Input-output analysis of the networks of production, consumption and environmental destruction in Finland

Tuomas J. Mattila





DOCTORAL DISSERTATIONS

Input-output analysis of the networks of production, consumption and environmental destruction in Finland

Tuomas J. Mattila

A doctoral dissertation completed for the degree of Doctor of Science (Technology) to be defended, with the permission of the Aalto University School of Science, at a public examination held at the lecture hall E of the Aalto University School of Science on 12 September 2013 at 12.

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Aalto University publication series **DOCTORAL DISSERTATIONS** 124/2013

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ISBN 978-952-60-5279-3 (printed) ISBN 978-952-60-5280-9 (pdf) ISSN-L 1799-4934 ISSN 1799-4934 (printed) ISSN 1799-4942 (pdf) http://urn.fi/URN:ISBN:978-952-60-5280-9

Unigrafia Oy Helsinki 2013

Finland



441 697 Printed matter



Author

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Name of the doctoral dissertation Input-output analysis of the networks of production, consumption and environmental destruction in Finland

Publisher Aalto School of Science

Unit Department of Mathematics and Systems Analysis

 $\textbf{Series} \ \text{Aalto University publication series DOCTORAL DISSERTATIONS} 124/2013$

 $\textbf{Field of research} \ \ Systems \ and \ Operations \ Research$

Permission to publish	granted (date) 24 April	2013	Language	English
Monograph	🛛 Article dissertati	on (summary	+ original articles)	

Abstract

Modern systems of production and consumption are complex and global. Supply networks cross continents, linking the consumption in Finland to land use in the Latin-America and South-East Asia. Economic input-output analysis was originally developed to track the production networks of a single country, but it has recently been applied to include environmental impacts over several regions. Current environmentally extended input-output (EEIO) models connect consumption, production and environmental impacts into a transparent system of equations, which can be used to examine the direct and indirect effects of different economic activities. The combination of EEIO with life cycle assessment (LCA) has allowed the industrial ecological analysis of various environmental footprints (for example ecological, carbon and water). However these footprints capture only a narrow share of the overall sustainability. The aim of this study was to broaden the scope of previous footprint analyses by focusing on new environmental impacts such as biodiversity, land use and ecotoxic effects. Impact assessment models for each impact category were connected to the general EEIO framework. This allowed the detailed analysis (structural path, structural decomposition and sensitivity analysis) to find the key components causing environmental destruction through production and consumption. These identified key components can be used to model and manage the environmental issues from a whole system perspective.

Keywords input-output analysis, ecological footprint, ecological economics, life cycle assessment

ISBN (printed) 978-953	2-60-5279-3	ISBN (pdf) 978-9	952-60-5280-9	
ISSN-L 1799-4934	ISSN (printed) 1799-4934	ISSN (pdf)	1799-4942
Location of publisher	Helsinki	Location of printing	Helsinki	Year 2013
Pages 120		urn http://urn.fi/UF	N:ISBN:978-952	-60-5280-9



Tekijä

Tuomas J. Mattila

Väitöskirjan nimi

Suomen tuotannon ja kulutuksen aiheuttamien ympäristövaikutusten panos-tuotos analyysi Julkaisija Perustieteiden korkeakoulu

Yksikkö Matematiikan ja systeemianalyysin laitos

Sarja Aalto University publication series DOCTORAL DISSERTATIONS 124/2013

Tutkimusala Systeemi- ja operaatiotutkimus

Käsikirjoituksen pvm	26.2.2013	Väitöspäivä 12.9.2013		
Julkaisuluvan myöntä	mispäivä 24.4.2013	Kieli Englanti		
Monografia	🛛 Yhdistelmävä	itöskirja (yhteenveto-osa + erillisartikkelit)		

Tiivistelmä

Nykyinen tuotannon ja kulutuksen järjestelmä on globaali ja monimutkainen. Suomessa kulutettujen tuotteiden hankintaketjut ulottuvat eri mantereille, aiheuttaen vaikutuksia esimerkiksi Etelä-Amerikan ja Kaakkois-Aasian maankäytölle. Panos-tuotos analyysi kehitettiin alun perin yksittäisen valtion talouden tuotannon rakenteiden tarkasteluun, mutta sitä on viime aikoina laajennettu käsittämään useampia alueita ja myös ympäristövaikutuksia. Nykyaikaiset ympäristölaajennetut panos-tuotos mallit (environmentally extended inputoutput, EEIO) kytkevät toisiinsa kulutuksen, tuotannon ja ympäristövaikutukset läpinäkyväksi mallikehikoksi, jolloin tätä kokonaisuutta voidaan analysoida systemaattisesti. Eri taloudellisten toimintojen suoria ja epäsuoria vaikutuksia voidaan tarkastella samalla läpinäkyvällä kehikolla. Viime aikoina ympäristölaajennettuja panos-tuotosmalleja on kytketty elinkaariarvioinnin vaikutusarviointimenetelmiin (*life cycle assessment*, LCA). Aikaansaaduilla teollisen ekologian malleilla on tarkasteltu laajasti vaikutuksia erilaisiin ympäristövaikutuksiin. Yleensä tarkastelu on kuitenkin rajoittunut erilaisiin jalanjälkilaskelmiin (hiili-, vesi- ja ekologinen jalanjälki). Tämän tutkimuksen tarkoituksena oli laajentaa näkökulmaa kattamaan myös harvemmin tutkittuja, mutta tärkeiksi koettuja vaikutuksia, kuten biodiversiteettivahingot, maankäyttö ja haitallisten aineiden vaikutukset. Työssä kytkettiin vaikutusarviointimalleja panos-tuotos malliin ja tulosten avulla purettiin kunkin vaikutusluokan kannalta keskeisimmät taloudelliset prosessit ja tuoteketjut. Näitä keskeisiksi tunnistettuja tekijöitä voidaan hyödyntää, kun haetaan systeemitason keinoja ympäristön ja talouden kehityksen ohjaamiseksi.

Avainsanat panos-tuotos analyysi, ekologinen jalanjälki, ympäristötaloustiede, elinkaariarviointi

ISBN (painettu) 978-952-6	0-5279-3	ISBN (pdf) 978-952-60-5280-9			
ISSN-L 1799-4934	ISSN (painettu)	1799-4934	ISSN (pdf) 1799-4942		
Julkaisupaikka Helsinki	Pain	opaikka Helsinki	Vuosi 2013		
Sivumäärä 120	urn	http://urn.fi/URN	I:ISBN:978-952-60-5280-9		

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Publications

The dissertation consists of the present summary article and the following papers:

- Mattila, T., Leskinen, P., Mäenpää, I., and Seppälä J., 2011. An Environmentally Extended Input-Output Analysis to Support Sustainable Use of Forest Resources. *The Open Forest Science Journal* 4 (1): 15–23.
- Mattila, T., Seppälä, J., Nissinen, A. and Mäenpää, I. 2011. Land use impacts of industries and products in the Finnish economy: A comparison of three indicators. *Biomass and Bioenergy* 35 (12): 4781–4787.
- 3. Mattila, T., 2012. Any sustainable decoupling in the Finnish economy? A comparison of the pathways and sensitivities of GDP and ecological footprint 2002–2005. *Ecological Indicators* 16 (1): 128–134.
- Mattila, T., Verta, M. and Seppälä, J., 2011. Comparing priority setting in integrated hazardous substance assessment and in life cycle impact assessment. *The International Journal of Life Cycle Assessment* 16 (8): 788–794.
- Mattila, T., Koskela, S., Seppälä, J. and Mäenpää, I., 2013. Sensitivity analysis of environmentally extended input-output models as a tool for building scenarios of sustainable development. *Ecological Economics* 86: 148-155.

Contributions of the author

The author was responsible for the design of the study, computation of results and writing in all of the studies. The idea for paper I came from professor Pekka Leskinen, for paper II from Dr. Ari Nissinen. Professor Ilmo Mäenpää constructed the detailed economic input-output table and the environmental extensions for waste production and climate emissions. The author constructed the environmental extensions for land use, ecological footprint and hazardous emissions.

Acknowledgements

I'm grateful to my colleagues at the Finnish Environment Insitute SYKE for all the discussions, the Ministry of Environment for funding the projects in which the articles were written, my co-authors for a wealth of ideas and healthy critique, the anonymous peer reviewers for their valuable comments, the supervising professor Raimo Hämäläinen for ideas for bringing the work forward, and to my friends and family for the patience in listening to and tolerating an absent minded scientist.

List of abbreviations and definitions

GDP gross domestic product GWP global warming potential EF ecological footprint LCA life cycle assessment LCC life cycle costing LCI life cycle inventory LCIA life cycle impact assessment HANPP human appropriation of primary production LCSA life cycle sustainability assessment MCDA multiple criteria decision analysis MRIO multiple region input-output model NPP net primary production EDP ecosystem degradation potential IOA input-output analysis EEIO environmentally extended input-output SA sustainability assessment MRIO multiple region input-output model MCDA multiple criteria decision analysis SNA system of national accounts SDA structural decomposition analysis SLCA social life cycle assessment SPA structural path analysis SPD structural path decomposition TBT tributyltin DALY disability adjusted life year expectancy

1. Introduction

1.1 General background

Imagine throwing a coin into a still pond. As the coin lands it pulls the water surface down. Surface tension pulls the water back up, starting a wave front. As the wave collides with the boundaries of the pool, it rebounds in new shapes, combining with the old wave, forming a complex mosaic of wave patterns.

The complex shapes of ripples in a pool of water are driven by simple natural laws and the structure of the pool boundaries. Similarly the complex cause-effect chains of the modern economy can be evaluated, if the structure behind the mosaic is known. Economic input-output analysis (IOA) is a tool for assessing the networks and cause-effect chains of the economy, connecting consumption to production and finally to environmental degradation and resource extraction.

Modern supply chains have become increasingly complex and global. For example most industries in developed countries are purchasing business services from India (Timmer, 2012), therefore connecting consumption in developed countries to electricity production, resource use and pollution in developing countries. The increasing complexity posed a challenge for industrial ecology, which investigates the interrelations between humans and the environment. Therefore IOA and especially environmentally extended input-output (EEIO) models have rapidly become one of the main research methods in industrial ecology (Suh, 2009). Their use has revolutionized the understanding about system boundaries in global supply chains (Suh et al., 2004), patterns of production and consumption (Lenzen et al., 2007; Peters, 2008) and about the complexities of the life cycles of most modern products (Lenzen, 2003; Lenzen et al., 2012).

EEIO is closely linked to life cycle thinking, a practice which emerged in the 1960s with energy analysis and quickly progressed to include various aspects of environmental sustainability (Guinée et al., 2011) . Life cycle

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assessment (LCA) follows the products from the cradle-to-the grave, quantifying the resource flows needed to manufacture a service or a product and to dispose of it safely. It can be easily understood that collecting a full LCA inventory is a tremendous effort, and possible only by building on previous LCAs. Even in full LCAs, important flows are often overlooked due to the lack of data (Suh et al., 2004). The combination of EEIO and LCA in the last decade has mainly focused on using the EEIO to obtain inventory data to supplement a process-based LCA inventory (Suh and Huppes, 2005). However the opportunities of using LCA to improve the impact assessment in EEIO have not been explored to a similar extent.

Most EEIO studies have focused on climate change. However climate change is by no means the most severe threat to humanity and ecosystems. Considerable problems persist in nutrient cycles, land use change and biodiversity, and the ecotoxic pressure is largely unquantified (Rockström et al., 2009). In comparison, LCA has been attempting to include these aspects for quite a while with detailed impact assessment models (LCIA models) available for land use related impacts and ecotoxicity (Rosenbaum et al., 2008; Finnveden et al., 2009; Mattila et al., 2012).

