UNIVERSITAT ROVIRA I VIRGILI

ENVIRONMENTAL TAX ON PRODUCTS AND SERVICES BASED ON THEIR CARBON FOOTPRINT: THE CASE OF SPAIN

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DEPARTMENT OF CHEMICAL ENGINEERING

UNIVERSITAT ROVIRA I VIRGILI

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CERTIFICATE

This is to certify that the thesis entitled:
"**Environmental tax on products and services based on their carbon footprint: the case of Spain**" submitted and presented by Mr. Eskinder Demisse Gemechu to the Chemical Engineering Department of Universitat Rovira i Virgili towards partial fulfillment of the requirements for the degree of Doctor of Philosophy in Chemical, Environmental and Process Engineering is carried out by him under our supervision and guidance, and that it fulfills all the requirements to be eligible for the International Doctorate label.

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TO MY BELOVED FAMILY, SPECIALLY TO MY WIFE **SELAME** AND MY LITTLE ANGEL **BLEINE**.

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Summary

From the publication of the Kyoto Protocol in 1997, the issue of climate change has been increasingly subject to policy debates. Several market instruments, such as the EU ETS, taxes on pollution, etc. have been introduced aiming to achieve the GHG emissions targets of the Kyoto Protocol, i.e. by 2020 reduce GHG emissions 20% below their level in 1990. In this line, the main aim of this thesis is to investigate the definition of an environmental tax on products based on their carbon footprint and to analyze the potential impacts of introducing the tax. These impacts are analysed both in terms of emissions reduction and change in public welfare. The analysis focuses on Spain, but it also considers trade with other countries and its effects on the carbon footprint of products consumed in Spain.

The carbon footprint of products and services can be estimated either through process-based LCA or by using economic IO approaches. This thesis applies the two methods concluding that IO approaches are more appropriate than process-LCA for economy-wide carbon footprinting exercises. However, the research here highlights that IO issues such as the aggregation of sectors and product grouping in the IO databases, the treatment of international transport, and the adjustment of prices from one to another year of analysis, could induce important errors in the estimation of product carbon footprint. To overcome these issues, this work recommends using IO-based hybrid models or EIO-LCA which conserve product specific data in an IO framework, easily applicable to economy-wide environmental policy analysis.

Based on the carbon footprints of products, an environmental tax is defined. The research highlights the significance of considering non-CO_2 GHG emissions in environmental tax policies. Despite the fact that CO_2 is the most important GHG and it is in the spotlight of most climate change mitigation actions, there are also other gases with higher global warming potential which are usually not considered. Sectors such as Mining of coal and lignite; extraction of peat, Agriculture, livestock and hunting and the Food sectors are among the sectors most responsible for the emission of a considerable amount of non-CO_2 GHGs in the Spanish economy. An environmental tax applied to these sectors would vary considerably if GHG emissions other than CO_2 were considered.

The potential impacts of the taxation are assessed by using a Leontief price model, which allows the introduction of the tax on selected sector to induce changes in the production price. The effects of the tax both on the economy, on the environment and on the society are estimated. The results show that there is a clear trade-off between the environment, economy and society. The environmental and economic goals cannot be met at the same time with the environmental taxation unless there is a way in which the public revenues could be used to compensate those who are negatively affected by the tax.

The thesis also addresses the issue of international trade and its implications on climate change mitigation. Such issues have been overlooked by the Kyoto Protocol, i.e. emissions due to international trades were not explicitly treated, resulting in carbon leakages. The thesis analyzes the emissions embodied in the Spanish imports and the responsibilities of the Spanish economy by using a fully integrated MRIO model. The empirical results reveal that Spain is a net importer of CO_2 emissions and China is the most polluting exporter to Spain. These results evidence a clear carbon leakage, as energy and emission intensive products are shifted to be produced in developing countries which are not signatories of international climate change policies. This suggests the need to go beyond the Kyoto Protocol and apply a border adjustment CO_2 tax on products which are imported from countries which are not covered by such Protocols.

Resumen

Desde la publicación del Protocolo de Kyoto en 1997, el tema del cambio climático ha estado cada vez más sujeto a debates políticos.

Para alcanzar los objetivos de emisiones de gases de efecto invernadero (GEI) del Protocolo de Kioto, se han introducido varios instrumentos de mercado como el comercio de derechos de emisión de la Unión Europea (o EU ETS), los impuestos sobre la contaminación, etc. Más concretamente, se pretende para el año 2020 una reducción de los GEI de un 20% por debajo de su nivel en 1990. En esta línea, el objetivo principal de esta tesis es investigar la definición de un impuesto ambiental a los productos en función de su huella de carbono y analizar los impactos potenciales de la introducción de este impuesto. Estos impactos se analizan tanto en términos de reducción de las emisiones como en el cambio en el bienestar público. El análisis se centra en España, pero también se considera el comercio con otros países y sus efectos sobre la huella de carbono de los productos que se consumen en España.

La huella de carbono de los productos y servicios puede calcularse a través de técnicas de análisis del ciclo de vida (ACV) basado en procesos o utilizando métodos en base a IO económico. Esta tesis utiliza los dos métodos, concluyendo que los enfoques IO son más apropiados que los procesos ACV para aplicaciones de cálculo de huella de carbono en sectores económicos en general. Sin embargo, la investigación pone de relieve que cuestiones relacionadas con IO, como la agregación de sectores y la agrupación de productos en las bases de datos IO, el tratamiento del transporte internacional y el ajuste de los precios de un año para otro de análisis, podrían inducir importantes

errores en la estimación de la huella de carbono del producto. Para superar estos problemas, este trabajo recomienda el uso de modelos híbridos basados en IO o EIO-LCA que conservan especificaciones de productos de datos en un marco IO, fácilmente aplicable a análisis de política ambiental en sectores económicos.

Se ha definido un impuesto medioambiental basado en la huella de carbono de los productos. Este estudio pone de relieve la importancia de considerar las emisiones de GEI distintas del CO_2 en las políticas fiscales ambientales. A pesar del hecho de que el CO_2 es el GEI más importante y es el centro de atención de la mayoría de las acciones de mitigación del cambio climático, también hay otros gases con alto potencial de calentamiento global, que generalmente no se consideran. Sectores como la minería del carbón y el lignito, extracción de turba, agricultura, ganadería, caza y sectores de alimentos son uno de los sectores más responsables de la emisión de una cantidad considerable de GEI distintos del CO_2 en la economía española. Un impuesto ambiental aplicado a estos sectores variaría considerablemente si fuesen también consideradas las emisiones de GEI a parte del CO_2.

Los impactos potenciales de la tributación se han evaluado mediante el uso de un modelo de precios de Leontief, que permite la introducción del impuesto sobre el sector seleccionado para inducir cambios en el precio de producción. Se han estimado los efectos de los impuestos tanto sobre la economía, el medio ambiente y en la sociedad. Los resultados muestran que hay un claro equilibrio entre el medio ambiente, la economía y la sociedad. Los objetivos ambientales y económicos no se pueden cumplir al mismo tiempo con la fiscalidad ambiental, a menos que hubiera alguna manera de que los ingresos públicos pudieran utilizarse para compensar a aquellos que se ven afectados negativamente por el impuesto.

La tesis aborda también la cuestión del comercio internacional y sus implicaciones en la mitigación del cambio climático. Estas cuestiones han sido pasadas por alto por el Protocolo de Kyoto, i.e., devido a que las emisiones de las operaciones internacionales no fueron tratadas de forma explícita, esto ha dado lugar a fugas de carbono. La tesis analiza las emisiones incorporadas en las importaciones españolas y las responsabilidades de la economía española mediante el uso de un modelo completamente integrado, MRIO. Los resultados empíricos revelan que España es un país importador neto de las emisiones de CO_2 y China es el exportador más contaminante para España. Estos resultados muestran una clara fuga de carbono, por el hecho de que la manufactura de productos energéticos y de emisiones intensivas se desplaza a países en desarrollo que no son signatarios de las políticas internacionales de cambio climático. Esto sugiere la necesidad de ir más allá del Protocolo de Kyoto y aplicar un ajuste sobre el CO_2 en los productos importados de países que no estén cubiertos por dichos Protocolos.

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Nomenclature

Variables

α_j	The share of final goods from sector j 95
$\dot{\boldsymbol{n}}$	Emissions released per functional unit - Process-based LCA ... 19
$\dot{\boldsymbol{A}}$	Technology matrix - Process-based LCA 19
$\dot{\boldsymbol{e}}$	A vector of process emission factors - Process-based LCA 19
$\dot{\boldsymbol{y}}$	A vector of functional unit - Process-based LCA 19
$\boldsymbol{\chi}$	A column vector of total output - MRIO 116
$\boldsymbol{\chi_1}$	A vector of sectoral total output of region i - MRIO 116
$\ddot{\boldsymbol{n}}$	Emissions released per functional unit - Integrated hybrid 30
$\ddot{\boldsymbol{y}}$	A vector of functional unit - Tiered 27
ΔW	Change in private real income 95
$\boldsymbol{\delta}$	Block matrix of technology matrix - MRIO 116
$\boldsymbol{\delta_{ii}}$	Technology matrix (Domestic region i - MRIO 116
$\boldsymbol{\delta_{ij}}$	Technology matrix (Import from region i to j - MRIO) 116
$\boldsymbol{\eta}$	The total environmental emissions associated with the production of $\boldsymbol{\gamma}$ output from all n regions - MRIO 117
$\boldsymbol{\eta_1}$	Emissions released in region 1 to produce final demand - MRIO 118
$\boldsymbol{\eta_i}$	Emissions associated with imported products to region 1 - MRIO 118
$\boldsymbol{\gamma}$	A block vector of final demands - MRIO 116
$\boldsymbol{\gamma_1^{ex}}$	A vector of export final demand from region 1 - MRIO 116

$\boldsymbol{\gamma_{11}}$	A vector of final demand from region 1 to region 1 - MRIO . . 116
$\boldsymbol{\gamma_{ij}}$	A vector of imported final demand from region i to region j - MRIO 116
$\boldsymbol{\gamma_{import}}$	Total imported final demand to Spain - MRIO 120
$\{\boldsymbol{m_{ij}}\}_c$	Imports of intermediate good c from region i to Spain(j) - MRIO 120
∂	Averaged margins and taxes less subsidies on products 65
τ_j	Ad-valorem tax on production in net terms 94
$\tilde{\boldsymbol{n}}$	Emissions released per functional unit - Tiered 27
$\boldsymbol{\varepsilon}$	Environmental tax 75
$\boldsymbol{\varphi}$	Emissions cost 75
$\boldsymbol{\xi}$	A block vector of emission sectoral factors - MRIO 117
$\boldsymbol{\xi_i}$	A vector of sectoral emission factors in region i - MRIO 117
$\boldsymbol{\zeta}$	Unbalanced block transaction matrix - MRIO 123
$\boldsymbol{\zeta^{**}}$	Balanced block transaction matrix - MRIO 123
$\boldsymbol{\zeta^{*}}$	Unknown target block transaction matrix - MRIO 123
$\boldsymbol{A}$	A matrix of technological coefficient - EIO 21
$\boldsymbol{A^*}$	New technology matrix - IO-based Hybrid 28
$\boldsymbol{A^{\varepsilon}}$	Technology requirement matrix - price model 94
a_{ij}	Technological requirements by industry j from industry i 21
$\boldsymbol{B}$	Commodity-by-industry direct requirement matrix 23
b_{ij}	Input requirement of commodity i associated with output of industry j 23
$\boldsymbol{C^d}$	Downstream cut-off matrix 30
$\boldsymbol{C^u}$	The upstream cut-off matrix 27
C_j	Consumption of goods of sector j by households 95
CPI_{2007}	Consumer price index in 2007 64
CPI_{2010}	Consumer price index in 2010 64
$\boldsymbol{D}$	A matrix of commodity output proportion 23
d_{ij}	Share of total commodity j output produced by industry i 23

$\boldsymbol{e}$	A column vector of emission factor - EIO	22
$\boldsymbol{e^*}$	A column vector of emission factor - IO-based Hybrid	28
E_j^{ε}	New sectoral emissions after the environmental tax	96
e_j	Emission factor of sector j	96
$\boldsymbol{g}$	A vector of total commodity final use	23
$\boldsymbol{I}$	Identity matrix of required size	22
$\boldsymbol{i}$	A column vector of ones	21
$\boldsymbol{I^*}$	Identity matrix of required size	28
k_j	Capital coefficient	94
l_j	Labor coefficient	94
m	Number of sectors in each region - MRIO	116
$\boldsymbol{M_{ij}}$	The share of import of each product - MRIO	120
m_j	Import coefficient	94
$\boldsymbol{n}$	Emissions released per final demand - EIO	22
n	Number of regions - MRIO	116
$\boldsymbol{n^*}$	Emissions released per functional unit - IO-based Hybrid	28
$\boldsymbol{n^1}$	Emissions released per final demand - Commodity-by-industry	23
$\boldsymbol{p^{\varepsilon}}$	A vector of price after the environmental tax - price model	94
p_j^m	Price of imports	94
P_0	Tax exclusive price	77
P_1	Price before the environmental tax	77
P_{2007}	Price in 2007	64
P_{2010}	Price in 2010	64
P_2	A vector of price after the environmental tax	77
P_{basic}	Basic price	65
p_c	Consumption price	95
p_j	Unitary price of output in each sector j	94
$P_{purchaser}$	Purchaser price	65
$\boldsymbol{q}$	Product output vector	23

R	Total public revenues	96
r	Capital price	94
s_j	Tax rate of social security paid by sector j	94
t	Ad-valorem tax	77
t^*	Effective tax rate	77
t_j^m	Ad-valorem rate of the imports in sector j	94
$\boldsymbol{U}$	Use matrix	23
u_{ij}	Amount of purchase of commodity i by industry j	23
$\boldsymbol{V}$	Supply matrix	23
$\boldsymbol{v}$	A vector of value added per unit output	94
v_{ij}	Amount in € of product j produced by industry i	23
w	Labor price (wage)	94
$\boldsymbol{x}$	A vector of sectoral total output	21
x_j^ε	Output of sector j after the tax	96
x_j	Output of sector j before the tax	96
$\boldsymbol{y}$	A vector of final demand - EIO	21
$\boldsymbol{y}^\diamond$	A vector of inputs required to produce tissue paper - EIO-LCA	23
$y_i^\diamond$	Input from sector i to produce tissue paper - EIO-LCA	23
$\boldsymbol{Z}$	A transaction matrix - EIO	21

Other Symbols

CH_4	Methane	75
HFCs	Hydrofluorocarbon	75
N_2O	Nitrous oxide	75
PFCs	Perflorcarbons	75
SO_2	Sulfur dioxide	90

Acronyms

10YFP	Ten-year Framework of Programs	3
ACV	análisis del ciclo de vida	xiii
BE	Belgium	121

NOMENCLATURE

BR	Brazil	121
CBA	Cost-Benefit Analysis	13
CGE	Computable General Equilibrium	86
CHP	Combined heat and power	56
CN	China	121
CPI	Consumer Price Index	64
DE	Germany	121
ECF	Elemental chlorine free	45
EF	Ecological Footprint	13
EIA	Environmental Impact Assessment	13
EIO	Environmental Input-Output	6
ERA	Environmental Risk Assessment	13
ERM	Environmental Resources Management	39
EU ETS	EU Emissions Trading System	4
EU	European Union	4
FR	France	121
GB	Great Britain	121
GEI	Gases de efecto invernadero	xiii
GHG	Greenhouse gas	3
GPP	Green Public Procurement	4
INE	National Statistics Institute	63
IOLCA	Input-Output based LCA	6
ISIC	International Standard Industrial Classification	120
ISO	International Organization for Standardization	13
IT	Italy	121
JP	Japan	121
LCA	Life Cycle Assessment	6
MFA	Material Flow Analysis	13
MRI	Midwest Research Institute	13

NOMENCLATURE

MRIO	Multi-regional input-output	6
NL	The Netherlands	121
PT	Portugal	121
REPA	Resource and environmental profile analysis	13
RWP	Recycled waste paper	8
SCP/SIP	Sustainable Consumption and Production and on Sustainable Industrial Policy	4
SCP	Sustainable consumption and production	3
SEA	Strategic Environmental Assessment	13
SETAC	Society of Environmental Toxicology and Chemistry	14
SIOT	Symmetric input-output table	61
SRIO	Single-regional input-output	114
SUT	Supply and use table	22
VP	Virgin pulp	8
WIOD	World Input-Output Database	121
WSSD	World Summit on Sustainable Development	3

Chapter 1

INTRODUCTION

1.1 General Introduction

The ever accelerating growth of global economy, which is not consistent with environmental and social needs, has resulted in irrational resource exploitation and environmental deterioration. Decoupling the environmental impacts from economic growth yet remains the unresolved challenge faced by both economists and environmentalists. Global warming is one of the univocal examples proven to be caused by human activities (IPCC, 2007). Unsustainable consumption and production patterns[1] are considered as one of the major driving forces of anthropogenic greenhouse gas (GHG) emissions. Therefore, attention should be paid to both producers' and consumers' behaviors and how they can be changed in order to reduce their impacts and to build a sustainable environment. The issues of consumption and production as driving forces of climate change have been put on international agenda since the Rio Earth Summit in 1992 (UN, 1992). The Rio Earth Summit on Environment and Development identified unsustainable patterns of consumption and production as major causes of the continued deterioration of the global environment, particularly in developed countries. It declared that sustainable development and high quality of life could be achieved when States should reduce and eliminate unsustainable patterns of production and consumption and promote appropriate demographic polices (UN, 1992). Ten years later, the world's political leaders gathered in Johannesburg in 2002 at the UN's World Summit on Sustainable Development (WSSD). The commitments in Rio Earth Summit were renewed in Johannesburg by deciding that changing unsustainable patterns of consumption and production should be one of the three overarching objectives of sustainable development (UN, 2002). A ten-year Framework of Programs (10YFP) was called for in support of national and regional initiatives to accelerate the shift towards SCP. The Marrakech Process was established with the aim of supporting the development of 10YFP called for in

[1] The most prominent definition of Sustainable Consumption and Production (SCP) is the one proposed at the 1994 Oslo Roundtable on Sustainable Production and Consumption. It defines as "use of goods and services that respond to basic needs and bring a better quality of life, while minimising the use of natural resources, toxic materials and emissions of waste and pollutants over the life cycle, so as not to jeopardise the needs of future generations". Source: Norwegian Ministry of the Environment (1994) Oslo Roundtable on Sustainable Production and Consumption (http://www.iisd.ca/consume/oslo004.html)

Johannesburg and the implementation of SCP. Following the Marrakech Process, a number of regions have implemented SCP frameworks. EU has been taking important action towards achieving the SCP main goals, i.e., to ensure sustainable economic development while reducing environmental degradation through increasing the efficiency of resource use. The Action Plan on Sustainable Consumption and Production and on Sustainable Industrial Policy (SCP/SIP), which was launched in July 2008 could be a good example (EC, 2008b). The SCP/SIP Action Plan comprises different directives and policy instruments so as to advance SCP by further improving the environmental performance of products and by promoting demand for sustainable goods. These includes: the Ecodesign Directive which sets EU-wide consistent rules on ecodesign requirements for improving the environmental profile of energy related products (EC, 2009); the Energy Labelling Directive that establishes a framework for harmonized product labelling so as to provide information for consumers aiming at shifting the market from energy-intensive products toward more efficient products (EC, 2010); the Green Public Procurement (GPP), which provides guidance on how to reduce the environmental impacts caused by public sector consumption and stimulate innovation in environmental technologies, products and services; and market-based economic instruments.

Market-based economic instruments can play an important role in promoting SCP (ETC/RWM and EEA, 2007). A wide range of economic tools have been used in EU in relation to SCP, which includes fees and charges, environmental taxes and subsidies, tradable permits, deposit-refund systems amongst others (Watson et al., 2009). Being price is one of the most determinants of consumer choices; economic instruments can provide incentives for the consumption of more sustainable products.

The EU Emissions Trading System (EU ETS) is one of the best examples of economic instruments that are applied to the implementation of SCP policy. It is the largest international scheme for trading GHG emissions allowance to combat climate change (Ellerman and Buchner, 2007). The EU ETS was launched in January 2005 and now it is in its third phase, which will run from 2013 to 2020 and supposed to ensure a 21% lower emissions compared with its first implementation in 2005. The working principle of EU ETS is 'cap-and-trade' in which a limit is set on the total GHG emissions that could be released by the installations[1] under the scheme. The permits which can be traded are then allocated to all participants for free. When the emissions exceed the allowance then an installation should buy allowances from others and similarly if the emissions are lower than the allocated allowance, then the installation can sell to others.

Alternative to tradable permit is environmental tax. According to the economic theory, environmental taxes are often considered as a Pigovian tax (Helm, 2005). The concept behind tax introduction is linked with externality. In economics, externality is referred to as "situations when the effect of production or consumption of goods and services imposes costs or benefits on others which are not reflected in the prices charged for the goods and services being provided" (Khemani and Shapiro, 1993). When there is negative impact to the environment such as pollution, the social cost will be higher than the private cost (producer), know as market failures. Environmental tax is used to correct market failures by internalizing negative externalities and letting the market to play its role in changing consumers' purchasing patterns. Because the costs imposed on the society are internalized by introducing environmental tax, both producers and consumers will be aware of the full cost of pollution.

From both conceptual and theoretical point of views, tradable permit and environmental taxes share similarity as policy instruments to ensure climate change mitigation. However, from practical point of view, environmental taxes seem more advantageous than the tradable permit (Norregaard and Reppelin-Hill, 2000). One of the advantages of environmental taxes is price fixing. The prices of pollution; for example,(GHG or CO_2 emissions) are fixed, which is not the case in tradable permit. Being it is quantity oriented system; tradable permit fixes the

[1]The EU ETS scheme includes a wide range of factories or power plants. For CO_2 emissions power and heat generation; energy-intensive industry sectors including oil refineries, steel works and production of iron, aluminum, metals, cement, lime, glass, ceramics, pulp, paper, cardboard, acids and bulk organic chemicals; and commercial aviation are included. GHG emissions such as N_2O from production of nitric, adipic, glyoxal and glyoxlic acids and PFCs from aluminum production are also covered by the scheme.

total amount of targeted emissions reduction and allows the price of the emissions to fluctuate as a result of market forces. The volatility of the permit price is one of the challenges the current EU ETS is facing. Due to the prolonged economic crisis in EU, which began in 2008, the EU ETS suffers from the growing allowance surplus. From 2008 to 2012, the second phase of EU ETS, the price of EU allowance has showed a remarkable sink, from 20 and 30€ per ton CO_2 to 6.5€ per ton CO_2 (Sartor, 2012). This price volatility of tradable permit system could have a disruptive effect as it does not encourage carbon-saving investments (investment in alternative fuels and energy efficient technologies or R&D) that have high start-up-costs. The other advantage of environmental tax is its revenue generation potential compared with the free allocation case in tradable permit system. The revenue from the tax could be used to adjust the distortional effects of the taxation on the society. Simplicity and transparency are also other advantages of environmental taxation over the traditional 'cap-and-trade' system. The later, relatively requires administrative structure while the former could be treated in an already established and developed taxation systems. More importantly the 'cap-and-trade' system is based on the direct emissions principle in which an entire responsibility is given to the producers. However, consumers should also be responsible for the emissions associated with their consumption. This perspective can be reflected through the application of environmental tax on products and services based on their emission intensities. In this line, this thesis, therefore, aims at developing a model to calculate environmental tax of products and services based on their carbon footprint and assess the potential impacts of taxation. Traditional Life Cycle Assessment (LCA), Environmental Input-Output (EIO) and hybrid-IOLCA models will be used as methodological tools to assess carbon footprint of products and services. The carbon footprint results then will be translated into environmental taxes. Both the social and environmental effects of the defined taxes are assessed through the use of Leontief price model. Finally the implication of international trade in such environmental policy application are addressed by using Multi-regional input-output (MRIO) models.

1.2 Hypothesis

The best method to estimate the carbon footprint of a product or service is LCA. One of the hypotheses of this work was that it is possible to design an environmental tax proportional to the carbon footprint of products and services at national level. Other hypothesis was that the introduction of the tax would lead to the reduction of Spanish national GHG emissions. And finally, that it is necessary to apply an environmental tax to imported products, to make local businesses more competitive on international markets.

1.3 Objectives

1.3.1 General Objective

The general objective of the thesis is to define and calculate environmental tax associated with the life cycle GHG emissions of products and services in the Spanish economy and estimate the impacts of its introduction in Spain.

1.3.2 Specific Objectives

The general objective was met through accomplishing the following specific objectives:

- identification of the most appropriated method to assess GHG emissions of products and services in the Spanish economy,
- design of an environmental tax based on estimated carbon footprints,
- assess the economic and social impacts of environmental tax implementation in the Spanish economy, and
- assess effects of environmental tax application on imported and exported goods and services.

1.4 Outline of The Thesis

The thesis is organized into nine chapters including the general introduction presented in the first chapter. The outline of each chapter is presented as follow:

Chapter 2 provides a general review and description of methods used for products' and services' carbon footprint estimation. Increasing concerns on climate change and urgent need of response to reduce its potential damage have motivated for the development and improvement of already existing methods for assessing the life cycle GHG emissions associated with human activities. This chapter revises the developmental evolution of common methods that could address carbon footprint of products and services, namely: process-based LCA, EIO and Hybrid-IOLCA approaches. The methodological frameworks, pros and cons of one approach over the other, and their applicability are addressed.

Chapter 3 illustrates the environmental implications of producing tissue paper from virgin pulp (VP) or recycled waste paper (RWP) by comparing the detailed life cycle inventory of both production lines. It also aims at informing decision-makers at both company and national levels as to the main causes of emissions and to suggest the actions required to reduce pollution. The chapter addresses the following points: "How the choice of raw material from VP and RWP influences total GHG emissions of tissue paper production," "What are the main drivers behind these emissions," and "How do the direct materials, energy requirements and transportation contribute to the generation of emissions?". Besides addressing the above listed issues, results from the life cycle inventory will be used as a base in comparing with other approaches such as EIO and hybird-IOLCA in the next chapters.

Chapter 4 uses the detailed LCA result from Chapter 3 and attempts to investigate the relevancy of other methods in estimating GHG emissions associated with a specific product, tissue paper production from VP process. It outlines the discrepancy of results among different methods (EIO and hybrid-IOLCA) that could arise from the nature of modeling and other possible factors.

Chapter 5 outlines the definition of an environmental tax on products and services based on their carbon footprint. It examines the relevance of process-based LCA and EIO as methodological tools for identifying the emission intensities on which the tax is based. The price effects of the tax and the policy implications of considering non-CO_2 GHG are also analyzed using quantity-oriented price formulation.

Chapter 6 is dedicated to illustrating the potential effects of environmental tax defined based on products' or services' life cycle emissions. EIO model is used to identify CO_2 emission intensities of products and services and, accordingly, the environmental tax proportional to these intensities are defined. The short-term price effects of the tax are analyzed using an input-output Leontief price model. The effect of tax introduction on consumption prices and its influence on consumers' welfare are determined. Furthermore, the potential of environmental taxation to induce CO_2 emissions reduction is also illustrated.

Chapter 7 discusses the policy implication of international trade with regard to current international climate change policies. It describes the methodological framework of MRIO model to capture emissions embodied in imported goods. A 13 region MRIO model is constructed for Spain with the aim of analysing the most important trading partners of Spain in terms of emissions flows. The key domestic sectors that are behind the embodied emissions are identified.

Chapter 8 presents the general conclusions drawn from each case study. Finally, the overall discussion on the policy significance of the thesis, main considerations and limitations of the models used, and the main possible sources of uncertainties are discussed in Chapter 9.

Chapter 2

METHODS USED IN PRODUCTS' OR SERVICES' CARBON FOOTPRIT: GENERAL REVIEW

2.1 Introduction

The issue of climate change has become one of the priorities of public and business leaders as there are unequivocal evidences and scientific consensus on its potential damages (IPCC, 2007). The ever increasing awareness of climate change impacts and the urgent need for responses from governments, global companies and individuals to reduce its potential effect have motivated and contributed to the development and improvement of already existing methods for assessing the environmental impacts of different human activities. This includes: LCA, Environmental Impact Assessment (EIA), Environmental Risk Assessment (ERA), Cost-Benefit Analysis (CBA), Material Flow Analysis (MFA), Ecological Footprint (EF) and Strategic Environmental Assessment (SEA) (ISO, 2006a; Sonnemann et al., 2004; Finnveden and Moberg, 2005; Finnveden et al., 2009; Ness et al., 2007). In this chapter we describe the methodological issue of LCA.

According to ISO 14040 the word life cycle is defined as "consecutive and interlinked stages of a product system, from raw material acquisition or generation from natural resources to final disposal". LCA stands for "compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle"(ISO, 2006a).

The introduction of LCA concepts in determining environmental impact of products dates back to the early 1960s (Vigon et al., 1994), when the scarcity of energy and resource materials drew the attention of scientists to find a way to assess energy use and to project future resource supply and use. The late 1960s and early 1970s were the period when environmental issues such as resource and energy efficiency, pollution control, and solid waste became the foremost public concern. Impelled by the massive oil crisis in early 1970's, a number of energy analyses were performed for different industrial systems. The study by Midwest Research Institute (MRI) in 1969, which dealt with the comparison of resource and energy requirements and waste release from different beverage containers of the Coca-Cola Company was among the fist LCA studies. Later on the methodology was refined by MRI for the US Environmental Protection Agency to prepare the resource and environmental profile analysis (REPA) of nine beverage container options (Hunt et al., 1974). From late 1070s to early 1980s the environmental issues were shifted to hazardous and solid waste management. Interest in LCA somehow detracted until the late 1980s when it again emerged as a tool for environmental problem analysis and its applications gained attention by consultants and researchers across the globe. Early 1990s was the time when LCA results were widely used to promote products. Concerns over the misleading use of LCA also raised and led to the development of standards and technical aspects of product LCA. The 1990s was the period when LCA had shown a remarkable growth in which a number of workshops were organized. A forum by the World Wildlife Fund and the Conservation Foundation on the policy issue of product life cycle (WWF, 1990), the workshop by the Society of Environmental Toxicology and Chemistry (SETAC) on the development of technical framework for LCA (Fava, 1991; Fava et al., 1993; Udo de Haes, 1996) and workshop held in Leuven on the practical aspects of performing LCA (de-Smet, 1990) are a few examples. LCA guidelines and handbooks were also produced (Heijungs et al., 1992; Vigon et al., 1994; Consoli et al., 1993; Lindfors et al., 1995). The involvements of SETAC and the ISO have played a substantial role for the further development of LCA. The LCA framework has shown continuous improvements and harmonization after the introduction the SETAC's guideline (code of practice) (Consoli et al., 1993) and ISO 14000 series.

The current stage of LCA is described as a stage of elaboration by Guinée et al. (2011). The demand on the use of LCA methods as a policy tool by decision makers at different level (government or firm) or individuals is increasing. However, the lack of well standardized methods in specific issues such as system boundaries, allocation and other issues have resulted in the variation of LCA approaches. The emerging applications of EIO, hybrid-IOLCA are good examples when it comes to carbon footprint or GHG emissions estimation of products or services. Although used different approaches, all methods are based on the main principle in which the traditional LCA is founded.

Detailed reviews on the recent development of LCA can be found elsewhere (Rebitzer et al., 2004; Pennington et al., 2004; Finnveden et al., 2009; Guinée et al., 2011). In this chapter we discuss the description and methodological foundation of the LCA approaches applied in the thesis. It is organized as follow: Section 2.2 describes the basics of process-based LCA. Section 2.3 and 2.4 illustrate EIO and hybrid-IOLCA approaches, respectively. Section 2.5 discusses the main advantages and disadvantages of one approach over the other.

2.2 Process-Based LCA

Process-based LCA is the most commonly used tool for evaluating the environmental impacts of specific products (Cederberg and Mattsson, 2000; Guinée et al., 2002). In recent years, it has emerged as a leading policy instrument in business decision-making processes, in research and development (R&D), in the environmental improvement of products, and in product labelling and environmental declarations, to name but a few. With the application of Process-based LCA, it is possible to analyze both the material and energy requirements and the associated emissions at all stages of a particular product or service from the extraction of materials, through to the production processes, the use of the product and its final disposal either by reuse, recycling, or in a waste management stream. It allows the most polluting phases in a product's life cycle to be identified and also provides both an overall environmental evaluation and a detailed analysis of each stage, thus preventing changes from being made at a given stage with no regards to the overall life cycle impact. Such a complete LCA of a product is often referred to as a cradle-to-grave analysis. However, LCA can also be applied to selected boundaries of product systems; for example, from materials extraction to production processes, a method known as cradle-to-gate analysis.

According to the ISO 14040 (ISO, 2006a) and ISO 14044 (ISO, 2006b), process-based LCA process can be described in four phases: goal and scope definition, inventory analysis, impact assessment and interpretation as described in Figure 2.1. In what follow, the definition and brief summary of each phase of the framework are presented.

Goal and scope definition: Being the first stage of the analysis, it is mainly concerned with the examination of the main system and boundary, the choice of

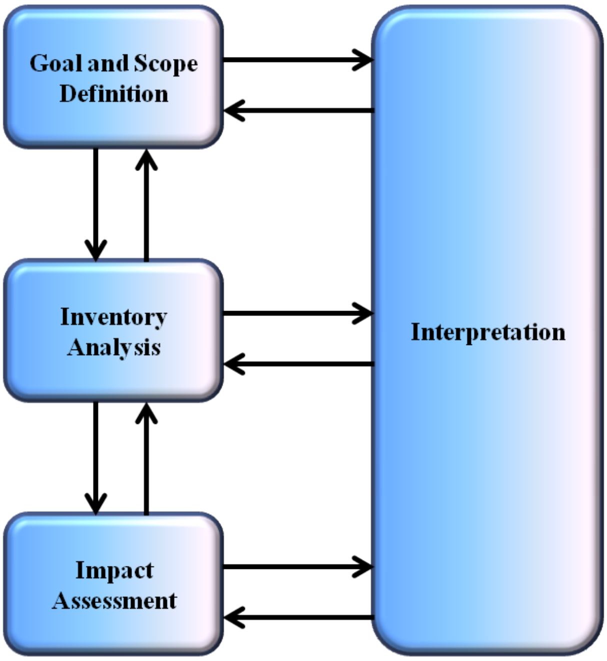

Figure 2.1: System boundary in LCA

the functional unit of the system and setting the main procedure to be followed. The goal of the study should answer questions such as: What are the main intention of doing the analysis and the reasons behind? To whom are the studies going to be addressed? How the results are going to be used?

