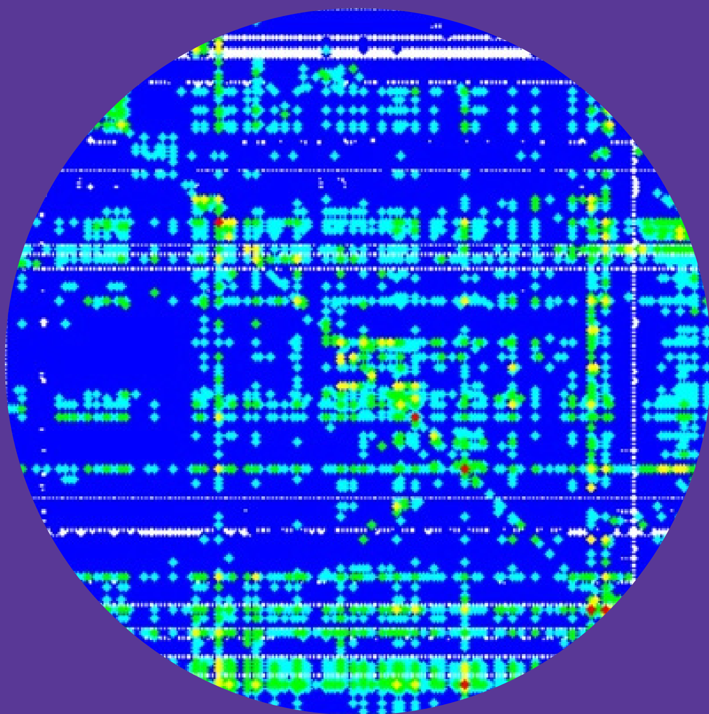
Department of Mathematics and Systems Analysis

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

Tuomas J. Mattila



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Abstract

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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

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Tiivistelmä

Tekijä
Tuomas J. Mattila

Väitöskirjan nimi
Suomen tuotannon ja kulutuksen aiheuttamien ympäristövaikutusten panos-tuotos analyysi

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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 input-output*, 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

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Publications

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

1. 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.

2. 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.

4. 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.

5. 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.

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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 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 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. Theoretical foundation

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 m^2 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.

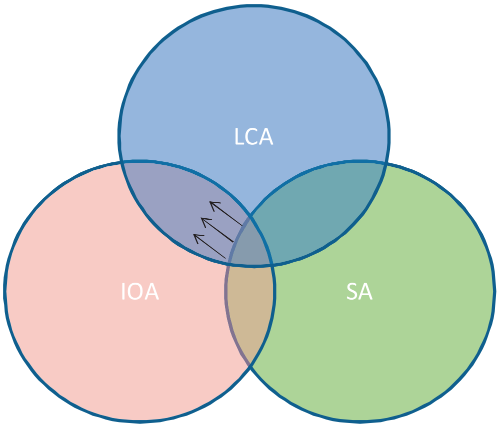

Figure 1. The methodological foundation for this dissertation is in moving the boundaries of sustainability assessment towards the detailed methods used in input-output analysis and life cycle assessment.

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 (**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:

$$\mathbf{x} = \mathbf{A}\mathbf{x} + \mathbf{f} \qquad (1a),$$

where **x** = total output (industry by 1) [M€]

A = intermediate use matrix (industry-by-industry) [M€/M€]

f = final demand (industry-by-1) [M€]

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:

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

where $\mathbf{I}$ = identity matrix (industry-by-industry)
$(\mathbf{I}\text{-}\mathbf{A})^{-1}$ = Leontief inverse (industry-by-industry) [M€/M€].

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:

$$\mathbf{p} = (\mathbf{I} - \mathbf{A})^{-1}\mathbf{v} \qquad (2),$$

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:

$$k = \mathbf{v}^T (\mathbf{I} - \mathbf{A})^{-1}\mathbf{f} \qquad (3),$$

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:

$$\mathbf{U} = \mathbf{A_p V} \leftrightarrow \mathbf{A_p} = \mathbf{UV}^{-1} \quad (4),$$

where $\mathbf{V}$ = make matrix (industry-by-product)

$\mathbf{U}$ = use table (products-by-industry)

$\mathbf{A_p}$ = product-by-product technology matrix

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

$$\mathbf{A_i} = \mathbf{V}\hat{\mathbf{q}}^{-1}\mathbf{U}\hat{\mathbf{x}}^{-1} \quad (5),$$

where $\mathbf{q}$ = total output of products (product-by-1)

$\mathbf{x}$ = total output of industries (1-by-industry)

$\mathbf{A_i}$ = industry-by-industry technology matrix

(the ^ symbol denotes a diagonal vector)

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 **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 matrix* $\mathbf{A_{pp}}$ *(product-by-product)*	Use matrix **U**	Product final demand **e**	Product output **q**
Industries	Make matrix **V**	*Technology matrix* $\mathbf{A_{ii}}$ *(industry-by-industry)*	*Industry final demand* **f**	Industry output **x**
Value added		Value added **v**	GDP	
Total output	Product output **q**	Industry output **x**		
Employment		Employment **s**		
Emissions and resources		Environmental 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:

$$\mathbf{B} = \mathbf{G}\hat{\mathbf{x}}^{-1} \qquad (6),$$

where **B** = emission or resource use intensity (environmental flow-by-industry) [kg/M€]

G = emission or resource use matrix (environmental flow-by-industry) [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:

$$\mathbf{g} = \mathbf{B}(\mathbf{I} - \mathbf{A})^{-1}\mathbf{f} = \mathbf{M}\mathbf{f} \qquad (7),$$

where **g** = overall emissions caused by the final demand (environmental flow-by-1).

M = 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:

$$\mathbf{g} = [\mathbf{B_d} \quad \mathbf{B_i}]\begin{bmatrix}\mathbf{I}-\mathbf{A_d} & \mathbf{0}\\ -\mathbf{A_i} & \mathbf{I}\end{bmatrix}^{-1}\begin{bmatrix}\mathbf{f_d}\\ \mathbf{f_i}\end{bmatrix} \qquad (8),$$

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 $\mathbf{B}_\mathrm{i}$ 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 $\mathbf{B}_\mathrm{d}$(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):

$$\mathbf{\Delta g = B(I\text{-}A)^{-1}\,\Delta f_s}f_t + \mathbf{B(I\text{-}A)^{-1}\,f_s}\Delta f_t + \mathbf{B\Delta(I\text{-}A)^{-1}\,f_s}f_t + \mathbf{\Delta B(I\text{-}A)^{-1}\,f_s}f_t \qquad (9),$$

where $\mathbf{f_s}$ = the structure of final demand [M€/M€]

f_t = total amount of final demand (scalar) [M€/M€]

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:

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

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:

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

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

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

where b and a are the elements of the corresponding matrices $\mathbf{B}$ and $\mathbf{A}$.

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):

$$S_f = \frac{\delta g_k / g_k}{\delta f_i / f_i} = M_{ki} \frac{f_i}{g_k} \qquad (13)$$

$$S_a = \frac{\delta g_k / g_k}{\delta a_{ij} / a_{ij}} = M_{ki} x_j \frac{a_{ij}}{g_k} \qquad (14)$$

$$S_b = \frac{\delta g_k / g_k}{\delta B_{kj} / B_{kj}} = x_j \frac{B_{kj}}{g_k} \qquad (15),$$

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 input-coefficients 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.)

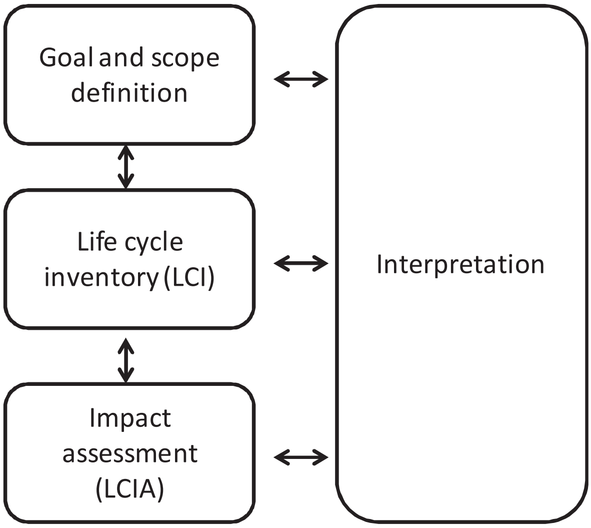

Figure 2. The four iterative stages of life cycle assessment (ISO, 2006)

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):

$$\mathbf{As} = \mathbf{f}_l \qquad \textbf{(16)}$$

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

$$\mathbf{s} = \mathbf{A}^{-1}\mathbf{f}_l \qquad \textbf{(17)}$$

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:

$$\mathbf{g} = \mathbf{Bs} = \mathbf{BA}^{-1}\mathbf{f} \qquad \textbf{(18)}$$

where $\mathbf{B}$ = 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:

$$\mathbf{q} = \mathbf{Cg} = \mathbf{CBs} = \mathbf{CBA}^{-1}\mathbf{f} \qquad \textbf{(19)}$$

where **C** = characterization factor matrix (impact per unit emission or resource use)

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):

$$\mathbf{C} = \frac{\partial \text{disability}}{\partial \text{incidence}} \cdot \frac{\partial \text{incidence}}{\partial \text{exposure}} \cdot \frac{\partial \text{exposure}}{\partial \text{concentration}} \cdot \frac{\partial \text{concentration}}{\partial \text{emission}} \qquad \textbf{(20)}$$

The partial derivatives are obtained for example from chemical transport models (concentration/emission response) (Mackay, 2001) and from dose-response 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):

$$\mathrm{i} = \mathbf{w}\hat{\mathbf{n}}^{-1}\mathbf{q} = \mathbf{w}\hat{\mathbf{n}}^{-1}\mathbf{CBA}^{-1}\mathbf{f} \qquad \textbf{(21)}$$

where **n** = vector of normalization factors

w = vector of impact category weights

i = overall impact score

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:

$$i_k = \sum_{j=1}^{n} w_j \, v_j(k_j) \qquad (22)$$

where i_k = the overall index for alternative k

w_j = the weight of attribute i (n attributes)

$v_j(k_j)$ = a value function converting the attribute value k_j to a utility value in the range 0…1

k_j = attribute j for alternative k

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

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

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 **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:

$$\mathrm{i} = \mathbf{w_e}\hat{\mathbf{n}}_{\mathbf{e}}^{-1}\mathbf{q_e} + \mathbf{w_s}\hat{\mathbf{n}}_{\mathbf{s}}^{-1}\mathbf{q_s} + \mathbf{w_c}\hat{\mathbf{n}}_{\mathbf{c}}^{-1}\mathbf{q_c} \qquad (24)$$

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 ($\mathbf{(I-A)^{-1}}$ in eq. (1)).

Looking at employment multipliers, the forest industries were found to have a relatively low intensity (e.g. working hours/€ of production) especially compared to primary production but also to metal industry and construction (Table 2). The total employment figures ($\mathbf{M}$ 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 CO_2e/€		work hours/€		m^2/€		€/€	
		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).

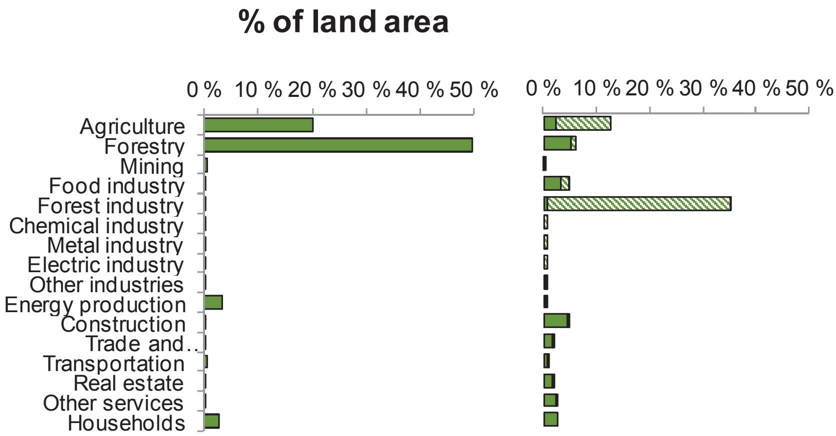

Figure 3. Land use allocated to industries (left) and products (right) in Finland 2002 as a fraction of the domestic inland surface area (305,000 km^2). The shaded region in the right figure is the extent of exported domestic land area.

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€), 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 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)).

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.

Product	**Land use km² €⁻¹**	**Biocapacity km² M€⁻¹**	**HANPP kt M€⁻¹**	**EDP Finland**	**EDP CE**
Other animal products	**199**	129	0.6	**15,000**	-5,200
Roundwood	75	**200**	**9.2**	850	520
Sawn wood	28	73	3.4	310	190
Crops	18	52	5.0	710	**760**
Dairy products	5	1	1.1	-1,600	130
Animal and vegetable oils	4	10	1.0	120	150
Refined petroleum	0.1	0.2	0.01	-1	0

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 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 inter-industry 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.

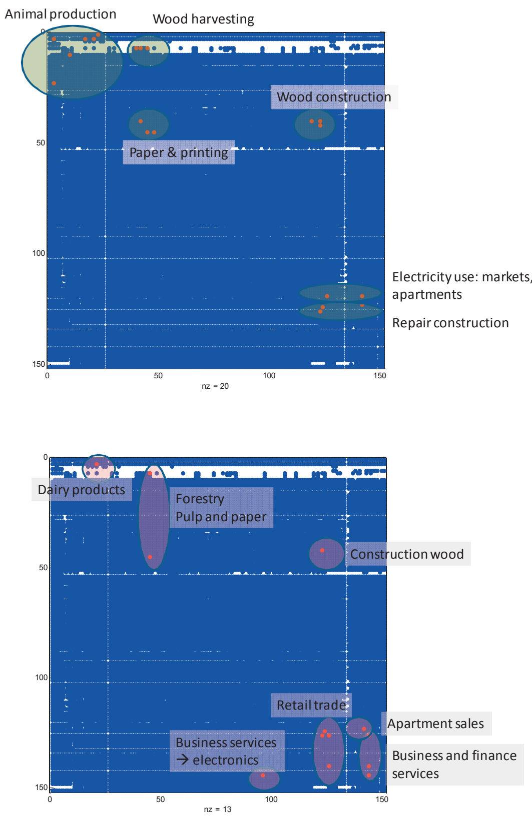

Figure 4. A graphical representation of the most relevant interindustry connections in the Finnish input-output tables identified with sensitivity analysis. Top: ecological footprint, bottom: gross domestic product.

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 €/cap/yr). In addition, most of the products were 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).

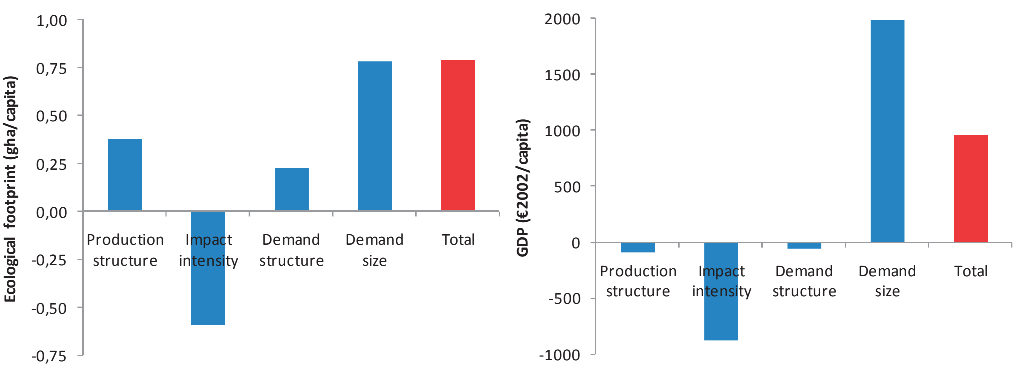

Figure 5. Decomposition of the change in (a) ecological footprint and in (b) GDP in the Finnish economy between 2002-2005.

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	Sawmilling	Forestry	-
3	0.06	Carbon	Δf	Electricity	-	-	-
4	-0.06	Carbon	ΔB	Electricity	-	-	-
5	-0.05	Carbon	ΔB	Renting and owning apartments	Electricity production	-	-
6	-0.04	Forest	ΔA3	Residential construction	Carpentry	Saw-milling	Forestry
7	-0.04	Carbon	ΔB	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	ΔA1	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 €$_{2002}$/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).

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.

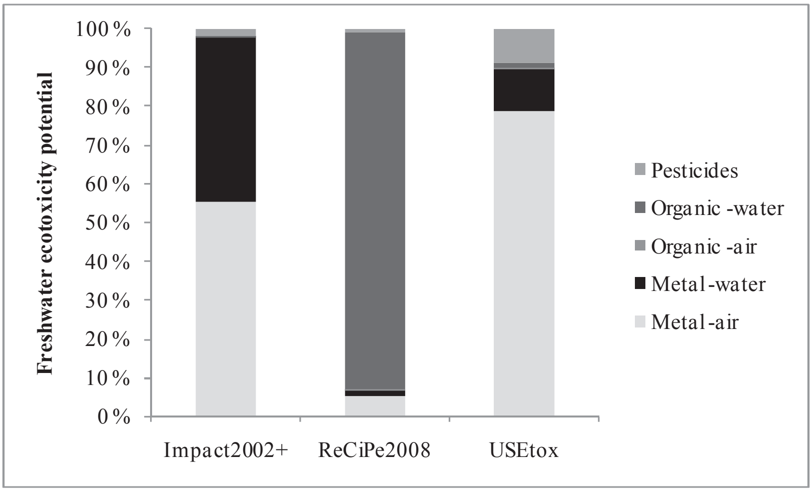

Figure 6. Comparison of the share of toxic load from substance groups in the three assessed LCIA models.

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, 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).

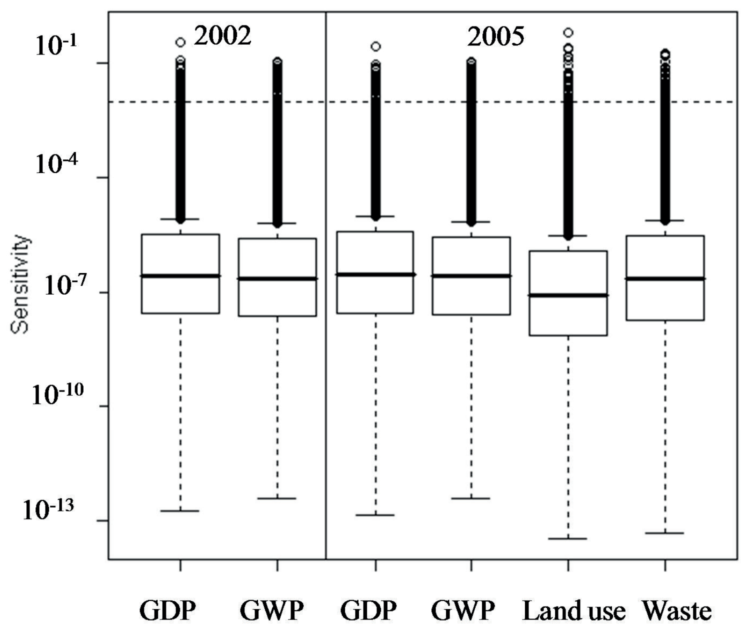

Figure 7. The distributions of sensitivity indices (S_a, S_b and S_y combined) for gross domestic product (GDP), global warming potential (GWP), land use and waste generation. The box and whiskers plot describes the median of the distribution, the 25% percentile and the 75% percentiles and the whiskers 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 input-coefficients (S_a, n=15). Emission intensities for electricity production, iron 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.

