils in organic farming systems have on average a higher content of organic matter (e.g. Mondelaers et al., 2009; Fliessbach et al., 2007; Mäder et al., 2002). Concerning biodiversity, organic farming contributes positively to agro-biodiversity and natural biodiversity (e.g. Mondelaers et al., 2009; Bengtsson et al., 2005; Hole et al., 2005), but it also depends on the type of organic agriculture and type of landscape. Furthermore, the risk of pesticides accidents and pollution is absent. With regard to the impact of the organic farming system on greenhouse gas emissions and nitrate and phosphorous leaching the conclusion is not that straightforward. When expressed per production area organic farming performs better than conventional farming for these items (e.g. Mondelaers et al., 2009). However, due to generally lower yields of organic farming, at least in developed countries, this positive effect expressed per unit product is less pronounced or not present at all (Mondelaers et al., 2009).

3.1.1 Greenhouse gas emissions

Greenhouse gas emissions for organic and conventional products is presented in Figure 3.1 that presents the results from a review of LCA studies, including the case studies in the present thesis (Paper II and III).

The case studies in China and Brazil find that greenhouse gas emissions from organic products at farm gate are 60-75% of a comparable conventional production in the case study area, which is in agreement with other studies (Figure 3.1). Interestingly, the study furthermore revealed that the inclusion of estimated soil carbon changes in the sensitivity analysis widens the difference in greenhouse gas emissions per kg product between organic and conventional. However, there is a need for methodological development on how to estimate and include soil carbon changes in LCA, which has traditionally not been included. This is the theme for Paper IV, which is discussed further in section 3.3 in the present thesis. Nevertheless, the results indicate that at least organic plant products might perform better than indicated in Figure 3.1, where most of the studies did not include soil carbon changes.

Figure 3.1 furthermore shows, the relative importance of farming system (organic vs. conventional) and product type (plant vs. meat products) for greenhouse gas emissions. Beef has the highest values followed by lamb, pork, poultry and eggs (Figure 3.1). To reduce the greenhouse gas emissions related to the consumption of food, the replacement of meat by plant products means more than replacing conventional products with organic ones. Interestingly, the combination of eating organic and at the same time also eating less meat might often be the case. A survey by a Danish supermarket shows that of the meat consumption by consumers that does not buy organic is twice as high (172 g meat per day) as the consumers buying much organic food (86 g meat per day) (FDB, 2010). Similar patterns might be found for canteens converting from conventional to organic food, due to higher prices of organic food and especially meat, implying a lowering of the meat proportion in the diets.



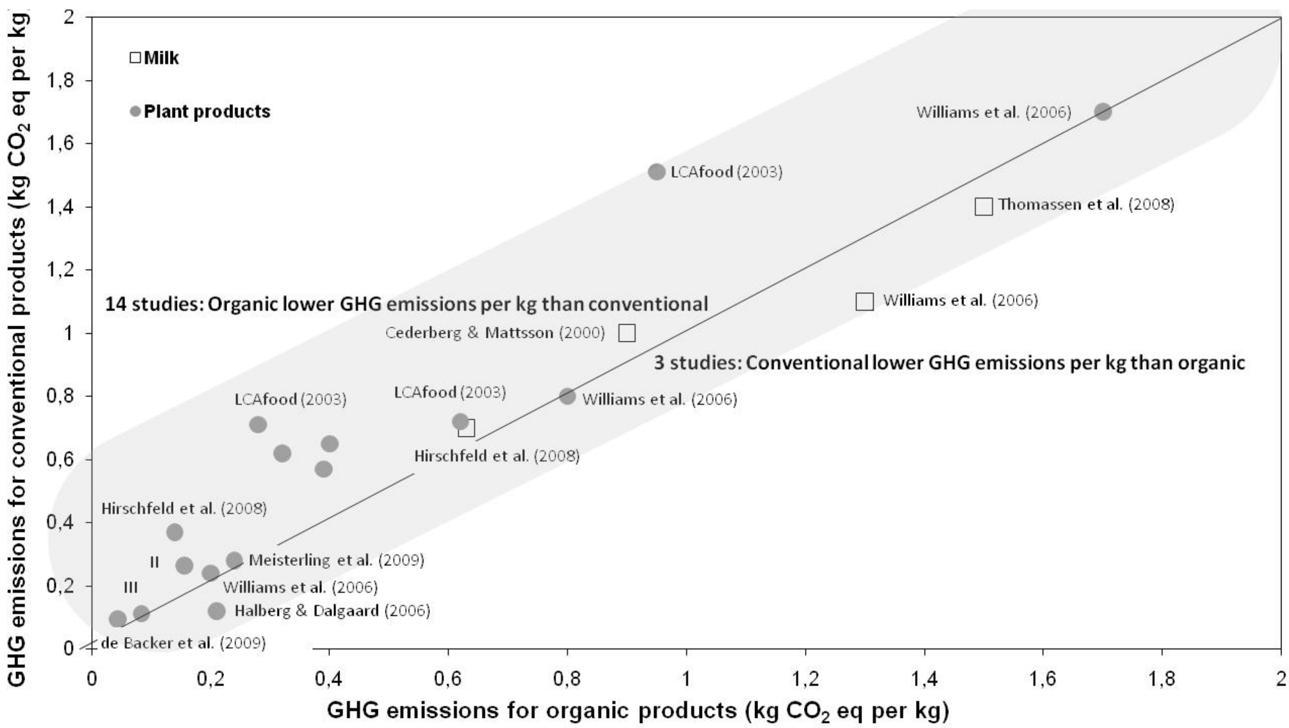


Figure 3.1 Literature review of greenhouse gas (GHG) emissions per kg organic and conventional product, including the farm gate results from the case studies assessed in Paper II (Knudsen et al., 2010) (II) and Paper III (Knudsen et al., submitted) (III). Organic perform better above the line and conventional perform better below the line. Idea after Niggli et al. (2008). The upper graph contains the total number of LCA studies, whereas the lower graph is a zoom of the studies of milk and plant products from the upper graph.

3.1.2 Nutrient enrichment, land use and biodiversity

The case studies in China and Brazil show that the eutrophication related to organic products are 38-82% of the comparable conventional production whereas land use per kg organic product is 10-13% higher. With regard to land use, higher crop diversity is found on small organic compared to small conventional orange farms in Brazil, which may have a positive effect on biodiversity along with the absence of pesticides and the interrow vegetation. No differences are found in biodiversity potential in the Chinese case study except the absence of pesticides. The differences in biodiversity potential among organic farms and practices suggests that the organic regulations should be more explicit concerning biodiversity aspects, if organic agriculture want to strive towards protecting *'...landscapes, habitats and biodiversity...'* as stated in the organic principles (IFOAM, 2005). Comparing large and small organic orange farms in Brazil, greenhouse gas emissions, eutrophication potential and copper use per hectare are found to be significantly lower on organic small-scale than on large-scale organic orange plantations, which also indicates possibilities for improvements within the organic farming practices. This suggests that future research should focus on differences within organic agriculture; between different systems and between farms with similar systems.

3.2 Food transport – what is the relative importance for greenhouse gas emissions?

The increasing globalisation of agricultural systems, which has also affected organic food systems, was discussed in Paper I (Knudsen et al., 2006). Here it was highlighted that the environmental effects of the increasing global trade with organic products needs to be addressed.

3.2.1 From food miles to life cycle assessments

The transport of the food we eat, often referred to as food miles, have been widely debated during the last 10-15 years, especially in the UK (e.g. Kemp et al., 2010; Coley et al., 2009; Weber & Matthews, 2008; Pretty et al., 2005; Smith et al., 2005; Garnett, 2003; Sustain, 1999). Within the organic sector, this issue of 'food miles' has also been widely discussed (Woodward et al., 2002; Rigby & Brown, 2003), especially with regard to the dilemma of airfreighted agricultural products from developing countries (Soil Association, 2007), which has also given rise to a new term 'fair miles' (MacGregor & Vorley, 2006) referring to the aim of development and fair trade with regard to developing countries (Edwards-Jones et al., 2009).

Garnett (2003) estimated that food transport accounts for 3.5% of UK's consumption related GHG emissions and concluded that a Life Cycle Assessment is needed. However, this number does not take transport of products outside the UK into account. Oxfam (2009) estimated transport to account for 12% of greenhouse gas emissions from food consumption (Figure 3.2)

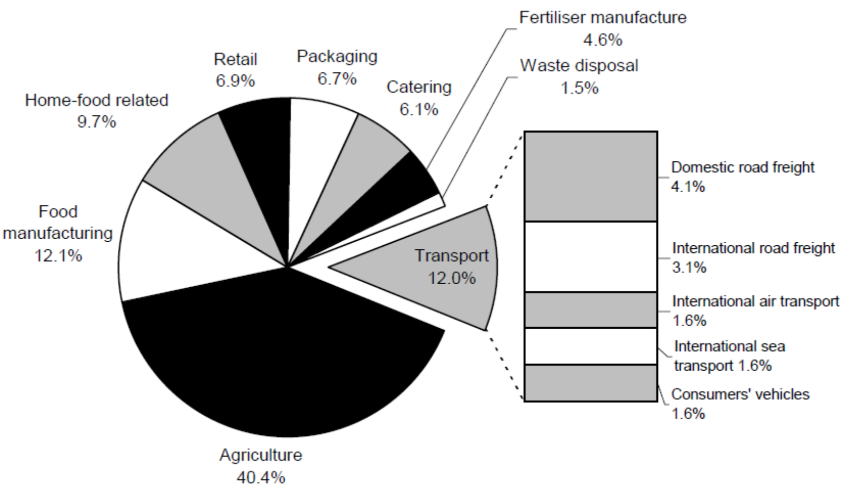


Figure 3.2 Greenhouse gas emissions from UK food consumption (Oxfam, 2009)

The validity of food miles was examined in a report by Smith et al. (2005) concluding that food transport has significant and growing impacts, but a single indicator based on total food kilometres is an inadequate indicator of sustainability. Essentially, the environmental impact of an agricultural product in a life cycle perspective depends on the sum of the following impacts:

a) Impacts before and during agricultural production, which depends mainly type of product (e.g. plant or meat product), type of production system (e.g. organic or conventional; outdoor or greenhouse production) and site of production.
b) Impacts during processing.
c) Impacts during transport, which depends mainly on transport mode and transport distance.
d) Impact during retail handling, home transport and processing, which are normally not included (Sim et al., 2007; Williams et al., 2008; Hospido et al., 2009).

3.2.2 The impact of transport mode

The impact on greenhouse gas emissions on different types of products and the effect of organic or conventional have been discussed in section 3.1, showing that the impact of product types (plant or meat products have much greater influence on greenhouse gas emissions than the effect of producing either organic or conventional. The impacts during transport depends on the distance, but even more on the transport mode, since air freight has a much higher impact per km than sea freight, as illustrated in Figure 3.3.

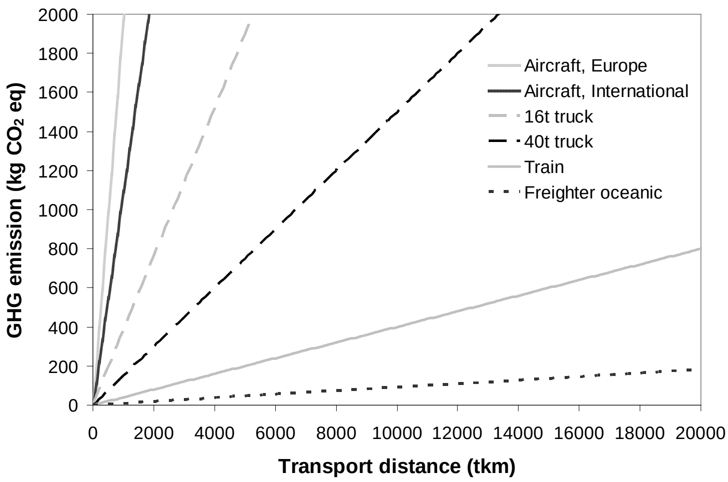


Figure 3.3 Greenhouse gas (GHG) emissions from food transport as affected by transport mode and distance (Ecoinvent Centre, 2007)

From these data (Ecoinvent Centre, 2007) it can be estimated that importing organic apples to Denmark from either Italy (by 40t truck from Rome to retail distribution centre (RDC) in Aarhus) or from Argentina (by ship from Buenos Aires to Rotterdam and by 40t truck to RDC in Aarhus) would amount to:

Apples, Italy (Rome): 2053 km x 150 g CO_2 eq. per tkm = 308 kg CO_2 eq. per t product

Apples, Argentina (Buenos Aires): (1000 km x 40 g CO_2 eq. per tkm)
(11718 km x 9 g CO_2 eq. per tkm) +
(834 km x 150 g CO_2 eq. per tkm) = 271 kg CO_2 eq. per t product

This simple calculation indicates that the greenhouse gas emissions associated with transport from Italy by truck is higher than from Argentina by ship. However, based on those calculations it is impossible to determine which apple has the largest environmental impact, since the contribution from the production phase for organic apples in Argentina and Italy also might differ, depending on yields, N-input, diesel consumption, storage time etc.

3.2.3 Contribution from transport for imported agricultural products

Relative contribution from transport: a review

Since the food miles report by Smith et al. (2005) concluding that food kilometre is an inadequate indicator of sustainability, a number of LCA studies on imported products have been carried out. Paper II and III in the present thesis are contributing to those studies. The life cycle assessment studies of organic soybeans from China (Paper II) and organic orange juice originating from Brazil (Paper III) imported to a regional distribution centre in Denmark revealed that transport accounted for 51-57% of the total greenhouse gas emissions. For organic orange juice, especially the truck transport of fresh oranges in Brazil and reconstituted orange juice in Europe contributed to this number, indicating that reducing the transport of 'water' in either oranges or reconstituted orange juice will improve the carbon footprint of orange juice. This furthermore indicates that the orange juice from concentrate has a potentially lower contribution from transport compared to freshly-squeezed orange juice consumed in Denmark.

Figure 3.4 gives an overview of the relative greenhouse gas contribution of the transport stage as a function of the total greenhouse gas emissions for the results from Paper II and III along with other relevant studies of imported agricultural products. All the other studies found are from the UK.

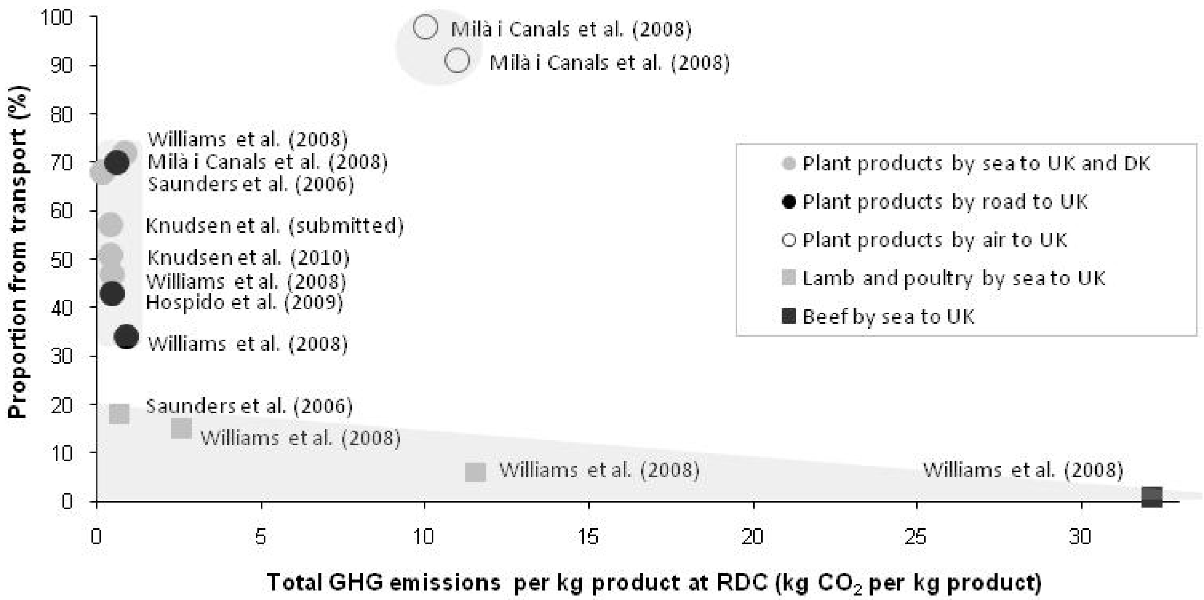


Figure 3.4 Proportion from transport of total greenhouse gas (GHG) emission per kg imported product at regional distribution centre (RDC) – a literature review. Further details are found in Table 8.2 in Appendices.

It is apparent from Figure 3.4 that the relative importance of transport for the GHG emissions of a product varies very much (1-98%) depending on mainly the product type (meat vs. plant products) and the transport mode (sea vs. air transport). Transport accounts for approximately 1-20% for meat products transported by sea, whereas transport accounts for approximately 35-75% for plant products imported by sea or road. For air-freighted plant products, transport accounts for 90-99% (Figure 3.4). However, the actual contribution of transport to the total GHG emissions for a product at the regional distribution centre (which can be found by multiplicating the percentage to the total GHG emission of the product in Table 8.2 in appendices) depends only on the distance and transport mode – and is in theory the same whether it is meat or apples that are being transported (assuming that the required conditions during transport are the same).

Actual contribution from transport for imported organic products to Denmark
Figure 3.5 presents the estimated actual transport contribution to the GHG emissions of organic agricultural products that are imported to Denmark, based on transport mode and distance. It is assumed that they are transported from the capital city or the centre of the country of origin to a retail distribution centre in Aarhus. The transport from Europe are assumed to be transported by road in 40t trucks, whereas the products from the remaining countries are assumed to be transported by sea and reloaded in Rotterdam harbour to 40t trucks. It is assumed that the organic agricultural products have the same required conditions during transport and additional energy for cooled or refrigerated transport is not included.

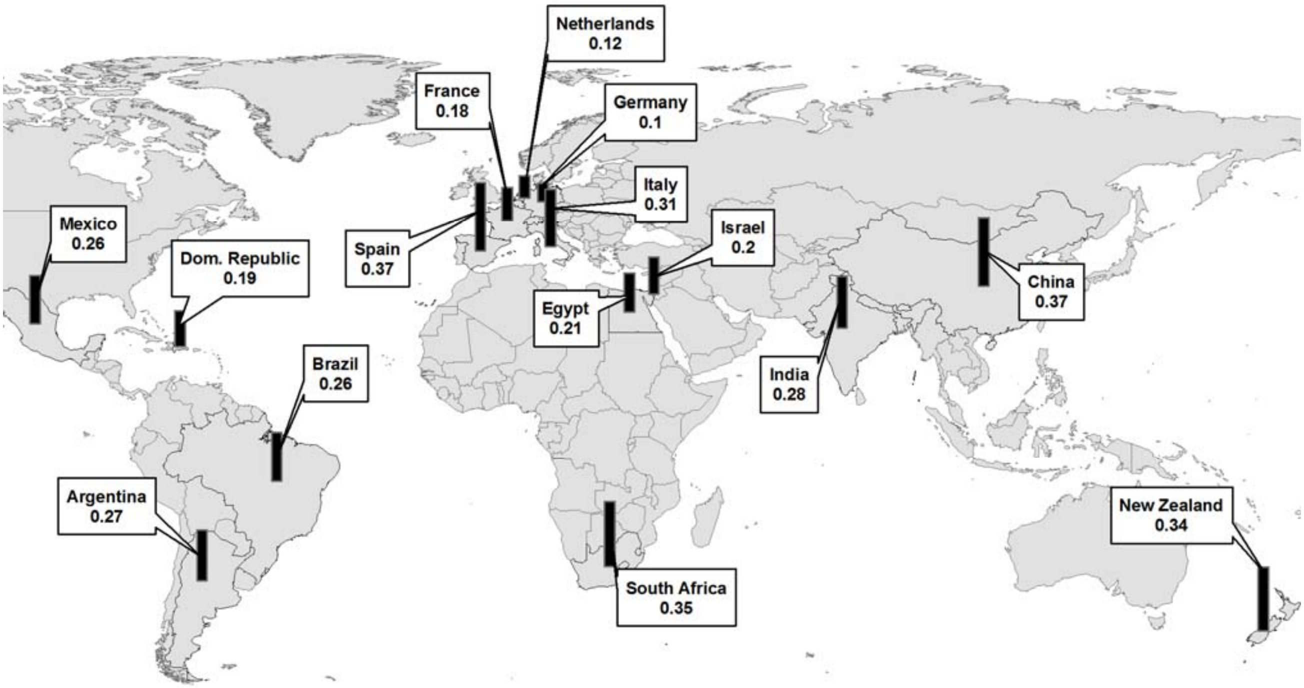


Figure 3.5 Actual contribution from transport (kg CO_2 eq. per kg product) when importing organic products to Denmark (Figure 2.1) from selected countries. It is assumed that road transport are in 40t trucks (0.15 kg CO_2 eq./tkm), sea transport by freighter oceanic (0.009 kg CO_2 eq./tkm) and sea transported goods are reloaded to trucks in Rotterdam harbor (Ecoinvent Centre, 2007). Further details are given in Table 8.3 in Appendices.

It is apparent from Figure 3.5, that the estimated GHG emission related to transport does not vary considerably depending on whether the organic product are transported from Southern Europe by truck or transported by ship from South America, Africa or Asia. The total GHG emissions per kg product are not given in Figure 3.5 and depends mainly on the contribution from the production stage, how large N-input, fuel consumption etc. and whether it is produced outdoor or in heated greenhouses, which can make a large difference as discussed in van Hauwermeiren et al. (2007).

Greenhouse gas emissions from retail to plate
In the present study, environmental impact until regional distribution centre was assessed. The last part of the chain from retail to plate have been assessed for only a few food items by Milà i Canals et al. (2008) (broccoli, green beans and lettuce) and by Mattsson & Wallen (2003) (potatoes). In the case of cooked vegetables, the home stage may have a considerable contribution. Milà i Canals et al. (2008) found a contribution of approximately 1 kg CO_2 eq. per kg broccoli or green beans, while Mattsson & Wallen (2003) estimated the home storage and cooking of potatoes to account for 0.09 kg CO_2 eq. per kg potatoes. However, for e.g. lettuce the contribution from home stage was negligible (Milà i Canals et al. (2008) as it would also be for orange juice. With regard to the home transport of the food, Mattsson & Wallen (2003) estimated that transport to home accounted for 0.05 kg CO_2 eq. per kg potatoes. As a comparison, Oxfam

(2009) estimated consumer's vehicles to account for 1.6% and home-food related to account for 9.7% of total greenhouse gas emissions form UK's food consumption (Figure 3.2).

Greenhouse gas emissions from domestic and imported food
Some of the imported organic products can also be produced domestically, such as apples, tomatoes, salad etc. The choice of whether to choose the domestically produced or imported products depends on the greenhouse gas emissions related to the products concerned during production, storage and transport. Domestic outdoor food production usually has lower greenhouse gas emissions per unit in the season, while imported produce may have lower environmental impacts if the domestic production uses heated or lit glasshouses or if the domestic produce are stored cold several months out of the domestic season (Milà i Canals et al., 2008). For apples consumed in Germany, Blanke & Burdick (2005) showed that the energy requirement of imported apples from New Zealand is 27% higher compared to German apples stored at 1^{o}C in five months. Milà i Canals et al. (2007a) also studied domestic versus imported apples and concluded that there are similarities in primary energy use of European and New Zealand apples during European spring and summer, while imported apples have a higher energy requirement during European autumn and winter, when local apples carry no CO_2 burden from cool storage.

Finally, it should be noted that greenhouse gas emissions are not the only environmental impacts related to transport. Furthermore transport also implies socio-economic impacts. Associated impacts of transport such as road building, noise and accidents are other impacts not considered here.

3.3 LCA as a tool for agricultural products – what are the challenges?

LCA is recognised as the best tool for assessing the life cycle impacts of products (Finnveden et al., 2009), especially with regard to greenhouse gas emissions. However, the LCA methodology also has some shortcomings as every other method. With regard to agricultural products, and especially organic products derived from slightly more complex systems, several challenges are identified. First of all, not all impact categories are well covered in a typical LCA due to a need for further methodological development (Reap et al., 2008b). Impact categories regarding land use, including biodiversity and soil, are problematic and needs to be improved (Finnveden et al., 2009). Secondly, some aspects are still in its infancy of being developed and implemented in already existing impact categories, such as implementation of soil carbon changes and direct and indirect land use change. Improvements in the methodological development of certain impact categories to be included in LCA may be achieved through further interaction with related fields (Finnveden et al., 2009; Horne & Grant, 2009), such as soil carbon research as in Paper IV. Thirdly, LCA of agricultural products implies some further challenges with regard to estimating the environmental costs of manure and the lack of data for the production and use phase of specific pesticides (Reap et al., 2008b).

In the present PhD thesis, the difficulties in integrating biodiversity in LCA are reviewed and discussed in a conference paper (Knudsen & Halberg, 2007) included in appendix 8.4. Furthermore, the aspects of estimating and implementing soil carbon changes is the aim of Paper IV: 'A methodological approach to include soil carbon changes in LCA'. Finally, the environmental costs of manure when used for plant production are discussed in further detail in Paper II on LCA of organic soybeans from China.

3.3.1 Biodiversity and other challenging impact categories in LCA

As mentioned in 3.1, research has shown that some of the main differences between organic and conventional agriculture are found with regard to increased soil organic matter, biodiversity and the absence of pesticides. Thus, when comparing organic and conventional products using LCA, the shortcomings of LCA with regard to estimating the effects on biodiversity, soil and including pesticides become even more visible.

The commonly used impact categories, such as global warming, eutrophication and acidification are all based on emissions from the system. However, for biodiversity and soil the impacts are not necessarily caused by emissions, but rather changes in land use within the systems, which is one of the complicating factors for those impact categories. Furthermore, more than one single indicator might be relevant for biodiversity (such as plant diversity, faunal diversity, connectivity etc.) and soil quality (such as soil organic matter, soil structure, soil erosion, soil pollution with copper etc.).

Biodiversity and land use

Agricultural land use affects several environmental impacts such as biodiversity, landscapes and soil quality, which are all impact categories not usually covered by LCA and with no widely accepted assessment method (Milà i Canals et al., 2007b). A simple indicator of land use is commonly used in LCA's of agricultural products, which is also the case for Paper II and Paper III. However, different kinds of land use have different impacts on biodiversity, landscapes and soils. E.g. one hectare of intensive wheat production does not have the same impact on biodiversity as one hectare grazed meadow, as illustrated in Figure 3.6.

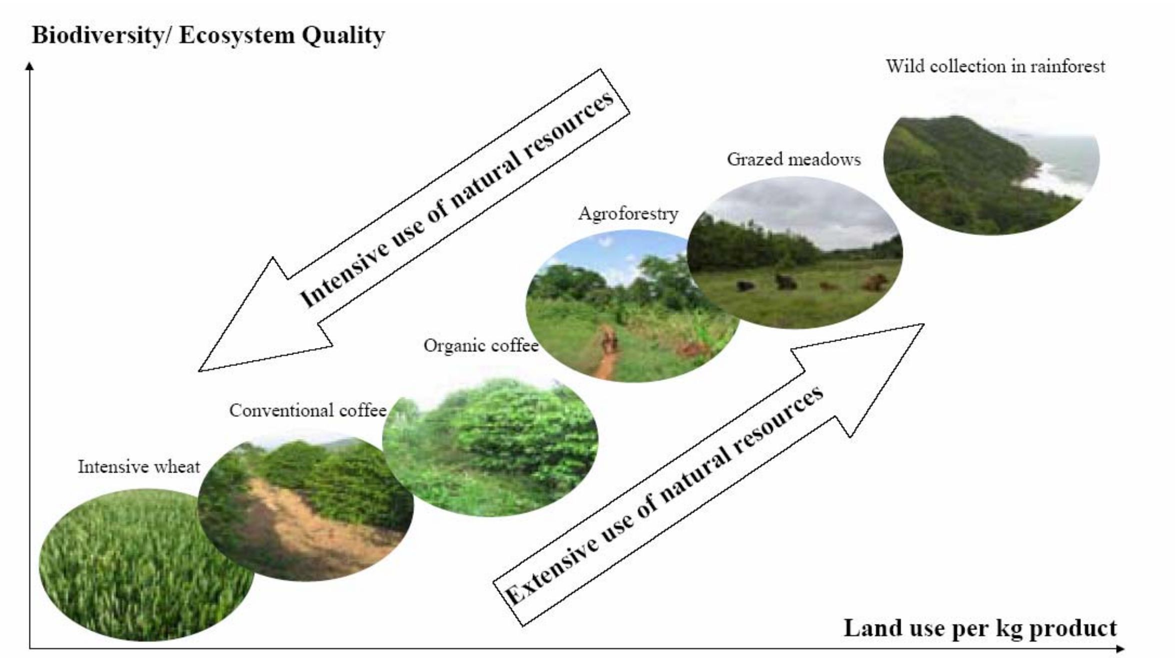


Figure 3.6 Illustration of the relationship between land use per mass unit and biodiversity or ecosystem quality (Knudsen & Halberg, 2007).

Environmental effects of biodiversity and landscapes are challenging to describe satisfactorily by a single or multiple indicators due to both geographical and cultural differences (Halberg et al., 2005; Geyer et al., 2010a), while at the same time the method should be able to produce a meaningful measure that can be practically implemented in LCA (Geyer et al. (2010a). Furthermore, the impact on biodiversity is local and site-specific compared to an impact category as global warming potential (Geyer et al., 2010a). Knudsen & Halberg (2007) (appendix 8.4) provides a short review of some of the suggestions for biodiversity assessment found in the literature. Since 2007, Milà i Canals et al (2008) have described the key elements in a framework for land use impact assessment in LCA. Schmidt (2008) has suggested a method for life cycle impact assessment of biodiversity focusing on plant species richness and categories of land use types (organic and conventional are not included), giving values for Northern Europe and Southeast Asia. Furthermore, Geyer et al (2010a & b) have suggested a method for coupling GIS and LCA for biodiversity assessments based on habitat types and four biodiversity impact indicators in a Californian context. None of the suggested methods can directly be applied to distinguish between organic and conventional land use in the case studies in the present PhD study, since it was not included in the land use or habitat types and due to the geographical context. Jeanneret et al. (2008) and Koellner & Scholz (2008) have also suggested detailed methodologies to account for biodiversity in life cycle assessment that can be used to distinguish between organic and conventional management, but those methods were specifically developed to the Swiss context.

As for biodiversity, no consensus exists on appropriate and simple soil quality indicators to be used on individual farms or fields (Halberg et al., 2005). Milà i Canals et al. (2007b) have proposed soil organic matter an indicator of soil quality as a life support function. However, soil organic matter is only one aspect of soil quality, where e.g. soil compaction or the amount of toxic substances such as ‘kg copper per hectare’ are not included. Aspects concerning soil organic matter and the global warming potential are discussed further in section 3.3.2.

Socio-economic aspects
Furthermore, socio-economic aspects are rarely included in LCA implying that for example aspects such as animal welfare, working conditions and landscape aesthetics are normally not included. The social impact methodologies for LCA are still in their infancy (Reap et al., 2008a). The difficulties in including social and economic aspects in LCA are listed by Reap et al. (2008a). This includes difficulties in obtaining consensus on more than 200 existing societal impact indicators, the less straightforward relation to the product and the challenges on integrating the social qualitatively approaches with LCA (Reap et al., 2008a).

3.3.2 Carbon sequestration

The category of global warming potential is well defined and central in life cycle assessments. However, not all affected carbon is yet fully included in the calculations. The methods of including the changes in the organic carbon stocks in vegetation, soils and litter caused by agricultural production are still in its infancy of being discussed and developed. Organic carbon changes have especially been discussed with regard to direct and indirect land use changes such as deforestation in relation to beef and soybeans from Brazil (Cederberg et al., 2009) and bioenergy crops (e.g. Searchinger et al., 2008), whereas soil carbon changes have gained attention in relation to comparisons of organic versus conventional production (e.g. Hörtenhuber et al, 2010) and with regard to the use of straw for bioenergy (e.g. Levasseur et al (2010). Agreement on simple and robust estimation method and the time horizon when including those changes in the LCA are crucial, as discussed further in Paper IV.

Soil carbon sequestration
As mentioned in section 3.1, soils under organic management will often increase the level of soil organic carbon compared to conventional management. The soil organic carbon changes for the case studies in Paper II and III was roughly estimated using the IPCC tier 1 methodology (IPCC, 2006) for the sensitivity analysis. A more accurate methodological approach for estimating and including soil carbon changes in life cycle assessments have been suggested in Paper IV. In this paper, the challenges of estimating and including soil carbon changes are also discussed, with regard to estimating the actual soil carbon change depending on the time perspective and development towards a new steady state level of soil organic carbon.

Direct and indirect land use change
At the moment there is an ongoing debate on direct and indirect land use change – which is especially relevant when talking about bioenergy, soybeans and beef from Brazil and palm oil from Malaysia/Indonesia, which have a direct linkage to deforestation of rainforest, where new agricultural land is needed for the production. This is termed direct land use change. The indirect land use change occurs if the demand for previous land use moves to other places and indirectly cause e.g. deforestation. It could be argued that it is always relevant when using land for a purpose to include an indirect land use change. However, many uncertainties and assumptions are involved in this argument and the methodology concerning land use change is still in its infancy. Therefore, aspects concerning land use change have not

been included in the present PhD study. With regard to the products concerned, no direct land use changes have occurred in the case studies.

3.3.3 Environmental cost of manure and other challenging aspects

One of the main differences between organic and conventional management practices is the absence of pesticides and mineral fertilizer. This makes organic farmers much more dependent on the crop rotation to prevent pests and diseases and provide especially nitrogen for the crops. Furthermore, organic farmers are more dependent than conventional on livestock production systems, to provide animal manure as a nitrogen source. Generally, the main nitrogen inputs to organic systems are derived from either green manure crops in the crop rotation or animal manure. The organic production systems including green manure crops imply challenges of how to allocate the environmental burden from the green manure in the crop rotation to the other crops. However, in the present PhD thesis the organic production systems in the case studies are mainly based on nitrogen from animal manure. This implies challenges of how to estimate the environmental cost of the animal manure.

Estimating the environmental cost of manure

In a comparative LCA of organic and conventional products, the question whether animal manure has an environmental cost or not affects the results. This issue has been further described and discussed in Paper II. In short three approaches have been suggested. Dalgaard & Halberg (2007) has suggested a consequential way of dealing with the manure issue, as used in Paper II and III. Secondly, van Zeijts et al. (1999) have also suggested a method of handling manure in an LCA, where the emissions caused by storage, transport and application of animal manure are allocated according to the economic value of the manure. A third approach would be including the environmental cost of producing the N in a green manure crop (Audsley et al., 1997; Jungbluth & Frischknecht (2007).

Overall, the approach depends on the context of whether the animal manure is derived from organic or conventional livestock or whether animal manure is regarded as a waste in society that would otherwise be dumped in the rivers and create environmental problems or whether animal manure is a precious source of fertilizer that prevents mineral fertilizer to be produced. When conventional animal manure is used in organic systems, the approach suggested by Dalgaard & Halberg (2007) of using the environmental production costs of mineral fertilizer as a ‘rate of exchange’ might be appropriate. However, when the organic systems are not importing manure from conventional livestock and thus not related to the production of mineral fertilizer at all, this approach might be misleading.

Pesticides

Other challenging aspects with regard to LCA of agricultural crops are pesticides, which is especially visible in a comparison of organic and conventional products. Some LCA studies includes the category of ecotoxicity based on pesticide application, while others excludes ecotoxicity as impact category, based on the argument that the uncertainty of especially the fate of the pesticides after agricultural use is too high and is highly dependent on the specific pesticide. Finnveden (2000) stated that even if development of more complete databases concerning chemicals (such as pesticides) may fill some of the data gaps, the impact categories human toxicity and ecotoxicity are not expected to be greatly improved due to the large number of chemicals used by society and the potential synergistic effects between them. Since 2000, improvements concerning pesticides in LCA have been made, but van Zelm et al. (2009) still states that other LCA studies shows a relatively large uncertainty range for fresh water ecotoxicity and that measures to reduce this uncertainty need to be taken before fresh water ecotoxicity are used as an impact category in decision support. Thus, in the present PhD thesis, ecotoxicity is not included as an impact category, but pesticides use is presented.

4 CONCLUSION

The increasing global trade with organic products holds a potential to offer economic and environmental benefits to developing countries. On the other hand, there is a risk of increasing the environmental burden due to long-distance transport. Furthermore, due to the challenges for small-scale farms to enter the global organic markets, there is a risk of pushing the organic food and farming systems towards simpler farming systems and thereby diminishing the environmental benefits of organic farming.

The case studies in China and Brazil show a total greenhouse gas emission of 429 kg CO_2 eq. per tonne organic soybean imported to Denmark and 424 g CO_2 eq. per litre organic orange juice. Transport accounts for 50-60% of the total greenhouse gas emissions from the imported plant products, which is in agreement with other studies of imported agricultural products to regional distribution centres in the UK. As a comparison, the proportion is much lower for imported meat products (1-15%), since the greenhouse gas emissions per unit meat is a lot higher than for plant products. However, the actual contribution is the same. The transport mode is important. When importing products to Denmark, sea transport from South America (reloaded to trucks in Rotterdam) is comparable to truck transport from Italy and France and lower than truck transport from Spain and sea transport from South Africa and China (also reloaded in Rotterdam) in terms of greenhouse gas emissions per kg imported product. This underlines that food miles as a single indicator of sustainability is inadequate.

Comparing diversity and land use of organic and conventional production in the case studies, land use is 10-13% higher per kg organic product than conventional. However, crop diversity is higher on small organic compared to small conventional orange farms in Brazil, which may have a positive effect on biodiversity along with the absence of pesticides and the interrow vegetation. No major differences are found in biodiversity potential in the Chinese case study except the absence of pesticides.

Within the organic orange production in Brazil, greenhouse gas emissions, eutrophication potential and copper use per hectare is found to be significantly lower on organic small-scale than on large-scale organic orange plantations. The Brazilian case study indicates that there is a need for constant and dynamic evaluation of how to secure environmental performance through organic regulation.

Comparing greenhouse gas emissions and eutrophication of organic and conventional, the greenhouse gas emissions per kg organic products at farm gate in the case studies in China and Brazil are 60-75% of the comparable conventional production, while eutrophication is 38-82%. This is in agreement with other studies showing either comparable or slightly lower level of greenhouse gas emissions and eutrophication per kg organic product compared to conventional. The studies in Brazil and China find that including estimated soil carbon changes widened the difference in greenhouse gas emissions per kg product between organic and conventional, but there is a need for methodological development on how to estimate and include this.

The difficulties in including the effects on soil carbon changes and biodiversity are some of the shortcomings of life cycle assessments (LCA) as a tool to evaluate environmental soundness of agricultural products. A methodological approach to include soil carbon changes in LCA is suggested.

