UNIVERSIDADE DE SANTIAGO DE COMPOSTELA

Departamento de Enxeñaría Química

FROM VINEYARD TO SEA
Application of Life Cycle Assessment to Wine and Seafood Sectors

Memoria presentada por:

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Para optar ao grao de Doutor pola
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Informan:

Que a memoria titulada “From Vineyard to Sea. Application of Life Cycle Assessment to Wine and Seafood Sectors” que, para optar ao Grao de Doutor en Enxeñaría Química e Ambiental, presenta D. Pedro Villanueva Rey, realizouse baixo a nosa inmediata dirección no Departamento de Enxeñaría Química da Universidade de Santiago de Compostela.

E para que así conste, firman o presente informe en Santiago de Compostela, o 10 de xuño de 2015.

Fdo. G. Feijoo | Fdo. M.T. Moreira | Fdo. P. Villanueva

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Aos galegos gústannos os nosos viños. Quizabes porque teñen máis ganas de falar ca nós, ou porque teñen un sabor fuxitivo, e buscándollelo, rememoramos tempos, lugares, xentes, amores, despedidas...

(Álvaro Cunqueiro)

If penicillin can cure those that are ill, Spanish sherry can bring the dead back to life.

(Sir Alexander Fleming)

La ciencia siempre vale la pena porque sus descubrimientos, tarde o temprano, siempre se aplican.

(Severo Ochoa)

Abstract

The path towards sustainability in the wine and seafood sectors requires the modification of the current operational and consumption patterns, involving the whole supply chain —where consumers play a relevant role. In this sense, lowering the consumption of energy and materials is necessary to meet sustainability goals and reduce environmental impacts. Hence, the environmental management tools have shown to be useful for measuring environmental performance of human activities. Among the wide range of these tools, Life Cycle Assessment (LCA) highlights due to its holistic point of view of the environmental evaluation of a given product, process, or service.

In this thesis, LCA was applied to wine and seafood (fishing and processing) systems in Galicia. Despite the fact that wine production was already evaluated through LCA in other production areas and countries, this dissertation delved into wine production in Galicia —which had not been assessed so far— offering a detailed inventory for the two largest Protected Designation Origin —PDO— (i.e. "Rías Baixas" and "Ribeiro"). Moreover, other tools and methodologies were implemented for wine sector. In this regard, Data envelopment analysis (DEA) —which allows analysing multiple data in order to include operational benchmarking and eco-efficiency verification— and Greenhouse Gas (GHG) emissions derived from Land Use Changes (LUCs) and vineyard operations were used for assessing the environmental performance of wine sector.

Furthermore, fishing and seafood processing supply chain were evaluated throughout LCA and other complementary tools such as Carbon Footprint (CF) —which involves the emissions of GHGs of a product supply chain— and the Energy Return on Investment (EROI) adapted to food sector, which implies accounting the edible protein energy return on investment (ep-

EROI). Additionally, seaweed derived products —alginate— were evaluated due to its link with fishing and seafood processing.

The application of LCA to wine sector permitted to identify the main hot spots for viticulture and winemaking stages, as well as the proposal of a series of improvement actions to reduce the environmental burdens linked to grape and wine production. Additionally, apart from conventional viticulture techniques, an "environmental friendly" technique such as biodynamic was assessed, characterized by a significant reduction of the main operational inputs and the environmental burdens.

The novelty of the LCA wine studies presented in this thesis was the timeline perspective, suggesting that environmental performance should be reported yearly due to the harvest yield variation, and the application of the LUCs methodology along with LCA to assess a whole PDO in terms of the GHG emissions, as well as the application of LCA to biodynamic viticulture.

The combined application of LCA+DEA methodologies has been useful to avoid problems with standard deviations, which commonly arise in LCA with average inventories. The use of this approach for grape production allowed identifying the operational inefficiencies among vine-growers and translating them into environmental efficiency as well as economic gains.

Furthermore, the energy efficiency of Galician fishing fleet was assessed through the indicator ep-EROI. For this particular case study, Galician fishing fleet inventory data, available from previous studies conducted in Galicia, were used in order to report the energy efficiency for fishing sector.

LCA has also proved to be a useful indicator to report the environmental performance of seafood supply chain. Thus, the sardine supply chain through its derived seafood products was evaluated, identifying the main hot spots of them. Additionally, the GHGs emissions of a multi-ingredient fish based product were analyzed —frozen fish sticks. For this product, fishing stage

has shown to be the most relevant in terms of CF; and when consumption phase is added, final CF presented a high variability depending on consumers behavioral patterns.

Finally, LCA was applied to sodium alginate extraction from seaweed. In this study, the wild harvesting and extraction process were assessed, identifying the extraction process as the main responsible for environmental burdens.

Keywords: Life Cycle Assessment (LCA), Carbon Footprint (CF), Data Envelopment Analysis (DEA), Energy Return on investment (EROI), ep-EROI, Land Use Change (LUC), wine, vineyards, wine-growing, biodynamic, fishing, LCA+DEA, seafood, seaweed.

SECTION I: INTRODUCTION TO THE STUDY

SECTION III: LIFE CYCLE ASSESSMENT APPLIED TO SEAFOOD

SECTION I

INTRODUCTION TO THE STUDY

Chapter 1

Introduction to winegrowing

Summary

Winegrowing is completely linked to the humankind history with numerous evidences from almost all historical periods. It is remarkable the contribution of Romans and Christianity through the Catholic Church to winegrowing expansion. Currently, grapevines can be found worldwide, but, traditionally, their cultivation was divided into two categories, depending on their origin: Old World wines and New World wines. The main wine production area is still Europe where Spain is the country with the largest surface devoted to vineyards. The European Union has fostered different actions to protect the main productive areas. In this sense, the different wine categories based on their origin allows identifying wines in an increasingly competitive market. In the Galician region, winegrowing is mainly located in the south, coinciding with the production areas of the five appellations of origin that have been developed in Galicia. Regarding cultivation methods, there are a number of different options (conventional, organic, biodynamic, etc.), but the most commonly one followed by winegrowers is still the traditional winegrowing and winemaking procedure.

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1.1 History of winegrowing

1.1.1 Development of winegrowing worldwide

The History of grape is linked to the humankind development. The main product obtained from grapes: wine, has historically received great relevance since the Classic Mythology considered it as a divine drink for gods. Hence, Greeks and Romans dedicated a deity called Dionysus or Bacchus, respectively —God of the grape harvest, winemaking and wine.

The earliest evidences of grape (*Vitis vinefera* L.) cultivation towards better productive and fermentation practices were found in The Near-East (McGovern, 2014). Furthermore, the first evidences of wine production were found in Iran circa 7400-7000 BCE (Before the Common Era). Also, seeds of wild grapes were found in Europe in archeological excavations from the Neolithic-Bronze age, suggesting the exploitation of grapes in the area (Marinval, 1997). The domesticated grapevine spreading from The Near-East quickly reached other cultures and civilizations: Egyptian, Mesopotamian, Phoenician, Greek, Roman and Chinese. One of the main drivers of grapevine spreading was the trade between the cultures located close to the Mediterranean Sea (Royer, 1988). Nonetheless, it was under the influence of The Roman Empire when the grapevine got inland areas, following the main trade routes in Europe, and vineyards were developed in the conquered lands (Royer, 1988).

After the collapse of The Roman Empire, during the Middle Age, the main driver for grapevine spreading was the Christianity through the Catholic Church. In addition, given the fact that wine is needed during the Eucharist, new vineyards were planted to supply wine to monasteries and churches. Thus, The Crusades allowed conquering new regions, increasing grapevine expansion and germplasm exchanging. Furthermore, during the Middle Age, the Islam expansion in North Africa, Iberian Peninsula and Middle East entailed an expansion of grape —mainly table grapes.

The discovery and colonization of the New World also brought grapevine to America, firstly as seeds (easily transportable) and then as cuttings. Again, it should be remarked the role of the

Catholic Church since their Missions introduced grapevines from their regions of origin (i.e. Spain, Portugal, Italy, etc.). Thus, grapevine was also introduced in other new regions such as South Africa, Australia and New Zealand.

After grapevine expansion, new varieties were introduced from America into the Old World during the late 19th century, which also brought unknown diseases (phylloxera and mildews) in Europe. Consequently, European vineyards were drastically devastated by the phylloxera plague. The solution was the introduction of cuttings of other species —used as rootstocks— resistant to phylloxera. However, the introduction of disease-resistant varieties —generating new hybrid species— implied a drastic reduction in genetic diversity during 20th century. Furthermore, grapevines in Europe suffered two World Wars that destroyed large areas of vineyards.

Nowadays, grapevine is dealing with the substitution from the local —and traditionally cultivated grapevines— towards more familiar and more extensively cultivated varieties (e.g. Chardonnay, Merlot, Cabernet Sauvignon, etc.) due to the global market demands, decreasing, again, grapevine diversity (This et al, 2006).

1.1.2 Development of winegrowing in Galicia

The first evidences of grapevine cultivation in Galicia are associated to The Roman Empire occupation since it is believed that Romans introduced and cultivated grapevine to provide wine to their troops (Díaz Losada, 2011). In fact, some archeological remains from Castro culture and Romans (e.g. amphorae, press rocks and seeds) indicate the existence of grapevine since 100 BCE. Furthermore, other evidences indicate the existence of grapevine before Romans since *Vitis sylvestis* (wild grapes) pollen was found in an archeological excavation in A Coruña, dated on 1300 BCE (Congil, 2007). After the fall of the Roman Empire during Middle Age, grapevine was cultivated all over the region due to the rise and influence of the Catholic Church (Díaz Losada, 2011). However, it was after The Spanish Reconquer when grapevines were widely established in Galicia.

During the 19th century, Galician vineyards suffered diseases (i.e. phylloxera and mildews), moving the vineyards to Southern regions and establishing them in their current location. Nonetheless, there are still grapevines in some Northern regions as isolated and small plots. These grapevines in some cases have achieved the quality category —according to the Spanish legislation (Ley 24/2003[1]) — named as "Vino de la Tierra" or Country wine (e.g. "Barbanza e Iria", "Betanzos" and "Val do Miño-Ourense").

Given the features of Galicia in terms of topography, soil, climate, etc., grapevines are fairly different based on the location characteristics. Thus, vineyards can be found in valleys: "Rías Baixas" appellation and some "Ribeiro" plots; alluvial terraces: "Valdeorras" appellation; and terraces: "Ribeiro" and "Ribeira Sacra". It should be highlighted that the geographical distinctiveness of the "Ribeira Sacra" appellation implies difficulties for mechanization due to the steeper slopes and terracing —the so-called heroic viticulture (Figure 1.1). In fact, it is common to use lifters to take the harvest from lower areas to places that can be accessible by road. In those cases where it is not possible or no affordable to install lifters, the solution is to ship the harvest by boat.

As aforementioned, climate has influence on Galician vineyards. Thus, two different types can be found: i) continental climate in appellations of "Ribeiro", "Ribeira Sacra", "Monterrei", and "Valdeorras"; and ii) Atlantic climate in "Rías Baixas" appellation. Furthermore, the soil type, in terms of fertility, also has influence on grapevines, selecting those varieties adapted to poor soils with acidic pH since, traditionally, rich soils were used for food production (e.g. maize, beans, potato, etc.) due to the dominant agriculture of subsistence, moving grapevines to mountains or hills (Díaz Losada, 2011).

[1] BOE, 2003. Ley 24/2003, de 10 de Julio, de la viña y el Vino, Ley 24/2003.

Figure 1.1. Picture of vineyards in "Ribeira Sacra" appellation with detail of the lifters. Source: González (2015)

Concerning trellis configuration, there are two different trends. On the one hand, the overhead trellis —typical of the Southwestern region and the "Rías Baixas" appellation— consists of posts (granite or concrete), joists (concrete or iron) and wire (iron or steel) (Figure 1.2), keeping vines canopy on a parallel plane to soil. The purpose of this system is to keep vines far from soil humidity that could favor diseases as well as it allowed the combined exploitation of other crops. Overhead trellis origin might be the Etruscan viticulture (Díaz Losada, 2011) introduced by Romans from Northern regions of Italy. During the mechanic revolution, this system developed —increasing in height and width— to allow a better level of mechanization.

On the other hand, the espalier system is typical of the vineyards of the continental areas and some areas of the "Rías Baixas" appellation. This system consists of posts —commonly made of wood or iron, but in some cases granite or slate— and wire (iron or steel), but unlike overhead system, it keeps vine canopy on a perpendicular plane to soil (Figure 1.3). The espaliers were introduced by Romans from Southern regions of Italy —influenced by the Greek viticulture (Díaz

Losada, 2011). As overhead trellis, the machinery revolution triggered changes in the features of espaliers, widening the distance between espaliers to enable machinery access.

Figure 1.2. Vineyard with overhead trellis system

Figure 1.3. Vineyard with espalier trellis system

1.2 Wine sector

1.2.1 New World and Old World wines

Viticulture has historically developed throughout Europe. In fact, wines arriving from this geographical area are commonly referred to as Old World wines. In contrast, the New World wines are those arriving from relatively modern areas of production, such as the Southern Hemisphere (South Africa, New Zealand, Australia, Chile, Peru or Argentina), the United States, Turkey and most of the Asian continent (e.g. China or Iran).

Old World vineyards represent just over 50% of the global surface area used for viticulture activities, as shown in Figure 1.4. Spain, France, Italy and, to a lesser extent, Portugal are the countries in the Old World with a large surface area linked to winegrowing; in the New World, Turkey and the United States lead the rankings (Figure 1.5). Nevertheless, the relative weight of Old World wine on an international scale has gradually decreased in recent decades. In addition, global vineyards have experienced a slight decrease in surface area (OIV, 2013).

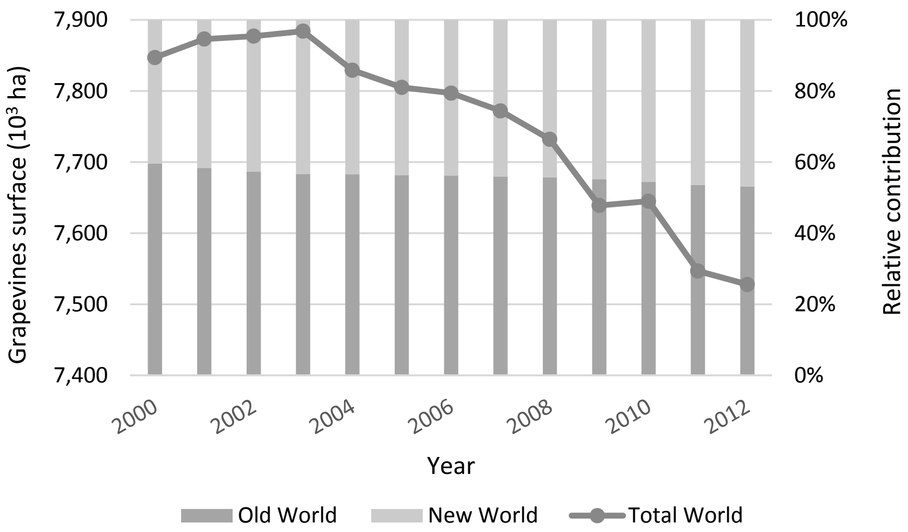

Figure 1.4. Worldwide distribution of vineyards divided by geographical location. Adapted from OIV (2013)

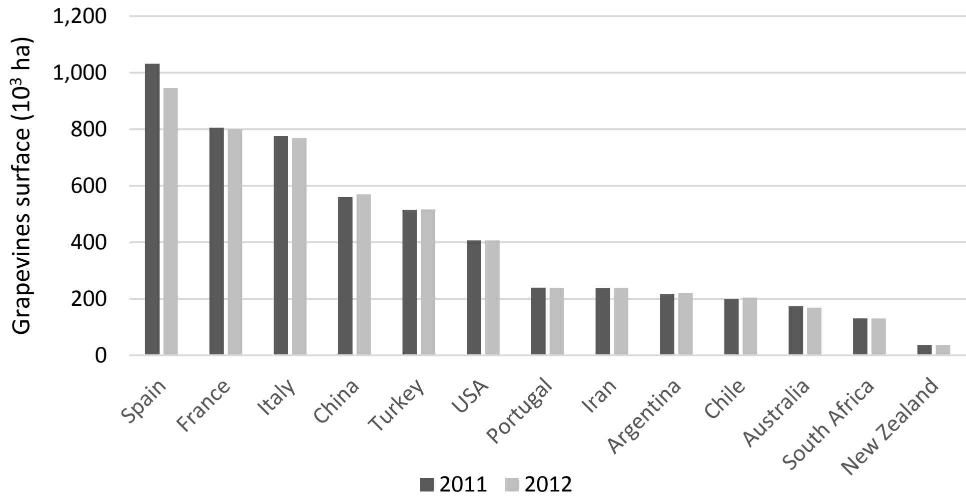

Figure 1.5. Worldwide vineyards surface per country during years 2011 and 2012. Adapted from OIV (2013)

In terms of production, 60% of the wine produced worldwide still arrives from Old World appellations and vineyards. Moreover, in terms of surface area, wine production has also experienced a dwindling tendency in the past few years (Figure 1.6). Interestingly, these values are in opposition to wine consumption worldwide, which shows a substantial increase in the past decade (OIV, 2013). The reason behind these opposing trends is linked to the optimization of production stocks, which are increasingly exported to meet demand in other areas (Figure 1.7).

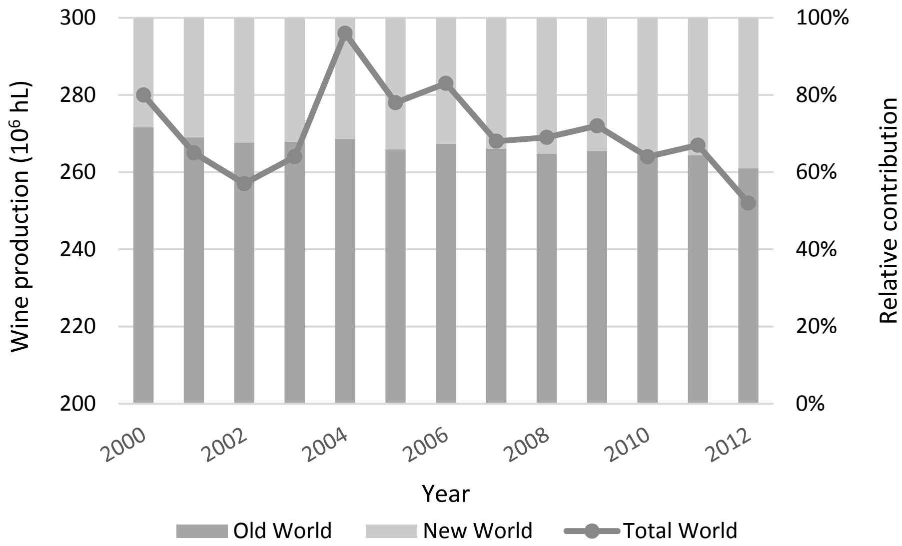

Figure 1.6. Worldwide distribution of wine production divided by geographical location. Adapted from OIV (2013)

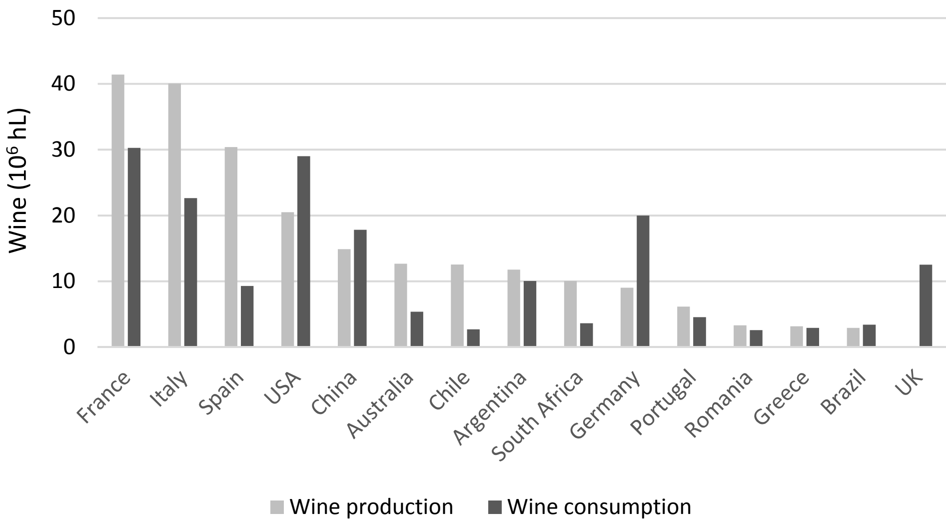

Figure 1.7. Absolute production and consumption of wine in a selection of countries during 2012. Adapted from OIV (2013)

1.2.2 Wine sector in Spain

Spain has the largest surface devoted to vineyards worldwide (i.e. 946,293 ha in 2013). However, it should be noted that this surface has decreased in 250,000 ha during the last decade. In terms of grape production, circa 99% is destined to wine elaboration whilst the remaining is consumed as table grapes or raisins (MAGRAMA, 2013).

Vineyards can be found all over Spain, but it is the Autonomous Community of Castilla-La Mancha which concentrates circa 50% of the surface —442,002 ha in 2013— devoted to winegrowing (Figure 1.8). For the remaining regions, the surface ranges from 81,672 ha (9%) in Extremadura to 70 ha (0.01%) in Asturias. Following the wine classification described above, the surface under a Protected Designation of Origin (PDO) was 580,288 ha in 2013 (MAGRAMA, 2014), which implies 61% of the total surface.

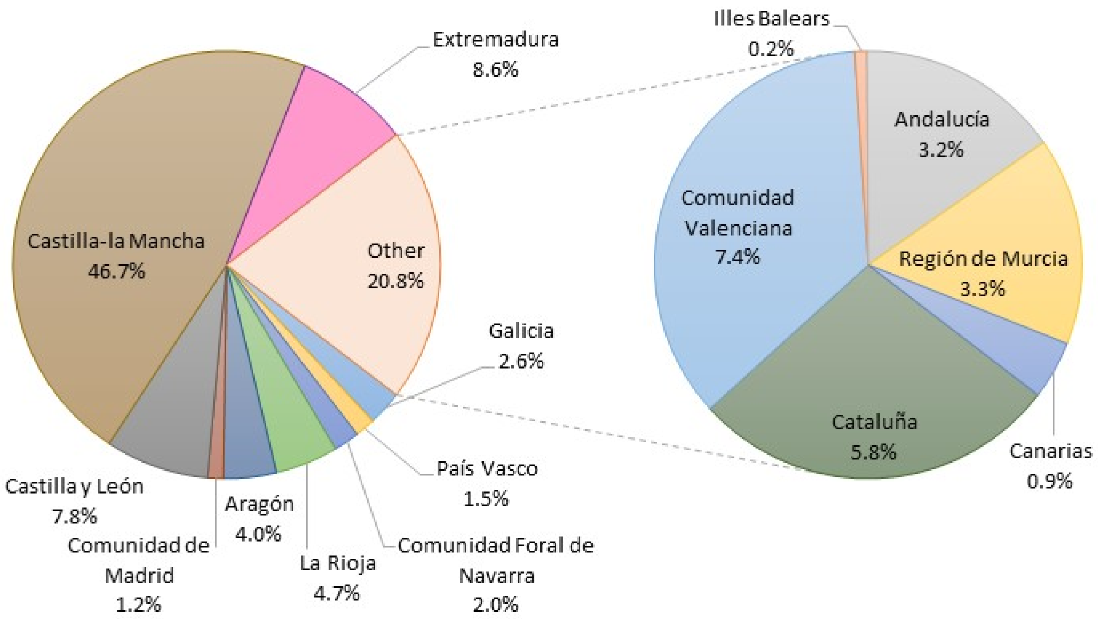

Figure 1.8. Grapevine surface share per Autonomous Community during 2013. Adapted from Eurostat (2015)

During 2013, Spain became the largest wine producer worldwide with 50.5 million hL, increasing by 41% over the previous year and passing other countries such as Italy or France (Núñez, 2014). In addition, the production of wine is generally related to the surface devoted to vineyards. Thus, the production of wine was focused on Castilla-La Mancha, whose production was 31.2 million hL, followed by Extremadura with 4.1 million hL in 2013. Focusing on the type of wine produced, 51.1% were red and rosé wines whilst 48.9% was white wine (ICEX, 2015). Furthermore, concerning grape variety, *Airén* and *Tempranillo* are the most common varieties with shares of 23.5% and 21%, respectively, based on cultivated area data.

Concerning production structure, there are about 4,000 wineries (ICEX, 2015) characterized by small size, domestic capital, family business or cooperatives. However, it should be remarked the existence of large companies, which comprise wines from all over the country to diversify their product portfolio, breaking the traditional features of wineries in Spain and taking a high market share. Furthermore, the grape supply is still mainly performed through independent winegrowers —who are not related to wineries— or winegrowers from cooperatives.

Regarding commercialization during the campaign 2012/2013, the domestic market for PDO wines entailed 54% of the sales —where red wine and white wine implied 58% and 22%, respectively— whilst exportations represented 46% of the sales (MAGRAMA, 2014), with the following distribution: red wine, 52%, sparkling wine, 23% and white wine, 13%.

The exports of wine under a PDO were focused on European Union countries (70% of market share), followed by USA (9%), other European countries outside the Union (7%), Switzerland (4%), Japan (3%) and other countries (7%) (MAGRAMA, 2014).

1.2.2.1 Wine classification

Given the wide variety of wine worldwide, it can be classified attending to different features. In this regard, three different classifications can be made following characteristics such as: i) origin, ii) type and iii) ageing.

Origin

The classification based on wine origin follows the current legislation on wine (European Commission, 2008a, 2009, 2010), and it divides wine into two categories as follows:

- Protected Designation of Origin (PDO) covers wines produced within a determined geographical area. Wine produced in these areas must be produced exclusively with grapes from them. Examples of this category are: “Rías Baixas”, “Ribeiro”, “La Rioja”, or “Bordeaux”. This category incorporates the subcategories for the PDOs included within Spanish legislation (BOE, 2003) as follows:
    - Appellation of Origin
    - Qualified Appellation of Origin
    - Quality wine with geographic indication
    - Single-state wine or “Vino de Pago”
    - Qualified single-state wine or “Vino de Pago Calificado”
- Protected Geographical Indication (PGI) covers wines produced close to a determined geographical area. These wines must be produced with at least 85% grapes from the area. The wines included within this category are also known as table wine, and, similarly to PDO wines, they can be subdivided according to the Spanish legislation as follows:
    - Table wine
    - Table wine with geographical indication
    - Country wine

Currently, Spain has 85 productive regions under a PDO —where 67 are Appellation of Origin, two are Qualified Appellation of Origin, seven are Quality wines with geographic indication, and

14 are Single-state wines or “Vinos de Pago”— and 41 regions, spread all over the country, under a Protected Geographical Indication (PGI).

It should be noted the fact that legislation worldwide is less restrictive in terms of wine labelling and/or classification as European and, therefore, it is quite difficult to harmonize the classification based on wine origin for countries abroad European Union. For instance, whilst products obtained from any fruit juice fermentation are allowed to be labeled as wine in United States, it is completely forbidden in Europe, where the fermented juice from grapes is only allowed to be labeled as wine.

Type

Wine can be classified in two large subcategories according to the vinification method and style (Hidalgo, 2011; Iris, 2013).

- Still wine includes all those wines that share a similar process during their elaboration and an alcoholic content ranging from 9% to 14.5%. These wines can be classified attending to their color. It should be noted the fact that wine color is not defined by grape color, being the vinification methods the responsible for wine final color. Thus, four more subcategories can be drawn:
    - White, obtained from white grape juice fermentation without the maceration of other parts.
    - Rosé, obtained from the fermentation of white and red grapes without the presence of grape skin.
    - Red, obtained as a result of the maceration and fermentation in presence of the grape skin.
    - Claret, obtained after mixing white and red grape juice and, generally, in presence of grape skin.

- Special wine includes wines with different features (e.g. alcohol and sugar content, sparkling, vinification method, etc.). Wines within this category can be grouped in the following subcategories:
  - Dessert wines: those wines whose sugar content is higher than regular wine. Generally, they are made with extremely ripe grapes (i.e. late harvest, infected by botrytis, etc.).
  - Fortified wines: those wines, generally sweet and with high alcoholic content, to which a distilled spirit (usually brandy) is added. Examples of this wine are: Sherry, Port and Vermouth.
  - Sparkling wine: those wines that contain carbon dioxide produced either from natural fermentation or forcibly injected later. Within this category, there are different types of wine based on the carbon dioxide pressure and origin (i.e. natural fermentation or injected). Examples of these wines are: “Vino de aguja” (Spain), Champagne (France), Cava (Spain), “Espumante” (Portugal), Sekt (Germany), etc.

Age

Wine can also be classified according to the length of time that wine spends in the winery. Thus, wine is divided into two categories as follows:

- Young wines: those wines consumed just after being elaborated (year-old wines) or bottled after fermentation (alcoholic and/or malolactic). In some cases, wine spends several months in tanks or barrels, but it cannot meet the features to be considered as an aged wine.
- Aged wines: those wines that have been resting in the winery (in wood barrels and in the bottle) during a time period (BOE, 2003; European Commission, 2010; Hidalgo, 2011). Thus, these wines can be divided, based on the duration of the ageing period, as follows:

- “Crianza” wine: wines aged for two years, remaining at least six months in barrel.
- Reserve wine: wines aged for three years, of which at least one in barrel.
- Great reserve wine: wines aged for five years, of which at least 18 months in barrel.

Given the different features of white and red wine in terms of ageing, the resting period either in barrel or in the bottle is slightly lower for white wines. For instance, the great reserve white wines are aged for four years, of which six months in barrels.

1.2.3 Wine sector in Galicia

Vineyards are spread all over Galicia, but extensive production areas are focused in Southern regions. According to data from Xunta de Galicia, Galicia had 24,942 ha —49% in Pontevedra, 34% in Ourense, 10% in A Coruña and 8% in Lugo— devoted to vineyards in 2012, of which roughly 40% is under a PDO (Xunta de Galicia, 2015). Furthermore, wines under a PGI are almost negligible, in terms of surface or winegrowers, when they are compared to the PDO wines. Following the wine classification, according to wine origin, there are four areas under a PDO and three under a PGI:

- PDO wines are mainly located in the provinces of Pontevedra and Ourense and some Southern councils of A Coruña and Lugo. Currently, there are five PDO in Galicia:
    - “Rías Baixas”
    - “Ribeiro”
    - “Ribeira Sacra”
    - “Valdeorras”
    - “Monterrei”

- PGI wines come from three different regions:
    - “Barbanza e Iria”
    - “Betanzos”
    - “Val do Miño-Ourense”

Table 1.1 shows the main figures of the five PDOs of Galicia, highlighting “Rías Baixas” as the largest one, followed by “Ribeiro”. Indeed, this dissertation is focused on the environmental assessment of the production of grape/wine from these appellations. Taking into account the revenues for the five appellations during 2013, the five PDOs represented 0.26% of the Galician Gross Domestic Product (GDP) and 0.01% of the national one (INE, 2015).

Table 1.1. Production features for the five Galician PDOs during 2013. Adapted from Xunta de Galicia (2015)

PDO	**Surface (ha)**	**Winegrowers**	**Wineries**	**Grape production (kg)**	**Wine production (hL)**	**Revenue (€)**
“Monterrei”	437	371	23	2,316,533	1,818,500	8,619,690
“Rías Baixas”	4,064	6,677	178	33,743,486	17,349,300	85,361,000
“Ribeira Sacra”	1,250	2,817	90	4,735,944	3,410,047	14,580,000
“Ribeiro”	2,762	5,960	102	12,491,548	8,450,500	21,126,250
“Valdeorras”	1,144	1,467	46	4,434,749	3,673,102	16,528,959
Total PDO	**9,657**	**17,292**	**439**	**57,722,260**	**34,424,349**	**146,215,899**

1.2.3.1 Production data and areas

“Rías Baixas”

Delving into the features of the appellations studied, in terms of production structure, “Rías Baixas” is divided into five production subareas: Val do Salnés, O Rosal, Condado do Tea, Soutomaior and Ribeira do Ulla. In addition, the Val do Salnés subarea is the largest in production, yield, surface, number of plots and winegrowers, but it has a lower surface/winegrower ratio —which indicates smallholder agriculture— in comparison to other subareas such as Ribeira do Ulla or O Rosal (Table 1.2). Also related to smallholding, the high ratio for plots/winegrower indicates a high land fragmentation and, therefore, difficulties towards mechanization and efficiency. Concerning production yield, it ranges from 2.46 t/ha (Soutomaior) to 9.39 t/ha (Val do Salnés).

When the appellation is assessed as a whole (Table 1.2), "Rías Baixas" presents a ratio of 6,000 m^2/winegrower and 3.51 plots/winegrower —which confirms the stated above smallholding and land fragmentation, that impedes better development of the appellation— and a production yield of 8.30 t/ha.

Table 1.2. Production structure for the appellations "Rías Baixas" and "Ribeiro" during 2013. Adapted from Rías Baixas (2015) and Ribeiro (2015)

Appellation	Winegrowers	Surface (ha)	Surface/ winegrower	Plots	Plots/ winegrower	Production (t)	Yield (t/ha)
"Rías Baixas"	**6,667**	**4,064**	**0.61**	**23,386**	**3.51**	**33,743**	**8.30**
Subareas							
Val do Salnés	4,759	2,280	0.48	15,607	3.28	21,404	9.39
O Rosal	514	594	1.16	1,607	3.13	3,704	6.24
Condado do Tea	1,241	1,007	0.81	5,795	4.67	7,374	7.32
Soutomaior	43	18	0.43	117	2.72	45	2.46
Ribeira do Ulla	110	165	1.50	260	2.36	1,217	7.40
"Ribeiro"	**5,960**	**2,762**	**0.46**	**65,000**	**10.91**	**12,492**	**4.52**

Concerning wineries, they are spread all over the five production subareas —177 in 2013. Nonetheless, they are mainly concentrated in the Val do Salnés, which is not strange due to the fact that it is the largest subarea in surface devoted to vineyards within the appellation, as well as the largest in production. In addition, it should be remarked that the three major cooperatives are located in this subarea.

The volume of wine elaborated was 229,899 hL in 2013 (Rías Baixas, 2014) —of which 173,493 hL were commercialized. The main market for wines elaborated under the PDO is still the national market, but exports have been increasing continuously during the last 10 years (Rías Baixas, 2014), representing 31% of the sales. By exports destination, the main markets are USA and the EU, which account for 48% and 31 of the exports, respectively.

Figure 1.9 shows the evolution of the Rías Baixas appellation in terms of production and surface for the period 2001-2013. The surface of the appellation has been increased significantly by 1,656 ha, from 2,408 ha in 2001 to 4,064 ha in 2013. Additionally, the production has also been increased considerably, ranging from 13,253 t in 2002 to 41,788 t in 2011 —which is the record harvest so far. However, as other extensive crops, winegrowing is affected by external factors such as weather, diseases, etc. that have influence on grapevine vegetative cycle and, therefore, on the final yield and production.

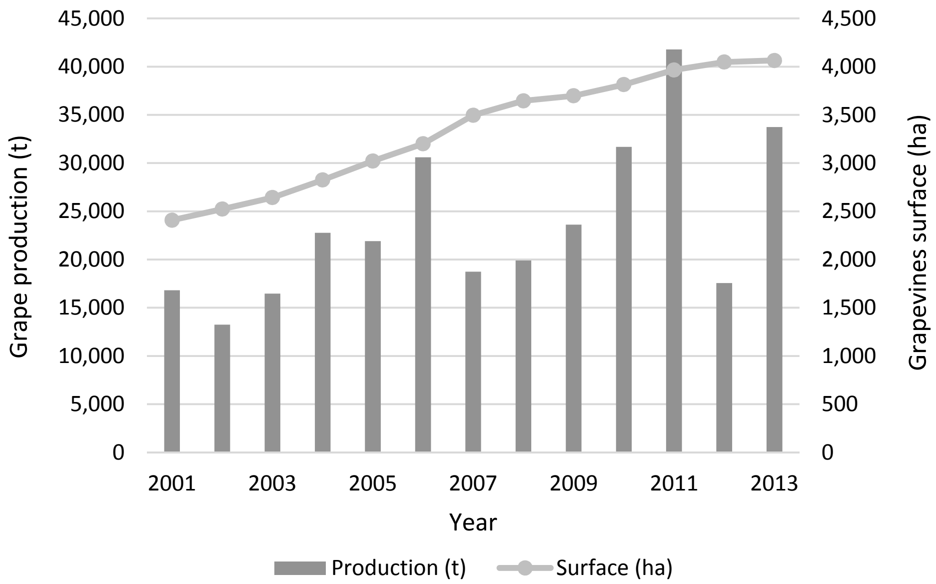

Figure 1.9. Evolution of production and surface in the appellation "Rías Baixas" during 2001-2013 period. Adapted from Rías Baixas (2015)

Given the restrictions and legislation of the PDOs, there are only several grape varieties under the umbrella of the appellation. In this regard, Rías Baixas includes either white or red grape varieties to elaborate their wines. However, it should be highlighted the fact that white wines are the most popular ones and produced within PDO portfolio —where white varieties represent 99% of the production. Thus, the most widely cultivated is the Albariño grape variety, representing 94% of the whole production (Rías Baixas, 2014) and giving name to its most famous wine: *Rías Baixas Albariño*. The varieties cultivated under the PDO are classified, according to their importance, as follows:

- Preferential varieties:
    - White: *Albariño*, *Loureira blanca*, *Treixadura* and *Caiño blanco*.
    - Red: *Caiño tinto*, *Espadeiro*, *Loureira tinta* and *Sousón*.
- Authorized varieties:
    - White: *Torrontés* and *Godello*.
    - Red: *Mencía, Brancellao* and *Pedral.*

"Ribeiro"

Regarding "Ribeiro" appellation, unlike "Rías Baixas", it only has one production area. This appellation presents even a lower ratio surface/winegrower and plots/winegrower: 0.46 and 10.91, respectively. Land fragmentation and smallholder agriculture are the main features —and, at the same time, problem— of this appellation, being a handicap towards mechanization and professionalization of the sector. The production yield is 4.52 t/ha, which is lower than for "Rías Baixas".

Figure 1.10 shows the evolution of the "Ribeiro" appellation, based on production and surface, for the 2005-2013 period. It was impossible to go further back due to lack of reliable data. The grape production of the appellation ranges from almost 20,000 t in 2005 to 9,759 t in 2008. The surface of the appellation has steadily increased in 157 ha during 2005-2012 period, but it decreased by 80 ha in 2013.