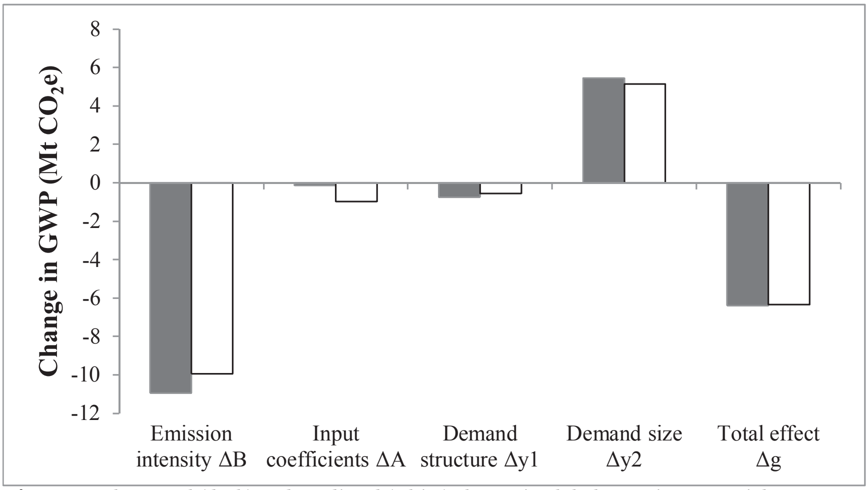

Figure 8. The actual (dark) and predicted (white) change in global warming potential decomposed into components of the input-output model.

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. Discussion

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 M€. The main argument is that (based on economic input-output calculations) the construction of a 1 G€ 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.)

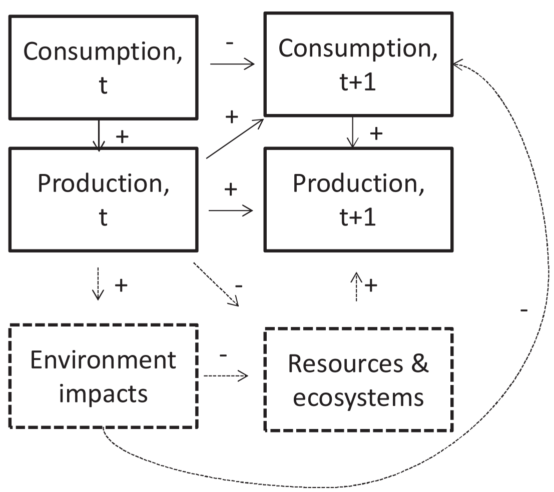

Figure 9. A simplified representation of a dynamic input-output model (solid lines) with environmental feedback mechanisms (dotted lines). For simplicity, the accumulation of capital is not explicitly presented but is present in the effect of current level production increasing future production through 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/€) 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 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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An Environmentally Extended Input-Output Analysis to Support Sustainable Use of Forest Resources

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Abstract: The use of environmentally extended input-output analysis was demonstrated for quantifying the overall sustainability impacts of forest industries in the Finnish economy in 2005. Direct greenhouse gas emission, land use, employment and import impacts of economic sectors were transformed into impact intensities of products. The intensities were used to construct a final demand based emission inventory demonstrating the relative importance of export, investment and consumption activities in causing environmental and social impacts. The calculations were presented using an aggregated input-ouput table, which makes it possible to repeat the calculations using standard spreadsheet software. Therefore the study can be used as an accessible primer to the use of input-output methods in sustainability assessment.

Keywords: Environmentally extended input-output analysis, land use, climate change, multiple criteria decision analysis.

INTRODUCTION

The global forest sector is under structural change due to a relocation of paper production plants and an increase of forest bioenergy and wooden construction materials production. Evaluation of the sustainability impacts of this change in production requires the development of different quantitative scientific tools (c.f. [1, 2]). The evaluation is limited by the complexity of supply chain interactions and also by the different definitions given to sustainability in general. Sustainable use of forest resources become an important topic in forestry after the United Nations (UN) Conference on Environment and Development in Rio [3]. Sustainability is a multi-dimensional concept and it includes economic, ecological, social and cultural dimension. As such, the different dimensions of sustainability are vague. In practical applications, the general criteria are typically measured by more concrete indicators such as economic profitability, biodiversity, climate change impacts and employment. Therefore a tool for evaluating sustainability of structural change should be able to present the different dimensions of sustainability simultaneously.

The environmental, economic and social impacts of forest industries are well known. However the direct impacts of an industry may only be a tip of the iceberg, compared to the indirect impacts caused in the supply chain [4]. In environmental sustainability assessment of products, life cycle assessment has been the method of choice for quantifying these impacts [5]. In macroeconomics, these supply chain impacts are quantified as multiplier effects and are routinely calculated for employment and value added [6,7], but in an increasing manner also for environmental emissions [4,8,9]. There is also an increasing literature on combining the product and economic level analyses into hybrid input-output life cycle assessment [10].

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Environmentally extended input-output analysis (EE-IO) was described already in 1970 by Leontief [11] (who originally invented economic input-output analysis and was awarded with the Nobel prize of economics). EE-IO incorporates environmental emissions or processes to the economic tables of the national accounts. A few examples of the application of EE-IO into national statistics are the NAMEA (national accounting matrix including environmental accounts) tables developed in the Netherlands [12] and the German PIOT (physical input output table)[13]. In addition a few EE-IO datasets have been developed by independent researchers for example for USA [8], Japan [14] and Finland [15]. Also the global trade analysis project (GTAP) includes emission inventories in addition to economic interactions between nations [16]. In recent years the combination of input-output tables with life cycle assessment [5] has increased the use of this method for product level sustainability assessment [10,17] as well as the analysis of the environmental impacts of consumption and global trade [18].

The purpose of this paper is to demonstrate the use of environmental input-output analysis in the sustainability assessment of forest industry. We present how EE-IO can be used to assess supply chain effects with multidimensional sustainability impacts in a concise and transparent manner. The following section gives an overview of the basic EE-IO equations, which are then applied to a simplified case of the Finnish economy in 2005, with an emphasis on forest industry. Results are presented on the ecoefficiency of forest industry, on the overall impacts of forest products in the national economy and on the possibility of using input-output analysis data as a background for detailed life cycle assessment.

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MATERIALS AND METHODS

Overview of Environmental Input-Output Analysis

This chapter presents and overview of the economic input-output analysis and its extension to environmental systems. A more detailed review of the development of the modeling approach is given for example in [7] or [9]. EUROSTAT has published a guide for compiling and using economic input-output tables [19] and economic tables are readily available for application from EUROSTAT [20] and OECD [21].

The main research topic of economic input-output analysis is the relationship between the scale of production output (x) and the final demand of products (f). This relationship can be expressed with an equation, which also takes into account the demand for products caused by the production itself:

$$x = A \cdot x + f \tag{1a}$$

where the intermediate demand is expressed in a matrix A. It describes the amount of products needed from other (and from the producing) sectors for the production of one unit of product. It is a square matrix with dimensions equal to the amount of sectors in the model. (The scale of production x and the final demand f are vectors with a length equal to the amount of sectors.) In the system of national accounts, the columns of the input-output table describe purchases (debts) and rows describe sales (credits) [7]. The input-output table (in M€) is transformed into the A-matrix (in M€/M€) by dividing each column (purchases by sector from all sectors, i.e. input, M€) with the sum of the corresponding row and final demand (total sales or production, i.e. output, M€). A basic assumption in input-output analysis is that the relationship between demand and production is linear and no thresholds exist. Therefore the model applies well to allocating production impacts to demand categories, but has limited scope for prediction, since the assumption of linear response to change is easily violated in nonlinear real-world systems if the change becomes sufficiently high.

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

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

where the matrix $(I-A)^{-1}$ is known as the Leontief inverse. Each column of the matrix describes the 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 [7] and are used for example to identify the key sectors of an economy [6].

While the economic model described in equations (1a) and (1b) is conceptually simple, it may have very large matrices. For example the Japanese [14] and US [8] tables include more than 400 sectors. National accounts in Europe are usually reported in a 58 sector resolution. With increasing resolution, the amount of overall production remains the same, but the allocation of production activities to demand gets more accurate [22]. However understanding of model complexity become limiting when matrix size is increased. In addition the inversion of large matrices requires special software, which are not as commonly accessible as spreadsheet calculators. In order to keep the model understandable and calculations easily repeatable, in the input-output table used in this study only the forest industry sectors were kept at a detailed level, while other sectors were highly aggregated.

If the emissions of economic sectors are known, equation (1b) can be extended into a environmental input-output analysis by simple matrix multiplication [17]:

$$g = B(I - A)^{-1} f \tag{2}$$

where g are the overall emissions caused by the final demand and B is a matrix of emission intensities (emission type by industrial sector). The emission intensities can readily be calculated from reported annual emissions by dividing each column with the production output of the corresponding sector. 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.

In the form presented in equation (2) the model does not include the emissions caused outside the national boundaries, which are sometimes called the "emissions embodied in imports" [18]. They can however be included in the EE-IO framework in order to construct consumption based emission inventories, where emissions are allocated to the consumers of goods. In the consumption based inventory, emissions embodied in imports are added to the national emissions and the emissions associated with export production (whether of domestic origin or embodied in imports) are removed from the inventory through the use of input-output analysis. The resulting figure describes the emissions caused globally by the production of goods for consumption in the studied country. Multiple region input-output (MRIO) models have been used to make these assessments with notable differences in the magnitude of emissions and mitigation options [23]. As the model presented in equation (2) does not include imports, it cannot be used to construct a consumption based inventory directly. However it can be used to quantify the extent of domestic impacts caused by export activities, as demonstrated later in this article.

An interpretation of equation (2) is that it describes the emissions (or other impacts) caused throughout the supply chain of producing the final demand items. If the final demand vector is replaced by a unit matrix (or left out completely), equation (2) will give the total emissions caused in meeting a unit of demand. The complex interactions and economic loops of production are completely captured with the Leontief inverse and there is no "cut-off" of processes [24], which limits conventional process based life cycle assessment [10]. Therefore environmental input-output analysis can be thought of as a simple method for producing life cycle assessments of services and products [8]. Compared to traditional process based life cycle assessments, it can be made with less resources, but it also has been found to be more comprehensive, since emissions caused by services and machinery are also included [9,10].

Because of the simple linear structure, operations from linear algebra can be used to analyze the system. Some of the more common analyses are the analysis of sensitivity and contribution of different industries to emissions [24], decomposition of observed temporal change [25], identification of key sectors [6] and the extraction of main impact pathways [26]. These advanced topics of analysis were not included in this study. Instead the focus is on demonstrating the basics of environmentally extended input-output analysis and its application to sustainability analysis.

In this study we applied the basic environmentally extended input-output analysis described above to the Finnish economy in the year 2005 [15]. Forest industries are compared to other economic sectors by their direct and multiplier impacts. Also the contribution of different final demand categories (consumption, export and investments) to sustainability indicators is studied. In addition, the use of environmental input output analysis in hybrid life cycle assessment [10,17] is demonstrated.

Data Sources for Emissions and Economic Interactions

The data used in this study was based on the ENVIMAT-project [15], where a highly detailed environmentally extended input-output table of Finland 2002 and 2005 was constructed. However for the purposes of demonstrating the role of forest industry in sustainability, the detailed data was aggregated. This chapter describes the data sources used in the ENVIMAT-study and the procedure of (and limitations caused by) aggregation.

The ENVIMAT study used the official economic supply and use tables [27] as the starting point. Those are regularly assembled by the statistical office as a part of the system of national accounts and are available for several countries from OECD and EUROSTAT databases. However since their function is economic accounting, the classification of sectors was adjusted to better suit environmental assessment. Several service sectors were merged and some industrial sectors were disaggregated. Details are given in [15]. Some of the most important new classifications were the separation of animal farming from crop production and fertilizer industry from basic chemicals. This resulted in a classification of 150 industries, which was used to construct an industry-by-industry input-output table using the fixed product sales structure assumption [19]. In this study this high level of detail was kept only for the forest industries, all other industries were aggregated into 14 macroeconomic sectors.

The aggregation of detailed data introduced errors in calculation [22]. For example, in the aggregated table agriculture included animal, grain and vegetable production. Starch purchased by the pulp and paper industry was accounted as input from the agricultural sector, and therefore a fraction of the methane emissions of animal agriculture was allocated with that purchase. Similarly purchases from metal industry resulted in an allocation of emissions from both ferrous and nonferrous metals. More accurate results could be obtained by the 150 industry division of the ENVIMAT -study [15], but the representation of calculation would have been more complex. Therefore inaccurate, but relatively simple results were presented in this study.

In addition to aggregation of economic sectors, also the amount of environmental interventions in the ENVIMAT-model was reduced for this study. From the environmental impacts considered in the ENVIMAT (e.g. climate change, acidification, human toxicity, ecotoxicity, eutrophication, land use, ozone depletion and depletion of mineral resources) only climate change and land use were chosen to represent environmental sustainability, while employment and value added were chosen for social sustainability. Several other alternatives for environmental indicator combinations would have been possible, but land use and climate change were chosen to represent the main differences between forest and other industries. The greenhouse gas emissions were based on the greenhouse gas inventory and energy statistics of the statistical office [28]. Separate gases (carbon dioxide, methane, nitrous oxide and F-gases) were aggregated into global warming potential (CO_2e) using the IPCC 2007 characterization factors [29]. Land use was estimated by allocating CORINE land cover data [30] to economic sectors based on agricultural and forest statistics and physical and economic production [31]. Employment and value added were based on the national account tables [27].

The aggregated environmentally extended input-output table is presented in Table **1**. The aggregated table is supplemented with final demand in three subcategories (domestic consumption, investments and exports) and the satellite accounts of greenhouse gas emissions, gross domestic product, employment and land use are presented below the economic input-output table. Imports to industries were not aggregated into the table, but were kept as a separate satellite account for transparency. Also as mentioned before, the emissions embodied in imports were not included in the analysis.

RESULTS

Economic and Environmental Multiplier Effects of the Finnish Forest Industry

The central role of forest industries in the national economy could be observed from the input-output table (Table **1**). Most of the forest industry sectors have a large share of export production and a high volume of export. Especially pulp and paper industry is strongly connected to the rest of the economy and within the forest industries. The main interactions of that sector are within itself (sales between companies belonging to the same industrial sector), and with chemicals, metals, forestry and transport. In addition the pulp and paper industry purchases goods from and sells products to other forest industry sectors.

The columns of the Leontief inverse provided a quantification of the overall structure presented in the previous paragraph (Table **2**). For purchases of sawn wood, one M€ of final product would result in more than 2.1 M€ of economic activity (the average multiplier for all sectors was 1.8). From the corresponding column it can be read, that most of the activity is focused on the sawmilling industry, but a significant portion occurs also in forestry and logging, trade services and transportation. A similar pattern could be observed for the pulp and paper industry, however with less economic impacts to forestry and more to chemicals and metals industries and to transport. The high multiplier for the pulp and paper industry's own demand demonstrates the

Table 1. An Aggregated Environmentally Extended Input Output Table of the Finnish Economy, Focusing on Forest Industries. Satellite Accounts Include Imports into Industries, Greenhouse Gas Emissions (GHG), Gross Domestic Product (GDP), Employment and Land Use. The Input-Output Table Describes Sales from the Row Industry to the Column Industry

FINLAND 2005	Input-Output at Basic Prices M€	1 Agr	2 For	3 Min	4 Food	201 Saw	202 Veneer	203 Carp	204&205 O wood	211 P&P	212 P.B.	221&222 Printing	6 Chem	7 Metal	8 Electric	9 Other	10 Energy	11 Const	12 T & R	13 Trans	14 RE	15 O Serv	Domestic Consumption	Gross Fixed Capital	Exports
1	Agriculture	963	0	0	2,246	1	0	1	0	3	0	3	7	8	19	3	2	7	80	13	13	58	622	163	371
2	Forestry and logging	0	580	2	0	1,183	186	72	3	833	0	0	2	0	0	2	33	9	0	1	0	0	161	101	57
3	Mining	1	1	143	3	1	0	3	0	127	0	0	193	150	4	2	180	162	3	1	1	6	8	5	138
4	Food products	471	4	6	1,380	11	3	8	1	68	4	29	101	124	189	28	24	44	916	93	85	376	3,312	99	1,240
201	Sawn wood	9	0	21	2	122	19	386	56	225	0	1	6	5	6	60	139	332	16	3	36	23	22	-23	1,440
202	Veneer sheets	0	0	0	0	2	18	24	3	1	1	0	2	1	1	73	1	21	3	0	1	4	12	2	731
203	Builders carpentry	1	0	1	2	4	1	96	2	9	2	2	4	8	10	6	5	1,045	16	7	7	34	15	14	540
204	Other wood products	0	0	3	9	2	0	1	0	22	1	1	25	37	12	3	0	7	1	0	0	38	3	2	26
211	Pulp, paper &cardboard	9	3	5	32	10	6	9	1	1,858	228	453	137	115	174	28	22	52	186	76	70	309	99	178	7,596
212	Paper and paperboard	1	0	1	94	3	1	2	0	91	36	14	22	21	51	8	3	11	51	10	9	57	81	14	288
221& 222	Printed goods	11	1	4	34	5	2	4	0	31	6	504	43	71	164	17	14	49	581	102	94	863	1,110	128	567
6	Chemical industry	203	30	78	207	27	30	86	2	649	21	46	1,972	635	243	196	126	1,655	712	888	107	516	1,044	189	7,875
7	Metal industry	66	3	84	185	46	19	77	4	501	19	71	505	6,705	516	601	242	1,432	247	69	41	493	453	1,469	13,781
8	Electric industry	19	2	9	54	13	4	9	1	57	4	44	98	355	1,867	80	31	328	175	94	92	534	252	765	17,589
9	Other manufacturing industry	3	1	2	10	3	1	8	0	11	2	26	39	235	34	653	5	241	294	187	20	277	760	1,059	2,895
10	Energy	54	15	25	120	69	40	21	2	453	9	33	310	275	71	53	73	139	491	128	817	975	852	64	58
11	Construction	35	0	1	7	5	1	1	0	12	1	6	11	20	28	5	2	1,464	181	426	2,719	311	466	15,125	41
12	Trade, hotels and restaurants	556	44	17	275	333	60	37	2	430	11	83	350	683	375	242	90	2,378	2,700	1,152	704	3,990	15,899	1,387	1,370
13	Transport and communi-cation	51	25	135	553	294	67	87	15	1,078	45	153	701	945	694	182	82	437	2,923	1,878	362	3,559	5,553	84	3,778
14	Real estate activities	5	11	15	100	10	3	19	2	54	9	109	124	265	280	114	59	138	998	523	203	1,999	17,598	1,481	0
15	Other service activities	309	53	55	441	66	18	79	5	342	37	549	776	1,466	2,959	317	300	933	2,509	1,246	2,358	8,301	44,789	2,555	3,352
SATELLITE ACCOUNTS																									
Imported products	M€	252	33	143	1,061	288	123	218	17	1,498	129	415	6,945	7,685	7,166	1,840	897	1,575	2,858	1,815	583	4,219			
GHG	Gg CO2 eq.	7,262	321	297	495	75	71	70	8	3,766	16	32	8,056	6,797	531	347	19,321	1,767	2,491	8,551	426	4,137			
GDP	M€	1,626	2,422	370	2,333	386	307	580	79	3,248	300	1,850	4,819	7,717	7,522	2,237	2,644	8,053	16,641	14,180	14,945	44,337			
Employment	1000 my	101	20	6	39	9	7	12	2	28	5	32	54	126	64	53	13	165	383	170	40	1,068			
Land use	km2	61,477	152,024	1,329	35	83	7	4	1	149	2	3	156	61	2	3	10,515	9	209	1,978	8,028	305			

feedback processes in the sectors supply chain: purchases of raw materials for pulp and paper will result in the increased use of pulp and paper. Therefore the industry has a limited potential for generating its own demand, which is a property of economic key sectors [6]. In comparison to other key sectors, forest industry has less internal feedback than metal industry but more than construction industry. However the external connections are so strong that pulp and paper and sawmilling industries stimulate more economic activity than the metal or construction industries. In addition, compared to metals industry their multiplier effects are focused more on the domestic economies and not on import products.