5 OUTLOOK

The present PhD work was initiated four years ago, but the import of organic products to Denmark and the global organic markets have only increased since then – making studies of the consequences of this development more relevant than ever.

The debate on the environmental implications of the growing global trade with organic products is relevant to both consumers deciding which products to buy in the supermarket, to organic animal producers or companies deciding which feed items to buy for the organic animal production, to the organic sector deciding in which direction the future development of organic food and farming systems should be heading and to major supermarkets deciding whether to introduce carbon labelling or not. This PhD thesis provides a contribution to the debate on environmental issues, but a wider perspective on sustainability should be considered when looking at the implications of an increased global trade with organic products.

In the following outlook, perspectives on the results from the present PhD thesis from each stakeholder’s point of view are briefly discussed.

For consumers in the North buying organic products, one of the messages from the present thesis is that transport of plant products from other continents or Southern Europe contributes approximately half the carbon footprint of the plant product. However, two main issues should be noted; firstly the domestic production of a comparable product might not be lower if extra energy has been used to facilitate the production or for storage out of season (e.g. heated glasshouses or cool storage). Here a life cycle based assessment gives a more qualified answer than a single indicator of food miles. Secondly, the organic production might contribute to improved environmental and economic development in the country of origin.

The imported products can be divided into two main categories. One category is products that need to be imported if we want to eat them, such as chocolate, coffee, tea, banana, several spices, oranges etc. The main consumer choice is here between an imported conventional or imported organic product and one of the main advantages of the organic product is the avoidance of pesticides (of which many of them is banned in Denmark) for the benefit of both consumers avoiding pesticide residues and farmers in the South avoiding to apply the pesticides. Another category is products that can be produced domestically, but they are imported when it is out of season (such as cucumber and salad from Spain) or they are imported due to competition from other countries due to higher production costs or tighter regulation in Denmark (e.g. no use of copper in Denmark) as for imported apples. The imported organic products are mainly fruits and vegetables, which is beneficial for the vitamin status of the population in the winter period. However, the degree of luxury consumption can be discussed within both categories and whether alternative products produced domestically in the season such as cabbage could replace e.g. the salad imported from Spain.

While transport of plant products contributes approximately half of the total greenhouse gas emission of the product, transport of meat products contributes only a relatively small part of total greenhouse gas emissions from meat products, even though the actual contribution is the same as for plant products. However, aiming at reducing the environmental impact of the food and drinks we consume, it might not always be a question of searching for the large contributions, but also searching for emissions which are possible to reduce. Many small contributions can also have a large effect. Nevertheless, it is obvious that reducing the meat consumption will reduce the greenhouse gas emissions related to food consumption considerably. Furthermore, it is important to reduce the travelling connected to shopping by car and reduce food waste.

For *organic animal producers* importing feed from afar, the message from this research is that transport has a considerable contribution to the carbon footprint for plants products and it does not matter much whether the feed are imported by truck from Italy or by ship from China with regard to greenhouse gas emissions. Organic protein fodder might be cheaper to buy from other countries, since production costs are high in Denmark (mainly due to high wages and land prices) and there are still no restrictions with regard to carbon footprint of feed products. However, a life cycle based assessment is needed to evaluate whether a domestically produced alternative has a lower environmental impact and the aggregated consequences of feeding with local versus imported feeds.

For the *retail sector*, carbon labelling of products has been suggested and implemented to a certain degree by major supermarkets such as Tesco and Carrefour. Carbon footprint accounting can be used both directly as information for the consumers or in business to business communication among e.g. wholesalers and retailers. Whether carbon labelling has a future depends very much of the interest of the consumers, wholesalers and retailers, since carbon labelling of products is also very data demanding and thus expensive. However, major supermarkets might be important players in addition to national and international politics with regard to sustainability. With regard to labelling of products, organic agriculture has a special interest in communicating the product properties to the consumers and LCA as a tool might have a role to play. However, a single focus on carbon labelling implies the risk of sub optimising in relation to sustainability when the result of only one environmental impact category is presented.

For the *organic sector*, one of the messages from the present thesis is that differences in environmental impacts were not only found between organic and conventional products, but the current research also indicated differences between organic systems producing the same product. This leads to the question of how well the organic regulation reflects the organic principles especially with regard to crop diversity and biodiversity and whether the organic production should be better documented in the future. Presently, the organic sector is mainly regulated by regulating the inputs and farming practices. However, an alternative would be regulation by an indicator based system using e.g. a life cycle approach, where the outcome of the system could be checked. The variation in environmental impacts between organic systems can be used for benchmarking and improvement of the organic production. However, further research is needed to verify this, to identify differences between organic systems and suggest improvements of the environmental performance in accordance with the organic principles.

This thesis only deals with one side of sustainability, namely environmental issues. However, other aspects of sustainability such as socio-economic issues are equally important when assessing the sustainability of the increasing global trade with organic products and sometime a trade-off is necessary. A recent example of this is the debate on air-freighted organic food products from mainly Africa to the UK, which has given rise to a heated debate on the trade off in sustainability between development aid for African countries versus increased greenhouse gas emissions from air-freight. Finally, the English Soil association decided that air freighted products were acceptable if they were fair trade labelled and thus contributed to a beneficial development for the countries or regions concerned. This illustrates the dilemma of striving towards solving two main problems: reducing poverty and reducing greenhouse gas emissions. If greenhouse gas emissions attract too much attention, other important environmental or socio-economic impacts might be overlooked or neglected.

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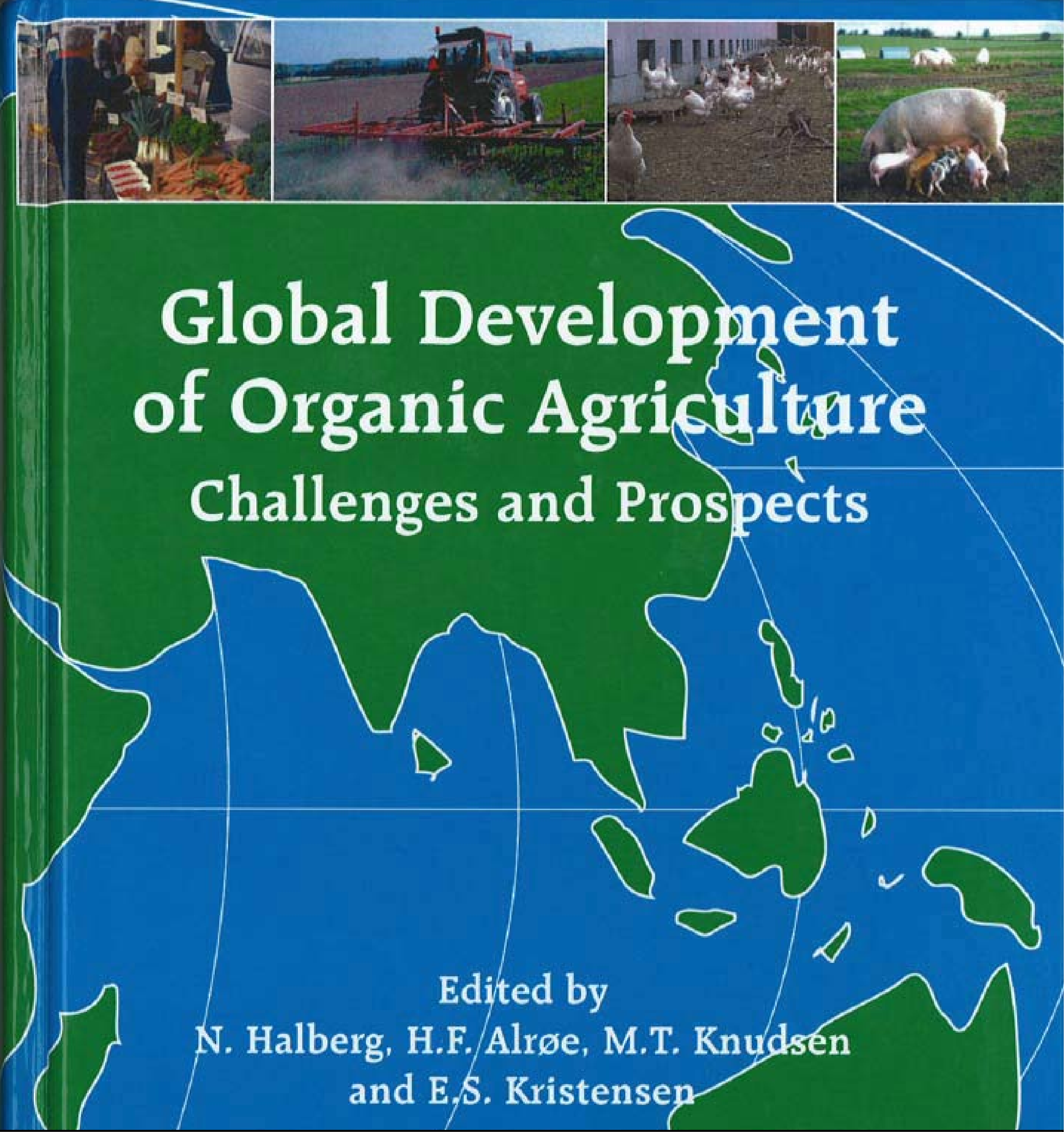
7 PAPERS



Paper I

Global trends in agriculture and food systems

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CABI Publishing Walllingford, UK, pp. 1-48

1
Global trends in agriculture and food systems

Marie Trydeman Knudsen, Niels Halberg, Jørgen E. Olesen, John Byrne, Venkatesh Iyer and Noah Toly*

Summary

Increasing globalization affects agricultural production and trade and has consequences for the sustainability of both conventional and organic agriculture.

During the last decades, agricultural production and yields have been increasing along with global fertilizer and pesticide consumption. This development has been especially pronounced in the industrialized countries and some developing countries such as China, where cereal yields have increased a remarkable twofold and 4.5-fold respectively since 1961. In those countries, food security has increased, a greater variety of food has been offered and diets have changed towards a greater share of meat and dairy products. However, this development has led to a growing disparity among agricultural systems and population, where especially developing countries in Africa have seen very few improvements in food security and production. The vast majority of rural households in developing countries lack the ecological resources or financial means to shift into intensive modern agricultural practices as well as being integrated into the global markets. At the same time, agricultural development

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* Corresponding author: Danish Research Centre for Organic Food and Farming (DARCOF), P.O. Box 50, Blichers Allé 20, DK-8830 Tjele, Denmark. E-mail: MarieT.Knudsen@agrsci.dk

has contributed to environmental problems such as global warming, reductions in biodiversity and soil degradation. Furthermore, pollution of surface and groundwater with nitrates and pesticides remains a problem of most industrialized countries and will presumably become a growing problem of developing countries. Nitrate pollution is now serious in parts of China and India. The growing global trade with agricultural products and the access to pesticides and fertilizers have changed agricultural systems. Easier transportation and communication has enabled farms to buy their inputs and sell their products further away and in larger quantities and given rise to regions with specialized livestock production and virtual monocultures of e.g. Roundup Ready soybeans in Argentina. Since 1996, the Argentinean area devoted to soybeans has increased remarkably from 6 to 14 million ha, covering approximately 50% of the land devoted to major crops in 2003. Since 1997, Brazilian Amazon has seen a deforestation of more than 17,000 km^2 each year with medium or large-scale cattle rangers presumably being the key driving force.

Organic farming offers a potentially more sustainable production but has likewise been affected by globalization. Organic farming is practiced in approximately 100 countries of the world and the area is increasing. European countries have the highest percentage of land under organic management, but vast areas under organic management exist in e.g. Australia and Argentina. Europe and North America represents the major markets for certified organic products, accounting for roughly 97% of global revenues. The international trade with organic products has two major strands: i) trade between European and other Western countries (USA, Australia, New Zealand); and ii) South–North trade, involving production sites, most importantly in Latin America, which ship to major Northern organic markets. The recent development holds the risk of pushing organic farming towards the conventional farming model, with specialization and enlargement of farms, increasing capital intensification and marketing becoming export-oriented rather than local. Furthermore, as the organic products are being processed and packaged to a higher degree and transported long-distance, the environmental effects need to be addressed. Organic farming might offer good prospects for marginalized smallholders to improve their production without relying on external capital and inputs, either in the form of uncertified production for local consumption or certified export to Northern markets. However, in order to create a sustainable trade with organic products focus should be given to issues like trade and economics (Chapters 4 and 5), certification obstacles, and ecological justice and fair trade (Chapters 2 and 3). Furthermore, the implications of certified and non-certified organic farming in developing countries need to be addressed (Chapters 6 and 9) including issues on soil fertility (Chapter 8) and nutrient cycles (Chapter 7) and the contribution to food security (Chapter 10).

Introduction

Increasing globalization has been one of the major trends in the latest decades, as a consequence of the dominating technological and social development. Globalization is here understood as ‘the erosion of the barriers of time and space that constrain human activity across the earth and the increasing social awareness of these changes’ (Byrne and Glover, 2002). The increasing globalization has consequences for the way that we produce and trade agricultural products and thereby also environmental consequences for the climate, biodiversity, and land resources among other things. Globalization has implications for conventional agriculture but contains also specific opportunities and problems for organic farming – related to e.g. trade with organic certified products from developing countries. The idea of ‘Sustainable development’ has been another key concept in the latest decades and can be seen as reaction to the dominating development. Sustainability is a concept that can have different meanings (Jacobs, 1995; Rigby and Cáceres, 2001). The definitions of sustainability include both the interpretation related to ‘functional integrity’, where man is seen as an integrated part of nature (Thompson, 1996) and the ‘resource sufficiency’, which addresses the rate of resource consumption linked to production. In the following, recent trends in agriculture in relation to globalization and sustainability will be presented. Focus will be given to issues that are relevant for the discussion of the role and conditions for further development of organic farming in a global context.

The overall aim with this chapter is to:

- Show global trends in agriculture and food systems related to globalization and their environmental and socio-economic impacts.
- Show global trends in organic farming related to globalization – and indicate potentials and challenges in global organic agriculture related to environmental and socio-economic issues.

World agriculture – trends and impacts

Agriculture and food systems have changed very much over the last 50 years. Agricultural development has seen a rapid advance of agricultural technology in industrialized countries with the green revolution in the 1960s being counteracted by an increasing public awareness of environmental protection and sustainable development that evolved in the 1980s (FAO, 2000). In the 1990s an increasing globalization occurred that has continued into the 21st century. The current wave of globalization was made possible by technological breakthroughs in transportation and communication technologies (notably the Internet, mobile telephone technology and just-in-time systems) and affordable fuel in tandem with various efforts to liberalize international trade and investment flows (FAO, 2003). Increases in long-distance food trade, global concentration in food processing and retail industries and diet change are signs of the globalization of the food system (von Braun, 2003).

In the following, major trends in agricultural production and food systems in relation to globalization will be shown along with environmental and socio-economic impacts. The conceptual model in Figure 1.1 shows the structure in this section and illustrates possible connections in the development of the global agricultural and food systems. The figure is not intended to cover all aspects on global food systems sustainability, but to illustrate possible problematic situations.

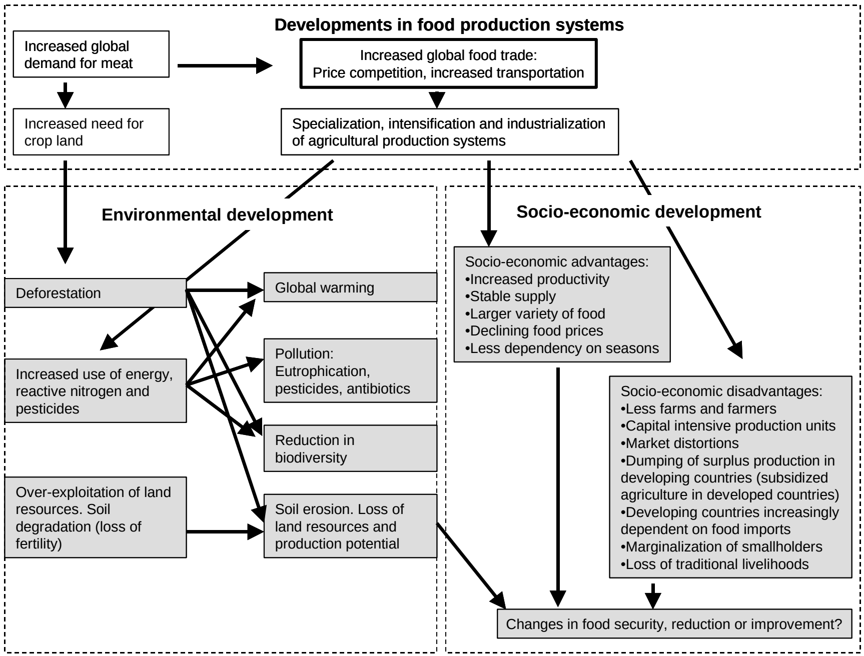

Figure 1.1. Illustration of possible problematic aspects in global food systems sustainability, environmentally and socio-economically. The arrows indicate possible effects.

Trends in agricultural production

Agricultural production has increased greatly over the last decades and in most continents the food production has been able to surpass the population growth. According to FAO (2000), the increase in production is attributable to the following factors, among others:

- the spread in developed countries of the modern agricultural revolution (involving large-scale mechanization, biological selection, use of chemicals, specialization);
- a modern agricultural revolution in some developing countries that is not dependent on heavy motorized mechanization but instead involves the use of chemicals and the selection of varieties;
- the expansion of irrigated surfaces, from about 80 million ha in 1950 to about 270 million ha in 2000;
- the expansion of arable land and land under permanent crops, from some 1330 million ha in 1950 to 1500 million ha in 2000,
- the development of mixed farming systems using high levels of available biomass (combining crop, arboriculture, livestock and, sometimes, fish farming) in the most densely populated areas that lack new land for clearing or irrigation.

The average yields of a milking cow and crop yields per ha and per worker have been increasing over the last 50 years (FAO, 2000). In the past four decades, increasing yields accounted for about 70% of the increase in crop production, compared to expanding the land area or increasing the cropping frequency (often through irrigation). However, yield increases have been most profound in industrial countries and e.g. China, whereas the yield increases in e.g. developing countries in Africa have been very limited (Figure 1.2).

The considerable advances in agriculture cannot hide the fact that most of the world’s farmers use inefficient manual tools and their plants and domestic animals have benefited very little from selection. The progress in agricultural production hides a growing disparity among agricultural systems and populations. The gap between the most productive and least productive farming systems has increased 20 fold in the last 50 years (FAO, 2000). The agricultural revolution with all its attributes and especially its motorized mechanization has not extended far beyond the developed countries, with the exception of small portions of Latin America, North Africa, South Africa and Asia, where it has only been adopted by large national or foreign farms that have the necessary capital (FAO, 2000).

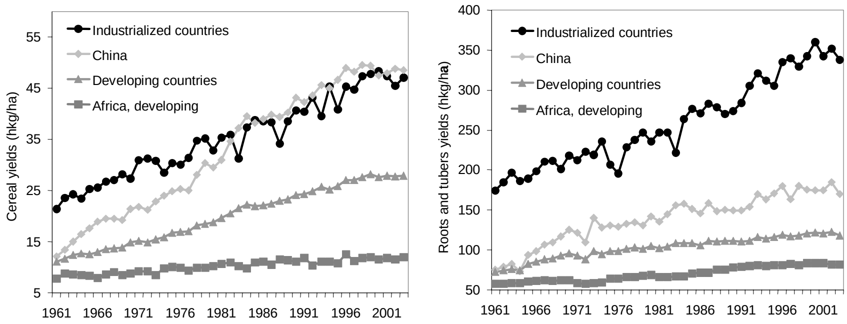

Figure 1.2. Yields of cereals plus roots and tubers in industrialized and developing countries plus in China and Africa, developing from 1961-2003 (hkg/ha) (FAOSTAT data, 2005).

Agricultural intensification

The agricultural revolution and globalization have had an enormous impact on agriculture and food systems in the developed countries. Developments in industry, biotechnology, transport and communication have affected agriculture in different ways.

Industrial developments have provided the means for motorization and large-scale mechanization, mineral fertilization, treatment of pests and diseases (pesticides, veterinary drugs etc.) and the conservation and processing of vegetable and animal products in developed countries. Developments in biotechnology supplied through selection, high yielding plant varieties and animal breeds have been adapted to the new means of production (FAO, 2000). The latest biotechnological developments are the genetically modified crops, grown primarily in USA, Canada and Argentina (see case from Argentina in Box 1.1).

The increase in fertilizer use and the use of improved varieties, through selection, have been among the important factors for the increased food production. A third of the increase in world cereal production in the 1970s and 1980s has been attributed to increased fertilizer use (FAO, 2003). World fertilizer consumption grew rapidly in the 1960s, 1970s and 1980s (Figure 1.3). The fertilizer usages in Europe have slowed down since the 1980s mainly due to reduced government support for agriculture and increased concern over the environmental impact. Fertilizer use in Asia, especially China, has been increasing (FAO, 2003; Figure 1.3), but the level of fertilizer use varies enormously between regions. North America, Western Europe and South–East Asia accounted for four-fifths of world fertilizer use in 1997–99 (FAO, 2003).

The highest rates are applied in East Asia, especially in China, followed by the industrial countries. At the other end of the scale, farmers in sub-Saharan Africa apply much less (FAO, 2003; Figure 1.3). The average fertilizer consumption is predicted to increase in developing countries (FAO, 2003). However, the average figure masks that for many (especially small) farmers the purchase of manufactured fertilizers and pesticides is and will continue to be constrained by their high costs relative to output prices and risks or simply by unavailability (FAO, 2003).

The global usages of pesticides have increased considerably during the second part of the 20th century (Figure 1.4). Some of the problems with diseases and insects have increased with the increased use of nitrogen fertilizers due to a higher susceptibility of the crop to attack at higher nitrogen input (Olesen *et al.*, 2003). Some countries in Western Europe have seen a reduction in pesticide consumption in recent years, primarily due to policies that promote or enforce management strategies with reduced pesticide use (Stoate *et al.*, 2001). Future pesticide consumption is likely to grow more rapidly in developing countries than in developed ones (FAO, 2003). The treatment of pests and diseases, in both plants and livestock, has become more important to safeguard investments in farm output.

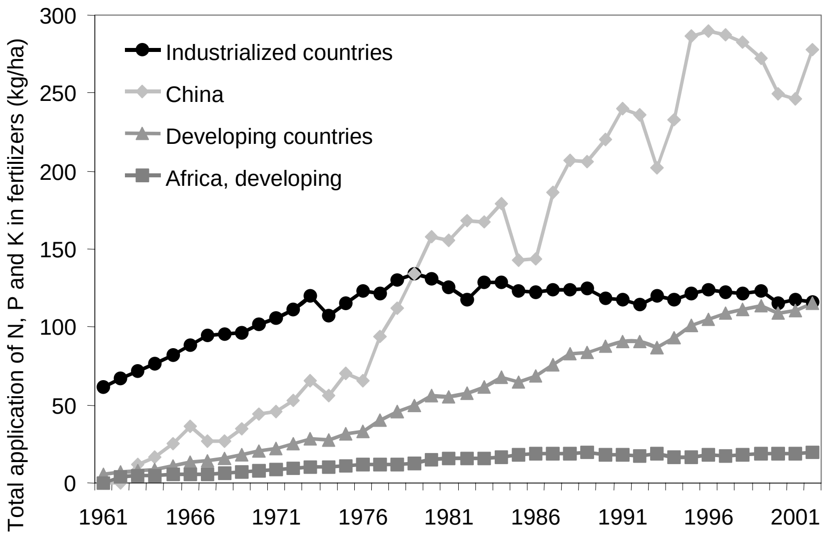

Figure 1.3. Total fertilizer application of N, P and K in industrialized and developing countries plus China and Africa, developing from 1961 to 2002 (kg/ha) (FAOSTAT data, 2005).

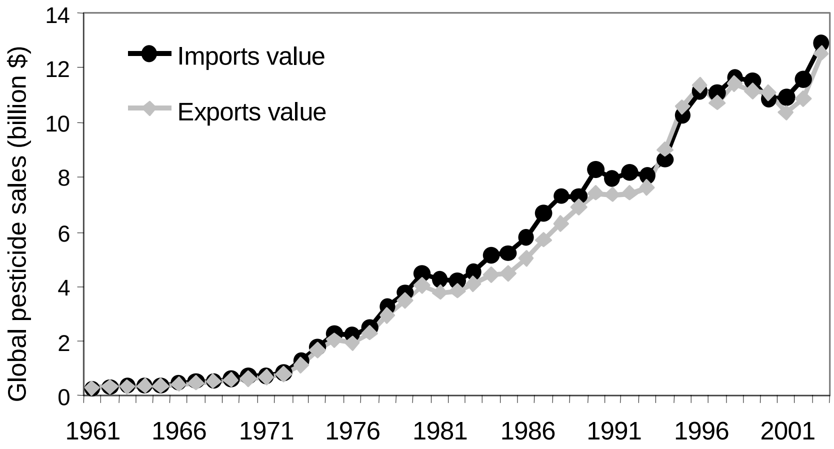

Figure 1.4. Imports and exports value of global pesticide sales from 1961 to 2003 (FAOSTAT data, 2005).

The more expensively bred and fed animal and the larger and more concentrated the animal production, the higher the risks. A great part of the antibiotics produced today are used as treatments against infectious diseases or as growth promoters in animal production, especially for pigs and poultry. Mellon *et al.* (2001) estimated that 70% of all antibiotics used in the USA are used for non-therapeutic livestock use. JETACAR (1999) found that approximately one-third of the antibiotics imported to Australia is for humans and two-thirds for animals. Denmark became the first country with a significant livestock industry to curtail the use of antibiotic growth-promoters in pig and poultry production in 1998. Approximately 70% of the antimicrobials used in Denmark are for therapeutic veterinary use (Heuer and Larsen, 2003).

Agricultural systems have changed with the introduction of mineral fertilizer, pesticides etc. With the use of mineral fertilizer cash crop production no longer relies on soil fertility building or use of manure. Furthermore with the introduction of mechanization, agriculture has also been freed from the need to produce forage for draught animals. Consequently, agricultural holdings suited for mechanized crop production have been able to abandon fodder and livestock production and specialize in cash crop production while other agricultural holdings have specialized in livestock production, often without sufficient land for manure application (FAO, 2000). Furthermore, the use of agricultural chemicals and GMO crops has partly released agricultural holdings from former crop rotation systems used to control weeds, insects and diseases. As a result cropping systems have been simplified and further specialized, culminating in monocropping or quasi-monocropping.

There has been a trend towards a narrower genetic base used for plant and animal production. Of the 270,000 known species of higher plants only three species (wheat, rice and maize) provide half of the world's plant derived energy intake (FAO, 1997; Cromwell *et al.*, 1999). At a national and regional level, only a few varieties are used over large-scale areas and the same trend can be seen in livestock genetic resources (CBD, 2001). The latest development in this aspect is the rapid spread of GMO crops, where a few (pesticide resistant) varieties of e.g. maize and soybean now cover large areas of land (see case from Argentina in Box 1.1).

Developments in transportation and communication have opened up the farms and agricultural regions and enabled them to procure their fertilizer, feed and other inputs from further away and in larger quantities. It also allowed for the sale of their products in increased amounts and wider areas. An increased globalization has freed agricultural holdings even more from comprehensive localized self-supply and made them able to focus on the most profitable product (or simplified combination of products). Virtual monocultures of soybean, maize, wheat, cotton, vineyards, vegetables, fruit and flowers and specialized productions of pig and poultry have thus spread over entire regions giving rise to new specialized regional agricultural systems (FAO, 2000).

Dietary changes

Just as world average calorie intakes have increased, so have also people's diets changed. Patterns of food consumption are becoming more similar throughout the world, incorporating higher-quality and more expensive foods such as meat and dairy products.

This diet change is partly due to simple preferences by populations. Partly, too, it is due to increased international trade in foods, to the global spread of fast food chains, and to exposure to North American and European dietary habits. Convenience also plays a part, for example the portability and ease of preparation of ready-made bread or pizza, versus root vegetables. Changes in diet closely follow rises in incomes and occur almost irrespective of geography, history, culture or religion (FAO, 2003).

These changes in diet have had an impact on the global demand for agricultural products and will continue to do so. Meat consumption in developing countries, for example, has risen from only 10 kg per person per year in 1964–66 to 26 in 1997–99. It is projected to rise still further, to 37 kg per person per year in 2030. Milk and dairy products have also seen a rapid growth, from 28 kg per person per year in 1964–66 to 45 kg in 1997–99, and with the expected consumption of 66 kg by 2030 in developing countries. The intake of calories derived from sugar and vegetable oils is furthermore expected to increase. However, average human consumption of cereals, pulses, roots and tubers is expected to level off (FAO, 2003).

Environmental impacts

Human activities and in particular the provision of foods for the growing world population put increasing demands on the natural resources of the earth. These effects are seen in several ways (Figure 1.1). In some areas of the world, agricultural land use increases at the expense of forests and other natural terrestrial ecosystems. In other parts of the world there is an overexploitation of the land resources leading to soil degradation and loss of soil fertility. However, the major way used to satisfy the need for food is through intensification of the agricultural production, primarily through the use of fertilizers and pesticides (see 'Trends in agricultural production'). All of these pathways have their own effects on the environment.

Four major indicators of environmental sustainability (EEA, 2005) are considered here as illustrated in Figure 1.1:

- Loss of land resources by soil erosion and soil degradation. Eroded soils are often lost for productive agricultural use for a very long time, whereas soils that are degraded through loss of soil organic matter, soil compaction, nutrient mining or salinization may be restored through proper agricultural management techniques. Loss of land resources has secondary negative effects on biodiversity and global warming.
- Loss of biodiversity involves a reduction in the number of living species on the earth and thus a loss of genetic resources (CBD, 2001) and a loss of ecosystem services in both natural and managed ecosystems (Costanza *et al.*, 1997). Both effects have negative long-term consequences for the interaction between the human population and the environment. Biodiversity is reduced by a number of agricultural activities, such as deforestation, reduction of field margins and hedgerows, drainage of wetlands, genetic uniformity in crop land, pesticides etc. (FAO, 2003).
- Global warming is a consequence of increasing emissions of greenhouse gases (primarily CO_2, CH_4, N_2O and CFCs) to the atmosphere. The global emissions of CO_2 in 1996 (23,900 million t) were nearly four times the 1950 total (UNEP, 1999). The use of fossil fuels is the primary cause of these emissions. However, agricultural production contributes about 39% of the methane and 60% of the nitrous oxide emissions released in OECD countries (OECD, 2000 cf. OECD, 2001b). Methane emissions from agriculture are mainly produced from ruminant animals and the handling of manure, while the main source of nitrous oxide emissions is nitrogen fertilizers (OECD, 2001b). In addition, CO_2 from deforestation, soil degradation and soil erosion also have major contributions to the global greenhouse gas emissions. Furthermore, the use of fertilizer is associated with high energy requirements for their production resulting in CO_2 emissions (Dalgaard *et al.*, 2000).

- The use of fertilizer in high amounts per ha and the large amounts of manure concentrated in specific geographical areas has increased the emission of ammonia and nitrate, which creates eutrophication and acidification in sensitive aquatic and terrestrial environments and pollution of ground and surface water (EEA, 2003; see more below). With increasing load of phosphorus in agricultural soils in particular with intensive livestock farming, there is also a risk of phosphorus losses to sensitive aquatic environments (Novotny, 2005).

Most of the environmental problems have increased considerably in recent decades. These problems are usually externalized, being greater for the society as a whole than for the farms on which they operate, and direct incentives for the farmers to correct them are therefore largely lacking (Stoate *et al.*, 2001). Impacts on biodiversity and global warming are trans-boundary or global in their nature, and efforts to deal with these therefore require international collaboration.

In the following, the effects on the above-mentioned environmental indicators caused by 1) agricultural land use and by agricultural intensification through 2) the global nitrogen cycle and 3) pesticides will be discussed.

Agricultural land use

The world's land area comprises 130.7 million km^2. However, less than half of this land area is suitable for agriculture, including grazing (Kindall and Pimentel, 1994). Nearly all of the world's productive land is already exploited. Thus, only a small increase in agricultural area has been seen over the past 40 years. Most of the unexploited land is too steep, too wet, too dry or too cold for agriculture. For arable crops, soils also limit land use, because many soils are unsuitable for tillage or depleted in nutrients.

Expansion of the cropland has to come at the expense of forest and grassland, which also have essential uses. The net gain in agricultural area comes from adding land through deforestation and loss of land from land degradation and reforestation. It has been estimated that 70–80% of deforestation is associated with agricultural uses (Kindall and Pimentel, 1994). There are several environmental problems associated with deforestation, of which loss of biodiversity and CO_2 emissions are the major ones. It has thus been estimated that CO_2 emissions from land use changes amount to 20% of the emissions associated with fossil energy use (Houghton *et al.*, 2001).

Degradation of existing agricultural land involves loss of productive land. According to some analysts, land degradation is a major threat to food security and it is getting worse (Pimentel *et al.*, 1995; UNEP, 1999; Bremen *et al.*, 2001). Others believe that the seriousness of the situation has been overestimated at the global and local level (Crosson, 1997; Scherr, 1999; Lindert, 2000; Mazzucato and Niemeijer, 2001). Brown (1984) estimated that about 10 million ha of agricultural land was lost by soil erosion every year, corresponding to 0.7% of global cropland area. Others argue that the area of cropland going out of use because of degradation is in the order of 5–6 million ha every year (UNEP, 1997). It is estimated that soil degradation is severely affecting 15% of the earth’s cropland area, and in Europe alone 16% of the soils are prone to soil degradation (Holland, 2004). UNEP (1999) estimated that 500 million ha of land in Africa have been affected by soil degradation since about 1950, including as much as 65% of agricultural land.

The degradation and loss of agricultural land arises mainly from soil erosion, salinization, waterlogging, and urbanization. In addition nutrient depletion, overcultivation, overgrazing and soil compaction contributes to the deterioration of soil fertility. Many of these processes are caused by agricultural management practices. Soil erosion is considered the single most serious cause of arable land degradation, and the major cause is poor agricultural practices that leave the soil without vegetative cover or mulch to protect it against water and wind erosion. In developing countries, the degradation is worsened by low inputs, partly due to lack of credits and partly because available crop residues and dung are used for fuel. This reduces soil nutrients and intensifies soil erosion.

The global nitrogen cycle

Nitrogen is one of the most abundant chemical elements in the atmosphere and biosphere. However, more than 99% of the nitrogen is present as molecular nitrogen, which is not available to most organisms. Only a small proportion of the nitrogen is thus present as reactive nitrogen, which includes inorganic forms (NH_3, N_2O, NO, NO_2 and NO_3) and organic compounds (urea, amines, proteins and nucleic acids).

In the pre-industrial world, creation of reactive nitrogen occurred primarily from lightning and biological nitrogen fixation, and the denitrification process balanced the input of reactive nitrogen. However, in the industrialized world reactive nitrogen is accumulating in the environment at all spatial scales (Galloway *et al.*, 2003). During the past few decades, reactive nitrogen has been accumulating in the environment (Figure 1.5), primarily due to the industrialized production of fertilizer nitrogen by the Haber-Bosch process, which converts non-reactive N_2 to reactive NH_3.

The remarkable change in the global N cycle caused by the higher inputs of reactive N has had both positive and negative consequences for people and ecosystems. A large proportion of the global population is sustained because reactive nitrogen is provided as fertilizer nitrogen or by cultivation introduced biological nitrogen fixation (Smil, 2002). However, nitrogen is accumulating in the environment, because the rate of input is much larger than the removal by denitrification, and this accumulation is projected to continue to increase as human population increases and per capita resource use increases. The accumulation of reactive nitrogen in the environment contributes to a number of local and global environmental problems (Galloway *et al.*, 2003):

- Increases in reactive nitrogen in the atmosphere leads to production of tropospheric ozone and aerosols that induce respiratory disease, cancer and cardiac disease in humans (Wolfe and Patz, 2002).
- Increases in nitrate contents of groundwater, which have potential health effects (Jenkinson, 2001).
- Productivity of terrestrial systems (e.g. grasslands and forests) is affected with loss of biodiversity in oligotrophic ecosystems.
- Reactive nitrogen contributes to acidification and biodiversity loss in lakes and streams in many parts of the world (Vitousek *et al.*, 1997). There are several examples of streams and lakes, where recent reductions in fertilizer inputs have led to reduced N concentrations (Iital *et al.*, 2005).
- Reactive nitrogen is responsible for eutrophication, hypoxia, biodiversity loss and habitat degradation in coastal ecosystems (Howarth *et al.*, 2000). This environmental problem appears to be increasing globally (Burkart and James, 1999; EEA, 2003).
- Reactive nitrogen contributes to global climate change and stratospheric ozone depletion, both of which have impacts on human and ecosystem health (Mosier, 2002).

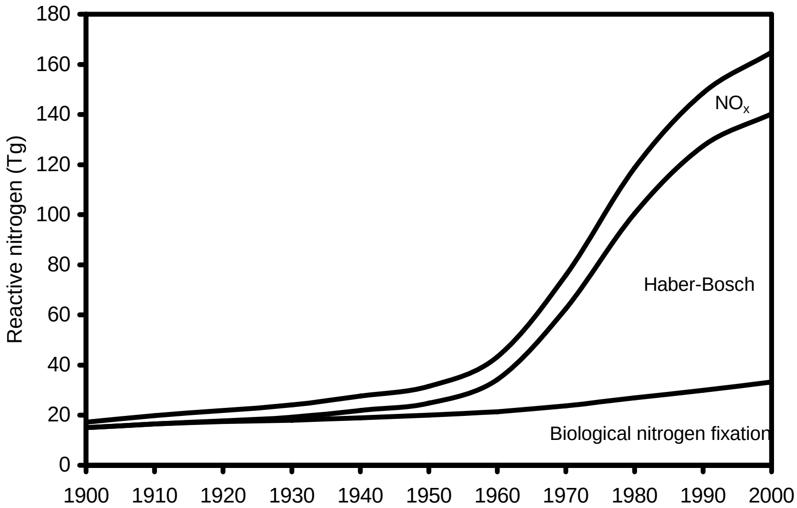

Figure 1.5. Global input for reactive nitrogen through biological nitrogen fixation, the industrial Haber-Bosch process and NO_x (based on Galloway *et al.*, 2003).

Intensively managed agro-ecosystems are the primary drivers of the changes that have occurred in the global nitrogen cycle. About 75% of the reactive nitrogen generated globally by humans is added to agro-ecosystems to sustain production or food and fibre. About 70% of this input comes from the Haber-Bosch process and about 30% from biological nitrogen fixation. There is only a small net residence of nitrogen in the agro-ecosystem, and most of the reactive nitrogen that is input to the system in a given year is lost again, either through consumption by humans or as losses to the environment.