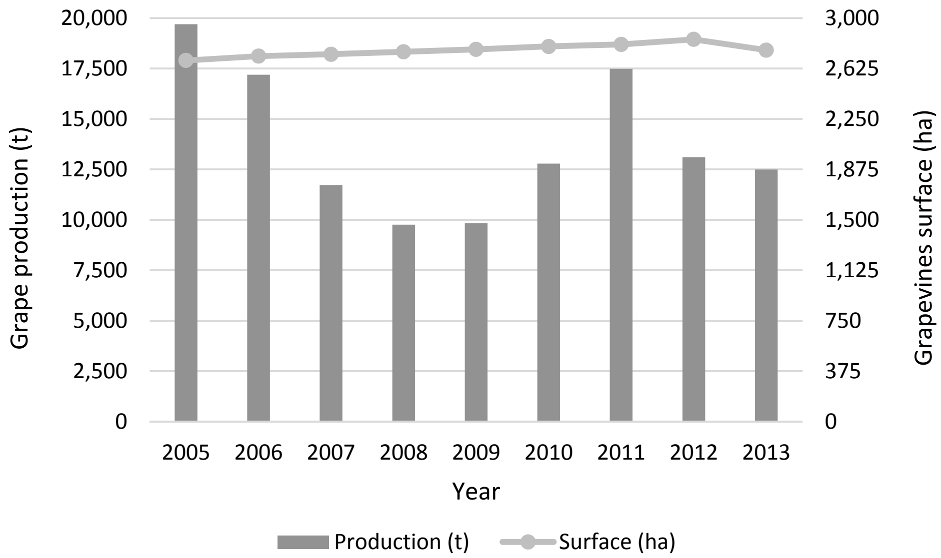

Figure 1.10. Evolution of production and surface in the appellation "Ribeiro" during 2005-2013 period. Adapted from Ribeiro (2015) and Xunta de Galicia (2015)

As "Rías Baixas", there are several grape varieties included under the PDO in the "Ribeiro" appellation that are the origin of the three types of wine elaborated: white, red and *tostado*. In terms of production share, white and red wines represent 85% and 15% of the wine elaborated, respectively, whilst *tostado* wines are almost negligible: 0.04% (Ribeiro, 2014; 2015). Due to the wide catalog of grapes, which includes indigenous and foreign varieties, wines are the result of the combination of the varieties described below (Ribeiro, 2014):

- Preferential varieties:
    - White: *Treixadura*, *Torrontés*, *Albariño*, *Loureira blanca*, *Godello* and *Lado*
    - Red: *Brancellao*, *Caíño longo*, *Ferrón*, *Sousón* and *Mencía*

- Authorized varieties:
    - White: *Macabeo, Albilla* and *Palomino*
    - Red: *Garnacha* and *Tempranillo.*

1.3 Winegrowing

1.3.1 Types of winegrowing

Operations linked to the viticulture phase are highly variable, depending on a wide range of issues such as climatic conditions, soil characteristics and altitude. In addition, the current tendency throughout wine regions to homogenize viticulture and vinification operations within one single appellation has led to the appearance of a series of common standards that winemakers must comply. Nevertheless, despite this homogenization in terms of operational inputs, it has become common to see viticulture practice divided according to the operations related to plant protection and fertilization. Hence, many studies in the field of viticulture distinguish between conventional wine and organic wine (Gabzdylova et al, 2009). The main characteristic of conventional viticulture, when compared with organic viticulture, is the fact that there are no legal restrictions regarding the use of fertilizing and plant protection agents (European Commission 2008a, 2008b, 2012). Another important characteristic of conventional viticulture systems is the use of machinery for most operational activities. Finally, despite advocating for certain quality standards, which are usually regulated by the appellations that manage specific wine types or areas, conventional viticulture prioritizes obtaining high yield rates.

Organic viticulture does not use mineral fertilizers on vineyards; it also strictly limits the use of synthetic substances as plant protection agents. Within organic wine, an interesting subcategory is biodynamic viticulture (Chapter 6 delves into it). The latter is even more restrictive than regular organic wine sites, seeking complete harmonization of the vineyards with their surrounding ecosystems, and using a series of biodynamic preparations to treat the vines (Lotter 2003;

Masson 2009). These cultivation sites, unlike conventional viticulture, are aimed at prioritizing grape and wine quality, as well as seeking an environmentally friendly approach to winemaking, rather than enhancing productivity. Organic and biodynamic viticulture are currently experiencing a strong proliferation, with many new and old wine farms promoting a change in operational activities (Gabzdylova et al, 2009). In fact, many stakeholders see in this transition an opportunity to improve their sales and attain a better position in the wine market.

1.3.2 Winegrowing and winemaking in Galicia

Despite the increasing number of farms adopting ecological friendly management systems, the conventional viticulture is the most commonly used for winegrowers in Galicia. Thus, the farming operations begin, just after harvest, with pruning (Figure 1.11), coinciding with the vegetative rest of the vines during fall/winter. Due to the features of the grapevines (i.e. training system, smallholding, land fragmentation, etc.), the mechanization of this operation is unfeasible nowadays, depending exclusively on human labor. However, the introduction in the market of the pruning shears —engineered specifically for winegrowing— has allowed gaining time and, therefore, finishing this operation in a short period. Furthermore, during spring/summer (Figure 1.11), vineyards have to be pruned again, but it is the so-called green pruning since some one-year-old branches are cut. In addition to the latter operation during spring/summer, vineyards are also cleared (i.e. some leaves clearing) to improve grape ripeness and prevent from fungal diseases —better ventilation of vineyards.

After finishing pruning, tying is the next fall/winter operation, which involves the branches tying to the training system. The material used to tie the branches has evolved from the traditional wicker to novelty materials such as plastics or biodegradable compounds. Similarly to pruning, the mechanization of this operation is not possible, and the only solution to get it easier and quicker is throughout electronic tying machines —very similar to the pruning shears.

Regarding pruning waste, there are several options to deal with it: i) burning, once pruning is finished, branches are picked up and gathered to be burned on field; ii) chipping, a wood chipper is used on field; iii) clearing/mowing, the same tractor-implement for mowing and pulling out weeds is used for this operation.

The fertilization of vineyards is an operation that depends on the requirements of soil and vines. Given the ancient tradition of use the fertile lands for crops, vineyards are established in poor soils. Thus, it is very common to apply mineral fertilizers to correct either deficiency of nutrients (N, P or K) or acidity. In addition, organic fertilizers are used not only to provide nutrients, but also to improve soil structure. As abovementioned, fertilization depends on soil requirements, so it might not be necessary to fertilize every campaign. This operation, in some cases, is still made by hand, but it is very common to use fertilizer distributors. Generally, fertilization takes place during late winter or early spring (Figure 1.11).

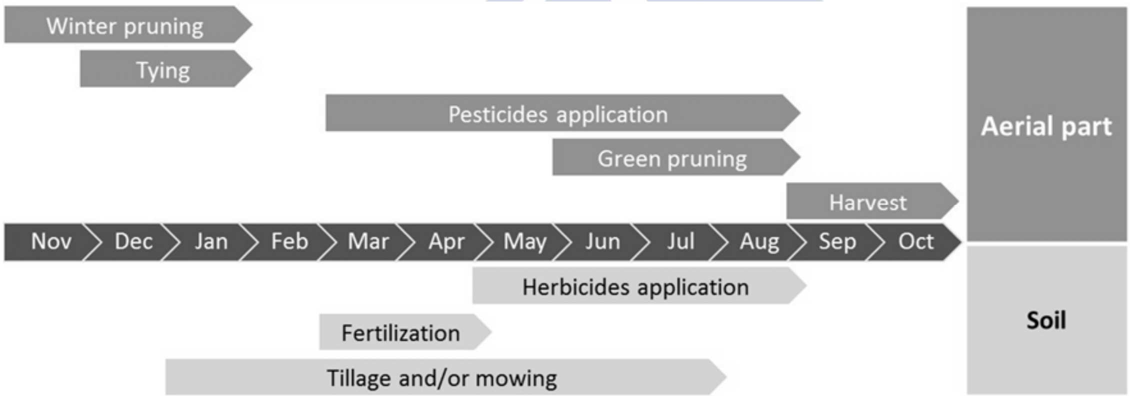

Figure 1.11. Schedule for the main winegrowing operations

Plant protection agents (i.e. pesticides and herbicides) are applied during spring and summer (Figure 1.11). Atlantic climate vineyards (e.g. "Rías Baixas") are prone to suffer from recurrent fungal diseases so that the number of phytosanitary treatments is higher than other appellations such as "Ribeiro" or "Valdeorras" (continental climate). Thus, whilst the number of treatments ranges from 12 to 18 per campaign in "Rías Baixas", it varies between 8 and 14 for "Ribeiro" appellation. Commonly, this operation is completely mechanized in Galician vineyards, but knapsacks and powder bellows are still used in some farms because of smallholding. Regarding herbicides, winegrowers are decreasing their use, restricting their application to those parts where the tractor-implements cannot reach —row of posts usually— or stopping their application in some cases. Additionally, after current findings regarding glyphosate[2], which is the most widely herbicide used by winegrowers, reported by the World Health Organization (WHO), it has raised international attention to its indiscriminate use and, therefore, a deeper review of glyphosate safety —or not— will be conducted by the regulatory agencies on this subject to decide what to do with this compound (Cressey, 2015).

Tillage is an operation that is getting less common to avoid root fungal diseases and soil degradation. Nowadays, winegrowers tend to keep the grass cover or even plant a cover crop, which can fix nitrogen (e.g. leguminous). In addition, the management of cover crops/grass cover requires mowing when it is necessary. As the previous operation, it is completely mechanized, but it is still made by hand in some wine-growing holdings.

The grape harvest takes place in late summer or early fall (Figure 1.11), depending on weather and grapes ripening. The operation is completely manual —unlike other production areas (outside Galicia) where it is performed by tractor-implement harvesters— being necessary a large amount of workers to pick up grapes. After being collected in plastic boxes, grapes are transported from vineyards to wineries to begin the vinification process.

Concerning vinification process (Figure 1.12), it begins with grape reception, where grapes are subjected to administrative —PDO requirements— and quality controls, as well as weighed and classified according to the sugar concentration. Generally, after grape reception, grapes are destalked to remove non desirable parts (e.g. stems) that could add flavors and aromas to wine.

[2] Glyphosate (N-(phosphonomethyl)glycine) is a broad-spectrum systemic herbicide used on food and non-food crops.

After that, some wineries macerates the fruit flesh, must —some berries are broken after destalking— and skins at low temperature (below 8°C for 7 h) to extract aromas, but this stage, as the previous one, is optional and not all wineries do so. Subsequently, flesh and skins are pressed by pneumatic presses to obtain must. It should be highlighted that several wineries perform neither destalking nor crushing stages and, therefore, pressing is the very next stage after grape reception.

After crushing, the product obtained is transferred to tanks —generally, made of steel— where the solid parts are separated from the must in an operation called *debourbage* or desludge. Then, the clean must is transferred to the steel fermentation tanks to develop alcoholic fermentation at 18°C for two weeks.

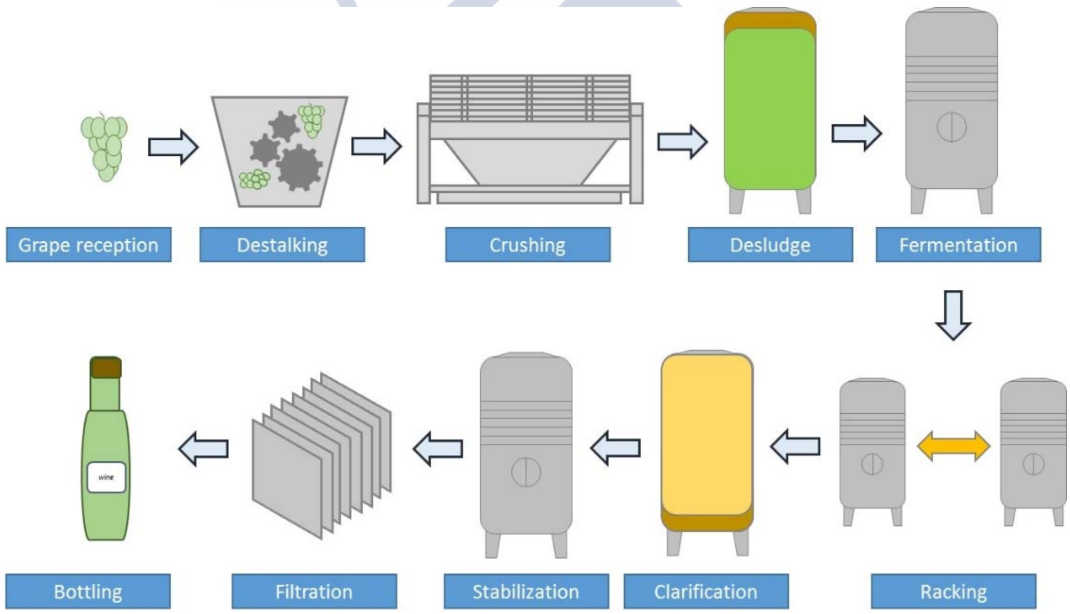

Figure 1.12. Vinification stages at winery

Given the necessity of adjusting the acidic level of wine, once the alcoholic fermentation is finished, a second fermentation is needed, the so-called malolactic fermentation. Nonetheless, those wines destined to aging in oak barrels are not subjected to malolactic fermentation since some Galician wines (e.g. “Rías Baixas”) do not have stability for aging during large periods.

After fermentation, wine is racked —several times to separate wine from lees— and clarified to remove suspended solids. Thereafter, wine is stabilized (4-6°C for 10 days) with the precipitation of salts of tartaric acid, which are not soluble at low temperatures. Finally, wine is subjected to filtration for removing any impurity before bottling. At this point, a new vintage can already be tasted and commercialized. Despite the fact that the process may vary considerably based on the winery, it generally takes several months (young wines), being bottled according to market demands.

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Chapter 2

Introduction to fishing sector

Summary

Despite technological improvements and humankind evolution, fishing is the unique hunting activity maintained at industrial level nowadays. The first evidences of fishing date from the latter Paleolithic period, being one of the triggers for changing from nomadic to settled communities. Fishing evolved rapidly, targeting new species and new fisheries, from coastal line to deep waters. In addition, linked to these activities, a processing industry was established to transform fish into more elaborate —mainly— food products, extending also shelf-time. Currently fishing has to deal with the problems derived from stock overexploitation and increasing operational costs. Thus, the reduction of the fleet and reconverting the sector are some of the current challenges.

In this context, Galicia as a prominent coastal region highly depends on the fishing sector, being the main fishing region in Spain and one of the most important at European level. Regarding the seafood processing industry, Europe is the second producer worldwide, where Spain and, therefore, Galicia, has a relevant role as main producer area.

Index

2.1 Fishing and processing industry

2.1.1 Brief History of fishing

Fishing has evolved dramatically since its first evidences in the late Paleolithic period, about 30000 BCE, using tools made of bones to fishing (Rincon, 2006). Nonetheless, it is the only hunting activity maintained at industrial level currently. At that time, when human communities depended exclusively on hunting —specially mammals— and were nomadic, fishing —freshwater and marine, close to the coastal line— allowed establishing settlements close to this resource since fish availability was guaranteed. In this way, some early human settlements are related to fish access in fluvial communities (Gartside and Kikegaard, 2007).

As technology evolved after the Neolithic period, fishing became a relevant activity to nourish early human settlements. Thus, fishing was one of the most important activities in the ancient Mediterranean civilizations such as Egypt, Greece, or the Roman Empire, being an important sector in terms of trade and food supply (Bekker-Nielsen, 2010). Furthermore, linked to the fishing industry, fish preservation methods (e.g. salted and dried) were developed and thereby, an important ancillary industry —being the beginning of the current fish processing industry.

Romans spread their fishing techniques throughout their conquered territories (Bekker-Nielsen, 2010). For instance, the Celtic culture began to entail Roman fishing tools —lines and nets— increasing their fishing effort offshore due to the fact that this pre-Roman cultures limited their activity to fishing along the coastal line and shellfish-gathering.

After the collapse of the Roman Empire, fishing activities began to expand to Northern Atlantic waters. In this way, the role played by Basque, Portuguese, and Viking fleets was remarkable, since they developed cod fishery, and, at the same time, the ancillary industries of cod drying and salting. The discovery of new fishing grounds —Newfoundland— by Basque fishermen increased cod stocks and resource availability, making cod the main species consumed (60% of all fish) in Europe by 1550 (Kurlansky, 1997).

Cod fisheries and related industries were a vast industry from the 16th century to the 20th century. Technological improvements —vessels, engines, and fishing gears— increased the pressure on cod stocks, which finished with the collapse of the cod fishery in Northern Atlantic waters. Moreover, the access to cod fishing grounds in waters close to Iceland triggered the so-called "Cod Wars" between UK and Iceland in the 50s and 70s (Jóhannesson, 2013). However, not only cod fishery triggered an international dispute regarding access to fishing grounds. In this regard, the Atlantic halibut (*Hippoglossus hippoglossus*) fishery in the Grand Banks (Southeast of Newfoundland waters and beyond of Canadian EEZ[1]), which were traditionally exploited by Galician and Basque fleets since the 16th century, was matter of dispute between Spain and Canada during the 90s —the so-called "Halibut War" (Parente, 1995). The rationale behind this was the overfishing on this ground and the Spanish fleet refuse —with the backing of the European Union— to comply quotas established by the Northwest Atlantic Fisheries Organization (NAFO).

Furthermore, it should be highlighted the importance of whale fishing in Northern Atlantic waters. Thus, the first evidences of commercial whaling were in Iberian Peninsula —the Basques, again— in the 10th century (Ellis, 1999). Quickly, whaling was spread all over the European coastal regions washed by the Atlantic Ocean —France, Denmark, UK, Greenland, and Iceland. However, whaling expansion and hunting techniques —which consisted in wounding and hunting the calves to approach the mother— implied the North Atlantic right whale (*Eubalena glacialis*) population depletion and, therefore, the industry decay *circa* the 17th century.

During the industrial revolution, the fishing industry changed dramatically with the developing of trawls. The introduction of steam-powered engines in fishing vessels allowed to drag the bottom of the sea and increase the catches significantly. The first steam-powered trawler was

[1] Economic Exclusive Zone (EEZ) is the sea area prescribed by the United Nations Convention on the Law of the Sea. This zone comprises the limits of the jurisdictional waters of a given country, establishing its limit in 200 nautical miles offshore from the territorial sea baseline.

built in UK in 1881 (Gartside and Kikegaard, 2009), which was rapidly extended in the North Atlantic waters by the end on the 19th century.

Thereafter, the next major change happened in the mid-20th century with the introduction of internal combustion engines —mainly diesel. At this time, it should be also remarked that the introduction of on-board refrigeration systems —which allowed to preserve catches and increase fishing days per tide— and factory vessels that enabled processing the catches on-board —gutted, filleting, packaging, etc. – were a regular practice in many fisheries.

Nowadays, the fishing industry has to face new challenges such as the continuous reduction of stocks —and, therefore catches— and the increasing cost of fuel, making difficult to get fishing as a profitable activity. Thus, the fishing industry —mainly large factory vessels— is changing towards new fisheries and target species, as well as alternative fuels. In this way, liquefied natural gas (LNG) may be an attractive technological option due to the existing inland infrastructure —for instance, the main ports worldwide are connected to the natural gas grid— and the availability of hybrid vessels powered by both diesel and LNG (Tellkamp, 2015).

2.2 The fishing sector worldwide

Fishing effort worldwide is linked to population growth. The exploitation of new fishing grounds or species because of overexploitation of the traditional ones to meet population demands has led to the current state of fisheries, in which roughly 80% are overexploited or fully exploited (FAO, 2010).

Figure 2.1 depicts the increasing fishing effort during the past 60 years as well as the importance of aquaculture. Marine species represented the main catches worldwide in 2012, accounting for 87% of total catches —the remaining percentage is related to inland captures. However, it should be noted that certain difficulties remain when it comes to obtain reliable data regarding inland captures (FAO, 2014).

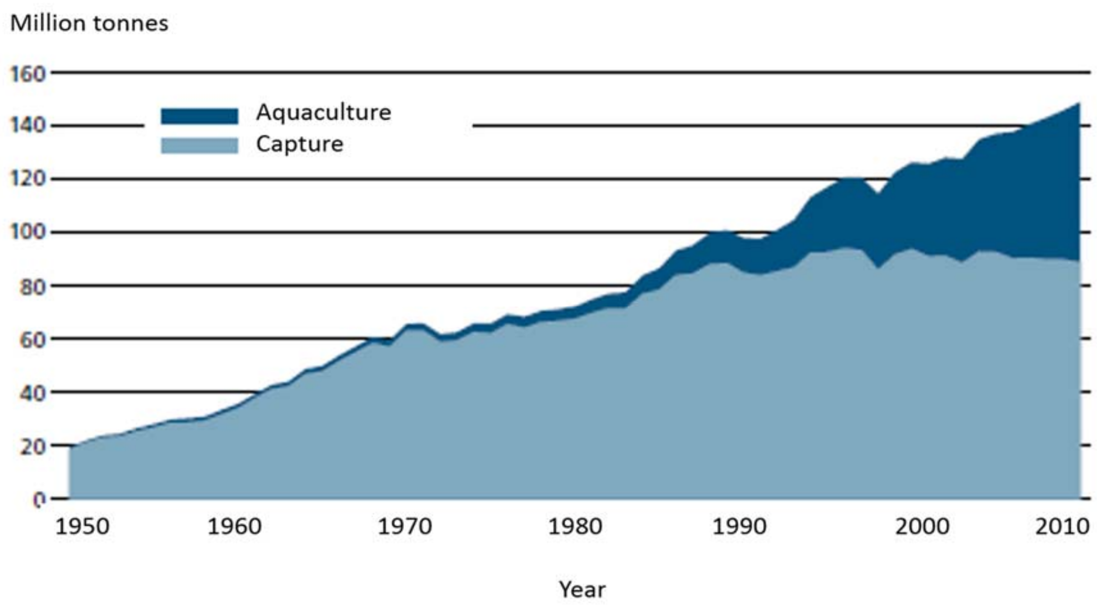

Figure 2.1 World capture fisheries and aquaculture production. Adapted from FAO (2010)

Per nation, marine catches were led by China —representing 17% of captures— in 2012, followed by Indonesia (7%), USA (6%) and Peru (6%) (FAO, 2014). Attending to the origin of catches, that means, the fishing area, the catches were centered in the Pacific Ocean, concentrating circa 60% of them —fishing area codes: 61, 67, 71, 77, 81, and 87 (Figure 2.2). Within the Pacific Ocean, the fishing area "Northwestern Indian Ocean" (FAO area 61) represented 27% of global catches. The second fishing area in relative importance in terms of landed capture was the Atlantic Ocean with 24% of the captures. More specifically, the fishing area named "Northeast Atlantic" (FAO area 27), with 10% of the total captures worldwide, constituted the main subarea in this ocean.

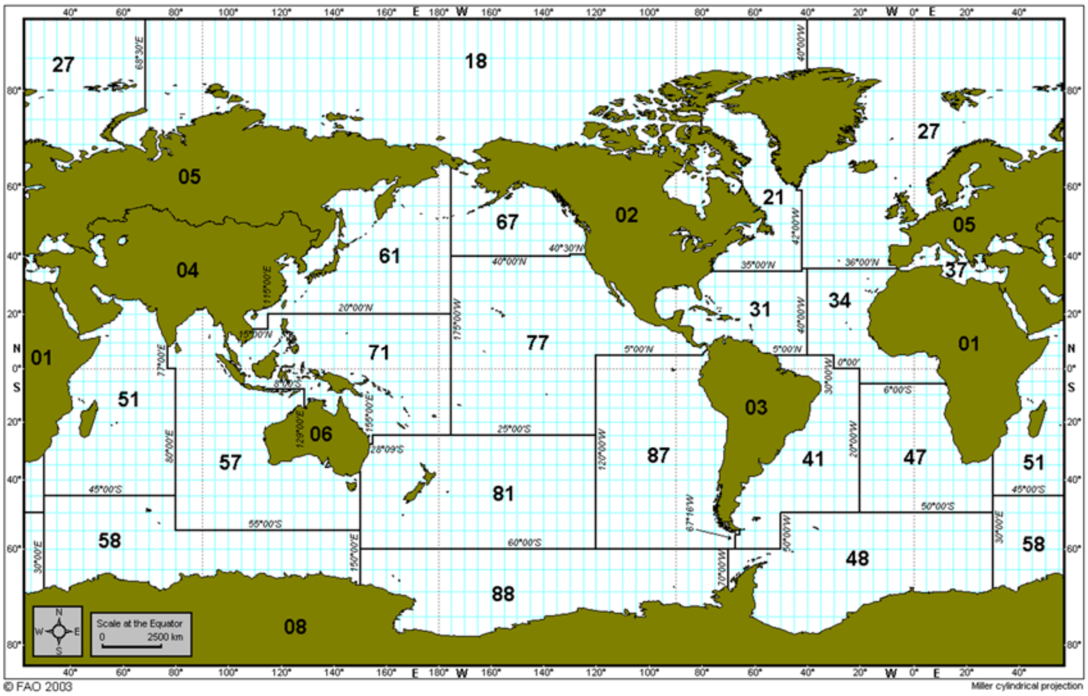

Figure 2.2 Major fishing areas. Source: FAO (2003)

By species, anchoveta (*Engraulis ringens*) was the main target species, representing roughly 6% of global captures in 2012. However, an important decrease in anchoveta catches in 2012 was identified in comparison to the catches in 2011, which meant 10% of the captures (FAO, 2014), due to the large interannual variations of this specie. After anchoveta, Alaska pollock (*Theragra chalcogramma*), skipjack tuna (*Katsuwonus pelamis*), and Sardinellas nei (*Sardinella spp*) were the most caught species, representing 4%, 3.5%, and 3%, respectively.

Regarding fish trade, the European Union (EU) is still the main market (Maneiro, 2015), being the leading importer of fishery and aquaculture products in 2012. In this regard, whereas imports represented 42% worldwide, exportations were 28%, which brings to light the relevance of EU role within fishing market. By countries, the EU's main suppliers —in market value— were Norway (20%) and China (8%) in 2012 (European Commission, 2014). Furthermore, the EU's main customers were United States and Norway, 10% and 9% of the sales value in 2012 respectively (European Commission, 2014).

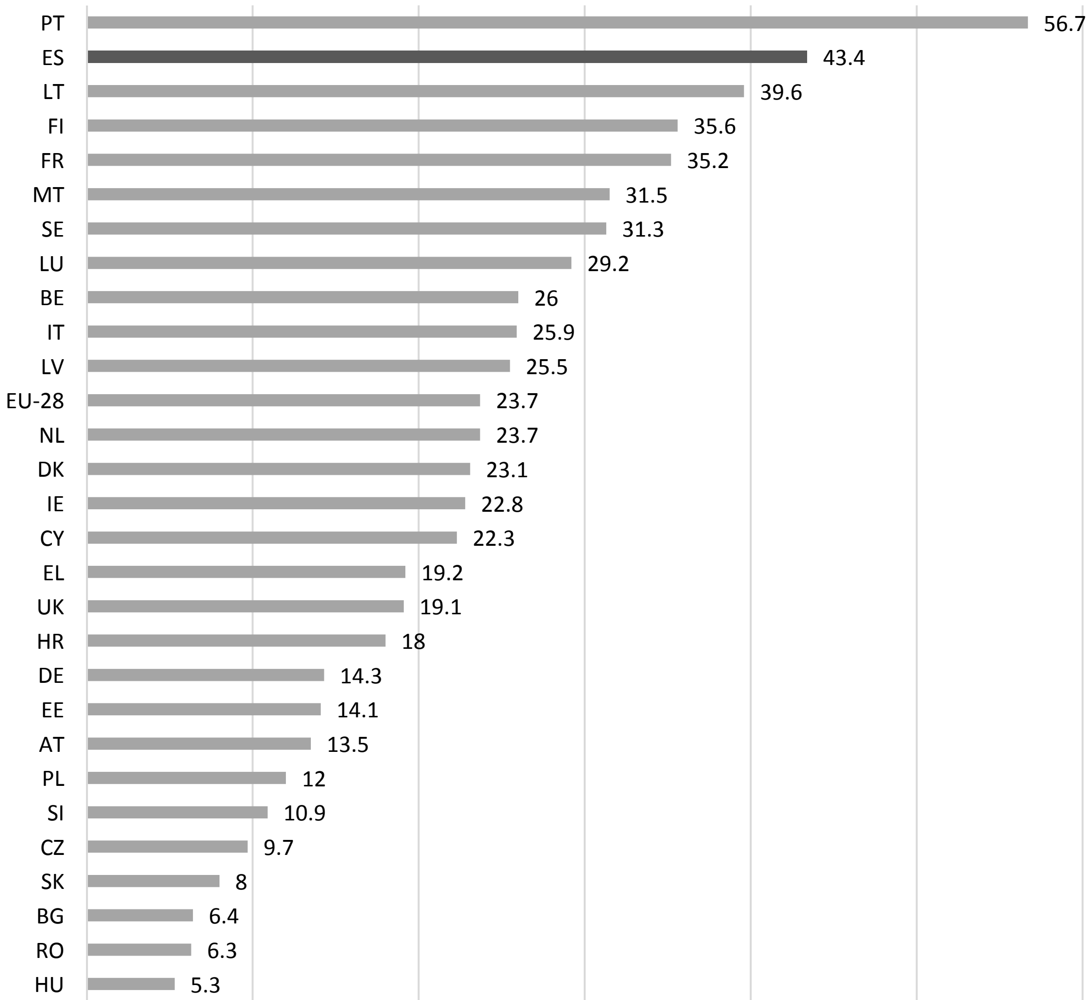

Figure 2.3 Fish consumption per capita in the European Union in 2010. Adapted from Maneiro (2015)

Fish products are a very important source of protein and an important part of a healthy diet. Thus, whilst average worldwide fish and seafood consumption was 19.9 kg/(person·year) in 2010, the EU average consumption was 23.7 kg/(person·year) (European Commission, 2014;

Maneiro, 2015). Within EU, fish consumption is led by Portugal and Spain, 56.7 and 43.4 kg/person/year, respectively (Figure 2.3).

2.3 The fishing sector in Galicia

2.3.1 The fishing sector

Traditionally, fishing capacity is higher than the availability of fish stock worldwide (UNEP, 2011). In fact, the EU's fishing sector is not alien to this trend because of the overexploitation of the main stocks (Figure 2.4) and, therefore, the EU's fishing fleet has been reduced by 19,284 vessels for the sake of reducing fishing overcapacity during the last 20 years (Maneiro, 2015). Nonetheless, it is still an issue within the EU's fleet, especially in countries such as Spain, Portugal, Ireland and UK (Figure 2.5).

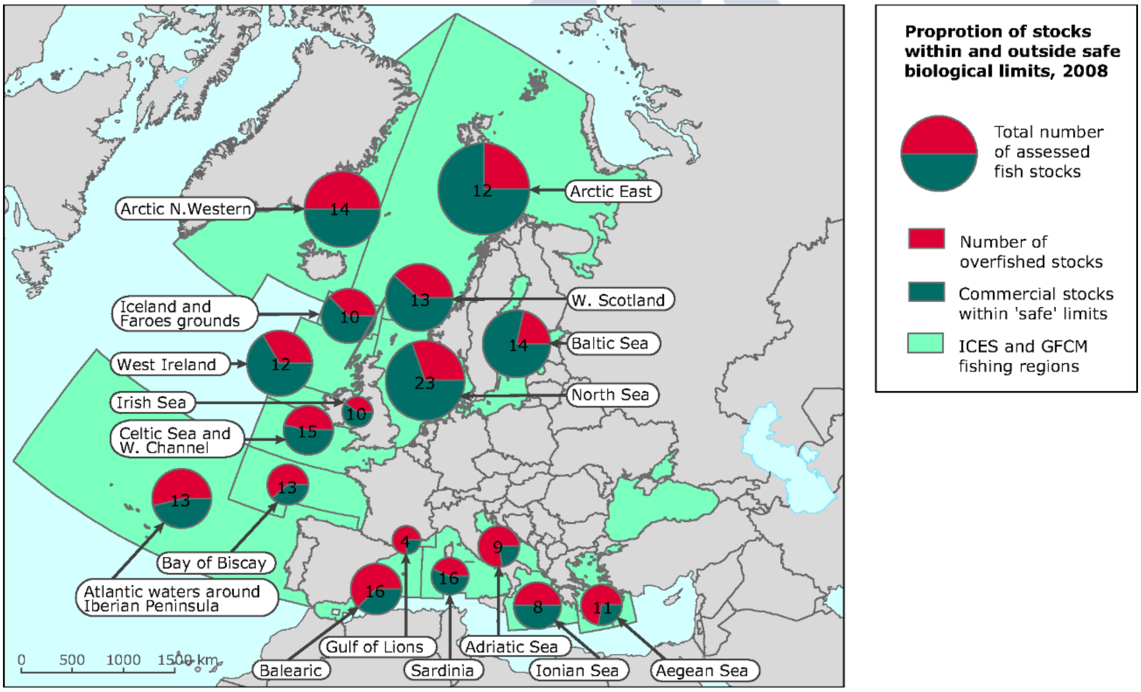

Figure 2.4 Status of the main EU fishing stocks in 2008. Source: EEA (2010)

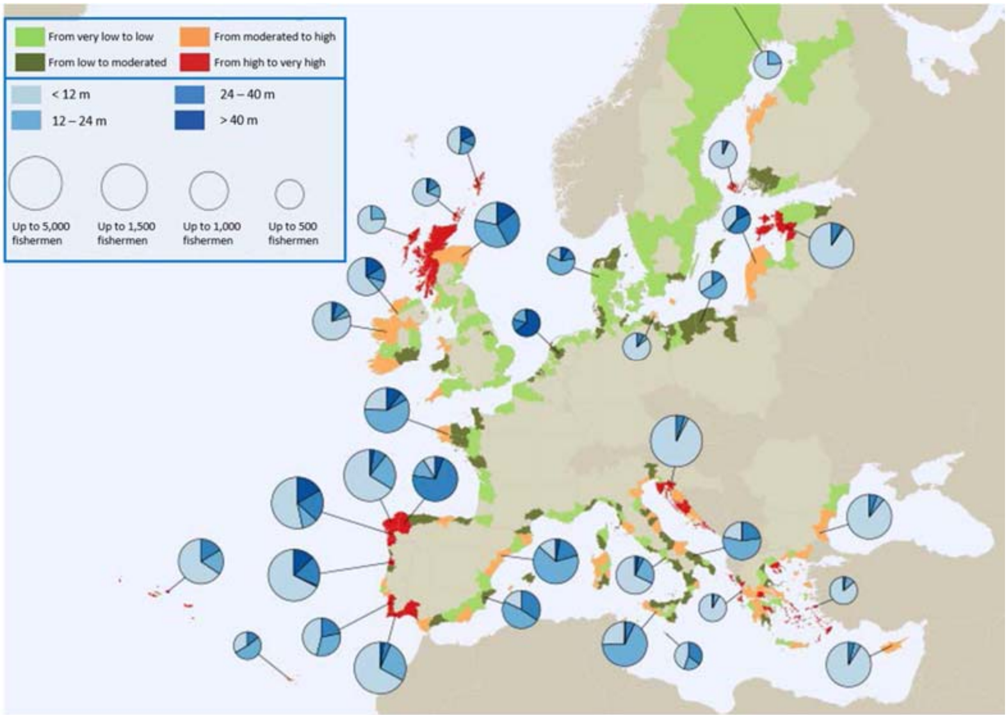

Figure 2.5 EU fishing capacity and features by region. Adapted from Maneiro (2015)

Table 2.1 presents the main data for the EU, Spanish, and Galician fishing fleets in 2012. It should be noted that in the EU-27 context (i.e. before the admission of Croatia), the Spanish fleet represented in number of vessels and capacity (Gross Tonnage, GT) 12% and 23%, respectively. Furthermore, the Galician fleet accounted for 6% in number of vessels and 10% capacity (Gross Tonnage, GT) whilst, at national level, was 46% and 42%, respectively.

Table 2.1 Features of the EU-27, Spanish, and Galician fishing fleet in 2012. Adapted from Maneiro (2015)

Fleet features	**EU-27**	**Spain**	**Galicia**
Number of vessels	80,240	9,986	4,603
Capacity (thousands GT)	1,635	382	159
Engine power (thousands kw)	6,231	866	286
Catch revenue (millions €)	6,871	1,985	677

The fishing grounds for the Spanish fleet are divided into three large categories based on the jurisdiction of the waters: i) National fishing grounds; ii) EU fishing grounds; and iii) International fishing grounds.

- National fishing grounds: includes those waters from the coastline up to 200 miles offshore (corresponding with the Spanish Economic Exclusive Zone, EEZ). Within these waters, the national fleet concentrates its fishing effort and catches. Simultaneously, it is divided into eight fishing areas over the Spanish coast (Figure 2.6)

Figure 2.6. Spanish fishing grounds within its EEZ

- EU fishing grounds: after Spain's entry into the EU, the fleet gained access to new fisheries. Fishing in these waters implied that the Spanish fleet had to meet a series of EU requirements: fishing quota based on country, total allowed catches (TAC) revised yearly depending on scientific surveys, fishing gear used, species and sizes caught, as well as the establishment of an EEZ of 12 miles. EU fishing grounds correspond with FAO fishing area 27 (see Figure 2.2 above).
- International fishing grounds: the Spanish fleet can fish in other areas of the world through agreements with third parties and the EU. In return, they receive economic and

technical aids, the development of mixed companies, and trade concessions. Thus, the list below shows the main areas where the Spanish fleet has legal access to fisheries:

- Northwest Atlantic (FAO area 21)
- Eastern Central Atlantic (FAO area 34)
- Southeast Atlantic (FAO area 47)
- Southwest Atlantic (FAO area 41)
- Western Indian Ocean (FAO area 51)

Delving into the main characteristics of Galician fleet in 2013, Table 2.2 shows the number of vessels per fishing ground, following the classification described above. Thus, the National fishing ground —Cantábrico-Noroeste— represented 95% of vessels, most of which fish using minor gears (Maneiro, 2015). Beyond Spanish waters, the EU fishing waters represented 2% of vessels whilst for International fishing waters was 3%. However, whenever the fleet is analyzed in terms of capacity and engine power, its structure changes considerably. Hence, the National, Community, and International fishing ground represented 32%, 16% and 52% in 2013, respectively. Regarding power, the National fishing grounds meant 52% whilst the Community fishing ground and International fishing ground 13% and 35%, respectively. Also, it should be remarked the differences among fisheries based on the crew. Thus, the National fishing grounds had a total of 9,981 people on-board (80%), mainly represented by minor gears, which meant 55% out of the total. The Community and International fishing grounds meant 1,249 and 1,322 people on-board, respectively.