The inclusion of environmental and social impacts makes the comparison of industries multidimensional. The direct impact intensity *B* can be obtained by dividing the impacts (presented in the satellite accounts of Table **1**) with the amount of total production (the row sum of both intermediate and final demand) of each industry. These direct intensity factors (Table **3**) can be compared with the total emission multipliers, which include also the supply chain. (The total multipliers are calculated by multiplying the direct factors with the Leontief-inverse as in equation (2), but leaving out the multiplication with final demand.) The primary production industries, agriculture and energy production have the highest greenhouse gas intensities in both terms, direct and total. Similarly agriculture has the highest employment multiplier and forestry has the largest land use multiplier in both terms. For process industries however, the total multipliers are considerably larger than the direct multipliers. For the pulp and paper industry, both total greenhouse gas emissions and imports are twice as high as the direct impacts, employment is threefold and land use is more than two orders of magnitude higher than direct impacts.

One of the peculiarities of EE-IO could be observed from the comparison of land-use intensities between forest sectors. The use of economic allocation (i.e. impacts are allocated to the purchasers based on the value of the purchases) results in significantly lower land use for pulp and paper than to sawmill products. This is not caused by differences in wood production yields but in the prices of products instead. Since timber is significantly more expensive than pulpwood, a

Table 2. Columns of the Leontief Inverse for the Aggregated Input-Output Table of Finland 2005 for Selected Industries. The Elements Describe the Amount of Economic Activity Generated in the Row Industries Through the Production of One Unit of Production from the Industry in the Column

		201	**211**	**221&222**	**7**	**11**
Code	**Industry**	**Saw**	**P&P**	**Print**	**Metal**	**Cons**
1	Agriculture	0.01	0.01	0.01	0.00	0.01
2	Forestry and logging	0.52	0.12	0.02	0.00	0.02
3	Mining	0.00	0.02	0.00	0.01	0.01
4	Food products	0.01	0.01	0.01	0.01	0.01
201	Sawn wood	**1.05**	0.03	0.00	0.00	0.03
202	Veneer sheets	0.00	0.00	0.00	0.00	0.00
203	Builders carpentry	0.00	0.00	0.00	0.00	0.06
204	Other wood products	0.00	0.00	0.00	0.00	0.00
211	Pulp, paper & cardboard	0.01	**1.20**	0.14	0.01	0.01
212	Paper and paperboard	0.00	0.01	0.01	0.00	0.00
221&222	Printed goods	0.01	0.01	**1.13**	0.01	0.01
6	Chemical industry	0.03	0.09	0.03	0.04	0.11
7	Metal industry	0.03	0.08	0.04	**1.33**	0.12
8	Electric industry	0.01	0.01	0.02	0.02	0.02
9	Other manufacturing industry	0.01	0.01	0.01	0.01	0.02
10	Energy	0.03	0.06	0.02	0.02	0.02
11	Construction	0.01	0.01	0.01	0.01	**1.08**
12	Trade, hotels and restaurants	0.16	0.07	0.05	0.05	0.16
13	Transport and communication	0.14	0.15	0.08	0.07	0.07
14	Real estate activities	0.02	0.02	0.04	0.02	0.02
15	Other service activities	0.07	0.08	0.19	0.10	0.10
	Domestic	2.12	1.98	1.82	1.73	1.88
	Direct imports	0.18	0.26	0.19	0.42	0.22
	Total	2.3	2.2	2.0	2.1	2.1

Table 3. Impact Multipliers Calculated from the Environmentally Extended Input Output Table of Finland 2005 Presented in Table 1

		GHG kg CO_2e/€		Employment Work Hours/€		Land Use m^2/€		Imports €/€	
		Direct	Total	Direct	Total	Direct	Total	Direct	Total
1	Agriculture	1,58	2,31	0,19	0,31	13,41	17,99	0,05	0,19
2	Forestry and logging	0,10	0,17	0,05	0,07	47,14	57,53	0,01	0,03
3	Mining	0,26	0,60	0,04	0,10	1,17	2,30	0,13	0,27
4	Food products	0,06	0,96	0,04	0,18	0,00	5,80	0,12	0,27
201	Sawn wood	0,03	0,31	0,03	0,10	0,03	24,75	0,10	0,18
202	Veneer sheets	0,08	0,40	0,06	0,11	0,01	12,97	0,14	0,22
203	Builders carpentry	0,04	0,28	0,06	0,12	0,00	8,25	0,12	0,24
204	Other wood products	0,04	0,26	0,09	0,14	0,00	8,36	0,08	0,18
211	Pulp, paper & cardboard	0,32	0,75	0,02	0,07	0,01	5,85	0,13	0,26
212	Paper and paperboard	0,02	0,35	0,05	0,09	0,00	1,78	0,15	0,27
221&222	Printed goods	0,01	0,22	0,06	0,12	0,00	0,87	0,09	0,19
6	Chemical industry	0,46	0,68	0,03	0,06	0,01	0,29	0,40	0,49
7	Metal industry	0,25	0,47	0,04	0,08	0,00	0,24	0,28	0,42
8	Electric industry	0,02	0,13	0,02	0,06	0,00	0,23	0,32	0,40
9	Other manufacturing industry	0,05	0,22	0,07	0,11	0,00	0,61	0,27	0,39
10	Energy	3,75	3,92	0,02	0,05	2,04	3,34	0,17	0,24
11	Construction	0,08	0,31	0,07	0,13	0,00	1,12	0,08	0,22
12	Trade, hotels and restaurants	0,08	0,29	0,10	0,15	0,01	0,45	0,09	0,16
13	Transport and communication	0,36	0,49	0,06	0,10	0,08	0,28	0,08	0,14
14	Real estate activities	0,02	0,23	0,01	0,06	0,33	0,71	0,02	0,08
15	Other service activities	0,06	0,20	0,13	0,17	0,00	0,24	0,06	0,11

larger fraction of the land use of forestry is allocated to sawmill products than to pulp and paper manufacturing.

The Role of Forest Industry in Production and Consumption Based Inventories

Although the emissions embodied in Finnish imports [18] were not included in this study, the model based on domestic emissions can be used to study the role of forest industry in consumption based inventories through the share of export production. By dividing the emissions into final demand categories based on the amount of products purchased, an overview of the emission trade balance can be generated. The balance is incomplete, since the emissions embodied in imports are missing. However the (domestic) emissions embodied in export are valuable stand-alone indicators, since they represent the limits of local consumer choices in influencing national emissions.

The Finnish greenhouse gas emissions were 72.5 Mt CO_2e in 2005. Of this the emissions of households were 7.7 Mt, leaving 64.8 Mt for the industries [15]. Of the emissions of industries, 41% were allocated to export production, 46% to domestic consumption and the remaining 13% to investments. For the forest industries, however, the extent of export was higher. For all forest industries except the production of printed goods and paperboard, the extent of exported greenhouse gas emissions was more than 85% of the total emissions associated with the products of these industries. This high share of exported emissions was shared with the chemicals and metals industry, while services were consumed mostly within national boundaries.

When emissions were allocated from producers to products, the overall view on the main causes of emissions was changed. In the common producer-based inventory, electricity production is the main emission source, followed by transport and communication. The pulp and paper industry is shown as a relatively small emission source, especially compared to the chemicals and metals industries and transport activities. However pulp and paper production uses a significant portion of the produced chemicals, metal products, transport activities and electricity, therefore a fraction of their impact is allocated from the producers to the products of pulp and paper industry. This allocation increased the impacts of pulp and paper products into a similar order of magnitude than the impacts of producing chemicals and metals.

Application to the Sustainability Assessment of Products

Environmental input-output data can be used to fill data gaps in conventional process based life cycle assessment.

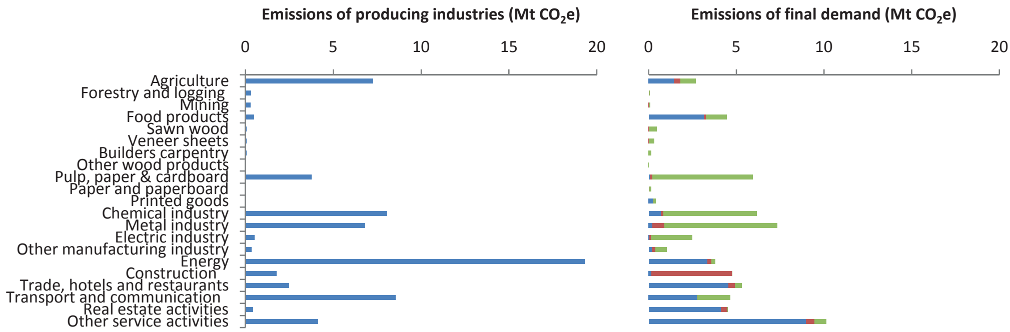

Fig. (1). The allocation of emissions reported by producing sector (left) to final demand of product categories (right). The final demand was further allocated to domestic consumption (blue), investments (red) and exports (green).

The most common source of error in process based analyses is ignoring the role of services and machinery [10,22]. Including machinery in a process-LCA is demonstrated here through an example of the environmental impacts caused by machinery in forest biomass production. For an overview of the strengths and weaknesses and the combined use of LCA and EE-IO see for example [10,22].

The costs of machinery for a small scale (30 kW maximum power) wood chip heat production unit would be approximately 20 000 €. In conventional life cycle assessment, this purchase might be ignored because of insufficient data. The order of magnitude of this purchase can however be estimated with input-output analysis by using the emission multiplier for the metals industry (0.47 kg CO_2 eq. €$^{-1}$, Table **3**). Therefore, the emissions caused by this purchase would be approximately 9.4 tons CO_2 eq., which is equal to burning three tons of light fuel oil. If using wood chips for heat instead of oil saves 8 tons of oil per annum, the emissions of machinery are quite irrelevant. However, the emissions of providing the wood for the system for 10 years amount to only about 1.5 tons of CO_2 eq. (i.e. approximately 600 m^3 of wood at 14 € m^{-3}, with the emission intensity of 0.17 kg CO_2 eq/€ from Table **3**). Therefore the magnitude of machinery may be significant and further life cycle assessment for this part of the product system is necessary.

DISCUSSION

Through environmentally extended input-output analysis an overview of the role of forest industry in the economy was presented. Based on the results, compared to the average sector, forest industries have slightly higher economic multipliers and slightly lower employment and climate change multipliers. Land use impacts were considerably higher than for other industries and the import dependency was lower than that of other production sectors (e.g. metals, chemicals). In spite of the relatively low greenhouse gas emission intensity, the large volume of production made the pulp and paper industry one of the largest sources of greenhouse gas emissions when emissions were allocated to final products.

Export production dominated the greenhouse gas emission inventory of Finnish forestry. In a consumption-based inventory [18,23], the domestic emissions of Finnish export products would not be included in the national inventory, but would be allocated to the importers of those products instead. As pulp and paper industry is one of the largest users of electricity, also a large share of emissions from electricity would be allocated to the export products. From a policy perspective, using consumption- instead of production-based inventories would shift the national mitigation focus from industrial electricity production to services. In addition, if importing nations would choose their suppliers based on carbon-intensity, Finnish bioenergy using pulp and paper industry would have a competitive benefit over natural gas using alternatives. Therefore the consumption based inventory would seem beneficial to the Finnish government. However including the emissions embodied in imports, might offset the benefits. Although the overall trade balance of emissions was not assessed in this study, it was quantified in the ENVIMAT -project. Based on the results of that project, the difference between production and consumption based inventories for Finland was minor, although emissions embodied in both exports and imports were considerable [15]. Due to the import dependency of the Finnish production and consumption patterns, the high share of emissions embodied in exports observed in this study has little effect on the difference between consumption and production based inventories of Finnish greenhouse gas emissions.

The usefulness of EE-IO in presenting multidimensional sustainability indicators was demonstrated. However in this study, no effort was made to connect the indicators into decision making and the indicators were presented separately. However, in real-life decision making conflicting indicators and tradeoffs between the dimensions of sustainability need to be resolved (e.g. is land use or employment more critical in a given supply chain). Therefore applying the tools of multi-criteria decision making (MCDA) [32,33] to environmentally extended input-output models would be an interesting topic of further research. In MCDA, the idea is to estimate the overall utility of different alternatives through weight coefficients for the sustainability indicators. Since several environmental, economic and social indicators have been connected to EE-IO, analyzing their relative importance would facilitate the interpretation of results for decision making.

Another interesting topic of future research related to decision making is the analysis of the uncertainties involved in EE-IO. Among others, uncertainties are related to selected system boundaries, estimated parameters and selected models [34]. In the case presented in this study, system boundary uncertainties are increased through aggregation and excluding the emissions of imports, parameter errors are introduced in the balancing of national accounts and choice uncertainty is introduced in the selection of indicators. Introducing weighting factors from decision analysis would add to the degree of uncertainty even more. More generally when making sustainability assessments based on national accounts, the level of uncertainty is expected to be high, given the complexity of the decision problem. If EE-IO would be used as a background for decision making in forest chain management, quantifying and reducing the uncertainties through models would be a priority. Monte Carlo simulation is commonly used in life cycle assessment [34] to carry out the uncertainty analysis and associated statistical analyses [35]. In the field of MCDA sophisticated techniques for measuring and analyzing uncertainties in terms of actual decision making have been developed [36,37]. The utilization of these techniques could benefit the use of EE-IO in multidimensional sustainability assessments.

CONCLUSION

Environmentally extended input-output analysis (EE-IO) was shown to be well suited to present the role of forest industries in producing both economic goods and environmental bads in connection with the rest of the economy. For most cases, the multiplier effects caused by the supply chain were considerably larger than those produced directly on the sector. This indicated that the forest industries are strongly connected with the rest of the economy. The analysis of emissions embodied in export revealed that for the forest industries, most of the environmental impacts are caused by production for export markets. Therefore local consumer choices have limited capability to control the overall emissions and land use impacts of Finnish forest industries through market mechanisms.

EE-IO was demonstrated to be applicable in simultaneously assessing the multiple dimensions of sustainability. This property could be further strengthened by using techniques from multi-criteria decision making.

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Land use impacts of industries and products in the Finnish economy: A comparison of three indicators

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ABSTRACT

The sustainability aspects of land use were assessed with an environmentally extended input–output model of Finland in 2002. The main economic industries and products causing land use were identified and the impacts were estimated with three indicators: biocapacity, human appropriation of net primary production (HANPP) and ecosystem degradation potential (EDP). The results correlated well with expert assessments on the threats to biodiversity, although the influence of animal farming was not clear in all indicators. Most of the domestic land use was caused by final demand outside Finland. Based on a simplified trade balance, Finland was a net exporter of land area, mainly through wood products. Two thirds of the domestic land use was driven by export production. Therefore a regional consumption based approach is not sufficient to mitigate and control the environmental impacts of land use even in a developed country like Finland.

1. Introduction

Land use associated impacts, such as habitat degradation, fragmentation and destruction, are identified as the main threats to biodiversity [1]. However, sustainability studies of bioenergy have to a large extent focused on energy and greenhouse gas balances [2] and in some cases on chemical pollution [3]. Indirect land use change, caused by increased demand on agricultural crop land, has been identified as a critical issue, but the focus has been on climate change [2] instead of biodiversity. Other impacts to ecosystems (e.g. soil compaction, loss of biodiversity, reduced productivity, salination) caused by land occupation have been ignored to a large extent. Partially this has been caused by the lack of a consensus on which indicators to use in life cycle impact assessment for land use [4]. Compared to other impacts, land use is a multifaceted environmental issue, as it can be seen as habitat degradation, resource competition or even as an alteration of biogeochemical cycles [5].

The purpose of this study was to identify the main drivers of land use in the Finnish economy and to assess environmental implications through a comparison of three indicators. The indicators were applied to an environmentally extended input–output model of the Finnish national economy [6]. The use of input–output analysis of a whole economy, instead of a product specific life cycle assessment, avoided favouring certain indicators due to system boundary selection [7]. Also the relevance of indicators could be compared to the national evaluation of the biodiversity action plan [8] and the IUCN red-list report [9]. While the focus on this study was on Finland, the methods may be applied to other countries in order to analyze the potential impacts of biomass utilization.

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The study extends previous environmentally extended input–output analyses by using land use impact indicators developed for life cycle assessment. In studies using input–output economics land use has been treated as an exogenous input to production, similar to employment or water [10]. Using the basic methods of input–output analysis, trade balances of land area [11] or ecological footprints [12] have been constructed for assessing, whether the consumption in a given country is within sustainable resource use limits and to what extent land use is caused by trade activities. In this study we extended the previous types of analyses by considering also the environmental impacts within a country and allocating the impacts to different industries and final demand categories. Also the land types were further disaggregated. While most environmentally extended input–output analyses treat land as a single class [10], and the ecological footprint divides it into six subcategories [13], we applied the CORINE classification [14] with thirty categories. This level of disaggregation allows the use of environmental impact assessment models for assessing the threats to biodiversity [4,15]. By combining impact assessment to the power of input–output analysis in studying trade flows, the main economic causes of land use impacts in Finland and embodied in imported products can be identified [12].

2. Materials & methods

2.1. Environmental input–output analysis

Land area, impacts, economic structure and final consumption were connected to each other with the basic equation of input–output based life cycle assessment [16]:

$$d = CF(I - A)^{-1}f \quad (1)$$

where,

d = impact indicators, three by one vector

C = impact characterization coefficients, 3 by 30 matrix, [impact $m^{-2}a^{-1}$]

F = direct land use intensity of industries, 30 by 151 matrix [m^2a M€$^{-1}$]

I = unity matrix, sector-by-sector, 151 by 151

A = intermediate use matrix, industry-by-industry, 151 by 151 matrix [M€ M€$^{-1}$]

f = final demand by industry, 151 by 1 [M€]

The monetary input–output table (MIOT) of Finland for the year 2002 [17] was used for intermediate interactions (A) and final demand (f). The resolution for calculations was 151 industries and products, arranged in an industry-by-industry format. Details for disaggregating certain industries are given in [6]. Details on deriving the land use matrix (F) and the impact indicators (C) are given in the following chapters. Following the general framework of land use in life cycle assessment [4], land use was reported as area multiplied by time. In the case of yearly national accounts this resulted in area × year.

The Equation (1) was used in analysis to allocate land use from production industries to consumption of products (i.e. the land use of forestry was allocated to the final products made from the harvested wood). The land use impact multipliers, which cover the whole land use needed in the supply chain to produce a certain (monetary) unit of production, were calculated by replacing f in Equation (1) with the unit matrix (I). For a rough trade balance, the extent of export was analyzed by substituting the total final demand in Eq. (1) with the final demand for export.