This dissertation analyzes the networks of production and consumption associated with the Finnish economy by combining environmentally extended input-output analysis with life cycle impact assessment. The main focus has been on land use, biodiversity and ecotoxicity, since few EEIO studies have been done on those impact categories. By combining the two research fields, several benefits are obtained. First, new insight is given to environmental problems by looking at them through the models of LCIA. Second, when the LCIA model is applied on a national scale, the output can be compared to observed impacts and policy responses. This dialogue between modeling and practice makes it possible to develop the models as well as increase understanding about the sustainability problems.

1.2 Research approach

As a whole the articles try to make sense of the complex network of consumption and production, which links consumer purchases to global environmental impacts. This is done by combining analysis from different environmental impacts and tools to a synthesis of the main contributors of change. The main research problem is to discover, whether the whole ecological crisis can be simplified to a limited set of subcomponents which can be understood and manipulated. This research problem is approached through the following research questions:

- 1) How can the sustainability of industries be described and compared in a concise form?
- 2) What causes biodiversity and land use impacts in Finland?
- 3) Are the mechanisms of economic growth and environmental degradation the same?
- 4) What kind of economic subsystems cause toxic emissions?
- 5) Can the LCIA models be trusted in hazardous substance management?
- 6) To what extent the environment-economy model system can be simplified and still maintain predictive power?

1.3 Research process and dissertation structure

The dissertation proceeds from the general to the more specific problems. The first article demonstrates the use of input-output analysis in preliminary sustainability assessment of industries. This is achieved by experimenting with a simplified and aggregated version of the ENVIMAT EEIO model. Only a few environmental, economic and social impacts are included and the focus is strongly on the forest industries.

The second paper digs deeper into the problems of evaluating land use impacts to biodiversity. Land use statistics and three LCIA impact assessment models were integrated to the disaggregated ENVIMAT. The third paper follows on the theme of land use, but looks at the economic mechanisms which drive both biological resource exploitation (ecological footprint) and gross domestic product in the economy. This is also the first paper in this dissertation which sheds some light on the internal structures of the Finnish economy through computational methods. Some mechanisms of change were also identified.

The fourth paper utilizes the same structural analysis techniques presented in the third paper, but applies them to ecotoxicological and human toxic impact assessment models. The chemical pollutants of the Finnish emission inventory were prioritized based on their calculated toxic impacts. In addition ENVIMAT was used to identify the main economic processes which are responsible for the toxic pressure on man and wildlife. The aim of this study (in the scope of the dissertation) was to provide a contrast to the land use and climate change impacts analyzed in the other

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papers. Since hazardous substances are not widespread in the economy, it was speculated that their networks would also be narrower.

In the fifth paper the experiences on analyzing the environment-economy systems interactions are brought together and a method for making sustainability scenarios is proposed. The method is based on identifying the main structural components causing the impacts of concern and then identifying ways of changing those components.

Finally the limitations of the approach are discussed, related mainly to the lack of dynamic feedback and the usefulness of history oriented static indicators in initiating change towards sustainable development.

2.1 Background

The dissertation lies on the foundation of two diverse system analytical tools, which are now becoming together. Economic input output analysis considers macroeconomic systems by looking at the interactions between industries. Life cycle assessment looks at the total environmental impacts in a supply chain from cradle-to-grave (from raw material acquisition to manufacturing, use and recycling).

The automated collection of inventory data has been the main application of combined input-output and life cycle assessment studies. LCA has suffered from the difficulties of collecting the necessary inventory data for the emissions and resources used in various stages of the supply chain. As the resolution of input-output databases has improved, this issue is left in the past. Multiple region input-output (MRIO) tables can quantify the networks of production and consumption very rapidly, beginning a new phase in life cycle assessment, where hybrid-LCA techniques are used to make more comprehensive assessments much faster. At the same time, the quantitative tools made for economic network analysis can be used to evaluate the accumulation of environmental impacts throughout the supply chain.

Life cycle assessment has however much to give to input-output analysis in impact assessment and interpretation. Throughout its history LCA has developed a consistent methodology for evaluating and comparing the overall environmental impacts integrated over global locations and over time. The methodology is rooted in multiple criteria decision analysis (MCDA), allowing consideration of tradeoffs between environmental impact categories (i.e. is 600 m² of primary rainforest converted to arable land worse than increase of climate radiative forcing by 10 t CO_2 eq.). LCA is therefore well suitable for diversifying the scope of input-output analysis, which has traditionally focused on only very few environmental indicators.

Recent applications show promise in combining these two aspects (Hertwich, 2010).

The multiple criteria approach links LCA with the broader scope of sustainability assessment and measuring development. Several indicators have been developed, ranging from the single indicator scores of GDP and ecological footprint to collated indexes (such as the sustainable societies index or the happy planet index). The combination of the three methodologies can offer new views to the sustainability crisis facing humanity.



Figure 1. The methodological foundation for this dissertation is in moving the boundaries of sustainability assessment towards the detailed methods used in input-output analysis and life cycle assessment.

In the following chapters, details on the methodologies used are given and illustrated.

2.2 Input output analysis

"And perhaps this art alone can show the links and chains by which one business hangs upon another and the dependence which all our dealings have upon each other" Charles Davenant, 1699 (Pyatt, 2000)

"Partial analysis cannot provide a sufficiently broad basis for fundamental understanding." W. Leontief, autobiography for the Nobel Foundation

Input-output analysis studies the interdependencies between industries and consumers. It is by no means a new idea; on the contrary, similar work began as early as in the 17th century with the Mercantilists and Physiocrats. Quesnay even compiled an input-output table (Tableau Economique) to describe the circular flow of goods in the economy in 1758. However it required more than a century, before the ideas were put into analytical form allowing further development and testing, eventually resulting in general equilibrium theory and input-output analysis. (Miller and Blair, 2009) When computers became available for research use, Wassily Leontief put the economic theory into practice by applying it to the US economy (Leontief, 1936). The learning process initiated by the application simplified the method and began the widespread use of input-output economics. Currently detailed input-output tables are compiled for most countries as a part of their system of national accounts (SNA)(Eurostat, 2010; OECD, 2010). In addition standardized practices for compiling and applying the tables have been published (Eurostat, 2008). This chapter describes the basic derivation and application of input-output analysis as well as its environmental extensions and the analytical tools applied in this thesis. The purpose is to familiarize the reader with the techniques and assumptions of the modeling framework.

2.2.1 Basics of input-output analysis

The main research topic of economic input-output analysis is the relationship between the scale of production output (\mathbf{x}) and the final demand of products (**f**). The analysis begins with a simple balance of products, which are used in intermediate or final use:

 $\begin{aligned} \mathbf{x} &= \mathbf{A}\mathbf{x} + \mathbf{f} & (1a), \\ \text{where} & \mathbf{x} &= \text{total output (industry by 1) [MC]} \\ \mathbf{A} &= \text{intermediate use matrix (industry-by-industry) [MC/MC]} \\ & \mathbf{f} &= \text{final demand (industry-by-1) [MC]} \end{aligned}$

Matrix **A** describes the amount of products needed from other (and from the producing) industries for the production of one unit of product. Also known as a technology matrix, it is obtained by dividing the purchases of each industry from other industries by their corresponding total output. The column sum of each row in **A** represents the purchases from other industries needed to supply one unit of product and is always less than one. The difference between one and the column sum is then the value added for that industry.

The eq. (1a) can be re-arranged to give the relationship between total production and final demand:

 $\mathbf{x} = (\mathbf{I} - \mathbf{A})^{-1} \mathbf{f}$ (1b),

where **I** = identity matrix (industry-by-industry)

 $(I-A)^{-1}$ = Leontief inverse (industry-by-industry) $[M \in /M \in]$.

Each column of the Leontief inverse matrix describes the overall economic activity resulting in the economy following the production of one unit of monetary product in a given sector. The column sums are also known as (backward) multiplier effects and are used for example to identify the key sectors of an economy (Oosterhaven and Jan Oosterhaven, 2004).

Equation (1) is known as the input-output quantity model, however it has a dual price model:

 $\mathbf{p} = (\mathbf{I} - \mathbf{A})^{-1} \mathbf{v}$ (2), where \mathbf{p} = unit prices (industry-by-1) \mathbf{v} = value added (industry-by-1)

The price model therefore allows the estimation of price changes following changes in value added or production technology. Combined with the quantity model, the value added can be used to estimate the changes to gross domestic product (GDP) from changes in technology or demand:

 $k = \mathbf{v}^{T} (\mathbf{I} - \mathbf{A})^{-1} \mathbf{f}$ (3), where k = the gross domestic product [M€].

Equation (3) not only allows the connection of total gross domestic product to total final demand. If final demand is entered as a diagonal matrix, equation (3) yields the value added for each category of final demand and can be used to quantify, where demand would cause the most of value added. This equation is extended in introducing environmental footprints for demand categories.

Although the **A** matrix constitutes the core of the input-output model, it is commonly not known, but must be calculated from the collected national accounts (although most national accounts report a finished technology matrix as well). The accounts contain rectangular make and use tables (the products made and used by various industries). These can be denoted as **U** and **V**. In order to make a symmetrical input-output table, assumptions about the production technologies and consumption structures need to be made. Nine possible alternative assumptions with their strengths and weaknesses have been identified (ten Raa and Rueda-Cantuche, 2003). The most commonly applied assumptions are the product technology model and the fixed product sales structure model (Eurostat, 2008). The first converts product-by-industry tables into symmetrical product-by-product tables by assuming that products have their own unique technologies, irrespective of the industry where they are produced. (i.e. tourist accommodation requires the same inputs, whether its produced by farms or hotels). The second alternative assumes that each product has the same sales structure, irrespective of industry where they are produced (i.e. all buyers of tourist accommodation buy them from all producing industries in respect to the market share).

Expressed in equations, the product technology assumption obtains the technology matrix **A** by solving the equation:

$$\begin{split} \mathbf{U} &= \mathbf{A_p} \mathbf{V} \leftrightarrow \mathbf{A_p} = \mathbf{U} \mathbf{V}^{-1} \eqno(4), \\ \text{where} & \mathbf{V} = \text{make matrix (industry-by-product)} \\ & \mathbf{U} = \text{use table (products-by-industry)} \\ & \mathbf{A_p} = \text{product-by-product technology matrix} \end{split}$$

The fixed product sales assumption assumes that the market share is constant, therefore:

In order to invert the **V** matrix in equation (4), the product technology assumption requires that the amount of products is the same as the amount of industries. This is commonly not the case in national statistics, since the detail of products is greater than the resolution of industries. In addition the product-by-product table is difficult to combine with other statistics, since they are collected on actual industries, while eq. (4) produces artificial single-product industries. In order to maintain a connection with other statistics, the EUROSTAT manual on collecting input-output statistics recommends the industry-by-industry approach (Eurostat, 2008) eq. (5), which was also the approach used in the studies of this dissertation.

The relationship between make and use tables, technology matrices, final demand, value added and the emissions and resources is presented in Table 1. In order to obtain eq. (1) with the industry-by-industry approach the final demand of products \mathbf{e} has to be converted into demand of industry output

f. This can be done by continuing the market share assumption to the final demand (i.e. $\mathbf{f} = \mathbf{V}\mathbf{q}^{-1}\mathbf{e}$).

Table 1. Overall structure of an environmentally extended input-output framework (adapted from (Miller and Blair, 2009)). The elements in italics are computed from the data presented in the national accounts.