Table 2.2 Galician fleet features. Adapted from Maneiro (2015)

Fishing ground	Vessels	Capacity (GT)	Engine power (hp)	Crew
National waters	**4,377**	**49,454**	**201,718**	**9,981**
Trawling	83	18,811	37,113	794
Minor arts	4,017	8,588	95,943	6,914
Purse seine	155	6,114	33,323	1,326
Gillnet	38	2,140	6,426	266
Bottom long-line	25	1,265	4,764	194
Surface long-line	59	12,537	24,149	487
Community waters	**87**	**25,496**	**50,392**	**1,249**
Trawling	39	12,693	24,185	570
Bottom long-line	48	12,804	26,207	679
International waters	**119**	**81,761**	**132,208**	**1,322**
Trawling	38	45,272	67,132	675
Purse seine	3	8,002	13,525	
Surface long-line	78	28,487	51,551	647
Total fleet	**4,583**	**156,713**	**384,318**	**12,552**

2.3.2 The seafood processing industry

Coastal regions are highly dependent on the fishing sector in terms of revenue and employment. Thus, the importance of this sector in regions such as Galicia, Cantabria or País Vasco in Northern Spain is considerable. Therefore, linked to fishing, the seafood processing industry has developed enormously in these regions. Currently, the Spanish fishing and fish processing industry led the production at European level, and are the second producer of canned fish and seafood worldwide.

The fish processing industry is under the category 102 of the National Classification of Economic Activities (CNAE-2009) (BOE, 2007). The products elaborated by it can be grouped based on the processes involved, the product's final presentation, or the type of industry (ANFACO-CECOPESCA, 2015; MAGRAMA, 2014) as follows:

- Fresh/chilled fish products: includes fillets and other formats.
- Frozen fish products: includes products such fillets (halibut, cod, and hake), loins (hake and cod) or steaks (hake and swordfish). Currently, it is common to carry out some processing operations abroad since catches may be either imported from third countries or pre-processed on-board in fishing vessels.
- Frozen crustaceous: shrimps and prawns are the species largely commercialized within this category. As the previous one, part of the production is imported from third countries, where shrimps or prawns are caught or raised, and processed in Spain afterwards.
- Frozen shellfish: octopus, scallops, squid, and cuttlefish are the main species commercialized within this category.
- Smoked, dried, salty, or brine fish: the most relevant products are smoked salmon, dried/salty cod, and salted pilchards or anchovy.
- Canned fish: includes canning and other preparations that use different sauces or packaging formats. The main canned species are tuna, pilchard, and mackerel.
- Canned seafood: includes the canned products derived from mussels, cockles, clams, queen scallops and cephalopods.
- Semi-preserved products: mainly includes products derived from anchovy.
- Other prepared products: include other fish processed products such as breaded fillets, sticks or fingers.

The total production of the seafood processing industry was 805,000 t in 2013 (MAGRAMA, 2014). Figure 2.7 depicts the relative contribution of each type of seafood processed product. Thus, canned and frozen fish represented together roughly 60% of the total production. Individually, canned fish accounted for 36% of the production, of which 90% were tuna related products. Furthermore, frozen fish products represented 20% of total production, highlighting the importance of fillets, loins and steaks. In terms of economic revenue, canned fish represented circa 50% of the total, where tuna products represented 85% of revenue (MAGRAMA, 2014).

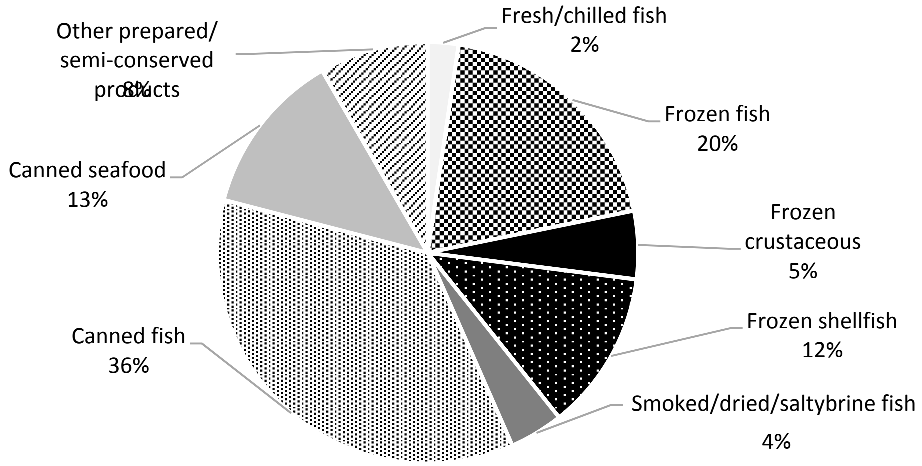

Figure 2.7 Seafood processing industry products share based on volume of production. Adapted from MAGRAMA (2014)

Concerning the number of companies and jobs, the fishing industry employed 18,324 people in 2012 (MAGRAMA, 2013). The number of companies involved in the fishing industry was 487 in 2012, being roughly 60% companies composed of 20 or less employees (MAGRAMA, 2013).

The fishing sector is featured —as other primary sectors— by a large consumption of raw materials (e.g. fuel) (Tyedmers et al., 2005). Moreover, if catches processing is added, the consumption of raw materials will be even higher due to addition of more stages to the process (Hospido et al., 2006). Fishing and seafood processing are not alien to this and, therefore, the application of methodologies to evaluate the environmental profile of primary sectors would be helpful to reduce their related impacts. In this regard, the next chapter delves into sustainability goals and how to meet them through the application of the environmental management tools.

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Chapter 3

Environmental management tools[1]

Summary

Given the necessity to meet sustainable development goals, environmental management tools have proved to be helpful in the evaluation of the environmental performance of human activities. Thus, Life Cycle Assessment (LCA), a standardized environmental management tool, is widely used to assess the potential environmental impacts associated with products, processes, or services. In this thesis, LCA and other complementary tools (Data Envelopment Analysis, Carbon Footprint, Land Use, etc.) are used to carry out the environmental profile of wine and seafood sectors. Additionally, a brief bibliographical review on these sectors —focusing on the main features of LCA methodology— is conducted. Finally, the objectives and structure of this dissertation are explained.

[1] Villanueva-Rey, P., Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2014. The use of carbon footprint in the wine sector: Methodological assumptions, in: Assessment of Carbon Footprint in Different Industrial Sectors, Volume 2, EcoProduction. Springer Singapore, Singapore, pp. 269–290.

Index

3.1 Sustainable development

The first concerns about pollution and environmental issues were related to unpleasant odors, mainly derived from wastewaters and sewage systems —which were, at the same time, origin of diseases. In the 19th century, the industrial revolution brought along environmental issues derived from air pollution. By this time, the first movements related to the conservation of forests and wildlife also raised, establishing Natural Reserves and National Parks. The first place with this category and protection level was the Yellowstone National Park (Wyoming, USA) in 1872.

The cultural movements raised in USA during the late 50s established the bases of environmentalism. The publication of *Silent Spring* by Rachel Carson (1962), which claimed the devastating effects of the indiscriminate use of pesticides —mainly DDT[2]— on ecosystems, brought to light the environmental problems related to pollution associated to human activity. Afterwards, the publication of *The Population Bomb* by Paul Ehrlich (1968) —who addresses the environmental problems to the overpopulation of the planet— entailed a change in people's perspective towards resources and environment. In 1970, the United States Environmental Agency (EPA or USEPA) was founded for the assessment and management of environmental problems, as a Congress action in response to the environmental awareness of the society —especially due to Rachel Carson's book.

During the decade of 1970, the oil crisis led by Arab countries triggered the rise of crude oil barrel price, with an enormous impact on global economy —extremely dependent on fossil fuels. This fact entailed a milestone in people and authorities towards fossil fuels dependency and the necessity to search for renewable energy sources. Also during the 70s, it was discovered the

[2] Dichlorodiphenyltrichloroethane (DDT) is an organochloride insecticide used indiscriminately to treat crops and eradicate malaria after World War II. DDT usage as pesticide was banned because of its damage on wildlife and humans, being only allowed to use it for disease controls.

effect of CFCs[3] on ozone layer depletion. However, CFCs were not completely forbidden until 1989, when the Montreal Protocol was ratified.

In 1983, the World Commission of Environment and Development (WCED) was established by the United Nations (UN) to develop proposals and solutions to deal with environmental deterioration of the human environment and natural resources. The WCED was headed by Gro Harlem Brundtland, former Norway Prime Minister, due to this fact it was also known as the Brundtland Commission. The WCED was dissolved after releasing *Our Common Future* in 1987, also referred as the *Brundtland Report*, where was firstly introduced and defined to the public the term "sustainable development" as "*the development that meets the needs of the present without compromising the ability of future generations to meet their own needs"* (World Commission on Environment and Development, 1987). In this way, sustainable development implies resources use involving environmental, social and economic factors (Figure 3.1).

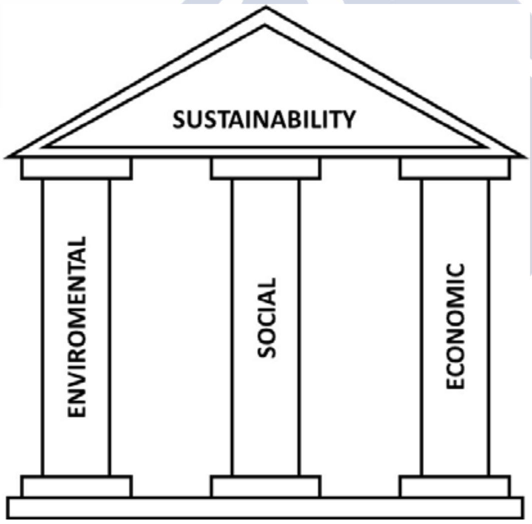

Figure 3.1. Schematic representation of the three pillars of sustainable development

[3] Chlorofluorocarbons (CFCs) are organic compounds widely used as refrigerants, propellants (aerosol applications) and solvents. Due to their effects on ozone layer, they were banned and substituted by other products, especially CHCFCs (hydrofluorocarbons).

Afterwards, following Brundtland Commission call for a large international meeting to define initiatives and set goals, The United Nations Conference on Environment and Development (UNCED) was held in 1992, also known as the Rio Summit or Earth Summit. The main outcomes from the summit were: The Convention on Biological Diversity, The Framework Convention on Climate Change, Principles of Forest Management, The Rio Declaration on Environment and Development, and Agenda 21 —an ambitious strategy to promote sustainable development. Following this Earth Summit, other meetings were held in Johannesburg (2002) and Rio de Janeiro (2012) —also known as *Rio+20*.

The increasing awareness of climate change and global warming during the decade of 1980 led to the creation of the Intergovernmental Panel on Climate Change (IPCC), as a scientific body under the umbrella of UN, in 1988 —aiming at producing reports that support the United Nations Framework Convention on Climate Change (UNFCCC). This context gave rise to the Kyoto Protocol in 1997, which committed the state partners to reduce greenhouse gas (GHG) emissions at least 5% in comparison to levels in 1990 over the 2008-2012 period. After finishing Kyoto's first round commitment, a second was proposed for the 2013-2020 period, but the withdrawal of some countries, which had signed the first commitment, makes difficult to meet Kyoto's goals and thereby reduce GHG emissions.

The aforementioned actions are only a minor part of the human impacts on environment and nature. Thus, ozone layer depletion, global warming, and pesticides are significant examples of our effect on environment. The road towards sustainable development passes through modify current economic and social patterns of consumption to avoid resources depletion and human impact on nature. Sustainability entails resources consumption at a rate that does not jeopardize their depletion without exceeding natural replenishment limits. In this way, a set of environmental management tools have been developed aiming at getting sustainability goals and identifying the impacts related to a given product, process, or service (Feijoo et al., 2007).

3.2 Environmental management systems

Environmental management systems (EMS) are focused on all activities that affect environment, aiming at reducing environmental risks and identifying the impact sources. Environmental management is becoming useful for companies, organizations, and institutions to improve their image towards public, increase business opportunities, and reduce production costs since EMS allow identifying inefficiencies within production processes. During last 20 years, there has been a proliferation of tools for this purpose (Table 3.1) (Thompson, 2014), adopting —most of them— the concept life-cycle within their definition. In this way, Life Cycle Assessment (LCA) has demonstrated to be an appropriate management tool to address environmental impacts related to products, processes, and services —being a widely accepted by scientific community and authorities.

Table 3.1. Selection of environmental tools

Environmental management system	Acronym
Environmental Risk Assessment	ERA
Environmental Impact Assessment	EIA
Environmental Auditing	EA
Input-output Analysis	IOA
Life Cycle Costing	LCC
Life Cycle Assessment	LCA
Social Life Cycle Assessment	SLCA

3.2.1 Life Cycle Assessment

Given the goals of sustainability, LCA is a useful and standardized[4] methodology that allows addressing the impacts of a given activity. Due to its holistic perspective —taking into account all the stages from a cradle to grave approach— LCA evaluates the potential impacts related to any economic activity including raw material extraction and processing, distribution, retailing,

[4] LCA methodology is under the umbrella of ISO standards 14040:2006 and 14044:2006

wholesaling, use or consumption, re-use, recycling, and final disposal. Nonetheless, LCA studies do not always cover all stages, using the approaches of cradle to gate, gate to cradle, or gate to gate. Among the wide range of LCA applications, the following can be highlighted:

- The detection of opportunities to improve the environmental profile of the system under study through its life cycle.
- Being a source of information for decision-makers, including governments, business, and research.
- Marketing purposes, including environmental information for a specific product through eco-labels to get consumers attraction.
- The selection of environmental indicators to identify the environmental profile.

LCA methodology follows the ISO guidelines 14040 and 1044 (ISO, 2006a and ISO, 2006b), consisting in four stages (Figure 3.2): i) goal and scope definition; ii) inventory analysis; iii) impact assessment; and iv) interpretation.

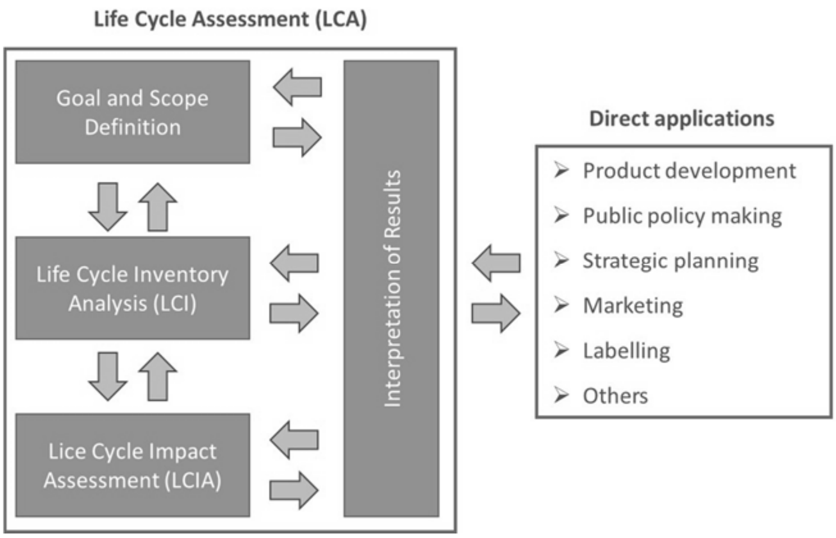

Figure 3.2. Framework for Life Cycle Assessment. Adapted from ISO (2006a)

Goal and scope definition

The goal and scope of an LCA is the first stage to be performed. Within this stage, the objectives and purpose of the assessment, the detailed description of the system under study and its limits, have to be exactly defined. However, given the inherent features of LCA, the scope may be changed during the study to improve analysis quality, or other constraints such as economical or temporary.

Furthermore, within this stage two key elements for LCA deserve special attention. Firstly, the Functional Unit (FU), which is defined as the basis of calculation to which the input and output data are referred. In addition, it should reflect the function/s of the product/s or system/s under study and be consistent with the goal and scope defined previously, as well as easy to reproduce and measure.

Secondly, the selection of system boundaries implies the inclusion or exclusion of a given process in the LCA analysis. In this regard, the criteria to establish the system boundaries have to be explained and discussed —especially for excluded processes. The system boundaries can be geographical, which means the area where the system under study is located; temporal, the period of time to which the LCA is referred; and physical, defining the level of detail of the study.

Inventory analysis

The Life Cycle Inventory (LCI) implies data gathering —qualitative and quantitative ones— to quantify the materials and energy flows related to the system under study. Data sources to carry out the LCI should be primary, which implies a higher effort in terms of time and costs, coming from direct measurements. However, those data impossible to obtain from a primary source must be collected from scientific literature, public sources, and/or LCA databases such as

Ecoinvent[5] (Weidema, et al., 2013). Also, calculation procedures and assumptions have to be detailed and explained throughout the study.

Furthermore, when two or more products are obtained from a given system or process —for instance, multifunction products—, it is necessary to allocate the input or output flows and, therefore, the potential environmental impact. Following ISO guidelines, allocation should be avoided as long as possible. To do so, either system expansion or division of multifunctional systems into sub-processes should be considered. Allocation procedure must be well documented and discussed since study accuracy and consistency depends on it. There are several allocation approaches, but the most widely used —and recommended by ISO standards— are mass and economic allocation. However, other allocation approaches —based on analyzed product features— can be used. For instance, when food products are studied, other physic features such as energy or protein content can be used for allocation.

Impact assessment

After performing data for the inventory, the next step is the calculation of Life Cycle Impact Assessment (LCIA), which implies the translation of impacts related to the LCI into environmental indicators. According to ISO standards (ISO, 2006a,b), LCIA consists of three mandatory stages: selection of impact categories, classification and characterization; and the following optional stages: normalization, weighting and grouping.

1. The selection of impact categories should be selected according to the goal and scope of the study. They represent the environmental issues related to the product under study such as Global Warming Potential (GWP). It should be highlighted that it is sometimes necessary a bibliographic review in order to make the study comparable with others in the same field.

[5] Ecoinvent is the world leading database that provides up-to-date Life Cycle Inventory data to perform LCA.

2. Classification is the assignment of the inventory data according to the impact category they affect. As an example of classification, emission of gases such as CO_2 and CH_4 would be assigned to the impact category of GWP. It should be noted the fact that inventory data might have effect on more than one category.
3. Characterization is the conversion of impacts within each category to the same units, obtaining one single value per impact category. This conversion uses characterization factors which, at the same time, are the indicators of the impact category considered. In this way, emissions of CO_2 and CH_4 are referred to $CO_{2\ eq}$ (CO_2 equivalents), which is the indicator of the GWP impact category.
4. Normalization implies calculating the magnitude of category indicator results relative to reference information.
5. Grouping is the aggregation of the impact categories into one or more sets.
6. Weighting aggregates the results obtained through the different impact categories selected previously into one single impact score. For this purpose, weighting factors — based on subjective value judgments— are used.

Interpretation

Results interpretation is the last stage in LCA, which allows identifying the process/es responsible/s for the environmental impacts as well as the options to reduce them. Interpretation also entails a deep review of LCA stages to check the assumptions and data quality in addition to the limitations and recommendations.

3.3 Life Cycle Assessment applied to winegrowing and winemaking

LCA methodology has been widely applied to a considerable amount of wine processes, including wine farms, appellations, viticulture practices or supply chains (Rugani et al., 2013), showing that despite the relative importance of plant protection agents and fertilizers in the viticulture stage of winemaking, there are many other activities throughout the production and supply chain that lead to important environmental burdens.

While LCA studies provide environmental information for a cluster of environmental dimensions, referred to as impact categories, in many cases some of these categories are individually analyzed due to the particular interest that they may generate in the production system under study (de Haes 2006; Weidema et al., 2008). Moreover, current worldwide environmental concerns, such as water scarcity or global warming have created increasing interest in using assessment methods that address these impacts specifically. Consequently, it is common to see life-cycle studies using single indicators, such as Carbon Footprint (CF) to measure emissions linked to climate change or Water Footprint to monitor the stress of water supply due to anthropogenic activities (Ridoutt et al., 2009; Weidema et al., 2008).

More specifically, CF has experienced an exponential proliferation in recent years due to the international concerns regarding climate change, as well as other derived factors, such as consumer awareness or the willingness of stakeholders to enhance product transparency and market quota through CF-oriented campaigns and certifications (Weidema et al., 2008; Laurent et al., 2012). In fact, most studies on CF in the wine sector have focused on assigning anthropogenic GHG emissions to the wine production and supply chain operations, for the identification of hot spots throughout the life-cycle of wine products or for communication purposes, mainly oriented towards stakeholders and consumers (Aranda et al., 2005; Benedetto 2013; Rugani et al., 2013).

As aforementioned, special attention will be paid to controversial issues in life-cycle thinking such as the selection of the system boundaries, goal and scope, functional unit (FU), allocation or assessment methods. In addition, a detailed description of Life Cycle Inventory (LCI) items will be provided, analysing the relevance of including them in the studies.

3.3.1 LCA studies of wine sector

3.3.1.1 Life Cycle Assessment in viticulture and vinification

From a viticulture and vinification perspective, LCA application is relatively recent, with the first studies dating from 2003. Nevertheless, wine LCA has shown an important proliferation in the past decade, as depicted in Table 3.2.

The geographical distribution of these studies is located mainly in Old World countries, mainly Italy, Spain, Luxembourg or Portugal. Nevertheless, some studies have also arisen in New World areas, such as Canada or New Zealand. Furthermore, it should be noted that most of the studies focus on single wine appellations (Point et al., 2012; Neto et al., 2013), individual wineries (Benedetto, 2013). In fact, only two recent studies have attempted to provide specific characterization values beyond the appellation level. This is the case of Vázquez-Rowe et al. (2013a), who provide a carbon footprint calculation on a national level for Luxembourg, while making a cross-country comparison between Luxembourg, Italy and Spain. In addition, a recent review on wine carbon footprint, attempted to obtain a worldwide average value for GHG emissions linked to the consumption of a bottle of wine, by aggregating the carbon footprint results available in the literature (Rugani et al., 2013). Moreover, these results were scaled up to the entire worldwide production and consumption of wine worldwide, concluding that the wine industry represents approximately 0.3% of total GHG emissions (Rugani et al., 2013).

Regarding the different viticulture practices, most available studies have focused on conventional practices (Notarnicola et al., 2003; Carballo-Penela et al., 2009; Bosco et al., 2011) or on the comparison between organic and conventional farms (Pizzigallo et al., 2008; Kavargiris et al., 2009; Vázquez-Rowe et al., 2013a).

Table 3.2. Selection of the main publications in wine LCA

Study	Country	Viticulture management	Grapes/wine type	Methodology(s) applied
Notarnicola et al. (2003)	Italy	Conventional	Unspecific	LCA
Aranda et al. (2005)	Spain	Conventional	Unspecific	LCA
Ardente et al. (2006)	Italy	Conventional	Unspecific	POEMS* / LCA
Gonzales et al. (2006)	France	Conventional & Organic	Red wine	LCA
Niccolucci et al. (2008)	Italy	Conventional & Organic	Red wine	Ecological Footprint
Pizzigallo et al. (2008)	Italy	Conventional & Organic	Red wine	LCA & Emergy
Carballo-Penela et al. (2009)	Spain	Conventional	Unspecific	Carbon Footprint
Kavargiris et al. (2009)	Greece	Conventional & Organic	Pink wine	Greenhouse gases & Energy
Gazulla et al. (2010)	Spain	Conventional	Red wine	LCA
Notarnicola et al. (2010)	Italy	Conventional	Red wine	LCA
Barry et al. (2011)	New Zealand	Conventional	Unspecific	LCA
Bosco et al. (2011)	Italy	Conventional	Red & white wine	Carbon Footprint
Comandaru et al. (2012)	Romania	Conventional	Not specified	LCA & Water Footprint
Pattara et al. (2012)	Italy	Organic	Red wine	Carbon Footprint
Point et al. (2012)	Canada	Conventional	White & Red wine	LCA
Benedetto (2013)	Italy	Conventional	White wine	LCA
Vázquez-Rowe et al. (2013a)	Spain, Italy & Luxembourg	Conventional & Organic	White & Red grapes	LCA
Neto et al. (2013)	Portugal	Conventional	White wine	LCA

*POEMS: Product Oriented Environmental Management System

Finally, a wide range of existing wine types have been examined using either LCA or carbon footprint studies, although some contributions do not state specifically the wine class analyzed. Nevertheless, most studies have focused on analyzing either red or white wines (Ardente et al., 2006; Benedetto et al., 2013) of medium price range, although Vázquez-Rowe et al. (2013a) also examined sparkling wine and an expensive red wine with long ageing periods, and Neto et al. (2013) analyzed the environmental impacts related to Portugal's *vinho verde*.

3.3.1.2 Carbon Footprint

Carbon footprint (CF) analysis in viticulture and vinification has been mainly related to the extraction of the Global Warming Potential (GWP) impact category from the CML baseline 2000 LCA assessment method (Frischknecht et al., 2007) or using the IPCC assessment method (IPCC, 2007). Moreover, to date most studies have developed CF calculations within the framework of the ISO 14040 and 14044 standards (ISO, 2006a; 2006b). Nevertheless, in recent years a considerable number of CF protocols have arisen, including PAS 2050:2011 (BSI, 2011), the Greenhouse Gas Protocol (Initiative GGP 2011), Bilan Carbone (ADEME, 2010) or the specific ISO standard for carbon footprint: ISO 14067 (ISO, 2013). Moreover, the International Organization of Vine and Wine (OIV) has developed its own standards (OIV 2011).

3.3.2 Scope of wine LCA studies

The scope of existing studies is highly variable. In most cases, as can be observed in Figure 3.3, studies have mainly focused on the viticulture and vinification stages of the supply chain, despite the fact, as stated by Rugani et al. (2013), that these two stages are not necessarily the ones that contribute most to the total environmental impact. Nevertheless, it should be noted that the vineyard planting phase, previous to the viticulture stage, is not always included in these studies. In fact, a general observation throughout the available literature is that the initial phases of land preparation, vine nursing or vine planting are treated with care or directly through omission on most studies, suggesting that the availability of data at these stages of the process is scarce.

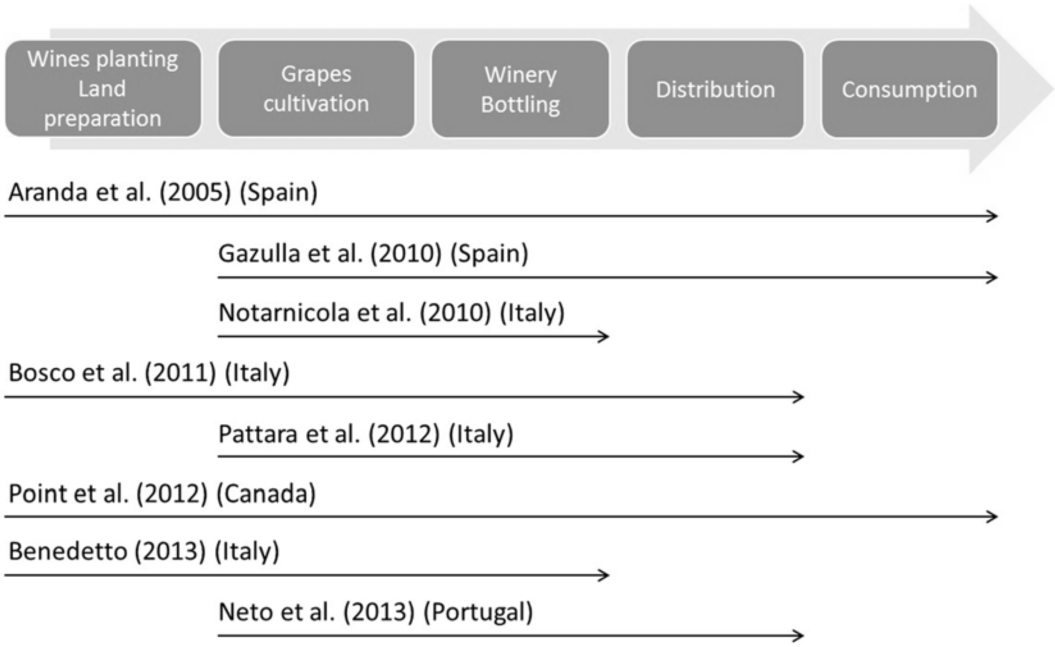

Figure 3.3. Scope of a selection of wine LCA studies

Regarding the perspective undertaken in these studies, an attributional approach was used in all of them (Rugani et al., 2013). Attributional approaches in LCA studies take into consideration the environmental impacts in a steady-state condition of the system under assessment, without considering the interaction that may occur with other interrelated production systems. As opposed to the attributional perspective, a more recent approach, namely consequential or prospective, aims at measuring the environmental consequences linked to changes in the production system, rather than monitoring direct emissions, based on a series of temporal or spatial-driven constraints and/or changes, among other potential shocks on the system under study (Sonnemann et al., 2011; Vázquez-Rowe et al., 2013b).

3.3.3 System boundaries

Grape production, given its annual characteristics, does not precise the establishment of temporal boundaries in a similar way to seasonal crops. Figure 3.4 shows a schematic representation of the system boundaries for the viticulture, vinification and bottling phases of wine production, up to the gate of the farm ready for transportation, distribution and consumption, including the most relevant operational activities that are undertaken throughout the production chain.

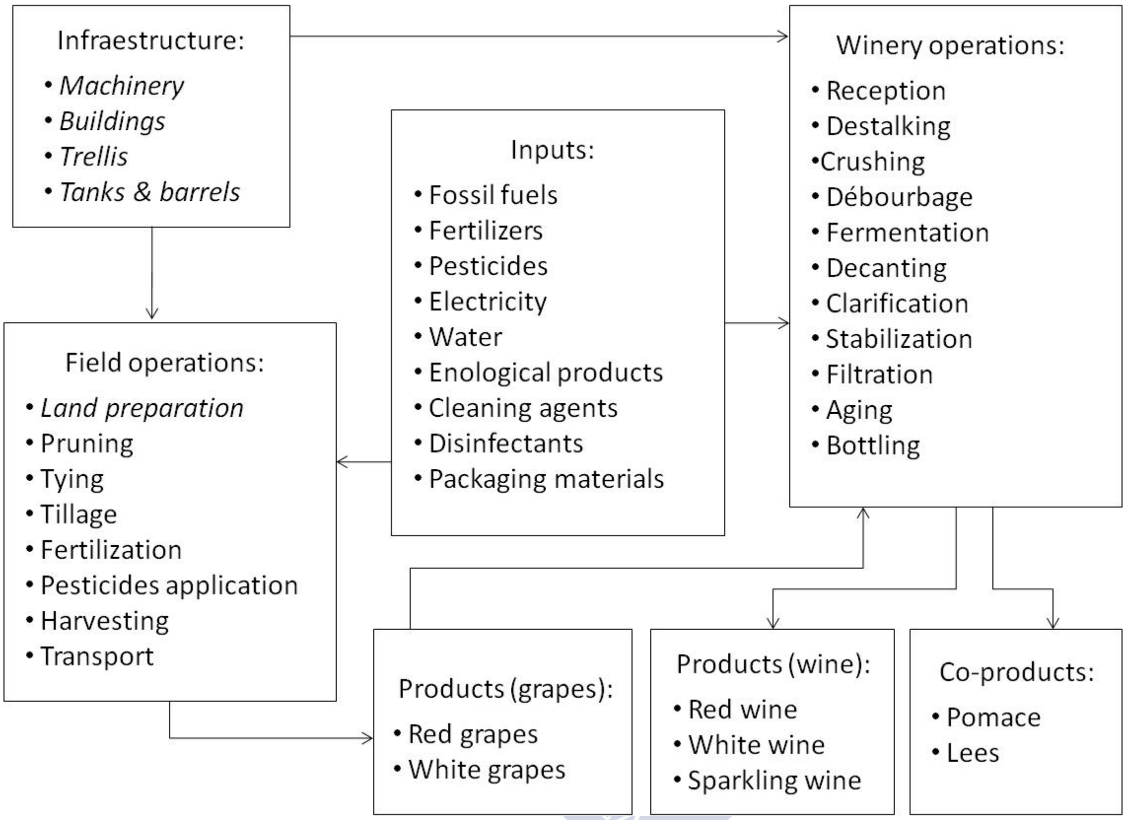

Figure 3.4. Schematic representation of a typical wine production system. Italic wording refers to operational inputs or processes commonly excluded from the inventories in wine LCA studies

Certain elements within these boundaries, such as infrastructure in the vinification stage are repeatedly disregarded in most studies (except in the case of Gazulla et al., 2010 and Vázquez-Rowe et al., 2013a). A similar situation occurs with the vine trellis since the wide variety of materials used for this type of infrastructure (i.e. wood, concrete, granite…) have shown to cause important differences in terms of final impact.

Machinery throughout these stages is also usually excluded when it comes to the production and transportation of the machinery itself, but included in terms of the use phase (i.e. diesel or electric consumption, etc.). Finally, vine nursing, land preparation and vine planting are phases prior to the viticulture stage that are not always included within the system boundaries. For instance, no studies reported in the literature report an LCI for vine nursing.

To sum up, all elements disregarded from the system boundaries seem to have two common denominators: i) difficulty in obtaining feasible data for these processes; ii) the scope and aims of the study allowed the exclusion of these stages (Neto et al., 2013; Rugani et al., 2013).

3.3.4 Functional Unit

The function of most wine LCA reported to date is basically oriented towards the environmental certification of the analysed product (i.e. the bottle of wine) or for an in-depth analysis of the main hot spots and, thereafter, improvement actions, throughout the supply chain. Therefore, an overwhelming majority of the analysed studies use the same functional unit (FU) to report the environmental profile of wine products: one standard wine package (usually glass) of 750 mL, since this is the main format used for sales (Aranda et al., 2005; Gazulla et al., 2010; Petti et al., 2010; Bosco et al., 2011; Point et al., 2012). Other studies use different volumes of wine (Gonzalves et al., 2006), whereas Notarnicola et al. (2010) used the percentage in alcohol volume and the hedonistic value of wine. Nevertheless, as can be seen here in Chapters 4, 5 and 6, the FU selected was the amount of grapes required to produce 750 mL of wine, rather than the selection of other options, such as based on cultivated surface.

After the analysis of the different functions and FUs available in the literature, it appears that the convenience of wine LCA has been limited to communication of results based on the final marketable product: wine bottles (Rugani et al., 2013). Therefore, it seems as if integral assessments of wine farms, appellations and wine region is still missing in the available literature, constituting an interesting field for future studies.

3.3.5 Allocation procedures

Beyond the main product derived from winemaking (i.e. wine), there are multiple by-products leaving the production system. Therefore, in many cases, always depending on the system function, it is necessary to allocate the co-products (Notarnicola et al., 2003). However, so far allocation has not been discussed in detail in most wine studies, since the entire burdens have been usually assigned to the main product (i.e. wine production). In other cases, the assessment was limited to the pre-harvesting phases and, therefore, no allocation was necessary.

In their study, Gazulla et al. (2010) allocated the environmental burdens to the different co-products based on their economic value. Anyhow, this allocation strategy derived in 98% allocation for the wine bottle, while the other co-products had an irrelevant role in the overall system. A more recent study by Bosco et al. (2011) chose mass allocation to allot the GHG emissions between the wine and stalk, skin and pip products. This implied that wine CF impacts were approximately 25%-30% lower than if all the impacts were allocated entirely to the wine product. Finally, other studies such as Notarnicola et al. (2003) excluded the treatment of by-products (i.e. compost from rasps or tartaric acid from marc) from the system boundaries.

According to a review provided by Rugani et al. (2013), the use of allocation in wine CF has been understudied due to the lack of adequate information to allow an expansion of the system boundaries. In fact, given the lack of information in many cases regarding the alternative production systems for by-products such as pomace, lees or press syrup, system expansion were considered not feasible in these processes (Gazulla et al., 2010).

3.4 Life Cycle Assessment applied to fishing and processed seafood

3.4.1 Life Cycle Assessment applied to fishing sector

The first LCA related studies were focused on energy consumption, which should not be strange because of the high dependency on fossils of fishing activities (Tyedmers, 2001). Traditionally, LCA has been applied to fishing sector worldwide, focusing on fleets —especially trawling and purse seining— whose target species were always those with high/medium economic value: cod, flatfish or tune (Vázquez-Rowe et al., 2012a). In addition, CF studies of fleet operations were also very common because of the abovementioned dependency of fishing on fossils fuels (Ziegler et al., 2003; Iribarren et al., 2011).

Furthermore, more recent studies have performed for other species and fleets as well as fishing gears; specifically, artisanal fleets (Ziegler and Food and Agriculture Organization of the United Nations, 2009), long liners (Vázquez-Rowe et al., 2011; Svanes et al., 2011) and creels (Ziegler and Valentinsson, 2008).

Regarding the species considered, in addition to the aforementioned species, new species and fleets were assessed in the last 5 years. Thus, it can be highlighted the studies carried by Vázquez-Rowe (2012b), who assessed the Galician fleet —trawling, purse seining, and long lining— focusing on the main species landed such as hake, horse mackerel, megrims, mackerel, and anglerfish.

3.4.2 Life Cycle Assessment applied to seafood processing

The number of LCA studies regarding seafood processed products is relatively low despite the varied type of products in the market. In this way, the product complexity —in terms of production processes— ranges from simple frozen fish, which only entails cleaning, gutting, and a basic packaging— to multi-ingredient products (canned fish, breaded fish, etc.) —to which other components from different origin are added—, becoming a constraint in the development of LCA. Additionally, in some cases it is difficult to follow the production chain of these products since many of them are partially processed on-board, making difficult to address the derived impacts. Table 3.3 depicts the main LCA studies —including Carbon Footprint and energy consumption— related to seafood processing.

Table 3.3. Selection of seafood LCA studies

Study	Country	Product analyzed	Methodologies applied
Ziegler et al. (2003)	Sweden	Frozen cod fillets	LCA
Hospido et al. (2006)	Spain	Canned tuna	LCA
Fikseaunet et al. (2007)	Norway	Fish sticks	LCA
Zufia and Arana (2008)	Spain	Pasteurized tuna with tomato	LCA
Sund (2009)	Sweden	Fish sticks	LCA & Energy
Winther et al. (2009)	Norway	Varied seafood delivered to consumer	Carbon Footprint & Energy
Iribarren et al. (2010)	Spain	Canned mussels	Carbon Footprint
Vázquez-Rowe et al. (2011)	Spain	Gutted fresh hake	LCA
Vázquez-Rowe et al. (2012c)	Spain	Frozen cephalopods	LCA

3.4.3 Scope of seafood LCA studies

Unlike LCA of wine production, LCA of fishing and seafood are commonly implemented taking into account all stages of the product under study —either from fish extraction to final product ready to dispatch at processing plant gate or including distribution and the subsequent phases. In this sense, LCA studies include phases such as distribution (Vázquez-Rowe et al., 2012c; Winther et al., 2009), wholesaling, retailing and consumption (Iribarren et al., 2010; Fikseaunet et al., 2007; Vázquez-Rowe et al., 2011).