The basic input–output analysis was supplemented with two other components: the household's direct land use and the land use embodied in imports. This extended the model to the following form:

$$d_T = d + Cb + Em \quad (2)$$

where,

d_T = the total land use impact (rows by impact)

b = the direct land use of households [m^2a]

E = the impact intensity of imported products, impact by commodity [m^2 a t^{-1} or g m^2 a t^{-1}]

m = the mass of imported products by commodity [t]

Since the land use outside Finland could be assessed only in total land areas and ecological footprints, it could not be included in all indicators.

Only domestic land use was included in the input–output analysis. Land use embodied in imports was separately estimated from the import volumes [18] and life cycle inventories for wood, fuels, ores, metals, chemicals, plastics and minerals [19]. Due to the large number of items in the trade statistics, only the largest commodity flows were analyzed (88% of total imported mass). The land use of food items was calculated with the yield and equivalence factors of the ecological footprint land accounts, which report the demand of global average land area by commodity [20]. Since the imported land use was calculated separately, it was not included in the input–output analysis and the results were only used to construct a rough trade balance.

2.2. Allocation of land use to industries

The European land cover dataset CORINE 2000 was used as the basis of land use accounts. As the land cover data was not from same year as the economic tables, the slight changes in land cover between 2000 and 2002 were ignored in the analysis. Land use classes were allocated to industries based on a combination of national statistics, mass balances and monetary allocation. Specific statistics were available for the largest land users, forestry and agriculture, but estimations had to be used for allocation of built up land to industrial production. These approaches are explained in the following paragraphs.

Arable land and pastures covered about 8% of the Finnish inland surface area. They were allocated to crop, vegetable and animal production based on agricultural statistics on crop areas and the usage of crops. Crops which were used as feed directly on farms were allocated to animal farming while crops which were sold to food and feed industry were allocated to crop production. This resulted in 47.5% of agricultural land area being allocated to crop, 1% to vegetable and 51.5% to animal production.

The low productivity regions in Northern Finland were occupied by reindeer husbandry. It belongs to the industry of "other animal farming", with beekeeping and fur animal production. All the moors and heathland and the transitional woodland/shrub in the northern areas were allocated to other animal production (36000 km^2, 15% of occupied land area). While the intensity of land use is low, trampling and grazing has been identified to have a major influence on plant diversity and lichen biomass in the northern habitats [8].

Forests were allocated between conservation areas and forest cultivation. In 2002, 11% of forests were protected, and the remaining (89%) was allocated to forestry. Although annual felling and harvesting affects a small portion of the total occupied forest area, the whole area managed by rotational cutting is affected by alteration of species and age distribution of trees and the reduction of deadwood. Based on these impacts, the whole commercial forest area was allocated to the forestry sector. An alternative approach would have been to include only the portion of forest area, which would be needed to replace the amount of wood removed in felling. This approach was not used, since the felling varies from year to year, but the biodiversity of the whole forest area is influenced by the overall operation.

For built up industrial and commercial land (0.3%) CORINE was supplemented with a more detailed national SLICES-database [21]. Industrial and commercial units were subdivided further into commercial, office, governmental, industrial and storage areas. Of these, industrial and storage areas were allocated to industrial sectors based on their physical output from the PIOT-table [6]. Commercial and office areas were allocated to commercial sectors based on employment. Also governmental buildings were allocated to schools, health care and public services based on employment. This allocation procedure was likely to produce erroneous results, but the influence of built up land was minor compared to other land use classes.

The more disaggregated SLICES-database was also used for allocating land to mining and quarrying sectors. SLICES allowed the separation of sand quarrying and other mining. Mining regions were allocated to mining industries based on their material output. In addition, 3% of mires and peat bogs were allocated to the peat extraction sector based on the reported excavation area [22].

Residential areas (1.8% of land surface) were allocated between households and industries based on their density. Sparse areas were assumed to be owned by individual households, while buildings in densely built up areas were assumed to belong to housing companies and were allocated to the industry of letting and owning of dwellings. In Finland the CORINE class "147 Sports and leisure facilities" is 0.6% of land area and comprises mainly of summer cabins. It was allocated directly to households although a minor fraction is rented commercially, since statistics of renting versus owning were not readily available. Finally roads (0.6%) were allocated to road and railway maintenance, and harbours to other supporting services for transport.

2.3. *Impact assessment methods*

Several methods for transforming land occupation to environmental impacts have been proposed, ranging from use of natural bioproductivity [23] to exergy retention in ecosystems [24] and to landscape naturalness [25]. Due to the lack of consensus for a single indicator, three impact assessment methods were used: eco footprint biocapacity [26], human appropriation of net primary production (HANPP) [27] and habitat loss (EDP) [15]. These represent the influences of land use to human resource use, ecological life support functions and to biodiversity, respectively. While land use has an impact also on greenhouse gas, nutrient and water cycles [28], these were not included in the impact assessment methods, since they are covered in other impact indicator classes (i.e. climate change, eutrophication and resource depletion). These three impact assessment methods were implemented in the characterization matrix C of Equation (1) as distinct row vectors with elements for each of the land cover types. Details for obtaining the characterization elements are given in the following paragraphs.

The ecological footprint describes the amount of productive land needed to produce the renewable raw materials for society and to absorb the biological wastes produced by society. In order to compare different countries with different yields, the land area is expressed in global hectares (gha), which describe the average productivity of global ecosystems. The method does not include damage to local biodiversity [12,29], but instead describes the global pressure to the biosphere through the use of renewable resources. In this study, land use was weighted with the yield and equivalence factors [20] to estimate the amount of bioproductive land occupied by economic activities in Finland. The yield and equivalence factors were obtained by personal communication from the Global Footprint Network.

Human appropriation of net primary production (HANPP) [27] describes, to what extent the natural net primary production is reduced and used by humans in a given area. Therefore it can be used as an indicator of the disturbance of energy storage in ecosystems, which is thought to relate to ecosystem diversity and stability [30]. In this study HANPP was adjusted to the Finnish conditions by making the following modifications: the amount of forest NPP and the harvested fraction were based on [31] and grain yields were based on FAOSTAT-statistics [32]. Boreal mixed forest was chosen for the reference land use and it was assumed that the current NPP measured as carbon (375 g m^{-2} a^{-1}) [31] is equal to the natural state (this assumption was likely to underestimate the actual natural production, but it was used for practical reasons, as in [27], since the actual NPP in natural conditions has not been measured in the region).

For habitat loss, the method based on species-area relationships of vascular plants [9] was applied. Species densities in a given land use class (S_{class}) were compared to the average density ($S_{average}$) of the region ($EDP = 1 - S_{class}/S_{average}$). The method was originally parameterized for Switzerland and applied to Central Europe [15]. Therefore the method was re-parameterized to the Finnish case by using the data on numbers of endangered species and areas of macrohabitats [8]. This resulted in a considerable drop in resolution since only nine main macrohabitats were reported (however, forests and built up land were subdivided into their components). Therefore same values were used for all CORINE-classes belonging to a given macrohabitat. The greatest differences compared to the Central European values were

Table 1 – Selected industries for the comparison of direct and total land use and land use intensity in the Finnish economy in 2002.

	Direct km^2a	Final demand km^2a	Intensity (m^2 a M€$^{-1}$)
Forestry	152,024	18,620	75
Pulp and paper	212	50,170	5
Crop production	11,825	4131	18
Animal production	13,634	2457	8
Electricity production	10,516	1953	3
Other animal production	35,844	32,104	198

found in discontinuous urban fabric (which has higher than average diversity in Finland, but not in Central Europe), coniferous forests (which have lower relative diversity in Finland than in Central Europe) and semi-natural grassland (which has a considerable amount of endangered species in Finland). These adaptations represented better the boreal environment, where coniferous forests are common but human habitats are scarce.

The ecological footprint biocapacity and HANPP are largely theoretical constructs, which have no direct link to the observed biodiversity decline. In comparison EDP is based on observations on species densities without providing for the mechanisms behind the observed impacts. The correlation with these measures and the most important issues identified for biodiversity conservation is therefore an important validation step for the models. This correlation is further discussed in the discussion and conclusions section.

3. Results and discussion

3.1. *Land use of industries and products*

Most of the inland area (70% or 236,380 km^2) in Finland was allocated to various economic activities and households in 2002. Forestry (64%), reindeer husbandry (15%), animal farming (6%), crop production (5%) and electricity production (4%) were the main land using industries followed by households (3%) (Table 1). However an analysis, where land use is allocated to the final products instead of the primary producers showed a different view of the main causes of land use (Fig. 1). For example, the land area of forestry was for the most part allocated to the processing industries, leaving 8% of land use to forestry and 46% to forest industries.

Most of the domestic land occupation was driven by export commodity production (154,506 km^2, this is about 65% of the inland area occupied by economic activities). The main export products were pulp, paper and sawn wood (70% of the exported land area), as well as food products (24%), especially reindeer products.

In addition to exports, 102,747 km^2 of land area was estimated (using Equation (2)) to be embodied in imports, corresponding to 43% of the Finnish land area occupied by economic activities. The imported products with the largest land occupation were roundwood, fish products and oilseeds, which together contributed to more than 90% of the imported land use. The area embodied in imports was smaller than that embodied in exports, suggesting that Finland is a net exporter of land area. Considering the high land area and low population, this was not surprising. Although the import accounts were not reported on a sectoral basis in this study, it could be estimated that most of the imported land area was used for production of export products (i.e. roundwood for pulp and paper).

In addition to absolute land areas occupied by the supply chains of different products, also the land use intensities can be used to compare products. Land use intensities are presented for the six most land use intensive products in Table 1. The data for all final products and the intermediate results are presented in the Supplementary Data spreadsheet. Wood, animal and crop products had the largest land use multipliers, indicating that their production is highly dependent on using large land areas. The aggregated product group of "other animal products" includes fur animals (with the largest fraction of economic output), bees and reindeer (with large land use, but low economic output). For reindeer production alone, the land use multipliers would have been almost tenfold.

These land multipliers can be used in life cycle assessment (LCA) as a background data source for hybrid analysis, by estimating the impact from the product price for products

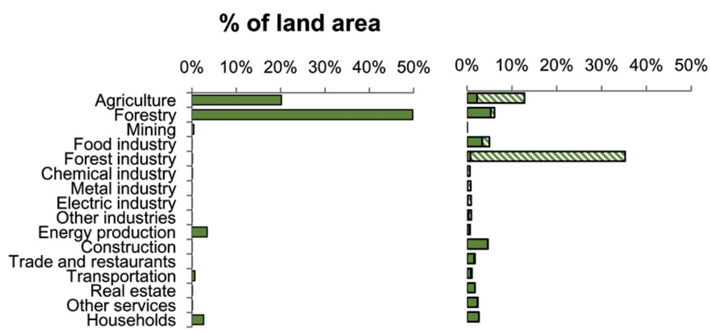

Fig. 1 – Land use allocated to industries (left) and products (right) in Finland 2002 as a fraction of the domestic inland surface area (305,000 km^2). The green region in the right figure is the extent of exported domestic land area. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

which do not have process-LCA inventories available [7]. The multipliers also give a crude estimate of the increases in land area requirements, if fossil fuels are replaced with bioenergy. For example, since the land use intensity of basic petroleum products is forty times smaller (0.1 m^2 a €$^{-1}$) than that of biological oils and fats (4 m^2 a €$^{-1}$), a shift to biological oils would increase the demand for domestic land area considerably. However, aggregated land use may not be a sufficient indicator for environmental sustainability impacts. Therefore more meaningful analyses of the multipliers require impact assessment of different land use types (e.g. cereal crop vs. managed woodland).

3.2. *Comparison of impact indicator results*

In spite of methodological differences, all indicators gave similar overall results. Based on all methods, a vast majority of the domestic land use impacts were associated with export commodity production. To a large extent this could be expected from the high level of land use in forest and animal production industries, which produce mainly for export markets.

Although the overall balance of land use impact was the same for all methods, notable differences could be observed in the impact intensities (which were calculated with Equation (1) by leaving out the multiplication with final demand) (Table 2). For example, other animal products were evaluated to be less damaging per monetary unit than roundwood in both biocapacity and HANPP, but more damaging in land use and Finnish ecosystem degradation potential. On the contrary, using Central European species densities for land use classes resulted in negative habitat loss (or habitat gain).

In the ecological footprint calculation, land occupation is weighted by the productivity of the land area [20]. Therefore, forest and arable land occupation has larger impacts than just occupied land area. Residential and commercial buildings are assumed to be constructed originally on arable land; therefore the impacts of buildings were amplified with the ecological footprint compared to aggregated land area. Because economic activities are concentrated on productive land, the ecological footprint gave a higher estimate of the available biocapacity in use (86%) than aggregated land area based metrics. However, the fraction of exported global footprint (64% of biocapacity in economic use) was similar to exported land area (66% of land area in economic use). The main products responsible for biocapacity occupation were wood products and construction.

The human appropriation of net primary production (HANPP) was relatively low compared to the figures of ecological footprint and aggregated land use, 'only' 36% of potential natural net primary production was estimated to be in use or destroyed. This is result is similar to earlier estimates (20–30%) for Finland [27]. (For comparison, most of the Central European countries have HANPP ranging from 40% to 70% [27].) Compared to the ecological footprint biocapacity, the HANPP indicated less severe impacts from forest products, since a majority of the net primary production (NPP) is left in the forest and only logs are harvested. While only one third of the potential NPP was used by society, the remaining NPP was in forest residues, semi-natural peatlands, sparsely vegetated regions, and in straw. Therefore the primary production remaining in forest and agricultural residues was not available for natural ecosystems, which were deficient in large-diameter deadwood, which is an important habitat for many endangered species [8,9]. Therefore the human impact on ecosystem function and biodiversity was somewhat underestimated. In spite of the low fraction of net primary production in use, the potential for expanding the human use of primary production into natural peatlands, sparsely vegetated regions and collection of remaining deadwood is limited by biodiversity concerns. For example, the use of deadwood competes with the goal of maintaining habitats for endangered forest species [8,9].

The estimation of habitat loss with the ecosystem degradation potential (EDP) indicator [15] was highly influenced by the use of Finnish species density data instead of Central European (Table 2; Supporting Data). In Central Europe, coniferous forests, lakes, alpine moors and heathlands have a high species diversity compared to the surrounding environment. In contrast, in Finland species diversity is the highest in cultural habitats, natural meadows and on beaches. Use of Finnish data resulted in negative habitat loss (or positive habitat gain) for dairy and meat production, since highly species diverse meadows were allocated to animal farming. Based on the Finnish specific EDP, most habitat loss was caused by other animal production (reindeer), which occupied large areas of low diversity land. On the contrary, using Central European data, this sector resulted in habitat gain, since the environment is rich in endangered species in Switzerland. 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 [4,28]).

Table 2 – A comparison of impact intensities of selected products assessed with different indicators using the environmental input–output framework for Finland.

Product	Land use km^2 M€$^{-1}$	Biocapacity km^2 M€$^{-1}$	HANPP kt M€$^{-1}$	EDP Finland	EDP CE
Other animal products	198.62	128.57	0.56	15,310	−5211
Roundwood	75.06	199.63	9.28	847	524
Sawn wood	27.66	73.39	3.42	310	192
Crops	18.48	52.26	5.04	714	762
Wood packaging	9.08	24.01	1.12	98	63
Plywood	8.6	22.56	1.05	97	56
Dairy products	5.28	1.22	1.09	−1616	130
Animal and vegetable oils and fats	3.78	10.47	1.00	120	147
Refined petroleum	0.12	0.22	0.016	−1	0

Overall, human appropriation of net primary production (HANPP) and ecological footprint gave similar results. However, neither indicator correlated very well with ecosystem degradation potential (EDP). The lack of correlation between ecological footprint and biodiversity has been noticed earlier [12]. In our case, this was mainly caused by outliers in the species density of certain land use classes with low productivity. Specifically, the species density of moors and heathland was very low compared to the productivity and the species density of meadows was extremely high compared to productivity. Therefore the occupation of these low productivity regions resulted in little impact in HANPP and biocapacity, but very high in biodiversity.

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*" [33], 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 [9], 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 in [33] and [8]. However the damaging impact of tourism and off-road driving [33] was not identified, 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 allocated only to the mining of energy minerals (peat) according to the area utilized. Effects of historical drainage were not included for peatlands, which were not in forestry or other economic use. Also shore habitats were not threatened by their use or occupation, but by transformation into residential areas [8,33]. Although methods for land use transformation impact assessment have been proposed [4], transformation impacts were not assessed, due to data limitations (Finland did not participate in CORINE-mapping prior to 2000).

Only the regionalized EDP-indicator identified the role 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.

4. Conclusions

The economic input–output table of Finland was extended to account for land use. CORINE land use statistics were allocated to economic sectors based on national statistics and material production. The sustainability impacts of land use were assessed with three indicators. All indicators were found to give similar overall results than those obtained by expert assessments on biodiversity management. Animal and wood products were identified as the products with the highest land use impacts. However no individual indicator provided all aspects of natural resource management. While HANPP could be used to identify the competition between forest biomass and deadwood, production based indicators were less applicable to low bioproductive habitats (alpine habitats and meadows).

The combination of land use impact indicators and national accounts made it possible to assess the "trade balance" of land use impacts. More than half of the domestic land use was found to be associated with export industries, which also consume imported land area. Due to the considerable impacts of imports and exports, a consumption based approach (similar to that proposed for climate change) might be more appropriate than the current national approach. Overall, existing land use statistics could be easily coupled to the national accounts and the use of impact indicators revealed relevant issues for sustainable land use on an international level.

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Appendix. Supplementary data

Supplementary data associated with the article can be found in online version, at doi:10.1016/j.biombioe.2011.02.052.

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Publication III

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Any sustainable decoupling in the Finnish economy? A comparison of the pathways and sensitivities of GDP and ecological footprint 2002–2005

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ABSTRACT

Ecological footprint was combined with economic input–output analysis in order to identify the economic structures causing the overuse of biological resources. These structures were analyzed with the use of structural decomposition, path analysis and sensitivity analysis and compared to the structures which drive economic growth. The scope of the analysis was the Finnish national economy during 2002–2005. Based on the results increases in gross domestic product (GDP) and ecological footprint were found to be different subsystems of the economy. This aspect was previously hidden by country level aggregate indicators. Ecological footprint was increased by the production and consumption of primary commodities, such as wood, paper, fish, crops, animal products and energy as well as construction. In contrast, GDP growth was caused mainly by increased demand in service sectors such as renting and owning apartments, trade and business services as well as governmental services, health, education and social work. The two systems overlapped only in dairy products and forest products, which had major influences to both indicators. Ecoefficiency improved overall in the economy between 2002 and 2005 especially in some industries, such as sawmilling and electricity production. However growth in consumption resulted in increased environmental impacts nevertheless.

1. Introduction

The ecological footprint measures the consumption of biological natural resources. It is expressed in productive land area needed to supply the goods and process the wastes of a given entity. Ecological footprint can be calculated for products, organizations and regions, but is most commonly used to estimate the ecological footprint of nations in national footprint accounts (NFAs). The national footprint accounts can be compared with the land area available for a given country (biocapacity) to determine, whether the country is exceeding its ecological limits (consuming more renewable goods than could be sustainably produced). Globally the ecological footprint exceeded the available biocapacity in the beginning of 1980s, resulting in an ecological overshoot which has continued since (Ewing et al., 2008). Population growth and use of fossil fuels have been identified as the main drivers of the overshoot. The ecological footprint has been found to grow continuously with increasing income, therefore negating any hypothesis of decoupling at the global scale (Caviglia-Harris et al., 2009; Bagliani et al., 2008). In spite of the global overshoot, some sparsely populated and bioproductive countries are still below their biocapacity (Ewing et al., 2008).