The scope of the study should be clearly set. This includes: the definition of each product or process and its boundary, the identification of the function and the choice of functional unit of the specific product, or system, the setting of allocation rules, the methodology going to be applied, the requirement of data and the unavoidable assumptions and restrictions of the analysis that should be taken into account during interpenetration of the results. The functional unit is the most important element in an LCA study. Therefore, it has to be properly defined. All the inputs and outputs of the system or process are referred to the functional unit. The definition of the functional unit is important when environmental profiles of different products are to be compared.

The system boundary reveals which processes or subsystems are included in the study. The setting of the system boundary should be in line with the defined goal of the study. Systems or processes omission is possible when it is thought that their relative contributions are not significant and will not affect the overall outcome of the study. Each system under analysis is usually described in process flow diagram (see Figure 2.2) for easy understanding of the entire system and the inter-relationships among products or subsystems.

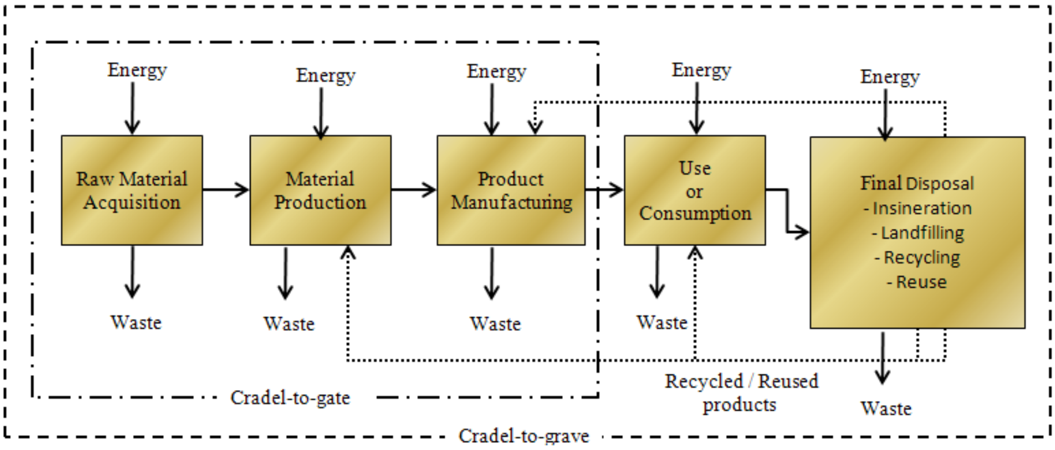

Figure 2.2: The general methodological framework for LCA according to ISO 14040

Inventory Analysis: This phase of the LCA process is the stage in which every data associated with inputs and outputs to the system are collected, recorded, calculated and validated. They are used as input to the next phase of the LCA study, impact assessment.

Data collection is the most difficult task in LCA study as it needs more time and labor to get all the necessary information that are related with the life cycle of the system under study. Both qualitative and quantitative data of each process or subsystem which is included in the defined boundary should be collected. The source of each data should be referenced with detailed information about the collection process, time of collection and about the reliability and quality of the data. This helps for the better interpretation of the final results.

All the relevant data should be related to the amount that are required or released to provide the defined functional unit. Once the collected data are normalized to the functional unit, the next step is the calculation of the inventory results using the methodology defined in the goal and scope definition stage. An important issue in data calculation is the allocation procedure. As defined in ISO 14044 (ISO, 2006b) allocation is the process of splitting input or output flows of a multi-functional process between the product system under study and one or more other product systems. A good example will be the study of environmental burden from milk production. The production of milk is a multi-function process in which it comprises the production of meat and other co-products. The allocation procedure will set to what extent environmental burden of such process should be assigned to the milk products, assuming that the defined functional unit of the system is production of 1kg of milk. Depending on the type of product or system analyzed, different allocation procedures could be applied such as: allocation based on the physical relationship between the product and its co-products, economic allocation, allocation through system expansion and so on.

Impact Assessment: In this stage all the environmental burden that are quantified in the inventory analysis are characterized with the aim of understanding the importance of each inputs and outputs to specific impacts. Important stages in impact assessment are selection and identification of the impact categories, classification (the assessment of the inventory analysis to the selected impact categories), and characterization (it comprises the quantification or analysis and aggregation of the inventory result in a given impact categories through multiplying by the characterization factor to have a common unit for each impact category.). The results from different impact categories reflect the environmental profile of the product or the system under study.

Depending on the goal and scope of the study, there are also optional elements which could be used. These are normalization[1], grouping[2], and weighing[3].

[1]Normalization process provides a reference situation on the environmental burden of each defined impact category and hence the absolute value from each impact category is translated to the reference situation.

[2]Grouping is the process of sorting and ranking of the selected impact categories.

[3]Weighing is the stage in which all the impact categories are converted or aggregated into a single indicator

Interpretation: It is the last stage of the LCA study. Interpretation comprises, the identification of the most relevant processes or subsystems based on results from the inventory analysis and impact assessment; evaluation of the completeness, consistency and sensitivity of the information that have been considered in different stages of the LCA processes; and conclusions, recommendation and limitations of study that could be considered in the application of the results.

Process-based LCA can be represented in the form of a matrix which covers an infinite order of process interactions within selected boundaries (Heijungs, 1994). The matrix description allows the approach to be connected to EIO models. The emissions $\dot{\boldsymbol{n}}$ released by producing a given functional unit $\dot{\boldsymbol{y}}$ are calculated as:

$$\dot{\boldsymbol{n}} = \dot{\boldsymbol{e}}' \dot{\boldsymbol{A}}^{-1} \dot{\boldsymbol{y}} \tag{2.1}$$

where $\dot{\boldsymbol{e}}$ is a vector of emissions per unit process and $\dot{\boldsymbol{A}}$ is the technology matrix, in which each element represents the inflow or outflow of energy or material from one process to another per functional unit.

The process-based LCA procedures discussed in this section are followed to assess the GHG emissions associated with the life cycle manufacturing process of tissue paper production both from VP and RWP. Since process-based LCA is the most common approach to assess the environmental performances of specific products, it is considered as a base to compare with other LCA approaches. All the practical aspects of the process-based LCA such as: the definition of the functional unit, boundary setting and system consideration, data collection and inventory analysis and impact assessment are well described in Chapter 3.

2.3 EIO-LCA

Other widely used carbon footprinting method is EIO approach. EIO is a top-down approach used to account for resources consumption and emissions release based on economic input-output (IO) tables (Miller and Blair, 2009). The approach uses generic data at national level to evaluate the emission intensity of all industries in an economy to produce the output necessary to satisfy a given final demand. The EIO model is derived from the Leontief's IO table, which was initially developed by Wassily Leontief in the 1930s, for which he received a Nobel Prize in Economics. The scheme of table is illustrated in Table 2.1. It is symmetric in nature as it is based on a one-to-one industry and product relationship, i.e. each industry is assumed to produce only one product and each product is produced by only one industry. Each row is read as the use of products from the corresponding industry either as intermediate input by the production system or as a final demand. The Final demand side comprises demand by household, government and export; often they are classified as domestic and foreign demands. The sum across the row refers to the total sectoral output. Each column in the IO table contains information about the purchase of products by the corresponding industry. It comprises purchase from the production system, from payment sectors (for labor services, tax, capital interest, land rental payment, etc, which is referred as value added), and finally imports. The sum along the column is the total input which is equal to the total output.

Table 2.1: The scheme of IO framework

	Industry	Final Use		Total
Industry	Intermediate Use	Domestic Use	Export	Total Use
	Import Intermediate	Import Final Demand		
	Value Added			
	Total Supply			

The formulation of IO model is based on two sets of data: data on transactions among sectors and data on the purchase of products and services by final consumers. Using the two data sets, the output from a given sector can be distributed to all sectors in the economy and final consumers. The IO model relates the total output from a given sector with its intermediate supplies and final demand. The simple relationship could be mathematically represented in matrix notation as:

$$\boldsymbol{x} = \boldsymbol{Z}\boldsymbol{i} + \boldsymbol{y} \tag{2.2}$$

where $\boldsymbol{x}$ is the vector of sectoral total output. Each element x_i represents the total output of sector i. $\boldsymbol{Z}$ is transaction matrix in which each element z_{ij} stands for the amount of sale by sector i to sector j. $\boldsymbol{y}$ is a vector of final demand, which constitutes demand by households, government and foreign trades. $\boldsymbol{i}$ is a column vector of ones.

Equation 2.2 can also be expressed to reflect the dependence of intermediate flows on the total output of each sector using the following equation.

$$\boldsymbol{x} = \boldsymbol{A}\boldsymbol{x} + \boldsymbol{y} \tag{2.3}$$

where $\boldsymbol{A}$ is a matrix of technological coefficients (the terms IO coefficient or direct input coefficient could also be used alternatively). It is derived from the transaction matrix, $\boldsymbol{Z}$ and total output vector $\boldsymbol{x}$ as: $\boldsymbol{A} = \boldsymbol{Z}(\widehat{\boldsymbol{x}})^{-1}$. Each element a_{ij} represents the amount of input requirement by industry j from industry i in order for industry j to produce a unit monetary output for final demand. Here the definition of the technology coefficient and then the Leontief IO model are based on two assumption: the constant returns to scale and fixed proportion. For any change in total production of industry j the amount requirement from industry i is in the same proportion with the change in the total output, this implies that the technological coefficient a_{ij} keeps a fixed relationship between the total output of an industry and its inputs, which ignores the economies of scale in production. On the other hand the model considers that for any change in total output of a given industry the proportion of inputs to the industries remain the same, which means that the input proportions are fixed with the technological coefficients (Miller and Blair, 2009).

The linearity assumption of the IO model allows to express the output of a given sector for an arbitrary demand. Therefore, equation 2.3 could be formulated as:

$$\boldsymbol{x}^{\diamond} = (\boldsymbol{I} - \boldsymbol{A})^{-1}\boldsymbol{y}^{\diamond} \tag{2.4}$$

where $\boldsymbol{y}^{\diamond}$ is the arbitrary demand, which could be demand from household, government or any demand on a specific sector, and so on. $\boldsymbol{I}$ is identity matrix, ones in the main diagonal and zeros elsewhere. $(\boldsymbol{I} - \boldsymbol{A})^{-1}$ is Leontief inverse or the total requirement matrix. It shows the dependency of total output with the final demand.

The emissions associated with any arbitrary final demand can be calculated:

$$\boldsymbol{n} = \boldsymbol{e}^{'}(\boldsymbol{I} - \boldsymbol{A})^{-1}\boldsymbol{y}^{\diamond} \tag{2.5}$$

where $\boldsymbol{e}$ is a column vector of environmental factor; for example, industrial direct GHG emissions per total output. Each element e_j represents the amount of GHG emissions released to produce a unit of € output of industry j.

According to the supply and use table (SUT) framework, the entire economic system is described by industries and product groups (commodities). The sectoral and commodity interdependencies are reflected by the total requirement matrices derived from the SUT, which is illustrated in Table 2.2.

$\boldsymbol{U}$ is the use matrix (often called input matrix) which shows how products are consumed by industries. Each element u_{ij} of the use matrix stands for the amount of purchase of commodity i by industry j. The $\boldsymbol{V}$ matrix (also called the output

Table 2.2: The SUT framework of an economy

	Industry	Commodity	Final Use	Total Commodity Output
Industry	-	$\boldsymbol{V}$	-	$\boldsymbol{x}$
Commodity	$\boldsymbol{U}$	-	$\boldsymbol{g}$	$\boldsymbol{q}$
Value Added	$\boldsymbol{v}$	-	-	-
Total Industry Output	$\boldsymbol{x}'$	$\boldsymbol{q}$	-	-

matrix) is a commodity-by-industry supply matrix, describes how each industry makes a product. Each element v_{ij} of the transaction matrix $\boldsymbol{V}$ represents the amount in € of product j produced by industry i. $\boldsymbol{q}$ and $\boldsymbol{x}$ represent the commodity and industry total output, respectively and $\boldsymbol{g}$ refers to the total commodity final use. Following the SUT structure presented in Table 2.2, the industry-by-industry EIO model in Equation 2.5 can be formulated as:

$$\boldsymbol{n} = \boldsymbol{e}'(\boldsymbol{I} - \boldsymbol{DB})^{-1}\boldsymbol{Dg} \tag{2.6}$$

In the same way, the industry-by-commodity EIO models can also be derived from the SUT. The detailed formulation can be found elsewhere (Miller and Blair, 2009).

$$\boldsymbol{n^1} = \boldsymbol{e}'\boldsymbol{D}(\boldsymbol{I} - \boldsymbol{BD})^{-1}\boldsymbol{g} \tag{2.7}$$

where $\boldsymbol{D}$ is a commodity output proportion (also called market share or supply coefficients matrix) derived from the supply matrix $\boldsymbol{V}$ and product output vector

$\boldsymbol{q}$. Each element d_{ij} is defined as $d_{ij} = v_{ij}/q_j$ and it represents the share of total commodity j output which is produced by industry i. $\boldsymbol{B}$ is the commodity-by-industry direct requirement matrix, which indicates the technological requirement of each product by industries. The matrix $\boldsymbol{B}$ is derived from the use matrix $\boldsymbol{U}$ and the vector of industry total output $\boldsymbol{x}$ and each element b_{ij} is defined as: $b_{ij} = u_{ij}/x_j$ and stands for the input requirement of commodity i associated with output of industry j.

EIO models have important features that make them potential methodological approaches for carbon footprinting of products and services. One of these features is their completeness. EIO models link all industries in a given economy and hence facilitate an assessment of the interdependency of industries. Another feature is their ability to assess both direct and indirect emissions explicitly. These features together allow EIO models to cover all emissions associated with the final demand of a given product. Through path analysis, they also allow a detailed tracing of the main sources and drivers behind each. Carbon footprint also aims to quantify all direct and indirect emissions through the life cycle of a product disregarding exactly where the emissions occur and these make EIO a suitable approach for carbon footprinting. Despite its important features, EIO also criticized for its poor level of aggregation. A collection of industries that have quite heterogeneous processes and outputs are usually merged as single industry with single output. For example, in Spanish IO framework, Manufacturing of mineral water, soft drinks, and alcoholic drinks are aggregated into Manufacture of beverages (See annex A.1 for the complete list of Spanish sectors and product groups). The environmental intervention from this industry is then approximated to be the average of interventions from all industries under this umbrella. This is one of the potential sources of uncertainties from the use EIO models, specifically when they are used in the analysis of environmental impact of a specific process, for example mineral water. However, errors due to industrial aggregation are still lower than the truncation error from cut-off in process-based LCA approach (Bullard et al., 1978; Lenzen, 2000). The treatment of imported goods is also another limitation of single-region EIO approach. When dealt with only single region, EIO approach usually assumes that imported products are produced with the same technology of the importing counties and therefore approximated to have the same environmental burden of corresponding products in the domestic economy. This could also be another possible source of uncertainty, particularly when there is technological differences between treading countries.

Other important aspect that needs to be considered is the limitation of EIO model to address the impacts from the use phase and end-of-life phases. The model explicitly addresses the impacts from material extraction, input production, and final product manufacturing. However, impacts from the use of products and disposal stages are not included; rather they need to be estimated by using other LCA methods.

EIO approach is applied in this thesis to estimate the carbon footprint of products and services in which the definition of economy-wide environmental tax is based (Chapter 4 and 5). It is also used to assess the price effect of environmental taxation on the production sectors (Chapter 6). In all cases a single-region EIO approach is considered. As mentioned earlier, it is based on the assumption that all imports are produced with the same technology as the Spanish production. This assumption can be adjusted by further extending the model by considering specific production process for imported products. For this purpose, we have constructed MRIO model for the Spanish economy. The detailed formulation and description of the model is explained in Chapter 7.

2.4 Hybrid-IOLCA

The methodological frameworks for estimating products and services life cycle emissions have often been described as either bottom-up (process-based LCA) or top-down (EIO)(Lave et al., 1995; Hendrickson et al., 1998). However the two distinct approaches have complementary features and can be used together in what is called hybrid analysis. There are several applications of hybrid LCA approaches (Moriguchi et al., 1993; Joshi, 1999; Matthews and Small, 2000; Lenzen, 2002; Nakamura and Kondo, 2002; Suh and Huppes, 2002; Heijungs and Suh, 2002; Suh et al., 2004; Finnveden et al., 2009). Generally both process-based LCA and EIO approaches have advantages and disadvantages one over the other. LCA provides more detailed and accurate analysis on a specific product level. But it is limited to few upstream processes since it is time and resource intensive to include all upstream inputs. Therefore, practically it is not possible to attain both detaility and system completeness at the same time (Suh, 2004; Suh and Huppes, 2005). Its constraint to cover the entire system boundary leads to errors due to system incompleteness. On the other hand, EIO approach can provide the holistic view of economic interdependence of industries and it is superior instrument to model environmental impacts at industrial level; for example, manufacture of beverages, but it is too limited in modelling impacts from specific process as it fails to provide detailed information on specific processes. EIO approach in general loses process specificity due to the high level of aggregation, limiting its application to study carbon footprint at national or regional levels. On the premises of avoiding disadvantages of process-based LCA and EIO approaches and combining their strength a hybrid approach has emerged as the state-of-the-art in the field of LCA. The system incompleteness of conventional LCA and lack of processes specificity of EIO could be overcome by combining both approaches. It is important to note that the term hybrid reflects both the methodological integration of process-based LCA with EIO as well as the combined use of physical and monetary values (Suh et al., 2004). There are three types of hybrid approaches: tiered hybrid LCA, IO-based hybrid LCA and integrated hybrid LCA (Suh and Huppes, 2005; Suh et al., 2004).

2.4.1 Tiered Hybrid

It is the simple summation of process-based LCA and EIO approach (Moriguchi et al., 1993; Suh and Huppes, 2002). Basically there are to cases where tiered hybrid could be used. The first is the case when process-based LCA is used to compliment EIO. This is the case; for example, where EIO data is used to estimate life cycle environmental intervention of entire activities from extraction to production process and process-based LCA data is used to cover use phase and end-off life as they could not be explicitly addressed with EIO model (Moriguchi et al., 1993; Joshi, 1999). The other case is when all foreground systems of production, consumption and end-off life stages of a product are modeled using process-based LCA data and the remaining background systems are included using EIO data in an attempt to overcome system incompleteness problem of LCA due to upstream cut-off (Suh and Huppes, 2005; Strømman et al., 2006). This can be illustrated in Figure 2.3 . The doted circle represents the system boundary of process-based LCA and the solid circle for the economic system. Inputs which are not covered or left out from the process-based LCA such as the payments made for maintenance services, analysis and investigation, and so on are included from the economic system. Based on Heijungs and Suh (2002) the tiered hybrid models can be expressed using the following equation:

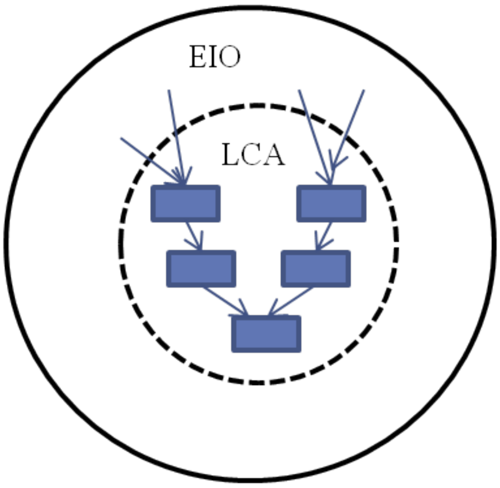

Figure 2.3: Tiered hybrid (IO approach to compliment process-based LCA), (adapted from Suh and Huppes (2005))

$$\tilde{\boldsymbol{n}} = \begin{bmatrix} \dot{\boldsymbol{e}} & \boldsymbol{e} \end{bmatrix} \begin{bmatrix} \dot{\boldsymbol{A}} & \boldsymbol{0} \\ -\boldsymbol{C^u} & (\boldsymbol{I} - \boldsymbol{A})^{-1} \end{bmatrix} \begin{bmatrix} \ddot{\boldsymbol{y}} \\ \boldsymbol{0} \end{bmatrix} \tag{2.8}$$

where $\boldsymbol{C^u}$ is the upstream cut-off and each element c^u_{ij} represents the cut-off in monetary unit of industry or commodity j which is required by the LCA process

i. All elements in $\boldsymbol{C^u}$ are negative entries. The total requirement matrix $\boldsymbol{A}$ of EIO model should be updated by subtracting the monetary values of physical flows to the process-based LCA (Strømman et al., 2009; Rowley et al., 2009).

2.4.2 IO-Based Hybrid

The general principle of IO-based hybrid analysis is that the life cycle environmental burden of a process or product of interest could be analyzed by disaggregating and representing it as a new sector entering into the economy. When detailed economic data on purchases and sales of the given production process is available, then its environmental burden could be analysed by splitting it from the corresponding representative sector in the economy. Detailed methodological approach and application of IO-based hybrid can be referred at (Joshi, 1999; Suh and Huppes, 2005; Treloar, 1997; Crawford, 2008; Crawford et al., 2006). According to the method described by Joshi (1999) models III and VI we can disaggregate the Manufacture of Pulp, Paper and Paper Product sector of the Spanish economy (sector 21 in Table A.1) and introduce the tissue production process; for example, as a new sector using all the inputs and output information provided by the tissue company. The Manufacture of Pulp, Paper and Paper Product sector is then split into a sector of "Tissue production", sector 21_a and "All Other Manufacture of Pulp, Paper and Paper Product sector except tissue paper", the new sector 21 (refer Figure 2.4 below).

The IO-based hybrid model could be expressed as:

$$\boldsymbol{n}^* = \boldsymbol{e}^{*'}(\boldsymbol{I}^* - \boldsymbol{A}^*)^{-1} \tag{2.9}$$

where $\boldsymbol{A}^*$ is the new matrix of technical coefficients that incorporate the new industry 21_a. Each element a_{i21_a} of the tissue company represents the upstream requirement of input from sector i to produce a unit of tissue paper and each element a_{j21_a} is output of tissue company to sector j. This output is estimated

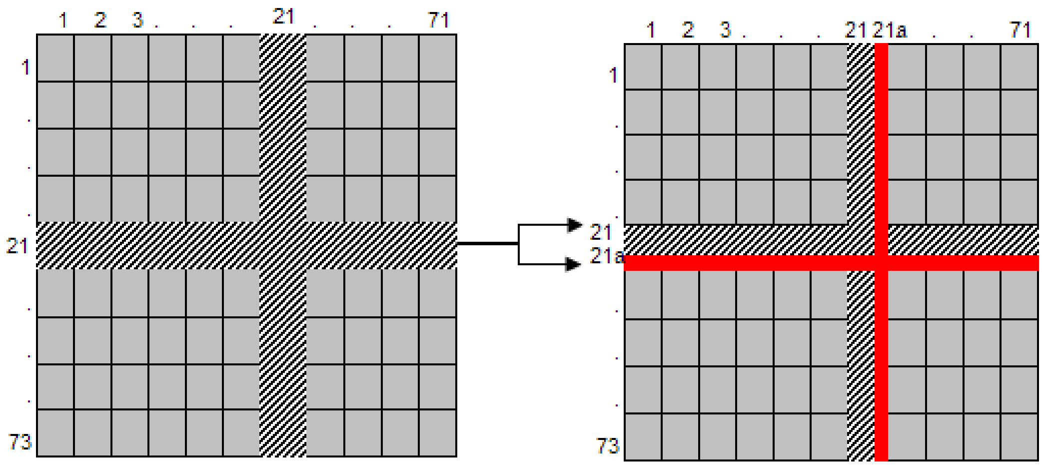

Figure 2.4: IO-based hybrid: The Manufacture of Pulp, Paper and Paper Product sector is disaggregated into the sector itself and Tissue production as a new sector

from the sales information of the company. This can be diagrammatically illustrated in Figure 2.4. $\boldsymbol{e}^*$ is the vector of emissions per total output of each sector. Unlike the previously defined vector $\boldsymbol{e}$, it comprises both the original emissions and direct emissions associated with the production of tissue paper.

2.4.3 Integrated Hybrid LCA

Integrated hybrid LCA is considered to be the state-of-the-art methodology in LCA and carbon footprinting. As proposed by Suh (2004) and Suh and Huppes (2005), it ensures a complete integration of process-based LCA with IO framework through the inclusion of both downstream and upstream cut-off. The upstream cut-off is the commodities in monetary unit that are left-off from the process-based LCA and which could be added to complete the system. Similarly the downstream cut-off represents the amount in physical units that are produced by particular process to the IO system. Detailed explanation of the upstream and downstream cut-off can be found elsewhere (Peters and Hertwich, 2006a; Suh, 2006, 2004; Suh and Huppes, 2005). According to Suh (2004); Suh and Huppes (2005); Suh (2006) integrated hybrid LCA could be expressed as:

$$\ddot{\boldsymbol{n}} = \begin{bmatrix} \dot{\boldsymbol{e}} & \boldsymbol{e} \end{bmatrix} \begin{bmatrix} \dot{\boldsymbol{A}} & -\boldsymbol{C^d} \\ -\boldsymbol{C^u} & (\boldsymbol{I} - \boldsymbol{A})^{-1} \end{bmatrix} \begin{bmatrix} \ddot{\boldsymbol{y}} \\ \boldsymbol{0} \end{bmatrix} \tag{2.10}$$

where $\boldsymbol{C^d}$ is the downstream cut-off and $\ddot{\boldsymbol{y}}$ is a vector shows the functional unit of the study.

Tiered and IO-based hybrid approaches are used to compare and assess their relevancy for economy-wide

2.5 Discussion

LCA of a given system can be performed at different levels depending on the scope and goal of the analysis. It could be done at a very specific product level, for example carbon footprint of a tissue paper production from VP or waste papers, or at meso-level[1] (e.g. carbon footprint analysis of tissue paper producing company in Catalonia, Spain) or at higher economic level (e.g. the environmental implication of pulp and paper producing sector in the Spanish economy). Based on the distinct methodological frameworks or formulations the life cycle environmental analysis at different levels can be better described by specific models. Figure 2.5 illustrates the general strengths of LCA approaches when they are used for different system level. Each framework has its own strengths and weaknesses over the other. For example, for systems at product level, process-based LCA is the most appropriate approach than EIO. Though the theoretical foundation of process-based LCA allows it to be used for any kind of system and capture all supply chain emissions; its large data, labor and resource requirement restrict it from being used for large and complex system. As a result some upstream or downstream sub-systems are left out from the analysis. This may result in the omission of important processes and may lead to truncation errors (Lenzen, 2000). The cut-off in process-based LCA could be as high as 20% in

[1]In economics meso- usually refers to a system which falls between the micro- and macro-levels. Here it represents the level between product and economy-wide system.

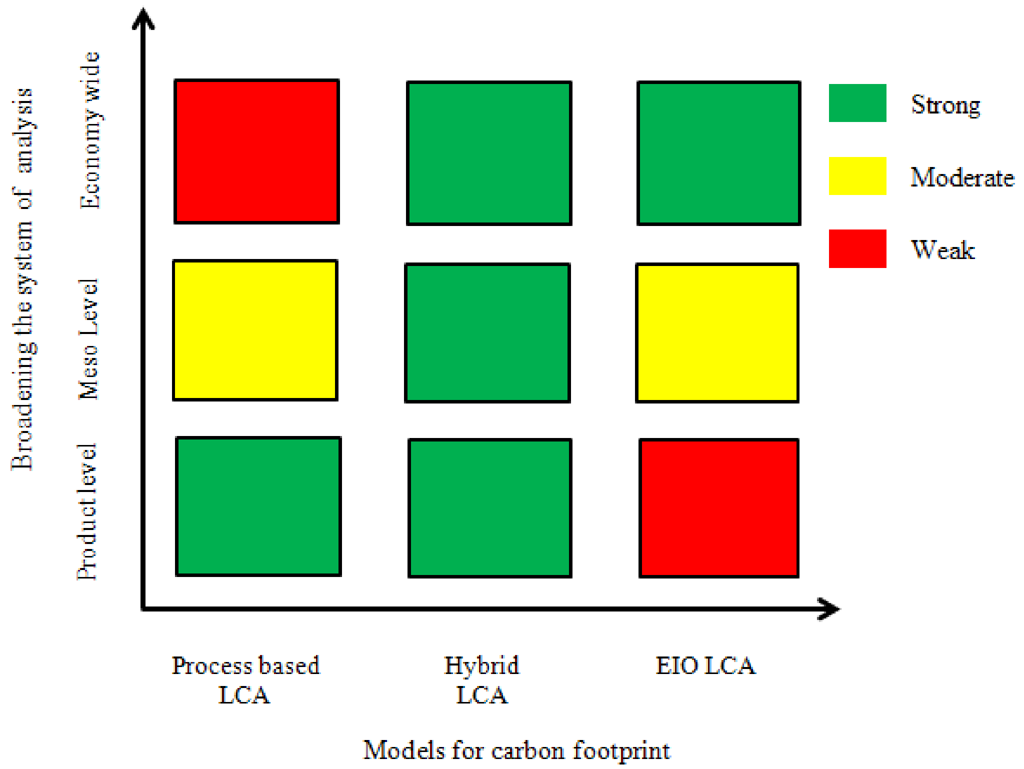

Figure 2.5: Description of LCA models and their systems (adapted from Guinée et al. (2011))

most environmental impact categories as reported by Suh et al. (2004). Such limitations of process LCA could be overcome from the use of hybrid approach, more specifically integrated hybrid LCA or tiered hybrid. Unlike process LCA, EIO approach is not subjected to boundary problems as it intrinsically covers the entire production systems in the economy. The consideration of capital goods (most of the time ignored in process LCA studies) as input to production systems, data availability and relatively less resource requirements are among the main advantages of EIO over process-based LCA. However, the high degree of data aggregation, which may result in losing process or product specificity, limits its applicability to systems at the macro-level. Again by the application of hybrid approaches, specifically from the use of IO-based hybrid, the completeness through the inclusion of all upstream and downstream inputs could be achieved while maintaining the process specificity. Therefore, the choice of one approach over the other mainly depends on the questions the study aims to address. Issues such as system boundary consideration, system completeness, data availability, uncertainty of source data, availability of analytical tools and time and labour requirements are among the main aspects that determine the applicability of one approach over the other (Suh and Huppes, 2005; Lenzen, 2002). Table 2.3 summarizes the main advantages and disadvantages of process-based LCA, EIO and hybrid approaches.

The next chapters discuss the practical aspects of the upper mentioned LCA methods. Their relevancy in estimating GHG emissions of a specific product or system are addressed through the analysis, a case study of paper and pulp sector in Spain. A detailed and complete process-based LCA of tissue paper production from VP and RWP processes using a real plant data of tissue producing company is discussed in Chapter 3. Then the comparison with other methods (EIO-LCA, IO-based hybrid and tiered hybrid) is performed in Chapter 4. Real monetary data on sales and purchases are used from the same company to simulate the hybrid models.

Table 2.3: Advantages and disadvantages of LCA methods, (adapted from Hendrickson et al. (2006); Greenroads (2011))

Methodology	Advantages	Disadvantages
Process LCA	• Gives detailed and product specific results • Is used for specific products comparisons • Allows to identify areas for process improvements/weak point analysis • Provides information that can be used for future product development	• System boundary setting is subjective • Needs high labor and cost for complex systems • Difficult to be used when initially developing a process or product • Often uses proprietary data • Cannot be replicated if confidential data is used • Uncertainty in data or missing data
EIO	• Provides economy-wide results • Allows for assessing both direct and indirect environmental effects • Allows systems comparisons at a coarser level • Data are publicly available and results are reproducible • Results can be used as information for future product development	• Data are highly aggregated • Specific product assessment is difficult • Difficult to linking monetary data with physical units • The same technology treatment of imported goods (in Single-region IO) • Lack of up-to-date data • Unable to consider use and end-of-life • Uncertainty in data

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Methodology	Advantages	Disadvantages
Tiered Hybrid	• Combines process LCA and EIO data • Reduces data collection time • Facilitates inventory analysis	• Subjected to double-counting errors • Omissions of important processes • Incorporates some disadvantages from both LCA and EIO models
IO-based hybrid	• Uses process specific date • Disaggregates IO key sectors and substitutes detailed economic information	• Incorporates disadvantages of EIO • Substitution of IO data for missing processes may reduce model reliability
Integrated Hybrid	• Combines process LCA and EIO data • Incorporates advantages from both LCA and EIO models • Connects process and EIO models • Consistent computational framework • No double counting	• Incorporates some disadvantages from both LCA and EIO models • Computationally complex • Data intensive • Time intensive

Chapter 3

GHG EMISSIONS OF TISSUE PAPER PRODUCTION: PROCESS-BASED LCA

3.1 Introduction

The increasing trend of population mainly in developing countries and a high level of consumption in industrialized countries are among the main drivers behind the rapidly increasing demand for natural resources and the associated GHG emissions (EEA, 2010). The pulp and paper producing industry is the fourth largest GHG emitter among global manufacturing industries and is responsible for around 9% of the total overall CO_2 emissions from manufacturing industries (EPN, 2007; IEA, 2007). The increase in paper consumption is the main driver behind the growth in the sector which results in having large ecological footprint on the planet (EPN, 2002). According to the OECD (2008), this demand-driven growth is projected to be at the rate of 2.3% a year in the coming decades. The consumption of paper varies hugely around the globe and it is correlated with income level. The United States and Western Europe are by far the biggest consumers of paper per capita, 334 and 202 kg per person per year, respectively (EPN, 2007). In addition to the burden placed on the environment due to resources extraction and production processes, the unsustainable consumption of paper creates a huge amount of municipal solid waste. Again according to OECD, waste from paper and paperboard make up 21% of the total municipal waste generated in Spain, which means that the fraction of paper waste is second only to the fraction of organic waste (49%). Therefore, policy measures that ensure not only a reduction in paper consumption but also an increase in the use of RWP, the implementation of cleaner production practices and the use of fibres from sustainable sources may be of great importance if the climate change impacts of the sector are to be significantly reduced (EPN, 2002).