	Products	Industries	Final demand	Total output
Products	Technology	Use matrix U	Product final	Product
	matrix App		demand e	output q
	(product-by-			
	product)			
Industries	Make matrix V	Technology	Industry final	Industry
		matrix A _{ii}	demand f	output x
		(industry-by-		_
		industry)		
Value added		Value added v	GDP	
Total output	Product output	Industry output x		
	q			
Employment		Employment s		
Emissions and		Environmental		
resources		flow matrix G		

When emissions, resource use or other sustainability indicators are known, they can be included in an environmentally extended input-output analysis (Suh and Huppes, 2005). The matrix of environmental flows is divided by the industry output to obtain unit emission/resource intensities:

 $\mathbf{B} = \mathbf{G}\hat{\mathbf{x}}^{-1}$

(6),

(7).

where \mathbf{B} = emission or resource use intensity (environmental flowby-industry) [kg/M \in]

G = emission or resource use matrix (environmental flow-byindustry) [kg]

These intensities can then be used similar to the value added in eq. (3) to give the environmental flows associated with a given technology and final demand:

 $\mathbf{g} = \mathbf{B}(\mathbf{I} - \mathbf{A})^{-1}\mathbf{f} = \mathbf{M}\mathbf{f}$

where \mathbf{g} = overall emissions caused by the final demand (environmental flow-by-1).

 $\mathbf{M} = \text{environmental multiplier ("footprint") matrix (environmental flow-by-industry)}$

If **f** is replaced with a diagonal matrix with the values of f at the diagonal, equation (2) will yield a matrix of emissions caused by production of final demand items. The final demand can also be reported for various subclasses of demand, most commonly household consumption, public consumption, investments and exports. The division of final demand to domestic and export demand allows the calculation of consumption based

inventories, if the emissions embodied in imports are also known (Peters, 2008).

In the analyses of this dissertation a tiered hybrid version of life cycle assessment and input-output analysis (Suh and Huppes, 2005) was used to evaluate the emissions embodied in imports. Using that approach, eq. (7) was modified to take into account domestic and imported products:

$$\mathbf{g} = \begin{bmatrix} \mathbf{B}_{\mathbf{d}} & \mathbf{B}_{\mathbf{i}} \end{bmatrix} \begin{bmatrix} \mathbf{I} - \mathbf{A}_{\mathbf{d}} & \mathbf{0} \\ -\mathbf{A}_{\mathbf{i}} & \mathbf{I} \end{bmatrix}^{-1} \begin{bmatrix} \mathbf{f}_{\mathbf{d}} \\ \mathbf{f}_{\mathbf{i}} \end{bmatrix}$$
(8),

where the subscripts d and i denote domestic and imported emission intensities, industrial outputs and final demand.

In the tiered hybrid approach, the intensities of imported products \mathbf{B}_i were mostly obtained from life cycle assessments of products, with the gaps (mostly in services) filled in by assuming similar intensities for imported products and domestic production \mathbf{B}_d (i.e. domestic technology assumption). The equation (8) is structurally similar to a multiple region input-ouput model (MRIO) (Wiedmann et al., 2011), except that the other "region" where imports were obtained from was approximated by a life cycle assessment database.

2.2.2 Analytical techniques of input-output analysis

The following four analytical techniques were used in the interpretation of results: structural decomposition, structural path, structural path decomposition, and perturbation analysis. The techniques are explained in detail in the following section.

Structural decomposition analysis (SDA) analyses the components of change over time. The basic components included are the environmental intensity, production technology and the size and composition of final demand. Since the input-output model is linear, the effect of changes can be expressed as differences (Miller and Blair, 2009):

$$\Delta \mathbf{g} = \mathbf{B}(\mathbf{I}-\mathbf{A})^{-1} \Delta \mathbf{f}_{s} f_{t} + \mathbf{B}(\mathbf{I}-\mathbf{A})^{-1} \mathbf{f}_{s} \Delta f_{t} + \mathbf{B}\Delta(\mathbf{I}-\mathbf{A})^{-1} \mathbf{f}_{s} f_{t} + \Delta \mathbf{B}(\mathbf{I}-\mathbf{A})^{-1} \mathbf{f}_{s} f_{t}$$
(9),
where \mathbf{f}_{s} = the structure of final demand $[\mathbb{M}\mathbb{C}/\mathbb{M}\mathbb{C}]$
 f_{t} = total amount of final demand (scalar) $[\mathbb{M}\mathbb{C}/\mathbb{M}\mathbb{C}]$

The differences are calculated between two points in time, but the static terms can be based on either the beginning or end year. This results in a large number of possible decompositions (16 decompositions for four components). Several methods have been developed in input-output

analysis to calculate the decomposition in a robust manner (Dietzenbacher and Los, 1998). In this thesis, the average of all possible first order decompositions was used. Also since the input-output tables are commonly reported in current prices, they are not directly comparable. This can be corrected by deflating the tables to given years prices. In the studies of this thesis, the tables were adjusted using double deflation (pre-multiplying intermediate use, final demand and total output by the producer's price indexes and recalculating the emission intensities) and the producer's price indexes (Statistics Finland, 2009). For a discussion on the methodological issues of double deflation, c.f. (Peters et al., 2007).

The decomposition analysis provides an overview of the causes of change, but does not identify the specific processes, which had changed. A recent addition to the environmental input-output methodology, structural path decomposition (Wood and Lenzen, 2009) can be applied to answer these questions. In structural path decomposition, the production structure of the economy is studied through series expansions of the Leontief inverse in order to identify the main environmentally relevant pathways (Lenzen, 2003). Changes in these pathways are then analyzed with structural decomposition. This method allows the study of change in a process level instead of country level aggregates.

The structural path analysis (SPA) begins with a series expansion of the Leontief inverse:

$$(\mathbf{I} - \mathbf{A})^{-1} = \mathbf{I} + \mathbf{A} + \mathbf{A}^2 + \mathbf{A}^3 + \cdots$$
 (10)

Applied to eq. (7) the total environmental flows can be expressed as the part directly caused by final demand, and the parts caused by higher order supply chains:

$$\mathbf{g} = \mathbf{B}(\mathbf{I} - \mathbf{A})^{-1}\mathbf{f} = \mathbf{B}(\mathbf{I} + \mathbf{A} + \cdots)\mathbf{f} = \mathbf{B}\mathbf{f} + \mathbf{B}\mathbf{A}\mathbf{f} + \mathbf{B}\mathbf{A}^{2}\mathbf{f} + \cdots$$
(11)

For a given flow k and industry i, the matrix expression of eq. (11) can be expressed as scalar sums:

$$g_{ik} = (b_{ki} + \sum_j b_{kj} a_{ji} + \sum_l \sum_j b_{kl} a_{lj} a_{ji} + \dots) f_i$$
(12),

where *b* and *a* are the elements of the corresponding matrices **B** and **A**.

Eq. (12) can then be used to express the overall impact of an industry as the sum of individual production paths. For example the path $b_{kl}a_{lj}a_{ji}$ describes the emission k originating from industry l, which is produced to supply products to industry j, in order for industry j to supply products for industry i. This is known as a second order or tier pathway, but orders can be continued indefinitely. However the amount of possible paths increases with path length. For example, an input-output system with 150 industries can potentially have over 500 million fourth order paths and relevant paths can still be found at the tenth order (Lenzen, 2003). Finding the appropriate pathways requires therefore an algorithm for screening out potential pathways without calculating them all. Usually the algorithms are based on comparing the upstream impacts to a given cut-off criteria (such as 1% of overall impact) and including for further analysis only the paths with potentially high upstream impacts (Lenzen, 2003; Wood and Lenzen, 2009). Once the important pathways are identified, structural decomposition can be applied on those to identify, to what extent the overall change can be explained with the changes occurring in the key pathways (Wood and Lenzen, 2009). This allows the identification of contrasting sector level development within the overall macro level change.

Sensitivity analysis attempts to answer the question: "what, if changed, can affect the outcome of a model?" Applied to sustainability scenarios, sensitivity analysis can identify the main components from an EEIO model. Several methods have been developed for sensitivity analysis (Saltelli et al., 2008), but we chose one of the simplest, a perturbation analysis based on partial derivatives (Heijungs and Suh, 2002; Heijungs, 2010). The perturbation analysis yields the sensitivity of the model output to relative changes in the input (i.e. $(\Delta f/f) / (\Delta x/x)$).

Applying partial derivatives for equation (7), the following sensitivity indices are obtained (Heijungs, 2010):

$S_{f} = \frac{\delta g_{k} / g_{k}}{\delta f_{i} / f_{i}} = M_{ki} \frac{f_{i}}{g_{k}}$	(13)
$S_a = \frac{\delta g_k / g_k}{\delta a_{ij} / a_{ij}} = M_{ki} x_j \frac{a_{ij}}{g_k}$	(14)
$S_{b} = \frac{\delta g_{k} / g_{k}}{\delta B_{kj} / B_{kj}} = x_{j} \frac{B_{kj}}{g_{k}}$	(15),

where the subscripts refer to the corresponding element of the matrix. For the eq. (14) a further correction was made on the diagonal elements $(1-a_{ii})$, scaling the sensitivity with the ratio of a_{ii} / $(1-a_{ii})$, in order to represent the actual change in the input coefficients and not in the Leontief matrix.

 S_f describes the sensitivity to final demand, S_a to inter-industry inputcoefficients and S_b to emission and extraction intensities. A subjective limit value of 0.01 was chosen for the sensitivity indices to separate the main components from less important parameters. With a sensitivity index of 0.01, a change of 100% in the component would influence the overall criteria by only 1%. Components which had a smaller potential for changing the overall criteria were not considered important.

The perturbation approach has its limitations; most importantly it is static and ignores the combinatorial effects of parameter changes. The static approach ignores possible rebound effects or marginal substitutions resulting from changing an input parameter. In a similar fashion, not taking into account combinatorial effects (e.g. the sensitivity of reducing electricity consumption will depend on the level of electricity emission intensity) presents the risk of overestimating the significance of combined changes. This is a general problem in combining individual measures to consistent scenarios (c.f. the popular stabilization wedges method, (Pacala and Socolow, 2004)). This problem can be avoided, if it is realized that the sensitivity indices are not additive. The combined effect of applying the measures must be analyzed in the actual scenario building phase as must the possible rebounds and substitutions. In spite of these limitations, the sensitivity analysis by perturbation is a useful screening level tool to identify the most important parameters for further analysis.

2.3 Life cycle impact assessment

Life cycle assessment (LCA) has developed from the first scientific studies in the 1960s (Guinée et al., 2011) to a standardized and sophisticated method for analyzing the environmental sustainability of products, regions and lifestyles (Finnveden et al., 2009). LCA is governed by a set of ISO standards (ISO, 2006) and method development is published in a specific journal for life cycle assessment (International Journal of Life Cycle Assessment). In the following, a brief description of a typical LCA study is given, followed by the mathematical details of LCA.

Typically an LCA study proceeds in four iterative sequences: Goal and Scope definition, Life Cycle Inventory (LCI) collection, Life Cycle Impact Assessment (LCIA) and Interpretation (Figure 2) (ISO, 2006). The goal and scope of the study define the questions it can answer, guides the methods that should be used to answer those questions and defines the functional unit for comparison. In the inventory stage, the product system supplying

the functional unit is mapped from "cradle-to-grave". In theory, all product and service flows are followed and the processes and flows needed to manufacture them are identified, until only "elemental flows" originating from or depositing to the ecosphere are left to follow. (In practice, cut-off criteria are used to simplify the analysis, commonly flows which are deemed insignificant are not taken into account.)