On a global basis, about 120 Tg (1 Tg = 10^{12} g = 10^{6} t) N from new reactive N (fertilizer and biologically fixed N) and about 50 Tg N from previously created N (manure, crop residues etc.) is added annually to global agro-ecosystems (Figure 1.6). Only about a third of this N input is converted into crop yield, whereas the rest is lost, primarily to the environment (Raun and Johnson, 1999). Animals consume about 33 Tg N per year of crop produce and humans consume about 15 Tg per year. Of the nitrogen input consumed by animals, only about 15% is converted to food used by humans. Of the 120 Tg N per year in new reactive nitrogen, only 21 Tg N per year is converted to food for humans (Figure 1.6). Since the change in soil nitrogen storage is very small, the rest is lost to the environment. On a global basis 6 to 12% of the added active nitrogen is denitrified to N_2 (Smil, 2002). The remaining losses of nitrogen occur as NO_3, NO_x, NH_3 and N_2O, and all of these emissions can cascade through natural ecosystems, where they alter their dynamics and in many cases reduce ecosystem services.

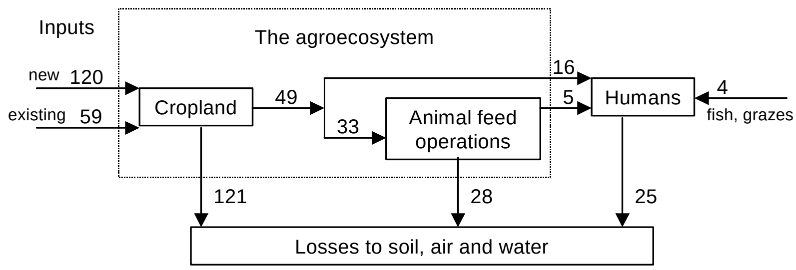

Figure 1.6. Major reactive nitrogen flows in crop and animal production components of the global agro-ecosystem (Tg N). Inputs represent new reactive nitrogen created through the Haber-Bosch process and through biological nitrogen fixation, and existing reactive nitrogen in crop residues, manure, atmospheric deposition, irrigation water and seeds. Portions of the lost reactive nitrogen may be reintroduced into the cropland component (modified after Galloway *et al.*, 2003 who refer to Smil, 2002).

Since the 1970s extensive leaching of nitrate from soils into surface and groundwater has become an issue in almost all industrial countries (OECD, 2001b). OECD (2001a) estimated that agriculture accounts for around two-thirds of nitrogen emissions into surface and marine waters and about one-third for phosphorus. In the EU countries, there is a large nitrogen surplus in the agricultural soils that can potentially pollute both surface and groundwater (Nixon *et al.*, 2003). Nitrate concentrations in rivers are highest in those Western European countries where agriculture is most intensive, but has during the 1990s been stabilized (Nixon *et al.*, 2003). Nitrate drinking water limit values (50 mg per litre) have been exceeded in around one-third of the groundwater bodies in the EU (EEA, 2003). In general, there has been no substantial improvement in the nitrate situation in European groundwater and hence nitrate pollution of groundwater remains a significant problem (EEA, 2003). Total nitrogen loading to the environment (air, soil and water) from livestock production in OECD regions is expected to increase by about 30% between 1995 and 2020 with particular large increases in Central and Eastern Europe and levels in Western Europe actually declining (OECD, 2001b). The problem of nitrate pollution of groundwater is now also serious in parts of China and India and a number of other developing countries and will presumably get worse (Zhang *et al.*, 1996). Nitrogen and phosphate enrichment of lakes, reservoirs and ponds can lead to eutrophication, resulting in high fish mortality and algae blooms, which may in the future be potentially more serious in warmer developing countries with more intense sunshine (Gross, 1998; FAO, 2003).

Pesticides

In most industrialized countries pesticides with serious toxic effects to vertebrates have been at least partially phased out. However, globally serious intoxications and incidences due to misuse of organophosphorous pesticides continue to be a problem (Satoh and Hosokawa, 2000). Both the intoxication rates and the fatality rates are highest in developing countries. UNEP (1999) estimated that global pesticide use results in 3.5–5 million acute poisonings each year.

Pesticides enter surface and groundwater from point source contamination following spillage events and from diffuse sources following their application to crops. They can be toxic to aquatic organisms and some are potentially carcinogenic (Cartwright *et al.*, 1991). In aquatic environments the leaching of pesticides into rivers, lakes and coastal waters is known to cause damage to aquatic biodiversity (OECD, 2001a).

Direct measurements of pesticides in surface or groundwater are not widely available across OECD countries, mainly because of the high costs of chemical analysis. Furthermore, many pesticides are not found in water bodies simply because they are not searched for, although when they are looked for they are frequently detected (OECD, 2001a). While the use of pesticides has fallen in many OECD countries since the mid 1980s, the long time lag between use and their detection in groundwater means that, as with nitrates, the situation could deteriorate before it starts to improve (OECD, 2001b). According to a survey of pesticide pollution of waters in the USA, in agricultural areas more than 80% of sampled rivers and fish contained one, or more often, several pesticides. Pesticides found in rivers were primarily those that are currently used, whereas in fish, organochlorine insecticides, such as DDT (now prohibited), which were used decades ago, were detected (USGS, 1999). The US survey also revealed that nearly 60% of wells sampled in agricultural areas contained one or more pesticides. The results of pesticide sampling in groundwater across a number of European Union countries, found a considerable number of sites with pesticide concentrations >0.1 µg per litre, which is the maximum admissible concentration of pesticides specified in the EU Drinking Water Directive (EEA, 1998). Finally, a French study found excessive quantities in the water environment, with surface waters being most affected where only 3% of the monitoring points showed no pesticides were present, and groundwater being better protected with 52% of all monitoring points considered to be unaffected (IFEN, 1998 cf. OECD, 2001a). Pesticide pollution is now appearing in developing countries as well and is likely to grow more rapidly than in developed countries (FAO, 2003).

The use of pesticides also affects terrestrial flora and fauna (OECD, 2001a). Herbicides are known to give rise to a decline in the flora of arable cropping systems (Andreasen *et al.*, 1996). The flora of farming systems are particularly diverse along the field margins, where herbicide uses also reduce biodiversity by removing or reducing the first step (plants) in the food web for e.g. birds and mammals (Chiverton and Sotherton, 1991). Farmland bird populations in the EU countries have fallen substantially in recent decades (EEA, 2004). The herbicide usages have been reported to have direct and knock-on effects on invertebrate abundance and species diversity (Moreby *et al.*, 1994). Broad-spectrum insecticides can cause substantial damage to populations of beneficial invertebrates and honeybees (Grieg-Smith *et al.*, 1995). Hence loss of biodiversity is not limited to the land clearing stage, but continues long afterwards.

Socio-economic impacts

Developments in agriculture and food systems such as industrialization and globalization have had socio-economic impacts all over the world, both for the millions who are engaged in farming and for the urban populations, as illustrated in Figure 1.1. More details on socio-economic impacts are discussed below.

The present phase of globalization, characterized chiefly by the proliferation of wireless communications, satellite television and the Internet, may be seen as the final outcome of a process that began in the mid-19th century with the first network technologies; the railroads and the telegraph. Beginning with these two early agents of mass transport and mass communication, the 20th century could well be characterized as the coming into being of a global mass society. Social, economic and political life has become increasingly dominated by the rise and spread of technologies of mass production and mass transport that are highly intensive in the use of energy, minerals and capital. With the accompanying trends of urbanization and rapid population growth, the impacts on agriculture and rural communities have been enormous worldwide (The Ecologist, 1993).

Industrialized countries

Agricultural modernization in the 20th century has brought major changes in socio-economic conditions in the industrialized countries of Western Europe, Oceania and North America. Along with the increases in agricultural production (see ‘Trends in agricultural production’), smaller farms have been consolidated into larger ones and there has been a dramatic decline in the percentage of the population engaged in agricultural activities (FAO, 2000). Thus, in the USA, the number of farms has shrunk from about 6 million in 1950, to about 2 million today (Pretty, 2002). With the shift of agriculture, from small- and medium-scale farms serving local needs to a mass-production industry aiming at global markets, has come the growth of international competition for selling surplus agricultural produce, and the constant pressures to lower costs. Agricultural modernization has thus resulted in an abundance of raw and processed foods in national and international markets, with declining food prices (FAO, 2003). Cheaper food allows consumers in industrialized countries to spend only a small percentage of their household disposable income on food (10% for American consumers in 2003). Furthermore, a larger variety of food, especially fruit and vegetables, independent of season, can presumably be beneficial for public health and may help to revive the cultivation of some marginalized crops, such as certain millets and legumes. Despite the falling commodity prices of agricultural produce such as maize and soybean, the price of food has continued to rise with inflation (FAO, 2003), an increase attributed to the marketing costs of agribusiness and food companies, such as transportation, packaging etc. Declining real prices of agricultural produce also implies that governments in the industrialized countries have had to constantly prop up their small rural populations engaged in high external inputs agriculture with large subsidies and other incentives. These farms, in turn, have been forced to consolidate into ever larger operations and enter into contracts with large agribusiness corporations in order to remain economically viable. Thus, in the USA, about 60 to 90% of all wheat, maize and rice is marketed by only six transnational companies; and about 90% of poultry production is controlled by just ten companies (Pretty, 2002). Trends in Western Europe have been similar over the past few decades (see e.g. Mies and Bennholdt-Thomsen, 1999).

Developing countries

Farm sizes in many developing countries are typically small (often less than 1 or 2 ha). In addition there is often a substantial rural population of landless households. Therefore, on-farm mechanization of agricultural activities has not occurred to the same extent as in industrialized countries. However, many trends of modern agriculture (often hailed by many agricultural scientists, governments and international donor agencies as the ‘Green Revolution’) have also been witnessed by most developing countries over the past three decades. With their large rural populations and small land-holdings, the arrival of high-input agriculture has brought sweeping socio-economic impacts upon tens of millions of families in Asia, Latin America and, lately, Africa as well. Certainly, some parts of the rural population have benefited greatly from better irrigation facilities and access to subsidized diesel and electricity for pumping water from canals or deep aquifers. But, the vast majority of rural households in developing countries, especially sub-Saharan Africa (SSA), lack the ecological resources or the financial means to shift to intensive modern agricultural practices.

Integration into the global markets can be a two-edged sword for farmers in developing countries (FAO, 2000). With declining real prices of agricultural produce, farmers in developing countries tend to focus on cash crops such as cotton, paddy, sugarcane and groundnuts to take advantage of the widening access to external trade, and are forced to adopt many modern practices such as the increased use of chemical fertilizers and pesticides. This entails significant increase in the costs of agricultural inputs such as high-yielding seeds, chemical fertilizers and pesticides. The socio-economic impacts of this have become plainly visible in South Asia, with its large population of small farmers and landless labourers (Shiva, 1991). Lacking sufficient access to financial institutions (e.g. microfinance and rural credit), small farmers and labourers tend to borrow from local moneylenders at exorbitant rates of interest, which they are often unable to repay due to the vagaries of weather or unfavourable market conditions. This results in a deepening of the economic problems for small farmers in developing countries (see e.g. Sainath, 1996). The farmers are thus obliged to concentrate their efforts on short-term returns and to neglect the maintenance of the cultivated ecosystem, leading to fertility decline (FAO, 2000). This process of impoverishment and exclusion is affecting primarily the most deprived, small farmers who are especially numerous in resource-poor regions and constituting the bulk density (three-quarters) of the undernourished people in the world (FAO, 2000).

Focusing on cash crops leads furthermore to a decline in local food production and increased dependency of food imports (FAO, 2000). Developing countries have become increasingly dependent on agricultural imports. A rapid growth in imports of temperate-zone commodities (especially meat) has been seen and is expected to continue far into the 21st century (FAO, 2003). Some regions have remained sheltered for a long time from the cheap imports of cereals and other staple foods from the more advantaged regions and countries, being able to maintain their production systems longer than others. However, as soon as these regions are penetrated by the advance of motorized transport and commerce, they also find themselves caught up in interregional trade, exposed to low-cost imports of cereals and other food commodities (FAO, 2000).

Food security

For the past few decades, global food production has generally been adequate to meet human nutritional demands, and has kept pace with the rapid growth in human population. Food security has been substantially increased for some developing countries over the last decades, whereas other countries such as sub-Saharan Africa have seen no improvements. With the socio-economic disparities and political asymmetries that continue to exist, nearly 800 million people remain undernourished (see Chapter 10), where the vast majority of this undernourished population lives in rural areas and urban shanties of South Asia and sub-Saharan Africa (FAO, 2000). Thus on one hand, increases in agricultural productivity and falling real prices of produce benefit global food buyers and even raise the economic status of the urban poor in developing countries, helping to reduce food insecurity for many. On the other hand, a combination of specialization, industrialization and increased price competition, accompanied by negative environmental externalities, holds the risk of marginalizing a large number of small agricultural producers in developing countries. Exacerbating this problem is the underdevelopment of regional food storage and distribution systems linking small producers to local and regional markets. Even where such systems exist (such as the public distribution system in India), small producers in developing countries are often unable to take advantage of them due to socio-economic inequities and political imbalances that exist in many rural areas.

On balance, the socio-economic implications of agricultural trends and the larger impacts of globalization are twofold. Based on the problems described above and the principles of organic farming, it is interesting to discuss the potential of organic farming for contributing to a solution to some of the issues. These opportunities, if utilized well, may reverse some of the ill-effects of modern agriculture witnessed in the 20th century as discussed in the (following) sections. This includes both the environmental problems in intensive agriculture and the problem that there does not appear to be sufficient safeguards and policies to ensure that small producers in developing countries can benefit from the present phase of globalization. A broad range of initiatives to foster sustainable land, energy and water use practices, and social equity policies at the regional, national and international levels will be required if global trends toward organic agriculture and renewable energy, for example, are to prove beneficial to small agricultural producers in developing countries. Additionally, the role of non-governmental organizations (regional, national and international) in helping address issues of smallholder farms can be critical if these producers are to benefit from the global trends toward organic agriculture.

Box 1.1. Case study on increasing Roundup Ready soybean export from Argentina.

Increasing Roundup Ready soybean export from Argentina
by Walter Pengue

The soybean production area in Argentina has shown a remarkable increase within the last decade caused by an increasing global demand for soybeans for the pig and poultry industry, an open market and a strong campaign of technological change to Roundup Ready (RR) soybeans among other things. Concurrently with the expansion of the RR soybean production in Argentina, the use of glyphosate has showed a remarkable increase too. However, excessive reliance on a single agricultural technology, like RR soybeans and glyphosate, can set the stage for pest and environmental problems that can erode systems performance and profitability. In the following a case study on the expanding soybean production in Argentina will be presented, focusing on the agricultural and environmental sustainability.

Expanding soybean export from Argentina
Over the last decade, soybean has become the most important crop in Argentina. The majority of the expanding soybean production in Argentina is exported to world markets for animal protein supplement and vegetable oil (Benbrook, 2005). Increasing demand for meat has increased the demand for fodder for e.g. the pig and poultry industry in Europe. At the same time globalization has expanded global markets for agricultural commodities and enabled production to be separated from consumption in geographical terms.

Argentina is the world's leading exporter of cake of soybeans, followed by Brazil (FAO, 2005a). Since 1997, the export of cake of soybean from Argentina has increased dramatically from 8 million t to 18.5 million t in 2003 (FAO, 2005a). The importing countries are primarily European countries, such as Spain, Italy, The Netherlands and Denmark (Figure 1.7). The majority (82%) of the cake of soybean imported to Denmark in 2003 came from Argentina (FAO, 2005a). Denmark is the world's leading exporter of pig meat and the cake of soybean is primarily used for the pig production (FAO, 2005b).

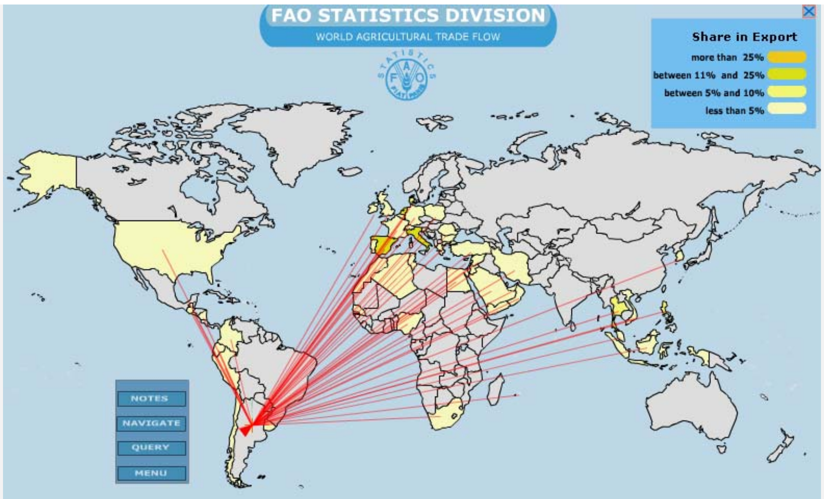

Figure 1.7. Export of cake of soybeans from Argentina in 2003 (18,476,000 t) (FAO, 2005a). The lines show the export of cake of soybeans to different countries, where Spain, The Netherlands, Italy and Denmark are the major importers of cake of soybean from Argentina.

Rapid adoption of RR soybeans and expanding soybean areas

The dramatic growth of the soybean industry in Argentina was made possible by the combination of two technologies – no tillage system and transgenic Roundup Ready (RR) soybeans. Since 1996, the area devoted to soybean production increased a remarkable 2.4-fold, from 6 million ha to 14.2 million ha in 2004 (Figure 1.8). Of the land devoted to major crops, approximately 50% was grown with soybeans in 2003. Over a 4-year period from 1997–2001, the adoption rate of transgenic RR soybeans rose dramatically from 6 to 90%.

The increase in the soybean area and the rapid adoption of transgenic soybean were a direct consequence of globalization in commodity trade, an open market and a strong campaign on technological changes. For the farmers, RR soybean came up with a solution for one of the main problems in the farm management, namely weed control. A cost reduction in the herbicide price, less fossil energy consumption and simple application made the offer of the technical package very attractive. For the private pesticide and seed production sector, it opened a unique possibility to concentrate and rearrange the business of production and commercialization of insecticides and herbicides to the new biotechnological alternative.

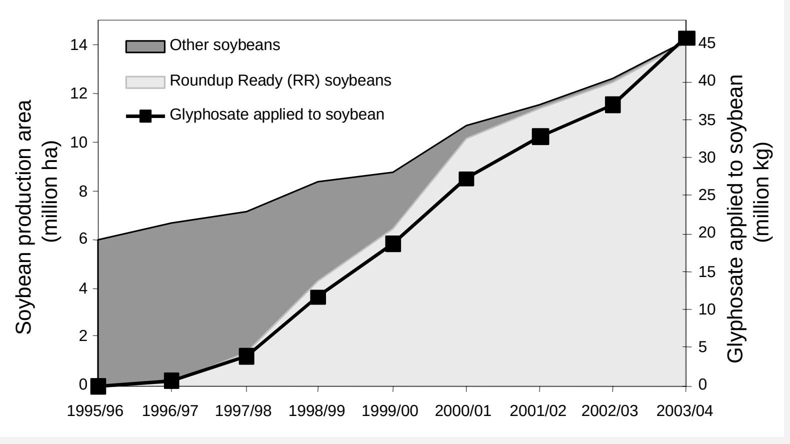

Figure 1.8. Soybean production area (million ha) and glyphosate consumption (million kg active ingredient) in Argentina from 1996 to 2004 (modified after Benbrook, 2005).

At first, soybeans were mainly produced on Pampas, one of the naturally most productive places in the world. But currently, due to the need for larger scale production, farmers are expanding the area and increasing the pressure on more environmentally sensitive areas.

During the period of expansion (1996–2004) in soybean production, the new areas needed for soybean production came from four main sources; i) approximately 25% came from conversion of cropland growing wheat, maize, sunflowers and sorghum; ii) approximately 7% came from conversion of areas growing other crops including rice, cotton, beans and oats; iii) approximately 27% came from conversion of former pastures and hay fields, and finally; iv) an estimated 41% came from conversion of wild lands, including forests and savannahs.

The Argentinean agricultural sector has set the goal of a total grain production of 100 million t by 2010, of which the soybean production is projected to be 45 million t. Achieving this goal would require an increase of the soybean planting area to about 17 million ha (Benbrook, 2005).

Increasing glyphosate consumption and resistant weeds

Given the expansion of the RR soybean hectares and the no-till systems, glyphosate herbicide usages has also risen dramatically (Figure 1.8). However, the reliance on a single herbicide year after year accelerates the emergence of genetically resistant weed phenotypes. It is predicted that continual glyphosate application for longer periods of time might lead to the development or higher increases in abundance of weeds tolerant to the herbicide (Puricelli and Tuesca, 2005). Tolerance to glyphosate in certain weeds in Argentina has already been documented (Puricelli and Tuesca, 2005; Vitta *et al.*, 2004). Given the steady increase in the intensity of glyphosate use in Argentina, the development of resistant weeds is essentially inevitable (Benbrook, 2005). The unresolved questions include how fast will resistant weeds spread, how will farmers respond and how will the spread of resistant weeds impact weed management costs, efficacy and crop yields?

Phosphorus export and depletion of Argentinean soils

In Argentina, soybean has been cropped without fertilization, although soil phosphorus (P) contents have decreased. Areas previously considered well supplied are at present P-deficient (Scheiner *et al.*, 1996). The demand for phosphorus and depletion of natural reposition is particular important in the Pampas, where the P extraction has been increasing during the last decade (Casas, 2003).

The intensification of the production system was followed by a decline in soil fertility and increase of soil erosion (Prego, 1997). Consequently, during the last decade, fertilizer consumption stepped up from 0.3 million t in 1990 to 2.5 million t in 1999. The increase in the soybean sector in the 1990s and the increase in fertilizer use thus drove the Argentinean Pampas into a more intensive agriculture that is typical of the Northern hemisphere. Before that the nutrient budgets of the Pampas were relatively stable, with a rotation of crops and cattle being the most common production system.

Each year the country exports a considerable amount of nutrients – especially nitrogen, phosphorus and potassium, in its grains – that are not replenished, except from the part of nitrogen that is derived from N_2 fixation. Argentina annually exports around 3.5 million t of nutrients – with no recognition in the market prices, increasing the 'ecological debt' (Martinez Alier and Oliveras, 2003). Soybean, the engine of this transformation, represents around 50% of this. If the natural depletion were compensated with mineral fertilizers, Argentina will need around 1.1 million t of phosphorous fertilizers and an amount of 330 million American dollars to buy it in the international market (Pengue, 2003). Estimations for 2002 showed that around 30% of the whole soybean area was fertilized with mineral fertilizers. Ventimiglia (2003) predicts that nutrients of Argentinean soils will be consumed in 50 years with the current trend in nutrient depletion in Argentinean soils and an increasing soybean area.

Increasing soybean production – and the environmental impacts

Soybean has had and will have, an emblematic role in relation with nutrient balance, loss of quality and richness of Argentinean soils, and in marginal areas it has transformed itself into an important factor of deforestation. During the last years, advances on natural areas in Argentina have known no limits. Forest areas and marginal lands are facing the advances of agricultural borders. The campaign to increase grain production to 100 million t by 2010 will demand more land for grain crops and especially soybeans. An important part of these hectares are new land, which implies deforestation and loss of biodiversity (in terms of bioecological and sociocultural concept), replacement of other productive systems (dairy, cattle, horticulture, other grains) or an advance on marginal lands.

From an ecological economics point of view, the agricultural border expansion without environmental and territorial considerations will produce not only environmental transformations but also social and economic consequences that Argentina, and the world, is currently not considering. On the one hand, Argentina is facing an important degradation of soil and biodiversity in the country that is being promoted to solve only with the application of mineral imported fertilizers, with more environmental impacts. On the other hand the countries importing the grain and nutrients are facing problems of eutrophication and loss of habitats and biodiversity due to accumulation of especially nitrogen and phosphorus in the environment (see further 'Environmental impacts').

Box 1.2. Case study on beef trade and deforestation of the Brazilian Amazon.

Beef trade and deforestation in Brazilian Amazon

The increased globalization and demand for meat has increased Brazilian beef exports significantly during the last decade, with the EU importing a significant fraction. However, according to a recent World Bank report, medium- and large-scale cattle ranching is the key driving force behind recent deforestation in the Brazilian Amazon (Margulis, 2004). Sustainable cattle grazing is however not necessarily linked to environmental losses, but is a widely used management tool in restoration and conservation of semi-natural grasslands to e.g. reverse the decline of northern European floristic diversity.

Beef production in the EU has decreased by nearly 10% between 1999 and 2003 and a further decrease is expected (Anonymous, 2004b). For the first time in 20 years beef production was lower than consumption in 2003 in the EU and it is projected that the EU will remain a net importer of beef until at least 2011. The main reasons are a declining dairy cattle herd, the impact of the market disruptions of the 2001 BSE crisis and an expected impact of decoupling of direct payments (such as suckling cow premium and slaughter premium) from 2005 (Anonymous, 2004b).

More than 55% of the beef imported to the EU comes from Brazil (Anonymous, 2004a). Beef production in Brazil has been rapidly increasing during the last 10 years (Figure 1.9; FAO, 2005b). According to FAO (2005b), Brazil was, in 2003, the third largest exporter of boneless beef and veal in the world, in volume terms after Australia and USA. More than one-third of these exports go to the EU (Figure 1.9) and the remainder is sold primarily to Chile, Russia and Egypt (FAO, 2005a). Projections show a steady increase in beef production in Brazil (at more than 3.2% per year on average from 2004–11) (Anonymous, 2004b). Demand is expected to grow rapidly in Asia, Egypt and Russia (Anonymous, 2004b).

According to Kaimowitz *et al.* (2004), Brazilian beef exports have grown markedly mainly due to devaluation of the Brazilian currency (Cattaneo, 2002) and factors related to animal diseases. Other factors in the Amazon have also given greater force to the dynamics, such as expansion in roads, electricity, slaughterhouses etc. and very low land prices and easy illegally occupation of government land (Kaimowitz *et al.*, 2004). The overwhelming majority of the new cattle are concentrated in the Amazon states of Mato Grosso, Para and Rondonia, which are also the states with the most deforestation (Figure 1.10).

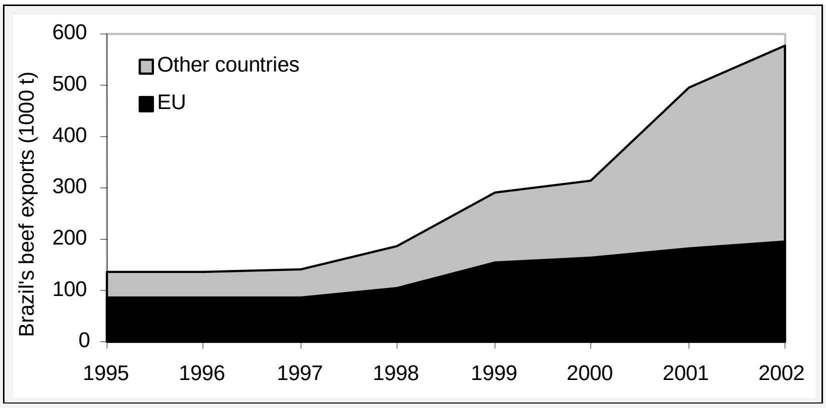

Figure 1.9. Brazil’s beef exports (1000 t) to the EU and other countries (based on a table in Kaimowitz *et al.*, 2004).

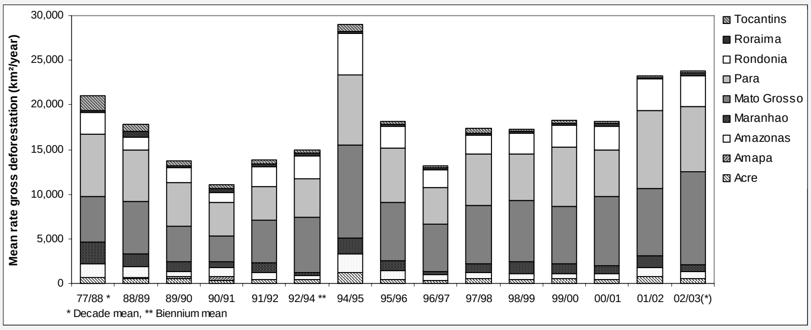

Figure 1.10. Deforestation rates in Brazilian Amazon (km^2/year) (modified after INPE, 2004).

According to a World Bank report, medium and large-scale cattle ranchers are the key driving force behind recent deforestation in Brazilian Amazon, and the overall social and economic gains are less than the environmental losses (Margulis, 2004). The expansion of the soybean cultivation into the Amazon explains only a small percentage of total deforestation according to Kaimowitz *et al.* (2004), who notes that logging is only partially responsible for deforestation, and is much less important than the growth of cattle ranching. Contrary to the occupation process in the 1970s and 1980s that was largely induced by government subsidies and policies, the dynamics of the recent occupation process gradually has become more autonomous, as indicated by the significant increase in deforestation in the 1990s despite the substantial reduction of subsidies and incentives by government. The study argues, that from a social perspective the private benefits from large-scale cattle ranching are largely exclusive, having contributed little to alleviate social and economic inequalities (Margulis, 2004).

Cattle grazing in the world however are not necessarily linked to environmental losses. Sustainable livestock grazing can enhance plant species richness and diversity of grasslands (Dupré and Diekmann, 2001; Pykälä, 2003; 2005; Rodriguez *et al.*, 2003; Pakeman, 2004) and is a widely used management tool in conservation programmes of natural grasslands (van Wieren, 1995; WallisDeVries, 1998). According to Pykälä (2003), restoration of semi-natural grasslands by cattle grazing is among the most practical options for reversing the decline of northern European floristic diversity.

Global trends in organic agriculture

Organic production and consumption has been increasing over the last decade. The organic products are not only being processed and consumed locally. Trade with organic products all over the world is a growing reality and organic products from developing countries like Uganda are being exported to e.g. Europe (see case study from Uganda in Box 1.3). However, apart from these globalization trends in organic agriculture, trends aiming at local production and consumption of organic food can also be seen (see cases from Denmark and USA in Box 1.4 and 1.5). In the following, status and developments of global organic farming will be given.

Status in global distribution of organic farming

Organic farming is practised in approximately 100 countries of the world and its share of agricultural land and farms is growing. The major part of the certified organic land is located in Australia followed by Argentina and Italy (Table 1.1). However, European countries have the highest percentage of agricultural area under organic management followed by Australia (2.5%) (Table 1.1; Willer and Yussefi, 2005).

Figure 1.11 shows the share for each continent of the total area under certified organic management. In Oceania and Latin America there are vast areas of animal pastures having a low productivity per ha, whereas the productivity per ha in European organic farming can be very high. Therefore, 1 ha in e.g. Australia cannot be directly compared to 1 ha in e.g. Denmark.

Table 1.1. 'Top ten countries worldwide' concerning percentage of agricultural area (%) or total land area (1000 ha) under organic management ranked according to highest percentage or total area (modified after Willer and Yussefi, 2005).

'Top ten worldwide' concerning land area under organic management			
Percentage organic area (%)		Total organic area (1000 ha)	
Liechtenstein	26.4	Australia	11,300
Austria	12.9	Argentina	2,800
Switzerland	10.3	Italy	1,052
Finland	7.2	USA	930
Italy	6.9	Brazil	803
Sweden	6.8	Uruguay	760
Greece	6.2	Germany	734
Denmark	6.2	Spain	725
Czech Rep.	6.0	UK	695
Slovenia	4.6	Chile	646

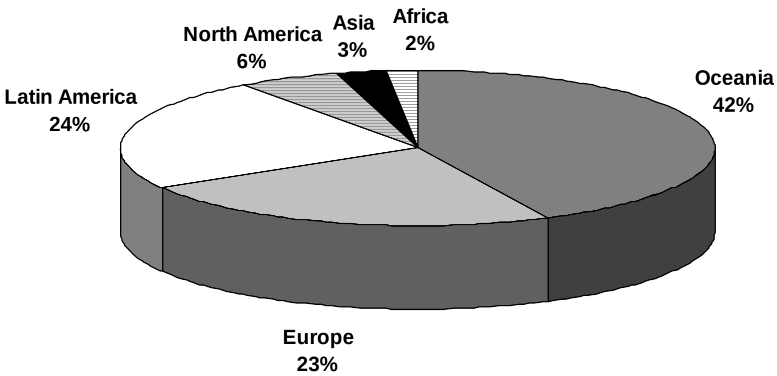

Figure 1.11. Total area under organic management – share for each continent (modified after Willer and Yussefi, 2005).

The major markets for organic food and drink are Europe and North America, which account for roughly 97% of global revenues and the markets are growing (Raynolds, 2004). Other important markets are Japan and Australia (Willer and Yussefi, 2004). Major northern markets offer good prospects for suppliers of organic products not domestically produced. These include coffee, tea, cocoa, spices, sugarcane, tropical fruits and beverages, as well as fresh produce in the off-season. Increasingly, governments in developing countries are creating conditions in support of organic export (Scialabba and Hattam, 2002). Regional markets of organic products are also expected to increase in developing countries like Brazil, China, India and South Africa along with increasing economic development and a more educated and affluent middle-class of consumers (Willer and Yussefi, 2004). Although certified organic products make up a minor share of the world food market (1–2%) it is the fastest growing segment of the food industry (Raynolds, 2004). Official interest in organic agriculture is emerging in many countries, shown by the fact that many countries have a fully implemented regulation on organic farming or are in the process of drafting regulations. Home-based certification organizations are found in 57 countries (Willer and Yussefi, 2004). The new international organic trade has two central strands, both supplying key markets in the global North. The first and largest strand is dominated by US exports to Europe and Japan, trade between European countries, and exports from Australia, New Zealand and South Africa to the top markets (Raynolds, 2004). The second strand is dominated by North–South trade and involves a growing number of production sites, most importantly in Latin America, which ship to major Northern organic markets (Raynolds, 2004). Latin America represents the hub of certified organic production in the global South, with Argentina having the greatest area and largest percentage of agricultural land under organic management (1.7%) (Willer and Yussefi, 2005). Uganda has the largest percentage of agricultural land under organic management in Africa (1.4%) (Willer and Yussefi, 2005) (see case study from Uganda in Box 1.3). A large part of African agriculture is however low external input agriculture (but not necessarily organic) where methods of the Green Revolution are risky, inappropriate or inaccessible (Willer and Yussefi, 2004). Ukraine and China are the major certified organic producers in Asia, measured by the number of certified organic hectares and enterprises, having 0.8% and 0.06% of agricultural land under certified organic management (Willer and Yussefi, 2005).

Global developments and challenges of organic farming

The organic food system has over the past two decades been transformed from a loosely coordinated local network of producers and consumers to a globalized system of formally regulated trade which links socially and spatially distant sites of production and consumption (Raynolds, 2004). Organic products were once largely produced locally, but as markets have grown, the range of organic items demanded has increased, moving beyond local seasonal products and bulk grains, to include a wide array of tropical products, counter-seasonal produce, processed foods etc. (Raynolds, 2004). Though preferences for local organic food persist, Northern countries are increasing their reliance on organic imports, particularly from the South (Raynolds, 2004). In 1998, 70% of the organic food sold in the UK was imported, 60% in Germany and The Netherlands and 25% in Denmark (Raynolds, 2004). At the same time supermarket sales of organic products have been increasing, dominating sales in the UK and Switzerland and controlling 90% of sales in Denmark. Supermarket sales comprise 20–30% of organic sales in the USA, Germany and Italy, but only 2% in The Netherlands (Raynolds, 2004).

'Conventionalization' and bifurcation in local- or export-oriented producers?

The extraordinary growth in the organic markets offers export opportunities to developing countries. At the same time the development of organic farming has led some analysts to warn that organic farming might be pushed towards the conventional farming model as agribusiness capital penetrates the organic community and its markets (Buck *et al.*, 1997; Tovey, 1997; Guthman, 2004). According to this scenario organic farming is becoming a slightly modified version of modern conventional agriculture, resulting in the same basic social, technical and economic characteristics – specialization and enlargement of farms (Milestad and Darnhofer, 2003), decreasing prices, increasing debt loads with increasing capital intensification, increased use of internal inputs and marketing becoming export-oriented rather than local (Hall and Mogyorody, 2001; Milestad and Hadatsch, 2003). Buck *et al.* (1997) are concerned that smaller alternative producers are increasingly being marginalized by larger producers who think and act like conventional producers in terms of production and marketing methods as they are forced to compete directly with larger more heavily capitalized producers within the same commodity and input markets. Although Buck *et al.* (1997) suggest that this process is leading to a bifurcation of the movement into two groups, they also argue that the alternative-oriented farmers are being pressured to adopt a number of conventional cropping, labour and marketing practices in order to survive.

In a case study of New Zealand, Coombes and Campbell (1998) found that there was some 'delocalization' in the relationship between organic producers and consumers, but due to a major growth in export-oriented organic production in New Zealand, the smaller producers were not being marginalized by the growth of larger production units or agribusiness penetration into organic agriculture. Agribusiness was focusing on converting their larger conventional growers for export-oriented markets, while the domestic markets were largely being ignored leaving the small-scale producers to continue to focus their attention on local consumers, retaining their alternative orientations and practices without any major threat or competition from agribusiness. When exporters attempted to dump certain products on the local market, there was no substantial effect on small-scale growers, as the export-oriented production was quite narrow in the range of crops, while the smaller-scale producers remained highly diversified (Coombes and Campbell, 1998). Hall and Mogyorody (2001) found little support for the idea of polarization between large export-oriented producers and small locally oriented producers in Ontario, but did find some support for the idea of 'conventionalisation' as organic field crop farmers tended to be export-oriented, large, mechanized, capitalized and specialized in cropping patterns. However, Campbell and Coombes (1999) argue that there are significant constraints and contradictions in any move to conventionalize organic farming, which creates significant space for the development of an alternative oriented organic movement. Hall and Mogyorody (2001) point out that organic farming is developing in distinct ways in different national contexts and one has to be cautious about drawing general conclusions regarding the development of organic farming. Campbell and Liepins (2001) argue that organic farming is still exceptional and provides a unique challenge to the standardizing food system. Even if it is not revolutionary, organic agriculture and food consumption highlight some ways in which the broad tendencies in food production and consumption are not linear, inevitable and uncontested – thereby providing an interesting terrain for examining the processes that are occurring at the margins of the globalizing food system. Raynolds (2004) suggests that while much of the literature on the preservation of organic movement values adopts a localist stance, these same values can be extended globally by linking small-scale peasant producers and conscientious consumers.

The development of farmers' markets, box schemes, farm gate sales, fair trade importing etc. may be seen as examples where those involved in the organic sector are attempting to develop alternative networks and patterns of control than exist in the conventional sector (La Trobe and Acott, 2000; Rigby and Bown, 2003). An example of an initially alternative trade network of organic milk in Denmark moving towards the trade patterns of the conventional sector is given in Box 1.4, where the degree of local links between food production and consumption are discussed. For some proponents of organic farming, it is exactly the potential for strengthening the local links between food production and consumption that is the promising issue. A large movement towards local production and consumption of organic food counteracts the trends of globalization in the organic sector. 'Eco-localism' is a concept presented by Curtis (2003) as an alternative economical paradigm as opposed to the global capitalist economy. The central argument is that economic sustainability is best secured by the creation of local or regional self-reliant, community economies (Curtis, 2003). An example of ongoing efforts to strengthen the local links between production and consumption of organic products in Iowa, USA, is presented in Box 1.5. The selected cases serve to illustrate the attempts to develop alternative supply networks and the problems associated with trying to 're-localize' the food chain, as the local markets have not (yet) proven adequate for sustaining a local production on a wider scale.