The pulp and paper manufacturing process has important features that attract the attention of environmental researchers (Szabó et al., 2009). The first one is its high energy consumption and, therefore, its high emissions. In terms of energy consumption, the sector ranks alongside other energy intensive sectors such as cement, iron and steel (EPN, 2007; IEA, 2007). The second feature is the sector's intensive use of natural resources. One of the most important input materials in the paper making process is biomass from wood and other fibre sources such as annual plants. Thus, the future of forest lands is directly linked to the production of paper, given that 40-42% of all wood harvested globally for industrial use is used by the sector (EPN, 2007). Recently, RWP has emerged as an alternative to virgin wood pulp as a consequence of ambitious recycling policies (EC, 2008c) and fundamental technical advances in the paper production process. For example, in Spain the recycling rate[1]: has reached to 79% in 2010 from 48.6% in 2000 according to (ASPAPEL, 2011). The third feature of the paper production process that attracts researchers is the energy self-sufficient nature of the paper making process. Despite the fact that the sector is energy and resource intensive, important environment improvements have recently been achieved by the introduction of combined heat and power (CHP), which allows the sector to improve its energetic profile and reduce its use of natural resources.

To date, various studies have investigated the general environmental impacts of pulp and paper production (Lopes et al., 2003; Jawjit et al., 2006; Dias et al., 2007; Jawjit et al., 2007; González-García et al., 2009; Merrild et al., 2009; Cui et al., 2011; Hong and Li, 2012) and the effect of waste paper recycling compared with other waste disposal alternatives (Arena et al., 2004; Finnveden and Ekvall, 1998; Ross and Evans, 2002; Bjrklund and Finnveden, 2005; Finnveden et al., 2005; Althaus et al., 2007; Merrild et al., 2008). Quite few efforts have been made to study the comparative environmental benefits of using RWP and VP for tissue paper. A work from the ERM (2007) is one of the studies that deal with such analysis. The central objective of the study by ERM is to determine the environmental performance of multiple types of tissue products manufactured by Kimberly Clark and to assess the environmental trade-offs associated with the use of VP and RWP. The main findings for the study show that there is no environmental benefits of using RWP over VP or vice versa, as both provide environmental advantages and drawbacks. Specific to global warming impact, tissue paper that contains high VP has relatively lower impact than tissue paper with high recycled fibre content, which is contrary to the result we have obtained. Hohenthal and Behm (2008) also performed an LCA analysis to assess the carbon

[1]Recycling rate is defined as the ratio of the total consumption of recovered paper as a raw material by industries to the total consumption of paper and paper board

footprint of toilet tissue paper made from 100% fresh fibre pulp and 100% recovered fibre pulp. The analysis is based on three scenarios: allocating 8.55% and 19.55% environmental burden of the waste papers' first life to the recycled paper and without allocating previous life cycle burden (cut-off). The carbon footprint results are almost the same for fresh fibre and recovered waste fibre in all cases. The slight variation mostly depends on the choice of the burden for the previous life cycle of the recovered fibre. When the cut-off method is applied to assume that the recovered fibre is not responsible for the environmental burden from the previous product's life cycle, tissue paper from recovered fibre has a lower carbon footprint (1.275 kg CO_2 eq per kg of tissue paper). However, when the burden from the life of the previous product is allocated to the recovered fibre then it will have higher carbon footprint than the fresh fibre.

In this chapter we use real data from Gomà-Camp S.A.U, a tissue paper manufacturing company in Tarragona, Spain, to investigate the GHG emissions associated with the production of tissue paper from both VP and RWP in 2010. The study deals with the following questions: How does the choice of raw materials for VP and RWP influence the life-cycle GHG emissions of tissue paper production? What are the main drivers behind the emissions? What is the contribution of the direct materials, energy requirements and transportation to emissions generation? All these aspects are analyzed by considering all the stages involved in the life cycle of tissue paper production and identifying the processes that make the most significant contribution to the overall GHG emissions of the product. This kind of analysis highlights where attention must be really focused so that decision-makers at both company and national levels are aware of the main causes of the emissions and can take the necessary policy actions. Beside the aforementioned objectives, the results from this analysis are also aimed to be used as a base to compare with other LCA approaches presented in the Chapter 4.

The rest of the chapter is organized as follows: Section 3.2 describes the main goal and scope of the study, the definition of functional unit, main systems considered and data sources used to estimate the GHG emissions associated with tissue paper production from both raw wood and waste paper. Section 3.3 is devoted to analysing and discussing the results, and Section 3.4 contains the conclusions and the policy implications of the research.

3.2 System Description and Data Acquisition

Process-based LCA was the methodology used to compare GHG emissions from the VP and RWP processes. A cradle-to-gate LCA is applied to selected boundaries - from material extraction to the production stage of tissue paper. Following the ISO framework on LCA described in Chapter 2, the detail of the systems is presented in this section.

3.2.1 Goal and scope

Aim and scope of the study

The main aim of the study is to determine the GHG emissions that arise from producing tissue paper using both virgin and waste paper pulp. In doing so, this study will have several positive outcomes. Firstly, it highlights the most polluting steps during the life cycle of tissue paper produced from both virgin and waste paper inputs. Secondly, it gives an insight into the amount of VP can be saved due to the substitution of waste paper. This enables decision-makers to take more appropriate action to further reduce the environmental impacts of tissue paper. Last but not least, it provides the international LCA community with Spanish data on tissue paper production.

The scope of the study is from "cradle-to-gate"; that is, from the extraction of the raw materials to the processing and manufacturing of the tissue paper. The disposal phases are not covered for two reasons. First, there is a lot of uncertainty regarding the after-use stage of tissue papers because it is not easy to determine whether they are sent to landfill, incinerated or disposed of by any other means. The second reason is that even if we do know the end-of-life; it may be assumed that the GHG emissions associated with the after-use stages of both products would be the same regardless of the origins of the material used to make them.

Functional Unit

As has been mentioned, the principal objective of this work is to determine the GHG emissions of tissue paper products and the trade-off that could result from using either VP or RWP. Here it is important to pay attention to the functional equivalency of the products in order to compare their life cycle emissions and to correctly interpret the results. In this regard, both products are considered to provide the same functionality as they are thought to be roughly similar except for slight differences in brightness and dust content. The high quality achieved with tissue paper made from RWP is the result of state-of-the-art technology-intensive processes. Since both products provide the same utility, the functional unit of the system is defined as the production of 1 kg of finished tissue paper.

3.2.2 System Boundaries and Systems Definition

As we are dealing with tissue paper produced using VP and RWP, we have two distinct system boundaries. The physical and structural property differences between VP and RWP mean they require separate production process lines. Figure 3.1 and 3.2 present the details of the main processes included in the production of tissue papers from VP and waste paper, and the system boundaries, respectively. For the VP process, we have considered the following systems: wood logging, transportation to pulp mills, extraction and transportation of chemicals for the pulp making process, the pulp making process, transportation of pulps to the tissue paper mill, the tissue paper production process, and the treatment of waste from the production process. For the RWP process, we considered the collection and pre-treatment of waste paper, the extraction and transportation of chemicals, the tissue production process, and the treatment of waste from the production processes.

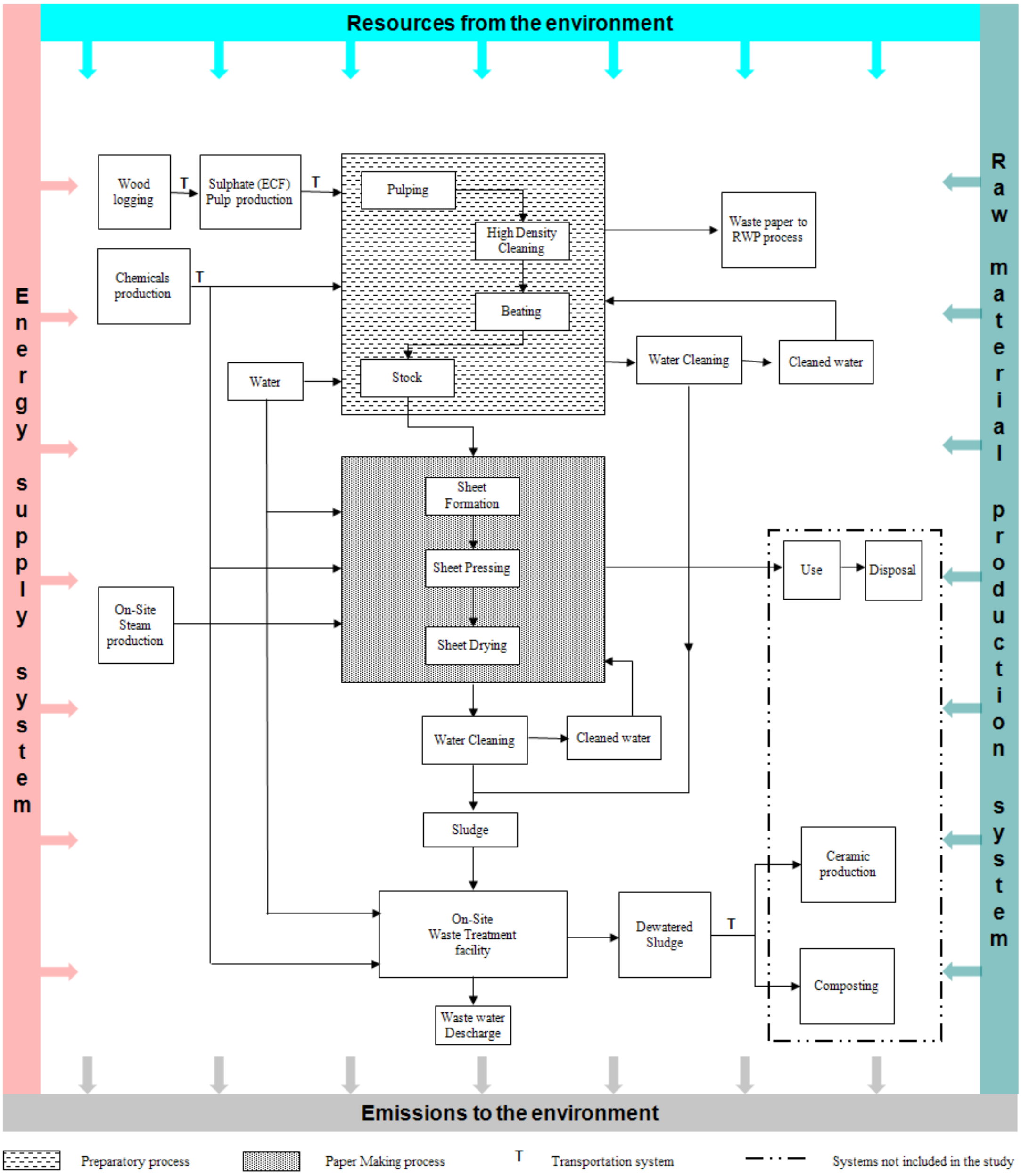

Figure 3.1: LCA system boundary and process flows of tissue paper production from VP

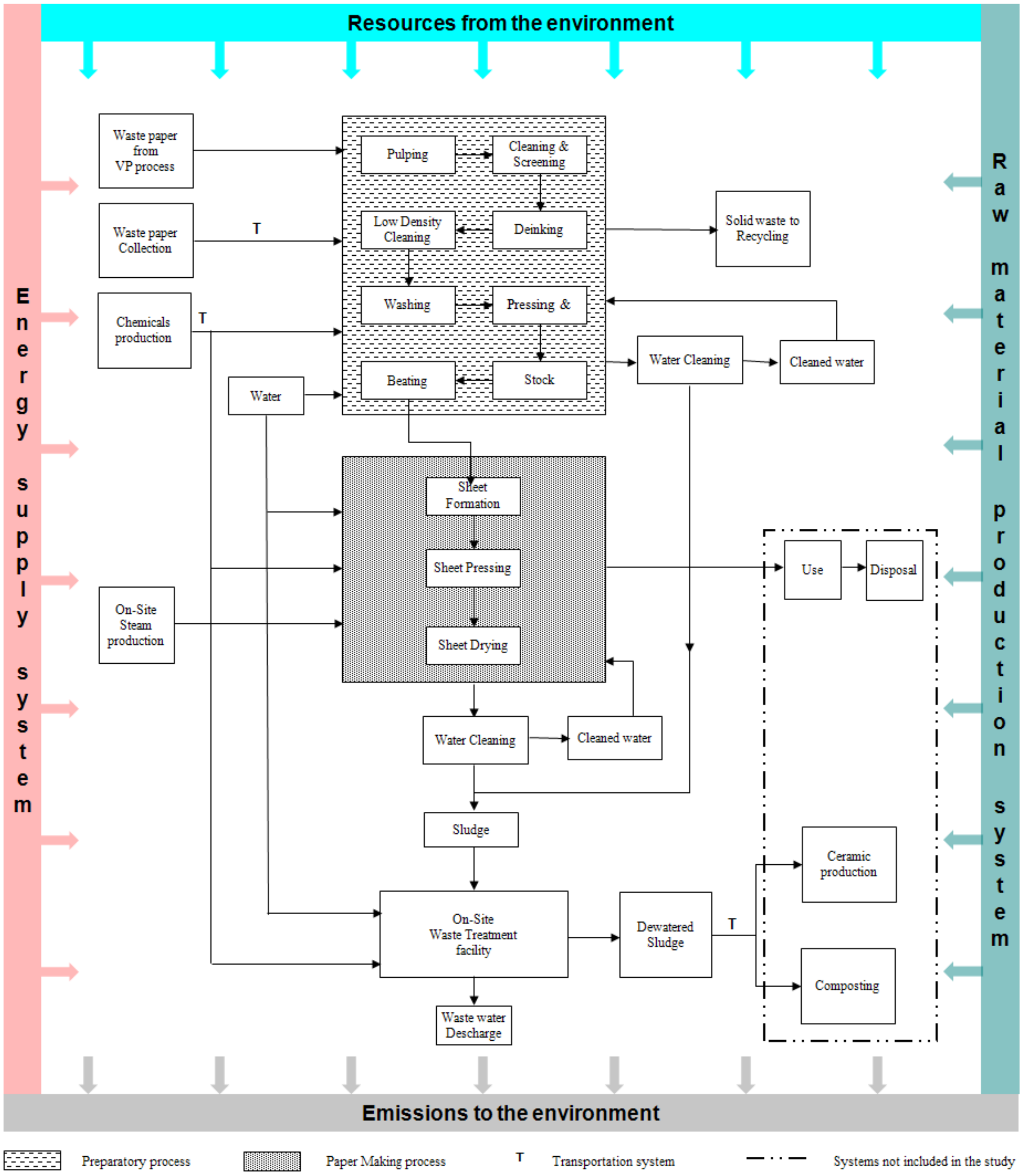

Figure 3.2: LCA system boundary and process flows of tissue paper production from RWP

Wood logging: Wood logging is the main system considered in the forest operation. Based on the data from ecoinvent woods from both pine (50%) and spruce (50%) were considered (Hischier, 2007). Logging includes the thinning process, final felling and the extraction of logs. Due to lack of data the cultivation is not considered, this might be changing the results somehow.

Pulp making process: The wood pulping process converts wood into fibre for the paper making process. This mechanism comprises highly energy intensive procedures such as debarking, chip refining, cooking, bleaching, washing and drying (Worrell et al., 2008). Pulping processes are broadly classified as mechanical and chemical pulping. Chemical pulping is the dominant process in the pulp and paper industries. It accounts for 75% of world's wood pulp supply (Das and Houtman, 2004). Kraft (sulphate) pulp is the most widely used chemical pulping process. On the basis of the data provided by the company, the present study considers the production of elemental chlorine free (ECF) sulphate pulp with a bleaching process. The main processes included are: wood handling, chemical pulping and bleaching, drying, on-site energy production, recovery and recycling of chemicals, and internal waste water treatments (Hischier, 2007). The VP used in the process is imported from Portugal (68%), Northern Europe (7%) and South America (25%) and similar pulping process is considered for all sources.

Waste paper: There are different sources of waste paper such as waste paper from curb-side collection, direct delivery from industries, secondary packaging, waste from paper production process and waste from printing and converting operations. Different models can be considered depending on the type of sources. For waste papers which are sourced from curb-side collection, both the energy and material requirements for the storing process and the subsequent transportation to the factory gate are normally considered. On the other hand, only transportation is considered for waste paper directly collected from industries or the paper producing process and delivered to the paper manufacturing plant (Hischier, 2007). Based on the data from the company the following types of waste paper were identified as the main inputs for the tissue paper making process: mixed office waste (30%), coated magazine (26%), white blank news (4%), light coloured office waste (8%) and broken (32%). Both the energy and material requirements for collection and transportation to the production site are included in the model. The emissions from the use of energy and material for collection and transportation efforts are considered. This allocation is in accordance with the recycled content approximation of PAS 2050 (BIS, 2011) and GHG Protocol (WRI and WBSCD, 2011). The recycled content method assigns the emissions associated with the recycling process to the life cycle that uses the recycled materials, which is tissue paper production process in our case (WRI and WBSCD, 2011). This approximation is chosen because of the fact that no recycling occurs at the end-of-life of tissue paper as it is considered as the last stage of the fiber.

Chemicals: Emissions from the extraction and transportation of chemical materials to the chemical industries and the direct emissions from the industries' chemical processes are included (Althaus et al., 2007).

Water: Emissions from the infrastructure and energy used during the treatment and transportation of water to the end user are considered.

Heat production: The extraction and production of natural gas and its distribution and combustion in the boiler at the tissue paper mill are included in the model. The material and energy requirements and the waste production during all the steps from extraction to combustion and the associated emissions are considered (Dones et al., 2007).

Electricity: The environmental burden associated with the production and transmission of the required amount of electricity is considered. Here the Spanish electricity production mix for the year 2010 is applied.

Transportation: The LCA considers the emissions associated with the operation of vehicles and ships to transport the pulp, chemicals and other input materials from their respective production sites to the tissue manufacturing gate. It also considers the transportation of waste (sludge) to waste recycling plants (composting and ceramic manufacturing). Furthermore, vehicle operations, emissions from the use of energy and materials during maintenance, disposal of vehicles, and road construction and maintenance are also considered. According to the data provided by the company around 75% of the wood pulp is imported from

Europe (Portugal (68%) and Northern Europe (7%)) by means of lorry. Average road distances of 1250 km and 3000 km are considered for Portugal and Northern Europe respectively. The VP imported from South America represents 25% of the total pulp. An average distance of 12000 km and ship mode of transportation is considered.

Infrastructure:The cut-off rule can be applied in LCA studies in order to avoid unnecessary effort to data gathering when the process, material input or energy input contributes to less than 1% of the total life cycle emissions. The infrastructure loads are thought to be negligible compared to the environmental burdens from the production process. Therefore, the infrastructure loads in all stages of the production processes are assumed to be of minor importance and have been left out of the analysis.

3.2.3 Data sources

Primary data such as the material input requirements for each process, the electricity and energy demand, and the transportation efforts are based on actual operational data from Gomà-Camps S.A.U. The annual data for the activity in 2010 were taken into account. Detailed information regarding the inventory of the foreground system of tissue production is presented below in Table 3.1. Extraction and production of ECF sulphate pulp (Hischier, 2007), chemicals (Althaus et al., 2007) and natural gas (Dones et al., 2007), and the collection and transportation of waste paper (Hischier, 2007) are modelled using secondary data from the ecoinvent database. The quantities of the material and energy inputs and the emissions specified in the inventory table are the quantities required and released during the production of the defined functional unit.

3.3 Results

The comparison between the VP and RWP process lines is based only on the GHG emissions resulting from the energy and material requirements for the production

Table 3.1: System inventory data for tissue production from VP and RWP processes

VP process			**RWP process**		
From Technosphere	Value	Unit	**From Technosphere**	Value	Unit
Biomass			*Biomass*		
Pulp	1.07	kg	Waste Paper*	1.5	kg
Transport - Road			*Transport - Road*		
Lorry 16 - 32t	1.16	tkm	Lorry 16 - 32t	1.1	tkm
Transport - Ship			*Transport - Ship*		
Transoceanic freight	3.23	tkm	Transoceanic freight		tkm
Energy			*Energy*		
Electricity	0.99	kWh	Electricity	1.1	kWh
NG (Heating system)	0.64	kWh	NG (Heating system)	0.7	kWh
Chemicals			*Chemicals*		
Soda	0.7	g	Soda	1.3	g
Organic Chemicals	0.51	g	Organic Chemicals	5.6	g
Resin	5.17	g	Resin	4.3	g
CO_2 liquid	2.97	g	CO_2 liquid	2.7	g
Urea	0.07	g	Urea	0.6	g
H_3PO_4	0.04	g	H_3PO_4	0.4	g
O_2	4.07	g	O_2	35.3	g
From Resource			**From Resource**		
Water	6.24	L	Water	8.6	L
To Technosphere			**To Technosphere**		
Solid waste (Recycle)	0	g	Solid waste (Recycle)	10	g
Sludge (Composting)	80	g	Sludge (Composting)	550	g
Sludge (Ceramic)	50	g	Sludge (Ceramic)	370	g
To Environment			**To Environment**		
Emissions to Water			*Emissions to Water*		
AOX	0.19	mg	AOX	0.23	mg
BOD	112.39	mg	BOD	139.6	mg
COD	374.64	mg	COD	465.36	mg
Nitrogen	0.2	ppm	Nitrogen	0.2	ppm
Phosphorus	0.2	ppm	Phosphorus	0.2	ppm
Water effluent	3.75	L	Water effluent	4.65	L

Note: all the values in the table are inputs and outputs associated with the functional unit, 1kg of tissue paper production from VP or RWP process.
*the fiber input is covered to 100% byRWP.

of tissue paper. It is important to note that the result would be different if we had also taken into account other environmental impacts such as land use and land use change. In the case of VP process harvesting forest lands are necessary in order to provide the wood for pulping, while the RWP processing line requires only waste paper as input, so consequently much less land use when compared to the VP. In terms of land use change, because the pulp comes from sustainably managed commercial forests (J. Gomá-Camps, personal communication, February 2012) we consider that there is no deforestation related to providing the wood for pulp production and therefore no related LUC-GHG impacts. In the following we present the results from analysing the VP production line, followed by the results of the RWP line and comparison of the two lines. The discussion of results is provided in the next section.

Figure 3.3 shows the GHG emissions associated with each input material and process. The total GHG emissions of tissue paper from VP process were calculated to be 1.875 kg CO_2 eq per kg. The emissions from the use of materials (pulp and chemicals) account for 32% of the total. The relative contribution of pulp production is 30%. The main sources of emissions in the pulp production process are the use of fossil fuels (hard coal, natural gas and heavy fuel oil), the combustion of lime in lime kiln stands, the combustion of biomass, and the chemical recovery units. This is also shown by González-García et al. (2009) and Jawjit et al. (2007). Only GHG emissions from non-renewable sources of fuels are considered. Biogenic emissions are neglected since they are assumed to be balanced by the amount of gases absorbed during the trees' lives, assuming that the biomass is sourced from sustainably managed forest. Emissions from the use of chemicals are relatively low and responsible for only 2% of the total impact.

Electricity consumption during tissue paper production accounts for 25% of the total GHG emissions. Electric power is mainly consumed to operate different motor drivers. Processes such as pumping, high density cleaning, refining, paper forming, pressing and finishing operations rely totally on electricity. Unlike most integrated paper mills there are no on-site power boilers that generate electricity for internal demand; hence, the entire source of electricity is the Spanish national grid, where electricity production is highly dependent on fossil fuels.

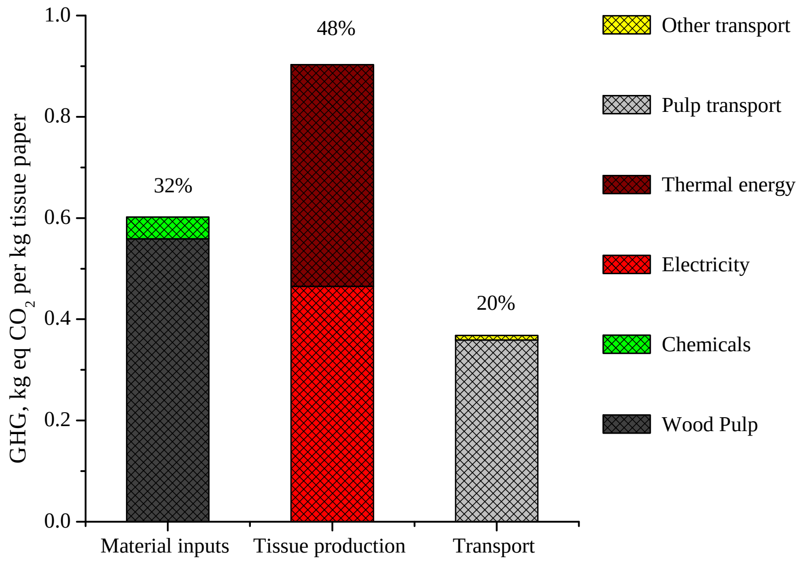

Figure 3.3: Contribution to the GHG emissions of inputs during tissue paper production using VP process

Steam and hot air production is also one of the major sources of GHG emissions and contributes to 23% of the overall result. Most of the steam heat is used in the drying stage of the paper making process. The drying operation is the most costly operation because it requires large amounts of steam in order to evaporate any excess water left over from the pressing operation. Steam heat generation in the mill is entirely dependent on the use of fossil fuel (natural gas).

Transportation is responsible for 20% of the GHG emissions. Around 75% of the total pulp is imported from Europe and the most commonly used form of transportation is by road. The remaining 25% of the pulp is imported from South America by ships. Although the imported pulp from South America represents only a quarter of the total amount of pulp used, it contributes 40% of the total transportation related emissions. Even though ship freight transport is the lowest impacting mode of transportation (ECMT, 1998), in this case its contribution is relatively high because of the long distance that the wood pulp has to travel.

The GHG emissions from RWP process are presented in Figure 3.4. The total emissions that arise from the use of waste paper as a main input in the tissue making process are calculated to be 1.31 kg CO_2 eq per kg. Production of electricity, steam and hot air are the most important inputs and contribute around 39.2% and 41.5% respectively of the total GHG emissions. As with the VP process, electric power from the national grid is used for driving motors in high and low density cleaning, screening, refining, paper forming, pressing and finishing operations. Steam is mainly used during the drying operation of the paper making process.

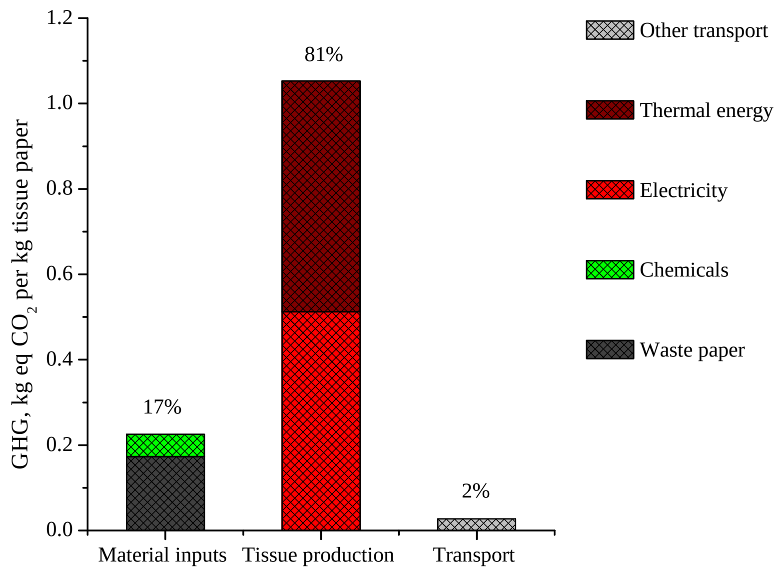

Figure 3.4: Contribution to GHG emissions of inputs during tissue paper production using RWP process

GHG emission associated with waste paper input is responsible for 13% of the overall GHG emissions. This is due to the fossil fuel required for collecting and transporting waste paper to the tissue production site. The emissions from the use of chemicals represent around 4% of the total GHG emissions. Chemicals are used both in the paper making processes and in the on-site-wastewater treatment plant. The GHG emissions from the transport category come from transporting chemicals both for the paper making processes and the wastewater treatment plant, and from the transportation of dried sludge from the on-site-wastewater treatment plant to the final use. An important amount of sludge (0.92 tons of dewatered sludge per ton of production[1]) is produced from the treatment plant and it is used as a raw material input in the ceramic industries (40%) and for composting plants (60%). The GHG emissions from the transportation represent 2% of the total emissions. Despite the fact that the relative emissions contribution of treated waste sludge is very small compared with the others, it is highly important economically as it costs the company up to 18€ to dispose of each ton of sludge.

Table 3.2 compares the GHG emissions for the VP and RWP processes. When only the manufacturing process is considered, the emissions from chemicals, electricity and steam are higher for RWP than for VP. This is mainly because manufacturing tissue paper from RWP process requires a relatively longer process, as can be seen in Figure 3.2. Additional processes such as screening, low density cleaning, deinking, washing, pressing and hot disperging are essential in order to clean and prepare fibre from waste paper for use in tissue production.

Table 3.2: Comparison of the GHG emissions for VP and RWP processes

VP process		RWP process	
Process	GHG kg eq CO_2	Process	GHG kg eq CO_2
Chemicals	0.043	Chemicals	0.052
Electricity	0.465	Electricity	0.512
Pulp	0.559	Waste paper	0.173
Steam heat	0.438	Steam heat	0.541
Transport	0.368	Transport	0.027
Water	0.002	Water	0.003
Total	**1.875**	**Total**	**1.307**

Functional unit: 1 kg of tissue paper

When the whole life cycle process is considered, a rather different picture emerges. The GHG emissions from the VP process are roughly 30% higher than from the

[1]Personal communications with the tissue producing company (2011).

RWP process, which implies a saving of 568 g CO_2 eq per kg of tissue paper produced. The emissions associated with the material and energy requirement for producing and transporting the main input materials are the most significant. For example, the transportation of pulps from South America and Portugal is responsible for the large divergence because it contributes around 93% of the total GHG emissions from transportation, whereas the GHG emissions caused by transportation during the RWP process are much lower because the paper waste is supplied by the local market.

In both VP and RWP process CO_2 emissions represent around 92% and 91% of the total GHG emissions in VP and RWP process, respectively. Next to CO_2 is Methane (CH_4) that contributes relatively high from non-CO_2 GHG emissions, 6% and 7% in VP and RWP processes, respectively. The contribution of other GHG emissions such as: Carbon monoxide (CO), Nitrous oxide (N_2O), Sulfur hexafluoride (SF_6) is very low.

It is also interesting to note the saving of trees (pulp) that could be achieved by using waste paper as primary input material to tissue paper making process instead of VP. Substituting used paper for VP can obviously help to reduce pressure on forests and the potential environmental impacts associated with forest operation and pulping processes. Here the most important issue is how to define the equivalence between VP and RWP. It is clear that a given amount of RWP does not provide the same quantity of tissue paper that can be obtained from a similar amount of VP, assuming that both are used to produce paper of a similar quality. The weight loss in the case of the RWP process is higher than in the VP process because fibre deterioration occurs during the use and recovery of the original paper mainly as a result of contaminants such as mineral charges, inks, plastics and other non-cellulosic materials. Hence, substitution ratio between RWP and VP is always less than one (Merrild et al., 2008). On the basis of yield data from the tissue plant, the output of tissue paper from 1 ton of RWP is equal to 0.7 ton from VP. This factor is used in the calculation. Around 87 ton of VP could be saved every day as a result of using RWP for tissue paper production. Being tissue paper is the last stage of fibre, in which recycling is not possible; the use of waste paper pronounces its advantage over VP.

3.4 Discussion and Concluding Remarks

Using RWP instead of VP has implications for various aspects of paper production such as use of water, use of chemicals, energy requirements, the effects on climate change and so on. As this paper estimates only the GHG emissions, we mainly focus on the emissions associated with the use of energy and materials. Here it is worth to mention that the biogenic CO_2 emissions are not considered in the analysis. The carbon neutrality of biogenic sources of CO_2 emissions are implemented on the basis that the wood pulp comes from sustainably managed forest. We have also neglected the temporary storage of carbon in the tissue paper, considering the short life span of the tissue paper as compared to other products from wood, such as furniture, buildings, etc. Hence only the CO_2 emissions which are released from the combustion of fossilized fuel sources were taken into account.

If we consider only the manufacturing process, the energy demand in the form of electricity and thermal energy is relatively higher for the RWP process than for the VP process. But when we compare the entire life cycle, a rather different picture emerges. The total life cycle energy requirement of the RWP process is lower than VP process mainly because the VP process needs more energy to prepare wood chips and transform them into clean lignin-free fibres. This process requires significantly more energy than transforming waste papers into fibres. The energy needed for transportation is also an important element that needs attention. A vast amount of fossil-fuel energy is required to collect and transport wood from forests to pulp making mills and then to the paper manufacturing site. To better understand whether RWP or VP uses less energy for transportation, it is important to understand the relative distance and the means of transportation used to convey the required amount of waste paper or wood pulp to the manufacturing site. For example, in the present study, around 75% of the wood pulp is imported from Europe by means of lorry and around 25% is imported from South America by ship. In contrast, most of the waste paper comes from the Catalonia area, and covers approximately 250 km during both collection and transportation to the site. The significant difference between the transportation distances for wood pulp and waste paper means that the associated energy demand for transportation is also significantly different. Our results indicate that substituting VP with RWP reduces significant fossil fuel emissions that arise from the transportation-related energy demands of the VP process.