Finland is one of the few countries, which is not in a state of ecological overshoot. Finland has a low population density with 5.3 million inhabitants and an inland surface area of 34 million hectares. The low population density is combined with production of resource intensive commodities such as pulp and paper, mining, metals and machinery. These commodities are mainly exported, contributing to economic growth but not directly to consumption based ecological footprint. Finland is also an interesting case study from the viewpoint of decoupling. The ecological footprint of Finland decreased by 6.5% from 2002 to 2005 (Global Footprint Network, 2010) while the gross domestic product increased by 9.5% (Statistics Finland, 2007). The ecological footprints of Germany and Netherlands also decreased, but their level of consumption was above their biocapacity (Ewing et al., 2008). Therefore Finland would seem to be a rare example of absolute decoupling at an already sustainable level of consumption.

In recent times there has been a synthesis of ecological indicators used in environmental systems analysis. Ecological footprinting is increasingly being used together with input–output economics to study the production–consumption patterns and subsequent biological resource use (Turner et al., 2007). In the same time, life cycle assessment has merged with environmental input–output analysis (Suh, 2009), enriching the methods in both fields. In this study we apply the rich methodological toolbox of life cycle assessment (LCA) (Guinee et al., 2002) and environmentally extended input–output analysis (EEIO) (Leontief, 1970)

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to the National Footprint Accounts of Finland 2002–2005 (Global Footprint Network, 2010). The aim is to demonstrate the benefits of analyzing the accounts at a detailed subnational process level and to identify the main pathways of change for ecological footprint and economic growth.

2. Materials and methods

2.1. The hybrid input–output model

Input–output analysis was developed to analyze the rich interactions between economic sectors. In order to produce a commodity (output), an industry will need raw materials (input) from other industries. These industries in turn need raw materials from other industries, sometimes including the original industry, resulting in complex cyclical flows. As a result, producing commodities for consumption requires a considerable amount of intermediate production. A key question in both economic and environmental input–output analysis is how much economic activity and environmental impact is caused by different parts of consumption. This information can then be used to identify the impacts of companies, consumers and countries. Of particular importance to ecological footprinting is the use of EEIO to construct consumption based national inventories, where the imported resources are added to the national inventory and resources used for export production are removed (Turner et al., 2007). The use of input–output based inventories is recommended, since the extent of domestic consumption and exports are more accurately followed (Wiedmann and Barrett, 2010).

The analysis was based on a tiered hybrid version of life cycle assessment and input–output analysis (Suh and Huppes, 2005). In this study the model was applied for ecological footprinting:

$$EF = [\, B_d \quad B_i \,] \begin{bmatrix} I - A_d & 0 \\ -A_i & I \end{bmatrix}^{-1} \begin{bmatrix} f_d \\ f_i \end{bmatrix} = B(I - A)^{-1} f \tag{1}$$

where EF = the ecological footprint (gha); B_d = domestic footprint intensity (gha/M€); B_i = imported footprint intensity (gha/M€); A_d = domestic input coefficient matrix (M€/M€); A_i = imported input coefficient matrix (M€/M€); I = identity matrix; f_d = final demand of domestic products (M€); f_i = final demand of imported products (M€).

The final demand (f) and input coefficient matrices (A) were based on the official national accounts of Finland (Statistics Finland, 2007). The term $(I - A)^{-1}$ is the Leontief inverse, which describes all the intermediate products needed to produce output from an industry when the whole supply chain is taken into account. B is the overall footprint intensity of domestic and imported commodities. The domestic input coefficient matrix was assembled as an industry-by-industry table according to the recommendations of Eurostat (2008) with a resolution of 151 economic sectors. The imported input coefficient matrix was reported as 733 commodity groups. The domestic footprint intensity (B_d) was based on the Finnish National Footprint Accounts (NFA) 2002 and 2005, calculated with the most recent footprint methodology (Global Footprint Network, 2010). The NFA reported the ecological footprint for six subclasses: carbon uptake, cropland, grazing, fishing, built and forest land. For the carbon uptake land, national emission inventories were used instead of NFA results. Therefore the calculation included also methane (CH_4) and dinitrogen monoxide (N_2O) emissions, which were converted to CO_2 equivalents using the most recent global warming potentials (IPCC, 2007).

The current study differs most from previous studies in the analysis of imports (B_i). Most previous studies combining IO and EF have either (a) assumed that imported commodities would be produced with similar emissions than domestic commodities, (b) used multiple region input–output (MRIO) models to estimate imports or (c) used the footprint coefficients from NFAs (Turner et al., 2007). In this study the imported commodities were estimated by combining NFA footprint coefficients with LCA databases on greenhouse gas emissions. NFA data was used for the crop, pasture, forest and fishing grounds embodied in imports, but the imported carbon intensities were based on a combination of life cycle emission inventories for greenhouse gases (Ecoinvent, 2008) and the domestic technology assumption (Seppälä et al., 2009). The use of more detailed life cycle data and the inclusion of other greenhouse gases than carbon dioxide were nonconventional, but acceptable improvements to the current methodology according to the Ecological Footprint Standards (Global Footprint Network, 2009). The NFA import footprint coefficients have been criticized for not including the carbon emissions of imported transport services and being based on inappropriate embodied energy figures (Wiedmann, 2009). The use of hybrid IO-LCA inventories instead of the NFA coefficients resolved these issues, since the LCA data both included transport emissions and was based on actual greenhouse gas emissions instead of embodied energy. Therefore it had the benefits of the MRIO method (Wiedmann, 2009) without adding too much computational complexity.

The NFA results were reported as aggregated totals for six land use classes, but the input–output tables included 151 industries. Therefore the aggregates had to be allocated to industries using national statistics. The carbon footprint was already reported by industry in the national emission inventory, therefore no adjustment was necessary. The aggregated data for croplands were allocated to crop production by using national statistics on the use of agricultural commodities. The crops which were reported as used directly as feed were allocated to integrated animal production, while the crops which were sold to other farmers or to the food and feed industries were classified as crop production. This resulted in 26% of cropland being allocated to animal farming. All grazing land was allocated to animal farming, all fishing land was allocated to fishing and all forest land was allocated to forestry. For built up land the more accurate CORINE 2000 database was used instead of the GAEZ database used in the ecological footprint. Residential areas were allocated to the industry of renting and owning apartments and to households. Industrial areas were allocated to industries based on the amount of material output (Seppälä et al., 2009) of the process industries and the economic output of the service industries. Finally roads, airports and harbors were allocated to the sectors responsible for maintaining roads and other transport areas.

2.2. Analytical methods: sensitivity and structural analysis

Both the GDP and ecological footprint are usually reported in an aggregated form, making it impossible to determine, why the results have increased or decreased. In this study we applied methods from input–output analysis and life cycle assessment to identify relevant model components and subsystems from the network of economic and environmental interactions included in the environmentally extended input–output model (Eq. (1)). To our knowledge, this is the first published application of these tools to ecological footprinting. The tools have been applied to greenhouse gas emissions and energy consumption in other studies.

Sensitivity analysis, or perturbation analysis, is used in life cycle assessment to determine the interactions between processes, which have the most influence on the environmental impacts studied (Heijungs and Suh, 2002). In the input–output model applied in this study, there are approximately 21 000 domestic inter-industry relationships and 20 000 relationships with import products. In the sensitivity analysis, each relationship (defined as an input-coefficient) was changed by 1% and the relative change in the

Table 1
Environmentally important linkages in the Finnish economy in 2005 and their influence on GDP. Linkages in bold are important for both ecological footprint and GDP. Sensitivity is reported as the impact which would be caused by 1% change in the linkage.

Use of product	Using sector	Footprint sensitivity (gm^2 a/capita)	GDP sensitivity (€/capita)
Roundwood	Sawmilling	7.42	0.126
Carpentry	**Residential construction**	4.12	0.161
Animal products	Meat processing	3.30	0.107
Animal products	**Dairy production**	3.30	0.134
Roundwood	**Pulp and paper manufacture**	3.30	0.142
Sawn wood	Residential construction	3.30	0.054
Animal products	Animal farming	2.47	0.107
Sawn wood	Carpentry production	2.47	0.054
Animal feed	Animal farming	1.65	0.040
Pulp and paper	**Pulp and paper manufacture**	1.65	0.268
Pulp and paper	Printing	1.65	0.054
Electricity	Renting and owning apartments	1.65	0.080
Construction work	**Renting and owning apartments**	1.65	0.295
Fish	Fishing and fish farming	0.82	0.008
Meat	Meat processing	0.82	0.067
Crops	Grain milling	0.82	0.016
Crops	Animal feed production	0.82	0.016
Crops	Beverage production	0.82	0.013
Roundwood	Carpentry production	0.82	0.019
Sawn wood	Electricity production	0.82	0.021
Trade services	**Residential construction**	0.82	0.348
Civil engineering	**Civil engineering**	0.82	0.187
Electricity	Trade	0.82	0.013
Natural gas (imported)	Electricity production	0.82	–
Deciduous pulpwood (imported)	Pulp and paper manufacture	0.82	–

ecological footprint or gross domestic product was recorded. The relationships with high sensitivity were labeled as influential to the indicator results. This allowed the identification of a smaller set of important factors.

The next stage in the analysis was to perform a structural decomposition analysis, in order to identify the macroeconomic causes of change in the ecological footprint and gross domestic product between 2002 and 2005. This allowed the identification of conflicting development processes between changes in production structure ($\Delta(I-A)$), impact intensity (ΔB) and the size and composition of final demand (Δf). The overall change in the ecological footprint could then be expressed as the sum of the individual difference terms:

$$\Delta EF = B(I-A)^{-1}\,\Delta f + B\,\Delta(I-A)^{-1} f + \Delta B(I-A)^{-1} f \qquad (2)$$

Several methods have been developed in input–output analysis to calculate the decomposition in a robust manner. In this study, the average of all possible first order decompositions was used (Dietzenbacher and Los, 1998). Since the input–output tables were reported in current prices, a price adjustment had to be made prior to comparing changes between years. The tables were adjusted using double deflation and the producer's price indexes (Statistics Finland, 2009). For a discussion on the methodological issues of double deflation, cf. Peters et al. (2007).

The decomposition analysis provided an overview of the causes of change, but could not be used to identify the specific processes, which had changed. For this purpose, a recent addition to the environmental input–output methodology, structural path decomposition (Wood and Lenzen, 2009) was applied. In structural path decomposition, the production structure of the economy is studied through Taylor 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 allowed the study of change in a process level instead of country level aggregates.

3. Results and discussion

3.1. Sensitivity analysis identified subsystems of growth and resource use

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 (Table 1). For the gross domestic product, 12 items were identified as important (Table 2). 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. These as well as the construction sectors were identified as important for both GDP and ecological footprint. Overall seven linkages 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.

Table 2
Economically important linkages in the Finnish economy and their influence to ecological footprint. Linkages in bold are important for both ecological footprint and GDP. Sensitivity is reported as the impact which would be caused by 1% change in the linkage.

Use of product	Using sector	Footprint sensitivity (gm^2 a/capita)	GDP sensitivity (€/capita)
Trade services	**Residential construction**	0.82	0.348
Business services	Electronics industry	0.04	0.321
Construction work	**Renting and owning apartments**	1.65	0.295
Pulp and paper	**Pulp and paper manufacture**	1.65	0.268
Trade services	Trade	0.58	0.268
Business services	Business services	0.25	0.214
Civil engineering	**Civil engineering**	0.82	0.187
Carpentry	**Residential construction**	4.12	0.161
Animal products	**Dairy production**	3.30	0.134
Roundwood	**Pulp and paper manufacture**	3.30	0.134
Post and communication services	Trade	0.16	0.134
Post and communication services	Business services	0.08	0.134

Some items were identified as significant in the analysis due to the inclusion on methane to the ecological footprint analysis. Imported natural gas produces methane emissions in its production and animal production emits methane due to rumination. If a conventional focus on only carbon dioxide would have been used, these interactions would have been ignored. This supports the inclusion of other greenhouse gases than carbon dioxide to the ecological footprint.

Sensitivity analysis of input–output models is not common, and only a few studies have been published. Therefore it is difficult to state whether the results of sensitivity analysis are generalizable outside Finland. Some conclusions can be made however. Firstly, in countries where forest industries are not dominant, the GDP influence of using pulp, paper or wood products is insignificant, since the commodities are imported. The sensitivity of EF to the use of wood products is still high, since the impacts outside country borders are included. This further strengthens the conclusion that a large portion of the EF is caused by economic activities which have very little influence on GDP. By contrast in most countries, as in Finland, flows associated with construction and animal production are likely to be significant for both GDP and EF, making their control politically controversial.

3.2. Structural decomposition analysis: improved efficiency versus increased growth

In contrast to the official national footprint accounts, the input–output based consumption footprint for Finland increased between 2002 and 2005 by 0.79 gha/capita. (In the footprint accounts it decreased by 0.44 gha/capita, from 6.73 to 6.29 gha/capita.) The difference was mainly caused by the different calculation of exported carbon footprint. Since the consumption based footprint is calculated by adding imported footprint to domestic production and subtracting the exported footprint, correct estimation of exported footprint is highly important. The current methodology of the footprint accounts estimates the exported carbon footprint by using embodied energy estimates and world emission intensity for energy production (Ewing et al., 2008). For Finland the main components of the exported carbon footprint were wood products and paper, which in reality are produced largely using biomass in the sawmills and pulp and paper plants. These embodied emissions were correctly tracked with the environmentally extended input–output model, but overestimated with the conventional footprinting method. This phenomenon has been observed in other studies comparing input–output based and conventional ecological footprint accounting (Wiedmann and Barrett, 2010). Since the input–output based results seemed reliable, the consumption based inventory actually increased between 2002 and 2005, thus negating the claims about absolute decoupling in the Finnish economy.

The main components which caused the changes in GDP and ecological footprint are presented in Fig. 1. The components were divided to production structure, impact intensity, demand structure and demand size. For example the production structure describes, how much grain is needed to produce a ton of animal feed, impact intensity describes how much land was needed to produce a ton of grain, demand structure describes what share of consumption goes to animal products, and demand size describes the overall amount of consumption. For the ecological footprint the impact intensity of production improved considerably, resulting in more commodities being produced with less resource use (Fig. 1a). However the production structure was more inefficient, meaning that more processed goods were used as raw materials. In addition both the structure and absolute size of final demand increased, therefore increasing the total ecological footprint. The competing effects of increasing efficiency and total demand have been observed also for other countries, such as China (Peters et al., 2007). Similarly, the GDP increased between 2002 and 2005 solely because of increased final demand volume (Fig. 1b). All other components influenced to reduce the GDP. This could indi-

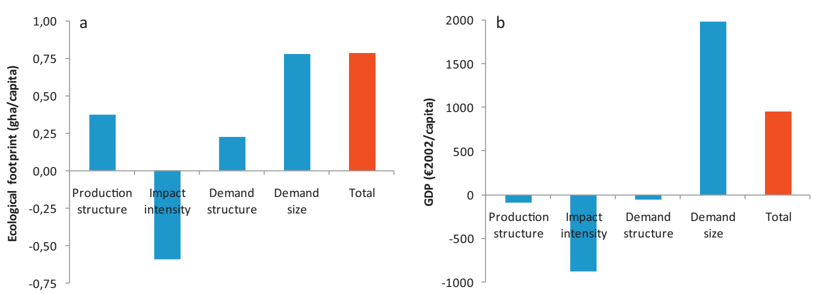

Fig. 1. Decomposition of the change in (a) ecological footprint and in (b) GDP in the Finnish economy between 2002 and 2005.

Table 3
Most influential economic pathways for ecological footprint in Finland 2005. Flow of goods is from the third to the second and first supplying sector and finally to the final product.

Ecological footprint					
3. Supplying sector	2. Supplying sector	1. Supplying sector	Final product	Ecological footprint (gha/capita)	Land type
–	–	–	Roundwood	0.72	Forest
–	–	–	Crops	0.20	Crop
–	Roundwood	Sawn wood	Residential construction	0.18	Forest
–	–	–	Electricity	0.17	Carbon
Roundwood	Sawn wood	Carpentry	Residential construction	0.14	Forest
–	–	Electricity	Renting and owning apartments	0.11	Carbon
–	–	–	Road maintenance	0.08	Built
–	–	–	Fish	0.08	Fishing
–	–	–	Imported fish products	0.08	Fishing
–	Roundwood	Carpentry	Residential construction	0.07	Forest
–	–	Roundwood	Pulp and paper	0.06	Forest
–	–	–	Trade services	0.05	Carbon
–	–	Animal products	Dairy products	0.05	Crop
–	–	Crops	Drinks	0.05	Crop
–	–	–	Imported fresh fish	0.05	Fishing

Table 4
Most influential economic pathways for GDP in Finland 2005.

1. Supplying sector	Final product	GDP (€/capita)
–	Renting and owning apartments	2177
–	Trade services	1431
–	Health services	1242
–	Education	1219
–	Public administration	1159
–	Social work	897
–	Residential construction	852
–	Road transport	740
–	Other services	699
–	Business services	562
–	Pulp and paper manufacture	538
–	Insurance	241
–	Road maintenance	235
–	Restaurants	214
–	Post and telecommunication	205
–	Civil engineering	191
Construction work	Trade	179
–	Other special machinery	164
Road transport	Business services	155
–	Real estate agencies	136

cate that the economy was using more and more of goods, which were produced from imported raw materials. In the same time, the impact intensity decreased sharply, indicating less value added per unit of production.

Overall the decomposition of the indicators revealed that both GDP and ecological footprint were driven by growth in final demand. Without growth in final demand both indicators would have decreased due to structural changes. For ecological footprint, these changes represented improved ecoefficiency, but for the GDP they represented a more inefficient economy.

3.3. *The most important pathways of economic production and environmental impact*

Structural path analysis confirmed that the structurally most important economic linkages were different for ecological footprint and gross domestic product. The largest single contributors to the ecological footprint were the consumption of wood, food and electricity (Table 3). Another important factor was the consumption of wood embodied in construction work through several intermediate products, such as builder's carpentry and sawn wood. In comparison, the main pathways of GDP formation were very short (Table 4). Most value was added just before the final product was consumed. In addition, most of the products were 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 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 caused mostly by the last stages of construction work, while the EF is caused by roundwood extraction two ladders further in the supply chain. Similarly, the growth in service industries increases GDP but the resource extraction is visible only through long supply chain interactions.

The use of structural path analysis allowed also a more detailed analysis of the decomposition results. The most important pathways were extracted for the years 2002 and 2005 and the change in those paths was analyzed with decomposition techniques (Wood and Lenzen, 2009). The results allowed the identification of major changes in the most important production and consumption pathways. 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.