Environmental issues

The environmental impacts of organic farming have primarily been assessed in developed countries, pointing out however a number of benefits. Studies have shown that regarding soil biology, organic farming is usually associated with a significantly higher level of biological activity and a higher level of soil organic matter (Stolze *et al.*, 2000; Hansen *et al.*, 2001; Mâder *et al.*, 2002; Pulleman *et al.*, 2003; Oehl *et al.*, 2004), indicating a higher fertility and stability as well as moisture retention capacity (Stolze *et al.*, 2000; Scialabba and Hattam, 2002). Furthermore, Stolze *et al.* (2000) concluded that in productive areas, organic farming is currently the least detrimental farming system with respect to wildlife conservation and landscape, and a higher species diversity is generally found in organic fields (van Elsen, 2000; Pfiffner and Luka, 2003). The absence of pesticides precludes pesticide pollution and increases the number of plant species in the agricultural fields (Stoate *et al.*, 2001), which benefits natural pest control and pollinators. Organic farming furthermore reduces the risk of misuse of antibiotics (see Chapter 9).

Organic farming systems must rely on a closed nitrogen cycle and on nitrogen input via N_2 fixation by legumes. This leads to management practices that also reduce emissions of reactive nitrogen to the environment (Drinkwater *et al.*, 1998; Olesen *et al.*, 2004). The use of cover crops and mulches in organic farming also has the capacity to maintain soil fertility and reduce soil erosion. The recycling in organic farming of animal manure contributes to maintaining soil nutrients and avoiding soil degradation. Furthermore, there are indications that arable organic farming systems may reduce net greenhouse gas emissions per unit of agricultural area for arable farming systems (Robertson *et al.*, 2000).

In developing countries, organic farming has a potential of increasing natural capital, such as improved water retention in the soil, improved water tables, reduced soil erosion, improved organic matter in soils, increased biodiversity and carbon sequestration (Scialabba and Hattam, 2002; Rasul and Thapa, 2004). The potential of organic farming to enhance soil fertility and reduce soil erosion is discussed in Chapter 8. Furthermore, the risk of pesticide accidents and pollution is absent.

However, the environmental benefits of organic farming are challenged by globalization. The patterns of organic trade that are developing between North and South are to a high degree replicating those of the conventional sector. As organic produce becomes a larger part of the global food system, and as such is processed, packaged and transported more, the environmental effects become worthy of attention. ‘Food miles’ is one measure of this increasing transportation of organic food that captures the distance food travels from producer to consumer (Rigby and Bown, 2003). When measuring and discussing ‘food miles’ it can be important to distinguish between agricultural produce that can be produced locally and those that cannot. With the intensification of intra- and international transportation of organic commodities, organic agriculture systems are increasingly losing their nutrient and energy closed-system characteristic (Scialabba, 2000b) and risk encountering the same problem of nutrient transfer, depleting the production resources, as discussed in Box 1.1. The potential of closing urban–rural nutrient cycles in organic farming, especially in low-income countries, is discussed in Chapter 7. Scialabba (2000b) points to the risk that the environmental requirements of organic agriculture are becoming looser as the organic system expands and that few certification schemes explicitly mandate e.g. soil building practices, shelter for wild biodiversity and integrate animal production. This points to the need to supplement the organic farming principles with more guidelines or rules concerning e.g. ecological justice as discussed in Chapter 3.

Socio-economic issues

Organic farming in developed countries has a potential of narrowing the producer–consumer gap and enhancing local food markets (Scialabba and Hattam, 2002; see Box 1.5). Furthermore, organic farming has a potential of decreasing local food surplus and expanding employment in rural areas (Scialabba and Hattam, 2002). A better connectedness with external institutions and better access to markets has been seen through strengthened social cohesion and partnership within the organic community (Scialabba and Hattam, 2002; Box 1.4 and 1.5).

The extraordinary growing organic markets offer export opportunities to developing countries. Provided that producers of these countries are able to certify their products and access lucrative markets, returns from organic agriculture can potentially contribute to food security by increasing incomes (Scialabba, 2000b). A large number of farmers in developing countries produce for subsistence purposes and have little or no access to inputs, modern technologies and product markets. As productivity of traditional systems is often very low, organic agriculture could provide a solution to the food needs of poor farmers while relying on natural and human resources (Scialabba, 2000b). In Chapter 11, the effect of organic farming on food security will be discussed.

In developing countries, organic farming has a potential to improve social capital, such as more and stronger social organization at local level, new rules and norms for managing collective natural resources and better connectedness to external policy instruments (Scialabba and Hattam, 2002). Furthermore, improvements in human capital have been seen, such as more local capacity to experiment and solve problems, increased self-esteem in formerly marginalized groups, improved status of women, better child health and nutrition, especially from more food in dry seasons, reversed migration and more local employment (Scialabba and Hattam, 2002). It is assumed that organic agriculture in developing countries facilitates women’s participation, as it does not rely on purchased inputs and thus reduces the need for credit (Scialabba and Hattam, 2002; FAO, 2003). However, insecure long-term access to the land is a major disincentive for both men and woman, since organic agriculture requires several years to improve the soil (Scialabba and Hattam, 2002). Chapter 6 illustrates the approaches of organic farming in developing countries.

Furthermore, organic agriculture has the potential to use fair trade conventions and to introduce ecological justice and the view of the theories of ecological economics. These issues are discussed in Chapters 2 and 4. Furthermore, Chapter 4 discusses the limitations of global organic trade and Box 1.3 shows a case on organic fair trade.

However, organic food and farming are challenged by globalization and development. The increasing export-orientation and supermarket domination of the organic market goes beyond the transportation effects. Supermarkets source primarily on the basis of range, quality, availability, volume and price and hence seek large volume suppliers who can supply at competitive prices all year round (Rigby and Bown, 2003). Raynolds (2000) points out that several studies suggest that due to substantial costs and risks of organic production, much of the international trade is controlled by medium and large enterprises, challenging the assumption that it is the small farms that benefit from the growing organic market. Organic farming may offer an opportunity for marginalized smallholders to improve their production without relying on external capital and inputs and to gain premium prices from trading with industrialized countries using organic production methods that have potential benefits to e.g. soil fertility and biodiversity. However, marginal organic farmers in the South are likely to be dependent on exploitative middlemen, corporate buyers and volatile prices as are conventional producers, unless they enter into fair trade networks (Raynolds, 2000). An example of organic farming as a development agent in Uganda is given in Box 1.3, where a fair trade network has been developed between organic farmers in Uganda and a Danish company. Producers, consumers and IFOAM acknowledge the convergence between the holistic social and ecological values of the fair trade and organic movements (IFOAM, 2000; Raynolds, 2004).

The certification issue is another challenge facing organic movements, especially with regard to developing countries. The term organic agriculture is backed with strict standards and rules that govern the ‘organic’ label of certified food found on the market. However, according to Raynolds (2004), onerous and expensive certification requirements create significant barriers to entry for poor Southern producers and encourage organic production and price premiums to be concentrated in the hands of large corporate producers. Furthermore, producers often have to comply with foreign standards not necessarily adapted to their country conditions (Scialabba, 2000b). Raynolds (2004) suggests that shifting certification costs downstream and empowering local producers to fulfil monitoring tasks should reduce barriers for small-scale producers. The issues of social justice in organic agriculture are further discussed in Chapter 2 and 3 and trade with organic products is discussed in Chapter 4.

The focus on certified organic products (and attendant costs and risks) has distracted attention on this system's potential to contribute to local food security, especially in low-potential areas in developing countries (Scialabba, 2000a). According to Scialabba (2000a), market-driven organic agricultural policies need to be complemented with organic agriculture policies that target local food security. The issues of food security are further discussed in Chapter 11.

Box 1.3. Case study on trade with organic products from Uganda.

Trade as an option of enhancing development? A case story from Uganda
by Åge Dissing and Ingelis Dissing

This case study tries to look at trade as a development tool. Most developing countries have been used to export agricultural commodities to e.g. Europe due to their former status as colonies, but often the population has hardly used the products themselves. Thus development of the products and processed produce is not incorporated in the society. In countries where agriculture is dominantly based on subsistence farming with a few cash crops for bulk export, handling of products for the market is a fairly new thing.

Scope of cooperation

Two Ugandan companies had formed a partnership with a Danish retail company in organic produce in order to supply the Danish partner with dried banana, pineapple and mango. The objectives were to process and export organic fruit in a fair trade arrangement to the Danish/European market.

The companies were both new-started, and to obtain the objectives they had to finalize and increase the infrastructure of the processing factory, including processing facilities, drying capacity and capacity-building of staff and management. On the supply side farmers should be trained in organic agriculture and be certified organic. The Danida Private Sector Development Programme has supported the cooperation.

Presentation of the Ugandan partners

Company X

The shareholder company X consists mainly of local people (like business people, teachers and agriculturists) in a middle-size town up country plus a few expatriates. The company X was initiated out of the interest of increasing agro-processing and of course of making a profit. The company was initiated as a start up trial and moved from there into a long-term cooperation of 5 years. Now 4 years after the start the company has built a factory in two stages including wet processing room, sorting/packing room, stores for fresh and dried produce and an office. Furthermore different types of dryers, a water tank, eco-toilets, a bathroom, a changing room and a store have been constructed. Factory staff and the board have been trained, and training and certification of factory and farms is an ongoing process. The monthly production is now on average 1000 kg of dried pineapple and banana, equivalent to 15,000 kg of fresh fruit from the farmers.

Company Y

The privately owned company Y has also a couple of other businesses to sustain the family, and to invest in the drying business. The company Y started like company X almost from zero like a start up trial and have reached almost the same infrastructure. However, the second stage of the factory is not finalized and fewer dryers are built. Director, staff and farmers have been trained, factory and farms are certified, but some inconsistency in the management policy has caused a high staff turnover. The monthly fruit production is now on average 400 kg of pineapple, banana and mango, equivalent to 6000 kg of fresh fruit, but the availability of fruit and the processing is very uneven over the year.

Hardships and obstacles for agro-business in Uganda

In fact there are many, but let us shortly describe the most obvious ones found in the two companies and the trade arrangement.

1. Financing a new business and especially in the countryside is almost impossible if you do not have the needed cash to invest. Banks give out only short-term loans and on very harsh conditions – at least as long as you are new in business. The main reason is that the government uses all money available to finance their part of donor investments like roads, hospitals etc. Most people do not opt for long-term investments; they still take actions from season to season, and prefer to invest in land, houses for renting or cows. A savings culture is not incorporated in society, partly due to family structures, where those who have money must give out to all relatives in need.
2. Management attitudes. It is hard to find people with a lot of management knowledge and experience, especially in agro-business. Uganda is recently recognized as a country with a very high level of entrepreneurs, but it is predominantly on very small scale like starting a tiny stall on the market or making bread on a veranda. Management experience to go into export is still difficult to find.
3. Consistency within the workforce on the factory and at farm level as well, is often very difficult and is therefore time-consuming. The inconsistency in the workforce is generally the case for both the leadership and the general staff in the factory. Industrialized working attitudes are new, as often seen in traditional agricultural societies.
4. Lack of proper logistics at all levels causes financial losses. Better logistics are needed to e.g. ensure that the needed fruit is available in time, and that the factory has all needed utensils etc. in place, not to lose too much time and money in the process.
5. Partnership and cooperation for mutual benefit is often difficult to create. At least it takes some years as farmers especially have been cheated by governmental ‘cooperatives’ and exploitative middlemen. Not only farmers have bad experiences of that kind, it also includes traders and other companies.
6. Cultural differences are big when African and European lifestyle and business attitudes have to find a mutual understanding. Industrialized countries have very difficult markets and are furthermore very protective and that is an additional constraint in a trade arrangement.

Conclusions

The current school system (especial secondary school) is not encouraging questions, curiosity and personal developments, on the contrary. This indicates that a boss can still handle staff in the usual feudalistic way, and thus developments are difficult, as we have seen in company Y.

A culture of subsistence farming is difficult to leave for farmers; they have to give up a lot of independence and freedom when going for commercial farming. In fact it is a very big change of lifestyle and work. Nevertheless, farmers connected to the two companies can now deliver the quantities and qualities required.

Factory work requires strict consistency in the workforce, which is not usual in countries that have not had the impact of long-term industrialized experiences. But company X has now built some capacity within the staff.

A fair trade agreement has a lot of good impact for the companies: fair prices for farmers and company, fair conditions for staff, transparency in the cooperation. It can include prepayment of the fruit, but in company X it has somehow caused delay in adjustment of the business as prepayment arrived anyway – for a period at least.

It is possible to see some good impact from these two trade arrangements. A lot of capacity building within farms, the staff and directors has taken place. In a country like Uganda it is definitely still needed, with some development supporting training and technical assistance to demonstrate success stories in international fair trade.

Box 1.4. Case study on local trade with organic milk products in Denmark.

Eco-localism and trade with organic products the case of Thise Dairy in Denmark

by Chris Kjeldsen

The dairy sector in Denmark is dominated by one big shareholder company that apart from the conventional milk also has organic shareholders and trades organic milk. However, the organic cooperative dairy Thise has successfully been established at the market through both alternative and more ordinary distribution channels. Thise Dairy was rooted in closer contact between producer and consumer and was initially a local dairy. Thise Dairy has over years increased its sales to all over Denmark.

Thise Dairy is an independent cooperative dairy, which was started in 1987, when a group of organic farmers in northern Denmark approached a privately owned dairy plant in an effort to acquire processing facilities for their production of organic milk. As a result of these negotiations, the cooperative, Thise organic dairy was formed in September 1988. The scale for the cooperative dairy was relatively small initially, reflecting the modest size of the market for organic milk at the time. There were only eight shareholders (organic and biodynamical farmers) in the cooperative, and the amount of milk weighed in at the dairy plant was only 1.6 million kg/year in the period 1989–90 (Jensen and Michelsen, 1991). The first 2 years were very costly for the cooperative, since they had to establish their own distribution network. One of the problems encountered in the early days of the dairy was that they distributed small amounts of milk over long distances to various rural locations in northern Denmark, e.g. shop owners with a sufficient dedication to organic products (Jensen and Michelsen, 1991). The main reason behind this very costly distribution strategy was that Thise could only gain access to stores without contracts with the major retail chains and their distribution networks. The problem was that individual shops within the major retail chains have only very limited autonomy regarding what to put on the shelf, since they are obliged to use centralized distribution networks.

Distribution costs for Thise Dairy were reduced by about 70% in 1990, when Thise joined a national distribution and sales organization for Danish organic dairy farmers, Dansk Naturmælk, which made it possible for Thise to sell their milk to some of the major retail chains, most notably ‘FDB’ (Jensen and Michelsen, 1991). The reduced distribution costs were mainly due to the fact that Thise, and with them other independent dairies, now could use the distribution network of the dominating (conventional) dairies ‘MD Foods/Kløvermælk’. Due to financial and organizational problems within Dansk Naturmælk, the organization was terminated in 1992. After the termination of Dansk Naturmælk’s agreement with the large retail chains in 1992, the future appeared quite bleak for Thise.

A crucial turning point for Thise Dairy happened in 1993. The Danish market for organic food expanded radically, when ‘FDB’ discounted organic products, which had a significant influence on Thise’s sales to ‘FDB’. The most important event in Thise’s history took place in 1995, when Thise Dairy signed a contract with the retail chain Irma in Copenhagen. Irma has since then been the most important distribution channel for Thise Dairy. Today, around 50% of Thise’s products are being sold in Irma shops in Copenhagen. Thise has more than doubled its sale, both in terms of turnover and the amount of milk weighed in at the dairy. At the same time, the number of shareholders in the cooperative has expanded to 42 (Anonymous, 2004c). In the late 1990s, Irma was bought by ‘FDB’, which marked an important change, since the forward sales of milk were sold in Irma’s own brand. However, Thise has maintained a high degree of branding of their own name, and is often praised in the media for their high degree of innovation in developing new product types. Compared to the much larger cooperative dairy Arla with 15–16,000 shareholders (that dominates the Danish dairy market and sells both conventional and organic dairy products), Thise launches a much wider range of new products each year, and has been quite a trendsetter in the organic dairy sector.

As the map below illustrates (Figure 1.12), the most important market for Thise Dairy today is in Copenhagen, where around 50% of their dairy products are being sold, approximately 400 km from Thise Dairy. The second most important markets are export markets in England, Germany and Sweden, where up to 20% are being sold (Anonymous, 2004c). The dotted line on the map indicates the initial ‘heartland’ of Thise, within which most of their sales were taking place in the early 1990s. The majority of the producers are still placed within or around that perimeter. Although Copenhagen is the major market, Thise is trying to diversify its operations, since it also sells its milk via different alternative distribution channels. One example is that Thise are selling dairy products to a company called ‘Anemonemælk’, a web-based milk delivery scheme, which delivers milk and other dairy products to people’s doorsteps in the wider Århus area. Other examples are health shops throughout the country.

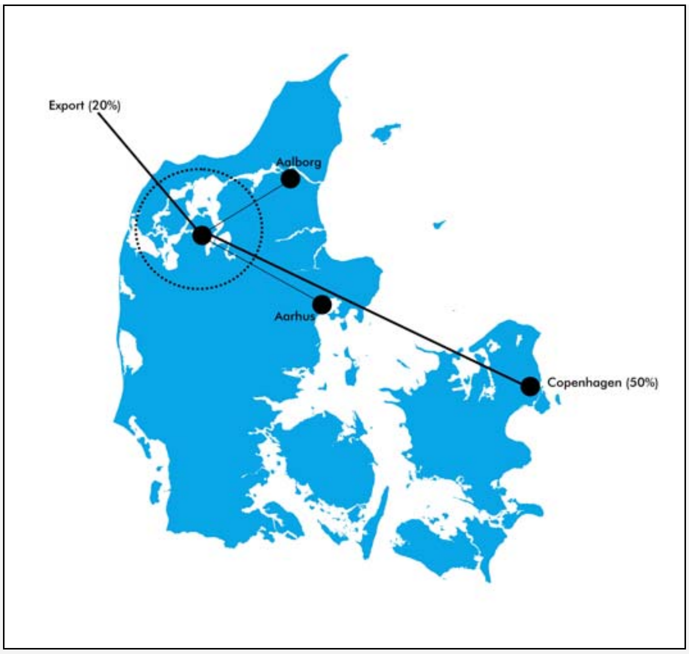

Figure 1.12. Pathways for products from Thise Dairy in Denmark.

Compared to the initial circulation of their products in a primarily rural (and regional) setting, Thise has moved beyond that context, now circulating its products in a primarily urban setting, geographically remote from the production sites in the network. One of the main reasons for this shift in direction was the inability of the local markets to support an economically viable scale of production, reflected in the fact that the shareholders for prolonged periods in the early history of Thise had to accept lower prices than at the other organic dairies, for example MD Foods (Arla).

Conclusion

Thise Dairy has moved from an alternative distribution network towards the supply patterns resembling those of the conventional sector. The initial supply pattern was characterized by a center–periphery structure dependent on place and personal relations as seen in some box schemes, farm shops etc. Thise Dairy has over time moved towards another distribution pattern characterized by standardization and regulation requiring no personal relations and less dependency of place, as seen in the supermarket distribution. The case illustrates the trend in Denmark where supermarket sales represent 90% of the sales of organic products (Raynolds, 2004). However, there are elements of the Thise Dairy, that exhibit some degree of 'regionalization' or dependency on place, as expressed in the idea of 'eco-localism'. It can be argued that Irma is a primarily regional based retail chain, and that 'Anemonemælk' is regionalizing Thise's products around Århus. Furthermore, one should of course not forget the very important regional importance of Thise in its 'home region' in terms of local jobs. But an important issue in this regard is that each of these distinctive patterns is not spatially adjacent to each other and their interaction is primarily based on the standardization. Thise Dairy can, however, in some ways be described as expressing some of the classical virtues of the Danish cooperative dairy sector, such as producer autonomy through cooperative organization, a high degree of innovation and also an orientation towards exporting their products. The challenge of Thise is to span across different geographical and social spaces in order to recruit enough consumers to obtain a viable economic scale. It has proven a very successful market strategy, but leaves other challenges to be met, both regarding how to 'regionalize' the circulation of organic milk and how to obtain a higher degree of social integration between producers and consumers.

Box 1.5. Case study on foodsheds and eco-localism in the USA.

The development of (local) foodsheds in Iowa
by Chris Kjeldsen

The notion of foodsheds has its origin in the use of watersheds as the organizing spatial unit for integrated biophysical and social systems in bioregionalism (Wackernagel and Rees, 1996; Hansson and Wackernagel, 1999). In the same manner, the notion of foodsheds has been proposed as an organizing spatial unit for closely integrated networks between production and consumption of food (Kloppenburg *et al.*, 1996). Taken at face value, the notion of foodsheds implies a strong degree of local embeddedness. In practice though, this might not be the case, since many food networks labelled as 'sustainable' might exhibit a large scale in terms of size of their foodshed. One obvious example is fair trade networks, where producers and consumers are half a world apart and products travel over very large distances.

Initiatives in Iowa
In recent years, Iowa has seen an increase in the number of food system initiatives aiming at 're-localizing' the circuit of food between producers and consumers (Hinrichs, 2003). Historically, Iowa is in a way not the most typical place for such initiatives to appear, since Iowa appears as 'the quintessential agricultural state in the US' (Hinrichs, 2003). Compared to many other Midwestern states, Iowa has less diversity in its terrain and climatic features, making it an obvious target for agricultural development. Because of its obvious potential for agricultural use, the prairie state of Iowa was rapidly ploughed and the early white settlers drained the abundant wetlands. From the early days of settlement, Iowa agriculture was oriented towards non-local (mostly national) markets. Commodity agriculture seemed to be a strong cultural force within the agricultural community, since Iowa agriculture was rapidly modernized, in terms of specialization and integration with the agri-food industry. From the mid-20th century and onwards, the range of crops grown in Iowa has decreased significantly, as well as the number of farmers active within the sector. One example is that many labour-intensive crops such as apple or other horticultural crops vanished to a large degree from Iowa (Pirog and Tyndall, 1999), being replaced

by imports from production sites within the USA, such as Washington State, or overseas producers like China. The heavily industrialized and export oriented grain–livestock–meat systems became the most typical food system in Iowa.

The interest for re-localizing Iowa food chains is very recent. Food system localization in Iowa first took place with direct marketing initiatives such as Farmer's Markets growing from a number of 50–60 markets in the early 1980's to some 120 markets by the mid-1990s (Hinrichs, 2003). The first direct markets were mainly producer-driven, more than consumer-driven and should be seen as part of a strategy aiming at finding ways to overcome the massive farm crisis for commodity agriculture during the 1980s. Important actors in this regard were county extension officers and chambers of commerce, who initiated the first direct markets. Even though direct markets remain a focus area for food systems activists, there was a growing disquiet about their limited ability to sustain the livelihoods of many Iowa farmers. Aided by the activities of other actors, such as researchers from Iowa State University and the Leopold Centre for Sustainable Agriculture, both sited in Ames, Iowa, food systems activists started initiating other projects, which were supposed to extend the possibility to channel farm production flows. One of the significant developments, which took place during the 1990s, was the growth of Community Supported Agriculture (CSA) projects. By 1996, there were nine CSAs in Iowa, whereas this number had grown to about 50 by the year 2000 (Hinrichs, 2003).

CSA was an improvement of the alternative market channels for locally produced food, but as in the case of direct markets, small, decentralized, face-to-face direct market initiatives like CSA could not sustain many Iowa producers. Instead, food system activists and organizers have increasingly focused on changing the patterns of institutional food procurement. One of the first initiatives was a publicly funded demonstration project in 1997–98, which determined that it was possible for a university dining service, a hospital and a restaurant in north–east Iowa to purchase a significant proportion of the food needs locally (Hinrichs, 2003). Another important development was the development of a type of event called the Iowa-grown banquet meal. The first of these events was held at the Leopold Centre for Sustainable Agriculture in 1997. As both a promotional event and a celebratory enactment of local Iowa foods, the banquet meals have helped to establish a new ritual that showcases and redefines local Iowa food. Since 1997, the Iowa-grown banquet meals have spread all over the state, coordinated by a brokering office of the farmer's organization Practical Farmers of Iowa, with 57 meals at 47 different events being held in 2000 (Hinrichs, 2003). A loosely knitted network of 23 farmers has supplied the food being served at these events.

As a symbolic way of redefining and sustaining a local food culture, the Iowa-grown banquet meals have been very important. Still, the banquet meal is episodic and supplemental for any individual Iowa producer (Hinrichs, 2003), and has not been able to sustain any larger number of local farmers. Organics as an element in the localization of food chains of Iowa have until now been overshadowed by the valorisation of local produce. So in that sense the banquet meals conform to what Michael Winter has termed 'defensive localism' (Winter, 2003), where localization is the top priority for development of food systems, more than progressive social and ecological priorities. The challenge for initiatives like the Iowa-grown banquet meal seems to be to balance between defensive localism and a more receptive attitude to wider social and ecological objectives.

Conclusions

- Increasing globalization and production in agriculture has primarily benefited the industrialized countries and certain developing countries such as China that are integrated into the global markets. In those countries, food security has increased, a greater variety of food has been offered and diets have changed towards a greater share of meat and dairy products.
- However, the development hides a growing disparity among agricultural systems and population, where especially developing countries in Africa have seen very few improvements in food security and production. The vast majority of rural households in developing countries lack the ecological resources or financial means to shift into intensive modern agricultural practices as well as being integrated into the global markets.
- At the same time, intensive agriculture especially in industrialized countries has contributed to environmental problems such as pollution of surface and groundwater with nitrates and pesticides, global warming, reductions in biodiversity and soil degradation, and virtual monocultures and specialized livestock productions have spread over entire regions.
- Organic farming offers a potentially more sustainable form of production. Organic farming is practised in approximately 100 countries of the world and the area is increasing. Trade with organic products all over the world is a growing reality with the major markets being Europe and North America. These major markets offer good prospects for suppliers of organic products from developing countries.
- However, the recent development holds the risk of pushing organic farming towards the conventional farming model, with specialization and enlargement of farms, increasing capital intensification and marketing becoming export-oriented rather than local. Furthermore, as the organic products are being increasingly processed, packaged and transported long-distance, the environmental effects need to be addressed.
- Organic farming might offer good prospects for marginalized smallholders to improve their production without relying on external capital and inputs, either in the form of uncertified production for local consumption or certified export to Northern markets. However, in order to create a sustainable trade with organic products focus should be given to issues like trade and economics (Chapters 4 and 5), certification obstacles, and ecological justice and fair trade (Chapters 2 and 3). Furthermore, the implications of certified and non-certified organic farming in developing countries need to be addressed (Chapters 6 and 9) including issues on soil fertility (Chapter 8) and nutrient cycles (Chapter 7) and the contribution to food security (Chapter 10).

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Paper II

Environmental assessment of organic soybean (*Glycine max.*) imported from China to Denmark: a case study

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Environmental assessment of organic soybean (*Glycine max.*) imported from China to Denmark: a case study

Marie Trydeman Knudsen [a,*], Qiao Yu-Hui [b], Luo Yan [b], Niels Halberg [c]

[a] Dept. of Agriculture and Ecology, Faculty of Life Sciences, University of Copenhagen, DK-2630 Taastrup, Denmark
[b] Dept. of Ecology and Environmental Sciences, China Agricultural University, Beijing 100083, China
[c] International Centre for Research in Organic Food Systems (ICROFS), DK-8830 Tjele, Denmark

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ABSTRACT

Growing global trade with organic products has increased the demand for environmental impact assessments during both production and transport. Environmental hotspots of organic soybeans produced in China and imported to Denmark were identified in a case study using a life cycle assessment approach. Furthermore, environmental impacts of organic and conventional soybeans at farm gate were compared in the case study. The total global warming potential (GWP) per ton organic soybeans imported to Denmark revealed that 51% came from transportation and 35% from the farm level. Comparing organic and conventional soybean at farm gate showed that GWP, non-renewable energy use, acidification and eutrophication was lower per ton organic soybeans, whereas land use was slightly higher.

1. Introduction

Global trade with organic products has increased substantially during the last decade with the major markets being Europe and North America (Willer and Kilcher, 2009). The growing markets are offering good prospects for suppliers from other parts of the world (Willer and Kilcher, 2009). China was in 2006 the country with the second highest area under organic management (Willer et al., 2008) with the primary driver being trade and export of especially organic seeds, such as soybean, sunflower and pumpkin seeds (Kledal et al., 2007). From 2003 to 2007, the organic land in China increased from 0.3 to 2.3 million hectare (Willer et al., 2008; Willer and Yussefi, 2004). In the same period, the global organic market almost doubled in value (Willer and Kilcher, 2009) and the import of Chinese organic products to Denmark increased sevenfold in value (StatBank Denmark, 2009). One of the main organic products imported from China to Denmark were soybeans for the organic livestock production (pers. comm. Agnete S. Nilsson, StatBank Denmark, 2009). Soybean (*Glycine max.*) is a common protein component in the fodder for organic livestock, especially for dairy cattle. Since soybeans are normally not produced in Denmark, due to climatic conditions, they are imported. The environmental effect of producing soybeans has gained increasing focus since they make up a considerable part of the fodder for livestock (Clay, 2004). Soybeans are used in the animal diets as a concentrate, mainly as a supplement to maize silage for dairy cows or in composite concentrates for pig and poultry. Furthermore, soybeans are used for human consumption.

Organic farming has four main principles of health, ecology, fairness and care (Ifoam, 2009). Research has shown environmental benefits from organic farming at the farm level, with the results primarily being based in the European context (Stolze et al., 2000; Hansen et al., 2001). However, the increasing global trade and import of organic products from spatially distant sites of production e.g. for the organic livestock production raises the need to estimate the actual environmental impacts of the imported organic products. The focus should be both on the environmental impacts in the production and during the transport. Life Cycle Assessment (LCA) is a method for integral assessment of several environmental impacts (e.g. climate change, eutrophication etc.) along the life cycle of a product. The LCA approach includes goal and scope definition, inventory analysis (LCI), impact assessment (LCIA) and interpretation of the results (ISO 14040 to 14043 standards). LCA

* Corresponding author. Present address: Faculty of Agricultural Science, University of Aarhus, Dept. of Agroecology and Environment, Blichers Alle 20, P.O. Box 50, DK-8830 Tjele, Denmark. Tel.: +45 8999 1308; fax: +45 8999 1200.
E-mail address: mariet.knudsen@agrsci.dk (M.T. Knudsen).

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has in recent years proven to be an internationally accepted methods also widely used in the agricultural sector for assessing the environmental impacts and for identifying a hotspot, where the environmental burden is especially high, for a product in a life cycle (Thomassen et al., 2008; Cederberg and Mattsson, 2000; Haas et al., 2001).

Few have studied the environmental impact of imported organic soybean, evaluating the whole chain from farmer to consumer. Pelletier et al. (2008) studied organic and conventional soybean produced in Canada using a life cycle impact assessment until farm gate. Likewise, Jungbluth and Frischknecht (2007) presented farm gate LCA results in a conference paper on organic and integrated soybean produced in Switzerland and conventional soybean produced in Brazil and USA, respectively. For conventional soybeans, Lehuger et al. (2009) presented LCA results of soybeans produced in Brazil and transported to France and Dalgaard et al. (2007) presented LCA results of soybeans and soybean meal produced in Argentina and transported to Denmark.

2. Goal and scope

2.1. Goal of the study

This paper aims at assessing the environmental impacts of organic soybean produced in China and transported to Denmark using a life cycle approach. The objectives are 1) to identify the environmental hotspots in the product chain of organic soybeans from China imported to Denmark and 2) to compare the environmental impacts at the farm gate of the organic soybean production with a comparable conventional soybean production in the same region in China.

Information about environmental impacts of organic product chains is needed to make informed choices in regulations and at the policy and consumer level. This information will feed into the ongoing discussion of two main environmental aspects concerning agricultural products: 1) global versus local procurement [e.g. Edwards-Jones, 2008] and 2) organic versus conventional production [e.g. Mondelaers et al., 2009]. Furthermore the study will be valuable for scientists and others developing LCA's of organic products where soybeans are part of the system.

2.2. Functional unit

The functional unit is 'one ton of organic soybean produced in China and delivered to Aarhus harbour in Denmark' for the environmental hotspot analysis. For the comparison of organic and conventional soybean, the functional unit is 'one ton of soybean with a protein content of min. 36% produced in the case area in China leaving the farm gate'.

2.3. System boundaries and delimitations

The main system studied was the production and transport of the organic soybeans cultivated in China and exported to Denmark. The main stages included in the cradle to gate hotspot analysis were 1) Production of agricultural inputs, 2) Farm stage, 3) Processing stage (drying, sorting and packaging seeds) and an overall 4) transport stage (including transport steps in/between every stage), as illustrated in Fig. 1.

In addition to the organic soybean system, a comparable conventional production of soybeans in China has been studied. This will be used for the cradle to farm-gate comparison of organic and conventional soybean production. This system only consists of two stages; 1) Production of agricultural inputs (including transport to the farm) and 2) Farm stage for the conventional and organic soybean.

3. Materials and methods

3.1. Methods

A life cycle assessment (LCA) approach has been used in the study. The impact categories included in the study were global warming, eutrophication and acidification, which is relevant to Chinese conditions (Yang and Nielsen, 2001) and will have an effect both globally and locally. In addition, results on non-renewable energy use and land use will be presented. Impact categories concerning toxic aspects were not included due to methodological limitations. The EDIP97 method (Wenzel et al., 1997) (updated version 2.3) has been used for the impact assessment (LCIA) by using the PC-tool SimaPro 7.1.8 (Pré, 2009). The EDIP97 method was updated according to the IPPC 2007 standards for greenhouse gasses (IPCC, 2007). Only for the non-renewable energy use, the IMPACT 2002+ method has been used. EDIP97, IPCC2007 and IMPACT 2002+ are all characterisation methods that convert and aggregate the results from the inventory analysis (data collection and estimation of emissions) into the chosen impact categories.

3.2. Selection of the case study

The case study represents an example of one of the main organic product chains from China to Denmark based on a) Danish national statistics on the main organic agricultural imports from China, b) contacts provided by one of the largest Danish companies in Denmark importing organic soybeans from China (primarily for cattle fodder) and c) information from one of the large organic certification bodies in China.

The case study was based on the production and export of organic soybeans related to a Chinese organic company who had 77% of their production focused on soybeans (15,900 tons in 2005).

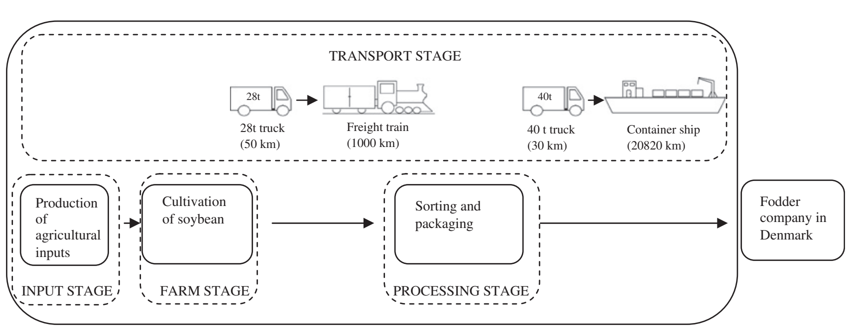

Fig. 1. Life cycle stages included in the hotspot LCA study of organic soybeans cultivated in the Jilin province in China and exported to Denmark.

The soybeans from the company were in 2005 exported primarily to USA (35%), Belgium (37%) and other countries (28%) incl. Denmark. The production of soybeans took place mainly in the Jilin province (30%) and in Heilongiang (70%) on the basis of contracts with the farmers. The farms at the production site in the Jilin Province in DaShan Country were chosen as the case study area. It was not possible to get access to the farms at the production site in the Heilongiang Province. The main difference between the two production sites was that more machinery in general was used in the Heilongiang compared to the Jilin production site e.g. for harvesting. The organic farms at the production site in the Jilin Province were selected randomly following the provision of a farmers list from the company and the certifiers. Following this, a sample of conventional farmers producing soybeans in close geographic proximity to the organic farmers was chosen randomly as a reference.

3.3. Soybean production

Data for the soybean farm stage, such as farming practices, agricultural inputs and yields have been collected by questionnaires and interviews at 20 organic and 15 conventional farms in the case area in the Jilin province in China in the growing season 2006 (Table 1).

Table 1 shows that both the organic and conventional case study farms producing soybeans have a very high share of soybeans on their land (80 and 83%, respectively), indicating that soybeans are an important crop in the area. The remaining land is mainly cultivated with maize, meaning that soybeans are cultivated in a crop rotation with maize as far as it is possible. A large share of the conventional farmers burn part of the crop residues in the field. On the contrary, organic farmers mainly either leave the crop residues in the field or harvest and remove it from the field and mix it with manure and forest soil (2:6:2) and use it for compost (Tables 1 and 2). The organic farms have larger farm areas and more livestock per hectare (mainly cattle and a few chickens) than the conventional farms, which generally have no cattle and only a few fattening pigs or chicken (Table 1).

Organic and conventional farms had the same level of nitrogen (N) input, although they used different N sources with different energy requirements and N-availability for the crops (Table 2). The diesel consumption was 10% larger for organic compared to conventional farms, and also the need for labour was higher on organic farms mainly due to time spend on weeding (Table 2). Furthermore, the yields were 10% lower on organic compared to conventional farms. Since data from the questionnaire showed that there was no difference in the amount of machinery and seeds between the organic and conventional systems, the production phase of seeds and machinery was not included in the study, as it was regarded to have a minor impact on the results (Macedo et al., 2008; Pelletier et al., 2008). However, the use phase of the agricultural and processing machinery was included in the study. Data on the agro-chemical production (e.g. fertilizers) was obtained from the Ecoinvent Database v2.0. Country specific data was not available, so European data were used. The standard values used for soybeans, compost and mineral fertilizer are shown in Table 3.

Table 1
Characteristics of the case study farms producing soybean in the Jilin Province, China (2006).

	Organic	Conventional
Number of studied farms	20	15
Main crops	Soybean and maize	Soybean and maize
Farm area (ha)	16.3 ± 6.6	5.8 ± 2.5
Soybean area (ha)	13.0 ± 6.6	4.8 ± 2.4
Share of crop residues burned in the field (%)	0	41
Animals (LU[a]/ha)	0.5 ± 0.3	0.1 ± 0.1

[a] Livestock units (LU), Definition: 1 LU = 0.5 cattle = 4 pigs = 100 chicken/broilers (FAO, 2003 for Asia).