Further reduction of emissions results when RWP is used to replace pulped wood from forestland, as the recycling of waste paper reduces the demand on forest wood and then eliminates subsequent energy and material requirements of the paper pulping processes. This could also increase forest carbon sequestration because the woods are left to grow to maturity, although such assumption cannot be guaranteed because it totally ignores the possible use of wood for other activities. In practice, when tissue paper is produced from 100% waste paper, then the pulp that has been saved could be used as an input for producing higher quality paper that cannot be made from 100% waste paper. Therefore, recycling waste paper can reduce the sectors overall dependency on forest resources but it does not completely avoid the use of VP as it is not possible to recycle paper ad infinitum (fibres can be recycled no more than seven times). Thus, there are two main environmental aspects to consider when making tissue paper from RWP. The first is the reduction in the total energy and material requirements and the associated emissions. As can be seen from the results, the life-cycle GHG emissions resulting from RWP are lower than those from VP. This becomes more important as technological advances allow RWP to be used to obtain paper with the same quality and functionality as that produced from VP. The second aspect is the non-recyclable nature of tissue paper in general. Unlike other types of paper, tissue paper is usually considered to be the last usable stage of the fibres (the end of the fibres' entire life cycle) because they are unlikely to be further recycled for use in the paper manufacturing process. So they could be used as the purge of the system. This is an important point that highlights the environmental benefit of producing tissue paper from RWP rather than VP. Therefore, if the policy is to reduce GHG emissions, then replacing VP with RWP is the best option for paper tissues production.

From the analysis of these LCA results with the Gomà-Camps company, several areas for improving the environmental profile were identified. Firstly, if the company implements cogeneration (CHP) unit to simultaneously generate both useful heat energy and electricity the environmental profile of the tissue product both from RWP and VP will be lower than the estimates presented here both in relative and absolute terms. This is because the use of CHP generally reduces energy losses from heat production by increasing the conversion efficiency of the fuel used (up to 93% of thermal efficiency), thus ensuring a reduction in GHG emissions (about 50% of the emissions from conventional power generation systems) (EC, 2001). The possibility of on-site power generation will also be of benefit as it totally or partially avoids the use of electricity from the Spanish national grid. Secondly, a change in the VP supply origin can decrease emissions by reducing the emissions from transportation. As shown in the results, emissions associated with the transportation of VP from South America are considerably high. Representing only a quarter of the total amount of VP used, it contributes 40% of the total transportation related emissions (8% of the total life cycle emissions). Therefore, reducing the import share of pulp from South America and switching to Europe market will reduce the emissions. However, the availability of sufficient supply in Europe market, the quality and the price issues are the most important factors that could challenge the practicability. Thirdly, the use of dewatered sludge for on-site energy generation. A significant amount of sludge is produced from the on-site waste treatment facilities and they are disposed of in ceramic and composting plant. The company will be both economically and environmentally benefited if the sludge is used as energy source, for the following reasons: it will reduce the total requirement of fossil fuels and thus avoiding the associated GHG emissions; it also reduce the transportation of the sludge to ceramic and composting plant, which then reduces the emissions due to transportation; and finally the company can also save the money paid for disposing the sludge. Therefore, we are confident that our results give the company a base for future planning and developments which will contribute to its positioning as an environmental leader on the Spanish Market of paper tissues.

Chapter 4

CARBON FOOTPRINT OF PRODUCTS AND SERVICES: COMPARISON OF LCA METHODS

4.1 Introduction

There is a need for strong environmental policies that support decision-makers to take action to combat climate change. This has motivated and contributed to the development and improvement of already existing LCA methods for assessing the environmental impacts of products. Since its concept has been introduced in early 1960s (Vigon et al., 1994), LCA has shown a significant development through its history. The current stage of LCA is described as a stage of elaboration according to Guinée et al. (2011), in which diverging approaches have emerged as a result of lack of common agreements on how to interpret some of the ISO requirements. These developments of LCA could be seen from different perspectives: from the consideration of temporal resolution (dynamic LCA)(Pehnt, 2006; Levasseur et al., 2010; Kendall et al., 2009), the treatment of allocation issues and spatial differentiation (Finnveden et al., 2009), system boundary selection and the application of EIO and hybrid based LCA (Suh et al., 2004; Finnveden et al., 2009; Hendrickson et al., 2006; Heijungs et al., 2006) and so on. This chapter mainly focuses on the latter case, the comparison of process-based LCA with EIO and hybrid-IOLCA approaches.

There are a number of studies that compare different LCA methods. Some have been carried out aiming at analyzing the potential of hybrid LCA in avoiding cut-off usually occurred in process-based LCA approach (Ferrão and Nhambiu, 2009; Mattila et al., 2010; Wiedmann et al., 2011). Mattila et al. (2010) analyzed the amount of cut-off from system omission in process-based LCA by comparing it with hybrid-IOLCA. The cut-off in process based LCA is not more than 25% for all environmental impact categories, but for metal depletion and terrestrial ecotoxicity. Results from EIO-LCA are fairly accurate with hybrid LCA except for land use and terrestrial ecotoxicity where they are underestimated and overestimated, respectively. More recently, Wiedmann et al. (2011) compared two hybrid-IOLCA approaches, namely IO-based hybrid and integrated hybrid with process-based LCA with the aim of building technology-specific processes into an economy-wide environmental modeling framework. Wiedmann et al. (2011) assessed the environmental impact of future energy supply for UK's economy.

According to the study CO_2 emissions from process-based LCA, which is based on the ecoinvent database, is 50% less than the ones estimated from both hybrid-IOLCA approaches.

This chapter investigates the relevancy of process-based LCA, EIO (pure EIO and hypothetical sector approaches), tiered hybrid and IO-based hybrid approaches for estimating the GHG emissions associated with a specific product, tissue paper production from VP. The main objective of the study is to compare the GHG emissions simulations of the specific product and analyze the discrepancy of the results that could arise from the nature of modeling, i.e., how the result from EIO, with less time and resource demand could be compared with the more detailed process-based LCA analysis presented in Chapter 3? We also aim at demonstrating how products or processes at micro system level could be simulated with macroeconomic models to determine their economy-wide environmental implications. The production of tissue paper in Spain was considered as a case study. The pulp and paper industry is among the sectors which are known for comprising high energy and resource intensive processes. This draws our attention to analyze the environmental implication of a product from the sector. More importantly what drives us to pick this sector is the availability of product specific detailed data (both physical and monetary) from the real tissue paper producing company in Spain. Such data allows us to apply hybrid-IOLCA approaches in order to compare with the process-based LCA and EIO approaches. Assuming national IO data are available, the methodology discussed here can be applied to any sector for which detailed LCA data are available.

The chapter is structured as follow: a brief description of systems considered for each approach and data acquisition are presented in Section 4.2. The comparison of results from different methods is discussed in Section 4.3, followed by Section 4.4 that is dedicated for conclusions.

4.2 System Description and Data Acquisition

For the process-based LCA, the case study presented in Chapter 3 was considered. The tissue production system is analysed in a cradle-to-gate approach, i.e., from the extraction of VP, through its processing and manufacturing of paper. This allows us to be consistent with the EIO approach, where the end-of-life and disposal phases are not intrinsically covered by the model.

For the EIO approach, the boundaries of the system are expanded to include all the Spanish domestic economic system. The EIO model described in Chapter 2 Equations 2.6 and 2.7 can be used to assess emissions associated with a specific product in two ways. One is by approximating the product of interest with its industry or product group. This implies that the product has similar technological requirements and environmental impacts (emissions) as the industry where it belongs to. It is also assumed that the requirements and emissions are proportional to the price of the product. This approximation does not need any specific information or data about the product, therefore it is quite simple and quick to apply. The Spanish input-output framework classifies the economy into 73 industries (Annex A.1) and 118 product groups (commodities)(Annex A.2). For the industry based EIO model, the production of tissue paper is included as an economic activity in the Manufacture of Pulp, Paper and Paper Product sector (sector 21 in the symmetric input-output table (SIOT) classification presented in Table A.1). Meanwhile, the commodity-by-industry EIO approach considers that industry 21 includes two product groups, namely Pulp, Paper and Paperboard (product group 29) and Articles of Paper and Paperboard (product group 30). Here the production of tissue paper is approximated by product group 29. This approach is here referred as a pure EIO approach. The other way in which EIO could be used for product assessment is when it is considered that the product of interest is not a representative output of the sector where it is classified. If economic data on its production are available then the product could be treated as a new hypothetical sector (Joshi, 1999). It can be assumed that the production of output from the new sector could induce an exogenous demand increase in the economy. Hence, the indirect emissions from the new sector could be evaluated by introducing exogenous demand vector $\boldsymbol{y}^{\diamond}$ in Equation 2.6 and 2.7 of Chapter 2. Each element $y_i^{\diamond}$ of $\boldsymbol{y}^{\diamond}$ refers to input requirement for producing € worth of the product (or a functional unit). The life cycle environmental impact of the product will then be the simple summation of the indirect impact linked with the production of its input requirements and the direct impact associated with its production. The system for this approach is illustrated in Figure 4.1. Similar with the process-based LCA system described in Chapter 3, the flow of important inputs such as energy (electricity and natural gas), raw materials (wood pulp, chemicals, waster and so on) are considered. Some additional inputs which are ignored in LCA system are also included (material used for cleaning, office work, payment for waste discharge, analysis and investigation). Of course the contribution of these inputs is expected to be low.

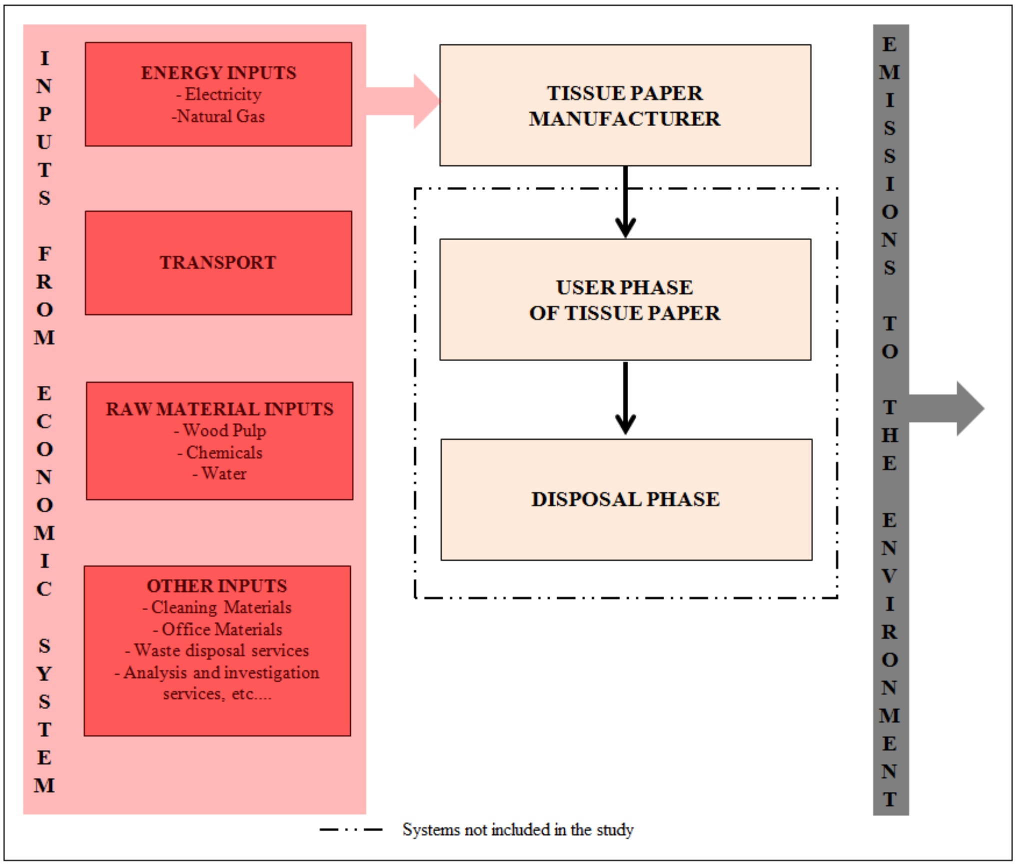

Figure 4.1: EIO-LCA system description of tissue paper production

The GHG emissions data for EIO approaches were obtained from the Satellite Atmospheric Emissions Accounts provided by the National Statistics Institute (INE) of Spain for the year 2007 (INE, 2010a). The emissions data were aggregated into 29 industries. Total sectoral outputs were used as a factor to disaggregate the emissions data into 73 industries (refer A.1) to bring them into line with the EIO models. These data were used to derive emissions vector $\boldsymbol{e}$ in Equations 2.6 and 2.7 of Chapter 2. The economic data on sectoral transactions came from the SUT published by INE for the same year (2007) (INE, 2010b).

Table 4.1: Input requirement of tissue paper production from VP

Inputs	Input Requirements %	Approximated Product & Sector SUT code	Approximated Product & Sector SIOT code
Electricity	6.1	13	9
Gas Natural	3.9	14	10
Water	0.1	15	11
Paper Pulp	52.3	29	21
Chemicals	2.2	32	23
Plastic Products	2.7	37	24
Spare Parts and Equipment	11.9	46	31
Cleaning, Security and Office Materials	4.3	67	42
Transportation	7.6	73	47
Analysis and Investigation	0.1	99	60
Waste Collection and Landfill Charges	1.2	108	67
Other	7.6		
Total	**100**		

For the IO-based hybrid analysis, detailed sales and purchases data associated with tissue paper production from VP are compiled by the company. They include purchases of energy and material inputs at each production stage and sales of the final and other intermediate outputs to the rest of the economy. Each purchase by the company and sale to the economy or final consumers are linked with the corresponding sector. For example the purchase of electricity is considered as the purchase of product group 13 (Production and distribution of electricity) in the case of commodity-by-industry approach or electricity input requirement from industry 9 (Production and distribution of electricity) in the case of industry-by-industry approach by the new sector (Tissue paper producing company). The total output from the tissue paper company represents around 1% of total output from the Manufacture of Pulp, Paper and Paper Product sector. The SUTs are used to introduce the sales and purchases information of the company as a new sector's supply and use data. All the inputs and outputs of the new sector are subtracted from the Manufacture of Pulp, Paper and Paper Product sector in order to avoid double counting. The main inputs and outputs are described in Table 4.1. As the Company's purchase and sales data are confidential they cannot be published, but only the percentage with reference to the total output are presented in the table. The direct GHG emissions data were estimated using process based LCA and they were also subtracted from the emissions of Manufacture of Pulp, Paper and Paper Product sector to avoid double-counting. Since the company produces tissue paper both from VP and RWP the direct emissions are allocated to both products by considering specific energy inputs to each system.

The sales and purchases data provided by the company were for the year 2010, while the latest SUT and sectoral emissions data of INE are for the year 2007. Therefore, they had to be transformed to 2007 values by adjusting the inflation. For this purpose the Consumer Price Index (CPI) approach was used. The value in 2007 is calculated as:

$$P_{2007} = P_{2010} * \frac{CPI_{2007}}{CPI_{2010}} \tag{4.1}$$

where P_{2010} is the value in 2010, and CPI_{2007} and CPI_{2010} are the indexes for the year 2007 and 2010, respectively. They are retrieved from INE database(INE, 2012b). It is important to note that this price adjustment may induce errors as the CPI is an aggregated index and the price change in each input may be different and not be well approximated by CPI.

The input values both for EIO-LCA and hybrid-IOLCA approaches are in purchaser's prices as they are obtained from the purchase list data of the company. They should be adjusted in order to reflect basic prices prior to introducing them into the SUTs. This could be done by subtracting the margins, delivery and taxes less subsidies on products using the following equation:

$$P_{basic} = P_{purchaser} - P_{purchaser} * \partial \tag{4.2}$$

where ∂ is the average margins and tax estimated from the Spanish SUT.

4.3 Results and Discussion

This section presents the results of GHG emissions associated with the production of tissue paper from VP process, using process-based LCA, EIO and hybrid-IOLCA approaches. The global result from each methodology is presented in Figure 4.2 and Table 4.2. Similar to the process-based LCA, in both cases; the cradle-to-gate life cycle GHG emissions are assessed.

The pure EIO approaches estimate the result to 797 and 794 g CO_2 eq per € tissue produced in industry and commodity based analysis, respectively. The GHG emissions results from both EIO models are under estimated to a great extent compared with process-based LCA. The emissions calculation from EIO models mainly depend on the level of aggregation, as they basically use averaged economic sector values across all paper and pulp production activities or product groups. This high level of industry aggregation is a problematic as it is unlikely that tissue paper production is well represented by its industry. Additionally, the results in EIO approaches also rely to a great extent on the overall input composition (both domestic and import structure) and emission profile of the

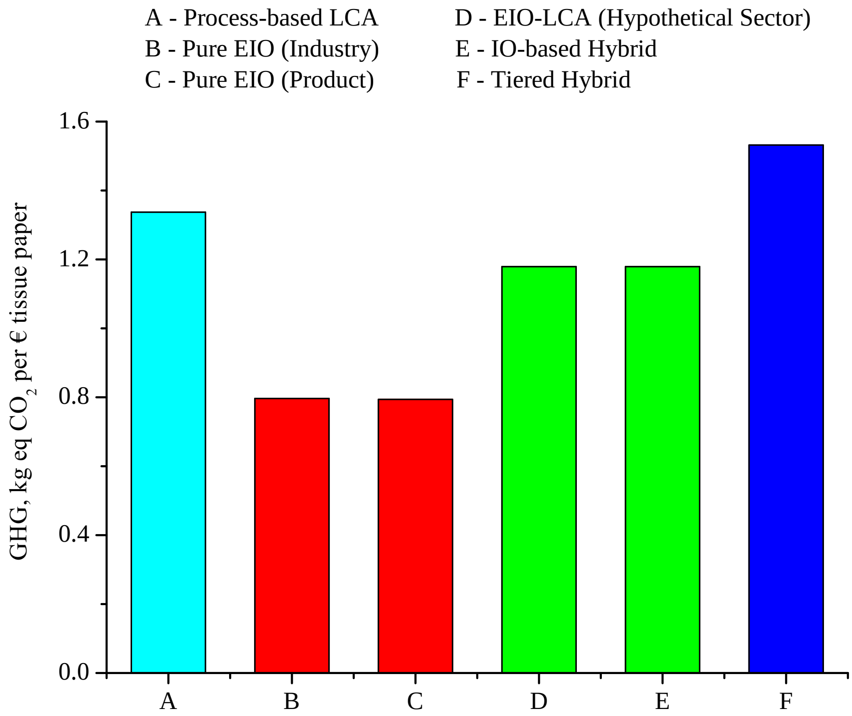

Figure 4.2: Comparison of GHG emissions inventories of tissue paper production

Manufacture of Pulp, Paper and Paper Product sector. Although not so detailed and precise, such kinds of approximations are very simple to apply as they do not need time and product specific data. Therefore, results from EIO approaches could be used as a first approximation as they allow for a quicker screening of GHG emissions related to all the products and services within the Spanish economy.

The problem with level of aggregation in pure EIO approaches to some extent could be overcome by extending the model in different ways. Considering the product as a new hypothetical sector entering to the economy could be an alternative solution. Being tissue paper is not a representative output product of aggregated industry 21 or product group 30, it can be treated as a new hypothetical sector. This approximation yield a better result than the pure EIO, 1,179 g CO_2 eq per € of tissue production, of which 74% is indirect emissions associated with input requirements from upstream supply chains. The remaining 26% is the direct emissions which are linked with the on-site combustion of fuels for heating and steam supply in the production processes. Even though this approach provides a result more precise than the pure EIO model due to the use of produce specific data, it is still lower than process-based LCA.

Inclusion of inputs which are not covered in process-based LCA results in increasing the emissions by only 14.6% in the case of Tiered hybrid. This suggests that most important inputs are considered in the process-based LCA.

Table 4.2: Contribution of main inputs to the total GHG emissions from tissue paper production (VP process), kg eq CO_2 per € tissue paper

Inputs	Process-based LCA	Pure EIO (Industry)	EIO-LCA (Hypothetical)	IO based Hybrid	Tiered Hybrid
Chemicals	0.03	0.04	0.03	0.03	0.03
Electricity	0.33	0.18	0.22	0.22	0.33
Pulp	0.40	0.15	0.06	0.06	0.40
Steam Heating	0.31	0.09	0.44	0.44	0.31
Transport	0.26	0.05	0.06	0.07	0.26
Water	0.00	0.01	0.01	0.01	0.00
Other	0.00	0.27	0.35	0.34	0.20
Total	**1.34***	**0.79**	**1.18**	**1.18**	**1.53**

* Average price of 1.4 € per kg of tissue paper is approximated based on the information on total sells and total productions.

Like the process-based LCA approach, the IO-based hybrid analysis is more product specific as it uses monetary input data from real tissue paper producing company in Spain. This enables the IO-based hybrid approach to estimate the GHG emissions more precisely than the traditional EIO. The results in Figure 4.2 show that GHG emissions estimated from the use of IO-based hybrid approach is relatively higher than both EIO approaches, but it is still lower than the process based LCA. Theoretically IO-based hybrid is expected to offer higher and more precise values than both process-based LCA and EIO as it combines both the process specificity and system completeness, which resulted from the use of process specific data and from the inherent nature of IO models to cover indirect

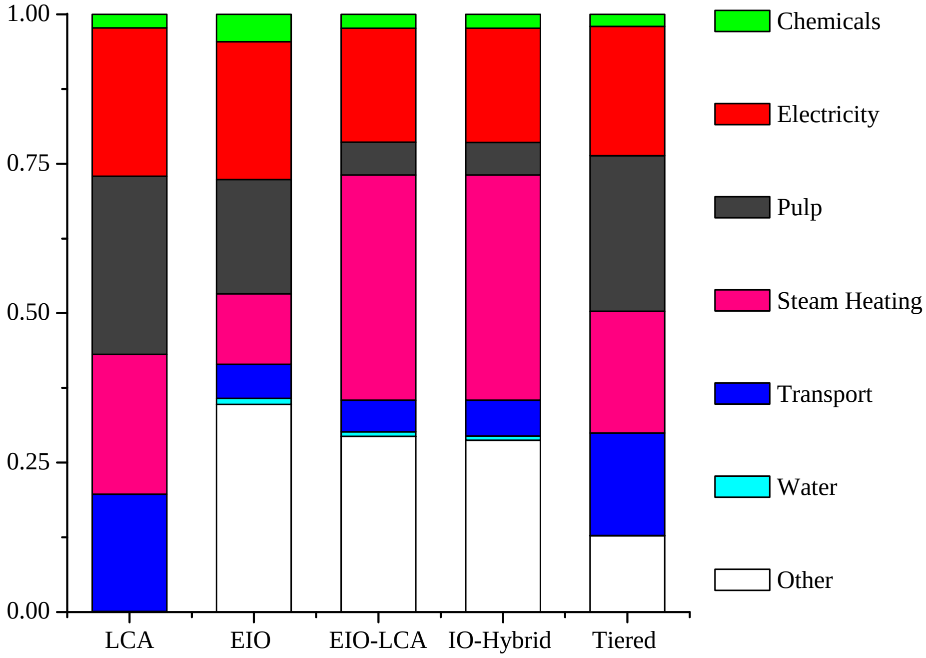

Figure 4.3: Relative contribution of main inputs to the total GHG emissions from tissue paper production (VP process), kg eq CO_2 per tissue paper

emissions from all upstream supply chain, respectively. Therefore, it is crucial to point out how this could possibly happen.

One of the possible reasons could be the treatment of international transport under EIO modeling framework. The detailed results from process-based LCA reveal that around 20% of the emissions are contributed by the transportation effort (Table 4.2 and Figure 4.3). Most of the pulp used by the process is imported from South America and Portugal using international freight transport. They are separately modeled in process-based LCA using the average distance and the amount of pulp imported (Chapter 3). However, the IO framework could not explicitly address the issue of international trade. As we see from the Table 4.2 and Figure 4.3, the relative contribution of transportation in both EIO and IO-based hybrid approaches are very small. The other possible reason for the underestimation of emissions in EIO and IO-based hybrid is the sectoral aggregation both in terms of emissions and input requirements. We would say the emissions profile of sectors which have substantial input contribution to tissue paper manufacturing process such as pulp and electricity are under estimated or too aggregated.

4.4 Conclusion

In this chapter we have applied different LCA methods which are used to estimate the life cycle environmental impacts of products and services. The traditional EIO model, EIO-LCA, IO-based hybrid and tiered hybrid approach are compared with the process-based LCA, taking the case of tissue paper production as an example. In principle, results from IO-based hybrid should yield a higher result than process-based LCA as they compliment both the advantages of process-based LCA and EIO (process specificity of LCA and system completeness of EIO approach). However, in this practical case, the IO-based hybrid is 13.4% less than the process-based LCA. There are a certain factors that may have contributed to this discrepancy. One is the aggregation of sectors in the Spanish IO framework. A good example could be the case of pulp input. As it is demonstrated in Table 4.2 and Figure 4.3 contribution of paper pulp is much higher in process-based LCA than EIO-LCA or IO-based hybrid cases, both in absolute and relative terms. Wood pulping is the most impacting process in paper making as it comprises energy intensive mechanisms such as pulping and drying. However, the Spanish IO framework does not differentiate the contribution of pulping process from others paper making processes which may have relatively lesser global emissions contribution to the sector. Whereas in the case of process-based LCA, the life cycle emissions associated with pulp input are separately modeled by taking into account all the processes such as wood logging operation, wood handling, pulping, bleaching, drying and so on as described in Chapter 3.

As explained, the handling of international transport also an important element that worth mentioning. The emissions from the transportation effort alone contribute to around 20% of the global emissions in the case of process-based LCA, while their contribution is quite low both in EIO and IO-based hybrid. The result could have been different if the emissions from the transportation of pulp from each region would have been considered. Besides, there are also other factors that likely affect the results such as the adjustment of inflation to transform price in different year using CPI and transformation of the input demands in purchaser price to basic price.

Based on the results obtained from this specific case study, we can conclude that Process-base LCA and Tiered hybrid approaches yield more precise calculation than other models considered, but they are time consuming and labor intensive to cover all the products and services in a national economy. In terms of operationally, process-based LCA might not be the best approach to calculate product or industry emission intensities on which to base an economy-wide policy implementation. Whereas, IO-based hybrid approach seems relatively easier to implement. Because it comparably needs less effort in introducing the product of interest into the IO framework than process-based LCA. Moreover, IO-based hybrid approach allows to assess the economy-wide implication of any change in production or consumption of a specific product since it is based on national input-output table.

Despite all these sources of uncertainties, generally LCA methods which are linked with IO framework have more advantages than process-based LCA approach as long as they are properly modeled and well approximated. Therefore, for this purpose we would recommend the use of EIO and IO-based LCA methods. The next chapters discuss how EIO models can be used for estimating the GHG emissions of products and services at a macroeconomic level.

Chapter 5

ENVIRONMENTAL TAX ON PRODUCTS AND SERVICES BASED ON THEIR CARBON FOOTPRINT

5.1 Introduction

Strong environmental policies that limit the growing dependence on fossil fuel are indispensable in order to reduce the growth of anthropogenic GHG emissions. This can be achieved through a regulatory based approach the so-called command-and-control as well as through market-based instruments. The command-and-control approach focuses on achieving reduced emissions through directly regulating the activities of companies and individuals by setting standards on the energy efficiency of manufacturing processes, on fuel content and use, and so on. Though it has been used in GHG emissions management, this approach is often criticized for the high administrative cost of its implementation. Alternatives to command-and-control approaches are market-based instruments, such as tradable permits and environmental taxes, which provide incentives to polluting industries for positive measures they implement to reduce their emissions. A properly designed market instrument can play an important role in moving the world closer to sustainability by reducing human-related emissions due to production and consumption. The cost effectiveness of technological innovation and the dynamic incentives inherent to market-based models are the two most notable advantages over command-and-control instruments (EEA, 1996; Jaffe and Stavins, 1995). However, distributional effects and global competitiveness are among the main shortcomings of environmental tax. The manner in which environmental tax treats different income groups in an economy is an important element that limits its applicability. Environmental tax is often considered as a regressive tax as it imposes a higher burden on low-income households than high-income households (Wier et al., 2005; Speck, 1999; Parry, 2004). In addition, imposing an environmental tax increases the cost of highly polluting energy sources and consequently increases the cost of production. Hence, domestic industries may lose their global competitiveness when they are competing with foreign producers from countries where similar environmental policies are not applied. Therefore, policy reforms are required in order to counter balance such negative implications and to make these instruments worthwhile (Fullerton et al., 2010; Parry et al., 2003; Poterba, 1993).

The principle behind an environmental tax is that a defined levy is introduced on environmentally polluting products. The definition is usually based on the potential cost of climate change effects caused by the production and consumption of these products. By internalizing the negative externalities (e.g. GHG emissions) and reflecting them in the price, the introduction of an environmental tax would raise the prices of polluting goods and services, while not affecting the prices of less or no polluting products. This would give consumers more information on the environmental profile of the products and services they purchase and could lead to a more sustainable patterns of consumption and production through promoting environmentally friendly products.

An important issue in the design of an environmental tax, and the central objective of this chapter, is how to differentiate between different products according to their particular emissions. The emissions associated with a product starts from the extraction and production of inputs necessary to produce the final product (e.g. a car). Emissions also occur during its production (e.g. emissions released during the production of a car), during its use (e.g. emissions released by burning the fuel in a car), and after its use (e.g. dismantling the car and recycling its components and/or disposing it in a landfill). The first question to be addressed is which of these emissions should be associated with a given product in order to place an environmental tax on it. Should we consider only the emissions from its production (also called "direct emissions") or the sum of emissions over the whole life cycle of the product, from materials extraction to its final disposal? Clearly, different boundaries of emissions assessment will lead to very different results, implying different measures (e.g. different environmental taxes). Therefore, we should pay attention to the choice of methods for the estimation of a product's emissions. In this chapter, we investigate the variability in an environmental tax when it is based on different methods to calculate GHG and CO_2 emission intensities of products and/or services in the Spanish economy in general. The case of tissue paper production is analysed in detail. EIO both sectoral and product/commodity approaches, and process-based LCA are used as methodological tools to calculate the emission intensities, which are then translated into environmental tax.

Furthermore in this chapter, the methodological approaches to define both CO_2 and GHG emissions related tax on sectors, product groups and specific products are outlined in Section 5.2. The description of EIO and LCA methods are presented in Chapter 2 and the choice of the most appropriate methodological approach that best determines the pollution embedded in a product or service for defining an environmental tax is discussed in Section 5.3. Finally, conclusions are drawn and the policy implications of the results are discussed (Section 5.4).

5.2 Environmental Tax Definition

In this section we analyze issues related to the definition of an environmental tax based on the carbon footprint [1] of products in the Spanish economy. The fundamental issue to be addressed is the choice of method, as the selection of one method over the other affects the outcome, due to different ways of defining a functional unit, different cut-off points and allocation rules. This also affects the policy implications that can be drawn from the results. Therefore, assessing the relevance of different available methods is an important step. This study is limited to process-based LCA and EIO (both the industrial and product approaches). Regardless of which method was used to calculate product emission intensities, the environmental tax ε calculated by multiplying the emission intensity vector obtained from LCA and EIO approaches by the cost of emissions as defined as:

$$\boldsymbol{\varepsilon} = \boldsymbol{\varphi C} \tag{5.1}$$

where $\boldsymbol{C}$ is a vector of emission intensity ($\dot{\boldsymbol{n}}$, $\boldsymbol{n}$ or $\boldsymbol{n^1}$ from equations 2.1, 2.6 and 2.7, respectively) and $\boldsymbol{\varphi}$ is the emissions cost expressed in €/ton CO_2 or €/ton

[1] Acoording to PAS2050 carbon footprint is defined as "the GHG emissions of a product across its life cycle, from raw materials through production(or service provision), distribution, consumer use and disposal/recycling. It includes the greenhouse gases carbondioxide(CO_2), methane(CH_4) and nitrous oxide(N_2O), together with families of gases including hydrofluorocarbons(HFCs)and perfluorocarbons (PFCs)."(BIS, 2011).

CO_2 eq. The same approach has been used by Creedy and Sleeman (2006), Hayami and Nakamura (2007) and Mongelli et al. (2009) to cite but a few.

One of the challenges with environmental tax implementation is the identification of a proper tax rate as setting the desired level of tax that could influence both consumers and producers is a very complex issue. There are different ways of evaluating the price of GHG emissions. Some approaches use costbenefit analysis, so that the environmental tax is set to be equal to the social cost of GHG emissions as a marginal social cost of emitting one extra ton of CO_2 or CO_2 eq (Clarkson and Deyes, 2002; Yohe et al., 2007). Others use the marginal abatement scenario, which considers the cost of reducing an additional emission unit (den Elzen et al., 2007; Hourcade and Shukla; Rao and Riahi, 2006). However, both measures are highly uncertain. In this study we consider the CO_2 tradable permit price of EU ETS (EC, 2008a) as equivalent to an environmental tax. The EU ETS was launched in 2005 with the target of reducing GHG emissions to at least 20% below the 1990 level by the year 2020. It works on the 'cap-and-trade' principle. The EU ETS established a uniform carbon price for selected industries across the EU which can be seen as an environmental charge for each industry and can be regarded as an equivalent to an environmental tax. The environmental tax on products and services based on their CO_2 emission intensities could also be considered to achieve the same reduction target as the EU ETS. The capacity of the tax in achieving the reduction target through the absolute reduction in consumption of goods (both final and intermediate consumption) could be assessed by capturing the response of production sectors and final consumers to the application of the environmental tax [1].

The multiplication of the intensity vector by the emissions price will increase the price of products or services in proportion to their emission intensities, assuming that the market is competitive and there are constant returns to scale.

[1] Taking into account that we are focused on quantifying the level of taxation that should be introduced according the sectoral pollution through the use of two alternative approaches, the calculation of the optimal price to achieve the required targets to reduce emissions is beyond the scope of this analysis.