Figure 2. The four iterative stages of life cycle assessment (ISO, 2006)

At the end of the inventory collection, all relevant natural resource extractions and emissions are mapped out, often resulting in a list of hundreds of substance flows. It is the aim of the impact assessment stage to convert this data into meaningful indicators (i.e. disability adjusted life year expectancy), which can then be evaluated in the final interpretation stage. The impact assessment is usually done with characterization factors derived from environmental impact assessment models and collected into characterization sets such as ReCiPe (Rosenbaum et al., 2008; Goedkoop et al., 2009) or USEtox. In the interpretation stage, impacts are evaluated to compare alternatives across impact categories.

Traditionally LCI was conducted as a branching tree, which was collected as a process flow diagram and then solved sequentially starting from the main product, scaling the flows to match the functional unit one node at a time. This sequential approach however had problems in solving loops in the process system and has subsequently been superseded by the matrix approach (Heijungs and Suh, 2002; Suh and Huppes, 2005). In this

approach, the system is described as a system of linear equations, which are then solved simultaneously through linear algebra.

The mathematical formulation of LCI begins with the question: "assuming linearity in input-output relations, how much should each unit process be scaled to yield only the functional unit as the final output of the system?" Expressed as an equation (Heijungs and Suh, 2002):

 $\mathbf{As} = \mathbf{f}_{I}$

(16)

where \mathbf{f}_l = the functional unit of the study (vector of flows)

A = the process and flow matrix

 $\mathbf{s} =$ scaling vector (vector of processes)

If the A matrix is square and invertible, the scaling vector can be solved (in other cases more detailed algebra is needed, c.f. Heijungs and Suh, 2002):

 $\mathbf{s} = \mathbf{A}^{-1} \mathbf{f}_l \tag{17}$

If the process and flow matrix is collected for a certain time period (i.e. each column represents hourly rates or yearly production), the scaling vector will represent the amount of running time needed from each process. Detailed **A** matrices are sold for background processes as life cycle inventory databases. One commonly used database contains over 4000 rows and columns in **A** (Ecoinvent, 2010).

LCA differs from the approach of IOA, in that it includes also the end-of-life ("grave") component of the life cycle. This part is commonly left out of IOA, although methods for including it have been described in so called waste input-output tables (Nakamura and Kondo, 2009). Waste input-output tables link the generation of waste of a given year to the waste treatment and recycling necessary to treat the waste generated at the end-of-life stage. Overall the LCA of a product system will cover processes occurring in different time periods, while the IOA will contain all the processes occurring in a given year.

If the unit emissions and resource extractions for each process are known, the elementary flows corresponding to the functional unit can be solved and the LCI stage is completed:

 $g = Bs = BA^{-1}f$ (18) where B = unit emissions for each process in the A – matrix. Comparing eq. (18) and (7), it can be seen that EEIO and LCI share the same matrix formulation, with only minor differences (monetary vs. mixed units, single or multiple output processes). This observation has been a cornerstone in developing hybrid IOA-LCA methods in the past decade (Suh, 2009).

However, full LCA proceeds from the inventory stage to impact assessment and interpretation. Assuming linear response between emission and impact (or piecewise linearizing the problem to yield marginal changes), the life cycle impacts can be calculated as:

 $\mathbf{q} = \mathbf{C}\mathbf{g} = \mathbf{C}\mathbf{B}\mathbf{s} = \mathbf{C}\mathbf{B}\mathbf{A}^{-1}\mathbf{f}$

where \mathbf{C} = characterization factor matrix (impact per unit emission or resource use)

(19)

q = life cycle impacts

The characterization factors are obtained by calculating relevant partial derivatives from more complicated environmental models. For example the characterization factor for human toxicity from chemical emissions can be calculated as a series of stages in the impact pathway (Huijbregts, Rombouts, et al., 2005):

c –	∂disability	∂incidence	∂exposure	∂concentration	(00)
с –	∂incidence	∂exposure	∂concentration	∂emission	(20)

The partial derivatives are obtained for example from chemical transport models (concentration/emission response) (Mackay, 2001) and from doseresponse curves (Huijbregts, Rombouts, et al., 2005). It should be noted that the emissions to once compartment will cause concentrations changes in virtually all environmental compartments (i.e. emission to waste water will eventually influence soil concentrations through processes of evaporation and deposition) (Mackay, 2001). Similar derivations for the impact pathway have been made for several environmental impact categories, including both endpoint (i.e. disability adjusted life year change) and midpoint indicators (i.e. the total greenhouse gas emissions expressed as carbon dioxide equivalents) (Goedkoop et al., 2009).

In the interpretation stage, the impacts are normalized and weighted (if considered necessary in the goal and scope):

 $i = w \hat{n}^{-1} q = w \hat{n}^{-1} CBA^{-1} f$ where n = vector of normalization factors w = vector of impact category weights i = overall impact score(21)

Eq. (21) summarizes then the full environmental LCA in a single equation. The normalization factor is commonly calculated by using the characterization models to a reference emission inventory, for example the emissions in EU in a given year.

2.4 Sustainability assessment

The classical definition of sustainability is "economic and social development to meet the needs of the present without compromising the ability of future generations to meet their own needs." (Brundtland, 1987) Commonly this is interpreted as the three pillars of economic, social and environmental sustainability. However the main problem is in measuring and defining progress towards these issues. Some recent approaches are to measure the distance to specific goals or boundaries (Rockström et al., 2009; Raworth, 2012) or to define heuristics for strong sustainability (Robèrt et al., 2002).

Heuristic approaches define a set of criteria, which (if met) guarantee the sustainability of the system. For example the Natural Step defines a sustainable system through four criteria: concentrations of naturally extracted substances are not increasing, concentrations of man made substances are not increasing, nature is not degraded physically and humans can increase their living qualities globally (Robèrt et al., 2002). The problem with heuristic approaches is that they do not take into account the subjective nature of defining sustainability. In addition as a general result, heuristics tend to provide sub-optimal results in complex decision making situations (Tversky and Kahneman, 1974; Hammond et al., 1998).

At the same time, life cycle assessment has progressed from considering only environmental aspects towards including economic and social sustainability (Kloepffer, 2008; Guinée et al., 2011). Conceptually life cycle sustainability assessment is seen as a combination of environmental LCA, life cycle costing (LCC) and social life cycle assessment (SLCA) (LCSA = LCA + LCC + SLCA) (Kloepffer, 2008). While the life cycle costing is a relatively mature method, the social life cycle assessment is still undergoing major development and is challenging to apply together with other aspects (Guinée et al., 2011).

Input output analysis has been proposed to be a good framework for sustainability assessment (Murray and Wood, 2010). Indeed it can track the

total indirect effects to economic, social and environmental aspects throughout the global supply chain. Readily available statistics (gross domestic product, employment, greenhouse gas emissions) can be used to make a "triple bottom line" assessment for any company, region or country (Wiedmann et al., 2007).

However the selection of indicators should not be based on just availability. Based on decision analysis theory and practice, the indicators should reflect the criteria and goals of the decision maker (Keeney and Raiffa, 1993). Multiple criteria decision analysis (MCDA) has been widely used in environmental decision making (Huang et al., 2011). Typically it consists of mapping the value system of the decision maker into a value tree, which connects the overall objective to criteria, subcriteria and finally attributes used to measure those subcriteria (Keeney and Raiffa, 1993). By evaluating the tradeoffs between the attributes, subjective weights for the value tree can be obtained and this information can be used to compare and measure progress towards the overall objective. Several options for weighting and structuring the decision problem are available, but one of the simplest is the additive preference model:

$$\begin{split} i_k &= \sum_{j=1}^n w_j \, v_j(k_j) \eqno(22) \\ \text{where} & i_k = \text{the overall index for alternative k} \\ & w_j = \text{the weight of attribute i (n attributes)} \\ & v_j(k_j) = \text{a value function converting the attribute value k_j to a} \\ & \text{utility value in the range 0...1} \\ & k_j = \text{attribute } j \text{ for alternative } k \end{split}$$

Combining equation (22) with (21) and solving for v(k) it can be seen that:

$$v(k) = \hat{\mathbf{n}}^{-1} \mathbf{C} \tag{23}$$

Therefore the general equation of LCA can be seen as a subset of a MCDA problem, where it is assumed that the relationship between environmental flows and their value is linear and can be determined externally from the decision makers' preferences. The purpose of the actual weights \mathbf{w} in eq. (21) is then to convert the indicator numbers to a subjective preference scale. If the linearity assumption could be followed for LCSA and LCC as well, then the overall sustainability could be expressed as:

 $\mathbf{i} = \mathbf{w}_{e} \widehat{\mathbf{n}}_{e}^{-1} \mathbf{q}_{e} + \mathbf{w}_{s} \widehat{\mathbf{n}}_{s}^{-1} \mathbf{q}_{s} + \mathbf{w}_{c} \widehat{\mathbf{n}}_{c}^{-1} \mathbf{q}_{c}$ (24)

where the subsets e, s and c represent environmental, social and economic weights, normalization functions and impacts.

A considerable problem in applying a decision analytical sustainability framework to input-output analysis or life cycle assessment is that many of the indicators which are relevant for the overall objective are not available. For example, although water scarcity and species loss are critical environmental issues, their impact assessment methods are still under development (Finnveden et al., 2009; Mattila et al., 2012). In a similar fashion, the methods for evaluating the overall ecotoxic impacts are still under development in LCIA (Rosenbaum et al., 2008; Finnveden et al., 2009; Diamond et al., 2010). Also Rockström et al. (2009) stated that the ecotoxic pressure is a relevant sustainability boundary, but they were unable to quantify the relationship of current emissions with the boundaries. Therefore the indicators available for sustainability assessment will only represent a fraction of the total impact and even that with uncertainty.

Since capturing all the relevant indicators seems difficult, several single score indicators have been proposed to be used as a proxy for the whole. For example carbon footprint and cumulative fossil energy demand correlate well with all impact categories except toxicity and land use (Huijbregts et al., 2006; Laurent et al., 2010). The ecological footprint has been found to correlate with non-toxic impact categories in LCA (Huijbregts et al., 2008). Therefore a relatively complete account of the environmental component of sustainability could be achieved by assessing carbon footprint, ecological footprint and toxicity impacts.

2.5 Theory synthesis

Input-output analysis (IOA) was found to be closely related to life cycle assessment (LCA) and sustainability assessment (SA) in general. In particular the impact assessment methods developed for LCA could benefit the linking of input output data to the overall sustainability criteria. The strong connection between LCA and decision analysis provides a theoretical background for this combination. The strengths of the IOA were its completeness, transparency, and the history of analytical tool development.

As input-output tables are a part of the national accounts (SNA), several sustainability indicators can be directly connected to them on a national level. Life cycle impact assessment models can then be used to convert the indicators into impacts, which can then be evaluated using decision analytical methods. Once the linkages have been constructed, the analytical techniques of input-output analysis can be used to identify main pathways,

networks and connections which contribute the most to given sustainability issues.

A main problem however in this process is the lack of a complete set of indicators for sustainability. Therefore proxies have to be used to represent the overall sustainability issue. The proxies for social and economic sustainability could be the employment and gross domestic product, while carbon footprint, land use and toxicity could approximate the overall environmental sustainability.

3. Research contribution

3.1 Article I: Input output analysis can reveal the sustainability of an industry in the perspective of the whole economy

In the first study of this thesis, environmentally extended input output analysis was applied to the Finnish economy with a focus on the forest industries. The analysis was focused on two economic indicators (GDP, import dependency), one social indicator (employment) and two environmental indicators (greenhouse gas emissions, aggregated land use). The analysis was conducted on an aggregated IO table, which had 8 forest industry sectors and 13 other sectors. The table was aggregated from the more detailed Finland 2005 IO table with 150 sectors (Seppälä et al., 2009).