Table 2
Resource use for 1ha of soybean production in the Jilin Province, China (2006).

	Organic	Conventional
Input		
Mineral fertilizer (urea and $(NH_4)_2PO_4$), N (kg/ha)	–	47 ± 12
Mineral fertilizer (urea and $(NH_4)_2PO_4$), P (kg/ha)	–	14 ± 4
Organic fertilizer (compost[a]) (m^3/ha)	13 ± 4	–
Organic fertilizer (compost[a]), N (kg N/ha)	45 ± 13	–
Organic fertilizer (compost[a]), P (kg P/ha)	8 ± 2	–
Seeds (kg/ha)	55 + 4	57 ± 3
Diesel (L/ha)	30 ± 15	28 ± 14
Labour (days/ha)	52 ± 17	17 ± 4
Percent of labour days spent on weeding (%)	50 ± 10	12 ± 4
Output		
Soybean yield (kg/ha)	2788 ± 306	3083 ± 310
Crop residues[b]		
Left in field (%)	23	13
Burned (in field) (%)	–	41
Burned (in kitchen) (%)	40	41
Removed for compost (%)	33	–
Removed for fodder (%)	4	5

[a] The compost consists of cattle manure (60%), forest soil (20%) and soy/maize crop residues (20%).
[b] % represents the mass of the crop residues.

3.4. Processing

After harvest the soybeans are sorted and packed on a simple factory, where 29 kWh of electricity and 2 l of diesel are used per ton soybean. A waste of 5% occurs during this process. Data for the processing stage have been obtained by questionnaires and interviews with the manager and staff at the company processing and exporting the soybeans. For the electricity, country specific data from China from the Ecoinvent Database v.2.0 was used.

3.5. Transport

The organic soybeans sold for the export market to Denmark are transported from the production site in the Jilin Province by truck and train to the processing factory in Dalian. From the factory they are transported by truck to the harbour of Dalian, China and loaded to a container ship and shipped directly to the harbour of Aarhus, Denmark. Distances and transport forms are shown in Fig. 1.

The conventional soybeans are primarily sold for the national Chinese market. Distances and means of transportation have been obtained from the farmers, the exporting company, the buying company in Denmark and relevant websites.[1] The inventory data on the transport means has been obtained from the Ecoinvent Database v2.0 (Ecoinvent Centre, 2009). For the transport by container ship, an average load of 65% has been assumed for the direct shipping from the harbour of Dalian to the harbour of Aarhus. For rail transport, country specific data from China from the Ecoinvent Database v.2.0 was used. However, for the truck transport country specific data was not available and European data was used in stead.

[1] www.searates.com, www.maps.google.com, www.metric-conversions.org, www.distances.com.

Table 3
Standard values used for the calculation of nitrogen budgets and emissions of the soybean production in the Jilin Province China.

	Values		References
	Mean	Range	
Soybean			
N from fixation (% Ndfa)	52		(Salvagiotti et al., 2008)
N content in seed DM (% N)	6.8		(USDA, 2009)
P content in seed DM (% P)	0.7		(USDA, 2009)
K content in seed DM (% K)	1.8		(USDA, 2009)
N in crop residue (% N)	0.8		(USDA, 2009; IPCC, 2006)
P in crop residue (% P)	0.06		(USDA, 2009)
Compost (cattle manure, forest soil, crop residues (6:2:2))			
N content of FW[a] (% N)	0.6	0.4–1.0	(Lampkin, 2003; Stamatiadis et al., 1999; Hao et al., 2004; British Columbia, 1996; Eghball et al., 1997)
P content of FW[a] (% P)	0.1		(Lampkin, 2003)
K content of FW[a] (%K)	0.7		(Lampkin, 2003)
Bulk density of fresh compost (kg/m^3)	600	430–900	(Schaub-Szabo and Leonard, 1999; Agnew and Leonard, 2003)
N availability for crops (%) (when used a fertilizer)	21	5–21	(Muñoz et al., 2008)
Mineral fertilizer			
Urea, N content (%)	46		(FAO, 2009)
Diammoniumphosphate, N content (%)	18		(FAO, 2009)
Diammoniumphosphate, P content (%)	20		(FAO, 2009)

[a] Fresh weight.

3.6. Calculation of nutrient budgets and emissions

Field level nitrogen (N) and phosphorus (P) budgets including inputs and harvested outputs per ha per year were established in order to assess the balance for potential N-leaching. The partial field nutrient budgets were the outcome of a simple accounting process, which details the inputs (mineral fertilizers, organic inputs (e.g. compost), biological nitrogen fixation, deposition and seeds) and the harvested outputs (crop sales and crop residue removal) from the field during a year (Watson et al., 2002). The standard values used for the calculations (e.g. for nitrogen fixation) are shown in Table 3.

Subsequently, the emissions related to the soybean production were estimated using the IPCC 2006 guidelines (IPCC, 2006) for the direct and indirect N_2O emissions and for the CO_2 emissions related to urea application. The NH_3 emissions were estimated to be 3% of fertilizer-N lost during fertilizer application (Andersen et al., 2001) and 2 kg NH_3 per ha lost during crop growth (Gyldenkaerne and Albrektsen, 2008; Sommer et al., 2004). The amount of crop residues and emissions caused by burning crop residues in the conventional soybean field were calculated according to (IPCC, 2006), except from SO_2 and NMVOC emissions that were calculated according to (Reddy and Venkataramen, 2002) and (US EPA, 1992), respectively. The sequestration of CO_2 in plant production was not included in the study, since the CO_2 will be emitted again during the use phase of the product. However, for the crop residues that were burned, the CO_2 input during growth was included in the calculation and withdrawn from the emissions from burning the crop residues.

For the sensitivity analysis, changes in the soil organic carbon (C) were estimated using the simple tier 1 methodology in the IPCC 2006 guidelines (IPCC, 2006). This IPCC estimation method covers a period of 20 years, whereafter the soil is assumed to have a new 'steady-state' C content. Since a 100 years perspective was used in the GWP calculation, the estimated soil C changes were also made comparable with the into a 100 years perspective using the above-mentioned simple assumption that the main soil C changes takes place within the first 20 years (Foereid and Høgh-Jensen, 2004). The changes in soil C within the first 20 years were therefore divided with 100 years. For the calculation of the soil C changes it was assumed the soil is a long-term cultivated Mollisol with annual crops in a temperate, dry climate region under full tillage. The organic soybean soils were assumed to have a medium input of organic matter, whereas for the conventional soybean soils a low input of organic matter was assumed, due to burning of crop residues in the field. Since this method is simple and can be questioned, it was only used in the sensitivity analysis.

3.7. Handling of manure in the LCA

The compost used for the organic soybeans contains manure, which is a co-product from the livestock production. However, it is problematic to estimate which part of the environmental costs of producing the manure should be attributed to the soybean production. Several suggestions was proposed for this allocation situation in the EU report on harmonisation of environmental LCA for agriculture, but no consensus was reached on how to solve it (Audsley et al., 1997; van Zeijts et al., 1999). Estimating the environmental costs can be done in at least two ways:

A. One could argue that the manure is a waste product from the livestock production and therefore all environmental costs from this should be allocated to the meat.
B. However, some would argue that since manure has a value as a fertilizer – it would be biased in a comparison of organic and conventional farming to regard manure as having no environmental costs in its production. The fertilization value of the manure was taken into account using a consequential LCA approach, described by Dalgaard and Halberg (2007). In this approach the environmental costs of producing plant available manure-N corresponds to the environmental costs of producing mineral fertilizer-N. Thus, the environmental costs of producing mineral fertilizer become some kind of rate of exchange when using manure in a system. The underlying assumption is that the manure could have replaced mineral fertilizer in another conventional field. This approach will be used in the following. The corresponding amount of mineral fertilizer that should have been produced is calculated by the total N contents in the manure/compost multiplied by the percentage that will be available for crops. For compost, the N availability for crops is listed in Table 3.

The effect of using the approach described under A. where manure is regarded as a waste product is shown in the later sensitivity analysis.

Table 4
Partial N and P budgets[a] at field level (kg N/ha) including inputs and harvested outputs from 1ha of soybean production in the Jilin Province, China (2006).

	Organic		Conventional	
	Nitrogen (N)	Phosphorus (P)	Nitrogen (N)	Phosphorus (P)
Input				
Mineral fertilizer			47	14
Organic fertilizer	45	8	–	–
Fixation	115	–	126	–
Deposition[b]	16	–	16	–
Seeds	4	0.4	4	0.4
Total input	180	8	193	14
Output				
Soybean yield	189	19	209	21
Crop residues removed (for fodder, compost or burned)	24	2	29	2
Total output	213	21	238	23
Partial field balance	−33	−13	−45	−9

[a] The partial field budgets does not include any emissions to the air, water or changes in the soil storage. Standard values used for the calculations are shown in Table 3.
[b] Ying et al. (2006).

Table 6
Characterized results at farm gate for 1 ton of organic and conventional soybean produced in the Jilin Province in China (2006).

	Input stage		Farm stage		Total
	Agro-chemicals	Crop production	Traction	Crop residue burning	
Land use (ha farmland/t soybeans)					
Organic	–	0.36	–	–	**0.36**
Conventional	–	0.32	–	–	**0.32**
Non-renewable energy use (MJ/t soybeans)					
Organic	235	–	538	–	**773**
Conventional	1250	–	460	–	**1710**
Global warming potential, GWP (kg CO_2 equiv./t soybeans)					
Organic	12	103	41	–	**156**
Conventional	66	110	35	52	**263**
Acidification potential (kg SO_2 equiv./t soybeans)					
Organic	0.1	1.9	0.4		**2.3**
Conventional	0.6	2.5	0.3	1.1	**4.5**
Eutrophication potential (kg NO_3-equiv./t soybeans)					
Organic	0.8	3.6	0.6	–	**5.0**
Conventional	6.1	4.8	0.5	1.6	**13.0**

4. Results

4.1. Comparison of organic and conventional soybeans at farm gate

In order to evaluate the losses from the production system, a field budget of nitrogen (N) and phosphorus (P) for the organic and conventional soybeans was made (Table 4).

Since the field N and P balances are negative for both the organic and conventional soybean production (Table 4) it is assumed that the risk of leaching of N and P from the organic and conventional systems is negligible. Estimated emissions to the air at the farm stage are presented in Table 5.

It is apparent from this table that N_2O emissions are comparable for the organic and conventional soybeans and that conventional soybeans have further emissions due to application of urea and burning of crop residues.

The characterized LCA results for organic and conventional soybeans at farm gate are presented in Table 6. This table shows that the land use requirements at farm gate for organic soybean production are 12% higher than for conventional production (Table 6). However, for the non-renewable energy use and the global warming, acidification and eutrophication potential, the organic soybean production has a lower impact per ton soybeans at farm gate (Table 6). The difference is mainly due to the production of agrochemicals and the burning of crop residues in the conventional soybean production system. An environmental impact for production of fertilizer has been added to the organic soybeans, since scenario B (where fertilization value of manure is included) is used for the calculations. However, since the corresponding amount of mineral fertilizer only reflects the N in the compost that are available to crops, the energy requirement for the organic input stage is five times lower than the conventional one.

For organic soybeans, the non-renewable energy use is mainly for traction at the farm (70%), whereas the production of agrochemicals is the main contribution (73%) for conventional soybeans (Table 6). The contribution to the global warming potential (GWP) at farm gate comes mainly from N_2O emissions (66% for organic and 42% for conventional soybean) during the soybean production (Table 6). Likewise, the main contribution to the acidification potential of soybeans at farm gate was during the crop production (Table 6). Almost half of the contribution to the eutrophication potential to conventional soybeans came from the fertilizer production since the risk of nutrient leaching from the soybean production systems were negligible.

Table 5
Emissions to the air from 1ha of soybean production at the farm stage in the Jilin Province, China (2006) calculated using mainly the IPCC 2006 guidelines (IPCC, 2006).

	Organic	Conventional
Nitrogen dioxide (kg N_2O/ha)[a]	1.0	1.1
Ammonia (kg NH_3/ha)[b]	2.8	4.1
Carbon dioxide from urea application (kg CO_2/ha)[a]	–	55.0
Carbon dioxide from crop residue burning, output-input (kg CO_2/ha)[a, c]	–	−212.1
Methane (kg CH_4/ha)[a, c]	–	3.8
Carbon monoxide (kg CO/ha)[a, c, d]	–	128.4
Nitrogen oxides (kg NOx/ha)[a, c]	–	3.5
Sulphur dioxide (kg SO_2/ha)[c, e]	–	0.8
Non-methane volatile organic compounds (kg NMVOC/ha)[c, f]	–	15.7

[a] According to IPCC 2006 guidelines (IPCC, 2006).
[b] According Andersen et al. (2001), Gyldenkaerne et al. (Gyldenkaerne and Albrektsen, 2008) and Sommer et al. (2004).
[c] Only from crop residues burned in the field.
[d] CO not regarded as a direct greenhouse gas, but it is assumed that it is converted to CO_2 with time.
[e] According to Reddy and Venkataramen (2002).
[f] According to US EPA (1992).

4.2. Environmental hotpots of imported organic soybeans to Denmark

As illustrated in Fig. 2, the organic soybeans produced in China and transported to Denmark had a total non-renewable energy use of 4377 MJ per ton soybeans, with the major contribution coming from the transport stage (71%). The transport by ship from China to Denmark was the major contributor with 2570 MJ per ton soybeans in the transport stage, whereas the transport by truck and train only had a minor contribution (250 and 300 MJ per ton soybeans).

The total global warming potential (GWP) of 1 ton organic Chinese soybeans delivered at the harbour in Aarhus was 429 kg CO_2 eq. per ton soybeans (Fig. 2). Interestingly, the main contribution to the GWP also came from the transport stage (51%) and secondly from the farm stage (35%) (Fig. 2). Within the transport

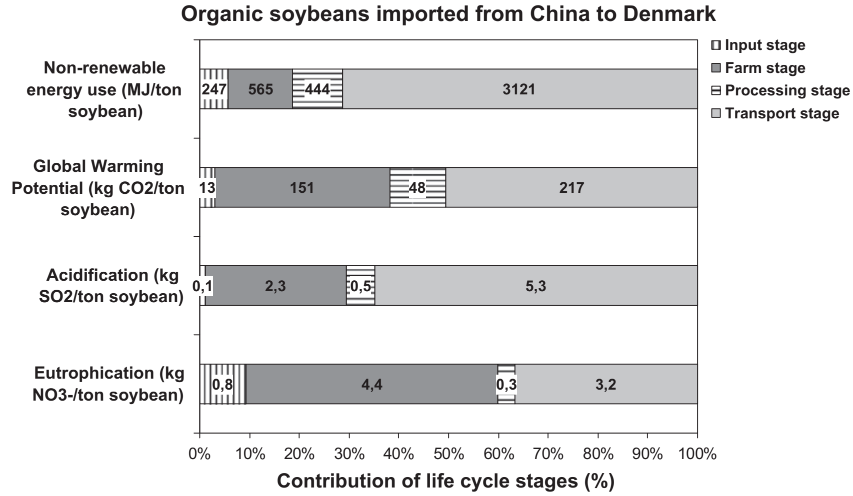

Fig. 2. Hotspot analysis and characterized results for organic soybeans produced in the Jilin Province, China and transported to the harbour of Aarhus, Denmark (2006).

stage, the freighter oceanic from China to Denmark was the main contributor with 187 kg CO_2 eq. per ton soybeans, whereas the rail only contributed 15 kg CO_2 eq. per ton soybeans and the trucks 16 kg CO_2 eq. per ton soybeans. With regard to GWP, N_2O is the main contributor at the farm stage, whereas CO_2 is the main contributor at the input production, processing and transport stage.

As shown in Fig. 2, the transport stage was also the main contributor (65%) to the acidification potential for the organic soybeans transported to Denmark having a total of 8.1 kg SO_2 eq. per ton soybeans. Also here the freighter oceanic from China to Denmark was the main contributor to the transport stage with 5.1 kg SO_2 eq. per ton soybeans.

However, with regard to the eutrophication potential for the organic soybeans transported to Denmark, the farm stage was the main contributor (50%) followed by the transport stage (37%) (Fig. 2). The total eutrophication potential for organic soybeans transported to Denmark was 8.8 kg NO_3^- eq. per ton organic soybeans.

4.3. *Sensitivity analysis at farm gate*

A sensitivity analysis focusing on GWP at farm gate was made to test the uncertainty related to the choice of models and assumptions (Table 7). Four main assumptions were included in the sensitivity analysis, which are considered to be the most important assumptions affecting the results.

Firstly, the sensitivity of the N_2O calculation method was tested. Using the IPCC 2000 guidelines (IPCC, 2000) instead of the IPCC 2006 guidelines (used in the reference scenario) showed that the GWP becomes approximately three times higher per ton soybeans (Table 7). The difference is caused by higher estimates of N_2O (4.40 and 4.63 kg N_2O per hectare for organic and conventional soybeans) when using the IPCC 2000 guidelines (IPCC, 2000).

Secondly, the effect of varying two central data estimates was tested. The N content of compost and fertilizer is central to the calculations of N_2O, which in turn is the main contributor to especially the GWP. However, since the N content in compost can vary (contrary to N content in mineral fertilizer) the effect of an increase in the N content in compost has been tested. Furthermore, the effect of increasing the estimate for N availability in compost for crops, which is used for calculating the corresponding amount of mineral fertilizer that should have been produced instead of the N in compost, have been tested. By assuming a 50% higher N in compost or a 45% higher availability of N in compost, the GWP at farm gate is slightly increased for the organic soybeans, but not to the level of the conventional soybeans (Table 7).

Thirdly, the main assumptions in system boundaries or allocation procedures were tested, which is mainly handling of manure and inclusion of carbon release or sequestration in the soil. Interestingly, the GWP for organic soybeans is only slightly decreased when testing the effect of scenario A, in handling the manure (where manure is regarded as a waste product with no economic value) (Table 7). The inclusion of soil carbon change in the calculations revealed a slight increase in the difference in GWP between the organic and the conventional soybeans (Table 7). For those calculations, it was estimated that the production of organic soybeans caused a decrease of 100 kg soil C per ha per year in a 100 years perspective, due to a medium input of organic matter, by using IPCC 2006 guidelines (IPCC, 2006). Likewise, it was estimated that the production of conventional soybeans caused a decrease of 120 kg soil C per ha per year in a 100 years perspective, due to burning of crop residues (low input of organic matter) (IPCC, 2006).

Table 7
Global warming potential, GWP (kgCO_2 eq./ton soybean) at farm gate for soybeans produced in the Jilin Province, China as affected by changes in certain assumptions in a sensitivity analysis.

	Organic	Conventional
Reference scenario[a]	**156**	**263**
Changes in assumptions:		
1) N_2O calculation methods		
Using IPCC 2000 guidelines[b] for N_2O calculations	523	609
2) Central data estimates		
50% higher N in compost (affects only organic)	198	263
45% N availability in compost for crops (affects only organic)	169	263
3) System boundaries or allocation procedures		
Manure regarded as waste (affects only organic)	144	263
Soil carbon change included (rough IPCC 2006 estimate)[c]	287	405
4) Production conditions		
No crop residue burning in conventional	156	232

[a] In the reference scenario, calculations of N_2O are based on IPCC 2006 guidelines and the fertilization value of manure/compost is included in the system in the form of mineral fertilizer in the amount that corresponds to total N in compost times the N availability in compost for crops (here: 21%). Furthermore, estimates on carbon sequestration are not included and 42% of crop residues are burned in the conventional soybean production.
[b] IPCC (2000).
[c] IPCC, 2006.

Fourthly, testing the effect of a scenario where the 41% of the crop residues are left in the field instead of being burned, results in a decrease in the GWP for conventional soybeans, but not to the level of the organic soybeans (Table 7).

5. Discussion

5.1. Methodology and sensitivity

N_2O emissions during cultivation of soybeans and emissions during the transport by ship to Denmark are the two largest contributions to the GWP for organic soybeans produced in China and transported to Denmark. Thus, the estimated emissions from these contributions have a large effect on the result.

With regard to the N_2O calculations, the IPCC 2000 guidelines resulted in much higher N_2O estimates than when using the IPCC 2006 guidelines (Table 7). This can be explained by three main issues that are different in the IPCC 2000 guidelines compared to the IPCC 2006 guidelines: 1) N inputs from fixation are included in the N_2O calculation, 2) the emission factor for N_2O of added N are 0.0125 in stead of 0.01 and 3) the N content in crop residues are estimated very high compared to the IPCC 2006 guidelines. Those factors give in total higher N_2O estimations. An evaluation of which is a better estimate for N_2O emissions is assisted by the recent results from Xiong et al. (2002) who found emissions of 0.93 and 1.27 kg N_2O per ha per year for a winter pea-summer soybean system without and with N fertilizer in upland cropping systems in China. This level is also reported by Bremner et al. (1980) cf. Eichner (1990) who found emissions ranging from 0.34 to 1.97 kg N_2O–N per ha per year with soybean grown in six different soil types. Lu et al. (2006) and Bouwman et al. (2002) found a strong correlation between N application and N_2O emissions (which is also the basis for the IPCC guidelines) and in the literature overview of Lu et al. (2006) the application of approx. 50 kg N didn't give rise to more than 1 kg N_2O–N. Those results correlate well with the results from using the IPCC 2006 guidelines and suggest that the IPCC 2000 guidelines overestimate the N_2O emissions.

Testing the central data estimates of a higher N content in the compost and a 45% N availability in compost (which is unrealistically high according to Muñoz et al., 2008) did not markedly change the GWP of organic soybeans compared to the level of conventional soybeans.

Regarding the scenarios of handling the manure/compost, it could be argued that scenario A (where manure is regarded as a waste product and all the environmental costs of the manure should be allocated to the meat) is the most reasonable one for this case. However, scenario B (where the fertilization value of manure is included) was used in the reference scenario to show the highest possible GWP of the organic soybeans and despite of this, it is still lower than the conventional one. The difference between the GWP of scenarios A and B is negligible and scenario A only gives a slight decrease.

Interestingly, the sensitivity analysis shows an effect of including soil C changes in the calculations, increasing the difference slightly between the GWP of organic and conventional soybean at farm gate (Table 7). However, the methodology used for calculating the soil C changes in this paper are very rough (IPCC, 2006). Thus, the results from the sensitivity analysis should be interpreted with caution. The long-term losses of soil organic carbon from arable land in Northwest China have been reported and are considered a problem for both soil productivity and contribution to GWP (Zhang et al., 2006; Tang et al., 2006). The results indicate that there is a need to develop the methodology for producing and including reliable estimates of soil C changes into future LCA studies.

With regard to the practice of conventional farmers to burn part of the crop residues, the sensitivity analysis (Table 7) shows that a ban of burning crop residues in the field would reduce the GWP of the conventional soybeans at farm gate by 12%. This would decrease the difference between the GWP of organic and conventional soybeans (especially if changes in soil C was included in the calculations), but there would still be a difference.

With regard to the data used for the LCA, it should be noted that the use of European in stead of Chinese data for the production of agrochemicals and truck transport could have a minor increasing effect on the results.

The use and effect of pesticides in the conventional as opposed to the organic soybean production was not included in this study, due to methodological limitations. Pelletier et al. (2008) showed that the production phase of pesticides only had a minor contribution to the GHG emissions. However, this does not mean that the environmental effects of pesticide use are not considered as important. Pesticides pose a risk for human health mainly by contamination of drinking water, pesticide residues in food or farmers' inhaling or exposure on skin (Pimentel, 2005). Furthermore, the ecotoxicity of a pesticide can directly and indirectly affect biodiversity (including natural enemies of pests, soil microflora, earthworms and pollinating bees) negatively (Pimentel, 2005). The ban of pesticides in organic agriculture only underlines the difference in the environmental profile of organic and conventional soybeans. The main pesticides used in the conventional soybean production were the herbicides acetochlor and fomesafen and the insecticide acetamiprid. Acetamiprid has a low acute and chronic toxicity in mammals and is moderately toxic to bees. Fomesafen is widely used for weed control in soybeans in China. It is less toxic to mammals, but highly toxic to aquatic organisms. Acetochlor is also widely applied as a herbicide in the agricultural production in northeast China. In the US, acetochlor is the third most frequently detected herbicide in natural waters. It is classified as a probable human carcinogen and is highly toxic to aquatic organisms.

Biodiversity was not included as an impact category in the study either, due to methodological limitations. Several studies have shown that biodiversity tends to be higher on organic compared to conventional farms, mainly due to the lack of pesticides and more diverse farming systems (Bengtsson et al., 2005; Hole et al., 2005). However, apart from the effect of pesticides on biodiversity in the system, no difference was seen in the organic and conventional system with regard to crop diversity, which could have an effect on the associated biodiversity in the cropping system (Altieri, 1999).

5.2. Comparison with similar studies

The lower environmental impact of organic compared to conventional soybean is consistent with the few other studies that compare organic and conventional soybean production (Pelletier et al., 2008; Jungbluth and Frischknecht, 2007).

The level of GWP was in agreement with Pelletier et al. (2008) who found very similar results (190 and 248 g CO_2 equivalents per kg organic and conventional soybean at farm gate, respectively), using IPCC 2006 guidelines for the N_2O estimation. This was supported by Lehuger et al. (2009), who found a comparable GWP of conventional soybean produced in Brazil at farm gate. Surprisingly, Jungbluth and Frischknecht (2007) found 6–8 times higher GWPs at farm gate in organic and integrated soybean produced in Switzerland. This might partly be explained by the fact that N_2O emissions were calculated on the basis of the IPCC guidelines from 1996 (Nemecek and Kägi, 2007). Contrary to the IPCC 2006 guidelines, both the IPCC 1996 and 2000 guidelines includes the N input from N_2 fixation and use an emission factor of 0.0125 in stead of 0.01 for N_2O of added N in the N_2O calculations (IPCC, 2006, 2000, 1996). Likewise using the IPCC 2000 guidelines for the N_2O calculations, Dalgaard et al. (2007) also found a higher level of GWP (642 kg CO_2 equivalents per t soybean) when analysing conventional soybeans produced in Argentina. When using IPCC 2000 guidelines for the present study of conventional soybeans from the Jilin province, China, the sensitivity analysis shows very similar results (644 g CO_2 equivalents per kg soybean).

Concerning the contribution of transport, Dalgaard et al. (2007) have shown results that are comparable to this study – when taking the longer distance by ship from China to Denmark into account (compared to the distance from Argentina to Rotterdam). Surprisingly, Lehuger et al. (2009) suggested very low contributions to GWP from transport of soybean meal (that corresponds to 80% of the mass of soybeans) by ship from Brazil to France.

5.3. *Outlook on the choice of protein fodder crops for Danish organic livestock*

The most likely alternative products for the organic livestock in Denmark evaluated in 2009 are organic soybeans from Italy (pers. comm. Henrik Kløve, DLG, 2009). The potential of using domestically produced protein feeds like organic rapeseed cake, faba bean or field pea in stead of Chinese soybeans are mainly limited by a higher price (pers. comm. Henrik Kløve, DLG, 2009). Transporting soybeans from China or Italy makes a difference in the distance travelled. Soybeans from China would travel 20,820 km by container ship (Fig. 1), whereas soybeans from Italy (Rome to Aarhus) would travel 1587 km by either truck or train. However, the GWP from the journey from China to Denmark by container ship would emit 187 kg CO_2 eq./t soybeans (9 g CO_2 eq./tkm, Ecoinvent Centre, 2009), whereas the travel from Italy to Denmark by truck in a 40 t truck would amount to 238 kg CO_2 eq./t soybeans (150 g CO_2 eq./tkm, Ecoinvent Centre, 2009). Contrary, would the travel from Italy by combined freight train (1529 km at 40 g CO_2 eq./tkm, Ecoinvent Centre (2009) and 28 t truck (69 km at 227 g CO_2 eq./tkm, Ecoinvent Centre, 2009) amount to 76 kg CO_2 eq/t soybeans. A local production of protein for the Danish organic livetstock would reduce the environmental costs for transportation, but one would have to be aware that the environmental costs at the farm stage do not exceed the environmental costs of the imported soybeans.

5.4. *Recommendations*

If the soybeans are to be imported from China, using train and container ship has reduced the environmental impact to a minimum. Mechanization at the farm stage was at a minimum in the present study. The environmental profile of the soybean would be improved if the use of N fertilizer was limited to a minimum, especially for the conventional soybeans. Although emissions from the manure management are not included in this study, optimal manure management such as covering the manure storage and assuring sufficient aeration are important to reduce both the loss of nutrients and methane emissions. A stop for burning crop residues in the field would improve the environmental profile of the conventional soybeans. Leaving crop residues in the field would build both soil carbon and fertility. Furthermore, an increased use of perennial crops in stead of only annual crops would build carbon into the system and increase diversity in the cropping systems.

6. Conclusions

The organic soybeans has a lower environmental impact, with regard to non-renewable energy use, global warming, acidification and eutrophication potential per ton produced compared to the conventional soybeans. The transport stage has a major impact on the environmental profile of the imported organic soybeans to Denmark. For the GWP of organic imported soybean to Denmark, 51% came from the transport stage (especially the transport by ship) and 35% from the farm stage. The sensitivity analysis showed that the estimation of N_2O had a major impact on the results.

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Paper III

Environmental assessment of organic juice imported to Denmark: a case study on oranges (*Citrus sinensis*) from Brazil

Knudsen MT, de Almeida G, Halberg N, Langer V, de Abreu LS (submitted)
Organic Agriculture

Environmental assessment of organic juice imported to Denmark: a case study on oranges (*Citrus sinensis*) from Brazil

Marie Trydeman Knudsen[1*], Gustavo Fonseca de Almeida[2], Vibeke Langer[1], Lucimar Santiago de Abreu[3] and Niels Halberg[4]

[1]Dept. of Agriculture and Ecology, Faculty of Life Sciences, University of Copenhagen, DK-Taastrup, Denmark, [2] Centre of Agricultural Sciences, Federal University of São Carlos, SP-330 Araras, São Paulo, Brazil, [3] Embrapa Environment, P.O. Box 69, Jaguariúna, São Paulo, Brazil, [4] International Centre for Research in Organic Food Systems (ICROFS), DK-8830 Tjele, Denmark.

* Corresponding author. Present address: Faculty of Agricultural Science, Aarhus University, Dept. of Agroecology and Environment, Blichers Alle 20, P.O. BOX 50, DK-8830 Tjele, Denmark. Tel.: +45 8999 1308; Fax: +45 8999 1200; E-mail: MarieT.Knudsen@agrsci.dk.

A B S T R A C T

Growing global trade with organic products has given rise to a debate on the environmental impacts during both production and transport. Environmental hotspots of organic orange juice produced by smallholders in Brazil, processed and imported to Denmark were identified in a case study using a life cycle approach. Furthermore, small-scale organic orange production was compared with small-scale conventional and large-scale organic orange production in the case study area in Brazil.

Transport was the main contributor (57%) to the global warming potential of organic orange juice from small-scale farmers imported to Denmark, followed by the processing stage (29%), especially the truck transport of fresh oranges in Brazil and of reconstituted orange juice in Europe. Non-renewable energy use per hectare was significantly lower on the organic small-scale farms than on the conventional, with a similar, although not significantly lower, pattern for global warming potential and eutrophication. Including soil carbon sequestration in organic plantations widened the difference in global warming potential between organic and conventional. Organic small-scale farms had a higher crop diversity than conventional, which may have a positive effect on biodiversity along with the spontaneous vegetation between the organic orange trees and the absence of toxic pesticides. Comparing small-scale with large-scale organic orange production, crop diversity was higher on the small-scale farms, while global warming potential, eutrophication potential and the use of copper per hectare were significantly lower, indicating that environmental impacts from small-scale differ from large-scale organic farms.

Keywords: conventional; LCA; orange juice; organic; transport

1. Introduction

The consumption of organic food and feed has increased over the last decade, especially in Europe and North America. Increasing demands for organic products offer good prospects for suppliers from other parts of the world. Latin America has 20% of the organically managed farmland in the world and exports most of its products and Brazil has the world's third largest organically certified area (Willer & Kilcher, 2009). The main organic products imported from Brazil to Denmark are orange juice and sugar (pers. comm. Agnete S. Nilsson, StatBank Denmark, 2010). Brazil dominates the market of frozen concentrated orange juice (FCOJ), sitting on more than 80% of total world trade. Most of the Brazilian orange juice originates from the State of São Paulo (Neves, 2008), which is also the production area for most of the organic orange juice sold in Denmark (pers. comm. Carina Jensen, Rynkeby, 2009).

The growing global trade with the geographically and socially widely dispersed sites of organic production and consumption has given rise to questions concerning the carbon footprint of long-distance transport (Soil Association, 2007). While research has shown farm level environmental benefits from organic farming primarily in a European context (Stolze et al., 2000; Hansen et al., 2001), it is questionable whether these benefits also hold true outside Europe and whether they will be offset by longer transport distances. Organic consumers may ask what the environmental benefits of organic production are at farm level compared with the impact of long-distance transport. At the same time, small-scale farmers are challenged when trying to enter the organic markets due to e.g. market demand for large and stable supplies (Kledal, 2009; Blanc, 2009) and there is an ongoing debate on the 'conventionalisation' of organic farming and how this might affect, e.g., environmental sustainability (Darnhofer et al (2010). Consumers may also ask whether the environmental impact of large-scale organic farms differs from that of small-scale organic farms. Thus, the environmental impacts of the imported organic products need to be assessed, both at farm level (including comparison of small- and large-scale organic with conventional) and along the food chain. Life Cycle Assessment (LCA) is a method used to assess several environmental impacts (e.g. global warming, eutrophication etc.) along the life cycle of a product. LCA has become an internationally accepted method also in agriculture for assessing environmental impacts and for identifying hotspots where the environmental burden for a product in a life cycle is particularly large, (Thomasson et al., 2008, Cederberg and Mattsson, 2000, Haas et al , 2001).

Few have studied the environmental impact of orange juice for the whole chain from farmer to consumer. Coltro et al. (2009) made a life cycle inventory of the farm management practices of conventional farms producing oranges for frozen orange concentrate in Brazil, but not a life cycle impact assessment. Beccali et al. (2009) made a life cycle assessment of conventional Italian orange juice concentrate, identifying transport, electricity and the production of agrochemicals as the main hotspots. In two papers on the LCA of organic and integrated orange production in Valencia, Spain, Sanjuan et al. (2005a, 2005b) assessed the environmental impacts of oranges. However, the studies are not directly comparable since slightly different methodologies were applied. Pereira & Ortega (2005) and La Rosa et al. (2008) studied orange production in Brazil and Spain, respectively, using primarily an emergy evaluation. Schlich & Fleissner (2005) and Schlich (2005) in their studies on orange juice from Brazil imported to Germany only reported the direct energy use, and the methodology and conclusions presented were later criticized by Jungbluth & Demmeler (2005). In summary, the few LCA studies found either focused on conventional orange juice or have been restricted to the production at farm gate. Thus, a full LCA of organic orange juice is needed, including the environmental impact of transportation and at farm level the organic versus conventional and small-scale versus large-scale production.

Therefore, our study set out to improve the basis for evaluating the ecological soundness of organic orange juice by assessing the environmental impacts of organic oranges grown and processed to frozen orange concentrate in Brazil, reconstituted in Germany and imported to Denmark, using a life cycle approach. The objectives were to 1) identify the environmental hotspots in the product chain of organic orange juice originating from small-scale farms in Brazil and imported to Denmark and 2) to compare the environmental impacts at the farm gate of the organic orange production with a comparable conventional and a large-scale organic orange production in the same region in São Paulo, Brazil.

Table 1. Characteristics of the case study farms producing oranges for juice in the State of São Paulo, Brazil (2007).

	Organic				Conventional	
	Small-scale farms (<75 ha)		Large-scale farms (>75 ha)		Small-scale farms (<75 ha)	
Number of studied farms	5		2		6	
	Mean	Range	Farm X	Farm Y	Mean	Range
Farm area (ha)	33	7-72	11,494	140	32	9-68
Agricultural area, excl. forest (ha)	29	6-64	8122	126	29	7-68
Orange area[a] (%)	33	14-53	60	100	72	57-90
Animals (LU[b]/ha[a])	0.07	0-0.3	0.3	0	0.6	0-2.9
Main crops	Orange, mango and lime			Orange		Orange
Other crops	Guava, vegetables			No (pasture)		No (pasture)

[a] of agricultural area excl. forest.
[b] Livestock units (LU), Definition: 1 LU = 1.4 cattle = 4 pigs = 100 chicken/broilers (FAO (2003) for South America).

2. Materials and methods

2.1. Selection and description of the case study

The case study on organic orange juice from Brazil was selected as a relevant example of smallholder production entering the global organic market. The specific case selection was based on a) information from the Danish National Statistics and major supermarkets, b) information from colleagues at Embrapa Meio Ambiente, São Paulo, Brazil, c) contacts provided by a Danish merchant importing organic orange juice originating from Brazil.

Given the infancy of the organic orange juice market, the total number of organic farms producing orange for juice was restricted. Based on the information gained, a case study area in the municipality of Itapolis in the State of São Paulo, Brazil, including a cooperative of small-scale farms exporting frozen concentrated orange juice to Europe was chosen for the analysis.

Small organic family farms that produce oranges for juice for export to Denmark or Europe were primarily selected. All organic farms in the cooperative with a productive orange plantation were selected for the analysis. Furthermore, a group of small conventional farms with a productive orange plantation from the same cooperative were chosen randomly for the comparison at farm gate. In addition to the small organic farms, large organic farms were also included for comparison and in order to represent the larger volume of exported organic orange concentrate. The large-scale farms was represented by two (for which it was possible to gain sufficient information) out of in total five large-scale organic farms producing organic oranges for juice in the State of São Paulo. Even though the number of organic large-scale farms is low, it represents a large proportion of the volume produced. Thus, the farms were selected to represent the two main ways of producing organic oranges in the case study area, which is the main orange producing region in Brazil.

For the hotspot analysis, only the organic oranges from small-scale farms were assessed. The factory processing the oranges from the small-scale farms was furthermore identified and assessed (detailed description in Figure 2 and section 2.6).

Table 1 presents the characteristics of the case study farms. The conventional small-scale farms were comparable in size to the small-scale organic farms (Table 1) and their management practices were similar to those of the organic farms prior to conversion to organic farming. However, the small-scale conventional farms had some financial restrictions on their management due to having 5-10-year contracts in US dollars with the juice processing industry, during a period when the value of the US dollar has declined. The large-scale farms were producing both organic and conventional oranges. Farm A had a very large farm area, focusing on oranges and cattle, while farm B only produced oranges.

The main difference between the three types of farms was in crop diversity, where the small organic farms grew mango, lime, guava and vegetables in addition to oranges, whereas the large organic farms and small conventional farms had orange as their dominating crop (Table 1).The small organic farms also allowed spontaneous vegetation to grow between the planted rows as a source of green manure which could, in addition to the absence of herbicides, give rise to both increased diversity and increased carbon sequestration. Generally, the farms had few livestock per hectare and did not use manure from their own animals as fertilizer for the orange plots.