Table 5
Twenty pathways which caused the largest change in the ecological footprint. (The sources of change are coded as following: f = final demand, $A1\ldots3$ = 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/capita)	Land type	Change	Final product	1. Supplying sector	2. Supplying sector	3. Supplying sector
1	0.192	Crop	Δf	Crop production	–	–	–
2	−0.067	Forest	$\Delta A2$	Residential construction	**Sawmilling**	Forestry	–
3	0.063	Carbon	Δf	Electricity	–	–	–
4	−0.058	Carbon	ΔB	**Electricity**	–	–	–
5	−0.049	Carbon	ΔB	Renting and owning apartments	**Electricity production**	–	–
6	−0.043	Forest	$\Delta A3$	Residential construction	Carpentry	**Sawmilling**	Forestry
7	−0.038	Carbon	ΔB	**Renting and owning apartments**	–	–	–
8	0.038	Forest	$\Delta A2$	Residential construction	**Carpentry**	Sawmilling	Forestry
9	−0.031	Forest	$\Delta A2$	Residential construction	**Carpentry**	Forestry	–
10	0.029	Forest	$\Delta A1$	**Residential construction**	Carpentry	Sawmilling	Forestry
11	0.025	Forest	Δf	Forestry	–	–	–
12	−0.024	Fishing	Δf	Fishing	–	–	–
13	0.023	Forest	$\Delta A1$	**Residential construction**	Sawmilling	Forestry	–
14	0.022	Forest	Δf	Residential construction	Sawmilling	Forestry	–
15	0.022	Forest	Δf	Paper (imported)	–	–	–
16	0.021	Fishing	$\Delta A1$	**Restaurant services**	Fish (imported)	–	–
17	0.021	Forest	$\Delta A1$	**Pulp and paper manufacture**	Forestry	–	–
18	0.021	Carbon	Δf	Crop production	–	–	–
19	−0.019	Fishing	$\Delta A1$	**Fish products**	Fish (imported)	–	–
20	−0.019	Forest	ΔB	**Forestry**	–	–	–
Total	0.79						

However the final demand of consumption (Δf) increased overall, and this resulted in a net increase of the ecological footprint by 0.79 gha/capita between 2002 and 2005.

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 $€_{2002}$/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 (Caviglia-Harris et al., 2009; Bagliani et al., 2008). 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 (Fig. 1 and Table 5). 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). Overall, more work is necessary to study the subnational level of ecological footprint in order to identify the processes which have succeeded in reducing the EF in spite of increasing overall production.

4. Conclusions

Compared to aggregated national inventory results, combination of National Footprint Accounts and input–output analysis provided a deeper understanding of the drivers and pathways of ecological footprint and gross domestic product increase. With a more accurate estimation of the exported ecological footprint with input–output analysis, the perception of Finland as an example of absolute decoupling of ecological footprint from economic growth was canceled. Instead both the ecological footprint and gross domestic product increased between 2002 and 2005.

Based on all the analyses made, ecological footprint and gross domestic product formation would seem to be mainly separate economic subsystems. The sensitivities overlapped in residential construction, pulp and paper manufacture and dairy production. Based on the structural decomposition analysis both indicators were increased during the analyzed time period mainly due to growing consumption volume. For the gross domestic product, consumption volume was the only major driver for growth.

The structural path and decomposition analyses allowed for a detailed subnational analysis on the most important pathways. Improved ecoefficiency was observed during the time period both at a macro-level and at individual process chains. Most notable improvements were observed in the reduced carbon emissions of electricity production as well as the more efficient use of wood in sawmilling. However on a national scale, the increased final demand resulted in an increased ecological footprint in spite of efficiency gains. On a process level, the picture was more complex, with tradeoffs in resource use in different product chains and process parts. Overall, a detailed process level analysis is recommended for policy making in order to track progress in individual sectors. Otherwise the aggregated results might hide many tracks of positive development.

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Comparing priority setting in integrated hazardous substance assessment and in life cycle impact assessment

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Abstract
Purpose The purpose of the study was to compare three recent Life Cycle Impact Assessment (LCIA) models in prioritizing substances and products from national emission inventories. The focus was on ecotoxic and human toxic impacts. The aim was to test model output against expert judgment on chemical risk assessment.
Materials and methods An emission inventory was collected for Finland describing the year 2005. The inventory included publicly reported emissions to air and water and it was complemented by the emissions of tributyltin, benzene, and pesticides from research papers and statistics. The emissions were characterized with three LCIA models: IMPACT 2002+, ReCiPe, and USEtox and priority substances were identified. The results were connected to an environmentally extended input–output model to study priority products and supply chains. A comparison was made with two integrated assessments of the chemical status and human exposure in the Baltic region.
Results and discussion The three assessed models presented very different priorities. In ecotoxicity, IMPACT2002+ and USEtox highlighted heavy metals while ReCiPe focused on tributyltin. The integrated assessment identified both groups. In human toxicity, IMPACT2002+ and the integrated assessment focused on organic air pollutants while USEtox and ReCiPe identified mainly metals.
Conclusions LCIA models can be used for priority setting in chemical emission control and consumption based analyses. However the models give differing prioritizations so care must be taken in model selection. The studied models differed from expert assessment mostly in substances which are bioaccumulative. Further studies in including bioaccumulation to LCIA models of toxic impact are recommended.

Keywords Ecotoxic impact assessment · Emission inventory · Environmentally extended input–output analysis · Human toxic potential · Priority substance · Structural path analysis

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1 Introduction

Life Cycle Impact Assessment (LCIA) of hazardous substances has progressed to the level, where the models used are fully comparable with those used in chemicals risk assessment (MacLeod et al. 2010; Rosenbaum et al. 2008).

The models are based on fugacity modeling, which has a long history in environmental chemistry and generally good correspondence with environmental monitoring (MacLeod et al. 2010). In the most recent models, the toxicological effect factors are based on statistical analysis of empirical measurements and population disease occurrences (Rosenbaum et al. 2008). In comparison, the integrated assessment of chemical risks in the environment and chemical management in environmental policy rely strongly on measured concentrations and agreed regulatory limits (e.g., EC 2004; EVIRA 2010; HELCOM 2010). The borderline between risk assessment and risk management is often blurred (Assmuth and Jalonen 2005).

LCA has been used more and more together with environmentally extended input–output analysis (EEIO; Suh 2009). In that context, LCIA models are used to analyze entire national economies with the aim of identifying the main sources of environmental impact in the production–consumption structures (UNEP 2010; EPA 2009). When entire national emission inventories are analyzed with LCIA, it offers an opportunity to test LCIA model output against other methods, especially national chemical integrated assessments and priority settings. This serves as a kind of a model validation, since ideally the prioritization from LCIA models should highlight the same issues as expert assessment (EVIRA 2010; HELCOM 2010) and perhaps point out targets for further enquiries.

In this study, we applied three recent LCIA models to an environmentally extended input–output analysis of the toxic emissions of Finland. The aim of this study was twofold: (1) to identify the main pathways causing ecotoxic and human toxic impacts in Finland and (2) test the applicability of impact assessment models against integrated assessments that were based on period from 1999 to 2007 (HELCOM 2010) and from 2002 to 2009 (EVIRA 2010).

2 Material and methods

2.1 Emission inventory and life cycle impact assessment

The emission inventory was based on public data sources and the emission registry of the Finnish Environmental Administration (VAHTI -database). The database includes reported toxic air and water emissions for industrial sites and estimated emissions for households, service production, and agriculture. In the Environmental Impacts of Material Flows Caused by the Finnish Economy (ENVIMAT) project, each site was connected to an industrial classification code to get the industry total emissions. The inventory included air emissions of arsenic, cadmium, cobalt, chromium, copper, mercury, nickel, lead, zinc, vanadium, PAH compounds, dioxins, and furans and hydrogen fluoride. Benzene emissions were estimated separately by combining fuel use (Statistics Finland 2006) and benzene emission factors (Pietarila et al. 2002). Since the exact distribution of PAH compounds was unknown and only the total was reported, it was assumed that the distribution would be similar to that estimated for EU (Sleeswijk et al. 2008).Water emissions of arsenic, cadmium, cobalt, chromium (III and VI), copper, mercury, nickel, lead, antimony, tin, vanadium, zinc, phenols, toluene, and vinyl chloride were included from the VAHTI database. Tributyltin emissions from ships were included based on expert assessment (Ministry of the Environment 2006).

Pesticide emissions to agricultural soil were included by using the sales statistics of 34 herbicides, insecticides, and growth regulators. Although detailed models for estimating air and water emissions from application rates were available (Birkved and Hauschild 2006), they were not applied. Instead the fate factors of the characterization models were used to transform the application rates (emission to soil) into water concentrations and human exposure. This was deliberately done to increase variability between characterization models and is an approach similar to that used in calculating recent LCIA normalization factors (Sleeswijk et al. 2008).

Three life cycle impact assessment (LCIA) methods were applied to characterize the emissions: IMPACT 2002+, ReCiPe, and USEtox. Only the freshwater ecotoxicological and human toxic midpoint impact categories were included in the comparison. All three models use fugacity-based multimedia environmental fate models to estimate environmental concentrations and exposure, but differ in the parameterization and detail of the models used. IMPACT2002+ is based on a spatially detailed multimedia-model of West Europe (Jolliet et al. 2003). ReCiPe 2008 (Goedkoop et al. 2009) uses a fugacity model based on the European System for the Evaluation of Substances; however, with better soil and air compartmentalization and other improvements (Huijbregts et al. 2005). USEtox is a consensus model, developed by an UNEP-SETAC group on toxic impact assessment (Rosenbaum et al. 2008). In USEtox, seven LCIA models were calibrated together and the simplified consensus model was used to produce characterization factors, which were recommended by the developers of the original LCIA methods.

Since the three models presented indicators in different units, the results were normalized for comparison. The normalization was done by calculating a reference impact based on European emissions in the year 2000 (Sleeswijk et al. 2008). In order to avoid bias in normalization (Heijungs et al. 2007), reference emissions were included only for substances, which were also present in the national inventory (e.g., emissions of atrazine were not included in the reference or in the national inventory, but emissions of copper were included in both).

2.2 Input–output model and analysis

Environmentally, EEIO was used to interpret the results of the emission inventory and to identify the main economic interactions causing the emissions. EEIO extends the input–output tables of the national accounts with the emission intensity per industry and with a characterization model (Suh and Huppes 2005). In this study, the ENVIMAT EEIO model with 150 sector resolution (Seppälä et al. 2009) was used, but only toxic emissions to air, water, and agricultural soil were included. The model equation was then:

$$q = CF(I - A)^{-1}f \qquad (1)$$

where q is the indicator result for human toxicity and ecotoxicity [kilogram reference substance], C is the matrix of characterization factors [kilogram reference substance/

kilogram] (2-by-62 matrix), F is the emission intensity matrix [kilogram per M€](62-by-150 matrix), I is the identity matrix, A is the input–output coefficient matrix [M€/M€ in basic prices] (150-by-150 matrix) and f is the final demand vector [M€] (150-by-1 vector).

Characteristics of the EEIO model were studied with structural path analysis (SPA). It is a method for extracting individual flows from the whole EEIO system described by Eq. 1 (Lenzen 2003). It can be used to identify the main economic interactions which contribute most to the studied overall environmental impact. SPA is commonly used in hybrid LCA to simplify input–output-based results and to focus the collection of more detailed emission inventories (Lenzen and Crawford 2009). Simplification is necessary to interpret the results, since the amount of economic interactions increases exponentially when more layers of supply chain are included in the analysis. For example, following the interactions of 150 industries for three production layers includes more than three million economic pathways. However, usually only a few dozen pathways cover the most of the environmental impact (Lenzen 2003). The SPA algorithm starts with the final demand supplied to consumption, investments, and export. It then follows the supply chain backwards until toxic emissions are encountered. The emissions are characterized and the path from final demand to the emission source is stored in a list. The list is finally sorted and the most important pathways are identified for further analysis.

3 Results and discussion

3.1 Comparing substance priority in LCIA and Baltic Sea integrated assessment for ecotoxicity

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 from ships (Fig. 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 tributyltin (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).

The difference between ReCiPe and the two other models was caused by differences in the ecotoxic potential

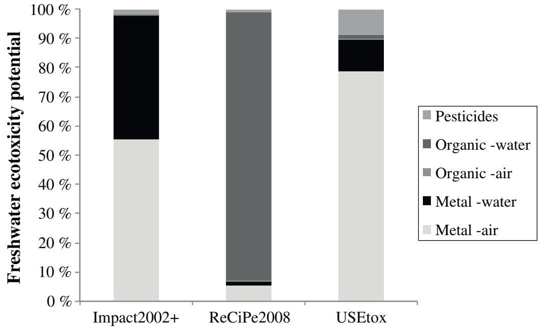

Fig. 1 Comparison of the share of toxic load from substance groups in the three assessed LCIA models

of TBT compounds. Although their use has been banned since 2003, emissions from leaching ship paints were still estimated to be 4,000 kg in 2005 (Ministry of the Environment 2006). 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). 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. The half-life of TBT in water is 9 days in the USEtox database, while in the ReCiPe database it is 128 days (Huijbregts et al. 2005; Hauschild et al. 2010). This reflects the considerable variability in the measured experimental degradation rates (ECHA 2008). As a consequence, the fate factor of TBT is significantly lower in USEtox than in ReCiPe. 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.

The share of agricultural pesticides varied considerably between models. In IMPACT2002+ pesticide emissions caused 1.7% of the ecotoxic impact mainly through the use of dimethoate, glyphosate, prochloraz, and propiconazole. In ReCiPe, impacts from pesticide emissions were only 0.6% of the total, caused mainly by linuron and dimethoate. In USEtox pesticide emissions caused 9% of the ecotoxic impact and caused by a broad scale of pesticides, but mainly prochloraz, mancozeb, and linuron. Therefore both the overall share and substance prioritization of agriculture varied between models used. The results of this study are in contrast with the recent materials and products prioritization studies, where agriculture was found to cause 82% of freshwater ecotoxic impact (UNEP 2010; EPA 2009). However this result was obtained using only one set of characterization factors and due to the extreme variability between models cannot be considered to be reliable. In further prioritization studies model intercomparison is recommended to test the robustness of results.

In the development of normalization factors for the year 2000 using ReCiPe (Sleeswijk et al. 2008), pesticides were found to cause 24–30% of freshwater ecotoxic impacts. The impacts were mainly caused by atrazine use, which has been banned in the EU since 2004 (Sass and Colangelo 2006). This highlights the importance of using recent emission inventories in priority setting. Without atrazine emissions the share of pesticides would have been 9–20% of total, which is comparable to the results of this study.

The LCA included emissions to air, water, and soil and evaluated the effects over time and space. In contrast, the integrated assessment was based on measured concentrations and quality targets in the Baltic. The two approaches cannot be directly compared, but can be seen as complementary. In the integrated assessment PCBs, lead, mercury, cesium-137, DDT/DDE, TBT, benz[a]anthracene, cadmium, and dioxins/furans were identified as substances of high concern (HELCOM 2010). The prioritization was done based on the proximity of observed concentrations to environmental quality limits. In contrast to current best practices in chemical risk management, the limits were set mainly based on human exposure, and not on ecotoxicological dose–response data. This was done as a precautionary approach to protect consumers from secondary poisoning (HELCOM 2010). Of the highlighted substances, TBT was the only compound identified as a source of concern. PCBs, DDT/DDE, and cesium-137 were not identified as they have no current emissions and the observed concentrations are caused by chemical residence in sediments. Lead, mercury, and cadmium are bioaccumulative heavy metals (Hendriks and Heikens 2001). The current LCIA models do not include ecotoxicity from secondary poisoning through food web bioaccumulation, which results in the underestimation of the toxic potential of food web accumulative substances. To some extent, this explains also the lack of benz[a]anthracene and dioxins/furans in the LCIA results, while they are a source of concern in the integrated assessment (HELCOM 2010). Basing the toxicity endpoints to critical body residues instead of dissolved concentrations would possibly make the results of LCIA more closely comparable to those of chemicals risk management in general.

3.2 Human toxicity potential

Compared to the several orders of magnitude of difference in estimating ecotoxic impacts, the models estimated quite similar human toxicities (Fig. 2). The share of Finnish emissions of the European reference emissions ranged

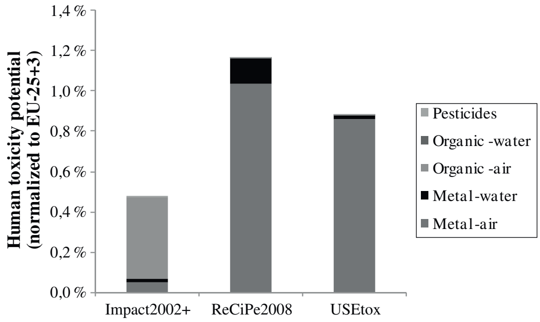

Fig. 2 Human toxicity potential of Finnish emissions in 2005 estimated with three life cycle impact assessment (LCIA) models. Impacts were reported as a share of the European reference emissions

between 0.5% and 1.2%. The main difference between models is the toxicity of polycyclic aromatic hydrocarbons (PAHs), dioxins/furans, and benzene. Dioxins, PAHs, and benzene amounted to 85% of the human toxic potential in IMPACT2002+, but less than 0.1% in ReCiPe and USETOX. Based on national air-quality assessments, PAH exposure is a cause for concern, since atmospheric concentrations of benzo[a]pyrene regularly exceed the regulatory limits (Alaviippola et al. 2007). The concentrations are the highest during wintertime and are caused by the incomplete combustion of wood. Also the exposure from food is at a high level (EVIRA 2010). In comparison, benzene levels are generally below regulatory limits (Pietarila et al. 2002). The results from ReCiPe and USEtox concerning PAHs were clearly in contradiction with the integrated assessments, while IMPACT2002+ captured the effect better.

The consumption of Baltic fish has been regulated due to high concentrations of dioxins and furans (EVIRA 2010), which have well known emission estimates, but lack characterization factors in USEtox. In IMPACT2002+ and ReCiPe, dioxins (2,3,7,8-TCDD) had the highest reported characterization factor for human toxicity, but dioxins were highlighted only using IMPACT2002+. The main exposure route to humans is through Baltic fish. Therefore it may be that the fugacity and food web models applied in the USEtox and ReCiPe do not represent the Baltic ecosystem and foodweb in sufficient detail. This could be tested by developing site-specific characterization factors for the Baltic, using the Popcycling Baltic model with foodweb components (Mattila and Verta 2008). As such, the IMPACT2002+ model represents the chemical risk management results for organics better than USEtox or ReCiPe.

Metal emissions dominated the results from ReCiPe and USEtox. Both models highlighted zinc, mercury, lead, arsenic, and cadmium, but ReCiPe identified vanadium as an additional priority pollutant. In IMPACT2002+ only arsenic zinc were identified. Cadmium, mercury, and

Table 1 The main economic pathways in the Finnish 2005 input–output table causing freshwater aquatic ecotoxicity calculated with IMPACT2002+, ReCiPe, and USEtox LCIA models

Emission	IMPACT2002+ industries	Share of impact
Copper–water	Non-ferrous metals	11.2%
Copper–air	Households	9.0%
Zinc–water	Artificial fibers	6.6%
Copper–air	Non-ferrous metals	4.2%
Zinc–air	Households	2.4%
Emission	**ReCiPe Industries**	**Share of impact**
TBT–water	Shipping	69.6%
TBT–water	Shipping → retail trade	3.5%
TBT–water	Shipping → pulp and paper	1.8%
TBT–water	Shipping → residential construction	1.1%
Vanadium–air	Oil refining	0.8%
Emission	**USEtox Industries**	**Share of impact**
Vanadium–air	Oil refining	13.3%
Zinc–air	Households	4.5%
Zinc–water	Artificial fibers	4.2%
Vanadium–air	Households	1.3%
TBT–water	Shipping	1.1%

arsenic were also identified as sources of concern in the recent integrated assessment on chemical exposure in Finland, since their uptake is close to the tolerable weekly intakes (EVIRA 2010). Zinc and vanadium were not included in the assessment and lead concentrations were found to be decreasing. Overall the results of the LCIA models are found to be in agreement with the integrated assessment in respect to heavy metals.