Regarding the perspective of LCA: attributional or consequential, existing seafood LCA studies follows the attributional approach, making its assumptions under a retrospective perspective. Furthermore, consequential approach is limited to one single study (Thrane, 2004), which deals with the environmental burdens of flatfish base products.

3.4.4 System Boundaries

System boundaries generally follows ISO standards for LCA (ISO, 2006a, b), recognizing four stages in fishing LCA (Avadí and Freon, 2013): construction phase, maintenance, use, and end-of-life. Additionally, other studies include stages such as transportation before transformation or processing (Eyjolfsdottir et al., 2003, Ziegler et al., 2003).

Seafood processing might entail the inclusion of other components to the final product such as ingredients, packaging materials, etc. The common trend followed in LCA is the exclusion of infrastructure due to the difficulties to carry out the LCA of a whole processing plant and the negligible effects on final impact, as well as the necessity of performing an allocation of such installations during their complete life span and products manufactured, making necessary to consider system expansion.

3.4.5 Functional Unit

The selection of the functional unit highly depends on the scope of the study. For seafood processing, it ranges from the product's commercial presentation (Fikseaunet, 2007; Zufía and Arana, 2008; Iribarren et al., 2010; Vázquez-Rowe et al., 2012) —Chapters 9 and 10 of this dissertation— to standard consumption portions (Vázquez-Rowe et al., 2011). In addition, other studies follow more common FUs such as 1 kg of final product (Sund, 2009) or 1 kg of edible product transported to the wholesaler (Fikseaunet et al., 2007). It should be highlighted that some environmental assessments use different approaches to report their FUs. For instance, Muñoz et al. (2010) established their FU in terms of food intake per person and year. In this way, Chapter 11 of this dissertation establishes a FU based on protein amount of analyzed products. Furthermore, for fishing LCA, the most common FUs are referred to the bulk landings at port (e.g. 1 kg or 1 t landed fish) (Vázquez-Rowe et al, 2012a).

3.4.6 Allocation

Allocation procedure is a key element for fishing and seafood processing. Given the features of fishing fleets, targeting several species, allocation is needed to allot the impacts based on catches and/or final products. Thus, there are two alternatives: i) economic allocation, which implies the economic value of captures or secondary products (Pelletier and Tyedmers, 2011); ii) mass allocation, widely used for those fisheries where captures have similar economic value (Eyjolfsdottir et al., 2003) or processing systems where the final products have the same features and value. Additionally, most studies follow mass allocation to avoid volatile prices of captures (Vázquez-Rowe et al., 2011) and encourage producers to use by-products of processing industry (Winther et al., 2009).

Recently, new biophysical allocation approaches have been introduced in fisheries and seafood such as protein, energy content, or hybrid allocation procedure, which implies the combination of a biophysical (e.g. mass, energy, or protein content) and economic approach. This latter approach is used in Chapter 11 of this dissertation.

3.5 Objectives and structure of this dissertation

The goal of this doctoral thesis is to assess the environmental performance of wine and seafood sectors in Galicia through LCA and other complementary tools. Figure 3.5 shows a schematic representation of this dissertation.

Section I (Chapters 1, 2 and 3) introduces the sectors studied —focusing on their specific features— and the environmental management tools used to carry out the assessment. Additionally, a brief review of previous LCA reports, specific for both sectors, is included in Chapter 3.

Section II (Chapters 4, 5, 6 and 7) is focused on the application of LCA to wine sector in two appellations of origin: "Ribeiro" and "Rías Baixas". Chapter 4 deals with the environmental profile of wine in "Ribeiro" appellation of origin analyzing the year-on-year variability, taking into account winegrowing and winemaking phases. Chapter 5 assesses viticulture operations in "Rías Baixas" appellation. To do so, LCA was combined with a management tool called Data Envelopment Analysis (DEA) to individually assess the environmental efficiency of 40 winegrowing exploitations and figure out which inventory items and/or operations are responsible of inefficiency. Furthermore, Chapter 6 analyzes different winegrowing methods in "Ribeiro" appellation. In this way, vineyards under different farming systems —conventional, bio-dynamic and a hybrid system: bio-dynamic and conventional— were evaluated year-on-year. Finally, Chapter 7 deepens in the estimation of carbon emissions in a whole appellation of origin —"Ribeiro". This chapter considers time-based LCA in combination with Land Use (LU) and Land Use Change (LUC) guidelines to report the carbon emissions/fixation derived from vineyard operations and LU during 1990-2009 period. Additionally, the main driving forces, socio-economic, which triggered LUC, are discussed through the chapter, as well as the technological improvements and management changes related to winegrowing over the years.

Section III[6] deals with the environmental assessment of seafood, comprising Chapters 8, 9, 10, 11 and 12. Chapter 8 introduces an environmental indicator which entails the energy consumption of fishing operations and energy provided —in terms of edible protein— by captures: ep-EROI (edible protein energy return on investment ratio). In this chapter, the Galician fishing fleet captures were analyzed throughout ep-EROI. The use of this indicator allows getting better understanding of energy efficiency in fishing sector. Furthermore, Chapters 9 and 10 assess the CF of a multi-ingredient product —fish sticks. Thus, on one hand, Chapter 9 evaluates the hake extraction (Patagonian grenadier) in Chilean waters, on-board processing, transoceanic freight, and final processing —addition of the remaining ingredients— at the processing plant in Galicia. On the other hand, Chapter 10 deals with fish sticks distribution, wholesaling, purchase and consumption —taking into account the different cooking options recommended by the

[6] Chapters 8 and 11 include inventory data analyzed and discussed by Vázquez-Rowe (2012).

producer, evaluating 24 different scenarios. Chapter 11 presents the evaluation of three different sources of marine protein based on sardine landings in Galicia: canned sardines, fresh sardines, and European hake caught using sardine as bait. The study evaluates the complete life cycle of the three products, considering all stages from fish extraction and processing to consumption at home —analyzing different cooking methods of each final product. Finally, Chapter 12 delves into the valorization of other marine resources such as seaweeds —a developing sector in coastal regions of many countries. Thus, this study evaluates the extraction of sodium alginate from seaweed (Kelp), taking into account harvesting and processing phases. Lastly, Chapter 13 presents the main conclusions derived from this dissertation.

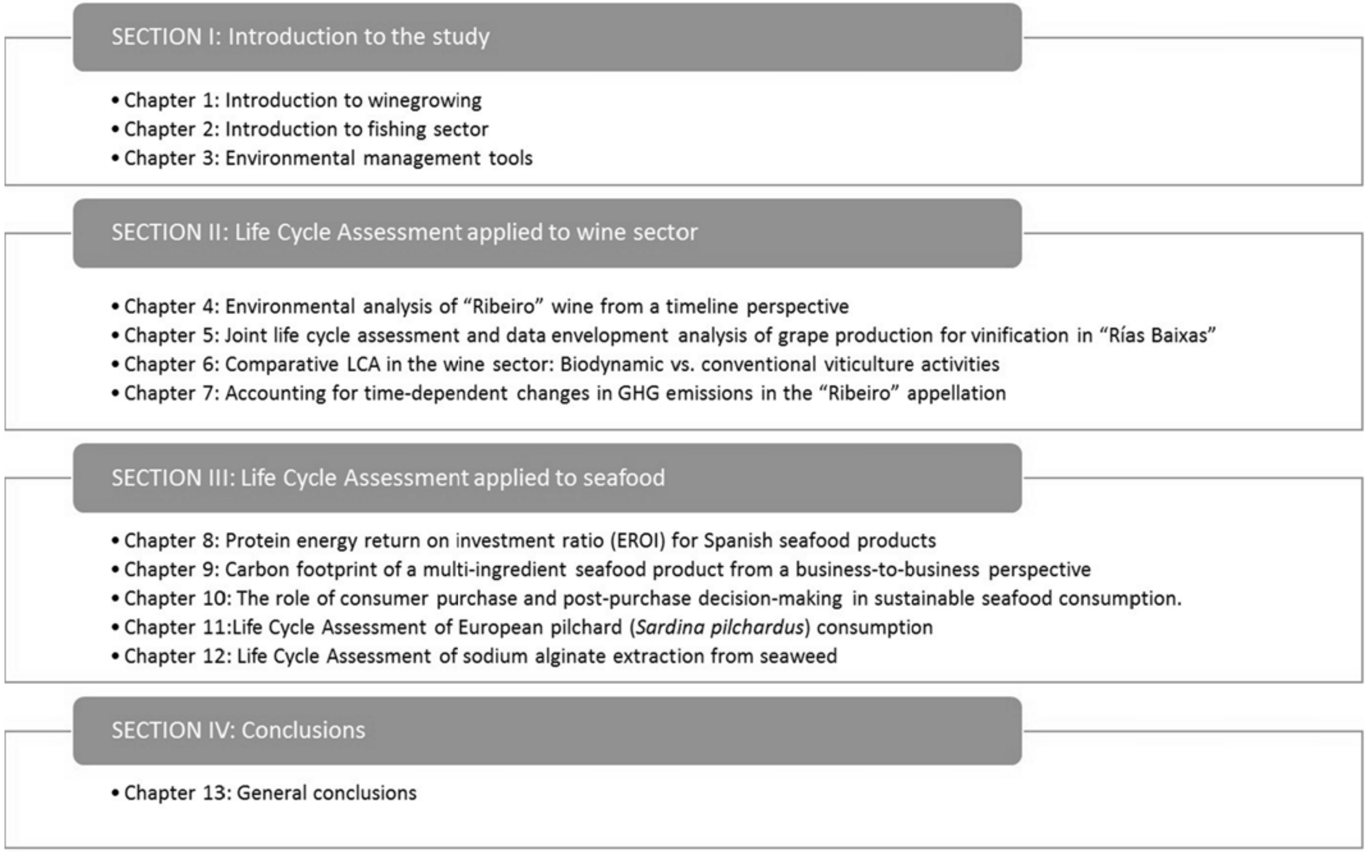

Figure 3.5. Schematic representation of the dissertation

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SECTION II

LIFE CYCLE ASSESSMENT APPLIED TO WINE SECTOR

Chapter 4

Environmental analysis of “Ribeiro” wine from a timeline perspective[1]

Summary

A series of Galician (NW Spain) wines, such as “Rías Baixas” and “Ribeiro” have acquired international renown in the last few years. In this particular study, viticulture, vinification and bottling and packaging in a winery of the “Ribeiro” appellation were studied from a life cycle assessment perspective, with the main objective of identifying the largest environmental impacts for four different years of production (2007-2010). The selected functional unit was a 750 mL bottle of “Ribeiro” white wine, packaged for distribution. Inventory data were gathered mainly through direct communication using questionnaires. Results showed considerable annual variability in environmental performance, stressing the importance of including timeline analysis in the wine sector. Therefore, environmental scaling was proposed for the assessed wine based on the individual environmental impacts for each harvest year. Furthermore, the main hotspots identified were the following: a) production and emissions of compost and pesticides in the agricultural phase and b) bottle production and electricity consumption in the subsequent stages of wine production. Suggested improvement opportunities included shifts in the compost transportation policy, recovery of natural resources for vineyard infrastructure, the introduction of new packaging formats in the bottling stage and the use of pesticides with lower toxicity potential.

[1] Vázquez-Rowe, I., Villanueva-Rey, P., Moreira, M.T., Feijoo, G., 2012. Environmental analysis of *Ribeiro* wine from a timeline perspective: Harvest year matters when reporting environmental impacts. J. Environ. Manag. 98, 73–83.

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4.1 Introduction

Wine production has historically been an important economic and social sector in Europe. In fact, three European countries (France, Italy and Spain) are the main world producers. Not surprisingly, despite a steady decrease since the 1980s due to the proliferation of the vinification industry in Asia and Oceania, Europe still accounted for 67.8% of the world's production in 2009 (OIV, 2010). Spain is currently the country with the highest surface area dedicated to wine production, approximately 2% of its territory. This translates into roughly 10,000 km^2 of surface area that produced around 35 million hL in 2009 (INE, 2011). Hence, Spanish wine production, as in most Mediterranean countries, is a key subsector within the agricultural sector. Most vine-growing regions in Spain are situated either along the Mediterranean coast or on the Castilla plateau. Nevertheless, Galicia, an Atlantic region in NW Spain which only accounts for 2% of the total land used for grape cultivation in the country, has acquired international renown in recent years thanks to the quality of some of its wines (Decanter, 2011), such as "Rías Baixas", "Riberia Sacra" or "Ribeiro" (Figure 4.1).

Figure 4.1. Location of the main appellations in Galicia (NW Spain)

Current quality standards for wine production are based on a series of indicators that comprise a wide range of dimensions, some of which are difficult to measure or report, making their definition complex (Charters and Pettigrew, 2007). Nonetheless, there have been attempts to construct and assess the different components of wine quality (Botonaki and Tsakiridou, 2004; Jover et al, 2004). For instance, Jover et al. (2004) divided red wine quality into 15 different dimensions, which affect both the expected and experienced quality of the wine, grouping them in two separate blocks: seven intrinsic factors, related to the physical characteristics of the wine, such as age, color, aroma or harvest; and eight extrinsic factors, linked to other characteristics, such as reputation, appellation, region or advertising and marketing.

Interestingly, to date, wine quality publications have not considered environmental aspects. However, the fact that consumers are increasingly aware of environmental issues, such as global warming, has led a growing number of stakeholders in the wine sector to analyze and communicate their environmental performance to gain market access and competitiveness (Garnett, 2008; Iribarren et al., 2010). Hence, environmental sustainability has developed into a priority in the wine supply chain (Forbes et al., 2009; Gabzdylova et al., 2009).

During the last decade, the bibliographic proliferation of wine LCA has definitely helped to picture the environmental profile of the wine sector, identifying the main hotspots throughout the process and suggesting the specific impact categories that are of special relevance. On the contrary, this phenomenon has also revealed the major limitations of applying LCA methodology to this agricultural sector. These limitations have been found to be mainly due to the lack of: i) specific data related to characterization factors for pesticides and fertilizers used in the viticulture stage; ii) inventory availability when assessing certain stages of viticulture, such as vine planting (Bosco et al., 2011) or delimiting the temporal boundaries of the study (Ramos et al. 2011); iii) an integrated and broad selection of impact categories, including those that are potentially suitable for wine LCA analysis (Petti et al., 2010); and iv) a unified criterion on how to include recovery and recycling processes (Jin and Kelly, 2009; Petti et al., 2010).

Hence, the main objective of this chapter is to perform the environmental assessment of a winery of one of Galicia's most renowned appellations ("Ribeiro") with the aim of: i) evaluating the main hot spots and improvement opportunities within the system; ii) approaching some of the above-mentioned limitations that previous wine LCA studies have identified, such as toxicity impact factors for pesticide emissions through the use of USEtox or temporal inventory availability to analyze variations in the harvest environmental performance; and iii) proposing an environmental scaling based on LCA results to identify differences in the environmental profile per harvest year.

4.2 Materials and Methods

4.2.1 Goal and Scope definition

As mentioned above, the objective of this LCA study is to assess the environmental burdens related to wine production in the "Ribeiro" appellation. Therefore, inventory data were collected during four different years of production (2007-2010), to identify potential differences in the environmental performance through several years.

The FU in an LCA study measures the function of the analyzed production system, providing a reference to which the inputs/outputs are linked to (ISO, 2006b). The selected FU was a 750 mL bottle of "Ribeiro" white wine[2], which is in accordance with previous wine industry LCA studies that consider a standard amount of wine; typically, the content of a regular bottle (Petti et al., 2010). Given that this study also comprises the agricultural stage of wine production, it is important to mention that the selected FU corresponds to 1.1 kg of harvested grape.

[2] The wine produced by the winery under study is a mixture of a total of seven grape varieties. Approximately 80% of the mixture corresponds to *Treixadura*, while the remaining 20% is a mixture of *Albariño*, *Albilla*, *Torrontés*, *Godello*, *Lado* and *Loureira*.

4.2.2 System boundaries

The production system studied embraced the different activities considered in the agricultural phase of wine production, including fertilization, field operations or soil management (Figure 4.2). Vineyard infrastructure and machinery for field operations were also included within the system boundaries. Therefore, the product was traced from the production of supply materials, such as diesel, pesticides or fertilizers, up to the delivery of the grapes at the gate of the winery after harvesting. Moreover, grape processing in the winery, including all the phases of its production, together with bottling and packaging were also included. Hence, the product was followed up to packaging ready for distribution to wholesalers, constituting a "cradle-to-gate" analysis (Guinée et al., 2001).

Excluded processes included the vine nursery stage (prior to vine arrival at the vineyards), as well as the post-bottling stages of the product. The rationale behind vine nursery exclusion was related mainly to the lack of reliable data regarding this phase of vine-growing. Nevertheless, given the longevity of most vines in the studied farm, and the small percentage of annual vine replacement, its exclusion should not significantly affect the final results (Bosco et al., 2011). Vine planting, however, as well as pre-production stage impacts are included within the grape production subsystem, since the replacement and growth of dead or damaged vines[3] is included within the daily tasks when analysing field operations, as reported by winegrowers.

Post-bottling stages were also kept outside system boundaries since most of the distribution was not handled directly by the winery. This implied that detailed data on these phases were not available. Hence, in order to maintain quality data in this case study, distribution, wholesaling, retailing and consumption were excluded. Moreover, the fact that one of the aims of this study

[3] The vine-growers of this particular winery reported replanting approximately 0.4% of the total on an annual basis.

is to propose an environmental scaling scheme for the winery, which also justifies the exclusion of the subsystems occurring beyond the gate of the winery.

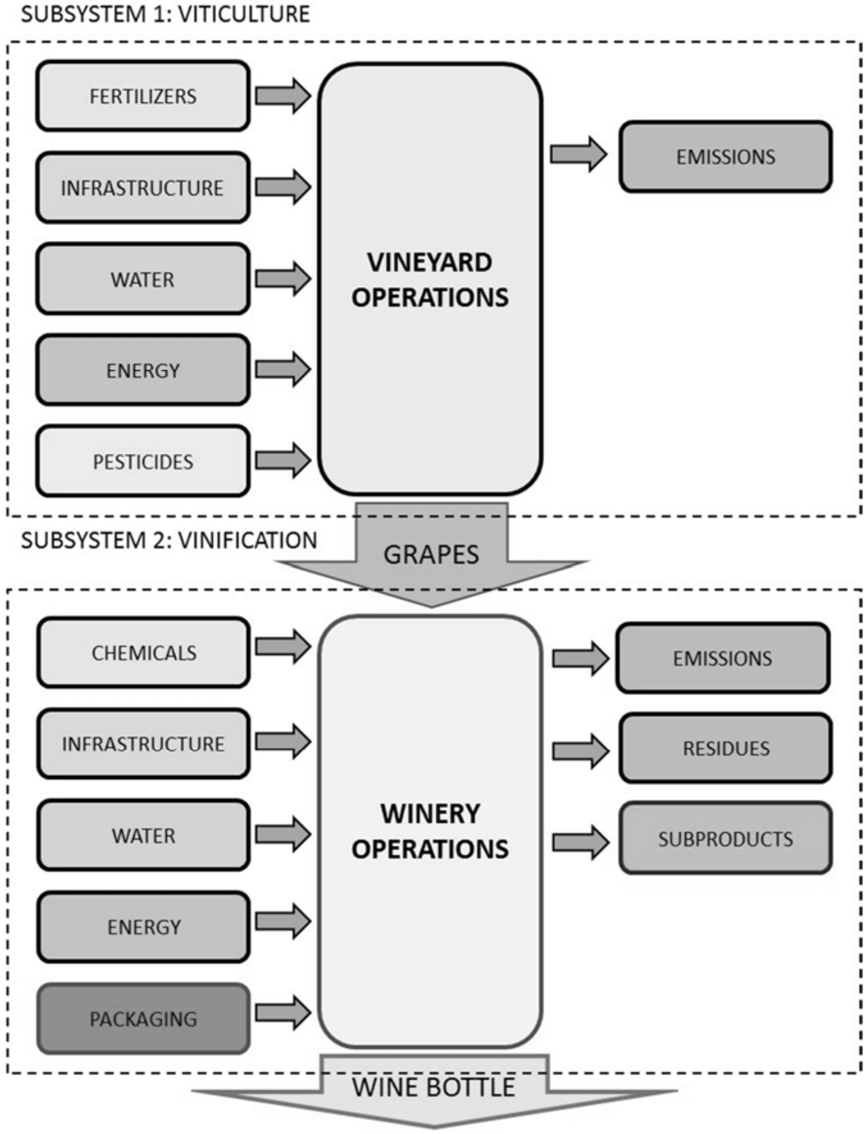

Figure 4.2. System boundaries for the production of a 750 mL bottle of "Ribeiro" wine

4.2.3 Data acquisition

Primary data were obtained through questionnaires answered by employees of a well-known winery in the municipality of *Leiro* (NW Spain), one of the main areas of the "Ribeiro" appellation (Table 4.1). Questionnaires embraced a thorough identification of operational aspects of the inventoried vineyards, such as fertilization, pesticide use, vineyard operations, machinery, water use and the infrastructure of the vineyard, when analysing the agricultural phase. The questionnaire for the vinification stage covered the quantification of all major inputs and outputs from the different processing stages, such as destalking, crushing, desludge, clarification and filtration. Finally, bottling and packaging data were provided by the winery for use also as primary data in this subsystem.

Table 4.1. Selected sample for the 2008-2010 period

	2007	**2008**	**2009**	**2010**
Surface area (ha)	14	14	14	14
Total grape production (tonnes)	112.5	98.0	96.0	120.0
Grape productivity (tonnes/ha)	8.04	7.00	6.86	8.57
Total of wine produced (hL/year)	787.5	686.0	672.0	980.0*
Wine yield	0.70	0.70	0.70	0.70
Commercialized bottles	105,000	91,470	89,600	130,670

* In the 2010 harvest, a total of 20 tonnes of grapes were bought from other producers to fulfil growing demand.

Direct emissions linked to fertilizer use in the agricultural step were included in the study by using two calculation methods. Nitrogen emissions were computed following the recommendations for nitrate, ammonia and nitrous oxide suggested by Brentrup et al. (2000). Phosphate emissions were taken from bibliographical data (Cowell, 1998; Cowell and Clift, 1997). Pesticide-related emissions were estimated by implementing PestLCI, a modular model that estimates pesticide emissions to the different environmental compartments (i.e. groundwater, surface water and air…) during field application. In order to compute pesticide emissions, a series of climatic and soil assumptions had to be introduced into the model (Birkved and Hauschild, 2006).

Climatic data (monthly precipitation, sunshine hours and mean temperature) were retrieved from the meteorological station of *Amiudal*, municipality of *Avión*, to calculate the annual water balance (Meteogalicia, 2011). This station was chosen since it is the only permanent meteorological station in the area. Evapotranspiration (ET) calculations to determine the annual water balance were based on the Thornthwaite method (Alkaeed et al., 2006). The rationale behind the use of this method is linked to the fact that the Penman-Monteith equation, standardized by the Food and Agriculture Organization (FAO56-PM), could not be computed for this particular meteorological station, since not all data were available to implement it (Allen et al., 1998). Hence, the Thornthwaite method was applied as a less precise method to calculate ET, but more feasible to apply in this study. Nonetheless, the fact that annual recorded data was introduced in the inventory, rather than using average climatic conditions, enhances the precision of the study. Information regarding the soil was obtained from Trueba et al. (1998). Additionally, a slightly acidic sandy clay loam soil (Rosa Calvo de Anta, personal communication) was assumed to meet data requirements for soil characteristics of *Ribeiro* County.

Regarding fuel use, emissions generated by agricultural machinery in field operations were estimated using the guidelines for air pollutant emissions provided by the EMEP-Corinair Emission Inventory Handbook 2006. Another energy source used in this study, electricity, was based on adapting the ecoinvent® electricity production mix (including imports) for Spain. Hence, based on data retrieved from the Spanish Government (REE, 2008 2009; IDAE, 2007), the Spanish electricity mix data for 2007, 2008 and 2009 were included in this research study[4].

[4] Electricity mix data for 2010 were not available when this study was conducted. Accordingly, mix data from 2009 were used.

Liquified sulphur dioxide use and emission factors linked to wine fermentation were gathered from USEPA (1995). Finally, background data for diesel production, fertilizers, pesticides, machinery, infrastructure, field operations (excluding diesel consumption), packaging and bottling materials were obtained from the ecoinvent database (Frischknecht et al., 2007).

4.2.4 Life Cycle Inventory

Life Cycle Inventory (LCI) collects data of the significant inputs and outputs of a production system, including resources use and emissions (ISO, 2006a, b). When dealing with primary sector processes, LCA practitioners may encounter added obstacles linked to data availability, due to the family-run characteristics of many farms and to small-scale food processing industries, such as small wineries or dairy farms (Hospido et al., 2003). The above-mentioned drawbacks constitute the main reasons for the limited bibliography available for primary sector product LCA studies in which more than one crop season is examined. Hence, detailed data were assembled for this particular study for a 4-year period, with the aim of detecting environmental performance variations throughout the assessed period.

LCA for the grape production phase

Inventory data for the agricultural phase are related to the grape production performed in the 14 ha of the winery during the assessed years. Table 4.2 shows the average allocated inventory data per FU, which include the main inputs and outputs of the analysed production system.

Table 4.2. Inventory for grape production for the 2007-2010 period (Data per FU: 1.1 kg of grapes at the winery gate to produce 1 bottle of wine-750 mL)

Inputs from the technosphere					
Materials and fuels	**Units**	**2007**	**2008**	**2009**	**2010**
Fertilizers					
Organic (compost)	kg	3.42	3.85	3.93	3.14
Energy					
Electricity	kw·h	$1.53 \cdot 10^{-2}$	$1.76 \cdot 10^{-2}$	$1.80 \cdot 10^{-2}$	$1.44 \cdot 10^{-2}$
Diesel	g	68.44	80.37	70.08	65.63
Pesticides					
Thiocarbamates	mg	61.19	70.24	57.39	57.37
Dithiocarbamates	mg	411.08	471.90	361.30	385.39
Acetamide-aniline	mg	9.86	11.31	11.55	9.24
Nitriles	mg	64.17	73.66	75.19	60.16
Cyclic-N compounds	mg	16.27	18.68	0.00	15.26
Phtalamide	mg	159.13	182.68	186.48	149.19
Triazine	mg	325.80	374.00	381.79	305.43
Glyphosate	mg	332.89	388.22	395.01	312.08
Fosetyl-Al	mg	691.44	791.78	649.14	648.23
Unspecified pesticides	g	2.68	3.08	2.57	2.52
Field operations*	m^2	2.26	2.57	2.63	2.1
Other inputs					
Trellis (steel)	g	124.11	142.48	145.44	116.36
Water (tap)	kg	1.29	1.48	1.27	1.21
Inputs from the environment					
Land use	m^2	1.37	1.57	1.60	1.28
Outputs to the technosphere					
Products	**Units**	**2007**	**2008**	**2009**	**2010**
Grapes	kg	1.10	1.10	1.10	1.10

Table 4.2. Inventory for grape production for the 2007-2010 period (Data per FU: 1.1 kg of grapes at the winery gate to produce 1 bottle of wine-750 mL) (continuation)

Outputs to the environment					
Emissions to the atmosphere					
CO_2 (diesel)	g	214.90	252.36	220.05	206.08
SO_2 (diesel)	g	684.40	803.70	700.80	656.30
VOC (diesel)	mg	497.56	584.29	509.48	409.30
NO_x (diesel)	g	3.44	4.04	3.53	3.30
CO (diesel)	g	1.10	1.29	1.12	1.05
NH_3 (diesel)	mg	0.48	0.56	0.49	0.46
NH_3 (fertilizers)	g	0.00	0.00	0.00	0.00
CH_4 (diesel)	mg	11.63	13.66	11.91	11.16
N_2O (diesel)	mg	88.29	103.68	90.40	84.66
N_2O (fertilizers)	mg	427.78	491.07	455.73	401.04
Emissions to water					
NO_3^-	g	21.84	26.54	24.82	19.86
PO_4^{3-}	mg	136.89	157.14	145.83	128.33

[*] Includes field sprayer, fertilizer broadcaster, rotary cultivator and rotary mower

Table 4.3 and Table 4.4 include the pesticide emissions per FU to air and to water, divided by species and chemical group. While the overall dose for years 2007, 2008 and 2010 was practically the same, cyclic-N compounds (triadimenol) were not applied to the vineyards in 2009 since lower amounts of anti-fungal treatments were required.

Table 4.3. Inventory data linked to atmospheric emissions due to pesticide use in grape production for the 2007-2010 period (Data per FU: 1.1 kg of grapes at the winery gate to produce 1 bottle of wine-750 mL)

	Units	2007	2008	2009	2010
Dithiocarbamate compounds					
Mancozeb	mg	324.58	372.61	285.28	304.30
Thiocarbamate compounds					
Cymoxanil	mg	9.73	11.17	11.40	9.12
Iprovalicarb	mg	40.56	295.31	33.91	36.20
Acetamide-aniline compounds					
Metalaxil	mg	7.78	8.93	9.12	7.30
Nitrile compounds					
Cyprodinil	mg	30.40	34.90	35.63	28.50
Fludioxinil	mg	20.27	23.26	23.75	19.00
Cyclic-N compounds					
Triadimenol	mg	12.85	14.75	--	12.05
Phtalamide compounds					
Folpet	mg	125.69	144.27	147.30	117.85
Organophosphorus compounds					
Fosetyl-Al	mg	542.64	622.93	512.56	508.73
Glyphosate	mg	264.48	303.61	309.94	247.95
Triazine compounds					
Terbuthylazine	mg	255.64	293.46	299.59	239.68

Table 4.4. Inventory data linked to water and groundwater emissions due to pesticide use in grape production for the 2007-2010 period (Data per FU: 1.1 kg of grapes at the winery gate to produce 1 bottle of wine-750 mL)

		2007		2008		2009		2010	
		W	GW	W	GW	W	GW	W	GW
Dithiocarbamate compounds									
Mancozeb	mg	T	0	$3.8 \cdot 10^{-2}$	$2.4 \cdot 10^{-6}$	0.96	$1.9 \cdot 10^{-3}$	1.42	$4.3 \cdot 10^{-3}$
Thiocarbamate compounds									
Cymoxanil	mg	0	0	T	T	T	T	T	T
Iprovalicarb	mg	T	T	$1.8 \cdot 10^{-2}$	$6.4 \cdot 10^{-7}$	5.6·10-^{2}	$8.0 \cdot 10^{-5}$	0.12	$2.8 \cdot 10^{-4}$
Acetamide-aniline compounds									
Metalaxil	mg	$2.0 \cdot 10^{-5}$	T	0.51	$1.1 \cdot 10^{-2}$	0.82	$3.0 \cdot 10^{-2}$	0.67	$2.6 \cdot 10^{-2}$
Nitrile compounds									
Cyprodinil	mg	$9.0 \cdot 10^{-5}$	T	0.46	$2.7 \cdot 10^{-3}$	1.53	$2.9 \cdot 10^{-2}$	1.37	$3.1 \cdot 10^{-2}$
Fludioxinil	mg	$1.4 \cdot 10^{-4}$	$1.3 \cdot 10^{-9}$	3.04	0.16	3.37	0.19	2.71	0.16
Cyclic-N compounds									
Triadimenol	mg	$5.2 \cdot 10^{-5}$	T	1.27	$4.5 \cdot 10^{-2}$	--	--	1.42	$6.9 \cdot 10^{-2}$
Phtalamide compounds									
Folpet	mg	0	0	$6.0 \cdot 10^{-7}$	T	$8.8 \cdot 10^{-4}$	$3.5 \cdot 10^{-8}$	$3.3 \cdot 10^{-3}$	$1.8 \cdot 10^{-7}$
Organophosphorus compounds									
Fosetyl-Al	mg	0	0	0	0	0	0	0	0
Glyphosate	mg	$8.9 \cdot 10^{-4}$	T	2.63	$1.1 \cdot 10^{-2}$	10.80	0.17	9.84	0.19
Triazine compounds									
Terbuthylazine	mg	$1.0 \cdot 10^{-3}$	T	24.91	0.85	34.05	1.57	27.79	1.33

W= emissions to water; GW= emissions to groundwater; T= trace.

LCI for wine processing phase

Annual inventory data respect per FU for the wine processing phase is shown in Table 4.5. Emissions of CO_2 associated to wine fermentation were calculated (Hidalgo-Togores, 2003), but were excluded from the environmental assessment, since biogenic CO_2 from viticulture was not taken into account. Identified residues, grape stalk and marc, were left out of the life cycle impact assessment since they are returned to the vineyards as fertilizers.

Table 4.5. Inventory table for the vinification subsystem in the "Ribeiro" appellation for the 2007-2010 period (Data per FU: 750 mL wine)

		Inputs from the technosphere			
Materials and fuels	**Units**	**2007**	**2008**	**2009**	**2010**
Energy use					
Electricity	kw·h	0.91	1.04	1.01	0.71
Propane	g	2.38	2.73	2.79	1.91
Materials					
Water (tap)	kg	8.40	9.28	9.16	7.65
Lubricant oil	mg	84.76	97.30	99.33	68.11
Bentonite	g	1.07	1.07	1.07	1.07
NaOH	mg	238.10	273.31	279.02	191.32
Ethylene glycol	g	2.05	2.35	2.40	1.65
Polypropylene	mg	685.71	787.17	803.57	551.02
Liquified SO_2	mg	37.50	37.50	37.50	37.50
		Inputs from the environment			
Land use	cm^2	190.48	218.65	223.21	153.06
		Outputs to the technosphere			
Products	**Units**	**2007**	**2008**	**2009**	**2010**
Wine	ml	750	750	750	750
By-products					
Grape stalk	cm^3	348	357	355	353
Grape marc	cm^3	350	350	350	350
Waste to treatment					
Wine lees	cm^3	12.35	11.23	11.86	11.48
Wastewater	dm^3	8.40	9.28	9.16	7.65

Table 4.5. Inventory table for the vinification subsystem in the "Ribeiro" appellation for the 2007-2010 period (Data per FU: 750 mL wine) (continuation)

Outputs to the environment					
	Units	**2007**	**2008**	**2009**	**2010**
Emissions to the atmosphere					
CO_2 (fossil)	g	23.81	27.33	27.90	19.13
CO_2 (fermentation)	g	76.75	76.75	76.75	76.75
CO	g	15.15	17.39	17.76	12.18
Ethanol-fermentation	g	0.17	0.17	0.17	0.17
Methanol-fermentation	mg	$5.78\cdot10^{-2}$	$5.78\cdot10^{-2}$	$5.78\cdot10^{-2}$	$5.78\cdot10^{-2}$
Acetaldehyde-ferm.	mg	$6.45\cdot10^{-3}$	$6.45\cdot10^{-3}$	$6.45\cdot10^{-3}$	$6.45\cdot10^{-3}$
H_2S	mg	0.12	0.12	0.12	0.12

LCI for the bottling and packaging phase

Inventory data for the bottling and packaging phase are shown in Table 4.6. Since this phase takes place within the winery's premises, it was impossible to obtain specific data regarding electricity use in the bottling and packaging subsystem. Therefore, the entire electricity consumption was assigned to the wine processing stage based on the assumption that only a small proportion of the total would be attributable to bottling and packaging (Bosco et al., 2011). The inventory includes separate data for white glass and green glass, since the production of the winery includes both bottles depending on the target market of the wine. Inventory data for the cork stopper were based on the data published by Rives et al. (2011).

Table 4.6. Inventory table for the bottling and packaging subsystem in the "Ribeiro" appellation for the 2007-2010 period (Data per FU: 750 mL wine)

Inputs[*] from the technosphere		
Materials	**Units**	
Glass (green bottle)[†]	g	570.99
Glass (white bottle)[‡]	g	570.99
Paper for labels	g	1.86
Cork	g	3.78
Aluminium cap	g	1.23
Packaging film (LDPE)	g	3.33
Corrugated board	g	36.65
EUR-pallet	p	$1.67 \cdot 10^{-3}$
Transport		
Cork transport to winery	tkm	$3.22 \cdot 10^{-3}$
Bottle transport to winery	tkm	0.40
Outputs to the technosphere		
Products	**Units**	
Bottle of wine (750 mL)	p	1
Residues		
Polyethylene	g	0.88
Paper	g	0.04
Corrugated board	g	0.85
Glass (mix)	g	2.85
Outputs to the environment		
Emissions to the atmosphere		
Ethanol	mg	9[**]

[*] No changes in the bottling and packaging system were observed throughout the analyzed period.
[†] The green bottle is the most commercialized bottle in this specific winery.
[‡] The white bottle is used exclusively for exports to the US, due to market regulations.
[**] Source: USEPA, AP-42, Section 9.12.2, 1995.

4.2.5 Allocation procedure

Allocation, an important issue in LCA studies, was not necessary for the grape production stage, since only one product was delivered to winery gate: grapes. In contrast, a series of residues are generated during the wine production stage. These products were incorporated into the vineyard as fertilizer, so no allocation was considered in this stage, since the only marketable product was wine. However, in 2010 a total of 20 tonnes of grape were used arriving from external vineyards and they were not included in the assessment. Given that all grapes underwent the same treatment, mass allocation was assumed in the winery stage. Finally, for the bottling and packaging phase no allocation was considered, given the fact that the white bottle and the green bottle are not competing products, filling the market coverage of different nations.

4.2.6 Impact category selection

LCIA was carried out using the CML baseline 2000 method (Guinée et al., 2001) for the following impact categories: abiotic depletion potential (ADP), acidification potential (AP), eutrophication potential (EP), global warming potential (GWP), ozone layer depletion potential (ODP), photochemical oxidant formation potential (POFP) and land competition (LC). Additionally, eco-toxicity (Etox) was evaluated according to the USEtox method (Rosenbaum et al. 2008)[5]. Impact category selection was based on a combination of commonly used categories in previous literature (Aranda et al., 2005; Gazulla et al., 2010), such as AP, EP and GWP, and a series of impact categories that will help broaden the scope of wine LCA studies, such as Etox or LC (Petti et al., 2010). Simapro 7 was the software used for the computational implementation of the inventories (PRè-Product Ecology Consultants, 2011).