The sectoral prices before the introduction of the environmental tax is defined as a function of the tax-exclusive price P_0 and an ad valorem tax t

$$P_1 = P_0(1 + t) \tag{5.2}$$

The new sectoral price after the introduction of the environmental tax will be

$$P_2 = P_1(1 + \varepsilon) \tag{5.3}$$

Combining equations 5.2 and 5.3, we find

$$P_2 = P_0(1 + t^*) \tag{5.4}$$

where t^* is the effective tax rate, defined as a percentage increase of the tax-exclusive sectoral price P_0 after the addition of an indirect tax, t, and an environmental tax, ε (Creedy and Sleeman, 2006)

$$t^* = t + \varepsilon(1 + t) \tag{5.5}$$

The data on the ad-valorem tax, t, on industries were calculated from the use table by dividing the taxes less subsidies on products by the total sectoral uses in basic prices. The values of t for the commodity-by-industry based EIO model were then computed by multiplying industrial t by product output proportion matrix $\boldsymbol{D}$.

5.3 Results and Discussion

Although not as detailed and precise as process-based LCA or Hybrid IOLCA, EIO allows for a quicker screening of CO_2 and GHG emissions related to all the products and services within the Spanish economy. The results of both the industry and product approach EIO models are summarized in Table 5.1 and 5.2 respectively. Specifically, Table 5.1 shows the results from the EIO model for the top 30 commodity groups that would be subjected to the highest environmental tax if they were levied based on their emission intensities (both CO_2 and GHG) per €. The products most affected by the introduction of an environmental tax based on CO_2 emission intensities would be those of the energy supplying sectors, such as electricity and gas with a 6.04% and 5.34% increase over the tax-exclusive price, respectively. As expected, energy intensive products like other non-metallic minerals, cements, glass and ceramic also show a relatively high percentage increase over the tax-exclusive prices. Cement, lime and plaster production is known as an energy intensive process, which results in high emission intensity both from the consumption of fuels and the calcination of limestone. Cement production alone accounts for 6.3% - 7.2% of global industrial energy use, with an average primary energy intensity of 4.4 gigajoules per ton of production (IEA, 2007).

The combined environmental and actual indirect taxes are mainly influenced by the environmental tax. This influence is especially visible in the case of Gas, for example, for which an environmental tax would increase its price by 5.80% compared to the 0.44% increase attributable to ordinary taxes. For Electricity, the combined tax rate is 5.57% due to an 6.04% environmental tax rate and 0.44% applied subsidies. These results show that the fiscal treatment of electricity does not point in the same direction as the environmental tax, and indeed implies that more subsidies on electricity and erroneous environmental evaluation will lead to a misleading conclusion. For example, electric cars are generally considered to be environmentally friendly products with almost zero emissions because the use of electricity as a fuel is usually considered as emission free. However, the production of electricity is highly polluting, especially in a country like Spain

Table 5.1: Environmetal tax rate for the top 30 Spanish commodity groups

Commodities	Code	CO_2			GHG		
		ε	t	t^*	ε	t	t^*
Electricity	13	6.04	-0.44	5.57	6.54	-0.44	6.07
Gas	14	5.34	0.44	5.80	6.47	0.44	6.93
Other non-metallic mineral	41	4.18	0.85	5.07	4.35	0.85	5.24
Cement, lime & plaster	38	3.97	2.73	6.81	4.15	2.73	6.99
Water	15	3.85	2.94	6.90	4.00	2.94	7.06
Glass products	39	3.83	1.71	5.60	4.02	1.71	5.80
Ceramic articles	40	3.70	1.15	4.90	3.90	1.15	5.09
Water transport	75	3.49	0.87	4.38	3.78	0.87	4.68
Coke & refined petroleum	12	3.41	2.06	5.54	4.79	2.06	6.95
Coal, lignite & peat	6	2.77	3.39	6.25	4.51	3.39	8.05
Air transport	76	2.66	0.29	2.96	3.01	0.29	3.30
Crude petroleum	7	2.42	2.42	4.90	4.16	2.42	6.69
NG, uranium ores	8	2.42	2.42	4.90	4.16	2.42	6.69
Land transport	73	1.65	9.40	11.21	1.84	9.40	11.42
Other chemical products	35	1.53	0.61	2.14	1.95	0.61	2.56
Pesticides	33	1.52	0.61	2.14	1.94	0.61	2.56
Pharmaceutical products	34	1.52	0.60	2.13	1.94	0.60	2.55
Basic chemicals	32	1.52	0.60	2.13	1.94	0.60	2.55
Basic metals	42	1.44	0.61	2.06	1.62	0.61	2.24
Articles of paper	30	1.31	0.65	1.97	1.59	0.65	2.25
Pulp, paper & paperboard	29	1.31	0.65	1.98	1.59	0.65	2.25
Fabricated metal products	43	1.26	0.42	1.68	1.39	0.42	1.82
Non-metallic ores	11	1.25	3.01	4.30	1.45	3.01	4.50
Recovered raw materials	60	1.10	0.20	1.31	1.25	0.20	1.46
Textiles	24	1.09	1.11	2.21	1.34	1.11	2.46
Alcoholic beverages	21	1.08	-0.77	0.30	1.43	-0.77	0.65
Mineral waters	22	1.07	-0.73	0.33	1.46	-0.73	0.71
Dairy products	17	1.04	0.33	1.37	1.78	0.33	2.12
Live animals & products	2	1.02	-2.07	-1.07	2.95	-2.07	0.82
Products of agriculture	1	1.02	-2.04	-1.04	2.95	-2.04	0.85

Note: ε is environmental tax rate (%), t is the ad-valorem tax rate (%) and t^* is the effective tax rate (%)

Table 5.2: Environmental tax rate for the top 30 Spanish industries

Industry	Code	CO_2			GHG		
		ε	t	t^*	ε	t	t^*
Electricity	9	6.08	-0.46	5.60	6.57	-0.46	6.09
Gas, steam & hot water	10	5.36	0.43	5.82	6.50	0.43	6.96
Water	11	4.40	2.15	6.57	4.49	2.15	6.73
Other non-metallic mineral	28	4.23	0.83	5.10	4.41	0.83	5.28
Cement, lime & plaster	25	3.98	2.76	6.85	4.15	2.76	7.03
Glass production	26	3.86	1.72	5.64	4.05	1.72	5.84
Ceramic production	27	3.71	1.15	4.91	3.91	1.15	5.10
Water transport	48	3.49	0.87	4.39	3.79	0.87	4.69
Coke & refined petroleum	8	3.42	2.07	5.57	4.81	2.07	6.99
Coal, lignite & peat	4	2.77	3.39	6.25	4.51	3.39	8.05
Air transport	49	2.66	0.29	2.96	3.01	0.29	3.30
Crude petroleum & NG	5	2.42	2.42	4.90	4.16	2.42	6.69
Land transport	47	1.65	9.63	11.43	1.83	9.63	11.63
Chemical production	23	1.52	0.61	2.14	1.94	0.61	2.56
Basics metals	29	1.44	0.61	2.06	1.62	0.61	2.25
Pulp & paper production	21	1.32	0.65	1.98	1.59	0.65	2.25
Other mining and quarrying	7	1.27	3.08	4.38	1.47	3.08	4.60
Fabricated metal	30	1.27	0.41	1.68	1.40	0.41	1.81
Recycling	39	1.10	0.20	1.31	1.25	0.20	1.46
Textiles production	17	1.10	1.11	2.22	1.35	1.11	2.47
Beverages production	15	1.08	-0.77	0.30	1.43	-0.77	0.65
Dairy production	13	1.04	0.37	1.41	1.77	0.37	2.15
Agriculture	1	1.02	-2.08	-1.08	2.96	-2.08	0.81
Mining of metal ores	6	1.01	0.35	1.36	1.18	0.35	1.53
Other food production	14	1.01	-1.39	-0.40	1.74	-1.39	0.33
Fishing	3	0.97	1.24	2.23	2.78	1.24	4.06
Rubber & plastic	24	0.96	1.04	2.01	1.16	1.04	2.22
Railway transport	46	0.96	0.23	1.19	1.07	0.23	1.30
Electrical machinery	33	0.96	0.36	1.32	1.11	0.36	1.48
Wearing apparel production	18	0.95	1.51	2.47	1.11	1.51	2.63

Note: ε is environmental tax rate (%), t is the ad-valorem tax rate (%) and t^* is the effective tax rate (%)

where the share of renewable energies in the national mix is low (20% of total energy production in 2007) (SEE, 2007).

If environmental taxes on CO_2 or GHG emissions were applied to industries instead of products, as Table 5.2 shows, these taxes would be almost the same and would not differentiate between products within the same sector; e.g. the Manufacture of pulp, paper and paper products (industry 21) is ranked as the 16^{th} CO_2 polluting sector and in the industrial environmental tax approach its products would experience a 1.32% price increase. In the commodity approach (Table 5.1), the same industry is split into two products: Pulp, paper and paperboard (product group 29) and Articles of paper and paperboard (product group 30), both are subjected to an increase of 1.31% in their prices if an environmental tax on their production were applied.

Table 5.1 and 5.2 also show that the positions of highly emitting industries and products change when the environmental tax is based on GHG emission intensities instead of CO_2 emissions alone. The industries that show a significant change when considering global GHG rather than only CO_2 are the Coal, lignite and extraction of peat (4), Agriculture (1) and the Food producing sectors such as, Diary production (13) and Other food production (14). The inclusion of non-CO_2 GHG emissions increases the environmental tax on Agriculture sector (1) by a factor of 3. The tax for Mining of coal and lignite, the extraction of peat (4) and the Food producing sectors (13 and 14) would increase by a factor of 1.7. This is mainly due to the emissions of CH_4 and N_2O. The main possible sources of CH_4 are from manure storage, particularly in anaerobic conditions and from enteric fermentation, especially in the case of ruminants. Similarly, N_2O is emitted due to the application of synthetic fertilizers and manure to soil as well as from crop residues applied to soil and from the storage and handling of manure.

The Manufacture of pulp, paper and paper products (industry 21) and its product groups (29 and 30) are not affected by the inclusion of non-CO_2 GHG for environmental tax calculation. The percentage increase of the tax from CO_2 to GHG is only 20% and it is very small compared to industries like the Mining of coal and lignite; Agriculture, livestock and hunting; and the Food production industries.

We next evaluated and compared the different approaches for implementing an environmental tax. The commodity approach has the advantage of differentiating between 118 commodities/products, while the sectoral aggregation includes 73 sectors. This would suggest the use of the commodity approach. However, when comparing the emission intensities in the two EIO approaches with the results of the process-based LCA, a much more detailed and specific environmental evaluation, there is no clear indication of which approach works better. Specifically, for the paper and paper products sector, we obtained the following results presented in Figure 5.1.

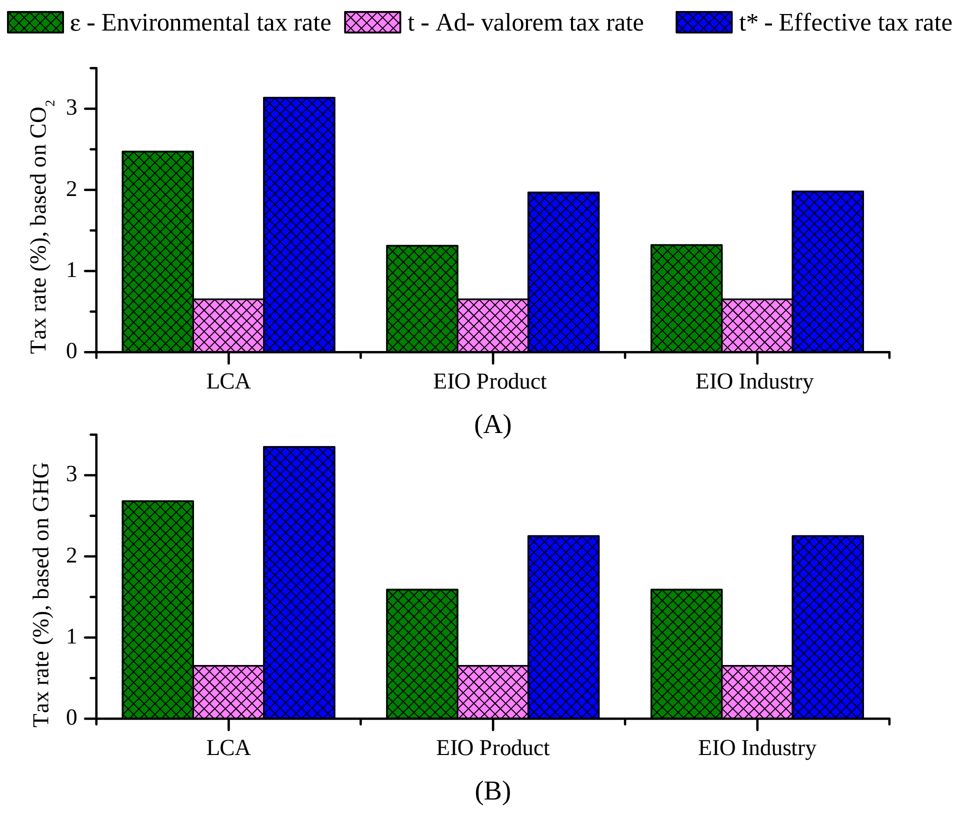

Figure 5.1: Comparison of tax rates for tissue paper product and paper industry based on CO_2 (A) and GHG (B) emission intensities from LCA, EIO industry and EIO commodity approaches.

Figure 5.1 shows the results that would be obtained if LCA were used as basis for calculating environmental taxes instead of the EIO approaches. Taking into consideration that we do not have the indirect taxes that are applied specifically to the tissue paper obtained through the VP process, we assume them to be equal to those applied to product group 30 (Articles of paper and paperboard see Table 5.1). LCA estimates higher emission intensities than the EIO approaches. Theoretically EIO should estimate higher emissions than LCA as it covers both direct and indirect emissions. However, there are several reasons why this is not the case here: first, the process LCA model uses on-site data of specific tissue paper production from a real plant, which include highly energy consuming processes, whereas EIO uses averaged economic sector values across all paper and pulp production activities considered within the sector. According to the results presented in Figure 5.1, as LCA estimates a higher CO_2 and GHG emission intensity than the EIO product group or sectoral approach, the environmental tax (ε) based on LCA would also be higher than both EIO approaches.

The tax comparison result between CO_2 and GHG emissions for the pulp and paper sector shows that the inclusion of non-CO_2 GHG emissions in the tax calculation has almost no effect on the product group or industry for either the EIO approaches or the LCA method, as the relative contribution of non-CO_2 GHG emissions to the total GHG emissions of the sector is very small (only 6.6%). However, it is important to take this into account for sectors with a high potential for non-CO_2 GHG emissions; for example, Agriculture, livestock and hunting, where the contribution of non-CO_2 GHG emissions is around 78.33% of the total GHG emissions of the sector. CH_4, with a global warming potential 25 times higher than CO_2, takes the largest share. Agricultural products serve as inputs to food industries, putting the manufacturing sectors of dairy products, meat products and other food products on the receiving end of high indirect non-CO_2 GHG emissions. Therefore, the environmental tax policy for such sectors should include all GHG emissions, as a tax based only on CO_2 emissions would not reflect the high potential impacts of non-CO_2 GHG emissions.

5.4 Conclusion

In this chapter we have defined an environmental tax on products and services of the Spanish economy based on their CO_2 and GHG emission intensities, with the aim of analyzing the relevancy of EIO and process-based LCA approaches for nationwide environmental tax application. We have compared both approaches by taking the production of tissue paper from VP in Spain as a case study, with a focus on determining how the methodological variation would be reflected in the tax levels. We have also compared the environmental tax rates that would result from considering GHG vs. only CO_2 emissions.

Our results show that emission intensity and the associated environmental tax for the Manufacture of pulp, paper and paper products sector are higher when calculated using the process-based LCA approach than either of the EIO approaches (sector and/or commodity). In theory, EIO models are supposed to yield higher values than the LCA model because they cover not only the direct emissions but also indirect emissions from all upper supply chains. But this is not the case in this specific study. It is mainly due to the use of more detailed and product specific data in LCA for paper pulp production and the more general and too aggregated approximation of EIO for the Manufacture of pulp, paper and paper product sector or the Articles of paper and paperboard product group. The underestimation of the sectoral emissions in the EIO model, which may have resulted from reporting or measurement errors, could also be one of the reasons for these differences. However, despite all these sources of uncertainties, we maintain that there are no significant differences between the results from process LCA and EIO and they are on the same order of magnitude. Process LCA yields a more precise calculation of emissions and environmental tax, but it is time consuming and labor intensive to cover all the products and services in a national economy. In terms of operationally, process-based LCA might not be the best approach to calculate product or industry emission intensities on which to base an economy-wide taxation. Whereas in the case of EIO, product specificity is lost, despite the fact that the model can provide a comparatively detailed and holistic picture of both direct and indirect environmental impacts associated with sectoral production, on which an environmental tax would be based. Therefore, for this purpose we would recommend estimating GHG and or CO_2 emissions based on hybrid IOLCA and only if not possible (due to poor data availability) then use EIO. A policy recommendation would be the compilation of more detailed IO tables and/or connection of IO to LCA databases which would allow differentiating between the environmental consequences of producing different products.

Another point raised by our research is the importance of considering non-CO_2 GHG emissions in environmental tax policies. CO_2 is the most important GHG and it is in the spotlight of most climate change mitigation strategies as it makes up the largest percentage of anthropogenic GHG emissions. However, there are also other gases with higher global warming potential that contribute to the greenhouse effects. Sectors such as the Mining of coal and lignite; extraction of peat (4), Agriculture, livestock and hunting (1) and the Food sectors (12 and 13) are among the sectors most responsible for the emission of a considerable amount of non-CO_2 GHGs in the Spanish economy. An environmental tax applied to these sectors would vary considerably if GHG emissions other than CO_2 were considered. Therefore, policy makers should be aware of the possible effects of any mitigating actions on the various different sectors.

The defined environmental tax based on EIO can be extended further to assess its potential impact on the economy. The methodology we use in the case study is a quantity-oriented framework and it does not allow reflecting the link between prices and quantities. Such links are important to show how a tax modifies emissions through the modification of the relevant variables in the productive decisions (prices and quantities). In this chapter we only define a tax, according to emissions intensities estimated from the use of process LCA and EIO and we approximate the price change by using quantity models. It is obvious that the sequence of effects is: the tax will modify (rise) the production prices and, as consequence, this will reduce both the demand (quantities) and the total emissions (in a second round of effects). But with the proposed approach it is not possible to illustrate how the taxation affects prices and consequently, how the price rise will modify quantities and emissions. A further development of our analysis will be to use the results as a basis of some frameworks able to capture the link between prices and quantities, on the one hand, and the link with the economy and the environment, on the other hand. The Leontief Price approach or the Computable General Equilibrium (CGE) approach is useful tool to capture both the complexities within the economy and the complexities of the interdependences between the economy and the environment. The next chapter will discuss the detailed formulation and application of the price model with the aim of assessing the potential impact of environmental tax when it is applied on selected sectors based on their CO_2 emission intensities.

Chapter 6

ECONOMIC AND ENVIRONMENTAL EFFECTS OF CO_2 TAXATION: AN INPUT-OUTPUT ANALYSIS FOR SPAIN

6.1 Introduction

Global warming has become of the most challenging issue facing humankind. Concern about climate change has increased as there are overwhelming scientific evidences that the earth is warming up, which may result in a devastating long term effects (Stern, 2006; IPCC, 2007). Global temperature rise should not exceed 2°C in order to avoid some of the most extreme consequences of climate change according to the recent scientific consensus. Human activities are believed to be the major cause of most of the observed increase in global average temperature since the mid-20th century (IPCC, 2007) and no efforts in emissions reduction will end up in unpredictable consequences (Stern, 2006; IPCC, 2007). Emissions from production and supply of energy have contributed considerably to the ever increase of the atmospheric GHG concentrations (IPCC, 2011). Hence, there is an urgent need for actions by governments, firms and individuals on climate change in order to avert its worst impacts and outweigh its potential cost (Stern, 2006; CLGCC, 2011). More recently more than 100 countries have adopted the 2°C limit as a guiding principle for climate change mitigation (Meinshausen et al., 2009). And over 200 companies from around the word have made a definitive and progressive statement to take action on climate change, sustainable development and green economy in a response to the urgent need of meeting the 2°C challenge (CLGCC, 2011). This has opened up a door for different environmental measures to be considered in the policy agenda. Environmental taxes and tradable permits are the economic instruments which mostly have been considered and implemented in order to control anthropogenic CO_2 emissions. Such economic tools can play a crucial role in achieving climate change mitigation in a more cost-effective manner than regulatory based approaches, as they equalise marginal abatement costs of industries (Jaffe and Stavins, 1995; McKinsey & Company, 2009; Fullerton et al., 2010). One of the possible effects of such policy measures is double-dividend, a dual effect produced; for example, from a tax increase on polluting goods (tax on CO_2). The fist dividend could be achieved when the tax on CO_2 increases the price of CO_2 intensive products, thereby reduce their production (quantity) and improve the environment. The second dividend is when the revenues raised from the tax are recycled to reduce the tax burden on the society, which further increases the welfare by lowering; for example, the income tax rate (Tullock, 1967; Terkla, 1984; Lee and Misiolek, 1986; Pearce, 1991; Bovenberg and Goulder, 2002). While the first dividend is remarkably supported both theoretically and empirically, the second dividend is still controversial (Bovenberg and Mooij, 1994; Goulder, 1995; Parry, 1995).

Environmental taxes have been put into practice in the developed world, especially in the EU since the nineties century (Ekins, 1999). A number of EU countries have implemented a wide range of environmental taxation such as taxes on motor fuels and motor vehicles, natural gas, coal, electricity, plastic bags, landfill wastes, batteries, pesticides, fertilizers, sulfur dioxide (SO_2) and CO_2 to cite a few. More recently the EU has established the Climate and Energy Package, which is the set of policy measures aim at achieving the 20-20-20 targets (to reduce EU's GHG emissions by 20% from the levels in 1990, to increase the share of renewable energy source by 20% and to increase the energy efficiency by 20%) to tackle climate change through transforming EU into highly energy-efficient and low carbon economy. One of the complimentary legislation which is intended to achieve the 20-20-20 targets is the EU ETS. EU ETS scheme is based on the 'cap-and-trade' principle and is considered to be the key and most cost-effective tool to reduce industrial GHG emissions in an effort to combat climate change. The EU ETS has been in place since 2005 and it covers 30 countries (27 EU countries and 3 non-EU countries). However, this chapter mainly focuses on the definition and impact assessment of CO_2 taxation using the Leontief input-output price model (hereafter referred to as price model).

The price model, firstly proposed by Leontief (1946) in an attempt to assess the US economy, can be used to evaluate the direct and indirect price effects of new policy measures in a given economy. For example, Manresa et al. (1988) used a generalized input-output model to evaluate the effects of the new indirect taxes in Spain after joining the European Economic Community. McKean and Taylor (1991) utilized the price model to study how changes in primary input prices influence internal cost of production in the Pakistan economy. This work revealed that tax levied on specific sectors could create significant input cost impacts in some sectors while it had relatively little effects on the others. An extended version of the traditional price model was also implemented by Boratyński (2002) to analyze the role of indirect taxes in the Polish economy. The study aimed at exploring the potential effects on final prices from 5% rate increase of indirect taxes on imported goods and effects from the adaptation of the new EU VAT rate on selected sectors. The simulation results showed that commodities or industries which have relatively largest share of import in their total supply exhibited the highest increase in final prices. Beside the share of the imports, the final prices were also driven by the increase in material costs. The sector with the largest increase in final price was the office machines and computers, which reveals 3.9% increase. Sectors that demand little material costs such as public administration and defence; education; and healthy care and social security were among sectors which have poor reaction in final price change. For the Spanish economy, Labandeira and Labeaga (2002) used the price model to study the effect of hypothetical carbon taxes levied on fossil fuel consumption based on sectoral energy-related CO_2 intensities for the Spanish economy in 1992. This work provided better disaggregated results on the emissions intensities and the price effects from the hypothetical tax introduction. The results from the empirical simulation suggested the need for the adoption of Spanish climate change polices both by emphasizing the necessity of improving the efficiency of energy system and highlighting the effectiveness and feasibility of carbon taxes as a policy measure in Spain. Llop and Manresa (2004) studied the influence of factor prices and imports for the Catalan economy. The results from the study showed that changes in import costs can create a greater impact on production costs than change in domestic input costs can do. This somehow discloses how the cost control within the economy could be ineffective when the changes in production costs are induced by inputs from foreign sectors. Llop and Pié (2008) also proposed a price model to analyze the consequences of a tax on energy uses in the Catalan economy. More recently, Llop (2008) used the price model to evaluate the potential effects of new water policy scenarios (20% decrease in water use and increase in water production, 40% tax on water use and the combination of first and second situations) in Spain. The empirical results showed that increasing technical efficiency and water production may end up in Jevons paradox. On the other hand, the tax imposition results in raising water price and in reducing its consumption. However, the combined effect suggests that both environmental and economic advantages cold be achieved if the water policy comprises both price and quantity measure.

In this chapter, we aim at investigating the potential impacts of CO_2 taxation in the Spanish economy. For this purpose we apply the price model to examine the effects on production prices induced from the environmental taxation. Unlike the tax model presented in Chapter 5, which is defined on quantity-oriented framework and hence fail to reflect the link between prices and quantities, the price model in this chapter allows to evaluate how production prices would respond to the implementation of the new environmental tax and consequently, the effects on individual consumer's welfare and on public revenues. Similar to the case presented in Chapter 5, the tax definition is based on sectoral life cycle CO_2 emission intensities, which are estimated from the use of EIO model. We also calculate the environmental consequences of the environmental taxation, measured in terms of the reduction in CO_2 emissions. All these variables allow us to reflect not only the economic impacts of the simulations but also the environmental impacts. Three sectors, namely: Production and distribution of electricity; Manufacture of gas, distribution of gaseous fuels through mains steam and hot water supply; and Manufacture of non-metallic minerals are selected to run the price model simulation. The selection is based on the emissions accounting data (INE, 2010a) and the result of the EIO analysis. These sectors are the most important sectors in terms of CO_2 emissions and together are responsible for 43%[1] of the total direct emissions in 2007 (INE, 2010a).

The chapter is organized as follows. Section 6.2 explains the methodological framework used for price model formulation. The empirical results are presented and discussed in Section 6.3. Finally Section 6.4 concludes.

[1]According to the satellite emissions accounting report of INE (INE 2010a) Production and distribution of electricity sector is the largest contributor of direct CO_2 emissions, which accounts for 26% of the total CO_2 emissions, followed by the Manufacture of other-non metallic minerals, which represents 10%. The emissions share of Manufacture of gas, distribution of gaseous fuels through mains steam and hot water supply is 7%.

6.2 Price Model Formulation

As already described in chapter 5, the definition of the environmental tax is based on the CO_2 emissions of products and services of the Spanish economy, see equation 5.1. Once the environmental tax for each product and service is estimated, then the impacts on the economy are analyzed using the Leontief price model. The impacts of such economic measure could also be examined by using CGE model. In our case, the choice of IO model mainly relies on the scope and depth of the study. If we compare IO and CGE, its simplicity to be implemented, relatively less data requirement and its instinct nature of capturing the inter-industy linkages are among the main strengths of IO price models. However, IO price models have several limitations when used in assessing alternative policy impacts. One of the drawbacks is the fixed coefficient assumption. This assumption totally disregards possible changes in industrial structure in response to changes in the economy (supply and demand interaction) that may result from any new policy action. Its limitation to explicitly examine the welfare implication of a new policy and overestimation of the policy impact due to the lack of supply side constraints are also other drawbacks of IO models that need to be considered while interpreting the empirical results. CGE models have quite strong features to be used for policy impact analysis. Unlike IO models, prices are not fixed as they are endogenous to the model, therefore, they are able to determine economic responses. CGE also allows the substitution effects in production and consumption, which enable to analyze the welfare implications of any change in policy. Such effects are not directly examined from the use of IO models. Despite their usefulness, CGE models in general require more data than IO models, and this may end up in high implementation costs than IO models. High level of input aggregation and lack of empirical validation are also among the weakness of CGE models. More importantly, it is not an easy task to properly interpret results from CGE models. As they usually contain a large number of variables and parameters (some of the parameters are usually determined by assumption), it is highly probable that important assumption could be hidden and makes it difficult to identify the main reason behind a particular result (Borges, 1986; Panagariya and Duttagupta, 2001). In this analysis, we have chosen the IO price model as a first approximation of the CO_2 tax impact assessment

Assuming that the sectoral prices are equal to the average cost of production, the normalized unitary price of output in each sector j, p_j, can be expressed as the total cost of intermediate inputs and total value-added expenditure as follows (Llop, 2008):

$$p_j = (1+\tau_j)\left[\sum_{i=1}^{73} p_i a_{ij} + (1+s_j)wl_j + rk_j + (1+t_j^m)p_j^m m_j\right] \tag{6.1}$$

where τ_j is the ad-valorem tax on production in net terms, a_{ij} are the input-output technical coefficients, s_j is the tax rate of social security paid by sector j, w is the price of labor (wage), l_j is the labor coefficient, r is the price of capital, k_j is the coefficient of capital, t_j^m is the ad-valorem rate of the imports in sector j, p_j^m is the price of imports and m_j is the import coefficient.

The impact of environmental tax rate (ε_j) on the cost structure of sector j could be evaluated using the following equation.

$$p_j^\varepsilon = (1+\tau_j)(1+\varepsilon_j)\left[\sum_{i=1}^{73} p_i a_{ij} + (1+s_j)wl_j + rk_j + (1+t_j^m)p_j^m m_j\right] \tag{6.2}$$

The above production price can be expressed in matrix form as:

$$\boldsymbol{p}^\varepsilon = (\boldsymbol{I} - \boldsymbol{A}^\varepsilon)^{-1}\boldsymbol{v} \tag{6.3}$$

where $\boldsymbol{A}^\varepsilon$ is the new technical coefficients matrix that incorporates both the ad-valorem and environmental taxes and $\boldsymbol{v}$ is the vector of value added per unit of output, which includes the capital, labor and import variables.

The impact of the environmental tax can also be analyzed in terms of changes in consumer's price index and in private welfare. Consumer's price index examines the weighted average prices of a basket of goods consumed by households and it is calculated by using a normalized basket of goods, which define the weights of the final prices:

$$p_c = \sum_{i=1}^{73} p_j \alpha_j \tag{6.4}$$

where p_j is the production price of sector j and α_j stands for the share of final goods from sector j as a ratio of the total goods consumed in the economy.

The impact of the tax on the private real income, that could be refereed as change in consumer's welfare, can be approximated using the following expression:

$$\Delta W = W - W^{\varepsilon} = \sum_{i=1}^{73} p_j C_j - \sum_{i=1}^{73} p_j^{\varepsilon} C_j \tag{6.5}$$

where p_j and p_j^{ε} are the consumption prices before and after the introduction of the environmental tax respectively, C_j is the consumption of goods of sector j by households. Any positive value in the change of welfare corresponds to a situation in which there is a consumer's benefit. A negative result represents a worse situation for consumers in which there is a reduction in individual consumer's welfare.

Changes in sectoral production prices induced by the tax could also be reflected in the total production output. Such effects can be simply evaluated by assuming that change in prices is in the same proportion as change in quantity, which keeps the sectoral output before and after the tax unaffected. A similar approach is used by (Llop, 2008). Therefore, the new sectoral output of sector j after the environmental tax x_j^{ε} can be calculated as:

$$x_j^{\varepsilon} = \frac{p_j x_j}{p_j^{\varepsilon}} \tag{6.6}$$

where x_j is output of sector j before the tax and taking into account that prices in the reference equilibrium are unitary (i. e. $p_j = 1$). This expression is further used to simulate the possible change in CO_2 emissions by arbitrarily choosing two alternative scenarios to the benchmark equilibrium condition in order to evaluate the sensitivity of the results to the total cost of production. The two additional scenarios are a 10% reduction and 10% increase in sectoral production output.

Using the proportionality assumption of the input-output approach, each sector's total CO_2 emissions are directly linked to the total output of that sector. Therefore, we can approximate the new sectoral emissions that would be released after the introduction of the tax:

$$E_j^{\varepsilon} = e_j x_j^{\varepsilon} \tag{6.7}$$

where e_j is CO_2 emissions intensity of sector j expressed as its total direct CO_2 emissions per its total output.

Finally, the total public revenues (R) that could be raised from the tax would be evaluated as:

$$\sum_{i=1}^{73} \varepsilon_j p_j^{\varepsilon} x_j^{\varepsilon} \tag{6.8}$$

6.3 Results and Discussion

EIO allows for assessing the environmental taxes based on CO_2 emission intensities for all the products and services within the Spanish economy. According to the source data from EEA (2009), the average EU ETS permit price for the year 2007 was 20€/ton. This price was used as a benchmark CO_2 price applied in order to estimate the environmental tax rate associated with each sector's CO_2 emission intensity. Figure 6.1 shows the qualitative analysis on the frequency distribution of sectoral tax rate when a price of 20€/ton of CO_2 is applied. As displayed in the graph, around 66% of the Spanish production sectors would experience an environmental tax rate of less than 1% and only 3% of sectors would exhibit a tax rate higher or equal to 5%. Almost 84% of sectors experience tax rates smaller than 2%.