The integrated assessment included several substance groups, which were not included in the LCIA models. Nitrates and fungal toxins are included in food safety, but no characterization factors exist for converting agricultural practices into human exposure. Polybrominated diphenyl ethers, perfluorooctanesulfonic acid, and diethylhexylphthalates are measured from food (EVIRA 2010), but they lack LCIA characterization factors and emission estimates.

3.3 Priority products and main emission pathways

Applying input–output analysis and structural path analysis to the emission inventory allowed a consumption-based perspective on the emissions of hazardous substances. This exercise could be used to identify the priority products, which have the highest embodied emissions in their supply

Table 2 The main structural paths in the Finnish input–output table causing human toxic impacts calculated with IMPACT2002+, ReCiPe, and USEtox LCIA models

Emission	IMPACT2002+ Industries	Share of impact
PAH–air	Households	19.1%
Benzene–air	Households	6.3%
Dioxins and furans–air	Electricity	5.6%
Dioxins and furans–air	Non-ferrous metals	3.7%
Dioxins and furans–air	Electricity → renting apartments	3.6%
Emission	**ReCiPe Industries**	**Share of impact**
Vanadium–air	Oil refining	8.2%
Arsenic–air	Non-ferrous metals	3.7%
Arsenic–air	Oil refining	3.4%
Mercury–air	Iron and steel manufacturing	2.6%
Arsenic–water	Non ferrous metals	2.6%
Emission	**USEtox Industries**	**Share of impact**
Zinc–air	Households	14.6%
Mercury–air	Iron and steel manufacturing	6.4%
Zinc–air	Non-ferrous metals	2.8%
Mercury–air	Pulp and paper industry	2.6%
Zinc–air	Iron and steel manufacturing	2.3%

chain (UNEP 2010). These would then be priority candidates for policy actions.

The models yielded overlapping results. IMPACT2002+ and USEtox highlighted zinc emissions from artificial fiber 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 TBT from shipping, consumed either directly or through purchases of retail trade, pulp and paper, or residential construction.

Compared to ReCiPe the USEtox put less emphasis to TBT emissions from shipping. Vanadium emissions from oil refining are considered as the main priority, followed by zinc and vanadium emissions from domestic fuel use and zinc water emissions from artificial fiber production. Overall, the focus is moved from shipping to petrochemical manufacture and use (Table 1).

Compared to ecotoxic priority setting, the models had more differences in the human toxic priority pathways. IMPACT2002+ highlighted the main emission sources of PAHs, benzene, and dioxins, identifying among others the electricity use in apartments as a key pathway. The other two models focused more on direct emissions of zinc, mercury, and arsenic from oil refining, metals industry and pulp and paper industry (Table 2).

All models could be used to identify a set of priority products and pathways. However the prioritization differed considerably as did the evenness of pathways. Five top ranking pathways covered 70% of the ecotoxic impact in ReCiPe, 33% in IMPACT2002+, and only 24% in USEtox. 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, while based on the two other method a focus on few key pollutant sources would be recommended. In human toxicity, completely opposite focus would be obtained using IMPACT2002+ or the two other models, with IMPACT2002+ focusing on household energy use and the others controlling on heavy metal emissions from industry.

4 Conclusions

Based on the results, LCIA models can be used for priority setting in chemical emission control and consumption-based analyses. However, careful selection of the model is advised since the models provide very differing prioritizations. For ecotoxicity, ReCiPe provided the prioritization most consistent with the integrated assessment. For human toxicity, IMPACT2002+ provided the priorities most similar to integrated assessment. A comparison of characterization model output to expert assessment is therefore recommended in further prioritization studies. Since the studied models differed from expert assessment mostly in substances which are bioaccumulative, further studies to include bioaccumulation to LCIA models is recommended.

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Sensitivity analysis of environmentally extended input–output models as a tool for building scenarios of sustainable development

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ABSTRACT

There is an urgent need to develop scenarios and roadmaps for a more sustainable future than where business-as-usual is heading. This paper addresses the use of sensitivity analysis for analyzing environmentally extended input–output (EEIO) models in order to develop cost-effective and comprehensive scenario building. Main components of resource use, emission intensity and final demand are extracted from the complete network of interactions contained in the input–output tables of the national accounts. The method is demonstrated using a detailed Finnish EEIO-model (ENVIMAT). Based on the results, only 0.3% of the 23 103 interactions were found to have a significant effect on Finnish greenhouse gas emissions. The same parameters were also relevant for waste generation and land use, but not for gross domestic product. The identified main components were tested by structural decomposition. Actual development of greenhouse gas emissions from 2002 to 2005 was compared to that predicted by updating only the identified components. Based on the results, the development of greenhouse gas emissions could be predicted with high accuracy using only the identified main components. Generalizing the results, sensitivity analysis can assist in identifying the main components to be included in future scenarios for sustainable development.

1. Introduction

The global environmental crisis is becoming more and more evident. Several planetary thresholds have been exceeded (Rockström et al., 2009) and the growth rate of global greenhouse gas emissions is faster than ever before (Peters et al., 2011a). Comparing the actual emissions to the IPCC scenarios (Nakicenovic et al., 2000), the annual emissions have reached the “worst case” scenario (“A1FI”), predicting a 4 °C global surface warming by the end of the century (IPCC, 2007). Looking at other aspects of environmental sustainability, the development through the last 30 years has been found to follow closely the “standard run” of the Limits to Growth simulations (Turner, 2008). The standard run predicts an overshoot of resource use in the 1990s and a collapse of society around the middle of the 21st century (Meadows et al., 1972). Therefore there is a growing body of scientific evidence that the current trend of global development needs to be changed rapidly. Following the current patterns of behavior would result in a very undesirable and possibly unlivable future environment.

At the same time the world has become highly interconnected. The production structure within individual countries is a complex network of interactions (Lenzen, 2003). In addition global supply chains connect the consumption of one country to the production of another (Peters and Hertwich, 2008). Attempts to control emissions with national and regional measures have even resulted in emission increases globally through international trade (Peters et al., 2011b; Wiedmann et al., 2010). Because of the high level of global interconnection, it is difficult to change the system by focusing on individual industries, emission sources or even countries. This makes emission reduction a typical “wicked problem”, where improvements in one part of the system result in new problems in other parts (Jackson, 2003). Solving wicked problems calls for a thorough systems analysis of the current situation, its trends and (most importantly) identifying possible futures, which avoid the problems of continuing current development (Ackoff, 1974, 1999).

Various sustainability scenarios have been built with different methodologies (Ahlroth and Höjer, 2007; Mander et al., 2008; NIES, 2008; Rijkee and van Essen, 2010). Forecasting from the business as usual as well as backcasting from a potential future state has been applied (Rijkee and van Essen, 2010). The widely referenced IPCC scenarios represent both of these options (Nakicenovic et al., 2000). Delphi expert panels and system dynamics have also been applied. The most known examples of these are the Limits to Growth simulations (Meadows et al., 1972) as well as several general equilibrium models (such as GTAP (Hertel and Hertel, 1999)). In addition several industries have applied foresight (Salo and Cuhls, 2003; UNIDO, 2005) to create their own sets of future scenarios, for example transport (Helmreich and Keller, 2011), energy (IEA, 2008) and food production (Beddington, 2011).

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There are in general two approaches for scenario building: forecasting and backcasting. Forecasting builds a dynamic model of historical development and with the aid of assumptions projects future development across the economy. Backcasting starts from the future and asks, what the system should look like to provide the desired characteristics (i.e. dramatically less resource consumption and greenhouse gas emissions). A roadmap is then later built towards that desired configuration. However a considerable problem in backcasting is that the desired state can be achieved with many combinations of system changes. And detailed representations of economic systems are complex, for example the single country EIOLCA EEIO-model has 129 976 inter-industry connections (Hendrickson et al., 2005). How can one identify all the relevant potentials for change with such an amount of potential variables? Consequently most scenarios oversimplify the problem, looking at single sectors or single regions (Warren, 2011). Without a full supply chain view, the approaches may miss important linkages which influence the system in focus. For example the emissions of food and fuel production are both very strongly interlinked and connected to the extent of climate change. Yet assessments of the combined effects of biofuels, food and climate change are rare (Haberl et al., 2011). Even in very comprehensive studies on agriculture, the energy scenarios of the background economy are rarely considered and the extent of climate change is limited to scenarios well below the possible 4 °C temperature rise (Haberl et al., 2011).

Economic input–output (IO) analysis seems like the ideal tool for backcasting studies (Duchin and Lange, 1995). It was originally developed to analyze the interlinkages between industries of a country and to identify the amount of production needed to satisfy increased consumption (Leontief, 1936). Today it forms the basis of the collection of national accounts and the calculation of gross domestic product (GDP) (Ten Raa, 2006). Especially when the input–output tables of multiple regions are connected (so-called MRIO-tables), the tool can capture the entire supply chain (Tukker et al., 2009). Therefore the interactions between countries and industries can be quantified and analyzed. The economic input–output tables can readily be coupled with satellite accounts of emissions and resource extraction (Leontief, 1970). Environmentally extended input–output (EEIO) analysis has become a key tool for sustainability assessment (Murray and Wood, 2010; Suh, 2009). The ability to follow global supply chains has allowed the consideration of local emissions as consumer, producer or shared responsibility problems (Lenzen et al., 2007). Applied to backcasting, once a sustainability goal is defined, various economy wide scenarios can be set up to see which would meet the goal. The comprehensiveness and detail of EEIO forces the analyst to consider the economy as a whole, thus avoiding the problems of partial analysis.

The comprehensiveness and detail of EEIO models is also their main weakness. When involving stakeholders in scenario work, the sheer complexity (amount of parts and connections) present in input–output tables makes it mentally difficult to capture the whole system. Several analytical tools have been developed to identify the main components from the network of an EEIO model. For example structural decomposition analysis (SDA) describes, which parts of the system explain most of the change between years (Dietzenbacher and Los, 1998; Peters et al., 2007). Conversely structural path analysis (SPA; (Lenzen, 2003)) data mines the environmentally most relevant production chains from the EEIO-system for a given year. The methods have also been combined to yield structural path decomposition (SPD) (Wood and Lenzen, 2009), which explains the main supply chains where emission change manifests. The problem in analyzing single supply chains is that network nodes, which belong to several supply chains are not identified (i.e. the emissions of business services are allocated between almost all industries). Sensitivity analysis on the other hand identifies important nodes in the model, without regard to the supply chains where they belong. It is commonly used in life cycle assessment (Heijungs, 2010) and recently also in EEIO (Mattila, 2012; Wilting, 2012).

A general result in complex models is that usually only a small fraction of all the variables are relevant for a given decision situation (Saltelli et al., 2008). This has also been observed in the case of EEIO models for single indicators such as the ecological and carbon footprints (Mattila, 2012; Wilting, 2012). The aim of this study was to identify relevant parts of the economy to be included in backcasting scenarios. It was also analyzed, whether the same parts of the economy could explain the impacts in several impact categories (global warming, land use, waste generation and gross domestic product). The process was checked by predicting the observed development in greenhouse gas emissions between 2002 and 2005, using only the components, which were identified as important. Finally a generalization of the results to a more general case of backcasting is presented.

2. Materials and Methods

2.1. Sensitivity Analysis of the EEIO Model and Testing of Its Results

Sensitivity analysis attempts to answers 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, 2010; Heijungs and Suh, 2002). The perturbation analysis yields the sensitivity of the model output to relative changes in the input (i.e. $(\Delta f/f)/(\Delta x/x)$).

The EEIO model can be described with a single equation (Leontief, 1970):

$$\mathbf{g} = \mathbf{B}(\mathbf{I}-\mathbf{A})^{-1}\mathbf{y} = \mathbf{M}\mathbf{y} = \mathbf{B}\mathbf{x}, \tag{1}$$

where $\mathbf{g}$ is the vector of indicator results (categories of GDP, employment, environmental impacts, resource use), $\mathbf{B}$ is the intensity of production matrix (impact/production amount; impact-by-industry), $(\mathbf{I}-\mathbf{A})^{-1}$ is the Leontief inverse matrix, $\mathbf{B}$ is a diagonal matrix, $\mathbf{A}$ is the input-coefficient matrix (industry-by-industry or product-by-product) and $\mathbf{y}$ is the vector of final demand by product (or industry). The $\mathbf{M}$ is the intensity multiplier matrix, which contains the life cycle emission intensities for all products or industries and $\mathbf{x}$ is the amount of total production needed to produce the entire consumption $\mathbf{y}$.

Applying partial derivatives for Eq. (1), the following sensitivity indices are obtained (Heijungs, 2010):

$$S_{y,ki} = \frac{\delta g_k/g_k}{\delta y_i/y_i} = M_{ki}\frac{y_i}{g_k} \tag{2}$$

$$S_{a,kij} = \frac{\delta g_k/g_k}{\delta a_{ij}/a_{ij}} = M_{ki}x_j\frac{a_{ij}}{g_k} \tag{3}$$

$$S_{b,kj} = \frac{\delta g_k/g_k}{\delta B_{kj}/B_{kj}} = x_j\frac{B_{kj}}{g_k}, \tag{4}$$

where the subscripts refer to the corresponding element of the matrix. Since the Eq. (3) gives the sensitivity in regard to changes in the $(\mathbf{I}-\mathbf{A})$ matrix (technology matrix, in Heijungs, 2010), the S_a of diagonal elements $(1-a_{ii})$ were scaled with the ratio of $a_{ii}/(1-a_{ii})$ to give the sensitivity of the original input coefficient.

S_y describes the sensitivity to final demand, S_a to inter-industry input-coefficients 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. (Final demand was not disaggregated to overall demand scale and demand structure as in structural decomposition.

Since the model in Eq. (1) is linear in regard to the scale of demand, overall demand size will always have a sensitivity index of 1.0.) Sensitivity was done on the **A** matrix instead of the transactions flow table, since **A**, **B** and **y** could be considered as independent, while changes in the flow table would influence the whole system through total output.

The 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 (cf. 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. As the input–output tables also do not include any behavioral changes, they have to be taken into account in the scenario building stage. 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.

The predictive power of the identified main components was tested by using an actual development trend between 2002 and 2005. Observed changes in selected indicators were compared to those obtained by updating only the identified main components of the EEIO model from 2002 to 2005 values. Structural decomposition and comparison to the indicator values of individual industries were used to evaluate the relevance of the identified main components. The structural decomposition was conducted by taking the average of all possible decompositions, which has been found to be the most reliable estimate (Dietzenbacher and Los, 1998). The equation used for SDA was (Miller and Blair, 2009):

$$\Delta\mathbf{g} = \mathbf{B}(\mathbf{I}-\mathbf{A})^{-1}\Delta\mathbf{y}_1 y_2 + \mathbf{B}(\mathbf{I}-\mathbf{A})^{-1}\mathbf{y}_1\Delta y_2 + \mathbf{B}\Delta(\mathbf{I}-\mathbf{A})^{-1}\mathbf{y}_1 y_2 + \Delta\mathbf{B}(\mathbf{I}-\mathbf{A})^{-1}\mathbf{y}_1 y_2 \quad (5)$$

where $\mathbf{y}_1$ and $\mathbf{y}_2$ are the structure and overall scale of final demand.

2.2. EEIO Model Used in the Case Study

The case study in this work was the Finnish economy in 2002 and 2005. We have studied the economic system extensively through the ENVIMAT EEIO-model (Seppälä et al., 2011). The model is an industry-by-industry input–output model with 151 industries and 78 environmental emission and resource extraction categories. A detailed description of the data sources and modeling assumptions can be found in Seppälä et al. (2011). Since its publication the model has been supplemented with analyses on trade balances of embodied emissions (Koskela et al., 2011), land use and biodiversity (Mattila et al., 2011a), ecological footprint (Mattila, 2012) and ecotoxic impacts (Mattila et al., 2011b). In this study, four indicators were chosen for analysis: greenhouse gas emissions (global warming potential, GWP), waste generation, land use and gross domestic product at factor costs (GDP). These indicators were chosen, since they are relatively commonly applied and generally understood. The greenhouse gas emissions and waste generation were based on the official statistics of the environmental administration (VAHTI-database), while the factor cost gross domestic product (i.e. value added per industry) was obtained from the National Accounts (Statistics Finland, 2007a). The land use was based on CORINE land cover data (Härmä et al., 2004) allocated to industries as explained in (Mattila et al., 2011a).

For the purposes of this study, the focus was on the impacts on the national level. Consequently the embodied impacts of imports were not included, but the impacts of producing exports were included. This approach is analogous to the national emission accounting approach (i.e. production based approach). Consumption based inventories have been thoroughly discussed and analyzed in previous work (Koskela et al., 2011; Mattila, 2012) and can be included in the sensitivity analysis when necessary.

3. Results and Discussion

3.1. Sensitivity Indices

Very few parameters had sensitivity index higher than 0.01 for GDP, GWP, land use or waste generation (Fig. 1). Most of the parameters had sensitivity indices below 10^{-6}, indicating that for practical purposes they have little significance to the results. Together the identified components represented less than 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 define most of the model output (Saltelli et al., 2008). (Decreasing the sensitivity limit to 0.001 would identify 1.3% of model parameters as important.)

The identified main components for GWP and the associated sensitivities for other key indicators are presented in Tables 1–3. Some main components were found to be common to most indicators. For example the use of animal products in various industries had relevance to GWP, land use and waste generation. Similarly the intermediate use of pulp and paper products influenced GWP, GDP and land use and the use of cement influenced both GWP and waste generation. The intensities of electricity production, pulp and paper production, animal farming and road freight were also identified as relevant for several indicators. Also the final demand of pulp and paper (exported), apartments, trade services and residential construction were relevant for all indicators. Final demand of mobile electronics, meat products and restaurant services was identified as a main component for most of the evaluated criteria.

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 input-coefficients (S_a, n = 15). Emission intensities for electricity production, iron and steel manufacture, animal farming and pulp and paper production had the highest sensitivity indices (ranging between 0.08 and 0.27), followed by the final demand of pulp and paper, iron and steel, apartments and trade services (sensitivity index between

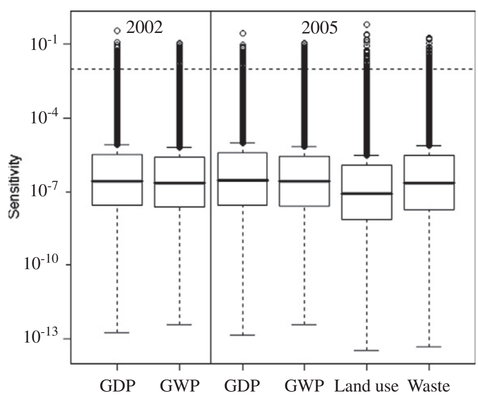

Fig. 1. The distributions of sensitivity indices (S_a, S_b and S_y combined) for gross domestic product (GDP), global warming potential (GWP), land use and waste generation. The box and whiskers plot describes the median of the distribution, the 25% percentile and the 75% percentiles and the whiskers 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).