2.2. Life cycle assessment approach

The environmental impact categories included in this study were global warming, eutrophication and acidification, all of which have an effect both globally and locally. In addition, results on non-renewable energy use and land use are presented. A life cycle

Table 2. Impact categories used in this study and the contributions from the main emissions (IPCC, 2007; Wenzel et al., 1997).

Impact category	Unit	Contributing elements	Characterization factors
Land use	m^2	Land occupation	1 for all types of land use
Non-renewable energy	MJ	Non-renewable energy consumption	1
Global warming	CO_2 equivalents	CO_2	1
		CH_4	25
		N_2O	298
Acidification	SO_2 equivalents	SO_2	1
		NH_3	1.88
		NO_x	0.70
Eutrophication	NO_3^- equivalents	NO_3^-	1
		PO_4^{3-}	10.45
		NH_3	3.64
		NO_x	1.35

assessment involves a range of inputs (materials, energy, chemicals and other) and outputs (products, co-products, emissions etc.) at every stage in the chain of the studied product and an estimation of the emissions. The emissions from the life cycle of the product or the production of inputs are then converted to the chosen environmental impact categories. The relation between the main emissions and the conversion into a certain impact category (using the characterization factors) is presented in Table 2. As an example, the impact category 'global warming potential' (GWP) is measured in kg CO_2 equivalents, and since CH_4 also contributes to global warming, it needs to be converted into CO_2 equivalents. The characterisation factor describes the relative strength of CH_4 compared to CO_2 in a 100-year perspective and the amount of CH_4 emissions should thus be multiplied by 25 to get the impact in CO_2 equivalents.

The characterization method EDIP97 (Wenzel et al. (1997) updated version 2.3) was used for the so-called impact assessment (where emissions are converted into a certain environmental impact category) by using the PC-tool SimaPro 7.1.8 (Pré, 2009). The EDIP97 method was updated according to the IPCC 2007 standards for greenhouse gasses (IPCC, 2007). Only for the non-renewable energy use was the characterization method IMPACT 2002+ used. Impact categories concerning toxic aspects were not included due to methodological limitations.

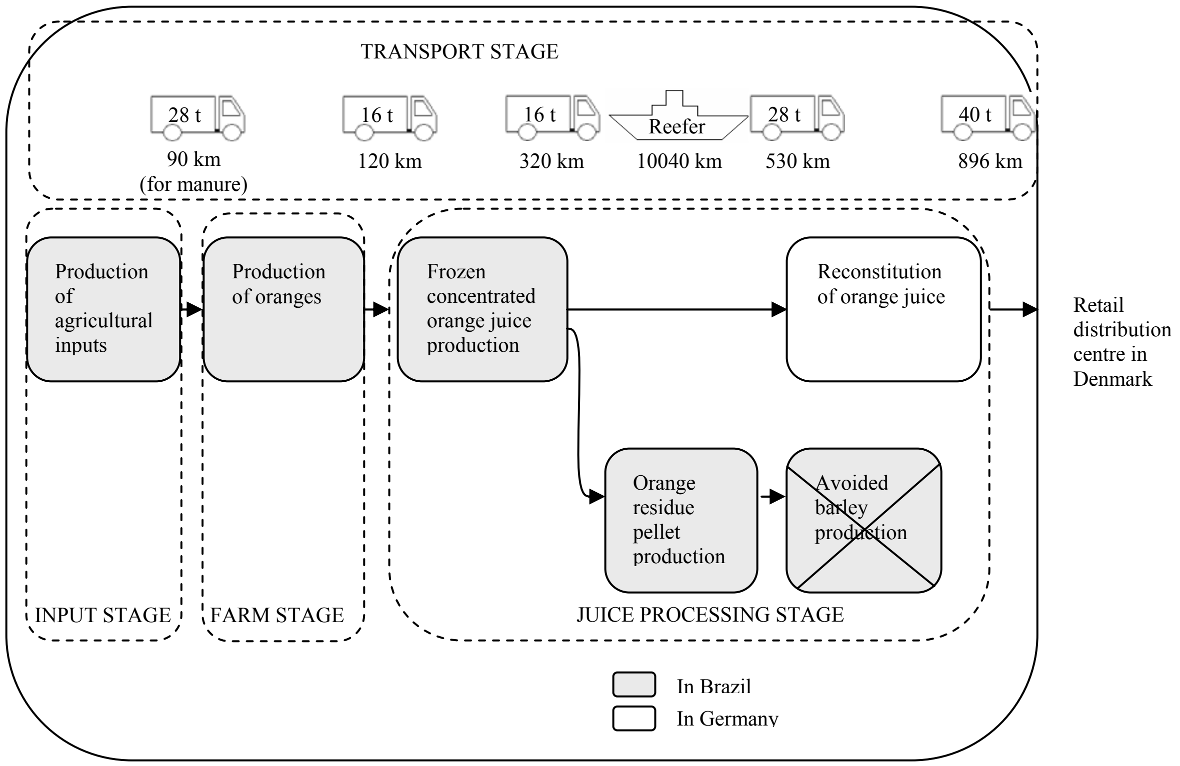

Figure 1. Involved processes, systems boundaries and transportation in the case study on orange production and the following processing of oranges from small-scale producers to orange juice imported to Denmark.

Environmental aspects concerning pesticide use and biodiversity will be described qualitatively. Statistical analyses compared the results from the farm types two by two in a t-test using R software. Small letters denote significant differences between farm types at $p \leq 0.05$.

2.3. Studied products (functional unit)

The environmental impacts are related to a so-called functional unit, which is the product studied. For the environmental hotspot analysis of organic orange juice, the functional unit is 'one litre of organic orange juice grown and processed to frozen concentrated orange juice in Brazil, reconstituted to juice in Germany and imported to Denmark'. In the comparison between organic and conventional orange production, the functional unit is 'one tonne of oranges produced in the state of São Paulo leaving the farm gate'.

2.4. Boundaries for the studied systems

For the hotspot analysis the main system studied was the production of oranges, processing and transport of the organic orange juice originating from Brazil and imported to Denmark. The main stages included in the cradle to gate hotspot analysis were 1) Production of agricultural inputs, 2) Farm stage, 3) Processing stage (production of frozen concentrate in Brazil and reconstitution and packaging in Germany) and an overall 4) transport stage (including transport steps in/between every stage and import to Denmark), as illustrated in Figure 1.

In the cradle to farm-gate comparison between small-scale organic, small-scale conventional and large-scale organic orange production, the system consists of only two stages; 1) Production of agricultural inputs (including transport to the farm) and 2) Farm stage (production of oranges etc.) for the conventional and organic oranges.

2.5. Orange production

Data for the farm stage include farming practices, agricultural inputs, yields etc. and were collected by questionnaires and interviews at five small and two large organic farms plus six small conventional farms in the case area in the State of São Paulo in Brazil in the growing season 2006-7. Since all orange farms consist of both young and productive plantations, the data were based on 4- to 20-year-old productive orange plantations. All farmers grew the cultivars Valencia, Pera Rio and Pera Natal, and in addition Westin and Murcot were used in the organic plantations and Hamlin in the conventional ones. Oranges were harvested in the period from June to November. The organic orange plantations were fertilized mainly with either chicken manure, cattle manure and/or filter cake from the sugar cane industry. The organic farmers typically allowed the interrow vegetation to grow tall; they then applied the manure/organic fertilizer and subsequently used an underbrush device to cut the vegetation, mix it with the manure and throw it under the trees. The conventional farmers typically applied mineral fertilizer just after harrowing the soil between the rows, aiming to ease the root access to the fertilizer and as a complementary weed management strategy to herbicides and the underbrush. The cost of managing 1 ha of oranges is shown in Table 3.

Table 3 shows that large-scale organic farms applied significantly more total nitrogen (N) than small-scale organic and conventional farms, although organic and conventional used different N sources. The same

Table 3. Resource use for 1ha of oranges in the State of São Paulo, Brazil (2007).

	Organic				Conventional	
	Small-scale farm (<75 ha)		Large-scale farm (>75 ha)		Small-scale farm (<75 ha)	
Number of studied farms	5		2		6	
	Mean	Range	Mean	Range	Mean	Range
INPUT						
Mineral fertilizer N (kg/ha year)	-		-		111a	72-135
Organic fertilizer[a], N (kg N/ha year)[b,c]	87a	23-110	185b	145-225	6	0-35
Mineral fertilizer P (kg/ha year)	-		-		23	15-28
Mineral fertilizer K (kg/ha year)	-		-		85	56-122
Pesticides (kg active ingredients/ha year)	-		-		6	3-11
Copper (kg Cu/ha year)[b]	0.3a	0-0.8	5.5b	2.5-8.5	0.9a	0-2.7
Diesel (L/ha year)[b]	185a	94-273	272a	163-381	185a	79-446
Electricity for irrigation (kWh/ha year)	-		143	0-286	-	
OUTPUT						
Orange yield (t/ha year)[b]	18a	12-21	23a	17-29	20a	14-26

[a] The organic fertilizer is chicken manure, cattle manure and/or sugar cane filter cake.
[b] Small letters denotes significant differences between farm types at $p \leq 0.05$
[c] Statistics is calculated on the total N input of both mineral and organic N fertilizer.

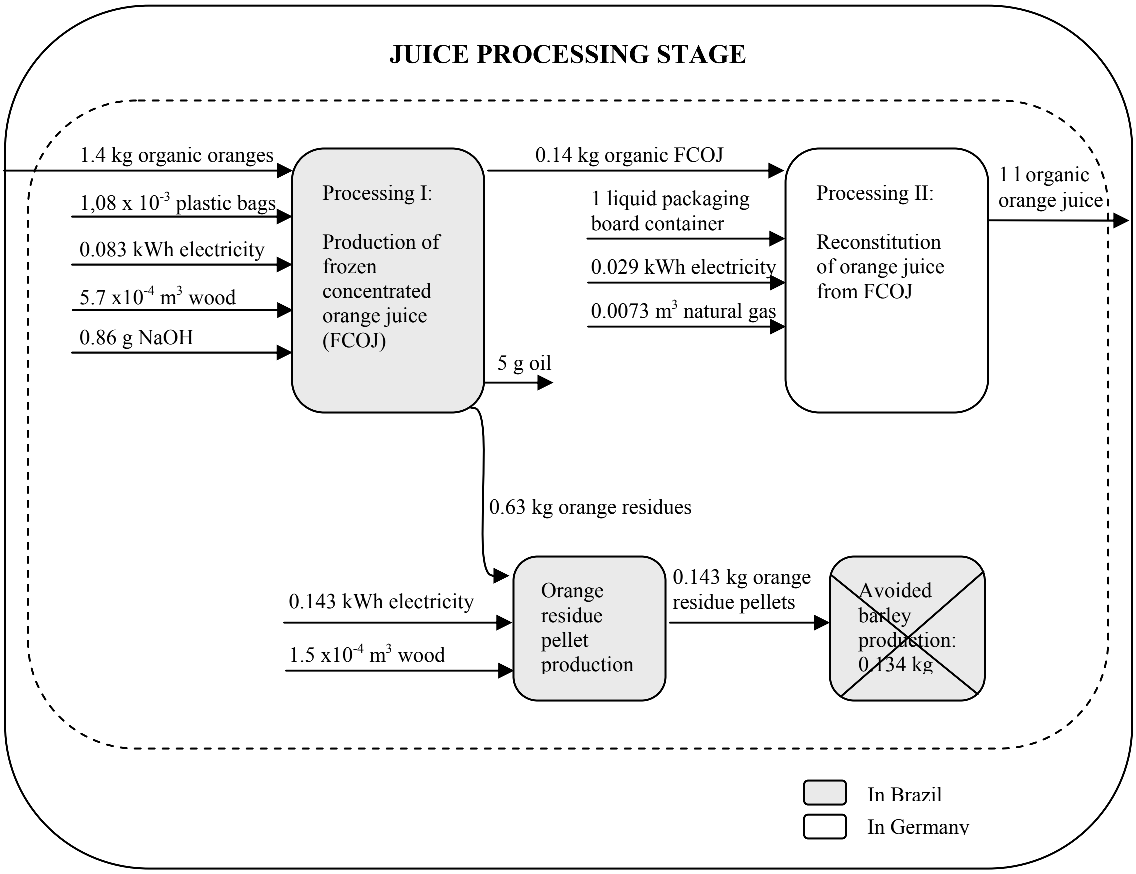

Figure 2. Illustration of the resource flows in the juice processing stage in the frozen concentrated orange juice (FCOJ) in Brazil (processing industry I) and the reconstitution to 1 litre orange juice in Germany (processing industry II) in 2007.

pattern was seen for copper (Table 3). The yield and diesel consumption were not significantly different for the studied orange plantations (Table 3). Since there was no difference in the amount of machinery used between the organic and conventional systems, the production phase of machinery was not included in the study, as it was regarded as having a minor impact on the results (Macedo et al., 2008). However, the operational phase of the agricultural and processing machinery was included in the study. Data on the agro-chemical production (e.g. fertilizers) was obtained from the Ecoinvent Database v2.0 as well. Country-specific data was not available, so European data was used.

2.6. Processing

Information on resource use and characteristics for the processing stage have been obtained by questionnaires and interviews with the managers at the processing plant I producing the frozen concentrated orange juice (referred to as concentrate in the following) in Brazil and the processing plant II reconstituting the orange juice from concentrate in Germany. A flow diagram of the resource use for 1 litre of organic orange juice is shown in Figure 2.

One of the main differences between the organic and conventional production of frozen concentrate is that organic oranges are washed with NaOH before being processed (Figure 2) where conventional oranges are washed with chlorine. The drums and pallets used in the production of the organic concentrate are reused and are therefore not included in the analysis. The wood used in the production of organic concentrate is from orange trees in the plantations. Thus, no production costs of wood are included in the analysis, only the average 30 km transport of the wood to the processing industry in 16-tonne trucks. For the electricity and ethanol, country-specific data from Brazil from the Ecoinvent Database v.2.0 was used. For the truck transport, country-specific data was not available, so European data was used.

2.7. Transport

Figure 2 illustrates the transport means and distances of the organic oranges from small-scale farms that were processed into frozen concentrate in Brazil, reconstituted into orange juice in Germany and transported to the retail distribution centre in Denmark (Figure 2). First the oranges were transported by truck

Table 4. Standard values used for the calculation of nitrogen budgets and emissions of the orange production in the State of São Paulo, Brazil

	Values		References
	Mean	Range	
ORANGE			
N content in oranges FW[a] (% N)	0.15	0.11 - 0.22	Alva et al (2006), Paramasivam et al (2000), Alva & Paramasivam (1998)
P content in oranges FW[a] (% P)	0.02	0.017 - 0.022	Alva et al (2006), Paramasivam et al (2000), Alva & Paramasivam (1998)
CHICKEN MANURE			
N content of FW[a] (% N)	2.25	1.5 - 2.4	Severino et al. (2006), Graciano et al (2006), Araújo et al (2006)
P content of FW[a] (% P)	1.89	2.2 - 3.1	Severino et al. (2006), Graciano et al (2006), Araújo et al (2006)
CATTLE MANURE			
N content of FW[a] (% N)	1.20	0.5 – 1.9	Severino et al (2006), Leão et al (2008)
P content of FW[a] (% P)	0.59	0.1 – 1.2	Severino et al (2006), Leão et al (2008)
FILTER CAKE (from sugar industry)			
N content of FW[a] (% N)	0.65		Macedo et al. (2008), Dinardo-Miranda et al (2003), Rosetto & Santiago (2006)
P content of FW[a] (% P)	0.17		Dinardo-Miranda et al (2003), Rosetto & Santiago (2006)
N availability of organic fertilizer for crops (%)	21	5-21	Munoz et al. (2008)

[a] Fresh weight

from the farms to the processing industry that produces concentrate. Then the frozen concentrate was transported by truck to the harbour of Santos, Brazil and loaded into a reefer towards Europe. In the Dutch port of Rotterdam, the frozen concentrate was reloaded onto a refrigerated truck to be taken to the processing industry that reconstitutes the frozen concentrate into ready-to-drink orange juice, which is reloaded onto a truck and transported to e.g. Denmark. Distances and means of transportation were obtained primarily from the farmers, the processing industry and relevant websites[1] . The inventory data on the transport means was obtained from the Ecoinvent Database v2.0 (Ecoinvent Centre, 2009). For the transport by reefer, an average load of 65% was assumed for the direct shipping from the port of Santos to the port of Rotterdam. For the refrigerated reefer an extra 12.5% energy use was added compared to unrefrigerated ships (Marintek, 2008). For the truck transport, country-specific data was not available and European data was used for all truck transport. For the refrigerated truck from Rotterdam to Frankfurt an extra 27% energy use was added compared to an unrefrigerated truck (Tassou et al., 2008).

[1] www.searates.com, www.maps.google.com, www.metric-conversions.org, www.distances.com

2.8. Calculation of nutrient budgets and emissions

Field level nitrogen (N) budgets including inputs and harvested outputs per ha per year were established in order to assess the balance for potential leaching risk. The partial field nutrient budgets were the outcome of a simple accounting process, which details the inputs (mineral fertilizers, organic inputs (e.g. manure) and the harvested outputs (crop sales and crop residue removal) from the field during a year (Watson et al., 2002). The standard values used for the calculations are shown in Table 4.

Subsequently, the emissions related to the orange production were estimated using the IPCC 2006 guidelines (IPCC, 2006) for the direct and indirect N_2O emissions. The NH_3 emissions during fertilizer application were estimated to be 4% of fertilizer-N (Cantarella et al., 2003). The N leaching loss was estimated to be 15% of the applied N in fertilizer or manure (Alva et al., 2006; Paramasivam et al., 2001; Dasberg et al., 1984; Cantarella et al., 2003). The sequestration of CO_2 in plant production was not included in the study, since the CO_2 will be emitted again. For the sensitivity analysis, changes in the soil organic carbon (C) were estimated using the simple tier 1 methodology in the IPCC 2006 guidelines (IPCC, 2006). The point of departure for the changes in soil organic carbon was the conventional plantation, with the perennial plantation covering half of the area and the other half of the area being a set-aside covered with either grasses or bare soil, due to harrowing once or twice a year. The organic plantations likewise had the perennial plantation covering half of the area. The remaining interrow was set-aside with perennial grasses, considered as improved grassland due to amendments with manure and no soil disturbance. The effect of including the soil carbon changes in the organic plantations is only presented in the sensitivity analysis for the GWP results at farm gate, since the method is simple and can be questioned. Results are given in both a 20-year and a 100-year perspective. The IPCC estimation method covers a period of 20 years, whereafter the soil is assumed to have reached a new 'steady state' C content. To calculate the results in a 100-year perspective, the changes in soil C within the 20 years are divided by 100 years instead (Knudsen et al., 2010).

2.9. Handling of co-products

In the environmental assessment of the product chain, some of the processes lead to more than one product. This is the case for the process leading to the production of manure and for the process leading to the frozen concentrate.

How to account for the environmental impacts related to the production of manure in LCA has been discussed by several authors (Audsley et al., 1997; van Zeijts et al., 1999; Dalgaard and Halberg, 2007). The approach used by Knudsen et al. (2010) will be applied in the present paper. In short, the estimation of the environmental costs of manure can be done in at least two ways:

A. The manure is regarded as a waste product from the livestock production and therefore all environmental costs from this should be allocated to the meat.
B. The fertilization value of the manure is taken into account using a consequential LCA approach, described by Dalgaard and Halberg (2007). In this approach the environmental costs of producing plant-available manure-N correspond to the environmental costs of producing an equivalent amount of mineral fertilizer-N because the manure may substitute mineral fertilizer in another form. For the organic fertilizer used, the N availability for crops is listed in Table 4. This approach will be used in the following, and the effect of using the approach described under A. (where manure is regarded as a waste product) is shown in the later sensitivity analysis.

Likewise, the production of frozen concentrate leads to a by-product; orange residues, used for ruminant fodder. The allocation of the environmental burden from the production of frozen concentrated orange juice and orange residues can also be done in at least two ways:

Table 5. Partial nitrogen (N) and phosphorus (P) budgets at field level (including inputs and harvested outputs) and emissions to air and water from 1 ha of orange plantation in the case study area in the State of São Paulo, Brazil (2007).

	Organic				Conventional	
	Small farms (<75 ha)		Larger farms (>75 ha)		Small farms (<75 ha)	
	N (kg N/ha)	P (kg P/ha)	N (kg N/ha)	P (kg P/ha)	N (kg N/ha)	P (kg P/ha)
INPUT						
Mineral fertilizer					111	23
Organic fertilizer	87	64	185	91	6	5
Fixation (green manure)	3	-	-	-	-	-
Deposition[a]	3		3		3	
TOTAL INPUT	93	64	188	91	120	28
TOTAL OUTPUT, Orange yield	27	4	33	5	29	4
FIELD BALANCE	67	60	155	86	90	24
Emissions, field						
Ammonia loss[b] (fertilization) (kg NH_3-N/ha)	2.7		3.5		6.4	
Nitrous oxide emissions[c] (kg N_2O-N/ha)	1.2		2.4		1.6	
Nitrate loss[d] (kg NO_3^--N/ha)	23		48		30	
Phosphate loss[e] (PO_4^{3-}-P/ha)		1.3		1.3		1.3

[a] Table value from Filoso et al. (2006).
[b] 4% of applied N (Cantarella et al., 2003).
[c] According to IPCC 2006 guidelines (IPCC, 2006)
[d] 15% of applied N (Cantarella et al., 2003; Alva et al., 2006; Paramasivam et al., 2001; Dasberg et al., 1984)
[e] According to Yu et al. (2006)

A. The environmental burden can be allocated according to the economic value (or the mass) of the products when they are sold.
B. The orange residues produced are sold for ruminant fodder and replace a carbohydrate in the fodder. In the present study barley is used as a marginal representative for a carbohydrate fodder with a nutritional value comparable, to the orange residues. The avoided environmental burden of producing barley is withdrawn from the total environmental costs of producing the products (consequential LCA approach). The remaining environmental burden is ascribed to the frozen concentrated orange juice. This last approach will be used in the following. The effect of using the other approach described under A. with economic allocation is shown in the sensitivity analysis for oranges at farm gate.

Since the by-production of oil for the industry in the frozen concentrate production represents only a small fraction of both the total production in mass (0.6%) and total income from the production (3%), no environmental costs are allocated to the oil production in the present study.

3. Results

3.1. Environmental impacts of organic and conventional orange production at farm gate

Nutrient surplus and losses at farm gate

A partial nitrogen (N) and phosphorus (P) field budget and estimated emissions from the orange plantations are presented in Table 5.

As seen in Table 5, nitrous oxide emissions and nitrate losses are larger from the orange plantations on large-scale farms due to higher N-applications, while small-scale organic and conventional orange plantations have similar emissions. As much as 50-100 kg N is unaccounted for in the N balance, which is in agreement with findings by Alva et al. (2006) and Dasberg et al. (1984), who suggests that this is accumulated in the tree trunks, stems and soil. In reality this should be subtracted from the carbon balance (GWP). However, since some of the C accumulated will be released later when orange trunks are used as renewable fuel for processing, this is not included. The effect of including the C accumulated in the soil in the GWP results is shown in the sensitivity analysis at farm gate

Table 6. Characterized results at farm gate for 1 tonne of organic and conventional oranges produced in the state of São Paulo (2007). Small letters denote significant differences between farm types at $p \leq 0.05$.

	INPUT STAGE	FARM STAGE		TOTAL
		Crop production	Traction and el	
Land use (ha farmland/t oranges)				
ORGANIC, small-scale	-	0.055	-	**0.055a**
ORGANIC, large-scale	-	0.044	-	**0.044a**
CONVENTIONAL, small-scale	-	0.050	-	**0.050a**
Non-renewable energy use (MJ/t oranges)				
ORGANIC, small-scale	257	-	507	**764a**
ORGANIC, large-scale	356	-	596	**952a**
CONVENTIONAL, small-scale	805	-	460	**1265a**
Global warming potential, GWP (kg CO_2 equiv./t oranges)				
ORGANIC, small-scale	15	30	39	**84a**
ORGANIC, large-scale	19	48	47	**114a**
CONVENTIONAL, small-scale	41	36	35	**112a**
Acidification potential (kg SO2 equiv./t oranges)				
ORGANIC, small-scale	0.1	0.1	0.3	**0.5a**
ORGANIC, large-scale	0.1	0.2	0.4	**0.7a**
CONVENTIONAL, small-scale	0.3	0.5	0.3	**1.1a**
Eutrophication potential (kg NO3-equiv./t oranges)				
ORGANIC, small-scale	0.2	7.3	0.6	**8.1a**
ORGANIC, large-scale	0.2	10.4	0.7	**11.3a**
CONVENTIONAL, small-scale	0.5	8.9	0.5	**9.9a**

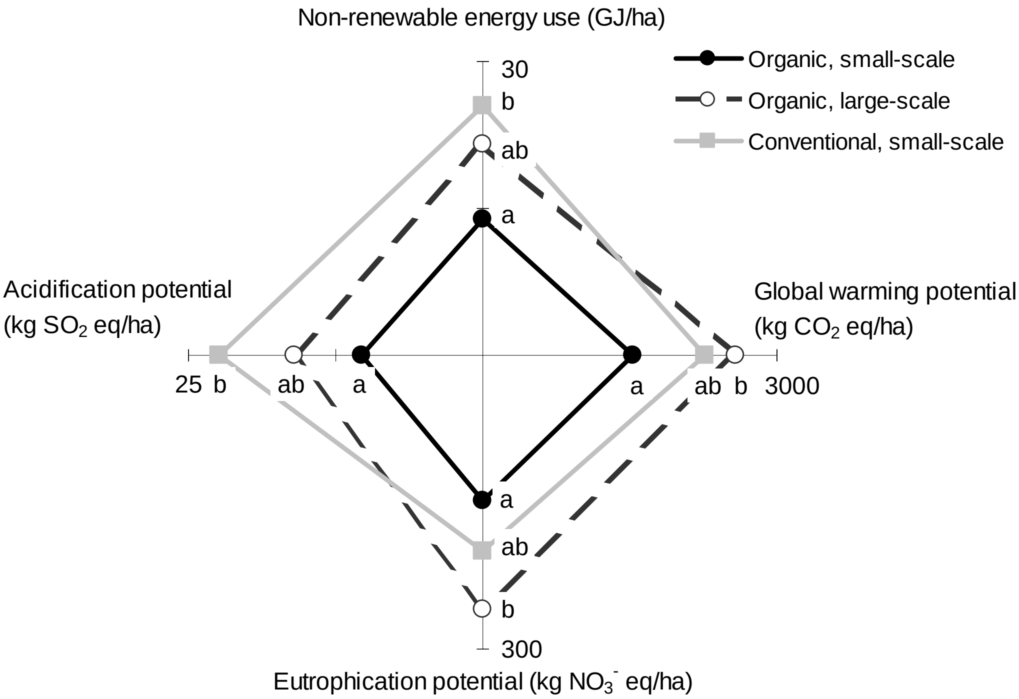

Figure 3. Environmental impact results per hectare of organic and conventional orange in the case area in the state of São Paulo (2007). Small letters denotes significant differences between farm types at p ≤ 0.05.

Characterized environmental impact results at farm gate

The results from the chosen environmental impact categories for 1 tonne orange at farm gate are presented in Table 6. In addition to the characterized environmental impact per tonne orange juice, the environmental impacts per ha are given in Figure 3.

The same overall pattern can be seen for the environmental impacts per tonne oranges (Table 6) and per hectare (Figure 3). The organic small-scale orange plantations consistently have the lowest value for the different categories (except for land use, which is slightly higher), followed by either the conventional small-scale and the organic large-scale with the highest value (for GWP and eutrophication) or the organic large-scale and the conventional small-scale with the highest value (for non-renewable energy use and acidification). The GWP and eutrophication potential per hectare of organic small-scale farms were not significantly lower than the conventional, with a p-value between 0.05 and 0.10. The higher GWP and eutrophication potential of the organic large scale farms is mainly due to the larger applications of N, (which results in the highest N_2O emission and N leaching of the three systems) (Table 6), while the highest non-renewable energy use and acidification potential of the conventional small-scale system are mainly due to the use of energy-consuming mineral fertilizer (Table 6). However, while the difference between the highest and the lowest environmental impact are significant when measured per hectare (Figure 3), no significant differences between farm types can be seen when the environmental impact is measured per tonne oranges produced (Table 6). It should be mentioned that since the organic orange plantations using manure were ascribed an environmental impact for the production of fertilizer corresponding to the amount of N available to crops (as mentioned in 2.9), the organic oranges also contribute considerably contribution to non-renewable energy use and GWP from the production of agricultural inputs.

3.2. Environmental hotspots of imported organic orange juice to Denmark

The environmental impacts from the organic orange juice imported to Denmark made from oranges from small-scale organic plantations in Brazil are illustrated in Figure 4 in a hotspot analysis for the chosen environmental impact categories. The environmental impacts of orange juice from large-scale organic, small-scale conventional and large-scale conventional farms (using farm data from Coltro et al., 2009) are presented in the later sensitivity analysis (Table 8), assuming that the post farm gate processing and transport are similar to that of the oranges from small-scale organic farms. Figure 4 shows that the transport stage adds the main contribution to both non-renewable energy use, GWP and acidification potential.

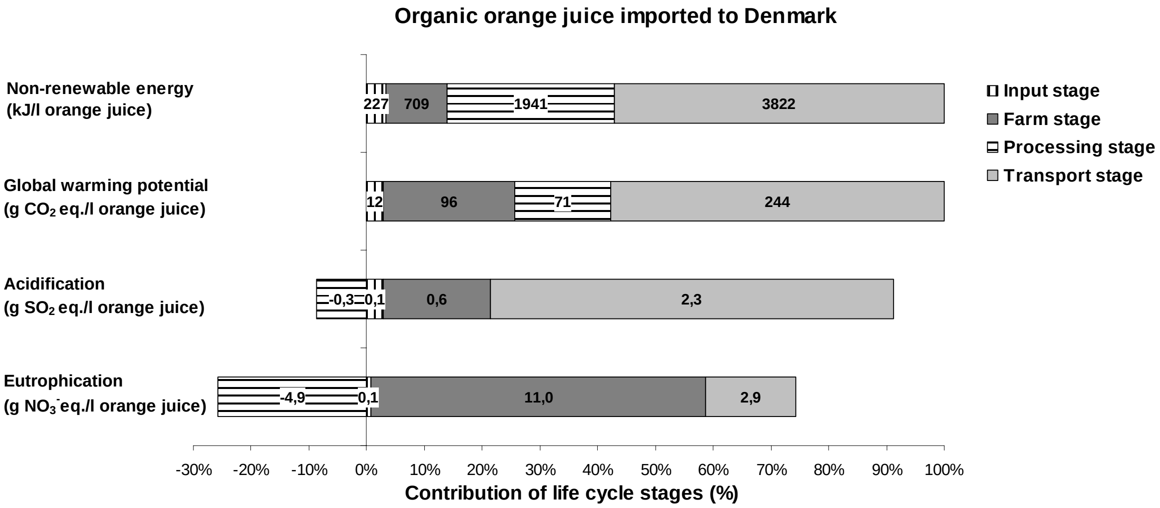

Figure 4. Hotspot analysis and environmental impact results for organic orange juice produced and processed to FCOJ in Brazil, reconstituted in Germany and transported to Denmark – based on data from small-scale organic farmers (2006/7).

The total non-renewable energy use of the organic orange juice was 6699 kJ per litre orange juice, with the hotspot being the transport stage (57%) followed by the processing (29%). Approximately the same pattern was seen for the GWP with 58% from the transport stage, 17% from the processing and 23% from the farm stage. The total GWP for organic orange juice imported to Denmark and made from Brazilian organic oranges from small-scale farmers was 424 g per litre organic orange juice. Interestingly, the major contribution to GWP per litre orange juice within the transport stage was the truck transport of reconstituted orange juice from Germany to Denmark (115 g CO_2 eq.), followed by the truck transport of freshly picked oranges from the farm to the frozen orange concentrate producer within Brazil (63 g CO_2 eq.), the truck transport of frozen concentrate from Rotterdam harbour to the reconstitution factory in Germany (22 g CO_2 eq.) and from the processing factory to the harbour in Brazil (17 g CO_2 eq.). Surprisingly, refrigerated ship transport of frozen concentrated orange juice was only the 5th largest contributor to the GWP within the transport stage (15 g CO_2 eq.), only higher than the transport of inputs to the farm and the processing industry (13 g CO_2 eq.). The results are mainly due to the difference in transporting water along with the oranges combined with the difference in CO_2 emission per ton kilometre (tkm) between trucks and ships (Ecoinvent Centre, 2009). The total acidification and eutrophication potential of 1 litre of orange juice imported to Denmark were 3 g SO_2 eq. and 9 g NO_3^- eq., respectively.

Table 7. Global warming potential, GWP (kg CO_2 eq./ton orange) at farm gate for oranges produced in the State of São Paulo, Brazil as affected by changes in certain assumptions in a sensitivity analysis.

ORANGES	**Organic**		**Conventional**
	Small farms (<75 ha)	Larger farms (>75 ha)	Small farms (<75 ha)
Reference scenario[a]	**84**	**114**	**112**
Changes in assumptions:			
1) Central data estimates			
50% higher N in compost (affects only organic)	100	130	112
45% N availability in compost for crops (affects only organic)	94	128	112
More electricity (and pesticides[b]) applied (like Coltro et al., 2009)	165	178	199
2) System boundaries or allocation procedures			
Manure regarded as waste (affects only organic)	75	101	112
Soil carbon changes[c] included (20 year perspective)	51	88	112
Soil carbon changes[c] included (100 years perspective)	77	109	112
Including the burden from the first 4 unproductive years	105	143	140

[a] In the reference scenario, calculations of N_2O are based on IPCC 2006 guidelines and the fertilization value of manure/compost is included in the system in the form of mineral fertilizer in the amount that corresponds to total N in compost times the N availability in compost for crops (here: 21%). Furthermore, estimates on carbon sequestration are not included.
[b] Pesticides only included for the conventional farms.
[c] IPCC (2006), tier 1 approach.

For the acidification potential, the hotspot was also the transport stage with the major contribution coming from the truck transport of reconstituted ready to drink orange juice transported from Germany to Denmark. However, with regard to the eutrophication potential the main contribution came from the farm stage. The negative values contributing to the acidification and eutrophication potential are due to the avoided production of barley because of the co-production of orange residue pellets (Figure 2).

3.3. Sensitivity analysis

In order to test the uncertainty related to the central estimates and assumptions considered to have the greatest influence on the results, a sensitivity analysis focusing on GWP of oranges at farm gate was performed (Table 7).

Since the estimates of N content in organic manure/compost and N availability is central to both the estimates of N_2O (crop production) and the estimates of impact from agricultural input production, the effect of increasing the levels are tested. This shows an increase of 14-19% and 12% for the two main assumptions for the two systems. However, even with the changed assumptions regarding N in manure and availability to crops, the average GWP value for oranges from organic small-scale farms does not exceed the average values for the other two systems.

Compared to similar studies, electricity and pesticide use are low in the present study. Therefore, the effect of using the same (high) level of electricity and pesticides, as on the conventional orange farms surveyed by Coltro et al. (2009) in the state of São Paulo, Brazil, is tested. Pesticides are of course only applied to the conventional system whereas the same high level of electricity is applied to all three systems. A higher electricity (and pesticide) consumption increased the GWP values of the oranges at farm gate from the three systems considerably by approximately 55%.

The effect of regarding manure as a waste product from livestock production with no environmental production costs was tested. Thus, the GWP values of the organic systems were lowered but only slightly (approx. 12%). The inclusion of soil carbon changes had a decreasing effect on the GWP of organic oranges and widened the difference in GWP between the organic and conventional oranges more or less depending on the chosen time perspective (Table 7). Finally, the effect of including the burden from the first fours unproductive years of the orange trees out of 20 years was tested, which increased the GWP in all farm types by approximately 25%.

The sensitivity analysis for the orange juice GWP estimate is presented in Table 8. A different way of handling co-products was tested, using economic allocation for the frozen concentrated orange juice instead of expanding the system to include barley production and a combination of this and adding no environmental costs on the manure (manure regarded as waste), which both increased the GWP estimate. Secondly, the sensitivity analysis showed that transporting the frozen concentrated orange juice directly to Denmark for reconstitution and consumption would decrease the GWP by 24%, if processing facilities for reconstitution are assumed to be unchanged. This is due to the avoided transport of water, when only the concentrate is transported.. Thirdly, the effect of using different farm data than from the small-scale organic plantations was tested assuming that all other factors in the rest of the product

Table 8. Global warming potential, GWP (gCO2 eq./l orange juice) at retail distribution centre in Denmark of orange juice from small organic plantations in the State of São Paulo, Brazil as affected by changes in certain assumptions in a sensitivity analysis.

ORANGE JUICE	Organic
	Small farms (<75 ha)
Reference scenario[a]	**424**
Changes in assumptions:	
Using economic allocation of FCOJ[a] instead	461
Combination: Manure regarded as waste and economic allocation of FCOJ[a]	442
Direct transport of FCOJ[a] to Danish processing plant instead of processing in Germany	321
Using farm data from large-scale organic plantations instead	466
Using farm data from small-scale conventional plantations instead	464
Using farm data from large-scale conventional plantation reported by Coltro et al. (2009) instead	516

[a] Frozen concentrated orange juice.

chain remained unchanged. Farm data from Coltro et al. (2009) representing large-scale conventional farms was included. The sensitivity showed that the lowest GWP was found for the small-scale organic farms while the highest was found for orange juice from large-scale conventional farms (22% higher) (Table 8). Likewise, assuming processing and transport chains to be similar, GWP of orange juice from conventional large-scale farms was 11% higher than orange juice from large-scale organic farms (Table 8).

4. Discussion

4.1 Methodology and sensitivity

The effect of handling the co-products in the LCA, such as manure and orange residues, are tested in the sensitivity analysis. Interestingly, the different approaches do not change the results markedly. Regarding manure as a waste product and excluding the 'production costs' of the manure (as described in section 2.9), decreases the GWP of the oranges at farm gate by approx. 11% (Table 7). However, when regarding both manure as a waste product and using economic allocation for the frozen concentrated orange juice co-products (which could be termed as a so-called attributional approach) the GWP of the orange juice is on the contrary increased by only 4%. With regard to methodology it should also be noted that the establishment of the plantation is not included in the calculation, but the sensitivity analysis estimated the effect by an approximately 25% increase in the GWP of oranges (Table 7), which results in an increase of 7% in the GWP of orange juice (424 to 453 g CO_2 eq. per litre orange juice). The inclusion of soil carbon sequestration widened the difference in GWP between organic and conventional (Table 7). The estimated increase in soil organic carbon under organic management is in agreement with findings by Canali et al. (2009) who found significantly higher total organic carbon values in organically managed citrus orchards compared to conventionally managed. However, the methodology used for estimating soil carbon changes in the present study is very rough (IPCC, 2006) and the results should be interpreted with caution. Furthermore, there is a need for discussion and consensus on which time perspective is the more appropriate. Thus, there is a need for further development of the methodology concerning soil carbon sequestration. Finally, the overall choice of the conventional reference plantations for all the environmental impacts in the present study can be questioned since the conventional farmers were small-scale and financially restricted. Thus, the difference between environmental impacts of organic and conventional orange production might be underestimated in the present study, which becomes visible when including the farm data of Coltro et al. (2009) in the sensitivity analysis (Table 8) that represents large-scale conventional farms in the state of São Paulo.