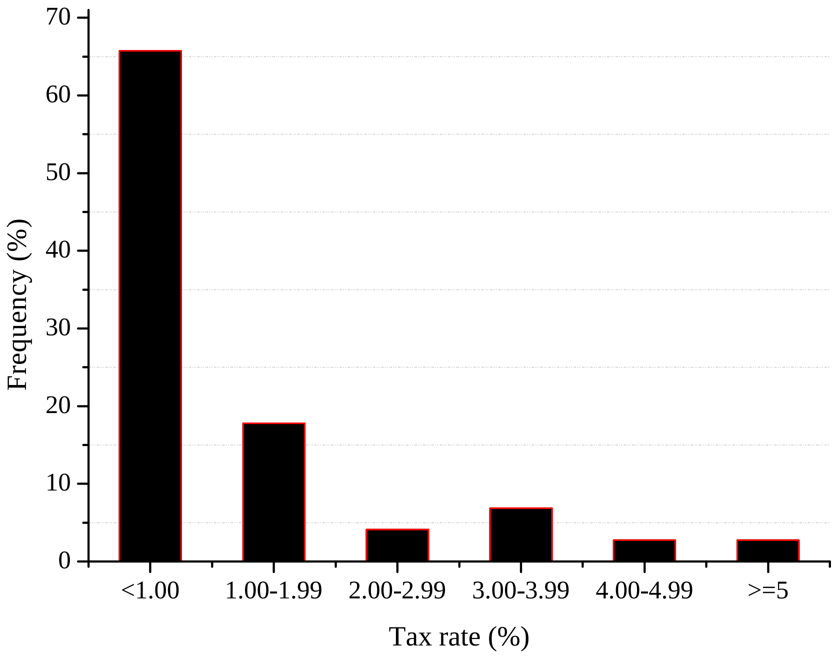

Figure 6.1: Frequency distribution of environmental tax rate of sectors in the economy

Figure 6.2 summarizes the top 25 sectors that would be subjected to the highest environmental tax. The highest environmental tax would be levied on Production and distribution of electricity, which exhibits a 6.08% tax rate. This is due to its poor environmental profile, in which the share of renewable energy source in the national electricity mix is significantly low, only 20%. Its production is mainly relayed on inputs from highly polluting sectors such as Manufacture of gas, distribution of gaseous fuels through mains steam and hot water supply; Manufacture of coke, refined petroleum products and nuclear fuel and Mining of coal and lignite, extraction of peat. According to the data from SEE (2007), energy from coal represents the largest share, which accounts for 24% of the total mix followed by combined cycle and nuclear power, representing 22 and 18% respectively. The share of renewable energy source in the national mix is only 19%. However, in recent years, the Spanish government has been taking a considerable effort in moving the sector a step forward to get a environmentally cleaner electricity, by increasing the share of renewable energy sources in the national mix. The recent data from the SEE (2010) shows that renewable energy has contributed by 33.4% in the total mix in 2010, with a 76% of increase compared with 2007.

Manufacture of gas; distribution of gaseous fuels through mains, steam and hot water supply and Manufacture of other non-metallic minerals are also subjected to high environmental taxes, 5.36% and 4.31% respectively. The gas manufacturing industry covers a wide range of operations such as exploration of gas, production, storage and distribution to end-users. The relatively high environmental tax of this sector is due to a considerable amount of emissions: one is from the direct combustion of fossil fuels used as energy source in process equipment and facilities in the industries, and the other is from equipment leaks and vented emissions.

The high emission intensity (and high environmental tax) of Manufacture of other non-metallic minerals is due to high energy (heat) and chemical requirements in the production process. This sector transforms mined or quarried non-metallic minerals such as sand, gravel, stone and clay into a wide range of products, for instance, concrete, mortar and blocks both for intermediate and final consumption. Energy intensive processes like grinding, mixing, cutting and shaping are

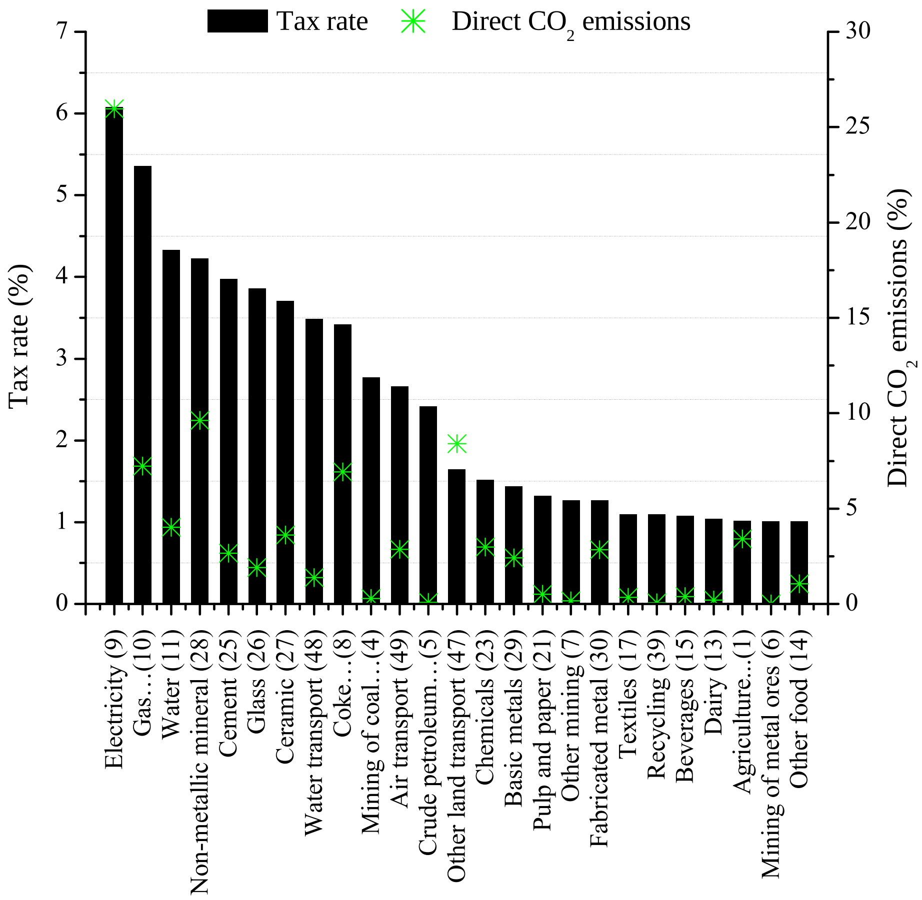

Figure 6.2: CO_2 emissions based environmental tax for the top 25 polluting industries

among the important processes responsible for the highest CO_2 sectoral emission intensities.

As expected, high energy intensive manufacturing sectors; for example, Cement, lime and plaster; Glass and glass products and Ceramic products, are also among the most affected sectors from the introduction of an environmental tax. These sectors are known for comprising high energy demanding processes. For instance, the cement production sector accounts for 6.3 to 7.2% of global industrial energy use, with an average of primary energy intensity of 4.4 gigajoules per tonne of production (IEA, 2007). High emissions of this sector are mainly due to the consumption of fossil fuel and the calcinations of limestone in cement production. The introduction of an environmental tax favours the less polluting sectors while it increases the production prices of goods and services in sectors with poor environmental profiles. The effect of the environmental tax is reflected on the production cost trough the market signals. Even though the sector itself on which tax is levied would be the most affected one, other sectors are also indirectly affected as the result of sectoral interactions. This can be captured with the application of a Leontief price model. To illustrate up how the price model works, we have considered three cases: an environmental tax imposed on Production and distribution electricity; on Manufacture of gas, distribution of gaseous fuels through mains, steam and hot water supply and on Manufacture of other non-metallic minerals. The selection is based on the CO_2 emission intensity results from the EIO model. As shown in Figure 6.2, these are the three activities that experienced the highest environmental tax rates.

In what follows, we present the price effects of the environmental taxation when it is applied to each of the selected sectors separately. A 6.1% environmental tax on Production and distribution of electricity caused a general increase in production prices of all sectors in the economy. As can be seen from Figure 6.3, the highest price raise is observed in the sector itself, which exhibits a production price increase of up to 7.6%. Particular sectors such as Transport via railways; Manufacture of cement, lime and plaster; Other mining and quarrying; Manufacture of other non-metallic mineral products; Mining of coal and lignite, extraction of peat; and Manufacture of pulp, paper and paper products are among the most sensitive sectors, and are subjected to relatively high price increases when environmental tax is introduced on the Production and distribution of electricity. These sectors are well known for their high electricity requirements in order to produce their outputs.

Figure 6.4 shows that a 5.3% environmental tax rate on Manufacture of gas, distribution of gaseous fuels through mains steam and hot water supply increases its production price by 5.4%. Though this is the highest impact, Production and distribution of electricity and Manufacture of ceramic products are also among

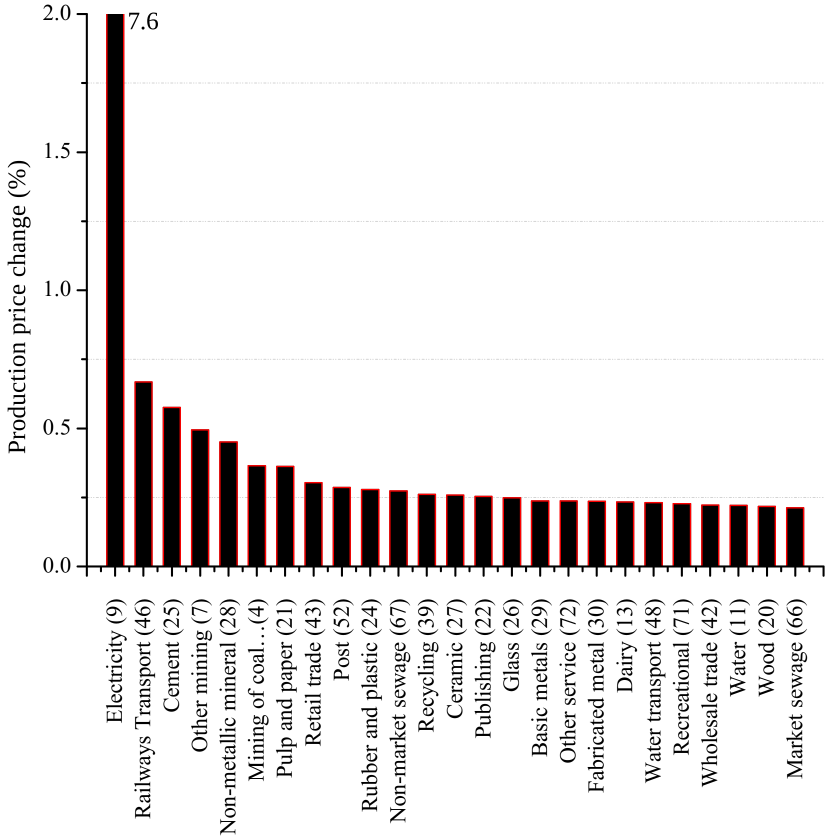

Figure 6.3: Changes in production prices (%) of an environmental tax (6.1%) on Production and distribution of electricity

the sectors affected, showing a price rise by 0.8% and 0.3% respectively. The effect on the other sectors is not significant.

A 4.3% tax on Manufacture of other non-metallic minerals is shown in Figure 6.5. As described, Construction is relatively sensitive to changes in production prices of Manufacture of other non-metallic minerals. Though not so high, sectors such as Manufacture of Cement, lime and plaster; Manufacture of glass and glass products and Manufacture of ceramic products are also sensitive to the price increase in Manufacture of other non-metallic mineral. The effect on the other sectors is negligible, as can be seen in Figure 6.5.

We also estimated the potential effects of the environmental tax imposition on

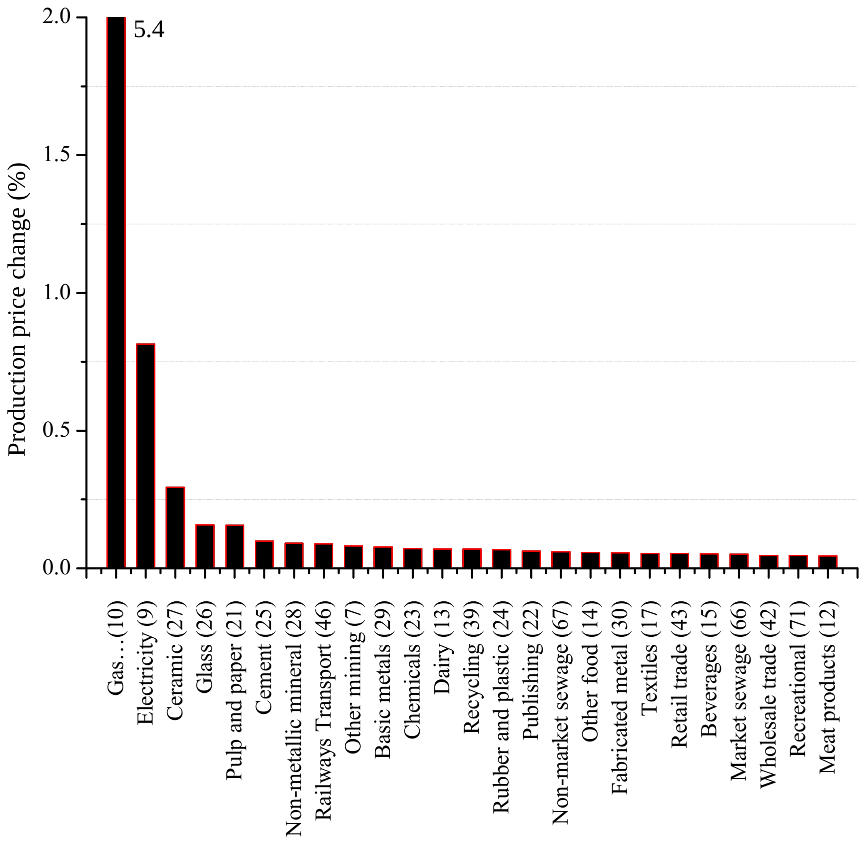

Figure 6.4: Changes in production prices (%) of an environmental tax (5.3%) on Manufacture of gas, distribution of gaseous fuels through mains, steam and hot water supply sector

other aggregated indicators such as the consumer price index, the individual welfare, the total tax revenues and CO_2 emissions reduction. The environmental tax induces a general increase in production prices and consequently in the consumption price index (Table 6.1). Environmental tax on production and distribution of electricity (case I) produces a relatively higher effect than when it is applied on Manufacture of gas, distribution of gaseous fuels through mains, steam and hot water supply (case II) and Manufacturer of other non-metallic mineral (case III).

The impact of the taxation on social welfare was approximated by estimating the

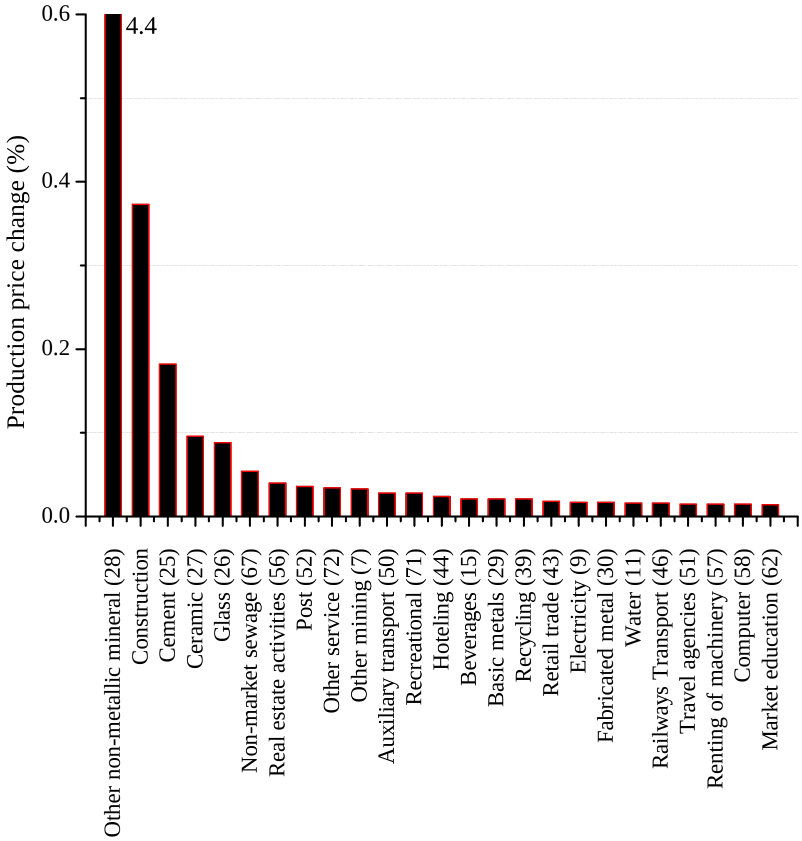

Figure 6.5: Changes in production prices (%) of an environmental tax (4.3%) on Manufacture of other non-metallic mineral

change in individual real income[1]. The aggregated loss of social welfare is often referred as deadweight loss. Measuring the deadweight loss is one of the common approaches to weigh up the economic distortion as a consequence of the new policy. As prices increase, the social welfare is negatively affected in all cases, as shown in Table 6.1. This can be interpreted as a decrease in economic well-being of the society due to the tax imposition. Again the effect of the tax on social welfare is by far higher in case I than in the other two cases. Electricity is an important sector in an economy given that whose outputs are highly demanded not only by the production system but also by the final consumers.

[1]Change in households' welfare is not a complete measure of social welfare, but it can be used as an approximation of the effects that the new policy measures cause on private agents.

A 6.1% tax rate in the Production and distribution of electricity raises a total revenue of 2,467 millions € which is equivalent to a 0.23% GDP share for the year 2007. The total revenues in cases II and III totalise only 25% and 40%, respectively, of revenues generated in case I.

The results in Table 6.1 also show the effect of CO_2 taxation on total emissions of the economy. The direct environmental effect of the tax is assessed by considering two arbitrary adjustments to the production at the benchmark equilibrium. When a constant cost of production is assumed a 6.1% of environmental tax on the Production and distribution of electricity, achieves a 2% reduction of direct CO_2 emissions. The direct effect on emissions at a constant production can be seen very small; however, a better effect could be indirectly achieved if the revenues raised through the environmental taxation are used for stimulating cleaner energy production by financing innovations in renewable energy technologies. This would increase the use of renewable energy sources and consequently it would decrease further CO_2 emissions. When the sectoral production is allowed to vary, even a higher emissions reduction (11.8%) could be achieved if the production decreases by 10%. However, a 6.1% tax on the Production and distribution of electricity could not ensure the require emissions reduction when there is an increase in production (10% increase in production resulted in 8% increase in direct emissions). The emissions reduction in case II and case III are relatively lower compared with case I in all production cost assumptions.

Table 6.1 illustrates a clear trade-off between the economic, social and environmental goals when an environmental taxation is levied on the economy. Specifically, the highest improvement in the environment is in case I and it coincides with the worst situation for consumers, despite being when public revenues increase to a greater extend. On the contrary, the best situation in terms of private welfare is obtained in case III where CO_2 emissions would be reduced by the smallest amount. We can also point out that, despite what it is expected under a new taxation on production, the three cases analyzed show slight price increases, and this is important if the objectives are focused on avoiding inflation. However, it is interesting to point out that the there is a global net gain of the taxation as the revenue from the taxation is always higher than the reduction in private

Table 6.1: Economic and environmental variables of the new taxation

Aggregated indicators	Case I*	Case II**	Case III***
Consumption price change (P_c), %	0.19	0.06	0.07
Private welfare change (ΔW), millions €	-1,348	-318	-140
Total revenue, millions €	2,467	603	914
CO_2 emissions , %			
Constant production	-2.00	-0.63	-0.43
10% reduction in production	-11.8	-10.56	-10.39
10% increase in production	7.8	9.23	9.53

Notes: * A tax rate of 6.1% is applied on Production and distribution of Electricity, ** A 5.4% tax rate is introduced on Manufacture of gas, distribution of gaseous fuels through mains, steam and hot water supply, and *** A 4.3% tax rate on Manufacture of non-metallic mineral is applied

welfare. Consequently, there is the possibility of using the public revenues to compensate the tax distortions caused on the rest of the economic agents. As illustrated in table 2, for example, a 6.1% tax rate on electricity sector raises revenue that could reduce income tax by 5% or corporate tax by 6%. The CO_2 tax in gaseous fuels could not raise enough revenue to compensate the tax burden on individuals compared with other two cases.

Table 6.2: CO_2 tax revenues as a share of income and corporate tax

	Revenue, millions €	Total Income tax, %	Corporate tax, %
Case I	2,467	5.1	5.5
Case II	603	1.2	1.4
Case III	914	1.9	2.0

Notes: Own elaboration based on INE data on income and corporate taxes for 2007.

These results are very interesting for policy makers, as they allow an analysis of trade-offs in different scenarios. For instance, if the tax is applied on gaseous fuels rather than on electricity, the social impact is 4 times lower meanwhile the CO_2 reduction is only 3 times lower, i.e., better CO_2 emissions reduction can be achieved with lower social effects. Probably the best case is applying the environmental tax on the non-metallic minerals (case III) when almost half of revenues and 25% of CO_2 emissions reduction are obtained with only 10% of social impacts of case I, when the tax is levied on Production and distribution of electricity.

To the best of our knowledge there are no similar studies which apply the price model to assess the impact of carbon tax specifically levied on electricity production, manufacturing of gas and other non-metallic minerals of the Spanish sectors. However, similar approaches were used to assess effects of alternative policies in Spain. Labandeira and Labeaga (2002) estimated the energy related CO_2 emissions of sectors in the Spanish economy in the year 1992. Based on the emissions intensity, they defined the hypothetical carbon taxes on fossil fuel consumption and calculated the associated price effects. They considered three different emissions tax rates, namely, the Pigouvian rate, the carbon budget (emissions caps) and the actual damage cost (shadow price). In line with this study, the results of Labandeira and Labeaga (2002) show that sectors such as Coal mining, Electricity, Natural gas, Oil processing, Manufactured gas and Cement are among the most affected sectors which exhibit price increases between 12.1%-4.1% when a tax rate of 34.8 US$/ton carbon (the rate proposed by the European Commission for the year 1998 as upper estimate) is applied. Llop and Pié (2008) also examined the consequences of a tax levied on intermediate energy uses in the Catalan economy by applying the input-output price model. They simulated the economic impact of three alternative scenarios on energetic activities, namely a 10% tax on energy uses, a 10% reduction in energy uses and finally a combined measure, a 10% tax and a 10% reduction in energy use. A 10% increase in the tax of energy induces relatively high changes in production prices of sectors such as Electrical energy, gas and water; Energy products, minerals, coke, petroleum and fuels; and Other non-metallic mineral products, which exhibit price changes of 4.7%, 3.8% and 1.8%, respectively. As a consequence of the change in production prices, the consumer price increase, the private welfare is also negatively affected, leading to a reduction in intermediate demand for energy and in pollutant emissions.

Despite the spatial and temporal differences in the related literature, the estimated effects of applying an environmental tax are in line with the results we have obtained in this paper. In all our simulations, the sectors showing high production price changes coincide with the ones in the previous studies. The interesting feature offered by the modeling framework discussed in this paper is the possibility to clearly visualise and analyse trade-offs between economic, environmental and social effects of environmental taxation. Furthermore, rather than discussing new types of environmental taxes, this framework uses the emissions prices established by the EU ETS, scheme already implemented in the EU. The framework presented here is flexible enough to allow simulations with different emissions prices and also to focus on specific sectors within a national economy.

6.4 Conclusion

In this work we have implemented both an EIO and a Leontief price model to investigate the potential impact of a CO_2 tax in the Spanish economy. We implemented EIO model to identify the three highest emission intensive sectors that are chosen to apply the environmental tax. The economic impacts of taxation, such as its impacts on production prices, private welfare and public revenues and its effect on the environment through the associated reduction in CO_2 emissions, were analysed from the use of the Leontief price model.

We have considered three illustrative cases. The first one consists of the application of an environmental tax on electricity production and distribution sector. The second case is based on a tax on Manufactures of gas, distribution of gaseous fuels through mains, steam and hot water supply and, finally, a tax applied on the Manufacture of other non-metallic mineral.

The results from the empirical analysis show that there is an unavoidable trade-off among the society, the environment and the economy. For example, a tax on the production and distribution of electricity would to some extent improve the environment by reducing the total CO_2 emissions by 2%, while it would negatively affect the social welfare. Our results suggest that the environmental taxation based on CO_2 emissions could not simultaneously ensure the environmental, economic and social goals. This implies the necessity of looking for complementary and alternative measures to environmental taxation in order to achieve not only environmental advantages but also to ensure economic and social rewards. A further analysis of trade-offs is also important in order to choose the optimum taxation for the best emissions reduction, the highest generated revenues, meanwhile assuring minimum impacts on public welfare. One of the challenges in putting such policies into action is how to minimise the distortional effects on the economy and society and how to avoid the (possible) trade-off with the environment. This is the most important point that any environmental taxation needs to address in order to enhance the effectiveness of the environmental policy. Therefore, the principal objective that any environmental measure needs to achieve is a win-win strategy for environment, economy and society. The trade-off between the environment and the society could be counter balanced by using the revenue from the tax to compensate those who are negatively affected. Likewise, a better improvement of the environment could also be achieved from the use of part of the revenue in stimulating cleaner production by financing innovations to replace non-renewable energy sources in the energy mix. This also makes an environmental tax a worthwhile policy in climate change mitigation.

The approach discussed here has important policy implications. It can be used in understanding the potential effects of a certain level of CO_2 price in achieving the targets through direct emissions reduction attained from the change in output induced by the tax or through the use of parts of the revenues to stimulate cleaner technology. However, the most challenging issue is identifying an effective level of CO_2 tax that could influence both produces and consumers to alter their behaviour through its price signal. The presented case study could be a showcase as it can be used as a pioneer in examining the effectiveness of the already existing polices. This could be explained by considering the current situation of EU ETS, one of the largest emissions trading scheme in the world to combat climate change.

Though EU seemed to be very close to meeting its emissions commitment of 2020, the EU ETS is now facing a huge challenge as a result of significant CO_2 market price drop in 2012 to around 5€/ton from 20€/ton in 2007. The main reason behind is the growing surplus of emissions allowance over the last few years coupled with economic slow down. According to our simulation, the potential effects of the environmental tax would be less than a factor of quarter if the current price is considered to be the tax rate. This may not ensure the necessary improvement both in the environment and in the economy, and questions the role of EU ETS as milestone EU's climate policy to achieve its emissions target. However, recently the EU is planning to intervene on the carbon market by proposing short-term and long-term options. The former consists of delaying the number of allowance in the swollen Emissions Trading System whereas the later includes setting of a more ambitious emissions reduction target, 30% GHG emissions reduction for 2020. Therefore, studies in the line presented in this paper can play a vital role in identifying important sectors where the emissions are concentrated and in setting the appropriate level of the environmental tax or allowance to each sector in an economy.

Finally, it is important to highlight that this study is limited to the same technology assumption. In all cases, imported goods both as intermediate and final demand sides are considered to be produced with the same technology as corresponding goods in Spain. As a result of such assumption, it is not possible to separately estimate the exact emissions contribution of domestic and imported goods. More importantly, when goods are imported from countries where the production technology and energy mix are hugely different from the domestic region (Spain), large error could possibly be produced. This could be avoided through the use of MRIO model. The next chapter will describe the methodological background and the application of 13 region MRIO model for the Spanish economy. Unlike the Single-region input-output models discussed so far, the MRIO allows to differentiating the pollution flows as a result of international trade between Spain and its most important trading partner.

Chapter 7

CO_2 EMISSIONS EMBODID IN INTERNATIONAL TRADE: A MULTIREGIONAL INPUT-OUTPUT MODEL FOR SPAIN

7.1 Introduction

Concerns on economic and social consequences of climate change are continuing to grow as there are scientific evidences on its potential impact (Stern, 2006; Peters and Hertwich, 2006a; IPCC, 2007). The Kyoto Protocol is an international measure adopted since 1997 in an attempt to mitigate the potential cost of climate change by committing the countries that have ratified to reduce their GHG emissions by 5% below 1990 levels in the first commitment period of 2008-2012. However, the Protocol is often criticized for creating carbon leakage in two ways. First, as it only commits a subgroup of high income countries and it does not impose mandate on developing countries that are economically emerging and dominating the global greenhouse gas emissions, there is a shifting of energy and emission intensive goods to be produced in countries that are not ratified (strong carbon leakage). Second, the responsibility to committing countries is only from production perspective, in a way that only emissions occurred within the national territories are considered. This totally ignores emissions embodied in imports, which is referred as weak carbon leakage. While the international trade is enormously growing due to free trade agreements, its indubitable effect on the environment (Antweiler et al., 2001; Machado et al., 2001; Copeland and Taylor, 2005) and the constraint of already existing global measures for climate change mitigation in capturing emissions associated with trade flow (Peters and Hertwich, 2006b, 2008; Lin and Sun, 2010) have been speculated for long time both by environmental and economic analysts.

This chapter investigates the significance of international trade between Spain and its key trading partners with respect to CO_2 emissions embodiment. The model applied is based on input-output framework (Leontief, 1941; Miller and Blair, 2009). EIO is a top-down approach which accounts for resource consumption and emissions release using generic data, IO tables that present the interrelationship of all industries in an economy (Leontief, 1970). EIO has been used for a long while to estimate pollution embodiment in international trade. In early 1970s, Walter (1973) applied EIO models to examine the US product profile of exports and imports and their environmental profiles. But Fieleke (1975) was the first who implemented the Leontief inverse in determining the US trade deficit in embodied energy. Since then a number of studies have been carried out using an EIO approach to analyze the environmental implication of international trades. Most studies have implemented the single-regional input-output approach (SRIO), which is usually based on the very simplified assumption of the same technology both for imported and domestic products (Wyckoff and Roop, 1994; Kondo et al., 1998; Lenzen, 1998; Machado et al., 2001; Sánchez-Chóliz and Duarte, 2004). However, such assumption is far from the reality, particularly when there are high discrepancies both in technology and energy mixes between the trading partners, and therefore, it may be subjected to large errors as estimated by Lenzen (2004) and Peters and Hertwich (2006c). Detailed review on the use of SRIO models for international trade and emissions analysis can be found elsewhere (Wiedmann et al., 2007; Wiedmann, 2009).

On the premises of avoiding errors due to same technology assumption in SRIO analysis, the development of MRIO approach emerged as the best alternative in environmental analysis associated with international trade. Unlike the SRIO, a complete MRIO model differentiates the production technology and then the related energy and environmental profile of imported goods and services from domestic ones. MRIO model fully integrates the domestic requirement matrix with imports, which is derived from international trade flows to simulate the interdependency of sectors in one region with all other sectors in trading partners. Hence, it allows seeing the entire supply chain of trades and emissions flows linked with goods and services imported to or exported from domestic region. Though the application of MRIO model dates back to mid 19's, it was only recently that different works have emerged applying MRIO models in the analysis of emissions embodied in trade, to cite but a few: (Dabo and Hubacek, 2007; Davis and Caldeira, 2010; Lenzen, 2004; Nijdam et al., 2005; Peters and Hertwich, 2006c,b,d; Hertwich and Peters, 2009; Peters et al., 2011; Wiedmann, 2009).

Particular to the Spanish case, there are few studies which analyze the environmental implication of international trade between Spain and rest of the world. Sánchez-Chóliz and Duarte (2004) used a SRIO model to study the potential impact of international trade in the level of CO_2 emissions generated by the economy for the year 1995. The study concluded that the pollution imported through the intermediate and final demand requirement of the Spanish economy is off-set by the pollution exported to satisfy demands outside Spain, leaving the net trade balance of only 4237 thousand ton (1.3% of the total emissions produced in Spain). More recently, Cadarso et al. (2012) analyzed the impact of international trade and the shared responsibility in the Spanish economy for the period 2000-2005 using SRIO model. In this study, they defined a set of criteria to share the responsibility of sectors for their direct emissions due to production as well as due to their input requirements. The responsibility was shared to all sectors along the global production chain depending on the value added on each step.

With regard to MRIO model application, in what we know, smaller number of studies have been conducted. Serrano and Dietzenbacher (2010) is the only work found by the authors in a peer-reviewed English-language journal. They developed a two regions (Spain and rest of the world) MRIO model to examine the emissions responsibility of the Spanish economy due to international trade taking into account both the net trade balance and responsibility balance concepts. They considered the years 1995 and 2000 and evaluated the effect for nine different types of gases. They concluded that both concepts (net trade balance and responsibility balance) yield the same results. Though not yet published in any peer-reviewed English-language journal, there are few studies that apply MRIO models to the Spanish case (Guadalupe et al., 2012; Navarro, 2012; Navarro and Madrid, 2012).

All the previous studies that analyze the effects of international trade on CO_2 emissions for Spain are either restricted to only two regions or inter-regional applications. However, a full MRIO model in the Spanish economy has not been yet constructed. Therefore there is a need to develop such a complete model that looks at a more broaden scope, from only two regions analysis to almost all trading partners. Applying 13 region multi-directional MRIO model, our study aims at analyzing CO_2 emissions stimulated due to international trade flow of the Spanish economy. The MRIO model allows us to understand the link between demand on production of goods and services among the regions and their environmental consequences (in terms of CO_2 emissions). Therefore, this study addresses the following questions. How would the Spanish economy be seen in terms of trade balance based on its consumption and production structure? Which partners have the most significant contribution in emissions due to trading with Spain? What are the key products responsible for the most of emissions embodied in imports? How the results from MRIO model would be used in CO_2 emissions reduction strategies by policy makers?

The chapter is structured as follows. Section 7.2 presents the methodological foundations of MRIO, data sources and the main assumptions considered. The main results and their policy implications are discussed in Section 7.3. Section 7.4 concludes the chapter.