Based on the results, the forest industries were strongly economically interlinked which each other and with the rest of the domestic industries. The import dependency was lower than those of most other sectors, with the exceptions of services and agriculture. Overall the forest sectors were found to act as key sectors, e.g. they were able to stimulate their demand through their own supply chain more than other industries (Oosterhaven, 2004). This was observed from the Leontief inverse multipliers ((I-A)⁻¹ in eq. (1)).

Looking at employment multipliers, the forest industries were found to have a relatively low intensity (e.g. working hours/ \mathbb{C} of production) especially compared to primary production but also to metal industry and construction (Table 2). The total employment figures (**M** in eq. (7)) were several times higher than the direct multipliers. Builders carpentry and other wood products had the highest employment multipliers and pulp and paper had the lowest. For most forest industries the greenhouse gas total multipliers were an order of magnitude higher than the direct multipliers. The total multipliers were still among the lowest sectors, indicating low carbon intensity. However for land use the total multipliers were two orders of magnitude higher than the direct multipliers. The land use intensity of

forest industries was the highest among industries, comparable only to the agriculture sector. However it should be noted that land use between agriculture and forestry differs considerably in intensity and environmental impact, therefore the figures are not directly comparable.

Another result from the input-output analysis is that the service sectors have a considerable indirect multiplier effects both to climate and land use. The transparent expression of different sustainability indicators allows a preliminary assessment of potential effects of changing the economy towards for example more services. MCDA could then be used to quantify the overall desirability of those changes looking simultaneously at all the sustainability pillars.

The multipliers allowed also the evaluation of total impacts caused by each industry and the division of those impacts to exports, domestic consumption and investments (eq. 7). From that perspective, the forest industries were found to contribute to a major share of aggregated land use and greenhouse gas emissions, but only a minor share of GDP or employment. Over 86% of the emissions associated with forest industries were found to be for exports. Therefore the sustainability of the Finnish economy and its forest industries was strongly linked to international trade.

Table 2. Selected impact multipliers for the aggregated environmentally extended input output table of Finland 2005. The highest impact multipliers were bolded. (Mattila et al., 2011)

		GHG		Employment		Land use		Imports	
		kg CO₂e/€		work hours/€		m²/€		€/€	
		direct	total	direct	total	direct	total	direct	total
1	Agriculture	1.6	2.3	0.2	0.3	13.4	18.0	0.05	0.2
2	Forestry and logging	0.1	0.2	0.05	0.07	47.1	57.5	0.01	0.03
203	Builders carpentry	0.04	0.3	0.06	0.1	0.00	8.2	0.1	0.2
211	Pulp, paper & cardboard	0.3	0.8	0.02	0.07	0.01	5.9	0.1	0.3
6	Chemical industry	0.5	0.7	0.03	0.06	0.01	0.3	0.4	0.5
7	Metal industry	0.3	0.5	0.04	0.08	0.00	0.2	0.3	0.4
10	Energy	3.8	3.9	0.02	0.05	2.0	3.3	0.2	0.2
11	Construction	0.08	0.3	0.07	0.1	0.00	1.1	0.08	0.2
15	Other service activities	0.06	0.2	0.11	0.2	0.00	0.2	0.06	0.1

3.2 Article II: Most of Finnish land use impacts are caused by the production of export products

The second study of this thesis focused on the land use impacts of industries. The previous calculations on land use were extended in two ways. First of all, the aggregated land area used in (I) was replaced by three LCIA indicators. Second, the calculations were based on the fully disaggregated 150 industry model, allowing a more thorough analysis of impact pathways.

CORINE land cover data was used to calculate the land uses of different industries. Details of the calculation are given in (II). The CORINE classification allowed the disaggregation of land use to 30 categories (Härmä et al., 2004). These categories were converted into impact indicators (eq. 19) using three impact assessment models: ecological footprint biocapacity (Ewing, Reed, et al., 2008), human appropriation of net primary production (HANPP) (Haberl et al., 2007) and ecosystem damage potential (EDP) (Koellner and Scholz, 2006). The biocapacity measures the productivity of the land and is used as a proxy for biological resource use, HANPP measures the disturbance to natural ecosystems through the utilization and reduction of net primary production (NPP) and EDP measures the value of land cover as habitats for species. All indicators were customized to Finnish conditions using national statistics on habitat density (Auvinen et al., 2007) and agricultural and forest productivity as well as individual studies on NPP distribution (Liski et al., 2006). The extent of land use embodied in imports was estimated using Ecological Footprint Accounts for Finland (Ewing, Reed, et al., 2008).

Based on the results, Finland was found to be a net exporter of land. An area corresponding to 43% of Finnish land economic use (70% of land cover) was reserved globally for the production of imports. However, 65% of domestic land occupation was reserved for the production of exports (Figure 3). The main drivers of land use occupation were the forest industries and agriculture (especially reindeer herding).



% of land area

Figure 3. Land use allocated to industries (left) and products (right) in Finland 2002 as a fraction of the domestic inland surface area (305,000 km²). The shaded region in the right figure is the extent of exported domestic land area.

Forestry, reindeer husbandry and agriculture were found to be the main direct causes of land use impacts in all three impact assessment methods. Approximately one third (36%) of Finnish net primary production was used by humans (HANPP), mainly in forestry and agriculture. Most of the unused NPP was estimated to be in undrained peatlands, forest litter and logging residues and on sparsely vegetated areas in North Finland. The biocapacity utilization rate (86% of productive land in use) was higher than general land occupation, since the land occupation was focused on more productive land areas (i.e. agricultural fields instead of sparsely vegetated areas).

Looking at life cycle impact intensity multipliers (CM, impact/ $M\in$), considerable differences were found between industries. While other animal products (reindeer) occupied the largest land area, it had a lower biocapacity occupation than roundwood production and a considerably lower HANPP impact multiplier than any forest, agriculture or food product.

Looking at ecosystem damage potential however, the impact of reindeer management was found to be beneficial to the environment using Central European impact factors. This however was an erroneous result, resulting from the difference in biodiversity between European sparsely vegetated regions (Alpine meadows) and Finnish regions. Using Finnish habitat density as a basis, the impact intensity of reindeer was found to be an order

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of magnitude higher than for other products. However using the same Finnish habitat data, dairy products have a negative impact multiplier, indicating biodiversity gain. This is caused by the maintenance of species rich pastures and meadows by grazing animals. Related to this, using Finnish habitat densities, also constructed areas have a net species gain, since they have more habitats per area than for example forests or agricultural areas.

This result underlined the importance of using regional species density data for impact assessment but also the challenge of creating universal indicators for biodiversity impacts in life cycle assessment of products (also identified in (Udo de Haes, 2006; Milá i Canals et al., 2007)).

Land	Biocapacity	HANPP	EDP	EDP
use	km² M€-1	kt M€-1	Finland	CE
km² €-1				
199	129	0.6	15,000	-5,200
75	200	9.2	850	520
28	73	3.4	310	190
18	52	5.0	710	760
5	1	1.1	-1,600	130
4	10	1.0	120	150
0.1	0.2	0.01	-1	0
	Land use km ² € ⁻¹ 199 75 28 18 5 5 4 0.1	Land Biocapacity use km ² M€ ⁻¹ km ² € ⁻¹ 29 75 200 28 73 18 52 5 1 4 10 0.1 0.2	Land Biocapacity HANPP use km ² M€ ⁻¹ kt M€ ⁻¹ km ² € ⁻¹ 200 9.2 28 73 3.4 18 52 5.0 5 1 1.1 4 10 1.0 0.1 0.2 0.01	Land Biocapacity HANPP EDP use km ² M€ ⁻¹ kt M€ ⁻¹ Finland km ² € ⁻¹ 129 0.6 15,000 75 200 9.2 850 28 73 3.4 310 18 52 5.0 710 5 1 1.1 -1,600 4 10 1.0 120 0.1 0.2 0.01 -1

Table 3. A comparison of impact intensities of selected products assessed with different indicators using the environmental input-output framework for Finland. The highest indicator results are presented in bold.

Were the analytical indicators consistent with expert assessments on biodiversity in Finland? According to the "Fourth National Report on the Implementation of the Convention on Biological Diversity in Finland" (Ministry of the Environment, 2009), based on nearly 100 habitat based indicators, halting the decline in biodiversity seemed unlikely to be met by 2010. Forests were identified as the main habitat of endangered species, threats to them resulting from long-term forest practices (species and age distribution and lack of deadwood). All impact indicators used in this study identified forest products as a main component of land use impacts. HANPP estimated that only a minor part of NPP in forests would be used by humans. However, since it is the large deadwood which is necessary for many endangered species (Rassi et al., 2001), the effect of forestry practices on the quality and size of remaining wood should be included for biodiversity assessment purposes.

In alpine habitats all indicators, except the Central European EDP, identified trampling and grazing by reindeer to have negative impacts. This influence on plant diversity was also confirmed by expert judgment (Auvinen et al., 2007; Ministry of the Environment, 2009). However the damaging impact of tourism and off-road driving highlighted in the expert evaluation (Ministry of the Environment, 2009) was not identified by the model, since the land use was allocated to the primary sector utilizing the biological productivity of the region. This allocation rule also resulted in the cut-off of mires and shores. Mires were threatened by the historical drainage to forests and agricultural areas, not by their current land use. Also shore habitats were not threatened by their use or occupation, but by transformation into residential areas (Auvinen et al., 2007; Ministry of the Environment, 2009). Although methods for land use transformation impact assessment have been proposed (Milá i Canals et al., 2007), transformation impacts were not assessed, due to data limitations (Finland did not participate in CORINE-mapping prior to 2000). With time series of land use and transformation the biodiversity impacts could be better allocated to industries. The allocation of the impacts of past land transformations remains however an open question, and is especially critical to historical high-biodiversity farmlands, which are declining because of changes in agricultural practices but at the same time maintained by agricultural practices.

Only the regionalized EDP-indicator identified the importance of animal production in maintaining biodiversity in farmland habitats (meadows). HANPP and biocapacity considered agriculture as a user of biological productivity, neglecting the aspect of habitat maintenance. In the Central European EDP, the biodiversity benefits of natural grassland and meadows were included, but their impact was less than in the Finnish ecosystem, where agriculture is only a minor fraction of the landscape. This confirms the need for a regionalized approach in assessing the life cycle impacts on biodiversity and also taking into account the benefits of human activity to biodiversity.

3.3 Article III: Value added and ecological footprint are caused by different parts of the economy

The third article focused still on land use. The ecological footprint (EF) was used as the main indicator. It reduces resource consumption into productive land area, which is needed to produce those resources. In addition to actual land occupation, it also includes the hypothetical land

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area needed to produce the fossil fuels used (Ewing, Reed, et al., 2008). The analytical techniques of structural path analysis (SPA), structural decomposition analysis (SDA), structural path decomposition (SPD) and sensitivity analysis (SA) were applied to reveal the most relevant interindustry connections. The aim was to see, which of the several thousands of model variables were actually relevant for the sustainability indicator. In addition, the gross domestic product (GDP) was analyzed by tracking the value added of industries with the same tools as the ecological footprint. This allowed the analysis of the interlinkage between GDP and EF. It also allowed the analysis, whether the decoupling of economic growth from ecological footprint would be a "nearly decomposable problem" (Simon, 1962), where the two would be driven by two different subsystems.

The sensitivity analysis revealed that there were relatively few important connections among the included 40 120 economic interactions. For the ecological footprint only 25 items were important in the input coefficients. For the gross domestic product, 12 items were identified as important (Figure 4). The overall ecological footprint was most sensitive to the industrial use of wood for pulp and paper, sawmilling and for residential construction, as well as to the use of animal products for meat and dairy production. Other notable influences were the use of crops and the production of electricity from both wood residues (from sawmilling) and from fossil sources. Two import commodities were of importance: the import of pulpwood and natural gas. For the GDP, fewer linkages were found to have significant effect than for the ecological footprint. These were mainly connected to trade, business and communication services as well as to construction. Pulp and paper production and dairy production were the only industrial processes, which had a significant impact on GDP.