[5] The USEtox + Interim method was the specific framework used in the study, since it covered all the pesticides used in the wine farm analyzed.

4.3 Results

4.3.1 Life Cycle Impact Assessment of Viticulture

Impact values per FU in the viticulture subsystem were the lowest in 2010, except for Etox, and the highest in 2008, except for EP, in 2009 (Table 4.7 and Table 4.8). For all other impact categories, values for 2009 were slightly lower than those reported for 2008 (ranging from 0.7% for GWP to 9.2% for ODP), while environmental impacts in 2007 lay between those of 2008/2009 and 2010. The greatest difference in environmental impact was identified for EP. The 2010 value was 21.7% lower than that for 2009. For the rest of the impact categories, the values for 2010 were approximately 18.5% lower than those for 2008.

Table 4.7. Total values per FU associated with wine production in the "Ribeiro" appellation per FU

Impact Category	Units	2007	2008	2009	2010
ADP	g Sb eq	13.65	14.50	13.87	12.01
AP	g SO_2 eq	20.78	21.63	20.80	17.87
EP	g PO_4^{3-} eq	6.54	7.23	7.16	5.76
GWP	g CO_2 eq	2,934	3,209	3,143	2,643
ODP	mg CFC 11 eq	0.28	0.32	0.29	0.26
POFP	mg C_2H_4	980	1,031	1,005	861
LC	cm^2a	4,040	4,093	4,078	3,961
Etox	CTUe	35.26	40.62	45.83	37.21

ADP = abiotic depletion potential; AP = acidification potential; EP = eutrophication potential; GWP = global warming potential; ODP = ozone layer depletion potential; POFP = photochemical oxidant formation potential; LC = land competition; Etox = eco-toxicity.

Regarding relative contributions of the different processes included in the viticulture subsystem, the environmental impacts were dominated in most impact categories by background processes linked to compost production and transport, especially for AP, GWP, LC and POFP. For these four impact categories, the total contribution of compost production and transport was above 50%. More specifically, GWP was identified as the impact category with the highest relative contribution of compost processes (67-69% depending on the assessed year). The impacts related to compost were also found to be the most significant for ADP (36-38%), a range similar to those linked to eutrophication (34-36%). Finally, environmental burdens associated with ODP summed up to 22-25%. Potential environmental impacts linked to fertilizer emission due to on-field application accounted for 47% (in 2010) to 51% (in 2009) of the total impact for EP, while contribution of these emissions to GWP was approximately 7%.

Another process that showed relevant impacts in most categories was the production process and consumption of diesel. The highest relative burdens for this process were observed for ADP (30-33%), while the lowest corresponded to EP (9-11%). The year with the lowest relative impacts for diesel processes was 2009 for all impact categories. Pesticide production involved low relative contributions, below 10%, for all impact categories, except for ODP (57% in 2009 and 59% in 2007). Infrastructure in the vineyards accounted for 27-29% of the total impact for POFP, 23-25% for ADP, 23% for LC and roughly 10% to AP. Other processes included in the analyzed subsystem, such as tap water use, field operations (excluding diesel consumption) or electricity only represented minor contributions in all impact categories. Finally, it is important to highlight the small differences observed regarding the relative contributions for the different processes from year to year (Tables A.1 to A.4 of Appendix).

Table 4.8. Total and relative impact values per FU linked to the three process subsystems included in this study

Impact category	Units	Grape production		Vinification stage		Bottling and packaging	
		Value	*% over total*	Value	*% over total*	Value	*% over total*
				Year 2007			
ADP	g Sb eq	5.05	*37.0*	3.79	*27.8*	4.81	*35.2*
AP	g SO_2 eq	11.92	*57.4*	3.93	*18.9*	4.93	*23.7*
EP	g PO_4^{3-} eq	4.82	*73.7*	0.85	*13.0*	0.87	*13.3*
GWP	g CO_2 eq	1,810.29	*61.7*	496.48	*16.9*	627.96	*21.4*
ODP	mg CFC 11 eq	$1.79{\cdot}10^{-1}$	*63.5*	$3.21{\cdot}10^{-5}$	*11.4*	$7.06{\cdot}10^{-2}$	*25.1*
POFP	mg 1,4DCB eq	518.18	*52.9*	256.62	*26.2*	205.36	*21.0*
LC	m^2a	383.00	*9.5*	129.60	*3.2*	3528.02	*87.3*
Etox	CTUe	35.26	*97.2*	0.35	*1.0*	0.63	*1.8*
				Year 2008			
ADP	g Sb eq	5.83	*40.2*	3.86	*26.6*	4.81	*33.2*
AP	g SO_2 eq	13.73	*63.5*	2.97	*13.7*	4.93	*22.8*
EP	g PO_4^{3-} eq	5.69	*78.7*	0.67	*9.3*	0.87	*12.0*
GWP	g CO_2 eq	2,084.19	*64.9*	497.22	*15.5*	627.96	*19.6*
ODP	mg CFC 11 eq	$2.06{\cdot}10^{-1}$	*65.1*	$4.01{\cdot}10^{-2}$	*12.7*	$7.06{\cdot}10^{-2}$	*22.3*
POFP	mg 1,4DCB eq	596.12	*57.8*	229.64	*22.3*	205.36	*19.9*
LC	m^2a	439.45	*10.7*	125.15	*3.1*	3,528.02	*86.2*
Etox	CTUe	39.71	*97.7*	0.29	*0.7*	0.63	*1.5*
				Year 2009			
ADP	g Sb eq	5.60	*40.4*	3.46	*25.0*	4.81	*34.7*
AP	g SO_2 eq	13.41	*62.0*	2.46	*11.8*	4.93	*23.7*
EP	g PO_4^{3-} eq	5.71	*79.0*	0.58	*8.1*	0.87	*12.2*
GWP	g CO_2 eq	2,070.71	*65.9*	444.89	*14.2*	627.96	*20.0*
ODP	mg CFC 11 eq	$1.87{\cdot}10^{-1}$	*63.3*	$3.78{\cdot}10^{-2}$	*12.8*	$7.06{\cdot}10^{-2}$	*23.9*
POFP	mg 1,4DCB eq	588.40	*58.6*	211.19	*21.0*	205.36	*20.4*
LC	m^2a	445.28	*10.9*	104.82	*2.6*	3,528.02	*86.5*
Etox	CTUe	44.94	*98.1*	0.26	*0.6*	0.63	*1.4*

ADP= abiotic depletion potential; AP= acidification potential; EP= eutrophication potential; GWP= global warming potential; ODP= ozone layer depletion potential; POFP= photochemical oxidant formation potential; LC= land competition; Etox= eco-toxicity.

Table 4.8. Total and relative impact values per FU linked to the three process subsystems included in this study (continuation)

Impact Category	Units	Grape production		Vinification		Bottling	
		Value	*% over total*	*Value*	*% over total*	*Value*	*% over total*
				Year 2010			
ADP	g Sb eq	4.76	*39.6*	2.44	*20.3*	4.81	*40.1*
AP	g SO_2 eq	11.21	*62.7*	1.73	*9.7*	4.93	*27.6*
EP	g PO_4^{3-} eq	4.47	*77.6*	0.42	*7.3*	0.87	*15.1*
GWP	g CO_2 eq	1,701.60	*64.4*	313.61	*11.9*	627.96	*23.8*
ODP	mg CFC 11 eq	$1.68 \cdot 10^{-1}$	*63.3*	$2.67 \cdot 10^{-2}$	*10.1*	$7.06 \cdot 10^{-2}$	*26.6*
POFP	mg 1,4DCB eq	486.68	*56.6*	168.57	*19.6*	205.36	*23.9*
LC	m^2a	358.68	*9.1*	74.42	*1.9*	3,528.02	*89.1*
Etox	CTUe	36.40	*97.8*	0.19	*0.5*	0.63	*1.7*

ADP= abiotic depletion potential; AP= acidification potential; EP= eutrophication potential; GWP= global warming potential; ODP= ozone layer depletion potential; POFP= photochemical oxidant formation potential; LC= land competition; Etox= eco-toxicity.

4.3.2 Life Cycle Impact Assessment of Vinification

The highest impact values for all impact categories were obtained in 2008 (Table 4.7). Environmental impacts for this year were 42% higher than for 2010 for AP. The lowest difference was observed for POFP (27%). Finally, the results for 2009 and 2007 showed intermediate values with respect to the 2010 and 2008 impacts.

The environmental performance of the vinification stage presented an overwhelming dominance of the consumed electricity when compared to the rest of inputs and outputs considered in the subsystem. Electricity inputs represented at least 93% of the total impact for AP, ADP, GWP, LC and ODP in all the assessed years. Environmental performance regarding EP was also dominated by the burdens attributable to electricity consumption. However, a decreasing trend was detected during the studied period, with its contribution declining from 85% in 2007 to 73% in

2010. In contrast, wastewater treatment, the other process with a significant impact in terms of EP, experienced an increasing relative impact, from 13% in 2007 to 24% in 2010. Concerning POFP environmental impacts, emissions identified in the winery, such as acetaldehyde, CO, ethanol or methanol, involved the greatest relative impact for the 2008-2010 period (60% of total contribution in 2010), while electricity was identified as the major contributor in 2007. Finally, other inputs such as ethylene glycol and propane production or the use of water presented minor contributions, never above 4%. Tables A.1 to A.4 of the Appendix show more detailed results of these inventory items.

4.3.3 Life Cycle Impact Assessment of Bottling and Packaging

Environmental impacts were mainly associated with the production of the glass bottle. In fact, regardless of the type of bottle used in the process, environmental burdens linked to their production ranged from roughly 42% for LC to 87% for AP. Additionally, bottle transport to the winery accounted for an impact range between 4.2% (POFP) and 12.1% (ODP). Packaging processes prior to storage and distribution showed important contributions for EP (10%), ADP (10%) and GWP (9%).

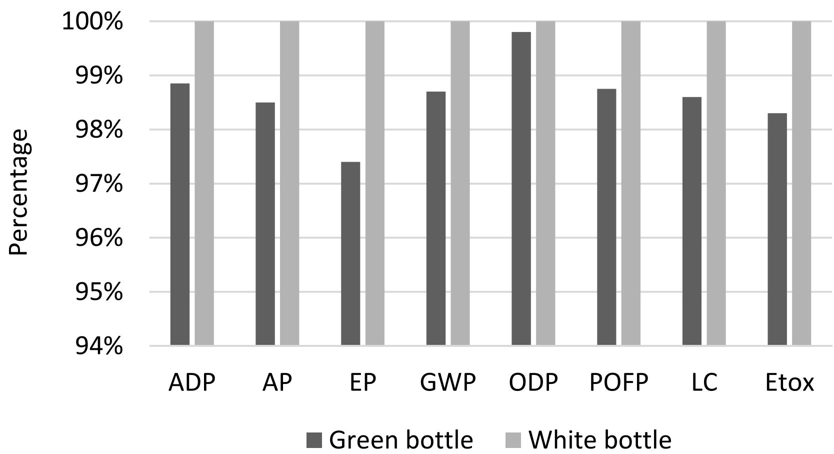

Figure 4.3. Environmental results for the selected impact categories for the bottling and packaging subsystem (%)

Finally, other minor environmental impacts were attributable to pallet production (39% for LC), cork production (7% for POFP) and bottle accessories (4% for POFP). Differences in the impact values depending on the type of bottle used were found to be very low (Figure 4.3). In fact, the highest variability in relative environmental contribution: 3%, was obtained for EP.

4.3.4 Impact values for aquatic eco-toxicity

Large differences between years were identified for the Etox impact category. The lowest overall characterization values per FU were identified for 2007 (35.3 comparative toxic units-CTUe), while those for 2009 were 30% higher (45.83 CTUe), as observed in Figure 4.4. Over 97% of the total impact for the assessed years was attributable to the viticulture subsystem. In fact, two specific pesticides: folpet and terbuthylazine, were responsible for at least 94% of the total environmental impact in each of the studied harvest years.

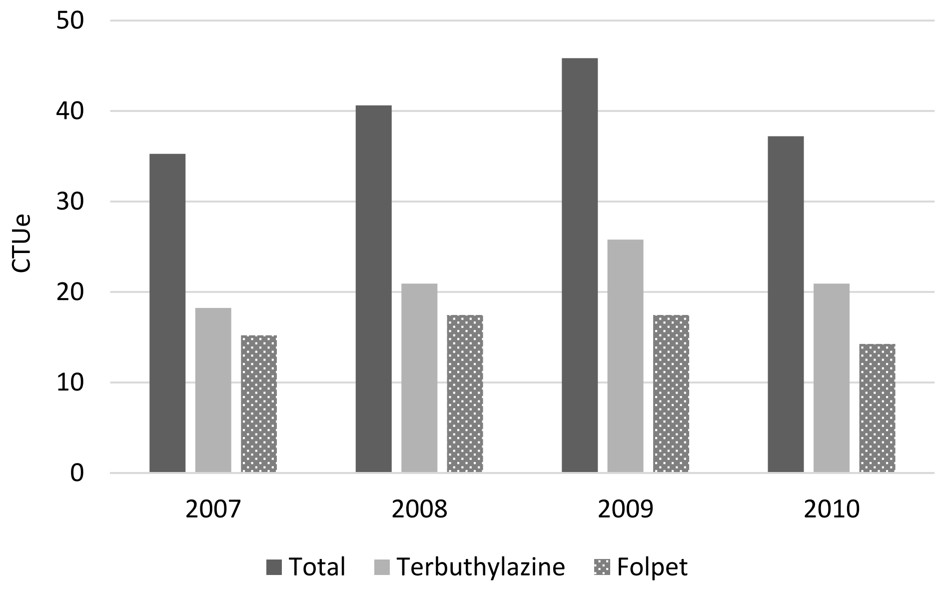

Figure 4.4. Total impact values for aquatic eco-toxicity (Etox) per FU. Impact values per FU for pesticides with significant environmental impacts regarding Etox

Moreover, the changing contributions for these three pesticides during the assessed period are remarkable. In the first place, terbuthylazine was found to contribute 52% of the total impact in 2007, while its contribution in 2009 and 2010 was 56%; and secondly, folpet contribution decreased from 43% in 2007 to 38% in 2010.

4.4 Discussion

4.4.1 Identification of hot spots and comparison with other published literature

Despite the significant variance detected on an annual basis, which is discussed further on, the viticulture subsystem was the main contributor to all impact categories, except LC, where bottling and packaging processes contributed by at least 86% for the entire period. Results for ADP, for instance, show a higher percentage of impact due to the viticulture phase (roughly 40%) for the 2007-2009 period, while in 2010 the greatest relative contribution was due to bottling and packaging processes. Considering the individual breakdown of the different subsystems, the main carriers of the environmental impact for ADP were bottle production, electricity production for the winery stage, production and transport of compost and, finally, diesel production and use.

Approximately two thirds of AP burdens were linked to the viticulture stage, where compost and diesel related processes were the main contributors. Electricity production for the winery phase and bottle production also presented important contributions.

EP impacts were attributable to emissions linked to fertilizing in the viticulture stage, followed by compost processes, and to a lesser extent, electricity production in the winery. Nevertheless, while fertilizer emissions did not suffer great variations on an annual basis, the reduction in the EP impact value in the 2007-2010 period was influenced by the strong changes in the Spanish electricity mix, due to the large decrease in hard coal and lignite use for energy production.

The main sources of GHG emissions were concentrated in the viticulture stage. For all the assessed years, this subsystem represented approximately 66% of total GWP impacts. More specifically, the major contributors to global warming were: compost production and transport, diesel related processes in field operations, bottle production and electricity production for the wine production stage. In a similar way as for EP, GHG emissions for 2010 benefited not only from higher grape —and wine— production rates due to favorable harvests, but also from the improvement of the Spanish electricity mix, as mentioned above.

ODP impacts were linked to the production of pesticides in the agriculture subsystem. *Circa* 33% of total ozone layer depletion impacts were linked to this sole process, while bottling and packaging and the processes associated with compost production and transportation also had significant impacts within this impact category.

POFP-related burdens were dominated by compost and infrastructure processes. The fact that the vineyard infrastructure is made up of steel is the main reason for the increased emissions in this specific impact category when compared to other studies. For instance, vineyards analyzed in Nova Scotia by Point (2009) had a negative POFP result, due to the use of wooden materials in the trellis instead of steel. Finally, it is also interesting to point out that emissions in the winery, linked to propane combustion and to the emission of acetaldehyde, ethanol and methanol during wine fermentation, showed high relative contributions in this impact category.

When the total characterization values of the studied system are compared to other wine sector LCA studies available in literature (Table 4.9), the white wine from the “Ribeiro” appellation appears at the top of the range for most of the assessed impact categories. However, given that a series of methodological assumptions may vary between studies, these comparisons must be done with caution. These studies also highlighted the viticulture and bottle and packaging subsystems as the main hot spots of the pre-distribution stages of the wine supply chain (Bosco et al., 2011; Gazulla et al., 2010; Point, 2009).

Table 4.9. Environmental comparison between wine LCA studies: impact results per FU (750 mL bottle up to the wineries gate)

	ADP	AP	EP	GWP	ODP	POFP
Current study (2008 harvest)	14.50	21.63	7.23	3,209	317	1,031
Current study (2010 harvest)	12.01	17.87	5.76	2,643	265	861
Current study (2010 harvest)*	11.39	11.85	4.35	1,645	251	642
Bosco et al. (2011)	--	--	--	798*	--	--
Gazulla et al. (2010)	--	5.90	30.90	934	--	260
Benedetto (2010)	10.30	11.90	1.50	1,640	--	--
Point (2009)	15.69	35.72	7.07	2,530	47	714

ADP = abiotic depletion potential; Units: g Sb eq., AP = acidification potential; Units: g SO_2 eq., EP = eutrophication potential; Units: g PO_4^{3-}, GWP = global warming potential; Units: g CO_2 eq., ODP = ozone layer depletion potential; Units: mg CFC eq., POFP = photochemical oxidant formation potential; Units: g C_2H_4 eq.

* Excludes compost production assuming these impacts are attributable to waste treatment phase of previous processes

Interestingly, the other Atlantic wine case study available in literature (Point, 2009) showed similar value ranges for ADP, EP, GWP and POFP when compared to the current case study. Wine from Nova Scotia, however, had higher AP impacts than "Ribeiro" wine due mainly to the bottle-production stage. Finally, ODP burdens were considerably higher for "Ribeiro" wine due to the high impacts detected during pesticide production.

4.4.2 Improvement actions identified

The main improvement actions that could be implemented are briefly described in Table 4.10. In the first place, despite the significant environmental burdens linked to compost production in most impact categories, the substitution of this product by mineral fertilizers was not considered. The rationale behind this decision lies in the fact that compost used for fertilization in the assessed vineyards is produced based on the use of organic —fraction municipal solid wastes (OFMSW), rather than sending OFMSW to landfill. This is in accordance with European policies that have regulated the gradual elimination of dumping OFMSW, while promoting their use for land fertilization (Directive 1999/31/CE). Hence, even though compost production has lower impacts than sending OFMSW to landfill, as identified by Martínez-Blanco et al. (2009), system expansion to include the avoided emissions from the landfill was not considered in the present study, since this would imply going back to waste disposal measures that are not recommended by current waste policy legislation. Following the perspective taken in this chapter, the entire environmental impacts linked to compost production were considered. However, another approach would assume that environmental burdens linked to compost production should be assigned to the impacts of the processes that generated the OFMSW as a residue. Hence, this latter approach would improve the environmental profile of the analysed bottle of wine considerably (Table 4.10).

Table 4.10. Environmental results per FU for the considered improvement scenarios. Note that results are based on potential improvements for the 2010 harvest year only

	Scenario 1		**Scenario 2**		**Scenario 3**		**Combined Scenario**	
	Value	**Red. (%)**	**Value**	**Red. (%)**	**Value**	**Red. (%)**	**Value**	**Red. (%)**
ADP	11.05	8	10.91	9.2	10.23	14.8	8.21	31.6
AP	17.15	4	16.74	6.3	15.67	12.3	13.83	22.6
EP	5.57	3.3	5.67	1.6	5.23	9.2	4.95	14.1
GWP	2,512	45	2,521	4.6	2,268	14.2	2,015	23.8
ODP	0.24	9.4	0.27	-1.9	0.26	1.9	0.24	9.4
POFP	839	2.5	725	15.8	780	9.4	623	27.6

Scenario 1 = Compost production plant within the "Ribeiro" region; Scenario 2 = Use of wood trellis for vine support; Scenario 3 = Substitution of bottle package for PET bottle.
ADP = abiotic depletion potential; Units: g Sb eq., AP = acidification potential; Units: g SO_2 eq., EP = eutrophication potential; Units: g PO_4^{3-}, GWP = global warming potential; Units: g CO_2 eq., ODP = ozone layer depletion potential; Units: mg CFC eq., POFP = photochemical oxidant formation potential; Units: g C_2H_4 eq.

A series of improvement actions related to compost production and transport processes were identified. Firstly, transport of compost to the winery could be minimized through the creation of a locally based compost plant to supply this product to the entire "Ribeiro" region, a strong agricultural area within NW Spain (Gellings and Parmenter, 2004). Additionally, the inclusion of measures to reduce the energy intensity of compost production would benefit the environmental profile of the analysed process. This issue is important given that other waste management alternatives, such as anaerobic digestion, use less energy (Hermann et al., 2011).

Regarding vineyard infrastructure, the use of wood trellis rather than steel to support the plants would help retain carbon in the vineyard. In fact, traditional practices in the "Ribeiro" appellation include the use of *Acacia dealbata* as training system. However, the ongoing rural flight in NW Spain, the abandonment of traditional practices and the set of technological innovations introduced for field infrastructure have enhanced the colonizing characteristics of this invasive species (Carballeira and Reigosa, 1999; Lorenzo et al., 2010). Hence, return to this field practice would improve the environmental profile of this "Ribeiro" wine, as well as contribute to recover harvest practices for *Acacia dealbata* and, therefore, minimize the displacement of other species.

Another major process that can potentially improve its environmental performance is bottle production. Bottle weight reduction, as proposed by Point (2009) or changes in its design may be interesting options. Moreover, the use of a plastic (PET) bottle, substituting the conventional glass bottle would reduce environmental impacts up to 15% (ADP and GWP). However, distributing wine in plastic bottles is not always well received by consumers. Moreover, wine quality and its shelf-life may also be reduced (Ghidossi et al., 2012), contributing to extended environmental impacts in the distribution phase due to increased non-consumed wine.

4.4.3 USEtox computation. Eco-toxicity impact values based on pesticide use

CML baseline 2000 impact categories assessed in this case study show similar trends regarding the annual impacts per FU, due to the difference in production levels between years and to the reduction of the dependence on fossil fuels in the Spanish electric grid. However, aquatic eco-toxicity measured with USEtox shows a completely different temporal pattern. For instance, the harvest year with highest impacts per FU was 2009. The reason behind this increased burden in

2009 and 2008 is mainly attributable to climatic conditions for these years, in which the "Ribeiro" region suffered from high precipitation rates. In contrast, the environmental impacts for 2007 were considerably lower due to the dry weather conditions identified in that campaign.

Folpet and terbuthylazine were the main compounds responsible for the Etox impacts (Figure 4.4). Furthermore, copper emissions —very common compound to treat fungal diseases— and potential impacts were not evaluated due to the lack of reliable data regarding emissions fate and characterization factors. Nonetheless, some literature on the subject highlights that high concentrations of copper in vineyard soils throughout Europe have been detected, including the "Ribeiro" appellation, in many cases exceeding legal limits (Arias et al., 2004; Díaz-Raviña et al., 2007; Fernández-Calviño et al., 2008; Komárek et al., 2010). Also, it should be highlighted that Folpet may be substituted by captan, another phtalamide which could potentially reduce eco-toxicity impacts by up to 13 times per ha, provided that the effects of this compound do not imply a threat to human health (Gordon, 2010; USEPA, 2004).

4.4.4 Impact results identified on a temporal basis. Timeline analysis

Environmental burdens linked to the production process of a 750 mL bottle of "Ribeiro" white wine showed similar trends to other wine appellations analyzed in previous LCA studies (Benedetto, 2010; Point, 2009). However, the fact that this case study covers an extended period of time aims at determining if there can be considerable variance in terms of environmental performance between years (Figure 4.5).

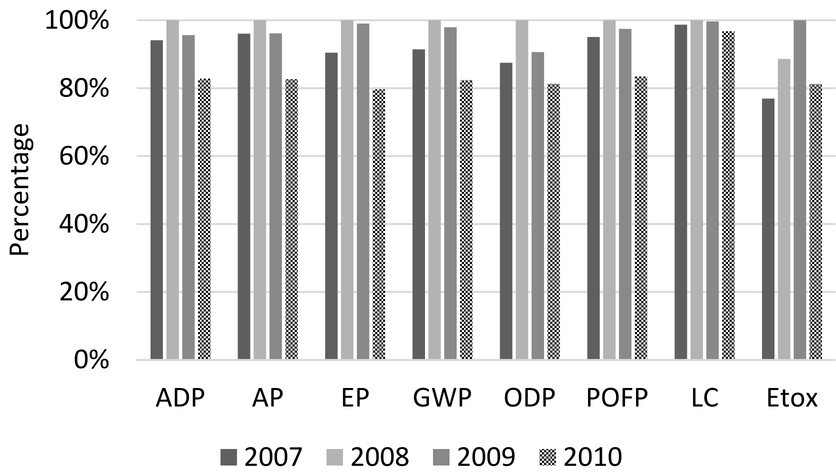

Figure 4.5. Environmental comparison between the assessed years for the selected impact categories (%)

Results for this winery suggest that reporting the environmental profile for a given appellation or winery should be done on the basis of the particular harvest year under study. In fact, reporting environmental burdens on the basis of a single harvest, without identifying the limitations of the study from a timeframe perspective, would shade potential annual fluctuations. This circumstance may interfere with correct environmental management in the wine industry, as well as give out a misleading message to stakeholders in the sector, and especially, to target consumers who in many cases base their wine bottle selection depending not only on the appellation and winery themselves, but also on the particular vintage of the harvest year (Fischer et al., 1999).

A recent study related to timeline LCA analysis of Atlantic mackerel (*Scomber scombrus*) in the Basque country suggested the use of moving averages to report annual environmental impact variations, to avoid multiple result interpretation problems (Ramos et al., 2011). While this approach may be used for reporting wine environmental performance in cases such as diet or meal case studies or beverage reviews, its extended use for providing environmental information about wine to consumers is not recommended, due to the differentiated marketable characteristics of most wines based on their harvest year.

4.4.5 Discrimination between harvest years based on environmental performance

A race by stakeholders to report the environmental performance of their products, to satisfy pressures from NGOs and growing concern from consumers and public opinion has occurred in recent years (Garnett, 2008; Iribarren et al., 2010). This new behaviour has created a scenario in which the perceived environmental standard of a product by the consumer has become an increasingly important factor when considering its overall quality. Therefore, many environmental labels have arisen to identify the best performing products from an environmental point of view within a similar group of products (Andersson, 1998; Iribarren et al., 2010).

The wine industry, which is usually concerned with the quality standards of the product, has traditionally reported wine quality for single harvest years in a given appellation based on the vintage, which intends to measure the quality of the evaluated wine. While this protocol is plainly accepted and used within the wine sector, the increasing environmental demands wineries are facing, lead us to propose a complementary scale to accompany the vintage. This environmental vintage is based on the GWP impact category results (i.e. GHG emissions) per bottle reported for each harvest year (Table 4.11).

Table 4.11. Vintage and environmental vintage for the assessed winery during the 2007-2010 period

Harvest year	2007	2008	2009	2010
Vintage*	Very Good	Excellent	Very Good	Very Good
GWP/FU (gCO_2 eq.)	2,934	3,209	3,143	2,643
Environmental	Very Good	Good	Good	Excellent

* Vintage classification according to the Spanish Wine Fair 2011, available from: http://www.anadas-do.com/50/ribeiro/

The proposed grading system is based on the environmental impact variations that can be identified between years. Therefore, the studied winery was divided into three environmental performance levels, based on the GHG emissions per FU: i) Excellent, for harvest years with relatively low GWP impact (< 2.750 kg CO_2 eq./bottle); ii) Very good, for those years with intermediate GWP results (2.750-3.000 kg CO_2 eq./bottle); and iii) Good for harvests whose GHG emissions are above 3.000 kg CO_2 eq./bottle (Table 4.11).

Even though global warming may provide a myopic view of the environmental profile of a product, the use of this single indicator has proved to be an effective method to reach consumers, since: i) there is worldwide public concern about reducing GHG emissions; ii) an expected outburst when reporting the global warming potential for food and beverage products, since they are significant contributors to global emissions (Garnett, 2008, 2009); and iii) the fact that wine has shown to be one of the beverages with the highest GWP (Hanssen et al., 2007; Point, 2009; Talve, 2001)

4.5 Conclusions and perspectives

The analyzed case study assesses the environmental impacts related to the viticulture, vinification and bottling and packaging stages of winemaking in a "Ribeiro" winery. Evaluated conventional impact categories showed that environmental impacts per bottle of wine varied considerably from year to year. These results demonstrate the high dependency on annual crop productivity, as well as proving the need to assess the environmental impacts of wine production in terms of the harvest year, in order to avoid misleading messages throughout the supply chain due to the use of mean values or harvest years that do not correspond to the marketed product. Improvement actions to reduce the environmental burdens associated with winemaking in this winery are proposed. However, it is important to highlight that most of them are highly dependent on a joint policy action of the entire appellation (i.e. county-level composting) or on a change in the packaging configuration of the wine.

The ecosystem toxicity relating to the agricultural phase showed a high dependence on the varying meteorological conditions. A series of improvements are suggested to reduce the impact of pesticide use on the ecosystem. Nevertheless, these would be subject to further research regarding human health hazards these shifts may generate.

4.6 References

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Chapter 5

Joint life cycle assessment and data envelopment analysis of grape production for vinification in “Rías Baixas”[1]

Summary

An important percentage of European wine appellations base their production on a broad number of wine-growers that annually sell their grapes to the wineries under the specific Denomination of Origin. Hence, the use of average values for the environmental evaluation of this type of multiple datasets can create large standard deviations that may impede an adequate interpretation of the results. Combined implementation of Life Cycle Assessment (LCA) and Data Envelopment Analysis (DEA), known as LCA + DEA methodology, has proven to be a suitable tool for assessing multiple input/output data in several agro-food systems, such as aquaculture, farming or fisheries.

In the current study, a total of 40 vine-growing exploitations belonging to the “Rías Baixas” appellation (NW Spain) were analyzed following LCA + DEA methodology in order to determine the level of operational efficiency of each producer. Furthermore, potential reductions in the consumption levels of the material inputs were benchmarked, while calculating the environmental gains linked to these reduction targets, thus verifying eco-efficiency criteria. Results led to average reduction levels of up to 30% per material input, which translated into environmental gains that ranged from 28% to 39% depending on the selected impact category. Additionally, a super-efficiency analysis led to identify the best performing units, which were used as a source of reference values for environmental impacts. Finally, potential economic savings of 0.14 € per functional unit (i.e., 1.1 kg of grapes for the production of a common 750 mL bottle of wine) were estimated on the basis of efficient vine-growing practices.

[1] Vázquez-Rowe, I., Villanueva-Rey, P., Iribarren, D., Teresa Moreira, M., Feijoo, G., 2012. Joint life cycle assessment and data envelopment analysis of grape production for vinification in the Rías Baixas appellation (NW Spain). J. Clean. Prod. 27, 92–102.

Index

5.1 Introduction

Southern European countries, such as Italy, France or Spain have traditionally been the main producers of grapes for vinification at a worldwide scale. Nevertheless, in recent decades there has been a continuous trend aiming at improving the quality, while reducing the global productions in these countries (OIV, 2010). In fact, surface area of vineyards has decreased in Europe at a much faster rate than in the rest of the world for the last 25 years, while appellations (also named Protected Designation of Origin, PDO) have nearly tripled since 1980 (FAO, 2011). These trends have occurred under the current globalization of agro-food products worldwide, generating a gradual modification of the wine production process. In this respect, pressure from governments and consumers has increased within the European Union (EU) in order to improve the environmental profile of wine production (Renaud et al., 2010). Environmental sustainability in the wine sector arises therefore as a major goal to be reached by the different actors involved in the wine supply chain (Forbes et al., 2009; Gabzdylova et al., 2009).

In this context, Life Cycle Assessment (LCA) has been applied to analyze the wine sector in a variety of countries, such as Spain (Aranda et al., 2005; Gazulla et al., 2010; Vázquez-Rowe et al., 2012), Italy (Benedetto, 2010; Bosco et al., 2011; Notarnicola et al., 2003; Pizzigallo et al., 2008) or Canada (Point et al., 2012). LCA studies in this field have focused not only on the final product (i.e., wine), but also on process wastes (Ruggieri et al., 2009) and even on specific wine-related components such as cork stoppers (Rives et al., 2011).

Published LCA studies on wine production have highlighted the importance of obtaining significant on-site data for the processes included in the system (Petti et al., 2010). However, a great number of appellations in Europe base their grape production phase on a broad number of vine-growers that annually sell their grapes to the wineries under the specific PDO. This situation makes environmental evaluations on viticulture (i.e., vine-growing) complicated since multiple data for multiple facilities have to be handled. The use of average values for this type of multiple datasets usually entails large standard deviations that may impede an adequate interpretation of the results (Reap et al., 2008a; Weidema and Wesnaes, 1996). In other words, the use of average inventory data when analyzing a multiple set of vine-growing plantations is likely to be subject to important data variability, distorting the individual performance of each of the assessed vineyards.

In order to face the assessment of multiple input/output data for a large number of similar entities, the joint application of LCA and Data Envelopment Analysis (DEA), known as LCA + DEA methodology, has been proposed as a valuable tool that avoids standard deviation concerns, while providing a detailed operational and environmental analysis of the sample (Iribarren, 2010). While LCA addresses the evaluation of the environmental aspects and potential impacts associated with a product (ISO, 2006a, b), DEA is a linear programming methodology to quantify the comparative productive efficiency of multiple similar units (Cooper et al., 2007; Sarkis and Weinrach, 2001). LCA + DEA methodology has already proven to be a suitable tool for assessing multiple datasets in aquaculture (Iribarren, 2010; Lozano et al., 2009; Lozano et al., 2010), fisheries (Vázquez-Rowe et al., 2010; Vázquez-Rowe et al., 2011) and dairy farms (Iribarren et al., 2011).

The goal of the current study was to apply LCA + DEA methodology over a relevant number of Galician vineyards belonging to the “Rías Baixas” appellation, in the Salnés subzone (NW Spain; capital: Vilagarcía de Arousa, 42°35′N 8°46′W). The “Rías Baixas” wine sector contributed to 0.2% of the Galician GDP in 2010, accounting for circa 100 million € (Xunta de Galicia, 2012). The analysis was conducted in order to: i) detect operationally inefficient grape cultivation plots; ii) benchmark target input consumption levels for the inefficient vineyards; iii) quantify the environmental benefits of moving towards operational efficiency in vine-growing, proving the eco-efficiency hypothesis, that is, that a reduction in input consumptions reduces potential environmental impacts; iv) estimate the economic gains brought about by efficient operational practices; and v) identify the best functioning vineyards to be used as operational and environmental references. The results from this study are expected to be of interest for a wide range of professionals, including LCA practitioners, researchers in the field of agriculture, policy makers and vineyard managers.

5.2 Materials and methods

5.2.1 Definition of the case study

5.2.1.1 Contextualization of the study

As in most Mediterranean countries, Spanish wine production is a key subsector within the agricultural sector. In Galicia, a series of Galician wines, such as "Rías Baixas", "Ribeira Sacra" or "Ribeiro" have acquired international renown over the years. In particular, "Rías Baixas" PDO accounted for a total of 3,814 ha of cultivated land, administered by 6,584 individual vine-growers and scattered in five distinct subzones across the province of A Coruña and Pontevedra in 2010 (Rías Baixas, 2011).

Vineyard exploitations in Galicia, unlike in other Spanish regions, are characterized by their small average size (0.58 ha/vine-grower) (Rías Baixas, 2011). Nevertheless, these smallholdings show a high variety of sizes, as well as geographical dispersion (Lloveras-Vilamanya, 1987). In fact, "Rías Baixas" wine is produced based on grape production arriving from 5 different valleys. However, only vine-growers belonging to the Salnés valley, which is the most important one —56% of the total production according to the Rías Baixas Regulatory Council (Rías Baixas, 2011)—, were included in this study. Representative vineyards do not stand for a compact plot of land in a given area, but to scattered vine-growing plantations throughout the entire valley.

5.2.1.2 Definition of the unit of assessment

Each homogenous entity whose input/output conversion undergoes assessment is named Decision Making Unit (DMU) or unit of assessment. The units of assessment chosen for this particular study consisted of a set of vineyards in the selected appellation. Any vineyard not included within the appellation was disregarded. Figure 5.1 depicts the input/output conversion process of each vineyard, summarizing the input and output flows subject to quantification for each DMU. Moreover, since DEA only involves the selection of the inputs and outputs in an LCA study, DEA and LCA elements are differentiated in Figure 5.1.

LCA inputs show a thorough identification of the operational and environmental aspects of the inventoried vineyards, following an attributional LCA perspective. Primary data were collected through face-to-face questionnaires answered by a set of vine-growers in the Salnés valley (capital: Vilagarcía de Arousa, 42°35′N 8°46′W). These questionnaires involved key operational aspects of the vineyards, such as fertilization, pesticide use, vineyard operations, machinery, water use or the infrastructure of the vineyard. LCA outputs comprised not only the cultivation of grapes for wine production, but also waste generation as well as direct emissions associated with fuel consumption, fertilizers and pesticides.

DEA embraced exclusively those inputs and outputs that are expected to generate relevant environmental and/or economic impact. Consequently, input and output amounts as well as prices and expected impacts per input/output unit were taken into account (Benedetto, 2010; Bosco et al., 2011; Gazulla et al., 2010; Notarnicola et al., 2003). Diesel, water, fertilizers, pesticides and concrete for infrastructure were selected as the DEA inputs, whereas commercial grapes were the unique DEA output. All the selected operational items for DEA implementation were assumed to be independent from each other. Therefore, occupied land was not included since its minimization would affect other inputs, such as fertilizers or diesel. Similarly, direct emissions to the different compartments were not included in the DEA matrix, given the direct proportion to some of the inputs included in the matrix. Hence, these emissions were indirectly minimized through direct minimization of the associated inputs (Lozano et al., 2009; Vázquez-Rowe et al., 2010). Future research on this topic could consider the inclusion of labor as an additional DEA input to provide the study with a stronger socioeconomic dimension.