7.2 MRIO Modle Description and Database

MRIO models are recognized to be a suitable tool to analyze emission embodied in trade both from consumption and production perspectives. They are able to trace the emissions linked with trades as a result of demand and supply interdependency of industries in domestic economy with foreign agents. In this section we present the construction of 13 region environmental MRIO model for the Spanish economy based on Peters and Hertwich (2009). Assuming that there are n regions, in which each region's production is classified into m sectors, the MRIO model can be formulated using the traditional IO framework as follows:

$$\boldsymbol{\chi} = \boldsymbol{\delta}\boldsymbol{\chi} + \boldsymbol{\gamma} \tag{7.1}$$

where $\boldsymbol{\chi}$ is a column vector of total output, $\boldsymbol{\chi} = \begin{bmatrix} \boldsymbol{\chi_1} \\ \vdots \\ \boldsymbol{\chi_n} \end{bmatrix}$, in which each element $\boldsymbol{\chi_i}$ represents the sectoral total output vector of region i. $\boldsymbol{\delta}$ refers to the total inter-industry requirement matrix, which integrates both the domestic and imported inputs for each sector in each region. It can be described as: $\boldsymbol{\delta} = \begin{bmatrix} \boldsymbol{\delta_{11}} & \cdots & \boldsymbol{\delta_{1n}} \\ \vdots & \ddots & \vdots \\ \boldsymbol{\delta_{n1}} & \cdots & \boldsymbol{\delta_{nn}} \end{bmatrix}$. Each $\boldsymbol{\delta_{ii}}$ in $\boldsymbol{\delta}$ is an mxm matrix of technology requirements on domestic production in region i, and $\boldsymbol{\delta_{ij}}$ (mxm matrix) denotes the inter-industry technological requirement from region i to region j. m refers to the total number of industries in each region. The total inter-industry technology requirement of region i can be equated from the column sum of $\boldsymbol{\delta}$ as $\sum_i \boldsymbol{\delta_{ij}}$. $\boldsymbol{\gamma}$ is the vector of total final demands which comprises both demand on domestic production and exports. It can be represented as: $\boldsymbol{\gamma} = \begin{bmatrix} \boldsymbol{\gamma_{11}} + \boldsymbol{\gamma_1^{ex}} \\ \boldsymbol{\gamma_{21}} \\ \vdots \\ \boldsymbol{\gamma_{n1}} \end{bmatrix}$. $\boldsymbol{\gamma_{11}}$ is an mx1 vector of final demand requirements on domestic production by domestic consumers and $\boldsymbol{\gamma_1^{ex}}$ represents the demand by foreign consumers. $\boldsymbol{\gamma_{ij}}$ stands for the final demand flows from region i to region j.

The above MRIO model can be written as:

$$\boldsymbol{\chi} = (\boldsymbol{I} - \boldsymbol{\delta})^{-1}\boldsymbol{\gamma} \tag{7.2}$$

where $\boldsymbol{I}$ is the identity matrix of the required size. An environmental extension of the model in Equation 7.2 can be expressed as [1]

$$\boldsymbol{\eta} = \boldsymbol{\xi}'(\boldsymbol{I} - \boldsymbol{\delta})^{-1}\boldsymbol{\gamma} \tag{7.3}$$

where $\boldsymbol{\eta}$ is the total environmental emissions associated with the production of $\boldsymbol{\gamma}$

[1] The symbol $'$ denotes transpose

output. $\boldsymbol{\xi} = \begin{bmatrix} \boldsymbol{\xi_1} \\ \vdots \\ \boldsymbol{\xi_n} \end{bmatrix}$ is a vector of direct emission intensities, in which each element $\boldsymbol{\xi_i}$ represents the vector of emission intensities of region i, where each entry is the emission intensity of a particular sector.

The MRIO model allows us to explicitly estimate the environmental impacts associated with domestic and import production for final consumption of goods and services. Emissions associated with total production in the domestic region($i = 1$) can be calculated as:

$$\boldsymbol{\eta_1} = \boldsymbol{\xi_1^{'}\delta_{11}\chi_1} + \boldsymbol{\xi_1^{'}}\gamma_{\mathbf{11}} + \boldsymbol{\xi_1^{'}}\sum_{j\neq 1}(\boldsymbol{\delta_{1j}\chi_j} + \boldsymbol{\gamma_1^{ex}}) \tag{7.4}$$

where $\boldsymbol{\xi_1^{'}\delta_{11}\chi_1} + \boldsymbol{\xi_1^{'}}\gamma_{\mathbf{11}}$ calculates emissions due to the production of outputs for domestic economy, both for intermediate and final demand requirements and $\boldsymbol{\xi_1^{'}}\sum_{j\neq 1}(\boldsymbol{\delta_{1j}\chi_j}+\boldsymbol{\gamma_1^{ex}})$ are emissions due domestic outputs production for both intermediate and final demand required by sectors and consumers in overseas economy.

Similarly, the emissions due to imported goods and services from region $i \neq 1$ to region 1 is estimated as:

$$\boldsymbol{\eta_i} = \boldsymbol{\xi_i^{'}\delta_{ii}\chi_i} + \boldsymbol{\xi_i^{'}}\gamma_{\boldsymbol{i1}} + \boldsymbol{\xi_i^{'}}\sum_{j\neq i}(\boldsymbol{\delta_{ij}\chi_j} + \boldsymbol{\gamma_{i1}}) \tag{7.5}$$

where $\sum_{j\neq i}(\boldsymbol{\delta_{ij}\chi_j} + \boldsymbol{\gamma_{i1}})$ comprises both the direct exports as final demand $(\boldsymbol{\gamma_{i1}})$ and intermediate demand $(\boldsymbol{\delta_{i1}\chi_1})$.

Production vs. Consumption based emissions

Generally there are two perspectives from which CO_2 emissions embodied in international trade could be estimated: the production and consumption perspectives. Production based emissions accounting, a method used in Kyoto Protocol, considers CO_2 emissions that are emitted during the production activities of domestic economy, without regarding where the produced goods and services are consumed, i.e. domestically or abroad. It includes emissions associated with the production of goods and services which are produced and consumed domestically and those which are exported to other markets. However, it does not take into account the emissions that occurred outside the national territory during the production process of goods and services which are consumed by domestic consumers.

Unlike the production based emissions accounting framework, the consumption based accounting is founded on the principle that consumers are responsible for all emissions associated with the production of goods and services which they are consuming regardless of the geographical location of production. It comprises emissions associated with all products which are produced and consumed domestically, products produced abroad but directly consumed within the economic boundary, and finally products which are produced abroad and supplied to industries as intermediate products and finally consumed domestically. This can be estimated as the sum of domestic emissions due to domestic final demand plus the imported emissions from all other regions to region 1:

The net emissions trade balance is a surplus or deficit emissions from the import and export, which is equal to the difference between the production and consumption approaches Rodrigues et al. (2009); Serrano and Dietzenbacher (2010).

Data

The 13 region MRIO tables are constructed through linking both the IO tables and bilateral trade data provided by OECD. The OECD is one supplier of consistent and harmonized IO tables that can be used for international trade and environmental analysis. The latest OECD IO data set covers inter-industrial transactions of goods and services of 48 countries (all OECD countries but Iceland and 15 non-member countries). It provides both the domestic and import tables separately for the years 1995, 2000 and 2005 in millions USD at basic prices. The data for the year 2005 were considered in this analysis. The IO tables and the bilateral trade data are in accordance with a harmonized industry structure of the ISIC[1] Rev.3 (the industries and their ISIC correspondences are listed in Table A.3). It is aggregated to 37 sectors (Table A.3). The OECD bilateral trade database provides monetary values of imports and exports of goods and services broken down by industrial sectors and by end-use categories (Zhu et al., 2011). Each off-diagonal block matrix $\boldsymbol{\delta_{ij}}$, which represents the import requirements from region i to region j are derived from the bilateral trade data and total output vector of region j. The bilateral data from OECD is only for intermediate goods and services; it doesn't include imported final demands. Therefore, the vector of imported final goods, $\boldsymbol{\gamma_{ij}}$ elements of total final demand vector $\boldsymbol{\gamma}$ in equation 7.2 are derived based on the assumption that imported final demands are in the same proportion as imported intermediate demands from the trading partners using the following equation [2].

$$\boldsymbol{\gamma_{i1}} = \widehat{\boldsymbol{M_{ij}}}\boldsymbol{\gamma_{import}} \tag{7.6}$$

where $\boldsymbol{\gamma_{import}}$ is imported final demand to Spain and $\boldsymbol{M_{ij}}$ is the share of import of each goods, which is calculated as:

$$\{\boldsymbol{M_{ij}}\}_c = \frac{\{\boldsymbol{m_{ij}}\}_c}{\left\{\sum_j \boldsymbol{m_{ij}}\right\}_c} \tag{7.7}$$

[1]ISIC stands for International Standard Industrial Classification of all Economic Activities.
[2]The symbol ˆ accompanying a vector denotes the diagonalisation of the corresponding vector

where $\{\boldsymbol{m_{ij}}\}_c$ is the total import of intermediate good c from region i to domestic region 1 (Spain).

The data on CO_2 emissions were obtained from the World Input-Output Database (WIOD). WIOD is a project funded by the European Commissions, Research Directorate General as part of the 7^{th} Framework Programme. It provides a set of harmonized supply and use tables and extensive satellite accounts for 27 EU countries and 13 non-EU major world countries for the period from 1995 to 2009 (WIOD, 2012). The CO_2 emissions data of WIOD includes both energy related and non-energy emissions and they are separately reported, which allows to simulate the environmental implication of switching from one energy source to another (from gas to coal in energy sectors). In line with the OECD IO tables, the CO_2 emissions data of WIOD are classified based on ISIC Rev. 3 (refer the classification from A.4). The CO_2 data are used to derive the emission intensity vector in equation 7.2.

A multidirectional industry-by-industry MRIO model of the Spanish economy with its 12 important trading partners is constructed for the year 2005. Figure 7.1 illustrates the import and export structure of Spain with its trade partners for the year 2005. According to the data from INE (2012a), around 66% of the total imported goods are from the EU and 76% of the total exports are to the EU. Germany (DE), France (FR), Great Britain (GB), Italy (IT), the Netherlands (NL), Portugal (PT) and Belgium (BE) are among the most dominant partners of Spain both in imports and exports. All together represent around 55% and 63% of the total imports and exports, respectively. The Asian market represents 17% of imports and 7% of exports of the Spanish international trade. China (CN) and Japan (JP) are important partners of Spain, which together account for 8% of the total Spanish import. Around 9% of the total imports and 10% of exports from Spain are represented by America. US and Brazil (BR) are among the important partners.

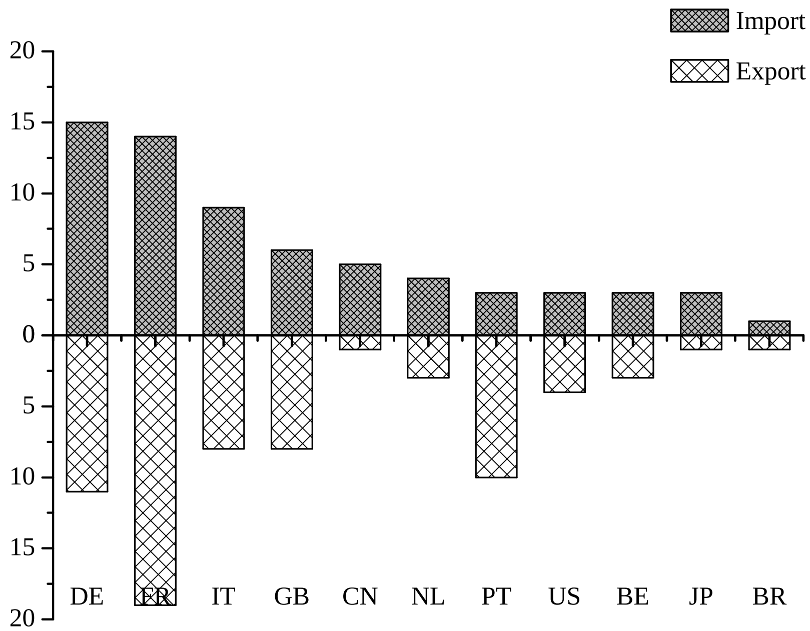

Figure 7.1: Import and export structure of Spain with its important trading partners (own elaboration based on data from INE data)

RAS method

In linking IO domestic table with the bilateral trade data, we have used the RAS procedure, an iterative method that updates IO tables. Due to unavoidable asymmetry problems, which may result from different trade systems definition, characterization of specific goods or transaction types, different valuation of imports and exports between countries, consideration of transit trades, re-exports and re-imports, and so on, there are some discrepancies between the values of imported goods by a given country and the corresponding exports from the other trade partner. As a result the total intermediate output of each sector does not match with the sum of its domestic and import outputs. Therefore, a balancing of the total intermediate requirement matrix ($\boldsymbol{\zeta}$) which is derived from the OECD domestic IO tables and bilateral trade data is required. Assuming that $\boldsymbol{\zeta}^*$ is the unknown target matrix, with the same dimension and structure of $\boldsymbol{\zeta}$, but only the column and row sum are known (from the total intermediate outputs and inputs of each country), the balanced matrix $\boldsymbol{\zeta}^{**}$, which has both similar dimensional and structural property of $\boldsymbol{\zeta}$, but the same margin as $\boldsymbol{\zeta}^{*}$ is obtained from the implementation of RAS method. RAS approach uses the margins of $\boldsymbol{\zeta}^{*}$ and the structure of $\boldsymbol{\zeta}$ to bi-proportionally adjust $\boldsymbol{\zeta}$ matrix and generate a balanced $\boldsymbol{\zeta}^{**}$ in such a way that it reflects both the property of $\boldsymbol{\zeta}$ with approximately similar margins of $\boldsymbol{\zeta}^{*}$ (Miller and Blair, 2009).

7.3 Results and Discussion

This section presents the main findings of the MRIO analysis for the year 2005. When the production base accounting framework is considered, a total of 292.5 Mton of CO_2 emissions occurred on the Spanish territory, 73.5% of which are emissions associated with the production of final demanded for domestic consumer and the 26.5% is due to demand by foreign market. Figure 7.2 shows the contribution of each sector to the total domestic CO_2 emissions linked with the production of both domestic and export final demand or in other word production responsibility. When the sectoral emissions intensities are analyzed, Electricity, gas and water supply; Other non-metallic mineral; and Coke, refined petroleum products and nuclear fuel are the most important sectors that together are responsible for around 68% of the total direct emissions occurred in Spain to provide other sectors energy and materials to produce final demand for both domestic and export consumption. In line with Alcántara and Padilla (2006), these are the key sectors that concentrate most of the emissions caused from the production perspective of the Spanish economy. As shown in the Figure 7.2 the emission intensity of the energy sector is by far higher than other key sectors. This shows how the outputs from this sector have great impact on the total CO_2 emissions generated in Spain. Emissions reduction measures in the sector such as increasing the share of renewable energy source in the national energy production mix, increasing the efficiency of energy generating plant by introducing more advanced technologies, switching from coal-powered turbine to natural gas or from single-cycle turbine to combined-cycle turbine and so on, can contribute to a decrease of total Spanish domestic emissions.

Different situation emerges when instead of emission intensities the total emissions, as a sum of direct and indirect, are considered instead of emission intensities. The total emissions express the emissions generated by the whole productive system in response to the production of final demand by the corresponding sector. When analyzed from the domestic demand side, the emissions concentrated in the key sectors are distributed among different sectors according to the final demand on each activity and their dependence on the key sectors to produce the demanded outputs. In this regard the construction sector of the Spanish economy is the most important sector, which relies on the other sectors in carrying out its activities. This implies that the sector buys pollution from other sectors through its input requirements to produce its final domestic demand for consumption. During its peak, in 2005, the production of energy and materials needed to satisfy the demand from the construction sector alone generated around 28% of the total emissions in Spain. Despite its burst after the crisis, from 2008, the sector is thought to be the significant contributor of the Spanish economy and was the most important factor linked with the economic boom in the country for almost a decade. Its high contribution to the total CO_2 emissions results from its tremendous economic growth. Next to Construction sector are Electricity, gas and water supply; Wholesale and retail trade, repairs; Hotels and restaurants; and Food products, beverages and tobacco, which together contribute to 37% of the total indirect CO_2 emissions. When analyzed from the exports demand side, Transport and storage, Other non-metallic mineral and Chemical and chemical products sectors generated an important amount of domestic emissions to provide goods internationally.

The consumption based emissions, all the emissions which are directly or indirectly produced elsewhere in order to provide goods and services consumed in Spain, are presented in Figure 7.3 for the same year 2005. According to the analysis, the Spanish economy is responsible for 378 Mton CO_2 emissions. The difference between the consumption and production based emissions reveals that

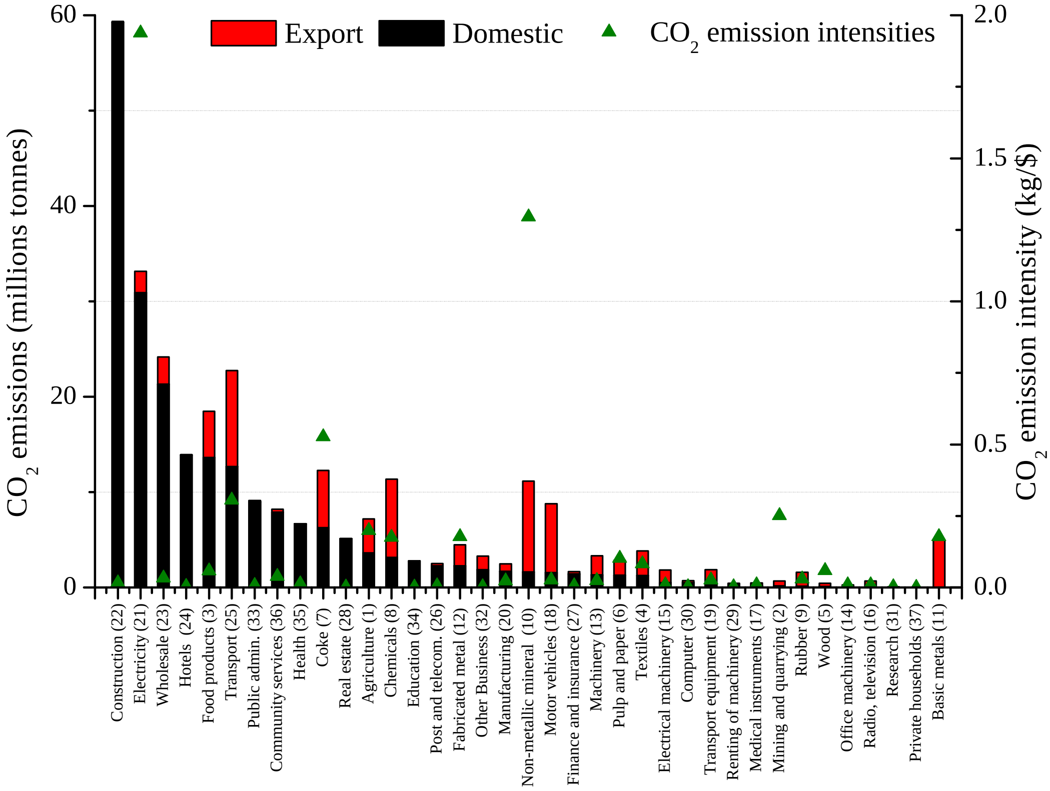

Figure 7.2: CO_2 emissions produced in Spain to supply both domestic and export demand, 2005

Spain was a net importer of emissions with a trade balance of 85.4 Mton of CO_2 emissions, which is approximately 29% of its production emissions. This reflects the significance of international trade in the context of CO_2 emissions and its implication for policy considerations. As can be seen from Figure 7.3 the largest portion of the emissions occurred in Spain is mainly to produce final goods or services for domestic consumption.

An important amount of CO_2 is released in China due to its export to Spain, of which 45% is due the supply of intermediate goods for the Spanish sectors and 40% is due to direct supply of final goods to consumers in Spain. The remaining 15% are due to the supply of intermediate goods for the most important trading partners of Spain, in order for them to produce export for Spain.

CO_2 emissions occurred in US due to consumption in Spain is also comparatively high. 52% of these emissions are due to supply of intermediate inputs for Spanish sectors to produce output for domestic consumption. Around 33% percent of the emissions are due to the production of final goods which are directly consumed by the Spanish consumers. The rest is due to import requirements of other regions to produce both intermediate and final goods for the Spanish economy. Though not as high as US and China, considerable amounts of CO_2 are also generated in Germany, Italy, Japan, France and UK.

A closer look to the import structure and emissions embodied in imported goods gives more insight on the key trading partners and their contribution to the global emissions derived from their exports to Spain. Figure 7.4 presents the

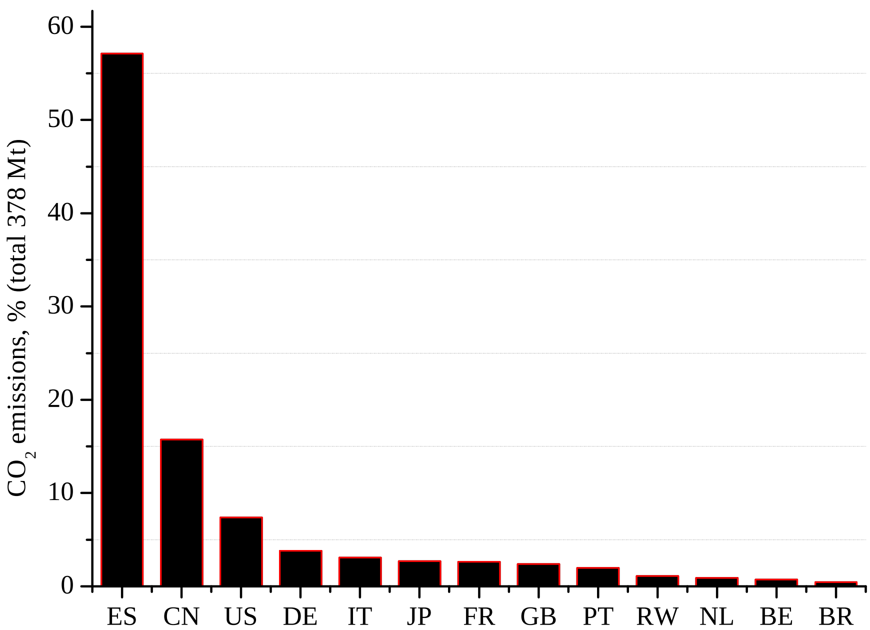

Figure 7.3: Consumption based emission by country where goods and services are produced, 2005

CO_2 emissions embodied in imported goods by country of origin. As shown, emissions associated with imported goods from China dominate to a great extent, which contributes to 36% of the total emissions due to imports. It is followed by US that accounts for 17%. Around 38% of the total embodied emissions are due to imports from the EU. When analyzed by end-use, the emissions due to the production of intermediate products that are reprocessed by domestic sectors before being supplied to domestic final demand account for 41% of the total emissions. Imported goods that are directly supplied to domestic final consumer represent 33%. Additionally the Spanish economy re-exports around 26% of the total imported emissions to other countries.

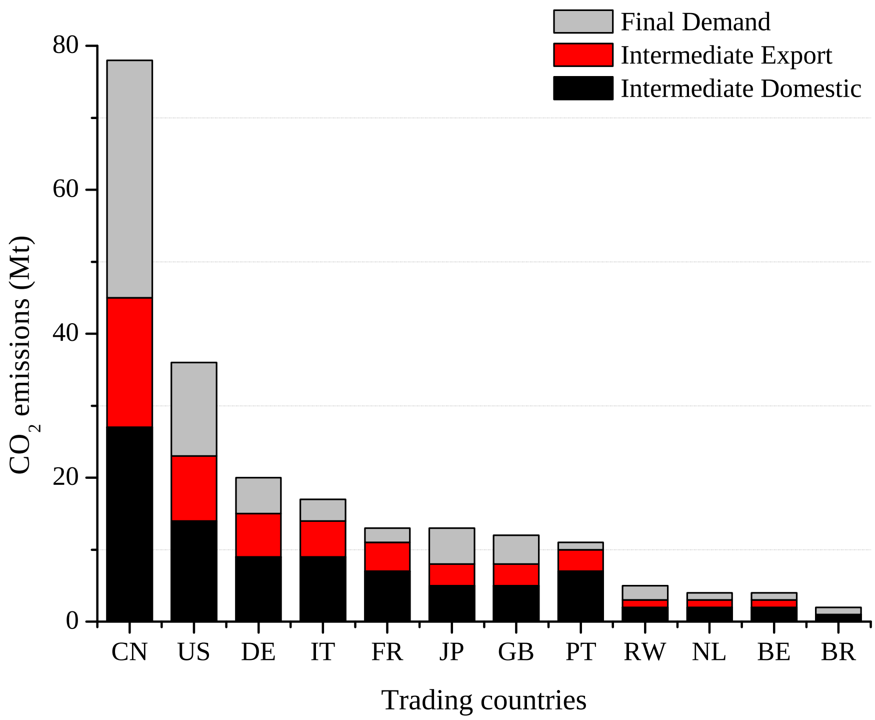

Figure 7.4: CO_2 emissions embodied in imported goods and services, 2005

The imported goods and services that dominate the total imported emissions from the selected countries are presented in Figure 7.5. As can be seen, sectors such as Motor vehicles, Chemicals, Machinery and equipment, Textile and leather products, Construction materials are the key imports that drive the emissions due to their production in the respective exporting countries.

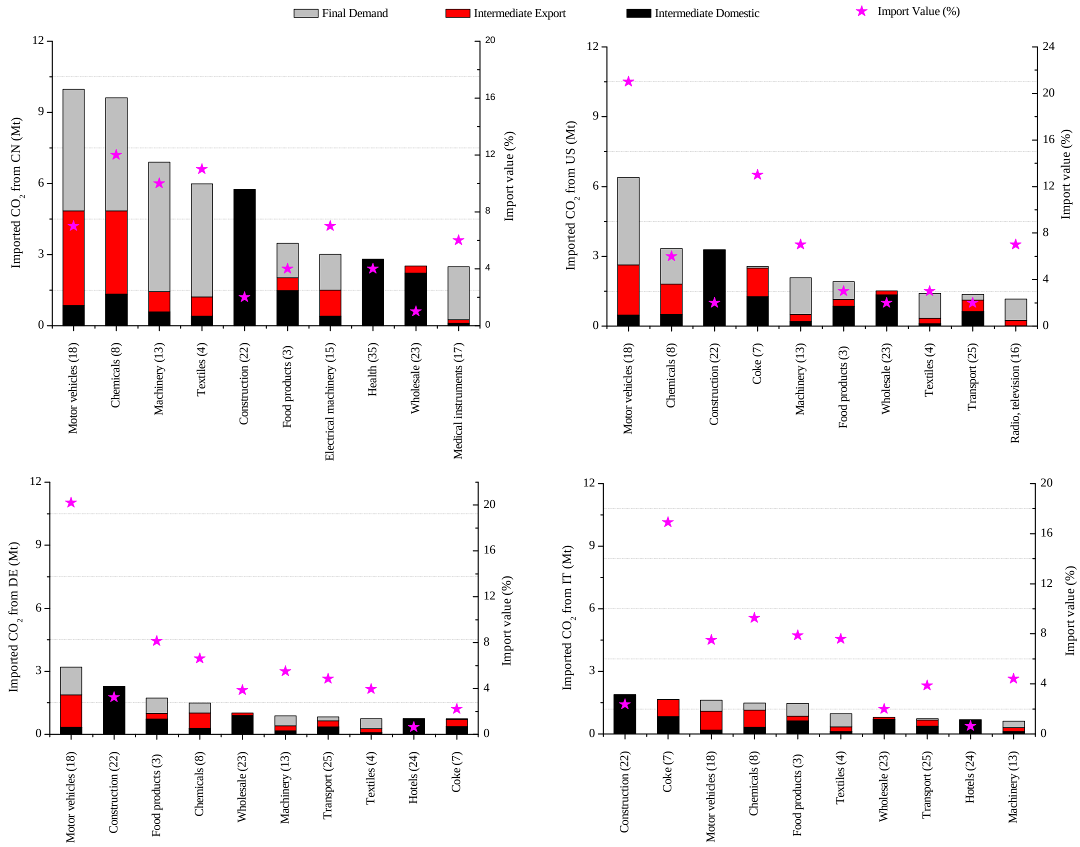

Figure 7.5: CO_2 emissions and import values of top 10 polluting products from selected countries, 2005

Some of these products are directly supplied as final goods to be consumed directly by the consumers, while others are imported as raw materials or intermediate products to undergo the Spanish production system before they are supplied to final consumers both for domestic and export destinations. As can be seen a considerable amount of emissions from the importation of motor vehicles and chemicals are re-exported to other countries. Additionally, emissions from the import of building and construction sector are finally consumed by the Spanish economy.

Figure 7.5 also illustrates that most goods and services imported from China have relatively low import value but high CO_2 emissions compared with other countries such as Germany, Italy and US. This could be explained as follows. Low price products might result from cheap labor market in China. Obviously, this could be one of the main reasons, but not the only one for having such a big divergence in import values and the associated emissions; there are also other facts behind. The relatively unclean energy mix of the Chinese economy plays a crucial role in this regard. Coal covers much of the demand in Chinese energy production and the share of other less CO_2 intensive energy sources is limited, as a result the CO_2 intensity of the Chinese energy sectors were noticeably high, especially compared with countries like Norway where hydroelectricity comprises the largest share (Peters and Hertwich, 2006c). This makes the Chinese energy sector to be the largest responsible sector which accounts for around 54% of the total direct emissions embodied in goods and services exported to Spain. The predominance of energy intensive sectors in China, which may lead to high energy demand, also highlights this fact. It is obvious that on one or another way, emissions associated with the production of goods and services from a given country are dependent on the national energy mix and the production technology. Most energy intensive product from counties like Germany, France, where the relative share of renewable energy source in the national mix is high, generally show less emissions as they incorporate low CO_2 per US$ output as can be referred from Table 7.1. As expected, the emissions intensities of most of Chinese sectors exhibit huge differences compared with Spain, unlike sectors in other trading partners. More importantly, the energy sectors in China exhibit a considerable variation compared with the corresponding sector in Spain. Sectors which heavily rely on energy sectors also disclose high intensities, as presented in Table 7.1. Therefore, the import and emissions profile of different sectors illustrated in Figure 7.5 reveals the fact that Spain is importing more energy intensive products from China than from other trading partners.

Figure 7.5 also suggests the shifting of energy and emissions intensive products to economically emerging countries such as China. A good example for this could be the emissions associated with motor vehicles imports. Representing only 7% of the total import value, vehicles from China [1] are responsible for around 10 Mt of embodied CO_2 emissions, whereas by far larger volume of the same product group imported from Germany shows quite smaller emissions values. Germany is one of the leading exporters of motor vehicles to Spain, which satisfies around 28% of the Spanish total demand on motor vehicles. Despite the fact that large value of motor vehicles are imported from Germany, small amount of direct emissions are associated with them. This suggests that the Germany economy imports energy intensive products such as different parts of vehicles; for example, from China, which are assembled and shipped to Spain.

[1] According to the ISIC rev.3 the Manufacture of motor vehicles, trailers and semi-trailers (sector 34) has three subgroups: Manufacture of motor vehicles; Manufacture of bodies (coachwork) for motor vehicles, manufacture of trailers and semi-trailers and Manufacture of parts and accessories for motor vehicles and their engines. Therefore, emissions associated with the final demand from China may not necessarily be motor vehicles, they could be parts or accessories of cars which are used for repairing personal vehicles, e.g. lights, screen wipers, wheels, etc. But the motor vehicles in the final demand could also be explained as the cars which are produced in Japan or somewhere else, but imported through China.

Table 7.1: Emission intensity, as the amount of CO_2 emitted to produce a unit output for final demand, kg per $

Industry	ES	BE	BR	CN	DE	FR	GB	IT	JP	NL	PT	US	RW
Agriculture, hunting, forestry and fishing	0.44	0.66	0.52	1.01	0.41	0.38	0.45	0.36	0.37	0.76	0.41	0.61	1.31
Mining and quarrying	0.73	0.58	0.61	2.69	0.86	0.57	0.51	0.44	2.64	0.46	0.97	0.61	0.88
Food products, beverages and tobacco	0.47	0.46	0.41	1.23	0.37	0.37	0.39	0.44	0.31	0.48	0.43	0.54	1.1
Textiles, textile products, leather and footwear	0.46	0.46	0.34	1.47	0.44	0.31	0.41	0.45	0.47	0.43	0.44	0.55	0.89
Wood and products of wood and cork	0.43	0.4	0.32	1.69	0.36	0.29	0.46	0.37	0.46	0.33	0.44	0.53	1.11
Pulp, paper, paper products, printing and publishing	0.47	0.47	0.43	1.91	0.35	0.32	0.33	0.45	0.35	0.34	0.53	0.42	1
Coke, refined petroleum products and nuclear fuel	1.15	0.8	0.79	2.5	0.95	0.95	1.13	1.16	0.98	1.1	1.21	1.44	2.55
Chemicals and chemical products	0.76	0.79	0.72	2.63	0.52	0.54	0.57	0.76	0.82	0.89	1.33	0.79	2.18
Rubber and plastics products	0.51	0.44	0.5	3.04	0.39	0.4	0.54	0.52	0.46	0.5	0.6	0.56	1.06
Other non-metallic mineral products	1.89	1.81	1.83	9.24	1.3	1.19	1.08	1.46	1.67	0.77	2.35	1.76	2.93
Basic metals	0.72	0.97	0.91	3.46	0.83	0.64	0.97	0.72	1.15	0.8	0.63	0.91	2.7
Fabricated metal products except machinery and equipment	0.63	0.83	0.85	3.45	0.66	0.52	0.75	0.51	0.82	0.72	0.48	0.77	1.6

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Industry	ES	BE	BR	CN	DE	FR	GB	IT	JP	NL	PT	US	RW
Machinery and equipment n.e.c	0.42	0.39	0.48	2.01	0.28	0.28	0.41	0.39	0.37	0.33	0.43	0.48	0.95
Office, accounting and computing machinery	0.35	0.29	0.33	1.46	0.23	0.28	0.3	0.33	0.48	0.43	0.26	0.42	0.79
Electrical machinery and apparatus n.e.c	0.54	0.38	0.44	1.69	0.29	0.33	0.41	0.41	0.46	0.48	0.42	0.42	0.9
Radio, television and communication equipment	0.39	0.27	0.4	1.3	0.22	0.25	0.33	0.26	0.36	0.36	0.29	0.22	0.74
Medical, precision and optical instruments	0.33	0.32	0.33	1.42	0.21	0.23	0.25	0.35	0.37	0.26	0.28	0.22	0.61
Motor vehicles, trailers and semi-trailers	0.48	0.47	0.45	1.8	0.35	0.37	0.48	0.46	0.38	0.4	0.43	0.55	0.71
Other transport equipment	0.42	0.34	0.39	1.77	0.31	0.29	0.41	0.42	0.44	0.34	0.32	0.38	0.8
Manufacturing n.e.c; recycling	0.4	0.48	0.35	1.34	0.27	0.42	0.41	0.35	0.49	0.27	0.39	0.34	1.07
Electricity, gas and water supply	2.74	2.13	0.6	12.38	3.36	0.88	2.47	2.16	2.09	2.77	3.09	5.85	8.07
Construction	0.42	0.43	0.39	2.38	0.31	0.27	0.25	0.33	0.4	0.32	0.63	0.36	0.78
Wholesale and retail trade; repairs	0.26	0.27	0.15	0.85	0.18	0.14	0.21	0.24	0.14	0.2	0.26	0.22	0.52
Hotels and restaurants	0.18	0.28	0.25	1.46	0.21	0.19	0.21	0.25	0.26	0.31	0.31	0.4	0.77
Transport and storage	0.65	0.72	0.8	1.95	0.53	0.47	0.72	0.47	0.58	0.8	0.81	0.98	1.21
Post and telecommunications	0.22	0.21	0.15	1.12	0.19	0.1	0.17	0.17	0.11	0.12	0.13	0.22	0.38
Finance and insurance	0.08	0.11	0.07	0.64	0.08	0.07	0.1	0.07	0.07	0.08	0.08	0.11	0.3
Real estate activities	0.08	0.09	0.02	0.44	0.06	0.03	0.05	0.04	0.04	0.07	0.08	0.24	1.39

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Industry	ES	BE	BR	CN	DE	FR	GB	IT	JP	NL	PT	US	RW
Renting of machinery and equipment	0.19	0.21	0.07	0.08	0.05	0.09	0.15	0.25	0.11	0.16	0.14	0.25	0.42
Computer and related activities	0.11	0.17	0.07	0.08	0.08	0.08	0.11	0.17	0.1	0.12	0.16	0.19	0.29
Research and development	0.19	0.24	0.07	1.27	0.16	0.17	0.16	0.16	0.2	0.22	0.14	0.22	0.59
Other Business Activities	0.18	0.18	0.17	1.2	0.09	0.09	0.11	0.17	0.11	0.16	0.18	0.19	0.49
Public admin. and defence; compulsory social security	0.16	0.11	0.14	0.92	0.12	0.1	0.18	0.12	0.25	0.19	0.22	0.37	0.4
Education	0.07	0.07	0.13	1.22	0.11	0.1	0.1	0.04	0.12	0.11	0.09	0.26	0.44
Health and social work	0.17	0.17	0.23	1.72	0.13	0.1	0.17	0.14	0.25	0.14	0.32	0.25	0.53
Other community, social and personal services	0.25	0.3	0.33	1.22	0.14	0.24	0.16	0.24	0.26	0.47	0.34	0.24	0.66
Private households with employed persons	0	0	0	0	0	0	0.02	0	0.22	0	0	0	0.19

7.4 Conclusion

In this study we analyze the CO_2 emissions generated as a result of international trade between Spain and its trading partners. For this purpose, we developed a 13 region multidirectional MRIO model that enables us to examine the carbon responsibility of Spain as the net emissions trade balance, by differentiating between the emissions resulted from domestic consumption and production.