4.2 Global warming potential and energy use

The main hotspots for GWP of organic orange juice were the transport stage followed by processing. The main contribution in the transport stage comes from the truck transport of fresh orange in Brazil and reconstituted orange juice in Europe. The lowest GWP was found for the oranges from small-scale organic plantations followed by the small-scale conventional and the large-scale organic ones, even though no significant difference was found. When using other farm data than from small-scale organic farms for the orange juice chain, the sensitivity analysis showed that the highest GWP was found for orange juice from large-scale conventional farms (using data from Coltro et al. (2009), while the lowest was from organic small-scale farms, assuming all other factors equal in the rest of the product chain.

Since the amount of N fertilizer and the diesel consumption are the main contributors to the GWP per tonne orange at farm gate in the present study, those two factors combined with the yields are also the main determinants of the outcome. Generally, the amount of diesel used in the present study (Table 3) was comparable to other studies (Coltro et al , 2009; Sanjuán et al., 2005a and b). However, the amount of N fertilizers applied in the present study (Table 3) were lower than in comparable studies from Spain and Italy applying 240-290 kg N per ha (Sanjuán et al., 2005a and b; Beccali et al., 2009). Since N use affects both N_2O estimates and agricultural input production, a lower N use has a profound effect on GWP and energy use. In the study of conventional orange production in São Paulo, Brazil, Coltro et al. (2009) found a slightly lower N use than seen on the large-scale organic orange plantations here, but still a higher N use than the small-scale orange farmers in the present study. However, the most striking difference to the present study is the high use of pesticides and electricity (used mainly for irrigation) in conventional production, reported by Coltro et al (2009), which contributes significantly to the GWP, as shown in the sensitivity analysis. The low use of pesticides (and fertilizer) of the conventional farmers in the present study is, as mentioned earlier, due to a lack of economic resources. The electricity use reported by Coltro et al. (2009) is remarkably high when compared to other studies of irrigated orange plantations (Sanjuan et al., 2005a and b; La Rosa et al., 2008).

Comparing conventional and organic orange production, the slightly lower GWP and non-renewable energy use of the organic versus the conventional orange plantations per tonne orange is consistent with Sanjuán et al (2005a; b) who conducted two separate LCA studies of integrated and organic oranges, respectively. The GWP level per tonne organic oranges is comparable to that of the study of Sanjuán et al. (2005b), when differences in methodology have been accounted for (pers. comm. Neus Sanjuán Pellicer, Universitat Politécnica de Valéncia, Spain, 2010.). In the study of organic oranges by Sanjuán et al. (2005b) no production costs for manure was included, but the input from electricity (due to irrigation) was higher than in the present study. Sanjuán et al. (2005b)'s study of integrated oranges shows a much higher GWP of integrated oranges than found for conventional oranges in the present study, mainly due to a higher use of N fertilizer and a higher estimation of emissions from the production of mineral fertilizer.

The only other peer-reviewed published study found, that was able to estimate the GWP of orange juice for the whole chain was Beccali et al. (2009) focusing on concentrated conventional orange juice in Italy. Beccali et al (2009) also identified transport as one of the main hotspots and found a value of approximately 1.2 kg CO_2 eq. per litre orange juice (depending on how much water is added to the juice concentrate). The higher GWP value can partly be explained by higher N application (three times higher than in the present study), higher diesel consumption, use of irrigation and the use of electric energy for the concentration of orange juice instead of renewable energy compared to the present study. With regard to the transport stage in the present study, the sensitivity analysis showed a GWP reducing effect of transporting the frozen concentrated orange juice and processing it as close to the consumption as possible. This furthermore indicates that GWP of freshly squeezed orange juice consumed in Denmark, which is a growing market, will have a considerable contribution from transport.

4.3 Biodiversity and land use

Organic farmers had higher crop diversity on the farm compared to large-scale organic and small-scale conventional farms. Organic farms also had a permanent plant cover in the interrows, while the plant cover in the conventional interrows was disrupted by harrowing and herbicide applications. Furthermore, the use of toxic pesticides is replaced by mainly $CuSO_4$ and $CaSO_4$ in the organic plantations. The absence of pesticides and diversified land use in organic agriculture is mentioned in Hole et al. (2005) as main reasons that give rise to a higher diversity and abundance of species under organic farming (Bengtsson et al., 2005).

Oranges are a crop with a high pesticide application (Wilhoit et al (1999). Clay (2004) reported that the orange production was the crop with the highest pesticide use per hectare in Brazil. Coltro et al (2006) found that the average pesticide use for oranges for juice production in the São Paulo region in Brazil was 1.3 kg active ingredients per kg orange, which corresponds to 43 kg active ingredients per ha in the study. However, the pesticide use by the small-scale conventional farmers in the present study was much lower. The main pesticides used in the conventional orange plantations were the herbicide Glyphosate, the fungicide Folpan (Folpet) and the insecticides Karate zeon, Marshal (Carbosulfan), Torque (Fenbutatin-oxide), Omite (Propagate), Decis (Deltamethrin), Supracid (Methidathion), Cascade (Flufenoxuron), Cipermetrina (Cypermethrin) and Vertimec. Torque, Omite and Supracid is classified as highly toxic in acute toxicity. Cascade and Folpan is not acutely toxic, whereas the rest is harmful or moderate acutely toxic. Omite and Folpan are classified as known carcinogens and Supracid and Cipermetrina as possible carcinogens. All the insecticides used are highly toxic to aquatic organisms, except from Supracid and Cascade, which are only moderately toxic to aquatic organisms (PAN, 2010). The pesticide use is replaced by mainly $CuSO_4$ and $CaSO_4$ towards orange pests and diseases. The highest use of copper is seen in the large-scale organic plantations (Table 3) and this long term contamination of the soil can be problematic in a long term perspective.

The more diverse farming systems of the small-scale organic farmers with a higher crop diversity compared to an orange monoculture at the conventional farms, would, according to Altieri (1999), in it self give rise to more associated diversity. Furthermore, the small organic farms have more focus on the vegetation between the orange rows in the plantation as a source of green manure which could give rise to increased diversity to the plantation and increased carbon sequestration, whereas the plant cover in the conventional plantations are interrupted by harrowing once or twice a year and herbicide applications. Booij and Noorlander (1992) note that the most important factors determining arthropod abundance and diversity in agroecosystems are the availability of food, shelter and suitable microclimate, which are factors closely related to the quantity, quality and duration of plant cover. Likewise, in a study of natural enemies of Diptera leafminer in an organic citrus orchard in Brazil, dos Santos et al. (2007) highlights the role of spontaneous vegetation for the establishment and multiplication of natural enemies. The non use of pesticides on organic versus conventional orange farms would only add to that higher diversity in organic systems (Hole et al., 2005, Bengtsson et al., 2005). A higher biodiversity in the soil under organic citrus orchards in São Paulo, Brazil compared to conventional ones was found by França et al. (2007) who found a higher richness and diversity of Arbuscular Mycorrhizal Fungus. With regard to the aboveground biodiversity, Genghini et al. (2006) found significantly positive effects on bird communities of organic compared to conventional management of orchards in northern Italy.

4.4 Eutrophication and water contamination

The farm stage was identified as the main hotspot for eutrophication potential of organic orange juice, which was also found by Beccali et al (2009). The lowest eutrophication potential per ton orange was found for the oranges from small-scale organic plantations followed by the small-scale conventional, while the highest eutrophication potenatial was found for the large-scale organic ones, even though no significant difference was found. The estimated eutrophication potential per hectare showed the same pattern, but here there was a significant difference between small-scale and large-scale organic plantations. Canali (2002) also found the amount of potentially leaching nitrates to be lower in organically managed soils than in conventional ones. It could be discussed whether interpretation of the specific impact category eutrophication is more relevant per hectare farmland than per yield unit (Knudsen et al., 2006).

The higher nitrogen supply in the large-scale organic orange plantations gives rise to higher N leaching and eutrophication compared to both small-scale conventional and organic farms. The eutrophication potential per ton orange in the present study is in average twice as low as the one found in the study of Sanjuan et al (2005a). However, the N application in the present study is also more than twice as low. In the citrus growing regions of central Florida, leaching of fertilizer nutrients and widespread NO_3-N contamination of drinking water wells are a serious concern (Paramasivam et al., 2001). In Florida the recommended nitrogen inputs have been around 280 kg N per hectare (Parsons & Boman, 2006). The present case study area with orange production in São Paulo, Brazil is facing the same problems, where there is a concern about the leaching of nutrient and pesticides to the adjacent rivers (CBT-TB, 2000; Cantarella et al., 2003).

Surface and groundwater contamination with pesticides are not included in the present study due to methodological limitations, but it should of course be considered as a potential environmental impact. Pesticides pose a risk for human health mainly by contamination of drinking water, pesticide residues in food or farmer's inhaling or exposure on skin (Pimentel, 2005). The pesticide use is described and discussed in section 4.3. The ban of pesticides in organic agriculture underlines the difference in the environmental profile of the organic and conventional oranges. With regard to other water contaminants, conventional oranges at the frozen concentrated orange juice processing plant are washed with chlorine, before being pressed, contrary to organic oranges which are washed with NaOH instead

5. Conclusions

The main contribution to the GWP per litre organic orange juice from small-scale farmers imported to Denmark was the transport stage accounting for 57% of the emissions followed by the processing stage (29%). Especially the truck transport of fresh oranges in Brazil and ready to drink orange juice in Europe contributed to this number. The GWP was reduced by 24% by transporting the frozen concentrated orange juice directly to Denmark for processing. Comparing organic and conventional small-scale orange production, the crop diversity was higher on organic farms, while non-renewable energy use and acidification per hectare was significantly lower. The same pattern was seen for GWP and eutrophication although not significantly lower. Including the increased soil carbon in organic plantations widened the difference in GWP between organic and conventional. Furthermore, the use of toxic pesticides was replaced by a small amount of CuSO4 and CaSO4, which may have a positive effect on biodiversity along with the higher crop diversity and the spontaneous vegetation between the orange trees and on the health of the farmers or workers that apply the pesticides. Comparing organic small-scale with large-scale orange production, the crop diversity was higher on the small-scale organic farms, while the GWP, eutrophication potential and the use of copper per hectare was significantly lower, indicating that environmental impacts from small-scale differ from large-scale organic farms.

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Paper IV

A methodological approach to include soil carbon changes in life cycle assessments of agricultural products

Petersen BM, Knudsen MT, Hermansen JE, Halberg N (manuscript)
Global Change Biology

A methodological approach to include soil carbon change in life cycle assessments

Bjørn Molt Petersen[1], Marie Trydeman Knudsen[2*], John Erik Hermansen[1] & Niels Halberg[3]

[1]Department of Agroecology and Environment, Faculty of Agricultural Science, University of Aarhus, DK-Tjele, Denmark. [2]Department of Agriculture and Ecology, Faculty of Life Sciences, University of Copenhagen, DK-2630 Taastrup, Denmark. [3]International Centre for Research in Organic Food Systems (ICROFS), DK-8830 Tjele, Denmark.

* Corresponding author. Present address: Department of Agroecology and Environment, Faculty of Agricultural Science, University of Aarhus, Blichers Alle 20, P.O. Box 50, DK-8830 Tjele, Denmark. Tel.: +45 8999 1308; Fax: +45 8999 1200; E-mail: mariet.knudsen@agrsci.dk.

A B S T R A C T

Globally, soil carbon sequestration is expected to hold 89% of the agricultural potential to mitigate agricultural greenhouse gas emissions. However, the majority of life cycle assessments (LCA) of agricultural products have not included possible changes in soil carbon sequestration. Recently, few studies have attempted to include soil carbon sequestration in LCA using slightly different methodologies and time horizons. In addition the time perspective of the soil CO_2 emissions are not included. Thus, there is a need for development on the methodology considering that the estimated soil carbon sequestration rates are very sensitive to the chosen time horizon.

In the present study, a methodological approach to estimate carbon sequestration to be included in LCA is suggested and applied to two examples where the inclusion of carbon sequestration is especially relevant: 1) Bioenergy: removal of straw from a Danish soil for energy purposes and 2) Organic versus conventional farming: comparative study of soybean production in China, where the carbon sequestration is estimated. The suggested approach considers the time of the soil CO_2 emissions for the LCA by including the Bern Carbon Cycle Model. A time perspective of 20, 100 and 200 years are used and a soil depth of 0-100 cm. Thus, the suggested methodology is well-suited for the LCA methodology. The application of the suggested methodology showed that the results were comparable to the IPCC 2006 tier 1 approach in a time perspective of 20 year, where after the suggested methodology showed a continued soil carbon change towards a new steady state. The suggested methodology estimated a carbon sequestration for the first example when storing straw in the soil instead of using it for bioenergy of 54, 97 and 213 kg C per tonne straw C in 200, 100 and 20 years perspective, respectively. For the conversion from conventional to organic soybean production, a difference of 32, 60 or 143 kg soil C per ha per year in a 200, 100 or 20 years perspective, respectively was found. The study furthermore indicated that soil carbon changes included in the LCA can provide a major contribution to the total greenhouse gas emissions per crop unit for plant products.

Keywords: soil carbon sequestration, LCA, straw, methodology, bioenergy, organic, conventional, soybean

1. Introduction

Soil carbon (C) sequestration in agricultural soils is expected to hold a major potential for agriculture's global warming mitigation potential to reduce agricultural emissions and increase C sequestration. Smith *et al.* (2007) estimated soil C sequestration to contribute about 89% to the global mitigation potential from agriculture, whereas mitigation of CH_4 emissions and N_2O emissions from soils account for only 9% and 2%, respectively. The global warming potential related to a specific agricultural product can be estimated by using life cycle assessment (LCA), which is a method for integral assessment of several environmental impacts aggregated into impact categories (e.g. climate change, eutrophication etc.) along the product chain. However, the importance of soil C sequestration is poorly reflected in current LCA's (Koerber *et al.*, 2009), since the majority of studies have not included soil C sequestration in the overall greenhouse gas estimations, mainly due to methodological limitations. The British publicly available specification (PAS 2050, 2008) for assessing product life cycle greenhouse gas emissions have not included soil C changes either, whereas agriculturally induced land use changes (e.g. forests to agricultural land) is included with a time perspective of 20 years (PAS 2050, 2008). The IPCC guidelines (IPCC, 2006) includes a tier 1 approach on how to estimate changes in soil C stocks using a 20 years default time perspective, which can be included in LCA, as it has been done in Knudsen *et al.* (2010) and Knudsen *et al.* (submitted). However, this method is rough and can be questioned especially with regard to the time perspective. Some main uncertainties and discussions with regard to including soil C changes in LCA's of agricultural products would be: a) The spatial system boundary: optimal estimated depth in the soil profile, b) temporal system boundary: optimal time horizon (20, 30, 100 or 200 years), c) towards a new equilibrium: saturation of soils, d) estimation of soil C changes (modelling or measurements) (Garnett et al., 2010)

Recently, a few LCA studies have attempted to include soil C changes - using mainly modelling and using time horizons of a few to 30 years, but the time horizon used is not explicitly stated in all of the studies (Hörtenhuber *et al.*, 2010; Röös *et al.*, 2010; Halberg *et al.*, 2010; Hillier *et al.*, 2009; Mila i Canals *et al.*, 2008 and Gabrielle & Gagnaire, 2008). The subjects of these studies are mainly bioenergy (Hillier *et al.*, 2009; Gabrielle & Gagnaire, 2008) or organic agricultural production (Hörtenhuber et al, 2010; Halberg *et al.*, 2010), since C sequestration is especially relevant to include in these studies (Whitaker *et al.*, 2010). However, the different approaches used can be discussed firstly with regard to the consideration of time in the LCA with regard to the temporary profile of the emissions or sequestration and secondly the time perspective used. The consideration of time in LCA has been discussed by Levasseur *et al.* (2010), who suggested a method to include time in LCA with regard to land use change in an example of biofuels. The time perspective has also been discussed with regard to C sequestration in forestry (Costa and Wilson, 2000) and the global climate change negotiations (Kirschbaum, 2006; Fearnside, 2002; Fearnside *et al.*, 2000), since the global warming potential (GWP) results are very sensitive to the chosen time horizon. Thus, there is a need for development and suggestions of a methodology for estimating and including soil C changes in LCA (Whitaker *et al.*, 2010).

The aim of the present paper is to suggest a methodological approach to estimate the effect of soil C changes on CO_2 in the atmosphere to be included in LCA's considering the above-mentioned uncertainties and discussions. The methodology is applied to two examples where the inclusion of soil C changes is especially relevant: 1) Bioenergy: removal of straw from a soil in Denmark for energy purposes and 2) Organic versus conventional farming: comparative study of soybean production in China. The effect of soil C changes is estimated and it is shortly described how this could be included in a future LCA. Furthermore, the effect of different time horizons is illustrated.

2. Materials and methods

2.1 Assessment of soil carbon changes in space and time

A number of issues are complicating the estimation of soil C changes related to a specific activity, which is also part of the reason why soil C changes have been included in only a few LCA studies.

First of all, one has to decide on a system boundary in space in which the soil C changes are estimated. At which depth should the soil C changes be assessed? Is it only necessary to consider the topsoil (normally the plough layer with a maximum depth of 30 cm) or do soil C changes also take place below 30 cm making it necessary to consider a soil profile of e.g. 0-100 cm? In the simple IPCC tier 1 guideline (IPCC, 2006) for estimating changes in the soil C stock, only the topsoil is taken into account. In contrast, a soil profile of 0-100 cm is considered in the C-TOOL model (Petersen,

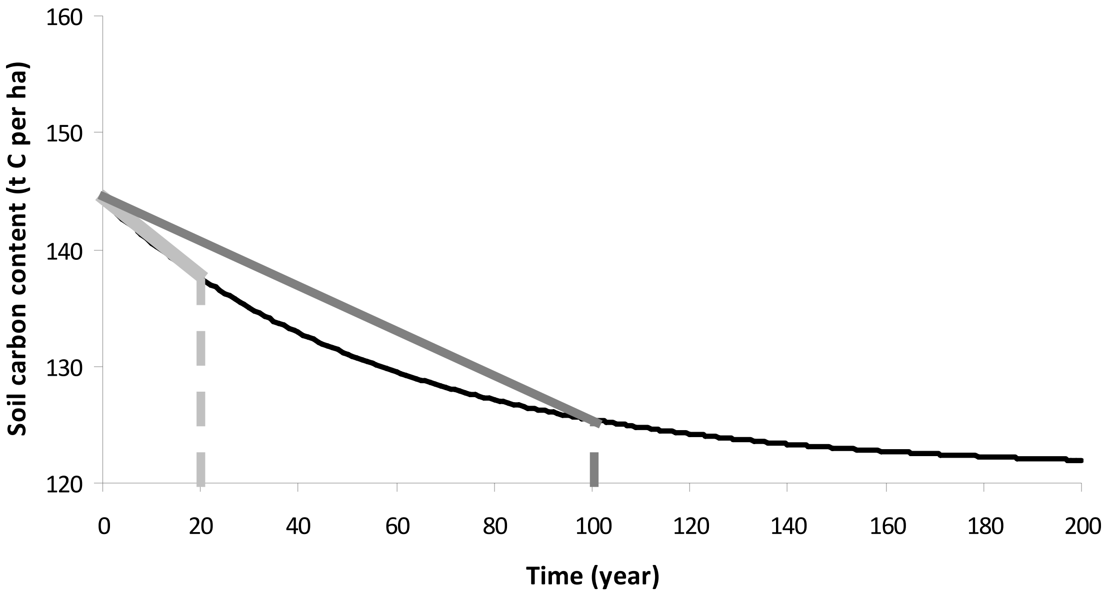

Figure 1. Illustration of the impact of the chosen time perspective when estimating soil carbon changes.

2010) used in this study. The difference between choosing 0-100 cm instead of only the topsoil is discussed further in the sensitivity analysis.

Secondly and even more importantly, one has to decide on a system boundary in time. Every agricultural practice in theory reaches a certain ‘steady state’ level of soil C after a number of years. A switch to a new agricultural practice such as conventional to organic (Example II) or removal of straw instead of leaving it in the field (Example I) will lead to a change towards either a higher or a lower level of soil C organic matter. C in soil organic matter is however not ‘stable’ but there is a constant turnover and the net changes in soil C will be a balance between deposited and emitted C. When the agricultural practice is changed, the level of soil C will increase/decrease more at the beginning of the period and then level out to reach a new equilibrium, as illustrated in Figure 1. Thus, the C change rate will be highest in the first few years and then the gains/losses will decline over time.

Thus, using a time perspective of 20 years, like in the IPCC tier 1 guideline (IPCC, 2006); the estimates of annual soil C changes will be higher compared to a time perspective of 30 years or 100 years. Thus, the time perspective chosen to evaluate the C sequestration or the payback time is crucial and different time perspectives have been used (Gabrielle & Gagnaire, 2008; Hörtenhuber *et al.*, 2010 & Halberg *et al.*, 2010). Fearnside (2002) have argued that a 100 year time perspective should be used for global warming mitigation calculations, like it is normally used for the calculations of the global warming potential, while others argue that the time perspective is more of a political decision than a scientific one (Levasseur *et al.*, 2010). In the present paper the consequences of choosing three time horizons of 20, 100 and 200 years are presented.

The final issue mentioned here is how the soil C changes in an agricultural system should be estimated. Should this be by measurements, categorical estimates or modelling? Measurements would be more relevant for calibrating and validating the models than for direct use in LCA of future scenarios. Categorical estimates are used in the IPCC guidelines (IPCC, 2006) tier 1 approach for estimating the soil C stock changes in typical situations. These are based on choices of four categories of land use, three categories of tillage and four categories of input with regard to crop residues and manure (IPCC, 2006). In contrast, the current approach uses dynamic modelling as the point of departure to increase the accuracy of the results. The soil C model C-TOOL is used for the estimation of the soil C changes, which is further described in (Petersen, 2010; Petersen *et al.*, 2002). The model consists of three C pools: FOM, which compasses freshly added matter and soil biota; HUM, native soil organic matter or “humus”; ROM, very slowly decaying matter with a halving time under Danish conditions of approx. 1500 years. Each of these pools exhibit first-order decay.

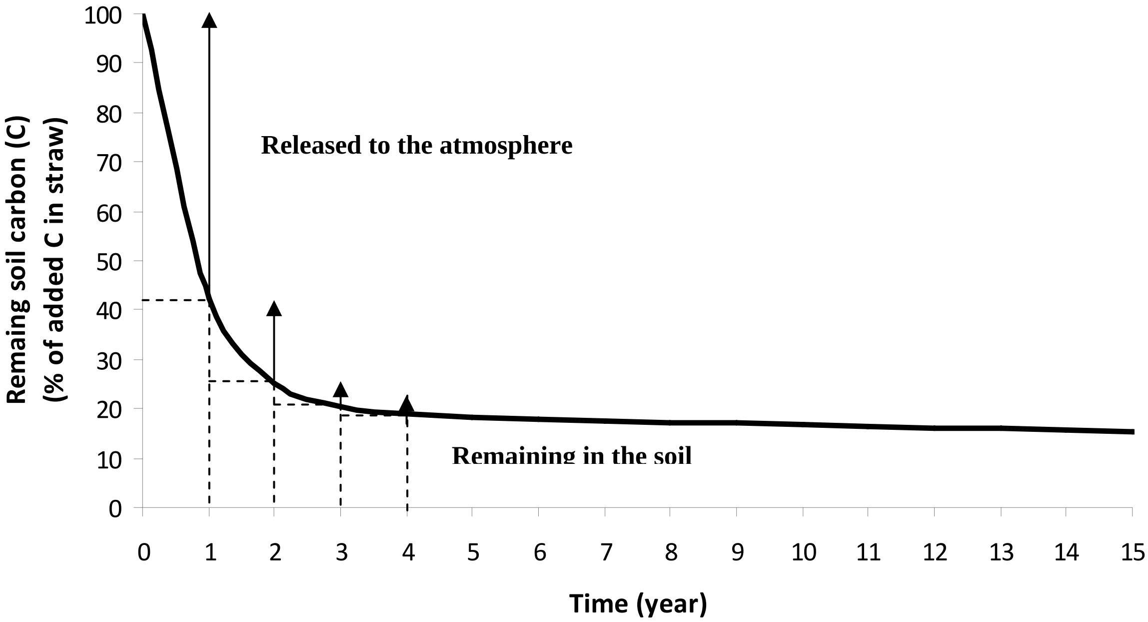

Figure 2. Generic illustration of the decay of carbon (C) (e.g. crop residues or manure) added to the soil as a single event in the first year. The area below the graph is C retained in the soil and the arrows above the graph illustrate the C that is released to the atmosphere.

The model considers both the topsoil (0-25 cm) and the subsoil (25-100 cm). Transport of C from the topsoil to the subsoil is included. In principle every soil C modelling tool can be used, as it is only a question of knowing the change in the C balance of the system. This modelling approach will be compared to estimates using the IPCC guidelines (IPCC, 2006) in a sensitivity analysis.

2.2 CO_2 emitted from soil in a time perspective: The Bern Carbon Cycle Model

Normally the time of emissions is not included in the LCA, since the emissions normally are emitted within the analysed time frame (e.g. a year). However, dealing with soil C changes, added C is released from the soil in different quantities over a longer period. The poor accounting for time-related conditions has been pointed out as one of the limitations in LCA (Reap *et al.*, 2008), since releasing a big amount of pollutant instantaneously generally does not have the same impact as releasing the same amount pollutant at a small rate over several years (Levasseur *et al.*, 2010). When C (in the form of e.g. crop residues or manure) in one year is added to the soil, parts of the C will remain in the soil, while other parts will be released to the atmosphere dependent on time, as illustrated in Figure 2.

This is not a linear development, as more C will be released to the atmosphere in the beginning of the period and then the emissions will gradually decline.

The soil carbon dynamics are modelled by C-TOOL (Petersen, 2010) in the present paper, but in principle any suitable soil C model can be used for this modelling.

When the C is released to the atmosphere in the form of CO_2, it will follow a decay pattern same as any other release of CO_2 to the atmosphere due to absorption in sinks (mainly the oceans). The decay pattern of CO_2 in the atmosphere is described by the Bern Carbon Cycle Model for which the following equation serves as a proxy (IPCC, 2007):

Equation 1:

$$f(t) = 0.217 + 0.186\exp(\frac{-t}{1.186}) + 0.338\exp(\frac{-t}{18.51}) + 0.259\exp(\frac{-t}{172.9})$$

where *f (t)* is the fraction of CO_2 left in the atmosphere dependent on time, *t*.

Figure 3 shows this decay of a pulse of CO_2 released to the atmosphere, when it is transferred to other pools, such as terrestrial ecosystems and the oceans.

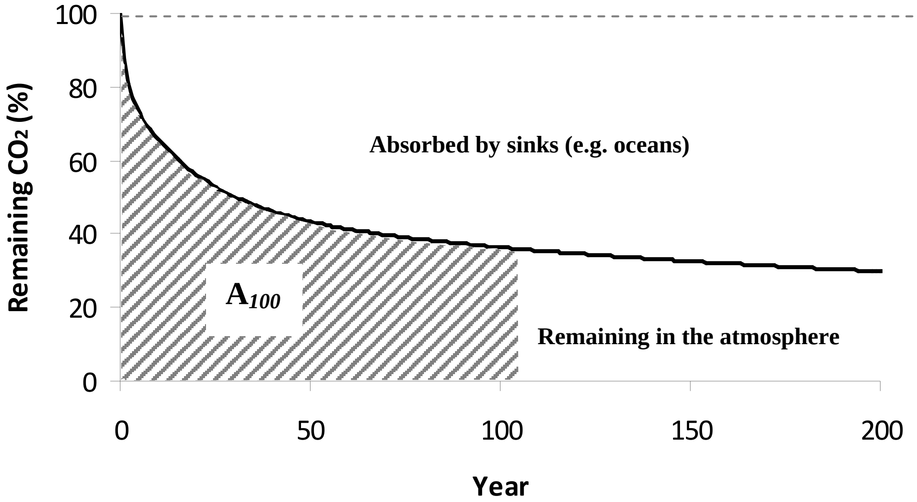

Figure 3. Decay of CO_2 in the atmosphere, based on the Bern Carbon Cycle Model, f(t) (IPCC, 2007). The CO_2 remaining in the atmosphere is the area below the curve as described by A_T (equation 2). An example of the fraction of CO_2 remaining in the atmosphere in a 100-year perspective, A_{100}, is given.

The area below the curve represents the CO_2 present in the atmosphere in a specific time frame perspective.

Equation 2:

$$A_T = \int_1^T f(t)\,dt$$

where T is the time horizon and $f\ (t)$ is derived from equation (1).

As illustrated in Figure 3, the area below the curve (equation 1) in a 100-year perspective, for instance, is only 48% of the hypothetical value without sinks.

The above description of the fate of C (e.g. straw or compost) added to the soil, that dependent on time will end up in either soil, atmosphere or C sinks (mainly oceans) is focused on a single years addition of C.

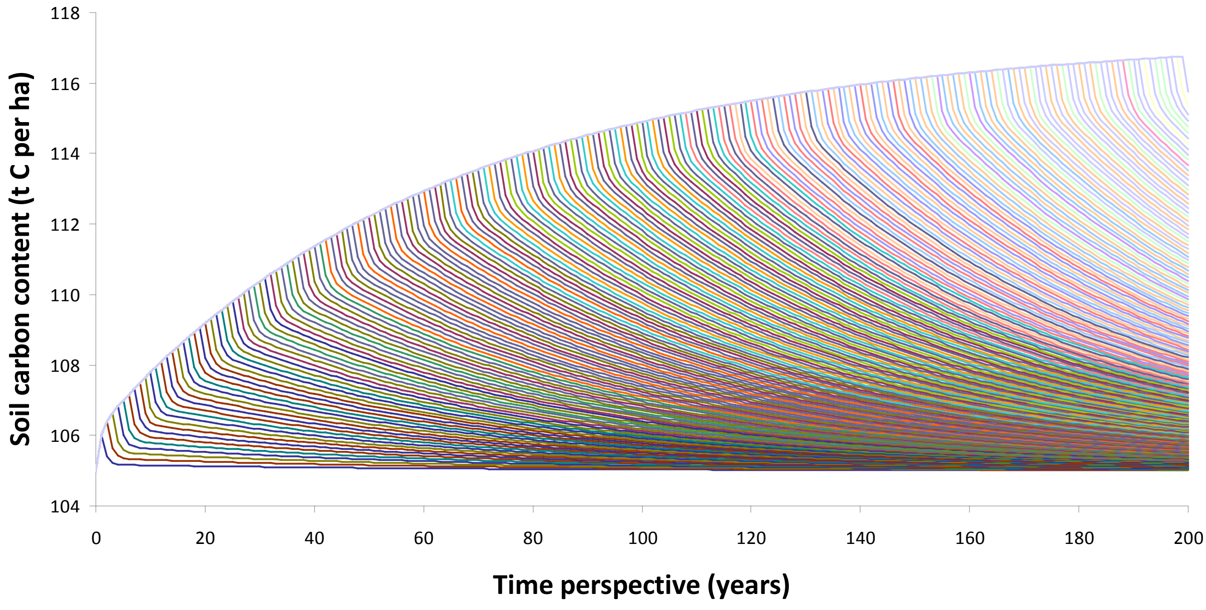

Figure 4. Illustration of the build-up of soil carbon (C) towards a new steady state from repeated additions of C (e.g. straw or compost) based on the decay curves of every single addition of C. The point of departure is a scenario where less C has been added to the soil.

Table 1. Grain and straw yield plus carbon (C) yield from straw in a typical Danish crop rotation for pig and cash crop farmers (from Wesnæs *et al.*, 2009).

	Grain yield (hkg/ha)	Straw yield (hkg/ha)	C in straw (t C/ha)
Winter barley	72	41.8	1.60
Winter rape	36	30.2	1.16
Winter wheat	89	53.4	2.04
Winter wheat	80	48.0	1.84
Spring barley with catch crop[a]	57	35.9	1.37
Spring barley	57	35.9	1.37

a. The catch crop is assumed to be either undersown perennial ryegrass or white mustard sown immediately following harvest.

However, this focus on a single year is fully additive if this event is repeated year after year. Figure 4 illustrates the build-up of soil C from repeated annual additions, approaching a new 'steady state' or equilibrium, where the decay curve from Figure 2 (of C added to the soil and released to the atmosphere) is recognisable for every year.

Thus, it is possible to focus on a single year's addition and release of C independent of where on the curve of Figure 4 the event takes place, as long as the time perspective of released C is taken into account.

On this basis a methodological approach to estimate the effect of soil C changes on CO_2 in the atmosphere was developed and applied to two typical LCA situations using a case study approach: I) the choice between using cereal straw for bioenergy or leaving it on the soil, II) comparison of two farming systems differing in their soil organic matter building practices.

2.3 Example I: Straw removal from agricultural soils in Denmark for bioenergy purposes

The first example is focused on a case study of removing cereal straw for bioenergy purposes versus leaving it in the field in a typical crop rotation in Denmark. The case study area is characterized by a coastal climate with a mean average temperature of 7.7 ^{o}C (average for Denmark, 1961-90). The climate zone is according to IPCC guidelines (IPCC, 2006) cool, temperate, moist. The soils are Alfisols with 12% clay and 62 tonne C per hectare in the topsoil (0-25 cm). The average yields of the typical crop rotation for pig and cash crop farmers in the case study in Denmark, which is winter barley, winter rape, winter wheat, winter wheat, spring barley with a catch crop and spring barley (Wesnæs *et al.*, 2009) is given in Table 1. An annual application of 500 kg C in slurry per hectare is assumed.

On average, 1.56 tonne C per hectare per year are available in straw. For simplicity, the crop rotation was chosen to include the crops in Table 1, leaving out minor crops for pig and cash crop farmers such as different legumes, seed crops and potatoes. In this study, it was assumed that one tonne C per hectare was removed for bioenergy respectively left in the field.

Table 2. Main inputs and outputs relevant for the C balance for 1 hectare of soybean in the Jilin Province, China (2006). Values are means ± standard deviation.

		Organic	Conventional
INPUT			
Mineral fertilizer, N (kg N/ha)		-	47 ± 12
Organic fertilizer[a] (m^3/ha)		13±4	-
Organic fertilizer[a], N (kg N/ha)		45 ± 13	-
OUTPUT			
Soybean yield (kg/ha)		2788±306	3083±310
Crop residues [b]	….left in field (%)	23	13
	…burned (in kitchen) (%)	40	41
	...removed for fodder (%)	4	5
	…burned (in field) (%)	-	41
	...removed for compost (%)	33	-

a. Compost, which consists of cattle manure (60%), forest soil (20%) and soy/maize crop residues (20%)
b. % represents the mass of the crop residues.

Table 3. Partial carbon budgets[a] at field level (kg C/ha) due to different fertilization and crop residue management practices of 1 ha of organic and conventional soybean production in the Jilin Province, China (2006).

	Organic	Conventional
INPUT		
Organic fertilizer[b]	675	0
OUTPUT		
Crop residues burned in kitchen	874	896
Crop residues removed for fodder	87	109
Crop residues burned in field	0	717
Crop residues removed for compost	721	0
PARTIAL FIELD BALANCE	-1007	-1722

a. Inputs from crop carbon assimilation during photosynthesis and output from harvested crops are not included, since organic and conventional soybean yields were not significantly different.
b. The C:N ratio in the compost used is estimated to be 15:1 according to Tang *et al.* (2006), Eiland *et al.* (2001), Stamatiadis *et al.* (1999) & Evanylo *et al.* (2008).

2.4 Example II: Organic versus conventional production of soybeans in China

Example II is focused on a case study of organic versus conventional soybean production in the Jilin province in China, which is further described in Knudsen *et al.* (2010). The case study area is characterized by a continental climate with a mean average temperature of 4.0^{o}C (in between climate station Changchun and Dunhua). The climate zone is according to IPCC guidelines (IPCC, 2006) cool, temperate, dry (on the border to moist). The soils are Mollisols with 14% clay and 3% C in the topsoil (Zhao *et al.*, 2006). In the case study, 20 organic farms and 15 conventional farms producing soybeans were included. The main crops for both farm types were soybeans and maize. The main inputs and outputs relevant for the C balance from soybeans are given in Table 2.

The crop residues from the conventional soybean fields were burned, whereas in organic these were incorporated into the soil as compost. The organic farmers were using compost, whereas the conventional farmers were using mineral fertilizer, which also affects the C balance of the soybean fields. The two most important differences between organic and conventional soybean production affecting the soil C balance is the fertilization method and the crop residue management practice.
The C assimilated in the crop and the C harvested in soybeans is assumed to be similar since there was no significant difference between organic and conventional soybean yields (Table 2). The relative changes in the organic and conventional soybean fields due to different fertilization and crop residue management practices are presented in Table 3. This approach is comparable to the soil organic carbon (SOC) deficit as suggested by Mila i Canals *et al.* (2007).

3. Results

3.1 Methodology to include soil C sequestration in LCA's

The suggested methodology to estimate the effect of soil C changes on CO_2 in the atmosphere to be included in LCA's takes its point of departure in the above description of the fate of a single year's C addition to the soil (section 2.4).

Thus, when some of the C added to the soil is released to the atmosphere due to the decay of C in the soil over time (Figure 2), some of this will be absorbed by sinks (e.g. oceans) due to the 'decay' pattern of CO_2 in the atmosphere over time (Figure 3). While the decay curve of added C to the soil (Figure 2) can be described by the C-TOOL model (Petersen, 2010) or any other soil C model, the decay curve of C released to the atmosphere (Figure 3) can be described by the Bern Carbon Cycle Model (IPCC, 2007).

The combination of the two curves for decay of C in soil (Figure 2) and the subsequent atmospheric 'decay' of the released CO_2 from the soil (Figure 3), respectively, with the subsequent atmospheric 'decay' of the released soil C (from Figure 2), is illustrated for the first four years after applying crop residues to the soil.
From Figure 5 it is visible that the approx. 60% of added C to soil that are released to the atmosphere in the first year (Figure 2) has a subsequent decay in the atmosphere according to the Bern Carbon Cycle Model. The second year an extra approx.15% of the originally added C is released to the atmosphere (Figure 2) following a decay pattern in the atmosphere (Figure 5) and so forth. In Figure 5 only the following four years after applying C to the soil is illustrated.