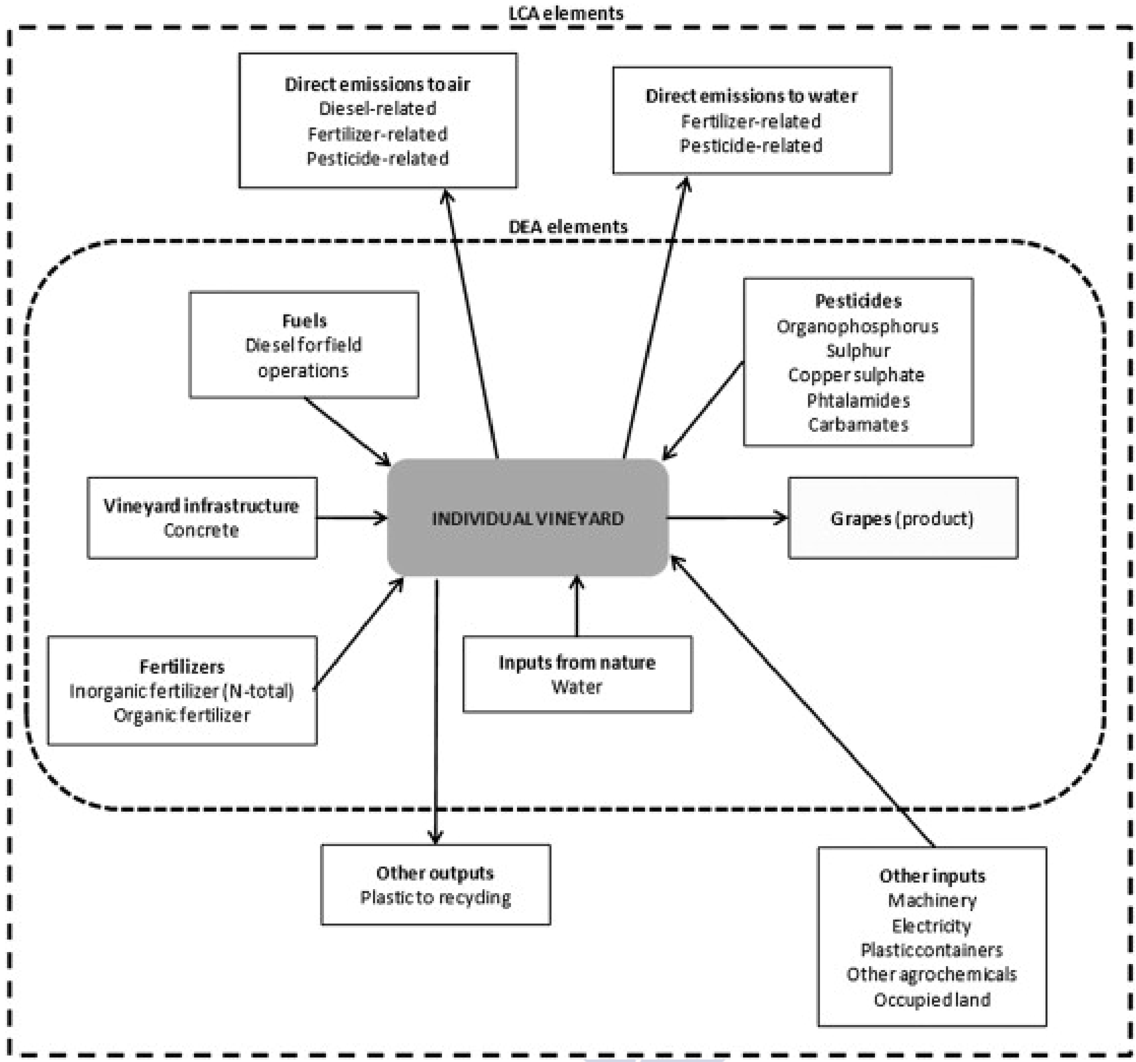

Figure 5.1. LCA and DEA items for each DMU (vineyard)

Despite the fact that wine is not the only product derived from winery transformation, grapes are the only product that is acquired from the cultivation phase, since all by-products are obtained once the grapes are delivered at the winery. Consequently, since a monofunctional system is involved, no allocation procedure was needed to distribute inventory data and environmental burdens.

5.2.2 LCA + DEA framework

Operational and environmental patterns in vine-growing in the “Rías Baixas” appellation were assessed according to the five-step LCA + DEA method (Iribarren, 2010; Vázquez-Rowe et al., 2010), outlined in Figure 5.2. The first stage of the methodology (step A) is based on data collection for the life cycle inventory (LCI) of each individual vineyard. Thereafter, the life cycle impact assessment (LCIA) for every vineyard is performed based on the LCI developed in the first step, constituting the environmental characterization of the farms for a given selection of impact categories (step B). The functional unit (FU) was 1.1 kg of harvested grapes, corresponding to the amount of grapes needed to produce a 750 mL “Rías Baixas” bottle (Rías Baixas, 2011; Petti et al., 2010).

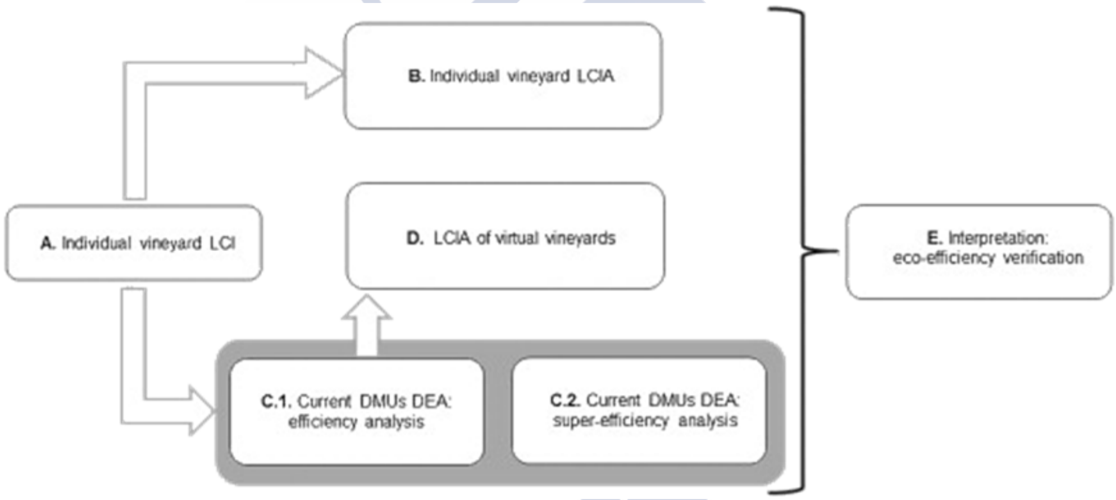

Figure 5.2. LCA and DEA items for each DMU (vineyard)

As step B, the third stage —DEA of the sample of vineyards— is based on the LCI of step A, but adopting an operational perspective on a 1-year basis. The operational efficiency of each DMU is calculated along with the projection for the inefficient DMUs (target values). DEA targets define virtual DMUs with lower consumption levels and/or higher production levels. Therefore, in step C.1, the performance of the DMUs is benchmarked from an economic/operational perspective. Furthermore, where a high number of efficient units is expected, step C.2 deals with the DEA super-efficiency analysis to rank efficient DMUs by assigning an efficiency score above 1, so that a shorter range of best performing entities can be identified (Cooper et al., 2007; Iribarren et al., 2010; Iribarren et al., 2011).

Target DMUs from step C.1 are subject to a second environmental characterization in step D. Finally, in step E, the potential environmental impacts for the virtual DMUs are compared to those for the current ones. Result interpretation at this point includes linking operational efficiency and environmental impacts, as well as quantifying the environmental consequences of operational inefficiencies.

5.3 Results

5.3.1 Inventory data

Data availability is a key requirement in LCA + DEA studies. Consequently, a thorough data collection phase is essential when it comes to applying this methodology. Table 5.1 assembles the main inventory data collected for the sample of 40 vineyard exploitations used in this study. These exploitations comprised a variety of sizes, ranging from 0.10 to 7.50 ha. The total sample size accounted for 55.55 ha. Data for this case study referred to the year 2010. Direct emissions from fertilizers were computed according to a series of different methods. On the one hand, nitrogen emissions were estimated based on the recommendations for nitrate (NO_4^{3-}), ammonia (NH_3) and nitrous oxide (N_2O) emissions performed by Brentrup et al. (2000). Therefore, potential emission rates were calculated using a series of structured estimation techniques. On the other hand, phosphate (PO_4^{3-}) emissions were calculated based on literature data (Cowell, 1998; Cowell and Clift, 1997).

Table 5.1. Selected inventory data for 40 "Rías Baixas" vineyards

DMU	Grapes (kg/year)	Diesel (l/year)	Concrete (kg/year)	Water (l/year)	Organic fertilizer (kg/year)	N-fertilizer (kg/year)	Organophosphorus pesticieds (kg/year)	Phtalamides (kg/year)	Carbamates (kg/year)	Cu-pesticide (kg/year)	S-pesticide (kg/year)
1	31,100	1,575	7,082	54,900	1,760	115.21	30.78	23.84	51.77	24.15	65.52
2	9,600	771	1,416	12,750	400	0.01	2.77	7.94	5.27	4.56	18.36
3	13,000	1,325	1,948	13,900	10,000	10.01	17.35	5.38	11.19	6.27	14.40
4	90,000	3,545	13,279	72,000	0	432.00	96.7	65.00	15.4	18.75	60.00
5	5,172	465	761	10,400	0	15.00	0	4.34	5.34	2.93	8.62
6	30,000	1,030	3,541	32,800	0	361.00	18.96	36.85	22.80	11.4	40.10
7	8,400	615	1,594	10,800	864	38.70	3.91	6.01	3.19	1.62	14.40
8	9,160	805	1,416	12,500	500	14.40	11.82	4.76	2.66	4.56	16.04
9	8,000	600	885	6,400	0	72.10	6.00	2.63	6.05	2.85	10.02
10	15,500	645	3,541	26,300	6,600	9.00	12.85	8.34	23.44	5.40	40.1
11	30,000	925	4,426	35,000	1,800	112.51	5.44	10.00	35.00	15.00	42.00
12	15,000	470	2,656	16,800	900	56.26	2.61	4.80	16.80	7.20	20.16
13	20,000	855	5,312	33,280	1,100	40.00	13.38	16.88	5.10	9.00	96.00
14	7,604	322	885	13,000	500	18.00	0	6.56	1.77	5.40	10.02
15	7,500	213	1,416	7,000	0	36.00	1.52	2.8	0.22	0.84	15.68
16	30,500	508	4,604	30,000	0	36.00	15.00	7.5	18.4	21.00	52.13
17	7,500	730	1,771	12,000	2,200	72.00	4.50	4.17	0.36	23.28	20.05
18	3,000	230	531	6,050	0	12.00	0	4.86	0.26	0.99	5.28
19	25,000	1,080	5,754	91,000	31,000	0	0	13.44	2.52	18.9	39.30
20	7,000	785	1,594	18,000	1,440	27.00	6.21	4.10	9.31	2.16	10.08

Table 5.1. Selected inventory data for 40 "Rías Baixas" vineyards (continuation)

DMU	Grapes (kg/year)	Diesel (l/year)	Concrete (kg/year)	Water (l/year)	Organic fertilizer (kg/year)	N-fertilizer (kg/year)	Organophosphorus pesticieds (kg/year)	Phtalamides (kg/year)	Carbamates (kg/year)	Cu-pesticide (kg/year)	S-pesticide (kg/year)
21	19,000	500	2,479.00	29,500.	1,000.00	45.00	12.23	9.99	10.57	29.07	28.07
22	16,000	955	2,125	12,025	1,600	0	2.28	6.00	14.08	1.92	22.40
23	1,200	35	177	3,000	40	4	0.00	0.00	1.05	0.45	14.11
24	4,000	235	1,115	12,000	675	0	2.78	9.00	9.89	3.59	26.88
25	13,800	190	2,479	12,000	880	0	10.80	5.40	5.09	1.44	20.16
26	6,200	600	885	24,020	800	23.00	4.05	0	12.74	10.56	44.80
27	5,000	525	708	12,300	350	0	11.13	6.80	1.14	2.70	16.80
28	5,200	520	708	9,600	6,000	0	1.80	8.70	3.70	1.44	8.02
29	2,900	275	443	5,200	220	0	2.40	2.80	3.01	1.42	14.56
30	12,000	345	1,771	13,200	1,125	0	9.00	4.50	0.72	18.48	20.05
31	8,000	315	779	9,000	160	0	0.00	5.63	5.10	1.80	16.80
32	9,120	334	1,239	10,700	0	36.00	7.45	4.06	10.16	3.99	15.36
33	3,362	237	708	3,600	800	14.00	0.00	2.73	0.18	1.40	5.12
34	55,000	1,560	8,853	77,000	2,500	225.00	15.95	22.00	77.00	33.00	107.80
35	2,500	130	390	5,000	234	0	3.55	3.00	1.60	0.88	9.60
36	20,000	1,150	2,231	37,500	800	0	7.25	24.80	29.00	3.00	56.00
37	1,900	30	212	2,750	50	0	0.36	2.00	0.06	0.45	3.37
38	13,000	610	1,771	9,600	2,000	0	0	5.29	6.23	5.70	11.20
39	18,400	965	4,426	49,000	800	0	70.90	24.00	20.64	7.20	67.20
40	4,026	165	443	4,400	500	0	0	3.18	0.24	1.80	7.20

Pesticide use constitutes one of the most relevant environmental issues in vineyards (Saint-Ges and Bélis-Bergouignan, 2009). Emissions related to pesticide use were calculated based on PestLCI, a modular model for estimating pesticide emissions derived from field application to the different environmental compartments: air, surface water and groundwater (Birkved and Hauschild, 2006). A series of specific data corresponding to the regional climatic and soil characteristics were taken into account in order to estimate these emissions (Calvo de Anta, personal communication; Meteogalicia, 2011; Trueba et al., 1998).

Emissions related to diesel consumption by agricultural machinery in field operations were calculated based on the EMEP/EEA Air Pollutant Emission Inventory Guidebook – 2006 (EMEP-Corinair, 2011). Finally, data for background processes were taken from the ecoinvent® database (Frischknecht et al., 2007).

5.3.2 Current environmental characterization

The life cycle inventory of each vineyard was implemented into Simapro 7 (Prè-Product Ecology Consultants, 2011) to carry out the LCIA. Acidification (AP), eutrophication (EP), global warming (GWP), photochemical oxidant formation (POFP) and land competition (LC) were the impact categories included by using the CML baseline 2000 method (Guinée et al., 2001), while eco-toxicity (Etox) was evaluated according to the USEtox method (Rosenbaum et al., 2008). CML impact category selection was based on the state of the art of the viticulture sector (Petti et al., 2010), while Etox was included to quantify the main toxicological impacts linked to pesticide utilization in the viticulture sector (Juraske et al., 2009; Komárek et al., 2010). The FU was 1.1 kg of harvested grapes. Figure 5.3 shows the environmental characterization results obtained for the assessed vineyards.

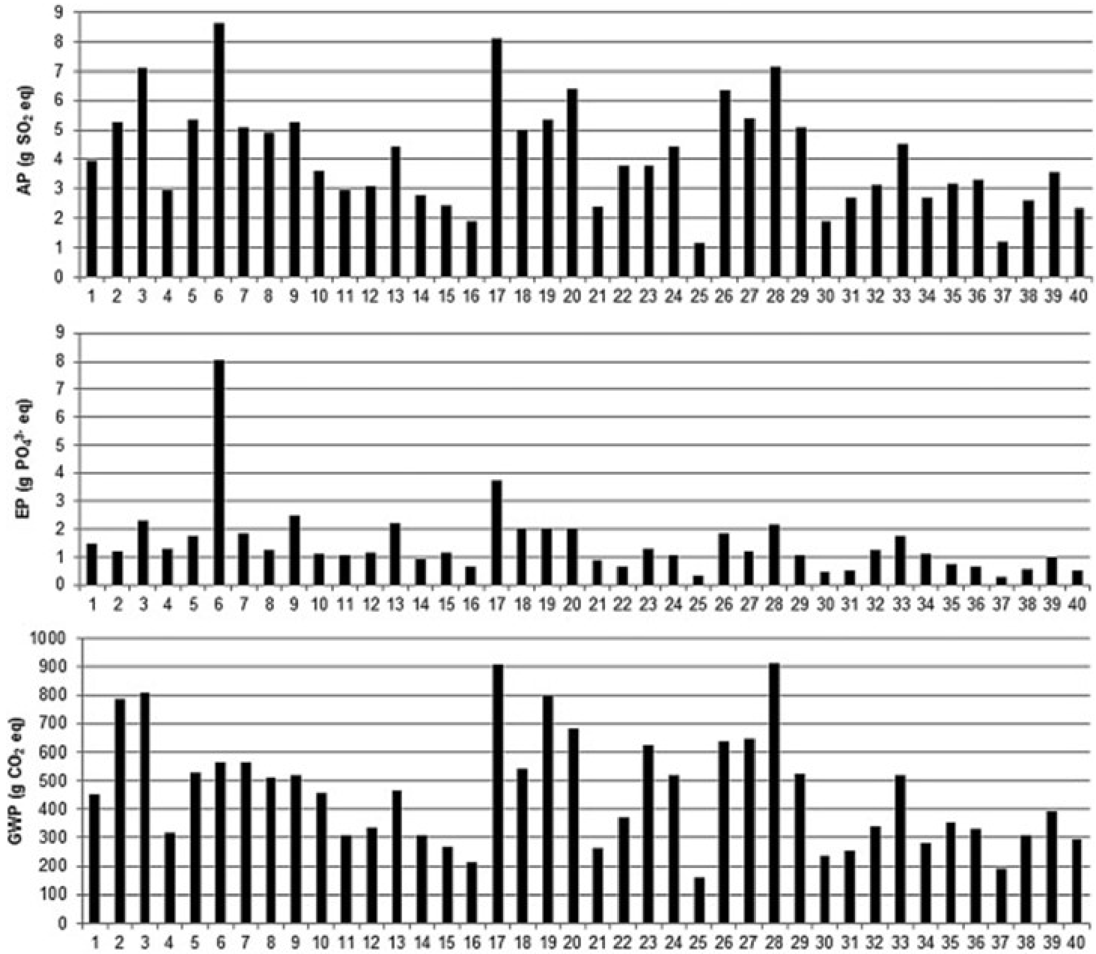

Figure 5.3. Current environmental characterization of the selected grape production exploitations (results per FU). AP: acidification potential; EP: eutrophication potential; GWP: global warming potential; POFP: photochemical oxidation formation potential; LC: land competition; Etox: eco-toxicity; CTUe: comparative toxic units

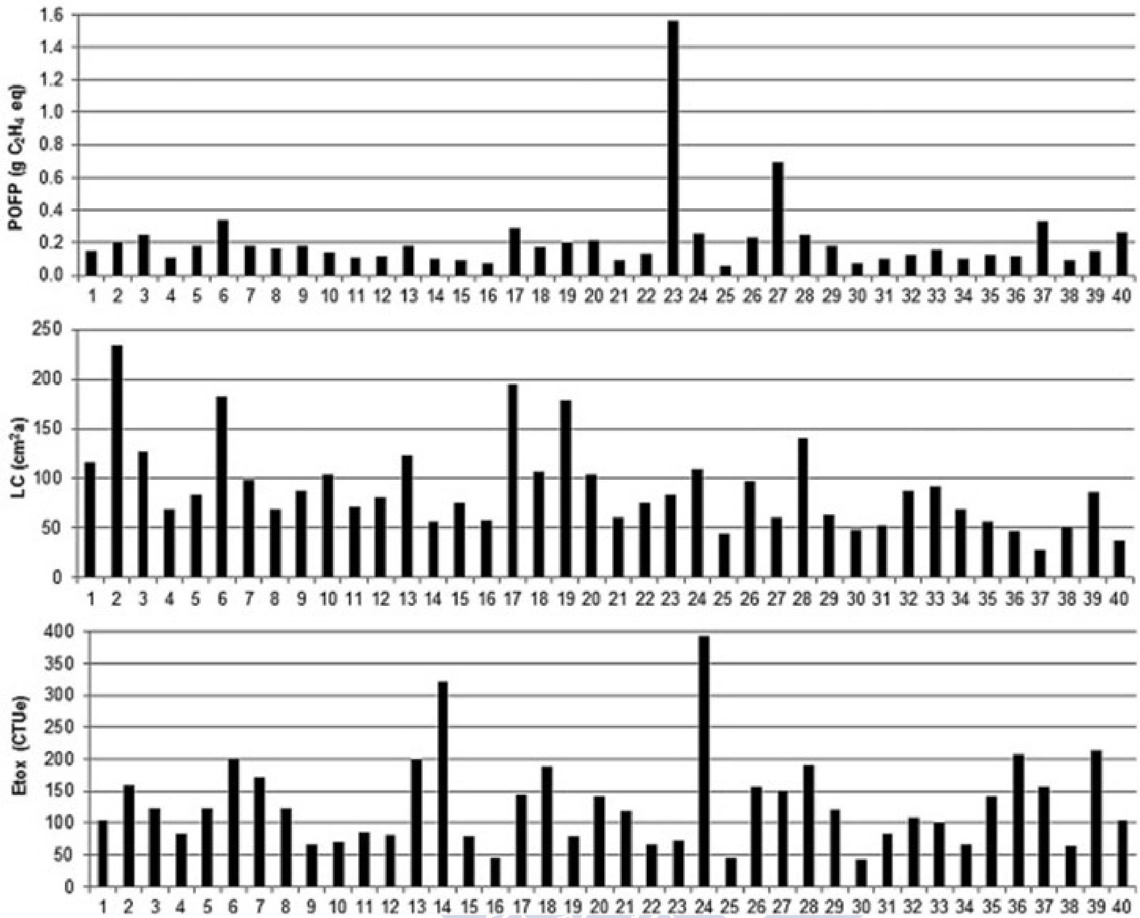

Figure 5.3. Current environmental characterization of the selected grape production exploitations (results per FU). AP: acidification potential; EP: eutrophication potential; GWP: global warming potential; POFP: photochemical oxidation formation potential; LC: land competition; Etox: eco-toxicity; CTUe: comparative toxic units (continuation)

5.3.3 DEA performance

An input-oriented slacks-based measure of efficiency (SBM-I) model with constant returns to scale (CRS) was selected in order to discriminate between efficient and inefficient vineyards (Tone, 2001). Therefore, a non-radial DEA approach was followed, and convexity, scalability and free disposability of inputs and outputs were assumed for the determination of the production possibility set. Details on the formulation of the model can be found in recent literature on the application of LCA + DEA methodology (Iribarren et al., 2011; Vázquez-Rowe et al., 2011). The DEA matrix, which consists of the data to be implemented in the model, only includes the most relevant outputs and inputs from the LCIs, and therefore it corresponds with Table 5.1. The selection of an input-oriented model is due to two main factors. In the first place, the study aims at achieving input minimization without hindering grape production. Secondly, the model selection is based on the fact that grape productivity per hectare is limited by a quota system. CRS were assumed taking into account that vine-growers operate in a competitive market and all the assessed vineyards work for the same appellation, under the same incentive and competitiveness conditions (Lozano et al., 2009).

DEA-solver Pro was the software used for DEA computation (Saitech, 2011). Results are gathered in Table 5.2, which only includes the efficiency score and the target operational benchmarks for the vineyards deemed inefficient (efficiency score $\Phi < 1$). Vineyards with $\Phi = 1$ (i.e., efficient units) were excluded since their target values are equivalent to their current values. A total of 16 exploitations (out of 40) were found to operate inefficiently, with scores ranging from 36% to 71%. However, most of the DMUs (24 vineyards, i.e. 60% of the sample) were found efficient.

Given that a high number of vineyards were deemed efficient, a super-efficiency analysis was conducted in order to determine which vine-growers showed the best performing patterns from an operational perspective (Iribarren et al., 2010). An input-oriented slacks-based measure of super-efficiency (Super-SBM-I) model with CRS was chosen for the discrimination between the efficient units (Tone, 2002). Table 5.3 shows the new efficiency scores (ψ) computed for the 24 efficient DMUs. These results are discussed later in section 5.4.2.

Table 5.2. Efficiency scores (Φ) and operational reduction percentages for the inefficient vineyards

DMU	Grapes (kg/year)	Diesel (l/year)	Concrete (kg/year)	Water (l/year)	Organic fertilizer (kg/year)	N-fertilizer (kg/year)	Organophosphorus pesticieds (kg/year)	Phtalamides (kg/year)	Carbamates (kg/year)	Cu-pesticide (kg/year)	S-pesticide (kg/year)
1	0.47	67.11	33.72	44.28	100	68.14	50.31	67.91	63.76	11.33	18.87
2	0.62	62.1	28.89	7.11	30.75	100	73.97	0	28.72	49.5	0
3	0.55	53.96	9.09	30.94	80	100	100	1.7	44.31	9.09	22.22
7	0.65	65.78	21.04	12.6	89.38	39.04	25.37	0	85.53	6.48	0
8	0.71	79.78	0	21.7	57.29	23.25	66.27	0	0	36.31	1.19
10	0.46	66.91	21.38	48.75	85.02	100	5.58	27.28	75.62	70.05	43.53
13	0.51	53.99	48.98	22.52	66.87	26.2	70.92	0	87.82	55.9	61.02
17	0.46	70.82	20	41.67	100	50	66.18	32.85	37.78	96.39	21.79
20	0.47	46.78	41.67	70.77	51.39	100	83.91	36.04	33.85	61.11	2.78
24	0.36	73.12	59.9	51.75	84.41	0	72.59	53.22	98.72	73.61	73.61
27	0.49	84.96	21.05	41.16	62.41	0	91.43	22.54	86.15	56.14	47.21
28	0.6	74.59	12.91	32.56	94.81	0	58.4	48.24	74.4	0	0
29	0.52	58.48	36.2	37.26	73.64	0	100	27.18	38.54	54.19	58.17
34	0.7	5.94	3.09	29.14	90.53	38.69	10.42	0	74.28	43.86	0
35	0.52	69.64	28.23	27.63	71.88	0	86.56	12.28	95.07	32.33	53.81
39	0.49	69.89	53.52	45.65	39.47	0	95.05	19.3	97.18	39.47	51.43

Table 5.3. Computed super-efficiency scores (ψ) for the vineyards that were identified as efficient in the SBM-I model

DMU	ψ	DMU	ψ	DMU	ψ	DMU	ψ
4	1.18	14	1.01	22	1.15	32	1.00
5	1.00	15	1.35	23	1.00	33	1.02
6	1.00	16	1.10	25	1.17	36	1.03
9	1.03	18	1.00	26	1.00	37	1.17
11	1.02	19	1.00	30	1.05	38	1.06
12	1.01	21	1.00	31	1.03	40	1.03

5.3.4 Target environmental characterization and eco-efficiency

The target projections computed through DEA in step C.1 led to carry out new LCIAs involving the modified LCI data for the inefficient vine-growing plots. Hence, a new environmental characterization was obtained for each of these inefficient vineyards, in order to determine their potential environmental impacts if they were managed under efficient operational conditions (Table 5.4).

The final stage (step E) aims at comparing the potential environmental impacts calculated for the virtual vine exploitations with respect to those for the current ones. Thus, the purpose of this step is to prove that environmental impacts depend on the efficiency with which operations are performed (Lozano et al., 2009; Vázquez-Rowe et al., 2010). This environmental approach of the five-step LCA + DEA method —focused on the verification and quantification of the environmental gains linked to reduced consumption levels— contrasts with other DEA approaches that aim at computing environmental impact efficiencies by implementing current impacts as components of the DEA matrix (Hua et al., 2007; Lozano et al., 2010).

Results regarding the entire set of vineyards (Figure 5.4) show an important average reduction in the environmental impacts for the six categories included in the study, with potential improvements of up to 39% in the case of GWP. Figure 5.4 shows how this environmental enhancement is linked to significant reductions in input consumption levels, with average reductions between 8% (phtalamides) and 30% (organic fertilizer) for the different inputs.

Table 5.4. Target environmental impact gains (%) for inefficient vineyards

DMU	AP	EP	GWP	POFP	Etox	LC
1	56.57	51.43	52.26	51.63	56.22	31.13
2	3.85	6.46	2.72	5.82	16.89	2.20
3	27.11	53.38	34.20	31.02	52.98	47.48
7	10.83	24.61	12.39	11.95	13.12	18.35
8	4.98	10.08	5.41	7.01	32.37	10.85
10	32.92	39.79	38.23	35.63	33.00	37.40
13	16.98	22.35	15.61	21.83	34.10	17.45
17	23.05	46.36	26.05	26.99	69.37	30.78
20	9.05	24.84	10.96	11.24	60.68	18.48
24	32.74	38.04	29.23	29.43	60.38	32.35
27	10.10	20.27	9.39	4.57	25.45	26.25
28	33.91	58.93	47.43	37.16	46.10	64.57
29	12.46	19.81	13.23	19.14	30.07	26.52
34	22.66	26.33	18.12	23.15	19.49	21.03
35	19.16	27.71	18.87	26.63	13.58	28.94
39	24.46	36.50	17.97	33.33	18.95	23.50

AP: acidification potential; EP: eutrophication potential; GWP: global warming potential; POFP: photochemical oxidant formation potential; LC: land competition; Etox: eco-toxicity

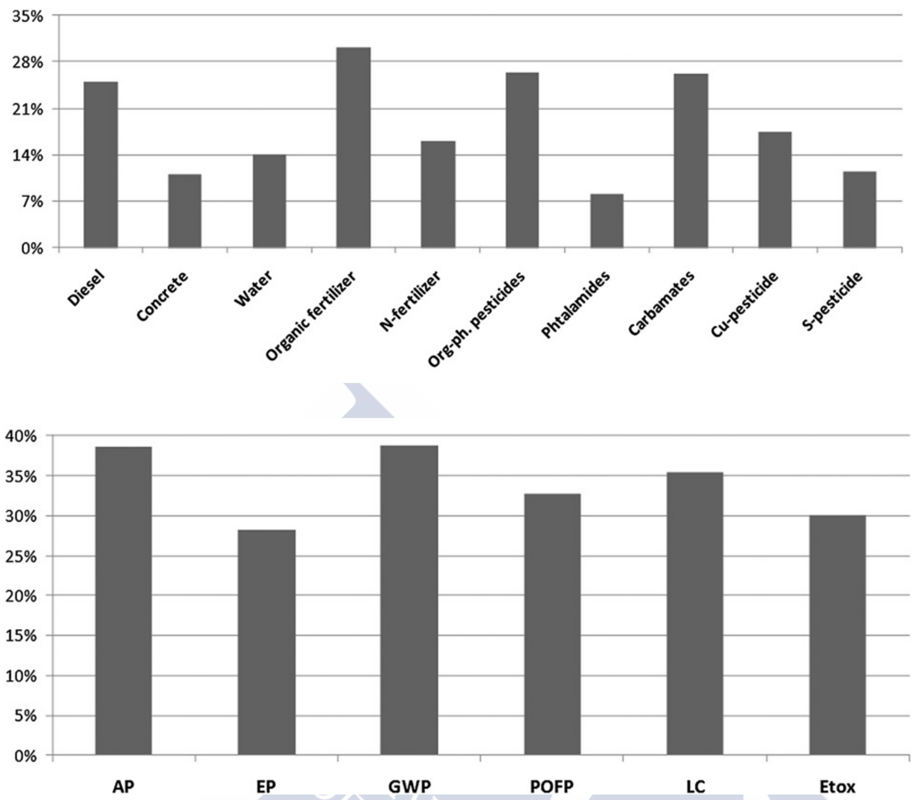

Figure 5.4. Average reduction in the environmental impacts and material inputs. AP: acidification potential; EP: eutrophication potential; GWP: global warming potential; POFP: photochemical oxidation formation potential; LC: land competition; Etox: eco-toxicity

5.4 Discussion

5.4.1 Environmental and operational performance of winegrowers

Table 5.5 gathers the average current environmental impacts associated with the cultivation and harvesting of 1.1 kg of grapes for vinification in the “Rías Baixas” appellation. These values are in the range of data reported in previous LCA studies on grape production for vinification (Gazulla et al., 2010; Point et al., 2012). However, as inferred from the high standard deviations in Table 5.5, these results suffer from strong variations between the different vineyards. This observation stresses the usefulness of using an LCA + DEA strategy in order to avoid the use of average inventories (Iribarren et al., 2011; Vázquez-Rowe et al., 2010).

Table 5.5. Average environmental characterization results per FU (current average impacts)

Impact category	Unit	Average ± standard deviation
AP	g SO_2 eq.	4.13 ± 1.81
EP	g PO_4^{3-} eq.	1.48 ± 1.28
GWP	g CO_2 eq.	462.7 ± 199.5
POFP	g C_2H_4 eq.	0.213 ± 0.245
LC	cm^2a	89.84 ± 45.23
Etox	CTUe	129.9 ± 72.2

AP: acidification potential; EP: eutrophication potential; GWP: global warming potential; POFP: photochemical oxidant formation potential; LC: land competition; Etox: eco-toxicity; CTUe: comparative toxic units.

Results obtained in the DEA study (step C.1) showed that the majority of the assessed vine-growing farms in the “Rías Baixas” appellation (60%) currently work under operationally efficient conditions ($\Phi = 1$). In fact, only 17.5% of the exploitations failed to reach an efficiency score of 50%. Therefore, on average, the potential for minimization of the input resources was low (Figure 5.4) when compared to input reductions estimated for other agro-food systems through the five-step LCA + DEA method (Iribarren et al., 2011; Lozano et al., 2009).

A series of differences were identified between the vine-growing farms depending on their production size. With this purpose, vine-growing exploitations were divided into three differentiated production levels: i) low production vineyards (<10 t/year of grapes); ii) intermediate production exploitations (10–30 t/year); and iii) high production farms (> 30 t/year). As depicted in Figure 5.5, vine-growers with low production exploitations presented an average efficiency of 79%, lower than that of intermediate production vineyards (83%) and high production sites (86%). These differences in the average scores are linked mainly to operational differences regarding diesel consumption, phtalamide pesticides use and, to a lesser extent, copper-pesticides application. In particular, for the phtalamide pesticides the difference between the average input efficiency for high production farms (90%) and low production ones (67%) was found to be the highest of all the selected inputs. However, regarding water use and fertilizer application, no clear tendencies related to production size could be extrapolated from the sample. Nonetheless, high production vineyards presented the highest individual input efficiency scores for 8 out of 10 inputs. This observation suggests the advisability of exploitations with high grape production rates from an operational perspective.

Although one of the key strengths of LCA + DEA methodology refers to the accomplishment of the operational and environmental benchmarking of individual units, this technique can be useful to identify performance trends of an average facility as well. According to Vázquez-Rowe et al. (2011), the efficiency score (Φ) of the average exploitation should be computed through DEA by adding an extra DMU defined by the values for the average exploitation. In other words, average inventory data should be used to define a virtual DMU to be included in the DEA study. For the present case study, a virtual DMU 41 was included based on average inventory data computed as the mean values of each of the considered DEA items in Table 5.1 The reason for following this approach was to attain potential minimization results and the efficiency score for the average vineyard, rather than simply calculating the average efficiency of the selected sample.

The results concerning the average vine-growing farm are summarized Figure 5.6. As observed, an efficiency score of 56% was computed for the average unit, which was therefore deemed inefficient. A total of 5 inputs presented an individual efficiency score below the efficiency score of the average exploitation. This was the case for diesel (32%), organic fertilizer (36%), Cu-pesticide (30%), and carbamates (31%). Additionally, the fifth input in this group, N-fertilizer, presented a zero efficiency score, which implies that this input should be avoided by the target average facility. In this respect, it should be noted that DEA is a non-parametric approach that relies on basic assumptions (section 5.3.3) so that the results are mainly linked to the observed data. In this way, the recommendation to avoid the N-fertilizer in the average vineyard is due to the production possibility set determined by the observed data and the assumptions of scalability, convexity and free disposability. The target vineyard for the average facility would use organic fertilizer, but not inorganic fertilizer. On the other hand, another group of 5 inputs showed efficiency scores above the efficiency score for the average farm. Phtalamides (100%), concrete (92%) and S-pesticide (85%) were identified as the best performing inputs, while organophosphorus pesticides (79%) and water (76%) also achieved high individual efficiency scores.

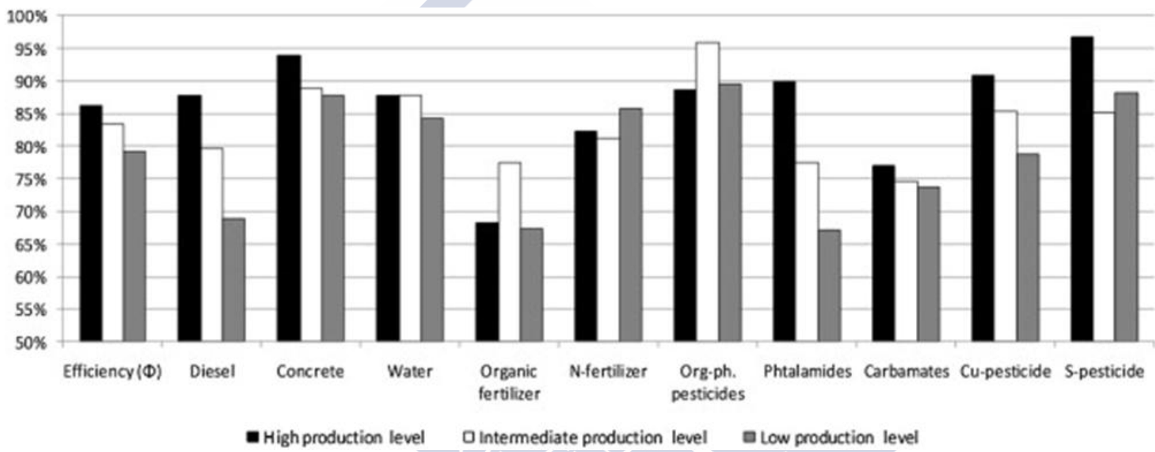

Figure 5.5. Average efficiency and individual input efficiency for the selected grape production farms based on production level. High production level: > 30 t/year; Intermediate production level: 10-30 t/year; Low production level: <10 t/year

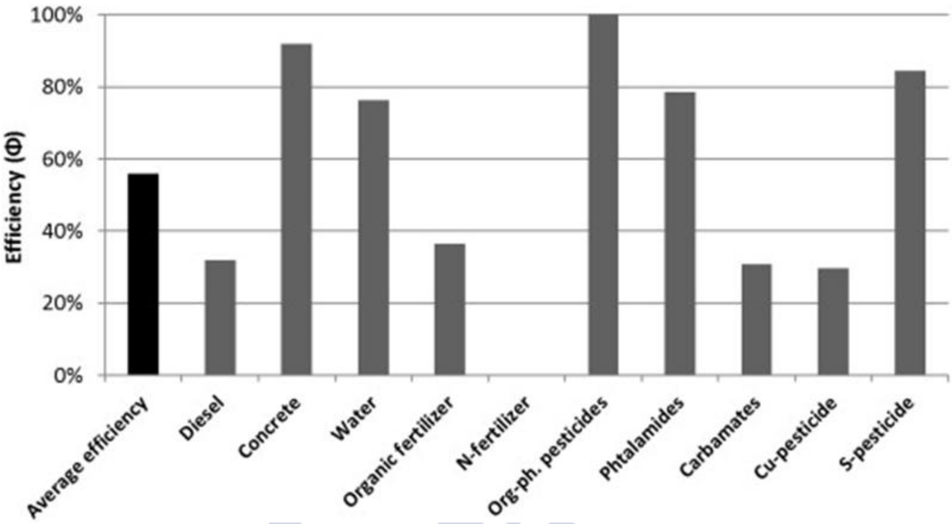

Figure 5.6. Efficiency score of the average exploitation as compared to the individual input efficiencies of the average exploitation for the selected sample

The results estimated for the average vineyard highlight the fact that diesel consumption, fertilizer usage and certain phytosanitary application practices should be revised in order to guarantee a net improvement in operational efficiency. Furthermore, the optimization of the use of these resources would also facilitate increased economic gains and reduced environmental impacts.