The emissions balance shows that Spain is a net importer of emissions, which is estimated to be 29% of its domestic emissions for the year 2005. This implies that a large amount of emissions are embodied in Spanish imports than the emissions embodied in its exports. In line with other related studies in Spain, the findings in the paper reveal that the trade between Spain and China take the largest share of CO_2 emissions. The relatively unclean energy production mix of Chinese economy could also highlight the emissions embodied in each product that are imported from China, despite the most important partners in terms of values are Germany, France, Italy and Great Britain. The embodied emission from countries with similar production structure and environmental profile are mainly driven by the total volumes imported. Regarding to their import and export flow presented in Figure 1, the net emissions flow the largest portion of embodied emissions is due to the demand by production sectors, which accounts to the 67% to the total imported emissions. The Construction sector is the most responsible sector behind these emissions. The year 2005 was the peak for the remarkable growth of this sector in Spain. Deliveries of the service from the sector demands large amount of input materials and, accordingly, emissions. Emissions from the import of large values of products such as motor vehicles and spare parts, machinery and equipment, textile and textile products, leather and footwear, chemical products are also considerably high. High value but relatively low emission intensive products such as computers, office machinery and other electronic devices have very little contribution.

The international trade between Spain and rest of the world has important policy implications from a global perspective. The following policy implication could be drawn from the empirical results obtained. As it is clearly observed, countries that are economically highly emerging and dominating the global trade but not legally bounded under international emissions reduction agreements are the main drivers of embodied emissions due to their looser environmental regulation. This has been a reason for many developed and major contributor to global GHG emissions to refuse to ratify the Kyoto Protocol; the US and Canada could be a good example. They consider that such unbalanced treatment favours the production of goods in non-Annex I countries while unfairly affects the competitiveness of producers that face restrictions on their emissions. Such inequity issue of international CO_2 regulations could be avoided by implementing border taxes. Border taxes or border adjustments taxes are levies imposed on imported products from countries that are not yet involved in any international CO_2 reduction protocols. Environmental tax or tradable permit price are believed by many economists and environmentalists to be the most effective and efficient way to reduce human related CO_2 emissions. They provide incentives to polluting industries for reducing their emissions through market signals. As the EU already established permit price on "cap and trade" principle with the target of reducing GHG emissions to at least 20% below the 1990 level by the year 2020, most companies are under this restriction. The border tax on CO_2 therefore charges companies that import goods and services from countries outside the EU the same price as current CO_2 price based on their life cycle emissions embodied in their production. The MRIO approach presented here could be used as a useful tool to analyze the global emissions flow linked with imports and exports between Spain and its important trade partners and the key sectors or product groups that are behind the embodied emissions. Such analysis could allow assessing the environmental profile of imported goods in a move to establish a general emissions taxation system.

Chapter 8

CONCLUSIONS

This chapter summarizes the main conclusions drawn in each case study presented in the thesis.

Process-based LCA - the case of tissue paper production from VP vs. RWP (Chapter 3):

- The results from comparison of tissue paper production from VP and RWP reveal that it is important to look at the whole life cycle process when assessing the GHG emissions advantages of one product over the other. Looking only at the input and energy requirements of the manufacturing processes my lead to wrong conclusions as in the case of RWP where the on-site energy demand is higher that VP. Whereas, when the supply chain emissions are considered the global emissions from VP is higher than RWP.
- The transportation effort in the case of VP process is proven to be an important element that contributes to large emissions as compared with RWP. The import from South America is the most responsible in this regard. The emissions profile could reduce if the company looks for possible market in EU.
- The emission both from VP and RWP process would be lower than the estimated values if the company could implement CHP. This will help them increase the energy generation efficiency, thus insuring a reduction in GHG emissions. The on-site electricity production could possibly avoid the equivalent consumption from the national grid.

The comparison of LCA methods (Chapter 4):

- Despite the fact that EIO based LCA frameworks theoretically should yield a higher emissions value than process-based LCA, the results from this case study illustrate on the other way round. EIO approaches and more importantly IO-based hybrid models provided lower results than LCA process.
- Hence this research highlights the most important factors that contribute for the discrepancy of the results among process-based LCA, EIO and hybrid-IOLCA approaches. The aggregation of sectors and product groups, the

handling of international transport are the most crucial aspects. Furthermore, factors such as the adjustment of inflation in price transformation from 2010 to 2007, changing from purchaser price to basic price are likely to introduce errors.

- The preciseness and product specificity of process-based LCA or Tiered hybrid LCA are counterbalanced by their time and labor intensiveness. Therefore, the balance shifts towards IO-based hybrid or EIO-LCA as they relatively need less effort in linking product specific data into EIO framework to apply for economy-wide environmental policy analysis. A good example could be an application of environmental tax based on products' and services' emission intensities presented in Chapter 5.

Environmental tax application based on sectoral emission intensities (Chapter 5):

- The environmental tax definition based on sectoral emissions intensity shows that polluting sectors such as Production and distribution of electricity, Manufacture of gas and Manufacture of other non-metallic minerals are among the sectors that are highly affected by the tax.
- The methodological discrepancies discussed in Chapter 4 are also reflected in the environmental tax values. In the case of Manufacture of pulp, paper and paper products sector the tax rates are higher when calculated using the process-based LCA approach than either of the EIO approaches (sector and/or commodity).
- In terms of practicability, process-based LCA might not be the best approach to calculate product or industry emission intensities on which to base an economy-wide taxation. Therefore, for this purpose we would recommend the use of EIO based LCA approaches.
- The results of this thesis highlight the significance of considering non-CO_2 GHG emissions in environmental tax policies. Despite the fact that CO_2 is the most important GHG and it is in the spotlight of most climate change mitigation, there are also other gases with higher global warming potential that contribute to the greenhouse effects.

- Sectors such as the Mining of coal and lignite; extraction of peat, Agriculture, livestock and hunting and the Food sectors are among the sectors most responsible for the emission of a considerable amount of non-CO_2 GHGs in the Spanish economy. An environmental tax applied to these sectors would vary considerably if GHG emissions other than CO_2 were considered. Therefore, policy makers should be aware of the possible effects of any mitigation actions on the various different sectors.

- The methodology followed in defining the effective tax rate is based on quantity-oriented framework and it doesn't reflect the link between prices and quantities. Therefore, the model should be extended so as to capture the link between prices and quantities, on the one hand, and the link with the economy and the environment, on the other hand.

In Chapter 6 the EIO model presented in Chapter 5 is extended so as to assess the potential impact of environmental tax introduction on selected sectors. The following conclusions are drawing based on the empirical results:

- The results suggest that there is unavoidable trade-off among the society, the environment and the economy, as environmental taxation based on CO_2 emissions does not simultaneously ensure goals in each pillar of sustainability. This implies the necessity of looking for complementary and alternative measures to environmental taxation in order to achieve not only environmental advantages but also to ensure economic and social rewards.

- A further analysis of trade-offs is also important in order to choose the optimum taxation for the best emissions reduction, the highest generated revenues, meanwhile assuring minimum impacts on public welfare.

- The trade-off between the environment and the society could be counter balanced by using the revenues from the tax to compensate those who are negatively affected. Similarly, a better improvement of the environment could also be achieved from the use of part of the revenues in stimulating cleaner production by financing innovations to replace non-renewable energy sources in the energy mix.

- This approach has important policy implication as it can be used in understanding the potential effects a certain level of an environmental tax on the economy.

The following conclusions are drawn from the MRIO model case study presented in Chapter 7:

- The Spanish economy is proven to be a net importer of CO_2 which is equivalent to 29% of its domestic emissions. This implies that less emission intensive products are exported while the economy imports high energy and emission intensive products.
- The trade between China and Spain takes the largest share of imported CO_2 emissions. The disproportional emissions to import volumes from China is explained by its relatively poor energy production mix compared with Germany, France, Italy and Great Britain with which Spain has high import share but quite low emissions flow.
- The Spanish production sectors are the most responsible for the largest portion (67%) of CO_2 emissions embodied in trade. The Construction sector, at its peak year in 2005, is the key sector behind.
- Emissions from the import of large values of products such as motor vehicles and spare parts, machinery and equipment, textile and textile products, leather and footwear, chemical products are also considerably high. Whereas, high value but relatively low emission intensive products such as computers, office machinery and other electronic devices have very little contribution.
- The research highlights that there is a clear carbon leakage which resulted from the shifting of energy and emissions intensive products to economically emerging countries like China. And this could possibly be avoided either by the implementation of a strict international measures beyond the Kyoto Protocol or by introducing a border adjustment tax.
- The MRIO approach presented here could be used as a useful tool to analyze the global emissions flow linked with imports and exports between Spain

and its important trade partners and the key sectors or product groups that are behind the embodied emissions. Such analysis could allow assessing the environmental profile of imported goods in a move to establish a general emissions taxation system.

Chapter 9

GENERAL DISCUSSION

9.1 Policy Implication of The Thesis

It has been almost 15 years since the first landmark international policy agreement to limit GHG emissions has been negotiated. The Protocol principally includes an extensive target, limiting the global GHG emissions 5% below the level in 1990 by 2012. However, it fell far short since only 37 industrialized countries committed to it. Economically emerging countries like China, India and Brazil are not ratified due to economic reasons while they are among the leading countries in GHG emissions. China is the first and India is the third largest GHG emitter. Recent trend on the global emissions reveals that in 2011 there was around 49% increase of CO_2 emissions from the level in 1990. Spain is also among those contributed to GHG emissions increase, from 1990 it's emissions increasing by 30% in 2011.

The current Kyoto Protocol expired in 2012, but EU pledged to extend the emissions reduction target under the Protocol by setting a new commitment period that would span from 2013 to 2017. Hence, the Spanish economy needs to establish a stronger environmental policy measure to enhance its emissions reduction and achieve its target beyond the 2012 goal. Implementing an environmental environmental tax, as proposed in the thesis, could be possible approach. The price signal on polluting goods could ensure emission reduction. The emission reductions could be achieved not only achieved from the direct change in consumption due to price effect, but also from the use of parts of the revenues generated to stimulate the use of cleaner technologies.

The results of this thesis show that there is a clear trade-off between the environment, economy and society. This means that environmental and economic goals cannot be met at the same time with the environmental taxation. While environmental taxation may reduce pollution in some sectors, it will have negative effects on the population. For compensating for these negative effects, this thesis suggests using revenues from tax implementation to compensate the most affected by the tax. Other possible use of revenues from taxation may be for rewarding the companies which reduce their GHG emissions, e.g. Subventions, tax reduction, etc., In order for this to be possible, a clear reporting of impacts and measures taken by companies to decrease them should be in place. Although this is not yet reality, methods to help companies to measure their impacts in terms of GHG and other environmental impacts are already available. This thesis shows the applicability of LCA for decision making for GHG emissions reduction. Although here LCA was applied to a specific tissue paper company, it can be applied to any producing activity. Besides compiling GHG across the supply chain, an LCA approach provides also insights on the most polluting stages of making a product. As usually the most polluting operations are also the ones involving bigger costs, LCA offers the opportunity to reduce pollution at the sale time with driving costs down.

Defining the CO_2 price that could achieve an emission reduction target is an important stage to implement an economy-wide environmental taxation. In this line the thesis contributes to understanding how a certain level of tax on selected sectors potentially induces effects on the economy. It also highlights that emissions reduction could be achieved directly from change in outputs resulted from price change or indirectly from the use of the generated revenue in cleaner production projects.

In the same line of wider political implications, this thesis analyses international trade and flows of pollution to and from Spain. It highlights the high pollution embedded in the products imported from China. If pollution was taxed, then these imported products would have higher prices, making more competitive local markets.

I think that the presented thesis could be a showcase for policy makers in Spain, as it helps them adopt such a taxation scheme into practice. As suggested this will give an opportunity to reduce an existing tax on household income while improving the environment. Even though the case study is based on the data from Spain, it could also be applied to other counties provided that there are available data on economic transactions and sectoral emissions profiles.

9.2 Basic Assumptions and Limitations

The EIO approaches discussed in the thesis are based on various assumptions and it is worth mentioning some of them and the associated limitations. One of the issues that I would like to point out is the assumption in which both quantity and price input-output models is based on. This relates to fixed proportion and constant returns to scale assumptions. These imply that any change in total output of a given sector induces changes in inputs in the same proportion to the outputs, which keeps technology matrix constant. The assumptions do not allow to reflect changes in the productive structure or to reflect improvements in technologies, whether such improvements result from increased efficiency or a switch to less emission intensive inputs. Consequently, the efforts of industries which shift from high emission intensity consumption to low emission intensity inputs for production (e.g. a shift from VP to RWP) would not be considered. Therefore, the EIO methods presented here only reflect the short-term effects of an environmental tax on price and the results should be interpreted taking this temporal reference into consideration. The constraint in relation with the long-term effects could be resolved in the future by developing a non-linear and dynamic IO model. Such model allows the production function to exhibit substitution between intermediate inputs. This makes it possible the use of the price model for long-term impact analysis. EIO models in Chapter 4, 5, 6 and 7 and the price model in Chapter 6 limit the analysis to a short-term effect and are not appropriate for long-term scenarios.

9.3 Uncertainties

Both LCA and EIO models are subject to uncertainties which may arise from different sources. One of the most important sources of uncertainty in the case of EIO is due to sectoral aggregation. The Spanish IO table aggregates all the industries and products into 73 sectors and 118 commodities assuming that each sector produces only one type of commodity, which ignores the heterogeneity of products among industries. This could result in underestimation or overestimation of environmental profile of a specific product as it leads to allocation errors. For example, in Spain, the manufacture of pulp, paper and paper board produces outputs such as pulp (chemical or mechanical), paper and paperboard (newsprint, handmade, toilet or tissue, container board, and so on), articles of paper and paper boards (corrugated board, sacks and bags of paper, and so on). These products cannot be homogeneous since each one requires different inputs which determine its environmental profiles. However, the EIO approach averages the input requirements and emissions release of all products to assign a single value. The uncertainty could be lower either if there is no significant discrepancy in the technological requirements and emissions profiles among the heterogeneous products or if the product of interest is the most representative output of the sector.

Other important cause of uncertainty in the EIO models is the assumption of the same technology both for domestic and imported products. However, in reality it is likely that products produced in foreign industries can have different input factors than those produced domestically. This may result in introducing large amount of error, especially if import input is not well represented by the output from the corresponding domestic industry. For example, in our case study, one of the important inputs is paper pulp which is imported from foreign industries. The output from the Spanish pulp, paper and paperboard is largely dominated by paper and paperboard since pulp is mostly supplied by foreign market. Here by assuming the same technology, there are two causes that may lead to high uncertainties. One is whether the corresponding pulp supplying sector is classified under the same industry category of the domestic economy or not. In addition to the classification variation, the other cause may be from the discrepancies in energy mix, which to a great extent affects the emissions factor.

There are also uncertainties associated with the data used in the EIO models. For example, the latest data, both for the IO matrix and sectoral emissions factor, are for the year 2007. Therefore, it is vital to take into account this. The pulp, paper and paperboard could be considered as a relatively stable sector in which the technological requirements are not rapidly changing in years. However, it is important to bear in mind while dealing with other sectors which are likely to be sensitive to time due to change in technology.

Besides, uncertainty could also be arisen from the measurement and reporting in the original data, due to the proportionality assumption of monetary and physical transactions and so on.

With regard to process-based LCA, the following could be the possible sources of uncertainties that need to be taken into account: uncertainty which may result from missing or unrepresentative data (database uncertainty), uncertainty due to measurement errors of sample data or misuse of standards to collect and quantify the data (model uncertainty), uncertainty from the choice of functional unit, cut-off rules, allocation of and so on (preference uncertainty).

It is of significant importance to do uncertainty analysis while comparing different LCA approach for a particular application. However, due to time constraint, they are not addressed in this thesis.

9.4 Further Development

Finally I recommend if the following issues to be confided in further development of the thesis.

- Uncertainty analysis: As discussed, IO based LCAs seem to be the most appropriate approach for economy-wide policy application. However, some of the crucial points regarding the uncertainty should be assessed in order to improve the robustness of the results. This more important while comparing IO based LCA with process-based LCA.
- Software Development: Developing software that calculates environmental tax using company specific data where available (IO-based hybrid) or general economic data (EIO approach) is of high priority in order enhance the practicality of the case studies presented.

- Comparison of environmental taxation with other forms of market instruments: for example, tradable permits (sector specific measures) with tax on specific products. As there are a wide range of economic instruments which can provide incentives for the consumption of more sustainable products, it may be of great importance to analyze the effectiveness of one over the other.

Appendix A

Classification

A.1 Industry and Product Classification

Table A.1: Symmetric IO Table (SIOT)/ Supply and Use tables (SUT) correspondences

Industry	SIOT codes	SUT codes
Agriculture, livestock and hunting	1	1
Forestry, logging and related service activities	2	2
Fishing	3	3
Mining of coal and lignite; extraction of peat	4	4
Extraction of crude petroleum and natural gas; mining of uranium and thorium ores	5	5
Mining of metal ores	6	6
Other mining and quarrying	7	7
Manufacture of coke, refined petroleum products and nuclear fuel	8	8
Production and distribution of electricity	9	9
Manufacture of gas; distribution of gaseous fuels through mains; steam and hot water supply	10	10
Collection, purification and distribution of water	11	11
Manufacture of meat products	12	12
Manufacture of dairy products	13	13
Manufacture of other food products	14	14
Manufacture of beverages	15	15
Manufacture of tobacco products	16	16
Manufacture of textiles	17	17
Manufacture of wearing apparel; dressing and dyeing of fur	18	18
Manufacture of leather and leather products	19	19
Manufacture of wood and wood products	20	20
Manufacture of pulp, paper and paper products	21	21
Publishing and printing	22	22
Manufacture of chemicals and chemical products	23	23
Manufacture of rubber and plastic products	24	24
Manufacture of cement, lime and plaster	25	25
Manufacture of glass and glass products	26	26
Manufacture of ceramic products	27	27

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Table A.1 – *Continued from previous page*

Industry	**SIOT codes**	**SUT codes**
Manufacture of other non-metallic mineral products	28	28
Manufacture of basics metals	29	29
Manufacture of fabricated metal products	30	30
Manufacture of machinery and equipment n.e.c.	31	31
Manufacture of office machinery and computers	32	32
Manufacture of electrical machinery and apparatus n.e.c.	33	33
Manufacture of electronic equipment and apparatus	34	34
Manufacture of medical, precision and optical instruments	35	35
Manufacture of motor vehicles, trailers and semi-trailers	36	36
Manufacture of other transport equipment	37	37
Manufacture of furniture; manufacturing n.e.c.	38	38
Recycling	39	39
Construction	40	40
Sale and retail of motor vehicles; retail sale of automotive fuel	41	41
Wholesale trade and commission trade	42	42
Retail trade; repair of personal and household goods	43	43
Hotels	44	44
Restaurants	45	45
Railway transport	46	46
Other land transport; transport via pipelines	47	47
Water transport	48	48
Air transport	49	49
Support and auxilliary transport activities	50	50
Travel agencies activities	51	51
Post and telecommunications	52	52
Financial intermediation, except insurance and pension funding	53	53
Insurance and pension funding, except compulsory social security	54	54
Activities auxilliary to financial intermediation	55	55
Real estate activities	56	56

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Table A.1 – *Continued from previous page*

Industry	**SIOT codes**	**SUT codes**
Renting of machinery, personal and household goods	57	57
Computer and related activities	58	58
Research and development	59	59
Other business activities	60	60
Market education	61	62
Market health and social work	62	64
Market sewage abd refuse disposal, sanitation and similar activities	63	67
Market activities of membership organization n.e.c.	64	69
Market recreational, cultural and sporting activities	65	71
Other service activities	66	74
Public Administration	67	61
Non-market education	68	63
Non-market health and social work	69	65-66
Non-market sewage abd refuse disposal, sanitation and similar activities	70	68
Non-market activities of membership organization n.e.c	71	70
Non-market recreational, cultural and sporting activities	72	72-73
Private households with employed persons	73	75

Table A.2: Product codes in SUT

Products	Code
Products of agriculture	1
Live animals and animal products	2
Agricultural and livestock services	3
Products of forestry, logging and related services	4
Fish and other fishing products	5
Coal and lignite peat	6
Crude petroleum	7
Natural gas, uranium and thorium ores	8
Iron ores	9
Non-ferrous metal ores,except uranium and thorium ores	10
Non-metallic non-energetic ores	11
Coke, refinement of petroleum and nuclear fuel	12
Production and distribution of electricity	13
Production and distribution of gas	14
Collected and purified water, distribution services of water	15
Meat and meat products	16
Dairy products and ice cream	17
Animal and vegetable oil and fat	18
Prepared animal feeds	19
Other food products	20
Alcoholic beverages	21
Mineral waters and soft drinks	22
Tobacco products	23
Textiles	24
Wearing apparel; furs	25
Leather	26
Leather products and footwear	27
Wood, and products of wood and cork	28
Pulp, paper and paperboard	29
Articles of paper and paperboard	30
Printed matter and recorded media	31
Basic chemicals	32
Pesticides and other agro-chemical products	33
Pharmaceutical products	34
Other chemical products	35
Rubber products	36
Plastic products	37

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Table A.2 – *Continued from previous page*

Product	**Code**
Cement, lime and plaster	38
Glass and glass products	39
Ceramic articles	40
Other non-metallic mineral products	41
Basic metals	42
Fabricated metal products, except machinery and equipment	43
Agricultural and forestry machinery	44
Domestic appliances n.e.c.	45
Other machinery	46
Office machinery and computers	47
Electric machinery and apparatus n.e.c.	48
Telvision and radio receivers; sound or video recording or reproducing apparatus and associated goods	49
Other electronic material	50
Medical precision and optical instruments, watches and clocks	51
Motor vehicles	52
Bodyworks and pieces for motor vehicles	53
Vessels and repair services	54
Railway and tramway locomotives and rolling-stock	55
Aircraft and spacecraft	56
Other transport material n.e.c.	57
Furniture	58
Other manufactured goods n.e.c.	59
Recovered secondary raw materials	60
Residential buildings	61
Non-residential buildings	62
Civil engineering	63
Renting services of construction or demolition equipment with operator	64
Trade and repair services of motor vehicles	65
Retail trade services of automotive fuel	66
Wholesale trade and commission trade services	67
Retail trade services; repair services of personal goods	68
Hotel services	69
Restaurant services	70
Market railway transportation services	71
Non-market railway transportation services	72
Other land and via pipelines transportation, market services	73

Continued on next page

Table A.2 – *Continued from previous page*

Product	**Code**
Other land and via pipelines transportation, non-market services	74
Water transport services	75
Air transport services	76
Market supporting and auxiliary transport services	77
Non-market supporting and auxiliary transport services	78
Market travel agency services	79
Non-market travel agency services	80
Post and courrier services	81
Telecommunication services	82
Financial intermediation services, except insurance and pension funding services	83
Insurance and pension funding services, except compulsory social security services	84
Services auxiliary to financial intermediation	85
Market real estate services	86
Non-market real estate services	87
Renting services of automobiles	88
Renting services of machinery and household goods	89
Computer and relates services	90
Market research and development services	91
Non-market research and development services	92
Market legal and accounting services	93
Non-market legal and accounting services	94
Architectural and engineering consultancy services	95
Advertising services	96
Investigation and security services	97
Industrial cleaning services	98
Other business services n.e.c.	99
Public Administration and defence services; compulsory social security services	100
Market education services	101
Non-market education services	102
Market health services	103
Non-market health services	104
Veterinaty services	105
Market social work services	106
Non-market social work services	107
Market sewage and refuse disposal services	108

Continued on next page

Table A.2 – *Continued from previous page*

Product	**Code**
Non-market sewage and refuse disposal services	109
Market membership organization services n.e.c.	110
Non-market membership organization services n.e.c.	111
Market artistic and news agency services	112
Non-market artistic and news agency services	113
Market cultural and sporting services	114
Non-market cultural and sporting services	115
Other recreational services	116
Other services	117
Private households with employed persons	118

Table A.3: The 37 sectors of the OECD IO table and their ISIC correspondences

No.	Sectors	ISIC Rev.3
1	Agriculture, Hunting, Forestry and Fishing	1+2+5
2	Mining and Quarrying	10–14
3	Food Products, Beverages and Tobacco	15+16
4	Textiles, Textile Products, Leather and Footwear	17+18+19
5	Wood and Products of Wood and Cork	20
6	Pulp, Paper, Paper Products, Printing and Publishing	21+22
7	Coke, Refined Petroleum Products and Nuclear Fuel	23
8	Chemicals	24
9	Rubber and Plastics Products	25
10	Other Non-Metallic Mineral Products	26
11	Basic Metals	27
12	Fabricated Metal Products, Except Machinery and Equipment	28
13	Machinery and Equipment, nec	29
14	Office, Accounting and Computing Machinery	30
15	Electrical Machinery and Apparatus, nec	31
16	Radio, Television and Communication Equipment	32
17	Medical, Precision and Optical Instruments	33
18	Motor vehicles, Trailers and Semi-Trailers	34
19	Other Transport Equipment	35
20	Manufacturing nec; Recycling (Include Furniture)	36-37
21	Utility	40-41
22	Construction	45
23	Wholesale and Retail Trade; Repairs	50-52
24	Hotels and Restaurants	55
25	Transport and Storage	60-63
26	Post and Telecommunications	64
27	Finance and Insurance	65-67
28	Real Estate Activities	70
29	Renting of Machinery and Equipment	71
30	Computer and Related Activities	72
31	Research and Development	73
32	Other Business Activities	74
33	Public Admin. and Defence; Compulsory Social Security	75
34	Education	80
35	Health and Social Work	85
36	Other Community, Social and Personal Services	90-93
37	Private Households with Employed Persons	95-99

Table A.4: Sectors classification in the WIOD CO_2 emissions accounting and their NACE correspondences

No.	Sectors	ISIC Rev.3
1	Agriculture, Hunting, Forestry and Fishing	AtB
2	Mining and Quarrying	C
3	Food, Beverages and Tobacco	15t16
4	Textiles and Textile Products	17t18
5	Leather, Leather and Footwear	19
6	Wood and Products of Wood and Cork	20
7	Pulp, Paper, Paper, Printing and Publishing	21t22
8	Coke, Refined Petroleum and Nuclear Fuel	23
9	Chemicals and Chemical Products	24
10	Rubber and Plastics	25
11	Other Non-Metallic Mineral	26
12	Basic Metals and Fabricated Metal	27t28
13	Machinery, Nec	29
14	Electrical and Optical Equipment	30t33
15	Transport Equipment	34t35
16	Manufacturing, Nec; Recycling	36t37
17	Electricity, Gas and Water Supply	E
18	Construction	F
19	Sale, Maintenance and Repair of Motor Vehicles and Motorcycles; Retail Sale of Fuel	50
20	Wholesale Trade and Commission Trade, Except of Motor Vehicles and Motorcycles	51
21	Retail Trade, Except of Motor Vehicles and Motorcycles; Repair of Household Goods	52
22	Hotels and Restaurants	H
23	Inland Transport	60
24	Water Transport	61
25	Air Transport	62
26	Other Supporting and Auxiliary Transport Activities; Activities of Travel Agencies	63
27	Post and Telecommunications	64
28	Financial Intermediation	J
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30	Renting of M&Eq and Other Business Activities	71t74
31	Public Admin and Defence; Compulsory Social Security	L
32	Education	M
33	Health and Social Work	N
34	Other Community, Social and Personal Services	O
35	Private Households with Employed Persons	P

Appendix B

About The Author

B.1 CV Resume

Eskinder Demisse Gemechu was born in 1979 in Dire Dawa, Ethiopia. He graduated as Textile Engineer in 2003 at the Engineering Faculty of Bahir Dar University (Ethiopia). Upon graduation, he was employed as a graduate assistant in the engineering faculty of Bahir Dar University. He was promoted to assistant lecturer after one year work experience as a graduate assistant. Being an assistant lecturer he offered different textile engineering courses to undergraduate students in the department. In 2007 he got a European Commission Erasmus Mundus scholarship to pursuit his MSc study. He obtained his MSc in Joint European Masters in Environmental Studies in September 2009 from the Universitat Autónoma de Barcelona, Spain and Technische Universität Hamburg-Harburg, Germany. In October 2009 he was awarded a teaching assistantship fellowship from Universitat Rovira i Virgili, where he started doing his PhD. His research activities were mainly focused on assessing LCA methodologies such as process-based LCA, EIO and hybrid-IOLCA for economy-wide environmental tax implementation. He also did a three months research stay at the Bren School of Environmental Science & Management, University of California Santa Barbara, where he developed MRIO model for the Spanish economy. He is a member of the International Society of Industrial Ecology, American Chemical Society and International Input-Output Association. He has research interest in Carbon footprint of products and services, Water footprint, LCA methods (Conventional LCA, Environmental Input-Output Analysis, HybridI-IOLCA, MRIO), Environmental Tax on products and services, Sustainable consumption and production.

B.2 Publications and Presentations

Publications

Villalba, G., Gemechu, E.D., Estimating GHG emissions of marine ports - the case of Barcelona. Energy Policy, vol 39 issue 3, (2011) p 1363-1368

Gemechu, E.D., Butnar, I., Llop, M. and Castells, F. Environmental Tax on Products and Services Based on Their Carbon Footprint: A Case Study of the Pulp and Paper Sector. Energy Policy, (2012) vol 50 p 336-344

Gemechu, E.D., Butnar, I., Llop, M. and Castells F. The impacts of environmental tax on production and consumption of goods in the Spanish economy: the application of EIO and price model (Accepted - January 2013 to J. Environ Plan Manag.)

Gemechu, E.D., Butnar, I., Gomà-Camps, J., Pons, A., Castells, F. A Comparison of the GHG Emissions caused by Manufacturing Tissue Paper from Virgin Pulp or Recycled Waste Paper. (Submitted on 27 - 01 - 2012 to Int. J. Life Cycle Assess.)

Gemechu, E.D., Butnar, I., Sangwon, S., Llop, M. and Castells, F. CO_2 emissions embodied in international trade: A multiregional Input-output model for Spain (Submitted on Nov - 2012 to Ecol Econ)

Gemechu E.D., Butnar I., Llop M. and Castells F., Carbon footprint of products and services: Comparison of life cycle inventory (LCI) methods (In preparation to be submitted to J. clean prod)

Conference participation

Gemechu, E.D., Butnar, I., Sangwon, S., Llop, M. and Castells, F. Application of Multiregional Input-output model to assess CO_2 emissions embodied in international trade: the case of Spain. 23^{nd} SETAC Europe Annual Meeting, May 12 - 16, 2013, Scottish Exhibition and Conference Centre, Glasgow, Scotland (Poster Corner)

Gemechu, E.D., Amores Barrero, M. J., Llop M., Butnar I., and Castells F. The impacts of environmental tax on production and consumption of goods in the Spanish economy. The 20^{th} International Input-Output Conference, Bratislava, Slovakia, June 26-29 , 2012 (Oral presentation).

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