Table 1
Main components of intermediate use (S_a) for greenhouse gases (GWP) in 2005 and the associated sensitivities for other indicators. The indicators in parenthesis represent the year 2002 values. Sensitivity indices higher than 0.01 are marked in bold.

Supplying industry	Using industry	Sensitivity index			
		GWP	GDP	Land use	Waste
Animal farming	Dairy	**0.045** (0.047)	0.005 (0.006)	**0.035**	**0.030**
Electricity	Real estate	**0.036** (0.003)	0.003 (0.003)	0.009	0.007
Animal farming	Meat processing	**0.034 (0.038)**	0.004 (0.005)	**0.026**	**0.023**
Animal farming	Animal farming	**0.033 (0.034)**	0.004 (0.005)	**0.026**	**0.022**
Electricity	Pulp and paper	**0.026** (0.004)	0.002 (0.004)	0.007	0.005
Electricity	Trade	**0.025** (0.002)	0.002 (0.002)	0.006	0.005
Pulp and paper	Pulp and paper	**0.022 (0.022)**	**0.010** (0.009)	**0.055**	0.040
Cement	Concrete	**0.013 (0.012)**	0.000 (0.000)	0.000	**0.017**
Waste disposal	Real estate	**0.011 (0.01)**	0.001 (0.001)	0.000	0.000
Concrete	Residential construction	**0.011 (0.01)**	0.003 (0.003)	0.001	**0.046**
Civil engineering	Civil engineering	**0.011** (0.008)	0.007 (0.006)	0.003	0.009
Meat processing	Meat processing	**0.011 (0.016)**	0.003 (0.004)	0.008	**0.010**
Iron and steel	Iron and steel	**0.010** (0.004)	0.001 (0.000)	0.000	0.006
Electricity	Other services	**0.010** (0.001)	0.001 (0.001)	0.003	0.002
Trade	Residential construction	**0.010** (0.009)	**0.013** (**0.015**)	0.003	0.003

0.07 and 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 and 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). To some extent this result was caused by the treatment of construction as a separate part of housing in the input–output models. If investments were endogenized in the model, the environmental impacts of apartments would increase.

To some extent the relationship between economic growth and environmental pressure is more complex that could be deduced from the static IO-tables. Economic growth allows investments, part of which are used to reduce the emission intensity of products. Over time therefore economic growth may reduce emission intensity. However a major problem in this (as identified by the sensitivity analysis) is that the value added and emissions are caused in different parts of the economy. For example, animal production has little value added available to invest in emission reduction. Therefore either a redistribution of value added or government intervention is needed to reduce the emissions of those sectors which have little influence on GDP but high influence on emissions.

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 ($S_y = 0.06$).

Table 2
Main components of intensity (S_b) for greenhouse gases (GWP) in the year 2005 and the associated sensitivities for other indicators. The indicators in parenthesis represent the year 2002 index values. Sensitivity indices higher than 0.01 are marked in bold.

Producing industry	GWP	GDP	Land use	Waste
Electricity production	**0.277** (**0.359**)	**0.019** (**0.018**)	**0.044**	**0.044**
Iron and steel manufacture	**0.096** (**0.073**)	0.006 (0.005)	0.000	**0.010**
Animal farming	**0.079** (**0.077**)	0.005 (0.007)	**0.057**	**0.010**
Pulp and paper production	**0.058** (**0.048**)	**0.023** (**0.035**)	0.001	**0.150**
Road freight	**0.046** (**0.041**)	**0.027** (**0.026**)	–	–
Refined petroleum production	**0.044** (**0.041**)	0.007 (0.006)	0.000	0.004
Waste disposal	**0.042** (**0.048**)	0.002 (0.002)	0.000	–
Basic chemicals production	**0.041** (**0.034**)	0.007 (0.008)	0.000	**0.037**
Trade	**0.038** (**0.024**)	**0.102** (**0.098**)	0.001	–
Air transport	**0.031** (**0.025**)	0.006 (0.006)	–	–
Crop growing	**0.029** (**0.034**)	0.002 (0.003)	**0.050**	0.005
Water freight	**0.028** (**0.013**)	0.007 (0.006)	–	–
Cement production	**0.025** (**0.021**)	0.001 (0)	0.000	0.002
Civil engineering	**0.020** (**0.014**)	**0.013** (**0.012**)	–	–
Heat production	**0.018** (**0.017**)	0.001 (0.001)	–	0.000

Table 3
Main components of final demand (S_y) for greenhouse gases (GWP) in the year 2005 and the associated sensitivities for other indicators. The indicators in parenthesis represent the year 2002 values. Sensitivity indices higher than 0.01 are marked in bold.

Consumption category	GWP	GDP	Land use	Waste
Pulp and paper	**0.095** (**0.121**)	**0.041** (**0.058**)	**0.233**	**0.171**
Iron and steel	**0.078** (**0.056**)	**0.011** (0.008)	0.002	**0.044**
Apartments	**0.075** (**0.092**)	**0.115** (**0.113**)	**0.057**	**0.024**
Trade	**0.065** (**0.050**)	**0.086** (**0.076**)	**0.017**	**0.017**
Electricity	**0.055** (**0.053**)	0.005 (0.004)	**0.014**	**0.010**
Residential construction	**0.050** (**0.044**)	**0.072** (**0.066**)	**0.054**	**0.079**
Dairy products	**0.036** (**0.032**)	0.006 (0.007)	**0.029**	**0.024**
Basic chemicals	**0.035** (**0.033**)	0.008 (0.008)	0.003	**0.053**
Public administration	**0.031** (**0.030**)	**0.066** (**0.065**)	0.009	**0.010**
Petroleum products	**0.030** (**0.028**)	0.006 (0.006)	0.001	0.004
Mobile electronics	**0.027** (**0.021**)	**0.061** (**0.066**)	**0.013**	**0.015**
Civil engineering	**0.026** (**0.021**)	**0.017** (**0.015**)	0.008	**0.022**
Other services	**0.025** (**0.024**)	**0.040** (**0.039**)	0.008	0.008
Meat products	**0.024** (**0.024**)	0.006 (0.006)	**0.018**	**0.023**
Water transport	**0.022** (**0.012**)	0.006 (0.007)	0.000	0.000
Health services	**0.021** (**0.023**)	**0.062** (**0.058**)	0.007	0.007
Air transport	**0.020** (**0.014**)	0.006 (0.005)	0.001	0.001
Education	**0.017** (**0.022**)	**0.059** (**0.056**)	0.006	0.006
Business services	**0.014** (**0.013**)	**0.038** (**0.029**)	0.005	0.006
Restaurants	**0.013** (**0.016**)	**0.016** (**0.017**)	0.007	**0.010**
Social work	**0.012** (**0.013**)	**0.041** (**0.038**)	0.003	0.003
Crops	**0.011** (**0.012**)	0.002 (0.001)	**0.017**	0.009

The list of relevant components is similar to that obtained for Spain for CO_2 emissions (Tarancón Morán and del Río González, 2007). Cement, electricity, steel and animal production were identified as the main components for Spain as in Finland. In addition in Finland, the pulp and paper industry was identified as a main component for climate change.

The same nodes were identified as important between years, which indicated that at least on a short term, indexes identified from a single year's data can be used for subsequent years. The highest sensitivity indices were in the impact intensities (S_b) and final demand (S_y). These correspond to the consumption ("highest life cycle impacts") and production ("highest direct impacts") based viewpoints of emission reduction (Wiedmann et al., 2010). In comparison, the sensitivities of input-coefficients (S_a) were lower. They represent the total flow of embodied emissions moving through a production node (e.g. the wastes produced in the life cycle of the commodities used at that industry). Based on the results of the sensitivity analysis, controlling the emissions and other impacts of the Finnish economy requires a combined producer and consumer responsibility approach to target both emission intensities and consumption.

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.

It should be noted however, that the low share of significant parameters was caused by the high level of disaggregation in the input–output tables. As the amount of disaggregated sectors doubles the amount of potential connections quadruples. Also with low disaggregation, non-important industries are grouped together with important, therefore resulting in a higher share of identified industries. Therefore with a lower level of disaggregation the share of important parameters is likely to be higher. On the other hand, the high level of disaggregation for example in the manufacturing industries could hide important overall development in the aggregated industry group. The approach of Wilting (2012), where sensitivity analysis is done also on blocks of parameters in the input–output table could provide another kind of way to identify the key parameters.

3.2. *Using the Identified Main Components to Predict Development Over Time*

The main components identified from the year 2002 EEIO model were updated from the year 2002 values to year 2005 values, while all other parameters were kept constant. In order to remove the effects of price fluctuations, the dataset of 2005 was converted into year 2002 constant prices with double deflation (Peters et al., 2007) and price indexes (Statistics Finland, 2009). The results were then compared to the actual development in greenhouse gas emissions from 2002 to 2005. The aim was to test, whether the identified components could explain the change between years.

Based on the results of the exercise, 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) (Fig. 2). The overall development in final demand (y_2) 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 (Statistics Finland, 2007b). (In Finland coal power plants are used primarily for handling the peak load demand, while nuclear power and biomass provide the base load (Soimakallio et al., 2011)). 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.

The influence of such external factors is however a challenge for scenario building. Although the sensitivity analysis could identify the importance of potential changes in electricity emission intensities, the method could not predict the changes over time. The input–output tables do not contain the causes or drivers of change, only a static representation of a situation in a given year. Therefore the reliability of the scenario work will always depend on the quality of the assumed trends. The proposed methodology only guides in prioritizing those parts of the economy, which have the potential to have a large impact on sustainability.

Looking at individual industries, the differences in prediction were large, but they were compensated by similar differences in other industries. The largest errors were the underestimation of the embodied emissions of electronics (by 17%, 0.3 Mt CO_2eq), trade services, dairy products and the manufacture of ferrous and nonferrous metals, pipes and other general machinery. These were somewhat countered by the overestimation of the emissions of education (by 22%, 0.3 Mt), manufacturing of ships, waste disposal, health services and the use of restaurant services. The errors were mainly caused by the very rapid change in either the emission intensity or the final demand of the corresponding industry. For example the emission intensity of the electronics industry tripled from 2002 to 2005, while the emission intensity of education decreased to 25% of the 2002 level. (The decrease in the emission intensity of education was again caused by the mild weather, decreasing the energy needs for heating.) Therefore although the sensitivity analysis identifies the crucial elements for the overall development, high variation may be found in individual industries. This is in line with earlier sensitivity analysis studies, where the sensitivities calculated for different supply chains were found to be highly variable (Wilting, 2012). Therefore the overall list of parameters, which are important for the full economy should be complemented with sector level data if more detailed analysis on some sectors if necessary.

Overall the sensitivity analysis provided a greatly narrowed down list of relevant parameters (ca. 60 main components out of 23 000 model parameters). The development in greenhouse emissions from 2002 to 2005 could be followed using only those main components, assuming that the components could be predicted with full accuracy. Therefore further backcasting should focus on those components since their assumed values determine to a large extent the outcome of the scenario.

4. Proposed Framework for Sensitivity Analysis Assisted Backcasting

The results of applying sensitivity analysis to EEIO can be generalized to a recommendation on comprehensive scenario building. The overview of the process is presented in Table 4. It borrows from Ackoff's interactive planning and can be used together with that methodology in its first stage or "ends planning" (Ackoff, 1974, 1999). The method is framed around the assumption that relevant stakeholders can be assembled to discuss the matters and agree on a sustainable

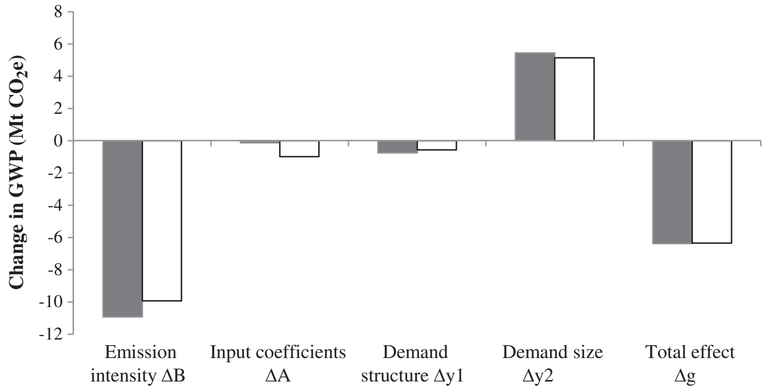

Fig. 2. The actual (dark) and predicted (white) change in global warming potential of the Finnish economy.

system state. Therefore it might not apply where the amount of stakeholders is extremely high or they are not willing to communicate openly (Jackson, 2003). In addition the method applies to cases where change at the sector level is considered important. Therefore it is not meant to substitute overall macro-level measures, such as carbon pricing.

First the current system is analyzed and the development trends are identified. On the basis of analysis key indicators are identified. The environmentally extended input–output tables provide a good basis for identifying the main components in the system which are responsible for the observed indicator values. Then sensitivity analysis is used to identify the main components of the system (i.e. which components need to be changed to alter the system). (If the initially proposed sensitivity limit of 0.01 provides too little or too many components, the limit can be adjusted accordingly to get a workable amount of components.) The identification is then tested by making a prediction of the change over a small time period and comparing it with actual development. Structural decomposition techniques (Dietzenbacher and Los, 1998; Wood and Lenzen, 2009) can be a great help in this stage. This comparison provides a quick way of identifying components, which were not noticed by the sensitivity analysis (such as very rapid change in some parts of the economy). After this stage the list of main components is updated.

Based on the list of main components, a comprehensive list of relevant stakeholders can be identified from consumers, government, industry and business. High sensitivities in final demand (S_y) indicate that consumers and government need to be included, main components in emission intensity (S_b) can be controlled within an industry and the intermediate use (S_a) requires a supply chain approach. Trends and mitigation potentials from individual main components are brought together in an EEIO model of the future system and the overall impact is evaluated. This stage allows the identification of possible conflicts between measures and the extent of indirect impacts on the rest of the economy. It should be emphasized that this stage is by no means an easy task. Since the input–output tables do not contain the causalities of production or consumption, the stakeholders involved have to take these into account in making comprehensive scenarios. In addition the stakeholders should evaluate to what extent the main components can actually be changed. The purpose of using input–output analysis at this stage is to describe the whole economy wide implications of those changes.

Ideally development trends for the future are explored and revised until a scenario can be constructed, which meets the sustainability criteria. After the scenario has been set up, the sensitivity analysis can be repeated to see if the sensitivities are still the same or whether the list of main components should be changed. For example a reduction in the emission intensity of electricity is likely to reduce the importance of electricity use from the sensitivities of S_y and S_a, while other sensitivities would be highlighted. A backcasting approach can then continue to generate a roadmap for reaching the scenario (Robinson, 1982).

We suggest that the backcasting would be done on all relevant criteria (e.g. greenhouse gases, land use, pollution, employment, gross domestic product) simultaneously. In an idealized case, all criteria would be met by a scenario. In the case that an idealized scenario cannot be found, tradeoffs between the given indicators and the

Table 4
Sensitivity analysis assisted backcasting.

Phase	Outcome	Tools
1. Systems analysis of the current economic situation	Key indicators, local development trends	EEIO datasets for two years
2. Analyze the components of the EEIO		
a) Identify components with highest sensitivities	A preliminary list of main components	Sensitivity analysis (Heijungs, 2010)
b) Try to predict the local trend by updating the identified components	The error between prediction and actual trend. Main sources of difference.	Structural decomposition (Wood and Lenzen, 2009)
c) Determine the final list of components or add general development trends for background data	A refined list of main components and background development	Time-series
3. Contact stakeholders, collect development trends	Trends for the identified main components. Overall development trend. Difference to desired sustainability criteria. Interaction between stakeholders.	Scenario building tools (Dortmans, 2005). Soft systems methodologies. (Ackoff, 1974) EEIO modeling
4. Generate sustainability scenarios	Revised development of main components, which meets the sustainability criteria or a detailed trade-off analysis between indicators and the amount of change. A list of new main components.	Backcasting (Robinson, 1982) Multiple criteria decision analysis (Keeney and Raiffa, 1993) EEIO modeling Sensitivity analysis

amount of change need to be analyzed. Multiple criteria decision analysis (Keeney and Raiffa, 1993) is particularly suitable for this task.

The proposed methodology offers a transparent approach for identifying the main components of change. It therefore can assist in prioritizing the right components for further scenario work. The use of comprehensive input–output tables ensures that all relevant aspects are captured, while the use of sensitivity analysis prevents the stakeholders from being overwhelmed by the amount of parameters included. EEIO reveals the indirect changes caused by altering individual components and requiring further adjustment until the system is in balance. This approach can reveal tradeoffs and synergies, for example between emissions reduction and avoiding trade deficit. This can highlight issues, which need a wider societal discussion in order to apply the scenario to reality (i.e. extending the system boundary of analysis (Jackson, 2003)).

The main weakness of the proposed methodology is that it does not identify the combined effect of very many small changes or major shifts at the industrial level. Incorporating these trends is the task of the analyst and the stakeholders in stages 2c and 4. Other scenario tools such as foresight (Salo and Cuhls, 2003; UNIDO, 2005) might be used together with the approach to capture these weak signals. Neither does it ensure that the identified components actually can be changed, or give indications on how to change them. Therefore the process requires a considerable amount of subjective stakeholder involvement and the use of other foresight tools. It is not meant as a replacement but an addition to those methods. Also measures which affect the economy as a whole (such as carbon pricing) are not identified by the intersectoral approach and have to be taken into account exogenously in the process.

We have illustrated the first two phases of the process in this study using a case study of Finland. The scenario work will continue with stakeholders from the Finnish industry (funded by the Ministry of Environment). However as mentioned in the introduction, the environmental crisis is global and highly interconnected. Further work based on the sensitivity analysis of multiple region input–output (MRIO) models (Tukker et al., 2009) and international stakeholder involvement is therefore urgently needed. There are no technical constraints on applying the proposed methodology on an international scale. The availability of new MRIO datasets (such as EXIOBASE (Tukker et al., 2009), WIOD (Timmer, 2012) and EORA (Lenzen et al., 2012)) opens up possibilities for identifying the main components for more comprehensive global scenarios. The use of a global focus in designing local interventions would avoid the problem of shifting the emissions through international trade (Peters et al., 2011b; Wiedmann et al., 2010).

There are however major challenges in the application and governance of such a scenario process at an international level. Also the co-ordination of the process with other ongoing policies is challenging. Experiences on limited sectors such as the freight transport sector (Helmreich and Keller, 2011) can be useful in applying the process on an international scale. Applying the proposed method first on local and national levels will provide experience on the problems and research needs of applying it with a larger set of stakeholders.

5. Conclusions

The use of a simple sensitivity analysis algorithm was demonstrated in identifying the main components from a detailed set of environmentally extended input–output tables. A very small fraction (0.3%) of the model parameters was found to explain the development in greenhouse gas emissions between 2002 and 2005. In spite of major errors in the prediction of emissions of some industries the overall development was predicted accurately. Based on the application of the sensitivity analysis to Finland, the following components should be given priority in a comprehensive carbon emission mitigation scenario: the emission intensities of energy, food, transport, waste, steel, paper and cement industries; the demand of apartments, electricity, electronics and food, government services and main export products; the intermediate demand of animal products, electricity, trade services, cement and steel. Including such a limited but comprehensive list of components in the scenario work ensures that main components are not omitted and relevant stakeholders are selected for further stages of the analysis. Further work should be focused on identifying relevant change in aggregate industries, including the emissions embodied in imports, as well as in facilitating stakeholder discussion over different industries.

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