Overall seven coefficients (carpentry and trade services in construction, wood and pulp use in paper industry, dairy production, apartment repair and civil engineering) were found to be significant for both indicators, but for the most part the sensitivities were different between ecological footprint and GDP. This indicates that GDP changes are governed by economic interactions, which do not have a clear influence on the ecological footprint. The only exceptions were the use of wood in pulp and paper manufacture and construction as well as the production of dairy products. This finding is in contrast with some other studies, which have found that on global scale, the increase in services usually increased footprint, while increases in materials industries (often related to export production) decreased the consumption based footprint (Jorgenson and Burns, 2007).



The only services which had a significant influence on the ecological footprint of Finland were housing and construction work.

Figure 4. A graphical representation of the most relevant interindustry connections in the Finnish input-output tables identified with sensitivity analysis. Top: ecological footprint, bottom: gross domestic product.

Structural path analysis confirmed that EF and GDP are driven by different subsystems. The largest single contributors to the ecological footprint were the consumption of wood, crops, imported fish and electricity. Another important factor was the consumption of wood embodied in construction work through several intermediate products, such as builder's carpentry and sawn wood (i.e. path "forestry-sawmilling-carpentry-residential construction" contributed to 0.14 gha of productive forest per capita in 2005). Together these top ranking flow paths contributed to a third of the total footprint. In comparison to the lengthy supply chains of ecological footprint, the main pathways of GDP formation were very short. Most value was added just before the final product was consumed, with the top ranking path being the owning and renting of apartments (2 200 C/cap/yr). In addition, most of the products were

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actually services provided by the government, such as education, social work and health services. Construction and renting and owning apartments were common to both datasets, but otherwise the identified pathways were different.

Finland may be an extreme example, where biological resource use and GDP are so clearly separated, since the economy uses so much wood. It is likely however, that the general pattern can be observed in other economies as well: value added is usually produced far in the supply chain from environmental impacts. For example in residential construction the value added is formed in the last stage of marketing the finished apartment, but the ecological impacts were caused by forestry three tiers up the supply chain. Similarly, the growth in service industries increases GDP directly, but the resource extraction is visible only through long supply chain interactions.

Overall both EF and GDP grew between 2002-2005 (Figure 5). Therefore there was no absolute decoupling between environmental impacts and economic growth, in spite of earlier reports (Ewing, Reed, et al., 2008). The ecological footprint impact intensity decreased considerably between the years, while the other factors pushed the footprint higher. Especially the increase in demand size was a critical term in increasing the footprint. Comparably the demand size was the only factor, which increased GDP. The production structure, intensity and demand structure would all have decreased the GDP. One mechanism for this was the substitution of domestic production with imports, which was indicated by the 30% of growth in imports between 2002-2005 (Statistics Finland, 2009).



Figure 5. Decomposition of the change in (a) ecological footprint and in (b) GDP in the Finnish economy between 2002-2005.

Looking at the causes behind the change with decomposition techniques (Wood and Lenzen, 2009), major changes in the most important production and consumption pathways were identified. The causes of change were separated to final demand size, land use intensity and structural change in the production layers. Structural change was indicated by changes in the use of inputs in the sector, for example more efficient use of wood in sawmilling. The largest single contributor to the change in the ecological footprint was the increased demand of crop products (Table 5). This was caused by the changes in storage fluctuations and not due to actual consumption changes. The second highest influence was caused by changes in the second layer of production (A2): Sawmilling for residential construction became more efficient in using round wood from forestry. The third and fourth most influential changes canceled each other out: the carbon intensity of electricity production decreased, but the demand of electricity by households increased.

Several top ranking pathways were associated with the product chain of wooden materials used in residential construction. More efficient use of raw wood in sawmilling and carpentry amounted to a decrease in the forest footprint, but this effect was offset by the increased demand of construction and the increased use of sawmilled products in carpentry and the increased use of carpentry in construction. A similar trade-off was observed in the reduced consumption of domestic fish (path 12) and an increase in the amount of imported fish consumed in restaurants (path 16).

Overall positive developments in ecoefficiency were observed in the process level. These were observed through improvements in impact intensity (ΔB) of electricity production and forestry, as well as the more efficient use of forest products (ΔA) in sawmilling. However the final demand of consumption (Δf) increased, and this resulted in a net increase of the ecological footprint by 0.79 gha/capita between 2002 and 2005.

Table 4. Top ranking pathways for change in the ecological footprint of Finland between 2002 and 2005. (The sources of change are coded as following: f = final demand, A1...3 = input use in supplying sector level, B = footprint intensity.) The sector where the structural change (indicated by changing input use to produce sector outputs) occurred is marked in bold.

Path rank	EF gha	Land type	Chan -ge	Final product	1. Order	2. Order	3. Order
1	0.20	Crop	Δf	Crop production	-	-	-
2	-0.07	Forest	ΔA2	Residential construction	Residential construction Sawmilling Forestry		-
3	0.06	Carbon	Δf	Electricity	-	-	-
4	-0.06	Carbon	ΔB	Electricity	-	-	-
5	-0.05	Carbon	ΔΒ	Renting and owning apartments	Electricity production	-	-
6	-0.04	Forest	ΔA_3	Residential construction	Carpentry	Saw- milling	Forestry
7	-0.04	Carbon	ΔΒ	Renting and owning apartments	-	-	-
8	0.04	Forest	ΔA2	Residential construction	Carpentry	Saw- milling	Forestry
9	-0.03	Forest	ΔA2	Residential construction	Carpentry	Forestry	-
10	0.04	Forest	ΔΑ1	Residential construction	Carpentry	Saw- milling	Forestry
11	0.04	Forest	Δf	Forestry	-	-	-
12	-0.02	Fishing	Δf	Fishing	-	-	-
Sum	0.79						

The main sources of change for the gross domestic product were associated with growth. All 12 top ranking causes of economic growth were the increased demand for services such as trade, health, public administration, education, transportation and business services. The demand for pulp and paper decreased, but this was compensated by increased demand of residential construction. Overall the gross domestic product increased by $950 \in_{2002}/\text{capita}$. Very few structural changes were in the most important pathways, the exceptions being the increased use of road transport and business services by the pulp and paper industry. The only top ranking pathway which was common for the two indicators was the reduced use of forestry products in sawmilling, which reduced the ecological footprint as well as the gross domestic product.

Previous studies on economic growth and EF have concluded that on a global level there is no Kuznets curve: increased income results in a larger ecological footprint (Bagliani et al., 2008; Caviglia-Harris et al., 2009). The results of this study support these findings, but also complicate the overall conclusion. The national economy was found to include processes, which would have reduced the ecological footprint through more efficient resource use, but that these processes were overrun by increased overall demand (Figure 5; Table 4). Similar results have been observed also for China, where the benefits of energy efficiency have been overcompensated by increased production levels, resulting in increased emissions (Peters et al., 2007).

In summary, looking at the ecological footprint and economic growth with different analytical tools, the two indicators would seem to be connected to mainly different subsystems of the economy, but both are driven by increased consumption. A few pathways and connection coefficients determine the most of the results for both indicators. With the economic and technological development ongoing between 2002-2005, if consumption would not grow continuously, both GDP and ecological footprint would decrease.

3.4 Article IV: A life cycle approach complements the priority setting of chemicals by expert judgment

In the fourth article included in this thesis, the viewpoint was changed from land use to chemical pollution. Increasing concentrations of hazardous substances has been identified as one of the main environmental problems, but also as very difficult to quantify (Rockström et al., 2009). In the study, three state-of-the-art life cycle impact assessment (LCIA) models were compared to each other and to the expert judgments on chemical hazards. All three models followed the same structure of eq. (20) but used different modeling assumptions in calculating the fate, exposure and damage associated. The IMPACT2002+ (Jolliet et al., 2003) and ReCiPe (Goedkoop et al., 2009) models were based on tools and methods used in chemical risk assessment. The USEtox was a consensus model (Rosenbaum et al., 2008) based on the harmonization of several previous models. It is currently the impact assessment model recommended by SETAC (Society of Environmental Toxicologists and Chemists).

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The three models were applied to an inventory of Finnish hazardous emissions for the year 2005, which included emissions to air, water and agricultural soil. Overall 62 emission categories (substance and receiving compartment) were included. Details on the collected emission inventory are given in article IV. Impacts were calculated for ecotoxicity to freshwater organisms and for human toxicity. The results were normalized by dividing them with the estimated toxic pressure caused by European emissions (eq. 21). In the following, only the results concerning ecotoxicity are presented. Results concerning human toxicity are presented in article IV.

Both IMPACT2002+ and USEtox identified copper and zinc emissions to water and air causing a major part of ecotoxic impacts. In addition USEtox identified vanadium air emissions as a priority and IMPACT2002+ highlighted also nickel emissions to air and water. In ReCiPe however, most of the ecotoxic potential was caused by water emissions of organic substances, especially tributyltin (TBT) from ships (Figure 1). Overall the normalized results expressed as a share of the toxic pressure from European emissions varied over four orders of magnitude between models (0.5% in ReCiPe, 1.4% in IMPACT2002+ and 2.1% in USEtox).

The small result in ReCiPe was caused by a small share of TBT compounds in Finland compared to European emissions. If TBT was ignored, ReCiPe had similar results to the other models (i.e. 2.0% of European toxic impact). Impact2002+ did not include TBT, but in ReCiPe it was the main pollutant, amounting to 92% of the ecotoxic pressure. Using USEtox, TBT amounted to only 1.4% of the ecotoxic pressure. The difference between the impact models is caused to a large extent by the different chemical properties for TBT in USEtox and ReCiPe. This reflects the considerable variability in the measured experimental degradation rates (ECHA, 2008). In the latest integrated assessment of the Baltic Sea, TBT compounds were identified as a source of high concern, since their observed concentrations in biota exceeded quality limits in most parts of the Baltic (HELCOM, 2010). If USEtox were used in national prioritization of ecotoxic impacts, the importance of TBT would be ignored and a focus would be on controlling air emissions of heavy metals. This is a strong caution against using LCIA models as a substitute for expert assessment.

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Figure 6. Comparison of the share of toxic load from substance groups in the three assessed LCIA models.

Applying input-output analysis and structural path analysis (eq.12) to the LCIA results allowed the identification of key economic pathways responsible for the toxic load similarly to that of ecological footprint (III). The models yielded overlapping results. IMPACT2002+ and USEtox highlighted zinc emissions from artificial fibre manufacture and household fuel use. Both USEtox and ReCiPe also identified vanadium from oil refining. IMPACT2002+ also identified copper emissions from metal industry and households. In contrast to other models ReCiPe highlighted the importance of tributyltin (TBT) from shipping, which was driven both by final demand and the supply chains of retail trade, pulp and paper as well as residential construction. In USETOX vanadium emissions from oil refining were considered as the main priority, followed by zinc and vanadium emissions from domestic fuel use and zinc water emissions from artificial fibre production. Overall by using an updated model, the focus was moved from shipping to petrochemical manufacture and use. The reduced role of copper emissions between IMPACT2002 and USEtox is notable, since USEtox includes a more sophisticated method for assessing the toxicity of metals, including only the dissolved and bioavailable fraction of metals.

All models could be used to identify top ranking supply chains for controlling pollution through sustainable consumption and production policies. Compared to earlier work using structural path analysis (Lenzen, 2003), the identified paths were very short, indicating that toxic emissions are mainly released in the final stages of the supply chain. Using USEtox for policy recommendations would then result in a broader scope of measures,

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while based on the two other method a focus on few key pollutant sources would be recommended.