5.4.2 Reference values on super-efficient exploitations

Best performers identified through a super-efficiency analysis can be taken as reference vineyards in order to propose benchmarks for environmental policy making (Iribarren et al., 2010). Thereby, the joint application of LCA and DEA would guarantee an appropriate relationship between environmental regulations and eco-efficiency (Wang et al., 2011). In this study, reference vineyards were selected based on the super-efficiency analysis in section 5.3.3 and their production capacity. Table 5.6 gathers the environmental characterization results for the best super-efficient farm in each of the three production levels previously defined in section 5.4.1.

Table 5.6. Environmental characterization results (per FU) for the selected super-efficient vine-growing plots

Production level	DMU	Φ (%)	AP (g SO_2 eq)	EP (g PO_4^{3-} eq)	GWP (g CO_2 eq)	POFP (g C_2H_4 eq)	Etox (CTUe)	LC (cm^2a)
Low	15	135.42	2.41	1.16	268.3	0.091	78.82	76.13
Intermediate	25	116.92	1.15	0.32	158.9	0.058	45.65	43.98
High	4	118.33	2.96	1.30	317.7	0.107	83.08	68.44

DMU: decision making unit; AP: acidification potential; EP: eutrophication potential; GWP: global warming potential; POFP: photochemical oxidant formation potential; LC: land competition; Etox: eco-toxicity; CTUe: comparative toxic units

Based on the results of the best performer in each production level, the selected intermediate production level farm attained characterization values up to 75% lower than the reference high production level vineyard. Meanwhile, the best-performing low production level farm presented intermediate results, with environmental gains ranging from 5% (Etox) to 19% (AP) with respect to the best-performing high production vine-growing site, except for LC, an impact category for which the low production best-performer showed the greatest impact.

When these results are crossed with those in Figure 5.5, the fact that high production size farms presented the best-performing farm with the highest associated environmental impact for all the assessed categories (except for LC) could be unexpected, since high production farms showed the highest efficiency figures. However, because operational efficiencies are relative and they depend on the shape of the production possibility set, parallelisms between low environmental impacts and high operational efficiencies —other than the verification of the traditional eco-efficiency hypothesis regarding current and target facilities— should be avoided. Nevertheless, it can be interpreted that the potential for minimizing the environmental impacts is rather low in high production farms, while operational and environmental minimization is potentially high for low production vine-growing farms. For intermediate farms, results indicate little space for environmental reduction and economic gains due to improved operational performance.

5.4.3 Economic gains

It is frequently recognized in existing literature that one of the main limitations of LCA as a sustainability assessment tool is its exclusive focus on environmental issues, rather than integrating environmental, economic and social indicators (Reap et al., 2008b). Nevertheless, the application of the five-step LCA + DEA method to a set of primary sector activities, such as extensive aquaculture (Iribarren, 2010; Lozano et al., 2009), fisheries (Vázquez-Rowe et al., 2010; Vázquez-Rowe et al., 2011) or milk farms (Iribarren et al., 2011), has proven this methodology suitable for the integration of environmental and economic issues.

Table 5.7 presents the economic reductions that could be attained if the inefficient vine-growing exploitations were to upgrade their performance in order to score efficient operation. To carry out this economic estimation, the average prices of diesel, concrete, fertilizers and pesticides in 2010 were obtained from a series of representative companies that sell these products in the analyzed appellation or from institutions that monitor these prices (Comercial Agro Leiro, personal communication; Construmatica, personal communication; FORVI, personal communication). Water inputs were excluded due to their minimal contribution to economic savings by vine-growers, since all growers reported obtaining their water from wells.

Approximate annual total savings for inefficient farms ranged from 400 to 5,150 €. The average economic saving per FU for inefficient vine-growing farms was 0.14 €. Consequently, if an average sale price of 1.40 € per kilogram of grapes in the “Rías Baixas” appellation is assumed (Xunta de Galicia, 2009), additional profits of 10% could be achieved by vine-growers. On average, the most significant economic savings were identified for carbamate pesticides (25% of the average total saving), concrete (21%) and diesel (19%). Nevertheless, if phytosanitary products were included overall, they would represent 50% of the average total saving.

Table 5.7. Economic savings linked to the accomplishment of operational targets

DMU	Economic savings (€/year)									*Total econ. savings (€/FU)*
	Diesel	*Concrete*	*Fert.*	*Orga. pest.*	*Phtal.*	*Carbam.*	*Cu-pest.*	*S-pest.*	*Total*	
1	792.70	1,218.00	532.40	101.20	745.10	185.00	27.60	95.90	3,698.00	*0.13*
2	422.70	208.70	37.20	25.50	0.00	25.40	0.00	79.10	798.50	*0.09*
3	536.20	89.40	160.00	322.30	4.20	52.60	7.10	20.00	1,191.90	*0.10*
7	303.40	171.00	233.60	12.30	0.00	97.10	0.00	3.70	821.10	*0.11*
8	481.70	0.00	5.70	104.60	0.00	0.00	0.40	58.00	650.40	*0.08*
10	323.70	386.10	112.20	16.40	104.70	205.80	39.00	132.60	1,320.50	*0.09*
13	346.20	1,326.90	14.70	102.80	0.00	710.00	130.90	176.30	2,807.80	*0.15*
17	387.70	180.60	44.00	88.80	63.10	0.00	9.80	786.20	1,560.20	*0.23*
20	275.40	338.70	223.90	117.20	68.10	32.80	0.60	46.30	1,102.90	*0.17*
24	128.90	340.80	172.40	25.10	220.50	515.50	44.20	92.60	1,539.90	*0.42*
27	334.50	76.00	66.10	194.20	70.50	76.70	17.70	53.10	888.80	*0.20*
28	290.90	46.60	113.80	31.30	193.20	27.40	0.00	0.00	703.20	*0.15*
29	120.60	81.70	49.00	71.50	35.00	12.20	18.90	27.10	416.10	*0.16*
34	69.50	139.50	684.60	20.60	0.00	3,725.90	0.00	507.20	5,147.40	*0.10*
35	67.90	56.10	50.90	90.50	16.00	129.90	11.50	9.90	433.70	*0.10*
39	505.80	1,280.20	95.50	1,534.60	213.20	1,063.50	77.20	99.60	4,797.80	*0.29*
Av. savings	**336.70**	**366.80**	**162.20**	**178.70**	**108.40**	**428.70**	**24.10**	**136.70**	**1,742.40**	***0.14***

DMU: decision making unit; FU: functional unit; €: euros; Fert: fertilizer; Orga. Pest: organophosphorus pesticides; Phtal: phtalamides; Carbam: carbamates; Cu-pest: copper pesticide; S pest: sulphur pesticide

5.5 Conclusions

The joint implementation of LCA and DEA has proven to be an adequate technique to assess the operational and environmental performance of vine-growing. A set of 40 vineyards from the "Rías Baixas" appellation was analyzed following LCA + DEA methodology.

A high percentage of the vineyards (60%) were found to operate efficiently. Discrimination between these efficient vine-growing farms was carried out through super-efficiency analysis in order to appoint the best-performing farms depending on their production size. The environmental characterization of the selected super-efficient vineyards led to propose environmental reference values for the different production sizes.

The projections computed for the inefficient units (target operational values) resulted in lower environmental impacts. Environmental reductions ranging from 28% to 39% were estimated. Hence, eco-efficiency criteria were accomplished.

Finally, input minimization was considered from an economic point of view, in order to determine to what extent operational optimization may translate into economic benefits for vine-growers. Results presented in this research confirmed that improving the operational performance of inefficient vine-growing farms is not only worthwhile from an environmental perspective, but also regarding economic profits. The inefficient vineyards assessed in this study would obtain on average an additional economic benefit of 10% with respect to the sale price for grapes, provided that these facilities turned to an efficient operation.

5.6 References

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Chapter 6

Comparative LCA in the wine sector: Biodynamic vs. conventional viticulture activities[1]

Summary

Viticulture is currently experiencing a gradual shift to more sustainable production practices. Many producers see in this shift an opportunity to increase their sales, especially in a context which is greatly influenced by the reduction in wine sales due to changes in human consumption patterns as well as the world economic crisis. Hence, both organic and biodynamic viticulture have begun to be applied in many vineyards as alternative attractive agricultural techniques. Nevertheless, it remains unclear which are the exact environmental benefits (or drawbacks) of applying these techniques for numerous environmental impacts, such as climate change or toxicity. Therefore, the main goal of this study is to perform an environmental evaluation by means of Life Cycle Assessment (LCA) for three different viticulture techniques within the appellation of "Ribeiro" (NW Spain): biodynamic cultivation sites, conventional vineyards and combination of biodynamic-conventional wine-growing (i.e. biodynamic site lacking certification). Moreover, two methodological improvements in the field of wine LCA studies are suggested and developed in terms of land use impact categories and labor inclusion in life-cycle thinking. Results demonstrate that biodynamic production implies the lowest environmental burdens, while the highest environmental impacts were linked to conventional agricultural practices. The main reasons for this strong decrease in the environmental impacts for the biodynamic site is related to the large reduction in diesel consumption (80%), due to the lower application of plant protection products and fertilizers as well as the introduction of manual work rather than mechanized activities in the vineyards.

[1] Villanueva-Rey, P., Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2014. Comparative life cycle assessment in the wine sector: biodynamic vs. conventional viticulture activities in NW Spain. J. Clean. Prod. 65, 330–341.

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6.1 Introduction

Currently, viticulture is experiencing a gradual change to more sustainable production patterns (Gabzdylova et al., 2009). In fact, many producers see in this shift an opportunity to increase their sales, especially in a context which is greatly influenced by the reduction in wine sales due to changes in human consumption patterns as well as the world economic crisis (OIV, 2011). Therefore, many producers have initiated or have already accomplished the conversion towards field operations that improve the environmental profile of wine production. Hence, both organic and biodynamic viticulture have begun to be applied in many vineyards as novel and attractive agricultural techniques. On the one hand, organic viticulture is characterized by the avoidance of mineral fertilizers and plant protection substances of synthetic origin. Organic products, including those related to the viticulture and vinification sector, are regulated by Member States in the European Union (EU) based on a common legislation (European Commission, 2007; European Commission, 2008; EUROSTAT, 2012), which covers the supervision of techniques, compliance with standards and labelling. On the other hand, biodynamic viticulture can be seen as a specific type of organic viticulture that is based on the radical consideration of two postulates that characterize organic viticulture, as explained in more detail in section 6.1.1. Moreover, these techniques seek higher independence from the use of machinery, and consequently, fossil fuels, by implementing artisanal field operation strategies. However, it is important to point out that despite the attractive gains in terms of input minimization when organic or biodynamic practices are applied to viticulture, there is also an important reduction in the harvest yield of these vineyards (White, 1995; Hassall et al., 2005; Badgley et al., 2007; Seufert et al., 2012).

Nevertheless, the wines obtained with these methods are characterized by an exceptional quality in terms of organoleptic characteristics, with higher doses of polyphenols and lower concentrations of sulfites. Currently, Spain is the European leader regarding organic agriculture, with 1.08 million ha used for this purpose; 9.5% of this surface area corresponds to viticulture (Eurostat, 2012 and INE, 2012).

6.1.1 Biodynamic viticulture

Biodynamic agriculture was developed in the 1920s based on a set of conferences performed by the philosopher Rudolf Steiner (Steiner and Gardner, 1993). This type of agriculture considers a holistic approach concerning the exploitation of the natural resources, taking into consideration the sustainability of different elements, such as the crops themselves, animal life preservation or the maintenance of a high quality soil, in order to recover, preserve or improve ecological harmony (Lotter, 2003). This perspective is achieved through a sharp reduction of external inputs into the production system, the use of a set of preparations for fertilization purposes and the application of homoeopathic treatments based on infusions or plant extracts (Table 6.1).

Table 6.1. List of the main biodynamic preparations. Source: Masson (2009)

Number of preparation	Main ingredient
500	Cow manure
500P	Preparation 500 with 502 to 507
501	Silica
502	Yarrow flowers (*Achillea millefolium*)
503	Chamomile flowers (*Matricaria recutia*)
504	Stinging nettle shoots (*Urtica dioica*)
505	Oak bark (*Quercus robur*)
506	Dandelion flowers (*Taraxacum Officinale*)
507	Valerian extract (*Valeriana officinalis*)
Compost	Cow manure with preparation 502 to 507

Cultivation sites that are certified as being biodynamic need to have been previously certified as organic agriculture production sites (European Commission, 2007; European Commission, 2008; EUROSTAT, 2012; Demeter International eV, 2012; CRAEGA, 2012) and have to go through a three year conversion period. Currently, there is on-going debate regarding the positive effects of applying biodynamic farming practices to different crops (Turinek et al., 2009), especially regarding the influence and appropriateness of using biodynamic preparations. Some studies have demonstrated substantial benefits of using biodynamic preparations in terms of soil structure and microorganisms, improving soil fertility or microbial biodiversity (Reganold et al., 1993; Mäder et al., 2002; Probst et al., 2008; Reeve et al., 2010), whereas other studies have highlighted the lame benefits that biodynamic agriculture can render under certain conditions (Carpenter-Boggs et al., 2000; Tassoni et al., 2013). There are several theories regarding the way in which the preparations may interact with the crops, including hormonal stimulation and enhanced crop growth, especially at root level (Stearn, 1976; Goldstein and Koepf, 1992; Deffune and Scofield, 1995; Fritz and Köpke, 2005). Other studies, however, suggest that biodynamic preparations act as regulators of bacterial activity (Miller and Bassler, 2001).

6.2 Materials and methods

6.2.1 Goal and scope

Despite the strong increase in wine LCA studies in past years, a recent review by Rugani et al. (2013) points out a series of gaps that remain unexplored when life cycle thinking is applied to the wine sector. One of these gaps is directly connected to an in-depth analysis of the different viticulture techniques that may be used (i.e. organic, biodynamic, conventional…), since some authors have suggested that organic practices may not be linked with lower environmental impacts (Venkat, 2012). Therefore, the main goal of this study is to perform a life-cycle environmental assessment for three different viticulture techniques within the appellation of "Ribeiro" (NW Spain): a biodynamic cultivation site (BD), a conventional grape production site (CV) and an intermediate biodynamic-conventional wine-growing plantation (BD-CV)[2].

[2] The intermediate biodynamic-conventional site considers biodynamic protocols for viticulture activities, but does not consider crop diversity or livestock farming. In addition, it has no organic or biodynamic certifications

Moreover, this novel approach is combined with three methodological improvements that are developed in this case study. In the first place, a comparison between land use impact categories is provided. Secondly, a repeatedly underrepresented activity in environmental management is the role of human labor in environmental impacts (Rugani et al., 2013). Finally, a third issue, in line with the work developed in Chapter 4, is the use of updated assessment methods for results computation, as described in Section 6.2.6.

The selected functional unit (FU), which is the reference unit to which the results are referred to (ISO, 2006), was 1.1 kg of collected grape, which was the amount of grape necessary to produce one bottle of Ribeiro wine (i.e. 750 mL of wine) in two consecutive harvest years: 2010 and 2011. Furthermore, the FU is in accordance with Chapters 4 and 5 of this dissertation and prior wine LCA and carbon footprint studies available in literature (Petti et al., 2010; Hayashi, 2013; Pattara et al., 2012; Rugani et al., 2013).

6.2.2 System boundaries

System boundaries in this study were limited to the gate of the winery (i.e. agricultural phase of wine production) in order to provide a direct comparison between the different viticulture techniques, disregarding post-agricultural stages of the life-cycle. More specifically, while substantial differences can be observed in post-agricultural stages of winemaking, these are not attributable to the implementation of different viticulture techniques. All the processes and field operations needed for grape production, including the production and use of the major inputs, such as fossil fuels, pesticides, water and the trellis of the vineyard (Figure 6.1) were included within the system boundaries. Excluded processes include the vine nursery stage due to the lack of data. Moreover, it should be highlighted that the number of vines that are replaced on an annual basis is very low, which minimizes the effect of this exclusion (Bosco et al., 2011). The greater part of the substances used in the biodynamic preparations were also excluded from the inventory, since most of these corresponded to the collection of minimal amounts of wild plants in the neighboring areas of the cultivation sites, such as nettles (*Urtica dioica*), horsetails (*Equisetum* spp.) or chamomile (*Chamaemelum nobile*). These plants are applied to the preparations in homoeopathic doses; therefore, the assumed impact of these inputs would be close to zero. In contrast, other substances used in biodynamic preparations, such as powdered quartz, salt or soy, were included within the system boundaries. Finally, concerning fertilizer use, the production of compost was excluded due to the fact that it was assumed to be a residue from ovine farming. Hence, only its transport and spreading on the vineyards was considered (Martínez-Blanco et al., 2007).

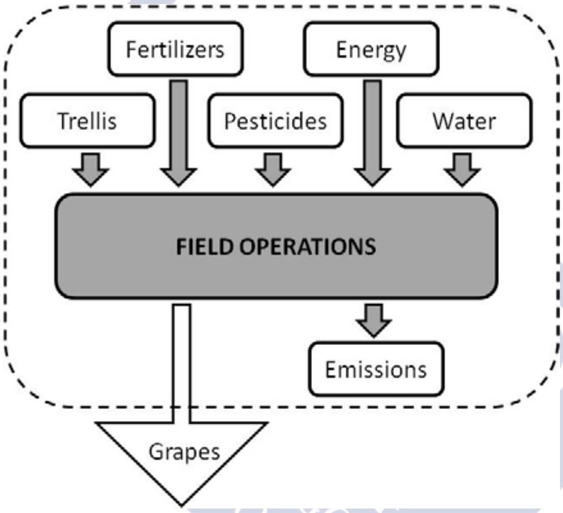

Figure 6.1. Graphical representation of the assessed production system

6.2.3 Data acquisition

Primary data were retrieved through a set of questionnaires that were distributed between the wine-growers of the exploitations inventoried in the study. The cultivation sites were located in Leiro and San Amaro, therefore, belonging to the “Ribeiro” appellation — (Table 6.2; Figure 6.2). These surveys embraced a wide range of inputs for the cultivation sites, such as fuel use, pesticides, field operations, machinery or trellis. Moreover, specific labor data, regarding working hours of employees, were included in order to include human labor activities in life-cycle thinking —see Section 6.4.2 (Rugani et al., 2012).

Direct emissions from field operations, such as those derived from fossil fuel consumption by agricultural machinery, were estimated based on the characterization factors proposed by EMEP-Corinair Emissions Handbook 2006 (EMEP-Corinair, 2006). Nitrogen emissions linked to fertilizer spreading on the vineyards were calculated following the methodology proposed by Brentrup et al. (2000). Nevertheless, as mentioned in Section 6.2.2, only on-field emissions were considered for compost, since the compost processing stage was excluded from the system boundaries. Finally, phosphorus and phosphate emissions associated with fertilizers spreading were obtained from the bibliography (Cowell, 1998 and Cowell and Clift, 1997).

Table 6.2. Selected sample for the assessed period 2010-2011

	Surface area (ha)	**Plots**	**Grape production (tons/year)**		**Annual yield (tons/ha)**	
			2010	***2011***	***2010***	***2011***
Biodynamic holding (BD)	4	1	15	15	3.75	3.75
Biodynamic-conventional holding (BD-CV)	27.6	42	124	162	4.49	5.87
Conventional holding (CV)	14	7	120	152	8.57	10.86

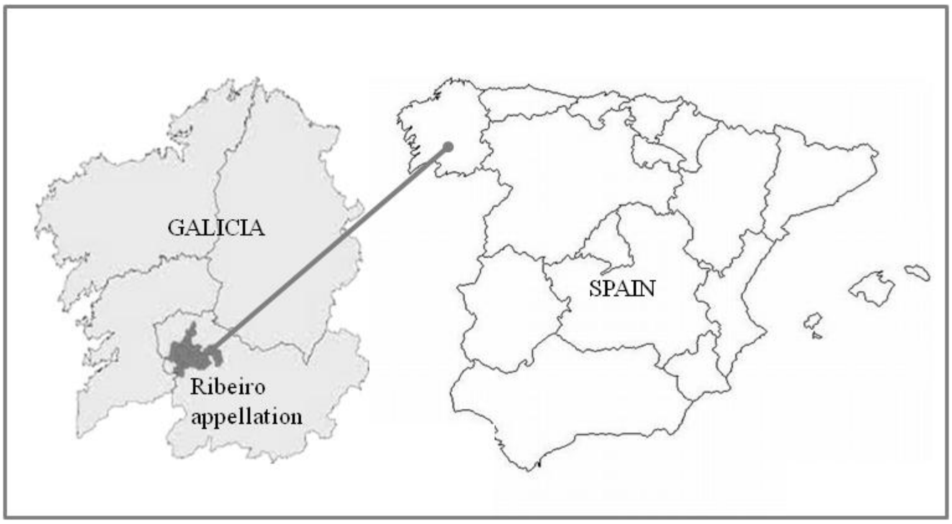

Figure 6.2. Geographical location of the Ribeiro appellation in Galicia and Spain

The emissions derived from pesticide use are estimated by using the PestLCI dispersion method (Birkved and Hauschild, 2006), adapted to the specific climatic and soil conditions of the analyzed area. On the one hand, meteorological data were retrieved form a weather station at Amiudal (Avión municipality), approximately 20 km away from the studied sites. Furthermore, the water balance for the assessed years was computed based on the Thornthwaite methodology. On the other hand, soil characteristics were taken from a sample point in the municipality of Cortegada (Trueba et al., 1998). Despite its relative distance from the inventoried sites (*circa* 25 km), the consulted experts assured that viticulture soils in the "Ribeiro" appellation are very similar in terms of soil texture, pH and organic matter content to the ones included in the inventory.

Finally, in order to include the active substances that are not available in the dispersion model, their characteristics were obtained from a pesticide database of the University of Hertfordshire (PPDB, 2012) and from the software available from the US Environmental Protection Agency, named Estimation Program Interface (EPI) Suite v.4.10 (USEPA, 2011). Emissions to the soil were not considered since soil is considered a sink for this type of emissions (Birkved and Hauschild, 2006).

Emissions related to the use of sulfur and copper-based pesticides were not taken into consideration for two main reasons. On the one hand, their emissions cannot be computed with the current PestLCI methodology. On the other hand, the retention rate of the soil for these pesticides is very high (Fernández-Calviño et al., 2008), limiting their importance in terms of air and water emissions. Additionally, the complex reactions and interactions occurring between Cu and the soil hinder the capacity to establish the endpoint of this substance (Kiaune and Singhasemanon, 2011). Finally, electricity inclusion in the study was integrated by adapting the electricity grid available for Spain in the ecoinvent® database, with data from the two years under analysis, based on official statistics (Frischknecht et al., 2007; REE, 2011; REE, 2012).

Secondary data referred to the production of plant protection products, trellis or diesel were obtained from the ecoinvent® database (Frischknecht et al., 2007). The synthetic pesticides used in conventional viticulture were grouped in compound families (e.g. thiocarbamates, triazines…); while for the copper- and sulfur-based pesticides specific processes were used for disaggregating these substances from an inventory perspective. For instance, sulfur was assumed to be obtained as a sub-product in the oil refining industry, and copper from the extractive mining sector (Manuel Montaño, Afepasa SA, April 2013, personal communication). For the biodynamic treatments and due to the lack of inventory data for certain products, only quartz, soy and salt were included in the inventory.

Data regarding the trellis of the vineyards were retrieved by contacting a specialized company that delivers this type of materials to the inventoried wineries —e.g. iron, steel or wood (Pedro Mosteiro, Viniequip SL, April 2013, personal communication). Consequently, detailed data regarding the specific types of materials used in each cultivation site as well as their transportation, were available. Whenever the vineyards showed a wooden trellis, the specific inventory process from ecoinvent® was the wood with the most similar density to the one used in this area (*Acacia dealbata*). This species is selected for vine support in this area for two main reasons: on the one hand, it constitutes an invasive species in the "Ribeiro" region; therefore, its availability is guaranteed and the need to control its expansion is augmenting; on the other hand, its resistance allows its use without any type of pre-treatment (e.g. copper salts, arsenic or chromium), reducing its toxicity potential (Point et al., 2012). Steel and iron trellis background processes were extracted and modified from ecoinvent®, in order to include the wire characteristics of the product as well as galvanization process for iron. The non-woody materials used for the trellis were allocated to a total of 50 years of usage on the vineyards, in order to compute the proportional environmental impact per harvest year.

6.2.4 Life cycle inventory (LCI)

6.2.4.1 Biodynamic farm (BD)

This cultivation site is completely adapted to biodynamic viticulture, including the certification process as an organic agricultural producer. The specific certificate for biodynamic production was under revision when this study was developed. Furthermore, it is important to note that the yield of the vineyards is relatively constant, due to the strict controls regarding the productivity of each vine (Table 6.2). In fact, the clarification of the grape clusters is conducted to ensure grape quality and their correct ripening.

Table 6.3 shows the LCI for grape production in this wine farm for the 2010 and 2011 harvest years. Most inputs remained constant when the two harvest years are compared, due to the strict quality controls. However, copper-based pesticides and diesel showed important variations between the two years, since year 2011 was characterized by low proliferation of fungi, such as downy mildew (*Plasmopara viticola*) or powdery mildew (*Uncinola necator*). This led to a lower dose of copper applied in the fields. Moreover, in this particular wine farm it is important to note the absence of fertilizer spreading on the vineyards. Despite the fact that organic fertilizers may be used in this type of grape cultivation, the soil analyses performed concluded that no fertilizer spreading was necessary in these two years of assessment. In fact, one of the reasons behind this may be the fact that the previous land use was pasture land for livestock. Furthermore, as part of the holistic biodynamic approach, currently sheep and poultry pasture is on-going within the vineyards (Petherick, 2010), enhancing not only fertilization activities, but also helping in terms of weed control, reducing the number of interventions and, therefore, the use of machinery and labor.

6.2.4.2 Biodynamic-conventional farm (BD-CV)

This winery does not present the entire range of biodynamic cultivation elements, since it does not consider crop diversity or livestock farming, but it follows biodynamic protocols for vineyard activities. Additionally, it lacks the certificates of organic agriculture or biodynamic farming. Therefore, it can be stated that this farm combines conventional and biodynamic operations in a hybrid manner. For instance, synthetic pesticides are not applied on the vineyards, using exclusively biodynamic preparations, copper- and sulfur-based pesticides and quartz powder.

As shown in Table 6.3, there is a considerable interannual variation in harvest yields, due to the high productivity in 2011. Furthermore, diesel and pesticide use variation is notable between the two harvest years, due to the good climatic conditions for the non-proliferation of vine pathogens in 2011, which reduced the amount of field operations. Finally, this winery did not include within its operations the spreading of fertilizers, since the performed soil analysis disregarded the need to do so.

6.2.4.3 Conventional viticulture

This winery presents conventional patterns of grape production, with the use of fertilizing agents and synthetic pesticides. These fertilizers, despite being organic, do not originate within the farm (Table 6.3). In fact, the inventory shows the amounts of synthetic pesticides that are used in field operations, as well as the use of herbicides, such as glyphosate and terbuthylazine. Additionally, inventory data regarding pesticides emissions for this winery is available on Appendix (Table B.1).

Table 6.3. Inventory data for the three viticulture sites for the period 2010-2011 (Data per FU: 1.1 kg of grapes)

Inputs from the technosphere							
		BD site		**BD-CV site**		**CV site**	
	Units	**2010**	**2011**	**2010**	**2011**	**2010**	**2011**
Energy							
Diesel	g	16.58	10.16	29.39	16.98	73.33	55.00
Electricity	kWh	–	–	0.33	0.25	$4.60 \cdot 10^{-2}$	$3.70 \cdot 10^{-2}$
Fertilizers							
Sheep manure	g	–	–	–	–	275.0	217.1
Transport (compost)	tkm	–	–	–	–	$2.80 \cdot 10^{-2}$	$2.20 \cdot 10^{-2}$
Pesticides							
Copper-based compounds	g	0.20	0.15	0.58	0.22	1.16	0.76
Soybean	g	0.88	0.88	–	–	–	–
Silica dust	g	$5.9 \cdot 10^{-3}$	$5.9 \cdot 10^{-3}$	14.69	5.62	–	–
Sulfur	g	–	–	14.40	3.67	1.36	1.07
Sodium chloride	g	–	–	0.00	0.07	–	–
Thiocarbamates	mg	–	–	–	–	57.37	36.25
Acetamide-anillide	mg	–	–	–	–	9.24	7.29
Dithiocarbamate	mg	–	–	–	–	385.4	228.2
Nitriles	mg	–	–	–	–	60.16	47.49
Cyclic-N-compounds	mg	–	–	–	–	15.26	12.04
Fosetyl-Al	mg	–	–	–	–	644.3	410.0
Gliphosate	mg	–	–	–	–	316.0	249.5
Phtalamide-compounds	mg	–	–	–	–	149.2	117.8
Triazine	mg	–	–	–	–	305.2	241.1
Trellis							
Iron (wire)	g	5.66	5.66	4.16	3.18	–	–
Steel (cables and tubes)	g	–	–	–	–	9.71	7.67
Wood	g	614.1	614.1	42.70	32.69	–	–
Water (tap)	g	586.7	440.0	979.4	524.7	1110	720.0
Machinery							
Field sprayer user	m^2	2.93	2.93	2.45	1.87	1.28	1.01
Fertilizing, by broadcaster	m^2	–	–	–	–	0.18	0.14

Table 6.3. Inventory data for the three viticulture sites for the period 2010-2011 (Data per FU: 1.1 kg of grapes) (continuation)

Inputs from the technosphere							
		BD site		**BD-CV site**		**CV site**	
	Units	**2010**	**2011**	**2010**	**2011**	**2010**	**2011**
Tillage, rotary cultivator	m^2	–	–	–	–	0.28	0.22
Rotary mower	m^2	2.93	2.93	2.45	1.87	0.18	0.14
Hoeing	m^2	–	–	2.45	1.87	0.18	0.14
Inputs from the environment							
Energy, gross calorific value, in biomass	MJ	20.35	20.35	20.35	20.35	20.35	20.35
Transformation, from pasture and meadow	m^2	2.93	2.93	2.45	1.87	1.28	1.01
Transformation, to arable, non-irrigated	m^2	2.93	2.93	2.45	1.87	1.28	1.01
Occupation, arable, non-irrigated	m^2	$2.7{\cdot}10^{-4}$	$2.7{\cdot}10^{-4}$	2.24	1.72	1.18	0.93
Outputs to the technosphere							
Products							
Grapes	kg	1.1	1.1	1.1	1.1	1.1	1.1
Outputs to the environment							
Emissions to the atmosphere							
CO_2 (diesel)	g	52.02	31.88	92.21	53.26	230.1	172.6
SO_2 (diesel)	mg	33.16	20.32	58.77	33.95	146.7	110.0
VOC (diesel)	mg	120.5	73.87	213.7	123.4	533.1	399.9
NO_x (diesel)	mg	834.0	511.1	1478.3	853.9	3.69	2.77
CO (diesel)	mg	265.3	162.6	470.2	271.6	1.17	0.88
NH_3 (diesel)	mg	0.12	0.07	0.21	0.12	0.51	0.39
CH_4 (diesel)	mg	2.82	1.73	5.00	2.89	12.47	9.35
N_2O (diesel)	mg	21.39	13.11	37.91	21.90	94.60	70.95
N_2O (fertilizers)	mg	–	–	–	–	53.99	42.63
Emissions to water							
NO^{3-}	g	–	–	–	–	14.06	10.28
PO_4^{3-}	mg	–	–	–	–	56.19	44.36

When Table 6.3 is examined based on a cross-site approach, the use of diesel appears to be up to 4 times lower in the BD and BD-CV when compared to the CV exploitation. This is due to the high degree of mechanization of CV wine-growing as compared to BD or BD-CV, where artisanal methods are implemented. In terms of the materials used for vine support, the CV site only uses stainless steel and iron, while BD and BD-CV use a mix of abiotic and biotic materials.

A final issue that shows strong variability between the three agricultural techniques is the use of plant protection agents. On the one hand, copper-based pesticides were used in much higher quantities in the CV site with respect to the other two techniques. In fact, copper application in the BD-CV and BD sites was below the maximum standards recommended for organic wine by the EU (European Commission, 2008). On the other hand, for sulfur-based products the highest use was found in the BD-CV site. In fact, the BD winery did not use any sulfur-based pesticides, using only powdered quartz in homoeopathic doses for the same purposes —increase the resistance of the plant to pathogens (Fauteux et al., 2005)— (biodynamic preparation 501 — Table 6.1), while the BD-CV winery made a mixed application of powdered quartz and sulfur, in order to reduce the amounts of sulfur needed

6.2.5 Allocation and other assumptions

Allocation is a critical issue in LCA studies. However, in this case it was not necessary to apply allocation to the outputs, since there is one sole product exiting the system: grapes for vinification. The existence of co-products in the vinification phase (e.g. skins), does not fall within the system boundaries of the production system analyzed. Allocation of the environmental impact of fertilizers has shown to be a controversial issue in agricultural systems (Luo et al., 2009). In this specific study, it was decided to include only those impacts linked to fertilizers that are directly connected to viticulture practices (i.e. transport of the compost and associated emissions in the vineyards), as described in Section 6.2.2. Consequently, this cut-off approach allowed disregarding previous upstream impacts of the sheep manure in the CV site, assuming that this item is a residue of a separate production system.

6.2.6 Impact category selection

The life cycle impact assessment (LCIA) stage was performed using the CML baseline 2000 methodology (Guinée et al., 2001). The selected impact categories from the CML methodology were abiotic depletion (ADP), acidification (AP), eutrophication (EP), global warming (GWP), ozone layer depletion (ODP) and photochemical oxidant formation (POFP). This selection was based on commonly used impact categories in previous wine LCA studies (Petti et al., 2010), as well as on a flexible interpretation of the recommendations from the ILCD handbook for impact assessment in Europe (ILCD, 2011). Furthermore, toxicity (Etox) was analyzed following the USEtox method proposed by Rosenbaum et al. (2008). The selection of this assessment method for toxicity categories is also linked to the ILCD recommendations due to a higher coverage of chemical substances (Rosenbaum et al., 2008), the evaluation of model uncertainty and the extensive review process by model developers (ILCD, 2011). Finally, regarding land use, the land competition (LC) impact category from the CML 2001 method was selected to obtain a quantitative assessment, while the Soil Organic Matter (SOM) model developed by Milà i Canals et al. (2007), considered as a soil quality indicator, was used for a qualitative assessment following the ILCD recommendations (ILCD, 2011). Finally, regarding labor computation in wine-LCA, human labor (HL) input–output datasets were used (Rugani et al., 2012). The latter methodological issues, while not being the core objective of the study, are presented in Section 6.4.2 using the ReCiPe assessment method (Goedkoop et al., 2009). The software used to compute the results was SimaPro 7.3 (PRè-Product Ecology Consultants, 2011).

6.3 Results

6.3.1 Biodynamic farm (BD)

The environmental impacts obtained per FU for the BD farm are lower for the 2011 harvest year as compared to 2010, with environmental gains ranging from 3% for Etox to 32% for AP (Table 6.4). The production and consumption of diesel for field operations constitutes the main carrier of environmental impacts, with relative contributions ranging from 49% (EP) to 78% (AP) for the CML categories.

Table 6.4. Characterization results per year for the assessed viticulture sites (Data per FU: 1.1 kg of grapes)

		BD		BD-CV		CV	
Impact category	**Units**	**2010**	**2011**	**2010**	**2011**	**2010**	**2011**
ADP	g Sb eq	0.62	0.47	0.92	0.55	2.17	1.64
AP	g SO_2 eq	0.88	0.60	2.00	0.98	5.04	3.82
EP	g PO_4^{3-} eq	0.23	0.17	0.35	0.19	2.29	1.68
GWP	g CO_2 eq	97,17	71.11	147.60	87.32	375.31	283.42
ODP	g CFC-11 eq	$9.89\cdot10^{-6}$	$6.94\cdot10^{-6}$	$1.60\cdot10^{-5}$	$9.33\cdot10^{-6}$	$5.82\cdot10^{-5}$	$4.45\cdot10^{-5}$
POFP	g C_2H_4 eq	$3.72\cdot10^{-2}$	$2.89\cdot10^{-2}$	$7.30\cdot10^{-2}$	$3.66\cdot10^{-2}$	0.18	0.13
Etox	CTUe	$3.51\cdot10^{-1}$	$3.40\cdot10^{-1}$	$3.29\cdot10^{-1}$	$2.17\cdot10^{-1}$	$3.62\cdot10^{1}$	$1.73\cdot10^{1}$
LC	m^2a	2.45	2.45	2.04	1.56	1.18	0.93

BD= biodynamic viticulture; BD-CV= biodynamic-conventional viticulture; CV= conventional viticulture; ADP= abiotic depletion potential; AP= acidification potential; EP= eutrophication potential; GWP= global warming potential; ODP= ozone layer depletion potential; POFP= photochemical oxidant formation potential; Etox= eco-toxicity; LC= land competition.