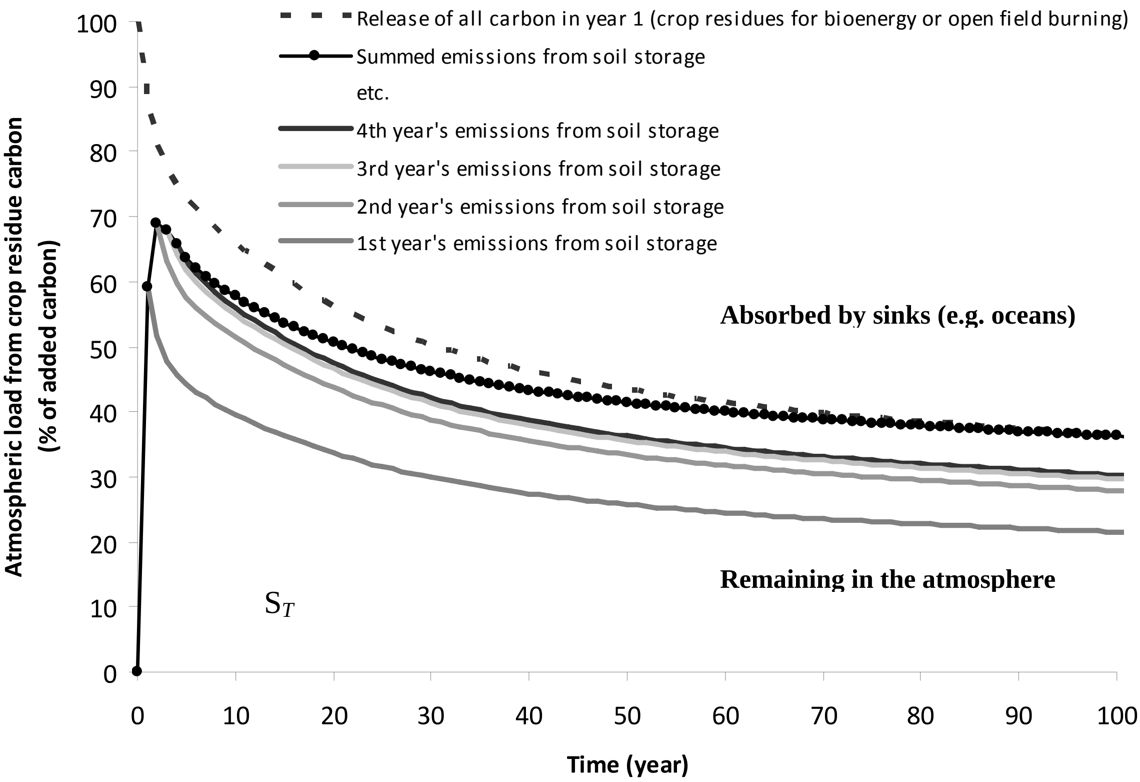

Figure 5. Illustration of the atmospheric load from either soil storage or burning of crop residue carbon (C). The soil storage curves are a combination of the decay curve of a single event of carbon (C) added to the soil in the first year and the decay curve of CO_2 in the atmosphere (Bern Carbon Cycle Model) – shown on a yearly basis for the first four following years only as an illustration. The curve of the summed emissions from soil storage contain all the following years. The upper dotted line represents the scenario where the entire C in crop residues is released in the first year (as for bioenergy use or open field burning).

However, all years are included in the summed curve illustrating the decay of the summed emissions from soil storage.

Thus, the area below the summed graph (Figure 5) expresses the time integrated relative atmospheric load of CO_2 as influenced by soil storage and can be formulated in one-year step numerical integration as:

Equation 3:

$$S_T = \sum_{i=1}^{T} (a(i) \sum_{j=1}^{T-i} f(j))$$

where T is the time frame, *a(i)* is the release of CO_2 in year i from a single addition of crop residues (as resulting from the decay of organic matter, and *f (j)* is given by equation 1.

In addition to the graph illustrating S_T, Figure 5 also includes a graph (the upper dotted line) illustrating the atmospheric load of CO_2 if the entire C in straw was released in the first year (if crop residues were used for bioenergy purposes or burned in the field), which corresponds to A_T (Figure 3; Equation 1).

Thus, the area below the dotted line in Figure 5 represents A_T, which is the atmospheric load if the entire C in crop residues was released in the first year. The area below the summed curve in Figure 5 represents S_T, which is the atmospheric load if the straw was left in the soil.

The relative emission reduction related to storing the crop residue C in the soil instead of releasing the C to the atmosphere is thus given by the area between the two curves (Figure 5). The soil storage effect equivalent to an emission reduction R_T over a T-year perspective is calculated as follows:

Equation 4:

$$R_T = \frac{A_T - S_T}{A_T}$$

where R_T is the fraction of the straw C that is stored in a T-year perspective and thus comparable to an avoided emission compared to releasing the entire C in crop residues in the first year. A_T is the atmospheric CO_2 load from crop residue C released to the atmosphere (e.g. for bioenergy) (the area below the Bern Carbon Cycle Model curve) and S_T is the atmospheric load from soil storage of the same amount of crop residues.

If the emission reduction, R_T, in a 100-year perspective was e.g. 10% through leaving 1 tonne of C in crop residues on the soil instead of releasing it all to the atmosphere in the first year, this would mean that a greenhouse gas (GHG) emission corresponding to 100 kg C is avoided in a 100-year time perspective for every tonne of C left in the field in a particular growing season.

As mentioned earlier, this approach furthermore focuses on the consequences of changing the C balance in a field in a single year, which will have an effect on CO_2 in the atmosphere. The resulting emission reduction (if e.g. crop residues were incorporated instead of burned) is assumed fully additive if this event is repeated year after year. This basic method for quantifying the consequences for CO_2 in the atmosphere of an activity that changes the soil C balance over a period was then applied to the two case studies.

Thus, the soil storage effect equivalent to an emission reduction, R_T is basically calculated in four steps:

1) The consequences for the C balance, in terms of the amount of C stored in the soil versus emitted to the atmosphere, is identified and estimated.
2) The soil C decrease and emissions over time is estimated using C-TOOL based on site-specific climate and soil data or another dynamic soil C model such as e.g. RothC (Coleman & Jenkinson, 1996).
3) The emissions from the soil decay of added C are combined with the Bern Carbon Cycle Model to estimate S_T, (given by equation 3).
4) Finally, the emission reduction, R_T, is calculated (given by equation 4).

Based on the amount of C found in step 1), the emission reduction, R_T, can then be converted into a sequestration potential per unit added C or per hectare.

The general method is well suited as a supplement to LCA studies and compatible with the IPCC methodologies. Values will be given for 20, 100 and 200 years.

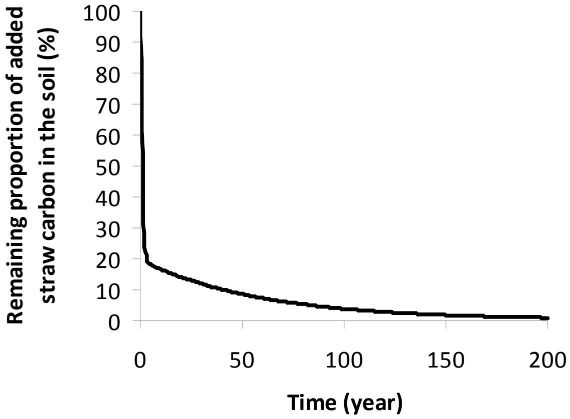

Figure 6. Decay of one tonne straw carbon when added to the soil as a single event in the first year according to C-TOOL modelling

3.2 Application of methodology to Example I: Straw removal for bioenergy in Denmark

The consequences for the C balance were based on one tonne of straw C stored in the soil instead of using it for bioenergy. The soil C decrease and emissions over time from added straw C in a Danish soils using C-TOOL is shown in Figure 6.

There was a relatively rapid decay of 80% of the added straw C during the first few years following the application and after 100 years up to 95% of the added C had been released to the atmosphere as CO_2.

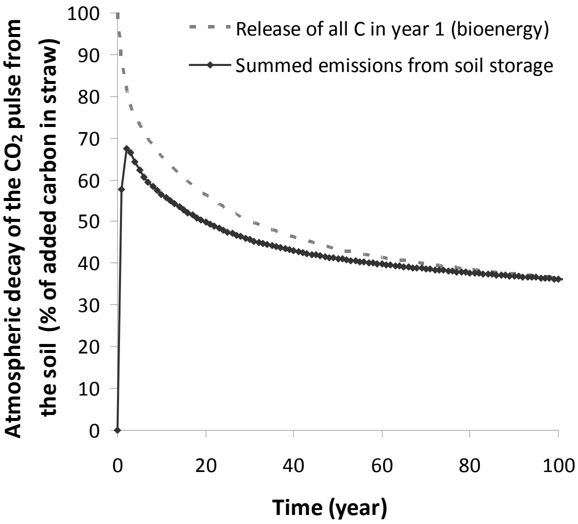

Figure 7. Atmospheric CO_2 emission load from soil storage of straw carbon (C) added in the first year in relation to releasing the entire straw C in the first year (for bioenergy) - using C-TOOL combined with the Bern Carbon Cycle Model.

Table 4. Emission reduction, R_T, carbon (C) sequestration and CO_2 reduction when incorporating one tonne of straw C in a soil in Denmark instead of using it for bioenergy (Example I).

Time perspective (years)	Emission reduction, R_T (%)	Carbon sequestration equivalents (kg soil C/ t straw C)	CO_2 reduction[a] (kg CO_2 / t straw C)
20	21.3	213	781
100	9.7	97	356
200	5.4	54	198

a. The carbon sequestration is multiplied by 44/12 to get the CO_2 reduction, based on the molecular weight of CO_2 to C.

When these emissions from the soil decay of straw C were combined with the Bern Carbon Cycle Model, as illustrated in Figure 7, the net emission reduction, R_T in atmospheric C from leaving straw C on the soil was shown to be 21.3% in a 20 years perspective and 9.7% in a 100 years perspective (Table 4).

Thus, when conducting a LCA study on straw for bioenergy in Denmark, an additional 781 kg CO_2 release should be added to the climate change impact per tonne straw C used for bioenergy if one uses a 20 year time perspective, due to soil C reduction in Denmark, or 198 kg CO_2 per tonne straw C if evaluated in a 100 year perspective.

3.3 Application of methodology to Example II: organic versus conventional soybean production in China

The consequences for the soil C balance when converting from conventional to organic soybean production in the case study in China is presented in Table 3 that shows a difference of 715 kg C per ha. The crop C assimilation during photosynthesis and the C output from harvested crops are assumed not to be affected by the change to organic, since crops yields were not significantly different (Table 2). Furthermore, the amount of crop residues burned in the kitchen stove and the amount of crop residues removed for fodder is comparable for organic and conventional soybean production (Table 3) and thus not affected by the conversion to organic management practices.

The main practice affected is the handling of the share of the crop residues which is burned on the field (717 kg C per ha) under conventional practice and removed for compost (721 kg C per ha) and returned to the field under organic management (Table 3).

The C-TOOL modelling of the decay and emissions of crop residue C in the Chinese soil over time using site-specific driving variables combined with the Bern Carbon Cycle Model resulted in a soil storage effect equivalent to an emission reduction, R_T, of 20.0% and 8.4% in a time perspective of 20 and 100 years, respectively. Thus, of the 715 kg C per ha per year that are added extra to the organic soils, 143 kg C per ha year are sequestered in a 20 years perspective and 60 kg C per ha per year in a 100 years perspective (Table 5) in the soil. Table 5 furthermore presents the resultant extra C sequestration per area and reduction in CO_2 emissions per area and per crop unit caused by soil storage of crop residues instead of open field burning.

Thus, converting from conventional to organic soybean production practices in the case study in the Jilin province in China causes a removal of an extra emission load of 524 or 220 kg CO_2 per ha per year from the atmosphere in a 20 or 100 years perspective, respectively.

When conducting a comparative LCA of organic and conventional soybeans from this case study in China the difference in greenhouse gas emissions per crop unit should be widened by 188 kg CO_2 per tonne soybean, using a time perspective of 20 years.

Table 5. Emission reduction, R_T, carbon (C) sequestration and CO_2 reduction when converting from conventional to organic soybean production practices and thereby incorporating 715[a] kg/ha year extra of soy residue C in a soil in the Jilin province, China instead of burning it in the field (Example II).

Time perspective (years)	Emission reduction, R_T (%)	Carbon sequestration equivalents (kg soil C/ ha year)	CO_2 reduction[b] per area (kg CO_2 / ha year)	CO_2 reduction per crop unit (kg CO_2 / t soybean)
20	20.0	143	524	188
100	8.4	60	220	79
200	4.5	32	117	42

a. From C balance (Table 3)
b. The carbon sequestration is multiplied by 44/12 to get the CO_2 reduction, based on the molecular weight of CO_2 to C.

Box 1.
IPCC 2006 methodology applied to Example I:

Temperature zone: Cool temperate, moist

$SOC_{ref,\ Alfisol\ (HAC\ soils)}$: 95 t C/ha (in 0-30 cm)

F_{LU}: 0.69 (long-term cultivated)
F_{MG}: 1.00 (full tillage)
$F_{I,\ no\ straw\ removal}$: 1.00 (medium input)
$F_{I,\ straw\ removal}$: 0.92 (low input)

No straw removal: 95 x 0.69 x 1.00 x 1.00 = 65.55
Straw removal: 95 x 0.69 x 1.00 x 0.92 = 60.31
Difference: 5.24 t C/ha

C sequestration, 20 years perspective:
5.24 t C/ha / 20 years = 262 kg C /ha year

CO_2 reduction:
262 kg C/ha year x 44/12 = 961 kg CO_2/ha year

Box 2.
IPCC 2006 methodology applied to Example II:

Temperature zone: Cool temperate, dry

$SOC_{ref,\ Mollisol\ (HAC\ soils)}$: 50 t C/ha (in 0-30 cm)

F_{LU}: 0.80 (long-term cultivated)
F_{MG}: 1.00 (full tillage)
$F_{I,\ organic\ soybean\ production}$: 1.00 (medium input)
$F_{I,\ conventional\ soybean\ production}$: 0.95 (low input)

Organic: 50 x 0.80 x 1.00 x 1.00 = 40
Conventional: 50 x 0.80 x 1.00 x 0.95 = 38
Difference: 2.0 t C/ha

C sequestration, 20 years perspective:
2.0 t C/ha / 20 years = 100 kg C /ha year

CO_2 reduction:
100 kg C/ha year x 44/12 = 367 kg CO_2/ha year
367 kg CO_2/ha year /2.788 t/ha year
= 132 kg CO_2/t soybean

Taken into account that the total greenhouse gas emissions of the Chinese organic soybeans at farm gate excluding soil C changes was estimated to 156 kg CO_2 eq. per t soybean (Knudsen *et al.*, 2010), the inclusion of C sequestration plays a major role in determining the result. Even if a 100 year perspective was chosen the consequences of not burning straw equal 79 kg C per ha per year corresponds to 51% of the total farm gate greenhouse gas emissions per ton organic soybean.

3.4 Sensitivity analysis

A sensitivity analysis was performed on the suggested methodology. First of all, the methodology was compared to the IPCC 2006 tier 1 approach to estimate change in soil C stocks (IPCC, 2006), as presented in Box 1 and 2. SOC_{ref} is the default reference soil organic C stocks for mineral soils (under native vegetation) and F_{LU}, F_{MG}, F_I is the relative stock change factors related to land use, tillage and input, respectively.

The present C sequestration results (per tonne straw C) from Example I (Table 4) need to be converted to an area basis to be comparable to the IPCC results. Since the available cereal straw yield per hectare in Example I is 1.56 t C per ha per year (Table 1), the difference in C sequestration (based on Table 4) from removal of all available straw to leaving it in the field in a 20 years perspective will be 213 kg C /t C x 1.56 t C/ha year = 332 kg C/ha year as compared to the IPCC approach estimating 262 kg C/ha year (Box 1).

The IPCC estimate of a C sequestration of 100 kg C /ha year when converting from conventional to organic soybean production in the case study area in Example II (Box 2) is directly comparable to the estimate of 143 kg C/ha year in a 20 year time perspective in the present study (Table 4).

The IPCC estimates were thus lower than our new estimates, which is partly due to the deeper soil layers included in C-TOOL (0-100 cm) compared to the IPCC approach (0-30 cm). If C-TOOL is parameterized for 0-25 cm only, the comparative value for C sequestration in 20 years for Example I would be 195 kg C/ton straw C and 305 kg C/ha year, which is only 8% lower/higher than the IPCC estimate of Example I.

Figure 8 illustrates for Example I, the soil carbon changes if all available straw carbon in the crop rotation is removed from the Danish field, using both the suggested approach with a soil depth of 0-100 cm, the IPCC tier 1 2006 approach with a soil depth of 0-30 cm and a modified suggested approach where the estimated soil depth is reduced to 0-25 cm to be comparable to the IPCC approach.

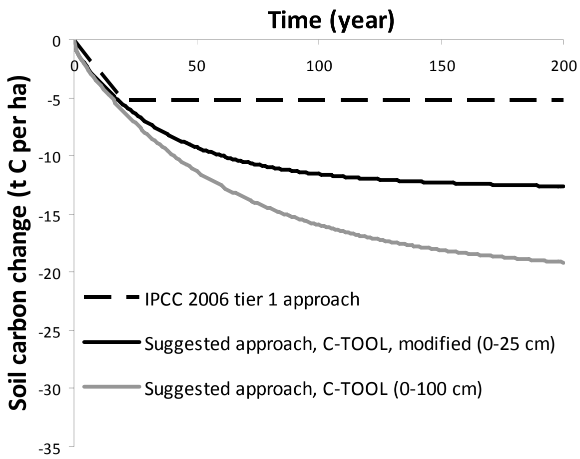

Figure 8. Relative soil carbon change for Example I, if all available cereal straw is removed year after year from the Danish field using either the suggested approach (incl. C-TOOL) with a soil depth of 0-100 cm, the IPCC 2006 tier 1 approach or the suggested approach modified to consider only 0-25 cm.

According to the C-TOOL modelling in Figure 8 it appears as if the IPCC tier 1 approach does not describe the entire soil carbon loss from straw removal. However, the magnitude of the yearly changes is comparable for the three different approaches for the first 20 years.

After 20 years, the C-TOOL simulation shows continued soil C losses towards a new steady state where the yearly soil C losses are lower. Interestingly, the C-TOOL simulations show that a new steady state will be approached sooner when considering only the topsoil compared to considering 0-100 cm (Figure 8).

Finally, the effect of possible future temperature increases on the modelling results is examined. As the soil C decay is affected by temperature, so is the emission reduction, R_T.
Figure 9 illustrates how R_T of Example I will decrease with increasing temperatures.

4. Discussion

4.1 Methodology

The main difference of the present methodology as compared with other approaches to include soil C sequestration in LCA (e.g. Halberg *et al.*, 2010; Hörtenhuber *et al.*, 2010; Gabrielle & Gagnaire, 2008) is primarily that the time perspective of the CO_2 emission and the decay in the atmosphere is taken into account by including the Bern Carbon Cycle Model (IPCC, 2007) and that any time perspective can be

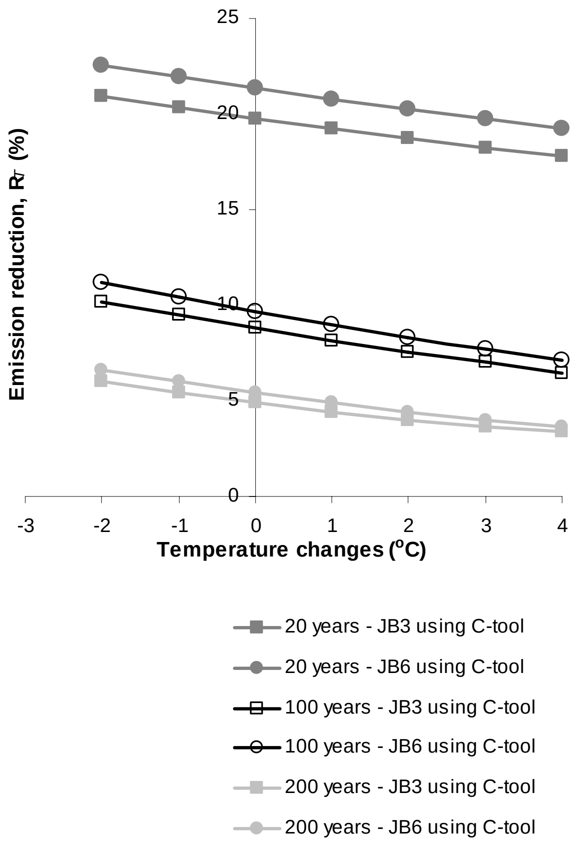

Figure 9. Emission reduction, R_T, of Example I (when leaving 1 tonne of straw C in the field instead of using it for bioenergy) as affected by temperature changes relative to the mean air temperature of 7.7°C.

chosen such as 20, 30, 100 or 200 years. Furthermore, the suggested methodology considers a soil depth of 0-100 cm enabling the method to capture a more precise estimate of the soil C changes in the soil depth.

The results derived from the suggested approach are comparable to the results of the IPCC approach (IPCC, 2006) when the soil C changes are estimated in a 20-year perspective. The slightly higher values in the suggested approach (Example I: 332 vs. 262 kg C/ha year; Example II: 143 vs. 100 kg C/ha year) can partly be ascribed to the deeper soil horizon considered in the suggested approach (0-100 cm) than in the IPCC approach (0-30 cm). However, as shown in Table 4 and 5, the chosen time perspective is crucial to the results. In the suggested methodology, the time perspective is not fixed to be 20 years as in the IPCC approach; since it can be discussed whether a 20 years time perspective is the more appropriate to use (Fearnside, 2002) when the global warming potential is normally estimated on the basis of a 100 year time perspective (IPCC, 2007). Furthermore, soil C changes towards a new steady state can take more than 20 years, as illustrated in Figure 8, which is in accordance with long term field studies in Northern Europe (Jenkinson & Rayner, 1977;

Jenkinson, 1990; Kirchmann et al., 1994), where the changes by contrasting residue management and manure application continue for at least 50 – 100 years. Assuming that a new steady state is reached after 20 years is perhaps better suited for the tropics, where soil C changes are faster due to the high temperature. The political motivations for mitigation options for utilising soil C sequestration is strongly affected by the chosen time horizon.

The suggested methodology, including the combination of soil and atmospheric C decay processes through the implementation of the Bern Carbon Cycle Model, gives more precise and dynamic estimates in time and soil depths of the global warming consequences of different treatments of C resources in agricultural systems.

The main challenge for using this method, is estimating the C deficit between the basis scenario and the new practice. The application of the Bern Carbon Cycle Model is straightforward so using equation 4 depends mainly on an appropriate soil C model (such as e.g. C-TOOL and RothC) to estimate the turnover of C in the specific site, dependent on e.g. soil properties and climate data.

The assumption that the soil C turnover is independent on the C content of the soil is used by the majority of soil C models (Paustian et al., 1997) including the C-TOOL model. It should be mentioned that this assumption is challenged, see e.g. Six et al. (2002), Steward et al. (2008) and Kimetu et al. (2009). These studies suggest a saturation effect by high levels of soil C. Both the latter studies are based on comparisons between soils of different origin though. The Steward et al. (2008) study compared soil from respectively the A and C horizon and the Kimetu et al. (2009) study compared forest soil with agricultural soil. The assumption of saturation at some point seems plausible, but arguably the comprehensive study demonstrating this effect and its quantitative implications on fully comparable soils is still lacking. Within the changes caused by agricultural practices this assumption should be acceptable.

4.2 Comparison with other studies

The estimated C sequestration due to soil storage of one tonne straw C in the present paper of 54, 97 and 213 kg C per tonne straw C for a time perspective of 200, 100 and 20 years respectively, is comparable to the estimate of Gabrielle & Gagnaire (2008), who estimated a C sequestration of 0.05 to 0.10 t C per t added straw, which corresponds to 111 to 222 kg C per t straw C. Gabrielle & Gagnaire (2008) used a time perspective of 30 years. The main difference to the present study was that the CO_2 decay in the atmosphere was not considered in Gabrielle & Gagnaire (2008).

Several studies have showed an increased soil carbon sequestration under organic as compared to conventional farming practices (Müller-Lindenlauf, 2009). The estimated C sequestration due to conversion from conventional practices in the present case study is mainly caused by using crop residues as soil amendment instead of burning them, but the additional C input to the organic system could also be caused by perennial or green manure crops, which is the case in Halberg *et al.* (2010) or compost or animal manure.

In the present study, the effect of converting from conventional to organic management practices is estimated to cause a C sequestration of 32-143 kg C per hectare (Table 5), depending on the time perspective, which corresponds to an emission reduction of 117-524 kg CO_2 per hectare per year or 42-188 kg CO_2 per tonne organic soybeans produced, compared to conventional practice (Table 5). This C sequestration is comparable to the difference of 600 kg CO_2 per hectare per year used by Hörtenhuber *et al.* (2010) based on a 20-year time perspective. The estimated C sequestration of organic compared to conventional pig production systems by Halberg *et al.* (2010) was higher than in the present study, mainly due to a shorter time perspective and a much more C enriched system, mainly due to perennial crops and a higher crop residue recycling. The results indicate that the inclusion of soil C changes in comparable LCA's of organic and conventional agricultural plant products will widen the difference between greenhouse gas emissions per crop unit of organic and conventional plant products. However, the time perspective chosen is crucial to the estimated effect of organic farming on carbon sequestration and a time perspective of 100 years will reduce the estimated carbon sequestration per year compared to a shorter time perspective of e.g. 20 years, used in the abovementioned studies.

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Appendices

Table 8.1 Literature review of GHG emissions per kg organic versus conventional agricultural product at farm gate.

Product	GHG emissions per kg product at farm gate (kg CO_2 eq./kg)		Ratio of organic to conventional	References
	CONVENTIONAL	ORGANIC		
MEAT				
Beef, UK	16.0	19.2	1.2	Williams et al. (2006)
Suckler beef, live weight	13.0	11.1	0.9	Casey & Holden (2006)
Beef, dairy cattle, Germany	4.8	3.1	0.6	Hirschfeld et al. (2008)
Fattening bull, dairy, Germany	7.9	11.0	1.4	Hirschfeld et al. (2008)
Pig meat, DK	2.7	2.5	0.9	Halberg et al. (2010)
Pig meat, UK	6.4	5.8	0.9	Williams et al. (2006)
Pig meat, Germany	2.72	1.7	0.6	Hirschfeld et al. (2008)
Poultry, UK	4.6	6.9	1.5	Williams et al. (2006)
Sheep, UK	17.0	10.2	0.6	Williams et al. (2006)
Eggs, UK	5.5	7.2	1.3	Williams et al. (2006)
DAIRY				
Milk, Germany	0.70	0.63	0.9	Hirschfeld et al. (2008)
Milk, The Netherlands	1.4[a]	1.5[a]	1.1	Thomassen et al. (2008)
Milk, UK	1.1	1.3	1.2	Williams et al. (2006)
Milk, Sweden	1.0[b]	0.9[b]	0.9	Cederberg & Mattsson (2000)
FRUIT/VEGETABLES				
Oranges, Brazil	0.11	0.08	0.8	Knudsen et al. (submitted)
Leeks, Belgium	0.094	0.044	0.5	de Backer et al. (2009)
Potatoes, UK	0.24	0.2	0.9	Williams et al. (2006)
Carrot, DK	0.12	0.21	1.7	Halberg & Dalgaard (2006)
Tomatoes, greenhouse, DK	3.45	4.96	1.4	Halberg & Dalgaard (2006)
AGRICULTURAL CROPS				
Soybeans, China	0.26	0.16	0.6	Knudsen et al. (2010)
Wheat, USA	0.28	0.24	0.8	Meisterling et al. (2009)
Wheat, Germany	0.37	0.14	0.4	Hirschfeld et al. (2008)
Bread wheat, UK	0.80	0.80	1.0	Williams et al. (2006)
Wheat, DK	0.71	0.28	0.4	LCAfood (2003)
Oilseed rape, UK	1.70	1.7	1.0	Williams et al. (2006)
Oilseed rape, DK	1.51	0.95	0.6	LCAfood (2003)
Winter barley, DK	0.62	0.32	0.5	LCAfood (2003)
Spring barley, DK	0.65	0.4	0.6	LCAfood (2003)
Oat, DK	0.57	0.39	0.7	LCAfood (2003)
Rye, DK	0.72	0.62	0.9	LCAfood (2003)

a. per kg fat protein corrected milk (FPCM)
b. per kg energy corrected milk (ECM)

Table 8.2 Literature review of the relative importance of transport for the greenhouse gas (GHG) emissions per kg agricultural product – not organic except from orange juice and soybeans.

Product	Country of origin and consumption	Primary transport mode	GHG emissions per kg product at RDC[a] (kg CO_2 eq./kg)	% from transport of GHG emissions per kg product at RDC[a]	Reference
MEAT					
Beef	Brazil to UK	Sea	32.15	1	Williams et al. (2008)
Lamb	New Zealand to UK	Sea	11.56	6	Williams et al. (2008)
Lamb	New Zealand to UK	Sea	0.69	18	Saunders et al. (2006)
Poulty	Brazil to UK	Sea	2.57	15	Williams et al. (2008)
FRUIT					
Apples	New Zealand to UK	Sea	0.87	72	Williams et al. (2008)
Apples	Chile to UK	Sea	-	72	Sim et al. (2007)
Apples	Brazil to UK	Sea	-	90	Sim et al. (2007)
Apples	New Zealand to UK	Sea	0.19	68	Saunders et al. (2006)
Strawberries	Spain to UK	Road	0.90	34	Williams et al. (2008)
JUICE					
Organic orange juice	Brazil to Denmark	Sea / road	0.42	57	PAPER III: Knudsen et al. (submitted)
VEGETABLES					
Potatoes	Israel to UK	Sea	0.48	47	Williams et al. (2008)
Onions	New Zealand to UK	Sea	0.19	68	Saunders et al. (2006)
Broccoli	Spain to UK	Road	0.6-1.2[b]	50-25[b]	Milà i Canals et al. (2008)
Lettuce	Spain to UK	Road	0.60	70	Milà i Canals et al. (2008)
Lettuce	Spain to UK	Road	0.45	43	Hospido et al. (2009)
Lettuce	Uganda to UK	Air	10.00	98	Milà i Canals et al. (2008)
Green beans	Uganda to UK	Air	11.00	91	Milà i Canals et al. (2008)
Green beans	Kenya to UK	Air	11.00	91	Milà i Canals et al. (2008)
Runner beans	Kenya to UK	Air	-	89	Sim et al. (2007)
Runner beans	Guatemala to UK	Air	-	91	Sim et al. (2007)
Watercress	USA to UK	Air	-	89	Sim et al. (2007)
FODDER					
Organic soybeans	China to Denmark	Sea	0.43	51	PAPER II: Knudsen et al. (2010)

a Regional Distribution Centre
b Including home transport by consumer

Table 8.3 A sample of import organic agricultural products to Denmark and the estimated actual contribution of transport to the greenhouse gas (GHG) emissions.

Examples of organic products[a]	Transport mode	Distance (km)[c]	Estimated transport contribution to GHG emissions[d] at RDC[e] (kg CO_2 eq./kg product)
EUROPE			
Asparagus plus faba beans from Germany (Kassel)	Road[b]	646	0.10
Red pepper, white/red cabbage, muchrooms from The Netherlands[1]	Road[b]	792	0.12
Wine from France (Paris)	Road[b]	1226	0.18
Kiwi, apple, orange, lemons, tomato, salad, fennel plus soybeans, soybean cake, rapeseed and rapeseed cakes from Italy (Rome)[1]	Road[b]	2053	0.31
Cucumber, salad, emon, melon, courgette, cabbage, aubergine, leaf sellery from Spain (Madrid)	Road[b]	2493	0.37
AFRICA			
Oranges, potatoes, spring onion, leek, rice, grapes from Egypt (via Rotterdam)	Sea / road[b]	204 (road) 5850 (sea) 834 (road)	0.21
Grapes, oranges and lemons from South Africa (via Rotterdam)	Sea / road[b]	785 (road) 11414 (sea) 834 (road)	0.35
SOUTH AMERICA			
Orange juice concentrate, cane sugar and coffee from Brazil (via Rotterdam)	Sea / road[b]	320 (road) 10040 (sea) 834 (road)	0.26
Apple, pear, plum, garlic, onion from Argentina (via Rotterdam)	Rail /sea / road[b]	1000 (rail) 11744 (sea) 834 (road)	0.27
Banana from Dom. Republic (via Rotterdam)	Sea / road[b]	7488 (sea) 834 (road)	0.19
Avocado from Mexico	Sea / road[b]	350 (road) 9373 (sea) 834 (road)	0.26
ASIA			
Tea and rice from India	Sea / road[b]	348 (road) 11888 (sea) 834 (road)	0.28
Soybean, sunflower and pumpkin seeds, linseeds, buckwheat, beans, tea, ginger, garlic, frozen strawberries and vegetables from China	Sea / road[b]	400 (road) 20044 (sea) 834 (road)	0.37
Grape fruit, tomato, avocado, red pepper from Israel	Sea / road[b]	100 (road) 6233 (sea) 834 (road)	0.20
OCEANIA			
Kiwi and apples from New Zealand	Sea / road[b]	150 (road) 20935(sea) 834 (road)	0.34

a. Based on information from retaliers, merchants, StatBank Denmark and observations in supermarkets.
b. Assuming tranportation by a 40t truck, even though transport in some cases could be by train, using data from Ecoinvent Centre (2007)
c. Distances measured from the capital cities or centre of the country to Aarhus using google maps (www.maps.google.dk). Sea transport via the nearest international sea port to Rotterdam.(using www.distances.com).
d. Only global warming potential (GWP) related to the transportation of products is calculated using Ecoinvent Centre (2007) and EDIP97 method updated with IPCC 2007 standards (IPCC, 2007). 0.009 kg CO_2 equivalents per tkm are used for sea transport and 0.15 kg are used for 40t truck transport. It is assumed that the required conditions during transport are the same, thus additional energy for cooled or refrigerated transport are not included.
e. Retail Distribution Centre, Aarhus.

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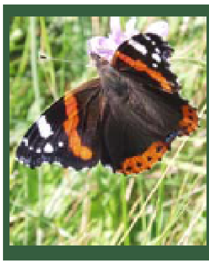


How to include on-farm biodiversity in LCA on food?

Marie Trydeman Knudsen[1] & Niels Halberg[2]
[1]Faculty of Life Sciences, University of Copenhagen (mariet.knudsen@agrcsi.dk). [2]Faculty of Agricultural Science, University of Aarhus

Objective To assess current approaches to include biodiversity aspects in Life Cycle Assessment and search for an approach to include biodiversity aspects in LCA on food.

Introduction

Life Cycle Assessments (LCA) of food and agriculture generally include potential effects on global warming, eutrophication, ecotoxocity and acidification some of which again affect biodiversity. However, LCA most often does not include specific indicators of the product's or agricultural system's impact (negative or positive) on biodiversity. Using LCA methodology on agricultural products makes it highly relevant to assess the impacts of land use. Some LCA's include a simple category of land use. This is sometimes interpreted as "nature occupation". However, if this is the only impact category addressing land use related biodiversity, the LCA cannot distinguish between different forms of agricultural systems, which may differ in their biodiversity impact (e.g. organic versus conventional products). Biologists as well as policy makers consider some agricultural land use, such as grazing semi-natural grasslands, is actually considered beneficial for biodiversity preservation.

Is land use always negative for biodiversity?

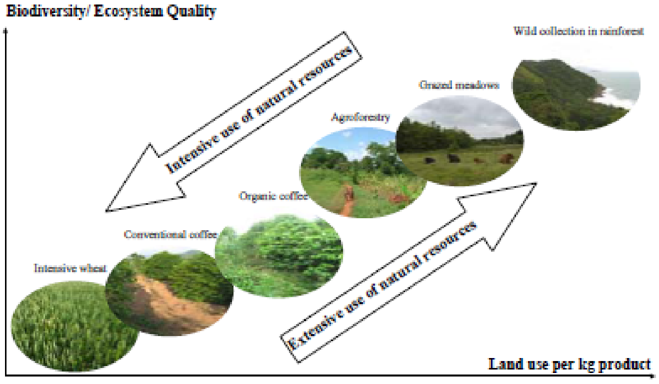
Does untouched nature automatically have higher ecosystem quality?

How can LCA account for biodiversity preservation effects of some agricultural systems?

Current approaches to include biodiversity in LCA

Indicators of biodiversity	Suggested by
Land use (ha year per kg product)	Current common LCA approach
Intactness, integrity, fragmentation, endemism, scarcity	Milà i Canals et al. (2006)
Indicators based on ecosystem thermodynamics	Wagendorp et al. (2006)
The biotope method (four categories of biotopes)	Kyläkorpi et al. (2005)
Species richness indicator (SRI) and ecosystem rarity indicator (ERI)	Vogtländer et al. (2004)
The Hemeroby Concept (scale of use intensity, %)	Brentrup et al. (2002)
Several indicators especially on farmers uncultivated area	Schenck (2001)
Species richness (SR), Inherent ecosystem scarcity (ES), Ecosystem vulnerability (EV) – combined in Quality ($Q_{biodiversity}$)	Weidema & Lindeijer (2001)
Qualitative descriptions only	Mattsson et al. (2000)
Species-pool effect potentials (SPEP)	Köllner (2000)
Species diversity of vascular plants (S)	Lindeijer (2000)
Area, number of listed rare species, number of species, number of individuals	Cowell (1998)

Land use per kg product can be difficult to interpret



How to select an appropriate indicator of biodiversity ? – to account for those differences?

Conventional coffee plantation, Brazil - with bare soil

Organic coffee plantation, Brazil - with wild vegetation

The selection of indicators

- Using DPSIR approach (pressure indicator)
- Using the most important factors affecting biodiversity, instead of direct estimations of e.g. species diversity
- Operational approach
- Using questionnaire for farmers instead of measurements (as suggested by Schenck (2001))

Suggestion for indicators

Several indicators? ⟶ A single indicator?

% small biotopes
% weeds
% unsprayed area

Ecosystem quality

As suggested by Schenck (2001) — As suggested by Brentrup et al. (2002)

Example of using several indicators

	Dairy farms in Denmark (1994-97)	
	Organic	Conventional
Land use per kg milk (m^2 year)*	2.1	1.4
% small biotopes	4	4
% weeds in small grains	10	1
% unsprayed area	100	35

*LCA Food Database (www.lcafood.dk) — Source: Halberg et al. (1999)

Conclusion Land use in food production systems can have both positive and negative impacts on biodiversity compared to leaving the land untouched by humans. Simple, operational indicators to account for the different impacts on biodiversity in food production systems could take the point of departure in the most important factors affecting biodiversity (easy obtainable pressure indicators) instead of estimating e.g. species diversity directly.

Contribution to the LCA Food Conference, 25-26 April 2007, Gothenburg, Sweden