Comparing the results to those of recent chemical risk assessments in the region (HELCOM, 2010), the main differences could be observed. The main strength of LCIA models is that they consider impacts over time, therefore metal emissions have a significant impact compared for example to pesticides, since they persist for hundreds of years in the system. This realization is important for broadening the scope of chemical risk assessment, which tends to focus on currently measured concentrations. On the other hand combining very long term impacts with current impacts makes the interpretation of results more difficult, since the future predictions cannot be validated by observations. In addition the effect of accumulating multiple stressors is not included in LCIA, which makes risk assessment more difficult. A second issue in current LCIA models is that they do not include foodweb bioaccumulation. Therefore the importance of persistent bioaccumulative organic pollutants is reduced in LCIA based studies. Finally, the current impacts from historical emissions (e.g. DDT, PCB, radioactives) are not included or identified in LCIA based EEIO studies.

Since all LCIA models could simplify the problem of managing over 60 substance emissions to a few key pollutants and emission pathways, their use could simplify environmental policy making. However, since the models also resulted in different priority setting (and in the case of USETOX the exclusion of the critical TBT emissions), the models can be seen a complementary tool and not a substitute for chemical risk management.

3.5 Article V: Input-output models can be simplified for building scenarios of sustainable development.

In the final article of this dissertation the possibilities of extracting meaningful information from EEIO-models was tested further. The aim was to see, if sensitivity analysis (eq. 13-15) could provide a simplified model of the economy, which could then be used to build scenarios of sustainable development. This idea was based on the observation that very few model components were identified as having a high sensitivity in regard to ecological footprint or GDP (c.f. Figure 4). The aim was to test, whether the same model components would apply to different impact categories and to

see if future development could be predicted by using only the components with high sensitivities.

Four impact categories were selected for analysis: GDP, greenhouse gas emissions, land use and waste generation. As was expected, very few parameters had a significant sensitivity in any category (Figure 7). Most of the parameters had a sensitivity index less than 10⁻⁶, indicating that they have little significance in practical purposes (i.e. an order of magnitude change in a parameter would change overall results by less than 0,001%). The components with a sensitivity index higher than 0.01 represented 0.3% of the total amount of parameters in the model (n = 23 103). This was in line with the general observation of modeling, that in most cases very few input parameters contribute to most of the variability in a model output (Saltelli et al., 2008).



Figure 7. The distributions of sensitivity indices (S_a , S_b and S_y combined) for gross domestic product (GDP), global warming potential (GWP), land use and waste generation. The box and whiskers plot describes the median of the distribution, the 25% percentile and the 75% percentiles and the whiskers extending to 1.5 times the interquartile range. The dots above the distribution are outliers, and very few parameters had sensitivities higher than 0.01 (marked by a dotted line).

For climate change impacts (GWP) the 57 main components were found in emission intensity (S_b , n=20), final demand (S_y , n=22) and inputcoefficients (S_a , n=15). Emission intensities for electricity production, iron

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and steel manufacture, animal farming and pulp and paper production had the highest sensitivity indices (ranging between 0.08-0.27), followed by the final demand of pulp and paper, iron and steel, apartments and trade services (sensitivity index between 0.07-0.09). In comparison all sensitivity indices for input-coefficients were less than 0.05, with the highest indices for the use of animal products in the food industry and the use of electricity in apartments.

Compared to the GWP, the gross domestic product (GDP) had a similar amount of main components (20 variables in intensity, 18 in final consumption and 4 in input-coefficients), but the identified components were different. The highest sensitivities were in the demand of apartments, trade and residential construction (ranging between 0.07-0.12). The direct intensity of apartment renting, business services and trade were in the same order of magnitude (i.e. 0.07-0.10). The only input-coefficients with high sensitivity were associated with the use of trade services for construction industry and the use of business services in the electronics industry. This highlighted the earlier conclusion, that the subsystems of environmental pressure and economic growth are largely separated in the economy (Mattila, 2012).

Land use and waste production had on average smaller sensitivities than global warming or GDP. Both indicator sets were dominated by a few main components with high sensitivities. For example land use had very high sensitivity to the direct land intensity of forest cultivation ($S_b = 0.67$) and animal production ($S_b = 0.15$). As a consequence also the sensitivities to the demand of pulp and paper ($S_y = 0.23$), sawn wood ($S_y = 0.12$) and animal products ($S_y = 0.14$) were high, as was the sensitivity to the intermediate use of timber for sawmilling ($S_a=0.23$) and pulp and paper production ($S_a=0.25$). Similarly waste generation was sensitive to the direct intensities of rock quarrying ($S_b = 0.19$), mining of fertilizers ($S_b = 0.17$) and pulp and paper production ($S_b = 0.15$). This was reflected as high sensitivities in the input-coefficient of fertilizer mineral use in fertilizer production ($S_a = 0.11$) and in the final demand of pulp and paper ($S_y = 0.17$), non-ferrous metals ($S_y = 0.07$), construction ($S_y = 0.08$) and fertilizers ($S_y = 0.06$).

The limited amount of identified main components is promising for scenario building: comprehensive scenarios can be built with a relatively small number of components. Based on the identified main components, the following subsystems should predict the trend of greenhouse gas emissions: process industry (pulp and paper, basic chemicals, iron and steel), electronics industry, construction, transportation, electricity production and animal production. Based on the shared high sensitivities over impact categories, that set of subsystems should also cover the development of waste production and land use with only minor additions. However in order to model the development of GDP, the public sector (education, social work and health services) as well as the trade and apartment sectors should be considered in the scenario work.

The accuracy of a simplified EEIO model was tested by updating only the identified components from a year 2002 model to year 2005 values. The predicted change in greenhouse gas emissions was then compared to actual development using structural decomposition analysis to highlight the components of change Based on the results the predictive power of the simplified update is highly accurate. The actual emissions changed by 6.4 Mt (from 71.8 Mt to 65.4 Mt), while the predicted change was 1% lower. Differences in the components of change where however slightly larger, but their effect was in the opposite direction (for example in the emission intensity and input-coefficients) (Figure 2). The overall development in final demand (y2) was captured reasonably well although it was not directly changed. Therefore the components which were identified in the sensitivity analysis represented also a major fraction of the final demand.

The decomposition results also demonstrated that the decrease in the national GHG emissions was caused mainly by the decreased emission intensity between 2002 and 2005. The main cause for the reduced emissions was the mild winter and the good availability of imported Nordic hydropower, both which reduced the need to operate coal fired power plants If the emission intensity had remained at the year 2002 level, the emissions would have grown by 4.5 Mt CO_2e , due to increased final demand size.

Overall the sensitivity analysis provided a greatly narrowed down list of relevant parameters (c.a. 60 main components out of 23 000 model parameters). The development in greenhouse emissions from 2002 to 2005 could be predicted relatively well using only those main components. The scenario development should then attempt to capture the relevant trends and mitigation potentials influencing those main components.



Figure 8. The actual (dark) and predicted (white) change in global warming potential decomposed into components of the input-output model.

3.6 Results summary

The combination of life cycle impact assessment and input-output analysis was shown to provide results, which are compatible with expert judgment on land use, biodiversity (II) and ecotoxicity (IV). At the same time, the structure of input-output analysis allows a transparent and concise evaluation of total "life cycle" or multiplier effects (I, II, V). Capturing all of the economy wide impacts with LCIA can also broaden the perspective on some issues, such as the biodiversity gains of grazing and the long time scales of metal toxicities following emission. With the analytical tools of IOA, key processes, supply chains and emission sources can be highlighted (III, V). Therefore EEIO can be seen as powerful tool for communicating and understanding the complex interactions of production, consumption and environmental degradation. At the same time however, difficulties in considering the impacts over time (e.g. chemical pollution IV) and the effect of historical events (e.g. chemical pollution IV, peatland drainage II) limit their applications and can result in erroneous priority assignments.

4.1 Theoretical and practical implications

Progress in the application of environmentally extended input output analysis (EEIO) in industrial ecology has resulted in two key observations (Suh, 2009): that it is difficult to capture the whole life cycle impacts without economic models (Suh et al., 2004) and that in order to track the whole environmental impacts you need a global input-output model (MRIO) (Wiedmann et al., 2011). The downside of this is that environmental problems seem to be very complex with thousands of direct interactions and an infinite amount of indirect interactions. The results of this study would seem to contradict these results to some extent. In spite of complex economic supply chains, a very limited set of nodes defines the overall environmental impact level (V). For some impact categories, such as ecotoxicity. most of the impacts are caused by the very last stages of the supply chain (IV), indicating that the supply chain approach is always not necessary. This is good news for managing environmental problems, since the systems can be simplified to the extent that they are understandable.

On the other hand, the results on land use demonstrated that most of the Finnish national land use was driven by production of exports. Using the consumer responsibility paradigm (Lenzen et al., 2007), those impacts would be the responsibility of the importing nations. However the consumption based inventories have been usually collected for greenhouse gases or ecological footprints (Ewing, Goldfinger, et al., 2008; Peters, 2008). Supply chain based analyses of biodiversity have only recently been published (Lenzen et al., 2012). The theoretical framework of controlling local land use impacts with consumer responsibility has not yet been developed (Sakai, 2012). How local land use impacts, which are driven by global demand, should be controlled remains a critical question for environmental policy as it increasingly also represents the problems with land use in Latin-America and South-East Asia.

The application of LCIA methods to EEIO has remained rare. Most of the applications have been on climate change or only fossil carbon dioxide emissions (Peters et al., 2007). The combined use of EEIO, LCIA and sustainability assessment can broaden the perspective in all three subfields. For example, the observed differences between expert judgment and LCIA results in ecotoxicology (IV) indicate that the long term effects of current emissions should be taken into account in risk assessment. On the other hand the exclusion of bioaccumulation from LCIA models was found to result in reduced impact scores for classical persistent organic pollutants. Therefore bioaccumulation probably should be included in LCIA. Similar observations hold for the application of land use impact assessment, for example the biodiversity benefits of grazing (II) should be further investigated and possibly updated in both LCIA models and sustainability assessment.

The level of aggregation is a critical issue in IOA: with increased disaggregation, the accuracy of the results generally increases, but the data availability decreases (Lenzen, 2001). Historically the extent of disaggregation in IOA has been governed by the needs of economic assessment and maintaining statistics. However with the application of environmental issues, more disaggregation is necessary in some parts of the economy while other parts can be aggregated more. The use of sensitivity analysis can guide in which parts of the economy to focus additional data collection.

The methods applied in this work for a single nation EEIO model could also be applied to a MRIO model of the world. Two such models have recently become publicly available: the EORA (Lenzen et al., 2012) and the WIOD (Timmer, 2012). An interesting topic would be to add the environmental impacts of land use, biodiversity and ecotoxicity also to those models. Also the methods of SPA, SDA and sensitivity analysis could be used to identify the main nodes and pathways responsible for global land use and biodiversity loss. This could provide important background information for environmental policy.

The combination of EEIO and LCIA makes it possible include global impacts better also in multiple criteria decision analysis (MCDA).With an increasing amount of readily characterized EEIO tables, a brief analysis of the global supply chain impacts of decisions ("footprint" calculations) can be made with very little time investment based on cost data (Hendrickson et al., 2005; Suh, 2009). Therefore there is no excuse for not including these impacts if they are considered relevant for the decision at hand. To date most of the applications of MCDA have focused either on local environmental issues (Huang et al., 2011) or on using process-LCA (Myllyviita et al., 2012). Including global environmental footprints routinely in MCDA might help to promote ecological intelligence in decision making (Goleman, 2009). However the application of EEIO and LCIA to decision making has some significant limitations as well, these are discussed in the next chapter with their possible remedies.

4.2 Limitations of the approach and recommendations for further research

4.2.1 Lack of dynamics

The EEIO models contain a detailed description of a static situation in a given time period. Although they can be used to highlight hot-spots and key pathways, they cannot directly model the consequences of a decision. From the viewpoint of decision making, this is a serious limitation.

To illustrate the point, at the time of writing, the Finnish government was considering whether to finance a construction of a cruise liner with 50 MC. The main argument is that (based on economic input-output calculations) the construction of a 1 GC cruiser would have considerable indirect employment effects, but the dock would need a loan for operating capital during construction. Could the methods applied in this study be used to evaluate the overall sustainability of the loan? Using EEIO the projected employment figures could be supplemented with carbon footprints, resource use and a variety of emissions. LCIA could be used to convert the results into impact to human health, ecosystem quality and resource depletion. The results could then be compared with those of other industries in the economy to give a comparison, whether the use of government funds in this way would be efficient compared to other alternatives for increasing employment. But would these footprint metrics answer the question about the sustainability of the investment?

By definition, sustainable development is a dynamic process and therefore a dynamic model would be more appropriate in quantifying it. Dynamic input-output models have a long history in economic assessment (Leontief, 1951), although there is no general agreement on the validity of assumptions needed to simulate development (Nakamura and Kondo, 2009). In a classical dynamic input-output, the system is "closed" in regard

to consumption and investments. In such a closed model, a purchase of a good will have impacts also on salaries, which will affect future levels of consumption. At the same time, the purchased good will enter a capital stock of goods, which will influence the amount of productive capital available in the future. On the other hand, as long as the good remains in the stock, there is less demand for purchasing new similar goods.

In a dynamic input-output model (Figure 9), consumption will reduce future consumption for durable goods. If a good is purchased now, it is not demanded in the next time step. However, consumption will also increase production levels, which will increase future consumption by paid salaries and increased marketing. This future consumption will then increase future production, resulting in economic growth over time. Also increased production levels increase the accumulation of productive capital, which can increase future production levels further. (In some cases, the item of consumption serves as productive capital in the future, as in the case of the cruise ship, which will increase the production potential of passenger ferry transport.)



Figure 9. A simplified representation of a dynamic input-output model (solid lines) with environmental feedback mechanisms (dotted lines). For simplicity, the accumulation of capital is not explicitly presented but is present in the effect of current level production increasing future production through increased capital formation.

Traditionally dynamic input-output models have not included environmental extensions, with a few exceptions such as the Dutch DIMITRI model (Faber et al., 2007). In the cases where environmental impacts have been included, the feedback between environment and economy has not been included. If they were included, the current production level would increase environmental impacts and reduce the

amount of natural resources available. The increased environmental impacts would decrease consumption through impacts on human health and decrease also the natural capital made available by functioning ecosystems. Decreased availability of resources and ecosystems would then decrease the possibilities of economic production (Figure 9).

Applying the descriptive model to the cruise ship example, an investment in a cruise ship now, would result in a reduced demand for cruise ships in the future. It would also "lock up" non-renewable resources from other uses until the ship is eventually recycled. The increased capacity of cruise ships would increase the production possibilities for cruise tourism, which would consume additional fossil fuels for the duration of the operation of the ship. This will result in earlier depletion of fossil fuels and an earlier shift to alternative fuels, which may be more costly. The increased economic activity would increase salaries and future consumption, but on the other hand, government would have less money available for other investments (such as social or health services). A full analysis would of course require implementing the model into a set of equations, but the benefit of applying a system dynamic perspective can be demonstrated already with a qualitative thought example.

Previous combined economic-ecological models did not benefit from sophisticated in environmental impact assessment models. The classical "Limits to growth" study included the feedback between environmental pollution and consumption through a coarse connection to potential food production (Meadows et al., 1972). The current LCIA impact assessment models would make it possible to make a more scientific and transparent link by converting pollution into effects to mortality, reproduction and ecosystem damage (Goedkoop et al., 2009; Guinée et al., 2011). Time dependent LCIA models such as those already used in ecotoxic pressure characterization (Huijbregts, Struijs, et al., 2005) would make possible to explicitly include the connection between environmental pollution in the present and economic growth in the future.

As such much of this discussion is still speculative, since no such dynamic models have been made. In addition, constructing such a dynamic model would require a considerable amount of assumptions and uncertainties. As a consequence the transparency and reliability of static-IOA would be lost. Therefore the proposed dynamic model could not be considered as a substitute for traditional IOA, but as an additional forecasting tool.

4.2.2 The LCIA indicators do not represent the definition of sustainability

A problem in applying LCIA impact indicators to monitor sustainable development is that the current LCIA models do not conform to the definition of sustainability. The most common definition of sustainable development is "providing for the needs of today without reducing the possibilities of next generations for providing for themselves" (Brundtland, 1987). It contains an explicit trade-off between the needs of today and the needs of next generations. On the contrary, the approach of LCIA has been to integrate impacts over time and space (Finnveden et al., 2009). In the process the trade-off setting between current and future generations is lost. As a consequence impacts happening slowly over millennia are given similar weight as effects occurring acutely in the present. Some approaches have been made to take into account the different time scales, for example by limiting the scope of analysis to the next 20 or 100 years (Goedkoop et al., 2009).

In order to assess sustainable development over time, the impact assessment models should be able to evaluate the impact of multiple stressors and occurring over time. In principle, the models used to calculate the characterization factors (eq. 20) are capable for simulating development over time (Mattila and Verta, 2008). And many of the current LCIA models are capable of linking the separate environmental pressures (midpoints) to overall environmental and human effects (endpoints). A problem in the analysis however is the combination of various ecological overshoots. If for example biodiversity loss from land transformation is well over planetary boundaries already (Rockström et al., 2009), how much additional damage would increased climate change or eutrophication cause?

Moving towards actual sustainability assessment includes a shift from the static indicators of LCIA and towards a modeling framework which can take into account multiple stressors over relevant time frames. This requires deeper understanding about the boundaries and thresholds of ecological systems.

4.2.3 Is negative feedback effective environmental education?

In order to support decision making, the modeling framework should provide information about the impacts which the decision makers consider to be relevant. For economic impacts the applied indicators (e.g. value added, employment) are usually positive, while for environmental impacts the indicators are negative (e.g. disability adjusted life years lost, species lost, economic costs to future generations, human rights violations).

From behavioral sciences it is well known that the framing of the decision problem can have a great effect on the interpretation of the results. This is known as a negativity bias, where negative outcomes (losses) have more significance than positive outcomes (gains) (Baumeister et al., 2001). Therefore an implicit message in LCA based sustainability assessment is that "less is good" and that companies should stay away from bad practices and parts of the supply chain. However the "award" for improving a product system is presented in reduced disabilities, less human rights violations, less chemical pollution compared to an alternative production form. In any case, the decision maker must make the decision based on mainly negative indicators using a damage minimization approach. The overall feedback structure is based on negative feedback, where information from the LCA should reduce the overall activity levels until the information is considered to be within acceptable limits.

An alternative problem structuring would start from welfare maximization approach and construct the sustainability indicators accordingly. A sustainability assessment would then measure the effect of a system to increases in welfare, education, sustainable use of natural resources and healthy ecosystem functioning. The assessment would then be based on a positive feedback, where information from the assessment would be used to increase parts of the supply network which show desirable development. This approach is currently used for example in future studies under the term backcasting, where a sustainable future vision is described and the indicators are constructed to follow development towards that goal (Robinson, 1982; Mattila and Antikainen, 2011). A new approach in life cycle thinking is to measure "ecological handprints" which track the benefits to the environment from human action (Goleman, 2012).

In principle EEIO could be used to track these positive indicators across supply chains. However since most current approaches to sustainability accounting track negative impacts (e.g. maternal mortality, HIV infections, proportion of species threatened by extinction), there is no ready set of indicators available. Application could therefore start with a few positive indicators (such as the forest identity (Kauppi et al., 2006)) to learn, if the reframing of the problem would result in different kinds of decisions about sustainable development.

5. Summary

The dissertation began with an analogy between economic ripple effects and the physics of surface tension. It is therefore appropriate to close with one:

Imagine seven billion people throwing 57 trillion US dollars into the global system of production and consumption. The patterns of production fluctuate, supply networks cross continents, resources are consumed and pollution generated. The overall pattern is too complicated to comprehend, but the structure creating the pattern is relatively simple, with a few main components responsible for most of the effects.

This analogy also reveals the limitations of static input-output analysis. Economies are evolving systems, so the pond does not remain static. In a sense, large fluctuations such as the total annual consumption change the boundaries of the "pond", creating a new kind of economic system. A snapshot of the system can be evaluated for structural identification, but that does not allow the prediction of system change.

The aim of the dissertation was to apply the methods of IOA and LCA to the Finnish economy, in order to see if meaningful main components could be extracted from the complex whole. Production, consumption and environmental degradation were combined into an EEIO model, with an emphasis on land use, biodiversity and hazardous emissions. Climate change, waste generation, employment and gross domestic production were included as additional sustainability indicators. Capturing the whole in a systematic EEIO framework allowed the transparent analysis of various sustainability aspects.

Although the results of this study apply to Finland, the applied methods can be used on an international scale. The recent availability of world scale EEIO models opens up the possibility of analyzing and identifying main components in the global system of production and consumption. Based on the aggregated model (I), forest industry was identified as an economic key sector, but with considerable climate and land use impacts in its supply chain. Especially the high land use intensity (km^2/\mathbb{C}) was a cause for concern, since the demand for productive land is rising with population and affluence growth. A more detailed analysis of land use impacts (II) confirmed that the forest industries were the main cause of land use impacts in Finland, when looked from the viewpoints of productive land occupation, biodiversity and use of net primary production. At the same time it was identified that some industries may be considered as highly beneficial to biodiversity (such as dairy production through grazing animals).

Observed over time, the Finnish economy was found to move towards more unsustainability (III). The ecological footprint increased from 2002 to 2005 as did the GDP. For the ecological footprint, the production and demand structures as well as demand size worked to increase environmental pressure. At the same time production and demand structure as well as production intensity evolved towards less GDP, with only demand size offsetting these impacts. This indicated that the economy was externalizing more and more of its production to other countries, resulting in less GDP but more environmental impacts. Positive development in ecoefficiency was observed in some industries (such as the use of wood in sawmilling), but overall the increased consumption level resulted in a higher ecological footprint.

In the analysis of ecological footprint and GDP it was observed, that analytical techniques (sensitivity analysis, SPA, SPD) can reveal the main components in the economy. This was then applied to other impact categories, such as ecotoxicity, human toxicity (IV), waste generation and climate change (V). Overall the result was found to be that out of the set of 23 000 economic interactions in the model, only a small fraction cause most of the effect in each indicator category. (III,IV,V) In addition, the identified main components could be used to estimate change over time with high accuracy (V). Therefore there is a good potential for making a simplified metamodel for managing sustainable development.

However the detailed input-output models provide only the "anatomy" of production, consumption and environmental networks. In order to find remedies to the evident sustainability crisis, also the "physiology" should be investigated. The hotspots identified from static models serve as an initial

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starting point, but the path towards fully dynamic models, which would represent relevant endpoints for sustainable development requires still a considerable amount of work. In this practice the experience of applied systems thinking might prove to be fruitful.

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ISBN 978-952-60-5279-3 ISBN 978-952-60-5280-9 (pdf) ISSN-L 1799-4934 ISSN 1799-4934 ISSN 1799-4942 (pdf)

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