The trellis of the vineyard was identified as the second most important source of environmental impacts in most categories. The contribution of the trellis ranged from 13% (AP) to 35% for POFP. Pesticide production represented 19% of the environmental impact for EP and 4% for AP. Finally, the sum of the remaining inputs, including machinery or water use, represented from 5% (AP) to 15% (ADP).

Regarding the ecotoxicity impact category, vine support materials were the main source of environmental impact (77%), followed by other inputs (machinery, electricity…) with 12% and pesticide production (9%).

6.3.2 Biodynamic-conventional farm (BD-CV)

The environmental profile obtained for grape production in the BD-CV farm shows, similarly to the BD farm, higher impacts for the 2010 harvest year (Table 6.4). In fact, the decrease in environmental burdens in 2011 is substantial, ranging from 40% for ADP to 51% for AP. Diesel production and combustion represented on average 71% of the total environmental impact, ranging from 55% (POFP) to 84% (ODP).

Production of plant protection products (i.e. pesticides) was the second source of environmental burdens in three impact categories: EP (20%), AP (24%) and POFP (26%). Vine support materials (wood and iron) represented 13% of the impacts for ADP and POFP, and 10% for GWP. Finally, other inputs, such as machinery, water use or electricity, summed, at the most, 8% of the total environmental profile (ADP).

Concerning eco-toxicity, the Etox environmental burdens were dominated by the vine support materials (64%), followed by the production of pesticides (20%) and other inputs (12%). Diesel production only accounted for 4% of the environmental impact.

6.3.3 Conventional farm (CV)

The overall environmental impact of grape production in the conventional winery was lower in the year 2011, in a similar way as to the decrease observed for BD and BD-CV (Table 6.4). More specifically, the reductions in the environmental profile ranged from 24% (ODP) to 52% (Etox). The main hot spot in conventional grape production was the production and consumption of diesel, ranging from 24% (EP) to 80% (ADP). On average, the relative contribution of diesel was 59%. The trellis of the vineyards (mainly stainless steel) represented relevant contributions to most impact categories, such as POFP (38%), AP (30%), ADP (11%) and GWP (10%). Fertilization and associated on-field emissions constituted the main environmental burden in terms of EP (64%). The production of pesticides represented the main impact in terms of ODP (40%), and it was also significant for EP (11%) and POFP (9%). Other inputs, such as electricity use, machinery or water consumption presented minor contributions to the global environmental profile of conventional grape production.

Finally, for the Etox impact category, the use of synthetic pesticides, such as folpet or terbuthylazine represented over 99% of the total environmental burdens (Figure 6.3). The remaining active substances used as plant protection agents presented a very low environmental impact despite the fact that mancozeb, fosetyl-Al or glyphosate are emitted in similar quantities to folpet or terbuthylazine. This is due to the lower characterization factors of the latter in terms of eco-toxicity. Finally, the environmental change from one harvest year to another was associated with a decrease in emissions to water in 2011 and to the higher characterization factors for water emissions in this specific impact category.

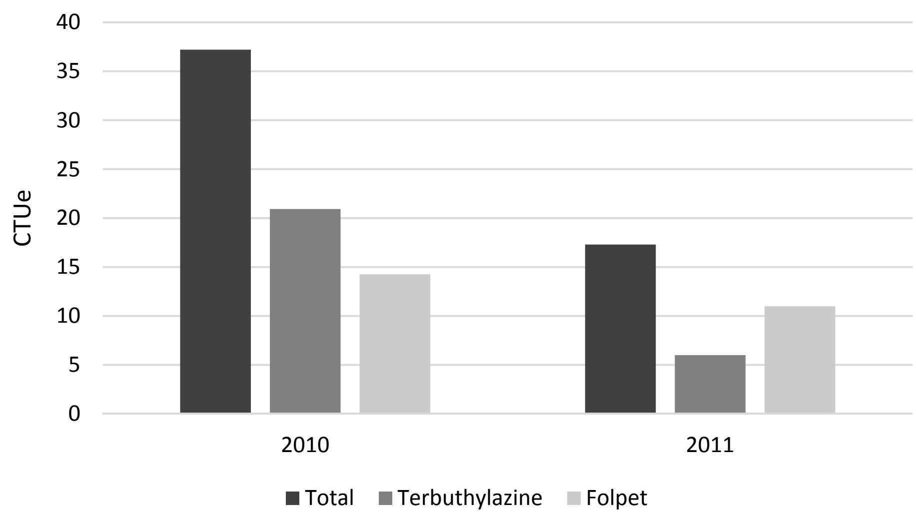

Figure 6.3. Characterization results for the main sources of environmental impact in terms of eco-toxicity (Etox) in the conventional viticulture site (Data per FU: 1.1 kg of grapes)

6.4 Discussion

6.4.1 Identification of the main hot spots

Diesel production and consumption used in field operations demonstrated to be the main source of environmental impacts in the three different agricultural management techniques for all impact categories, except for EP and Etox. In the latter categories, the importance of this activity depended on the management technique. For instance, at the CV site the main source of environmental impact in terms of EP was the on-field emissions associated to fertilization. In the case of Etox, vine support was the main carrier for BD and BD-CV, while the CV site profile was dominated by pesticide emissions.

When the current study is compared to other published studies, similar hot spots are observed: diesel and fertilizers (Aranda et al., 2005; Gazulla et al., 2010; Point et al., 2012). Nonetheless, the absolute environmental burdens per FU show that there is a substantial decrease in the environmental profile of wine produced with biodynamic techniques, in accordance with other studies analyzing other crop production systems (Alaphilippe et al., 2012; Stavi and Lal, 2012; Venkat, 2012). For instance, most conventional wines analyzed in the literature in recent years showed environmental impacts at least 100% higher for most impact categories when compared to the biodynamic wine evaluated in this case study (Chapter 5; Bosco et al., 2011; Point et al., 2012; Vázquez-Rowe et al., 2013). For example, in terms of global warming, as shown in Table 6.5, greenhouse gas (GHG) emissions linked to the agricultural stage of wine production ranged from 220 g CO_2 eq./bottle to 803 g CO_2 eq., at least 126% higher than the GHG emissions for the BD site in 2010 and 209% higher than in 2011. However, these values appear to be in a similar range to the conventional site, including those reported for another Galician appellation —"Rías Baixas" (Benedetto, 2013; Bosco et al., 2011). Finally, overall environmental impacts for the BD-CV site were considerably higher than for BD wine in both years of assessment, but in a similar range to organic wineries evaluated in the literature (Carta, 2009; Rugani et al., 2009; Rugani et al., 2013).

Table 6.5. Global warming potential (GWP) for the analyzed viticulture sites as compared to previous publications (results reported in g CO_2 eq./bottle)

	Publication	**GWP (g CO_2 eq)**	**Ratio Wine/BD-2010**	**Ratio Wine/BD-2011**
BD-2010	Current study	97.2	1.00	1.37
BD-2011	Current study	71.1	0.73	1.00
BD-CV-2010	Current study	148	1.52	2.08
BD-CV-2011	Current study	87.3	0.90	1.23
CV-2010	Current study	375	3.86	5.27
CV-2011	Current study	283	2.91	3.98
Vermentino (2009)	Vázquez-Rowe et al. (2013)	241	2.48	3.39
Nova Scotia (2006)	Point et al.(2012)	803	8.26	11.3
"Rías Baixas" (2010)	Chapter 5 of this dissertation	463	4.76	6.51
Monteregio di Massa Maritima (2009)	Bosco et al. (2011)	330	3.40	4.64
Morellino di Scansano (2009)	Bosco et al. (2011)	220	2.26	3.09

BD= biodynamic viticulture; BD-CV= biodynamic-conventional viticulture; CV= conventional; GWP= global warming potential.
NOTE: Not all wines have the same conversion rate of grapes to wine (kg/mL); therefore, the selected comparison basis was the amount of grapes needed to produce a bottle of wine

Finally, it is important to remark the relevant variations in environmental burdens identified when different harvest years are compared, in accordance with the analysis presented in Chapter 4 and other food and beverage products (Ramos et al., 2011), as can be seen in Table 6.4. A common reason for this decrease throughout the three viticulture sites is related to the lower consumption of diesel in 2011, which is linked to favorable climatic conditions that minimized the development of powdery and downy mildew and, therefore, produced a decline in the number of machinery interventions in the vineyards. Nevertheless, these decreases were more noticeable in the CV site due to the higher dependence on phytosanitary treatments. For instance, a very dry season favored lower inputs of folpet, as well as lower emissions associated with the application of terbuthylazine (73% lower impacts than in 2010 for this pesticide). Moreover, the decrease of diesel impacts is also influenced by a reduction of certain agricultural activities, such as mowing and in the case of the CV and BD-CV sites, to a higher harvest yield in 2011 (Table 6.2).

6.4.2 Comparing LCA results between viticulture techniques

Whenever the environmental profiles of grape production are compared for the three different production sites (Table 6.4), results suggest that biodynamic production implies the lowest environmental burdens, followed by the BD-CV site. The highest environmental impacts, therefore, were linked to conventional agricultural practices. The main reasons for this strong decrease in environmental impacts when BD and CV are compared is related to an 80% decrease of diesel inputs, which have to be matched with a lower application of plant protection products and fertilizers, and the introduction of manual work rather than mechanized activities on the vineyards.

Therefore, the environmental benefits of producing grape for vinification under biodynamic agricultural practices entail reductions ranging from 71% (ADP) to 99% (Etox) for harvest year 2010. Nevertheless, despite the considerable differences in global impacts between the production years, the relative environmental gains are similar for the two years of operation. In contrast, if the BD-CV site is compared to the CV winery, the decrease in environmental impacts is slightly lower, ranging from 58% for ADP and POFP to 99% for Etox in 2010. For the 2011 harvest year the reduction in environmental impacts is considerably higher, ranging from 67% (ADP) to 99% (Etox).

Whenever the LC impact category is compared between the different viticulture sites, the results show a completely different pattern. Due to the higher harvest yields for the CV site, this winegrowing holding needs the lowest amount of available agricultural surface per FU. BD, on the contrary, shows the worst environmental profile in terms of land use. These results must be interpreted with caution, since vineyards that have suffered conversions from conventional practices to biodynamic or organic activities, present very low harvest yields during the conversion period (Hokazono and Hayashi, 2012). Nevertheless, in the case studies that have been provided in this study, the BD site did not have a conversion period, since the vines were directly planted for biodynamic purposes. In the BD-CV case study, despite the existence of a conversion period, it did not affect the harvest years analyzed in this study. In fact, if the results in this study are matched to previous publications in which a certain crop or livestock activity is compared depending on the agricultural practices, it can be observed that in all cases the shift to biodynamic or organic production involves an important reduction in environmental burdens due to a reduction in operational inputs, but at the same time there is an increase in land occupation (Cederberg and Mattson, 2000; Pelletier et al., 2008; Meisterling et al. 2009; Boggia et al., 2010), as well as in human labor in some cases (Niccolucci et al., 2008).

Land occupation in LCA studies has traditionally been evaluated in a quantitative manner (Milà i Canals et al., 2007), without taking into consideration the degradation of the quality of the land that may occur during the occupation for human activities (Garrigues et al., 2012). Therefore, the SOM impact category, which is an easily defined impact category to compute land use and transformation burdens which influence life support functions (Milà i Canals et al., 2007), has been modelled for the CV and BD viticulture sites assessed in this study.

In order to conduct the SOC indicator, data available from a soil sample in Cortegada were retrieved (Trueba et al., 1998), as shown in Table 6.6. Additionally, in order to monitor the balance of SOC through time, and based on a previous study developed by Poeplau et al. (2011), the change rates for SOC were calculated for the two sites. In the first place, for the BD site the change rates in SOC were modelled in order to monitor the impact from the transition from grassland (prior to 2008) to the current biodynamic viticulture plantation. Secondly, in the case of the CV site, changes in SOC were modelled with respect to the transition from woody areas to viticulture that developed in 1988.

Table 6.6. Soil characteristics of the inventoried soil sample. Source: Trueba et al. (1998)

Parameters	**Horizon A**	**Horizon AB**	**Horizon C**
Upper depth (cm)	0.0	20.0	30.0
Lower depth (cm)	20.0	10.0	--
pH	4.4	4.9	5.0
Organic content (%)	7.3	1.2	0.7
Sand content (%)	73.5	60.0	69.5
Clay content (%)	17.8	27.7	21.5
Silt content (%)	8.7	12.3	9.0
Density (g/cm^3)	1.2	1.4	1.5

One single soil sample available for viticulture land was used for modelling the three different sites. On the one hand, the BD site was modelled based on the fact that it underwent a short transition period from pasture land in 2008. On the other hand, the CV site was initially a woody area when it was created in 1988. Changes in organic content of the vineyards were modelled based on the data available in Poeplau et al. (2011) concerning temporal dynamics of soils due to land use changes.

The results prove, in a similar way to the results shown in the LC impact category presented in the Results section, that the transition to biodynamic and other organic types of viticulture may imply an increased requirement for land, due to the lower harvest yields of these cultivation sites (Table 6.7). However, despite the strong difference shown in Table 6.7 regarding the SOM results, further field sampling should be done on site, in order to detect how the specific cultivation practices of conventional vs. biodynamic viticulture affect carbon retention of the soil.

Table 6.7. Indicator results for the soil organic carbon (SOC) impact category model in the conventional (CV) and biodynamic (BD) viticulture sites evaluated (Data per FU: 1.1 kg of grapes)

Cultivation site	**SOC per ha**	**SOC per ha**
	kg CO_2	g CO_2
Conventional (CV)	38.67	4.95
Biodynamic (BD)	53.12	15.56

Finally, it should be noted that the regular use of SOM in wine LCA studies would allow identifying not only the qualitative aspects of land occupation of the viticulture stage, but would also facilitate the inclusion of C dynamics when reporting the CF of viticulture products (Bosco et al., 2013). Moreover, the local characteristics of SOM (ILCD, 2011) allow unveiling the site-specific effects on land use of changing cultivation patterns. In fact, given the limited land availability in most European countries, including NW Spain, it is expected that land quality issues will become an important issue when applying consequential LCA approaches to viticulture (Rugani et al., 2013).

Human labor (HL) has been repeatedly disregarded from the system boundaries in previous wine LCA studies. The rationale behind this decision is that HL is not directly affected by changes in the FU. However, a recent study by Rugani et al. (2012) suggests its inclusion through a hybrid input–output LCI mechanism in order to provide a less anthropogenic perspective on how to deal with ecosystem services. The methodology in Rugani et al. (2012) is based on household expenditures and different levels of work skills, which eventually lead to variable human consumption behaviors. The method was adapted to HL in Spain, and thereafter computed for the specific characteristics of the CV and BD wineries. Finally, it is important to note that the assessment method used for this calculation was ReCiPe —midpoint H (Goedkoop et al., 2009), in order to maintain the same methodological criteria as Rugani et al. (2012).

The results, which can be observed in Figure 6.4 and in Tables B.2 and B.3 of Appendix, show that HL represents a higher proportion of the environmental impact for the BD winery, representing up to 54% of the environmental impact in the case of terrestrial eco-toxicity —TET (only 15% in the case of the CV winery) and 44% for marine eutrophication (ME). Regarding a commonly used impact category, such as climate change (CC), the relative environmental impacts of HL for the BD site represented 12% of the total impact, whereas in the case of the CV winery was only 1.5%. The discrepancies between the two viticulture sites are mainly linked to the higher labor input per unit of produced output, and to the lower overall environmental impact of biodynamic grape production.

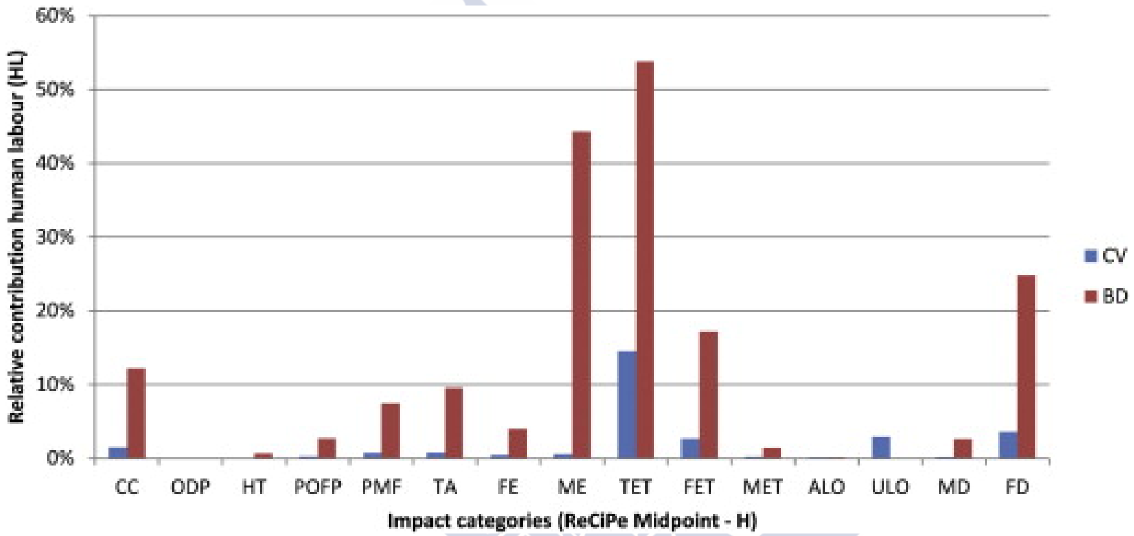

Figure 6.4. Relative contribution of human labor to the total environmental impact of conventional (CV) and biodynamic (BD) wine, using ReCiPe Midpoint (H). NOTE: CC= Climate Change; ODP= Ozone Depletion; HT= Human Toxicity; POFP= Photochemical Oxidant Formation; PMF= Particulate Matter Formation; TA= Terrestrial Acidification; FE= Freshwater Eutrophication; ME= Marine Eutrophication; TET= Terrestrial Eco-toxicity; FET= Freshwater Ecotoxicity; MET= Marine Eco-toxicity; ALO= Agricultural Land Occupation; ULO= Urban Land Occupation; MD= Mineral Depletion; FD= Fossil Depletion

Consequently, HL environmental impacts, despite their minor importance in most impact categories, show that they can be of crucial relevance in specific environmental dimensions. In fact, they support previous findings that suggested that the labor inputs in organic grape production are substantially higher (Loake, 2001; Guzmán and Alonso, 2008). Moreover, the expected increasing transition from conventional to organic/biodynamic viticulture in years to come will enhance the relative importance of labor activities (Fogarty, 2008).

6.4.3 Improvement actions

The main improvement actions to reduce the environmental impact should be focused on the CV site, since the impacts associated with this winery are substantially higher than for the other two sites. Moreover, the reduction in the environmental impacts between CV and BD or BD-CV is higher than the reduction attained through improvement actions proposed for conventional viticulture in previous publications (Point et al., 2012; Chapter 4 of this dissertation). Therefore, an improvement in the environmental profile of conventional viticulture can be accomplished, on the one hand, through the optimization of the main operational input in terms of environmental impact: diesel. This reduction can be performed through the reduction in the use of fertilizers with a correct management of nutrient balance and soil analysis. In fact, if plant protection products are managed taking into consideration meteorological conditions, anticipating pestilences, diesel inputs can be further reduced (Simon et al., 2011). Decision support systems, such as DOSAVIÑA, can be applied to calculate the optimal volume rate for spray applications in vineyards (Gil et al., 2011). In fact, these reductions in fertilizer and plant protection products would also imply an important reduction in environmental burdens related to the optimization of these products.

An alternative scenario would be the conversion of CV viticulture into BD-CV or BD. This approach would guarantee a strong decrease in most environmental impacts (except LC and HL). However, it is important to note that during the conversion period (at least 3 harvest years) the harvest yield is very low (e.g. in the analyzed appellation the yield during conversion period ranges from 1 to 2 t/ha), which involves very high environmental impacts per FU during this period. Hence, CV viticulture winegrowers who already have environmental monitoring and/or reporting schemes implemented, but want to shift to organic or biodynamic practices, may choose to perform a gradual conversion by plots in order not to damage the yield, and hence, the environmental profile of the entire winery.

It is important to remark that biodynamic farming implies a lower harvest yield (Table 6.2; Pfiffer et al., 1992). However, in terms of economic expenditure, the reduction of external operational inputs lowers production costs considerably (Scialabba et al., 2002), but they do not compensate for the increase in HL costs. Hence, current biodynamic wine prices are on average 25–30% higher than their conventional equivalents (Bernabéu, 2008). Nevertheless, it should be highlighted that a wide range of studies have alerted about energy scarcity that human populations will face in the following decades (Hall et al., 2003; Day et al., 2009). Consequently, it seems feasible that a growing number of wineries will shift to low-input viticulture techniques in an effort to avoid increasing oil prices (Wright, 2009). Moreover, recent publications have demonstrated that consumers are willing to spend more to acquire these types of wine (Bernabéu et al., 2007; ICEX, 2010). However, the increasing number of wineries that are shifting to biodynamic and other organic practices will eventually lead to a steady conversion between conventional and biodynamic wine prices (Greentrade market place, 2006).

6.5 Conclusions

As far as we were able to ascertain, the current study is the first one to analyze biodynamic viticulture from a life-cycle perspective, as well as its comparison with two other types of viticulture techniques: conventional viticulture and biodynamic-conventional viticulture. The obtained results do not only confirm prior findings that the environmental impact linked to a specific viticulture surface can have relevant variations on an interannual basis, but also demonstrate strong variability between viticulture practices. In fact, biodynamic viticulture, and to a lesser extent, an intermediate biodynamic-conventional winery, showed substantially lower environmental profiles for all the environmental impacts assessed, except for LC.

In decades to come, an increase in the scarcity of fossil fuels will affect many developed nations, including their agricultural practices and the price of food. Therefore, an alternative farming and food production and supply systems will be needed to face these important challenges. Despite the need to verify the results obtained in this case study for other climatic or geographical conditions, the shift to biodynamic viticulture seems an attractive alternative in terms of environmental sustainability and organoleptic characteristics of the wine. However, it remains unknown how a widespread shift in wine-growing activities towards biodynamic practices, which imply substantial yield decreases, would influence, on the one hand, land use changes in areas (i.e. appellations) that are strongly constrained by land availability, and, on the other, wine supply to meet the steadily increasing global demand after the economic downturn. Accordingly, from an LCA perspective, the implementation of consequential approaches in future studies would allow the use of life cycle management beyond the identification of environmental improvements in individual sites, triggering results that may be of use on a policy making level to guide land and environmental management at an appellation scale.

6.6 References

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Chapter 7

Accounting for time-dependent changes in GHG emissions in the "Ribeiro" appellation[1]

Summary

Land use changes (LUCs) constitute a crucial source of environmental impact in production systems, which are mostly associated with greenhouse gas (GHG) emissions. This circumstance is especially important for the agricultural sector, which accounts for an important ratio of the total GHG emissions occurring worldwide. Wine and grape production is a key sector in Spain, representing the largest surface area at European level. In the past decades, important wine related LUCs have been observed due to changes in farming methods/type, number of Denominations of Origin, and the expansion of large wineries with remarkable exports. The current study presents a temporally-based Life Cycle Assessment (LCA) study of the "Ribeiro" appellation, in which the gradual changes in the land use, as well as the technological improvements are analyzed in detail in order to understand how the environmental profile of this specific wine producing area has shifted in the past two decades (i.e. from 1990 to 2009). On the one hand, phenomena such as afforestation and agricultural intensification are analyzed throughout the appellation to estimate the impact due to GHG emissions linked to LUCs, based on IPCC standards. On the other hand, trends linked to technological improvements, operational changes, such as changes in the use and management of plant protection agents or fertilizers or the change in the energy sources for machinery on the vineyards, were assessed in detail.

[1] Villanueva-Rey, P., Vázquez-Rowe, I., Otero, M., Moreira, M.T., Feijoo, G., 2015. Accounting for time-dependent changes in GHG emissions in the Ribeiro appellation (NW Spain): Are land use changes an important driver? Environ. Sci. Policy 51, 215–227.

Index

7.1 Introduction

Accounting and reporting of environmental sustainability indicators has become a regular practice for numerous stakeholders in the wine sector (Rugani et al., 2013). In fact, increasing competitiveness has worked as a trigger for the communication of sustainable or ecological initiatives (Ginon et al., 2014). For instance, a growing number of wineries are evolving towards the production of organic and/or biodynamic wines to nourish an ever-growing green market for high and medium price wines (Gabzdylova et al, 2009). However, in many cases these initiatives have been criticised due to the qualitative rather than quantitative methods to assess the environmental sustainability of wine (Botonaki and Tsakiridou, 2004; Rugani et al., 2013).

When it comes to apply the Life Cycle Assessment (LCA) methodology to report the environmental profile of wine products (Rugani et al., 2013), most studies have delved into the operations throughout the wine production cycle, including viticulture, vinification or distribution, in order to identify the main environmental hotspots of the systems analyzed and, consequently, to propose improvement measures (Bosco et al., 2011; Point et al., 2012; Vázquez-Rowe et al., 2012a; Neto et al., 2013). Other reports have focused on specific auxiliary processes, such as the production of cork or the treatment of organic wastes (Ruggieri et al., 2009; Benetto et al., 2015). Finally, a final set of studies have discussed the comparison of the environmental profile of wines from different appellations, qualities or cultivation methods (Vázquez-Rowe et al., 2013; Villanueva-Rey et al., 2013).

However, as pointed out in an extensive review published by Rugani et al. (2013), most of these studies have focused on the corporative side of winemaking, while disregarding a series of large scale considerations that can have important consequences on the environmental profile of wine. One of these is that of land use changes (LUCs), a phenomenon defined by the United Nations Framework Convention on Climate Change (UNFCCC) as the human-induced change in the uses that a specific piece of land may suffer through time. In fact, recent IPCC reports and a wide range of additional studies have proven that LUCs can have important effects on the amount of greenhouse gas (GHG) emissions, as well as in terms of soil degradation (IPCC, 2015, Milà i Canals et al., 2007; Garrigues et al., 2012). In most cases, LUCs have been analyzed at a global and/or regional scale (Searchinger et al., 2008; Hertel et al., 2010). Nevertheless, their relative importance in worldwide anthropogenic GHG emissions, accounting for 4.3-5.5 Gt CO_2eq/year during 2000-2010 period (IPCC, 2015), as well as their key role in the agricultural sector, representing circa 50% of agriculture, forestry and other land use GHG emissions, has led to a situation in which LUCs are beginning to be included in the environmental accounting of smaller scale studies.

In the case of Spain, GHG emissions linked to the agricultural sector represented approximately 11% of total GHG emissions in year 2012 (MAGRAMA, 2014). In addition, LUCs have allowed GHG emissions in Spain to be substantially lower in the past decades due to afforestation of many agricultural lands that were abandoned due to enforcements from the European Union's Common Agricultural Policy. However, viticulture has shown in previous studies to be a potential threat to the loss of natural habitat and species diversity, as well as a net emission of carbon to the atmosphere (Carlisle et al., 2006; Underwood et al., 2009; Williams et al., 2011). Therefore, the repeated exclusion of carbon storage and emissions due to LUCs, woody biomass in the vineyards and soil management from LCA studies may provoke an underestimation of the GHG emissions that are occurring in vineyard landscapes.

Spain is the main country in Europe in terms of total area destined to vineyards, above France or Italy. More specifically, a total of 943,293 ha of land were used for grape production in Spain in 2013: a decrease of 0.1% with respect to the previous year. Despite this minimal decrease, a general tendency has been observed in the past decades that the actual overall mass production has declined considerably, a period in which winemaking has become highly specialized and intensive in most appellations. Therefore, the correlation between changes in viticulture land and LUCs, and their effect on the net carbon balance, appears to be a pertinent and relevant issue to be analyzed in detail.

To this end, we focused on an iconic appellation in Galicia (NW Spain), named "Ribeiro". This appellation, which currently comprises 2,762 ha destined to vineyards (Xunta de Galicia, 2014a), has observed how the surface area decreased roughly by 1,500 ha during the last 20-30 years, although important changes in the size of wineries and the distribution of vineyards throughout the valley have been identified. Hence, the aim of this study is to shed light on land transformations in the "Ribeiro" appellation and how significant they are in terms of GHG emissions throughout a significant timeframe of 20 years. For this assessment, the carbon stocks (both aboveground and in soils) across the viticulture area, as well as in those areas that suffered some type of vineyard-related land use change were quantified and crossed with the GHG emissions derived from operational activities in order to deliver a more accurate estimation of the climate change consequences of viticulture practices. Moreover, the results of this study are expected to aid policy makers in the wine sector in terms of stirring public policies towards improved farm stewardship and mitigation of GHG emissions in the agricultural sector. In addition, results will provide stakeholders with a long-term estimation of the life cycle GHG emissions that can potentially occur in the viticulture stage of winemaking from a temporal perspective, providing insights on how technology and changes in operational patterns may influence the environmental sustainability of their products.

7.2 Materials and Methods

7.2.1 Goal, scope and system boundaries

A joint interpretation of the IPCC LUCs and LCA methodologies was performed to determine the GHGs related to winegrowing operations in the entire appellation during the 1990-2009 year period. The function of the study was to understand the inter-annual changes in GHG emissions occurring in the appellation, as well as delving into the main factors that influence these variations. A land-based FU was selected rather than a more typical mass-based FU in viticulture studies (Petti et al., 2010; Rugani et al., 2013), given the joint appellation and temporal-oriented approaches taken into consideration in this case study. Consequently, the selected functional unit (FU) was land-based for the entire surface area that was destined to viticulture throughout the period 1990-2009 (i.e., 5,380 ha). In addition, given the fact the cradle-to-field gate perspective considered here (see below), an area-based FU was justified.

The system boundaries of the case study comprised the entire surface area destined to viticulture in the period 1990-2009, including the LUCs occurring in the aforementioned timeframe (Figure 7.1). Therefore, all the land that was no longer used for viticulture by 2009, but during a certain time of the period assessed was destined to grape harvesting, was included within the boundaries. Similarly, the surface that initially had other land uses and became viticulture land in the period assessed was also included in the system boundaries. Finally, all the operations that are part of the viticulture stage of winemaking were taken into consideration, such as the infrastructure of the vineyards (i.e., trellis), the use and maintenance of machinery and the production and combustion of the fossil fuels, production and application of organic and inorganic fertilizers, as well as the production and use of plant protection agents, among other operations (Vázquez-Rowe et al., 2012a,b; Rugani et al., 2013).

7.2.2 Land Use

Data to identify land use dynamics were obtained through digital maps of crops and utilization published by the Spanish Ministry for the Environment and Rural and Marine Affairs (MAGRAMA, 2014). However, it should be noted that several difficulties were encountered to obtain reliable data, in terms of vineyards surface, for the “Ribeiro” appellation before year 2005. This situation was due to the fact that winegrowers —in many rural areas— are not willing to register the total vineyards surface in the communication practices to avoid Regional Government and appellation authorities’ controls, or due to the lack of knowledge about these records (Araújo, 2001). Consequently, surface data before that year were complemented with data arriving from other sources such as the National Statistics Institute —INE (INE, 2014) or the

Regional Statistics Institute —IGE (IGE, 2014b). In addition, maps from two different time periods (1980s and 2000s) were used to compute changes between land uses. The software ArcGIS v.10.1 (ESRI, 2010) was used to carry out the land use study and georeference those dynamics in the Ribeiro appellation.

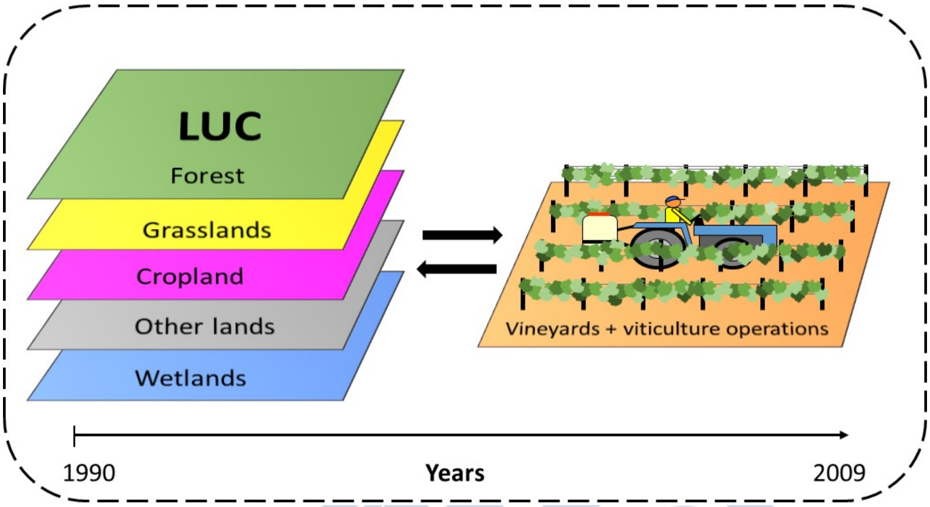

Figure 7.1. Graphical representation of the system boundaries

Once the land use dynamics were identified, the land use categories were grouped into larger categories, following the IPCC guidelines (IPCC, 2006), including vineyards as a new independent category: i) forest; ii) cropland; iii) grassland; iv) other lands; v) wetlands; and vi) vineyards. The category referred to as settlements was excluded since no links between vineyards and human settlements were identified. Furthermore, transition categories were established for the categories cited above in order to distinguish between old and new land uses. These categories are related to the areas that involve a change from vineyard to a new land category (i.e. forest, crops, etc.), or from other categories to vineyards.

The evaluation of the LUCs was performed on an annual basis, covering the years between 1990 and 2009. Thus, national and regional data for crops and utilization were used to complete the land use trends for the assessed period (Xunta de Galicia, 2014b; INE, 2014). In addition to this, data from the Ribeiro appellation was used (Ribeiro, 2014), as well as data obtained through interviews with researchers of the *Technologic Center for the Ribeiro Appellation* —EVEGA (Francisco Rego, personal communication, EVEGA, June 2013). The data gathered allowed performing a more accurate land use dynamics assessment and, therefore, an increase in the reliability on annual LUCs.

7.2.3 Carbon emissions/storage from LUCs

For the evaluation of the carbon emissions/storage derived from LUCs, the IPCC guidelines, specific for LUCs (IPCC, 2006), were considered. The tier 1 and 2 perspectives were used for carbon stock variations in soil and aboveground living biomass. Moreover, the specific data for forest (i.e. growth rate per species, soil organic carbon, etc.), croplands, vineyards (i.e. biomass losses, growth rate, etc.) and grasslands were obtained from the Spanish Greenhouse Gases Inventory (MAGRAMA, 2011). Following the assumptions and recommendations made by this inventory, dead organic matter was excluded from the system boundaries since it is considered a carbon sink. Finally, it should be noted that the lack of reliable data related to the organic matter decomposition rate per land category in Spain may increase the uncertainty of the results (MAGRAMA 2011).

Forest

Regarding forest lands, specific data for growth and production were used for conifers (*Pinus pinaster*) and leafy trees (*Eucaliptus globulus*). Both species were and are still predominant in the region according to the National Forest Inventory (MAGRAMA, 1997; 2008). Furthermore, soil organic carbon (SOC) content was obtained from the *Atlas de los Suelos de Galicia* (Trueba et al., 1998), a publication that includes soil profiles for the area considered.

Despite the fact that in this area there is no forest management, forest fires, wood and firewood extraction were also included to account for biomass losses and, therefore, carbon related emissions. These wood extractions are negligible and are reduced to subsistence management, which is mainly based on firewood for heating and livestock bedding, and, therefore, natural forest regeneration after wood/firewood extraction and forest fires was taken into consideration, as well as for the forest lands transition category. Furthermore, it should be noted that forest fires are relatively frequent in Galicia, so annual burnt forest area (Xunta de Galicia, 2014a) —specific per year— was considered, as well as the derived GHG emissions (IPCC, 2006). Appendix (Equation C.1 to Equation C.6) provides further insights regarding the mathematical computation of the GHG emissions due to different land use transitions.

Cropland

Given the fact that only annual crops were identified in the area, only changes in SOC were considered. However, it should be emphasized that other GHGs were taken into account, such as those derived from fertilization. In addition, crop specific data (yield, nitrogen content, fraction used as fodder, and remaining vegetable residue after harvesting) were used based on each crop type (e.g. maize, bean, wheat, rye, potato, and fodder) (IPCC, 2006; MAGRAMA, 2011; Trueba et al., 1997).

Grasslands

Grasslands in the region are either not managed or used as grazing lands. Therefore, the study only took into account SOC, which was obtained from the Spanish Greenhouse Gases Inventory (MAGRAMA, 2011) using specific data for the area studied.

Other lands

Following the IPCC guidelines for LUCs, the category other lands includes those land uses which do not comply with the requirements for being considered forest lands. For instance, unproductive lands and shrub were included within this category. Due to the lack of reliable data concerning shrub biomass and, therefore, carbon content, only the SOC content was taken into account for this category. Data for SOC, in a similar way as in forest lands, was obtained from the *Atlas de los Suelos de Galicia* (Trueba et al., 1998).

Wetlands

Carbon losses derived from vineyards transformation were the only feature taken into account. No other specific parameters were considered in this category.

Vineyards

Vineyards and vineyard-related LUCs are the central aim of the study. Hence, specific carbon biomass content, as well as losses derived from pruning, was used for vines (MAGRAMA, 2011). Moreover, different data were used for new and old vineyards since they are considered a carbon sink or a carbon source depending on the age of the vines. In this sense, new vines are considered a carbon sink because of their lower production yield and vine growth during the first 7-10 years (pruning and training is mainly focused on getting the mature age). Thereafter, vines are considered a carbon source due to the pruning waste. The SOC for vineyards was taken into account due to soil management changes during the assessed period —lower tillage operations. In a similar way to other crops, forest, and other lands, the data for SOC was obtained from *the Atlas de los Suelos de Galicia* (Trueba et al., 1998).

7.2.4 Data collection and Life Cycle Inventory implementation

The main operational inputs identified in viticulture —in terms of GHG emissions— were diesel, fertilizers, pesticides, and trellis, according to previous LCA studies in the area (Vázquez-Rowe et al., 2012a; Villanueva-Rey et al., 2013). Hence, the life cycle inventory (LCI) focuses mainly on these items, as reported in Table 7.1 and Table E.7 of Appendix. Additionally, the inventory follows the guidelines for LCA and Carbon Footprint in the wine sector, as proposed by Villanueva-Rey et al. (2014). Thus, the processes to carry out the LCI were modified on an annual basis, which implied the adaptation of the Spanish electricity country mix to the characteristics of the year under study, as well as the consideration of diesel and gasoline production standard changes in the period assessed. For instance, diesel and gasoline composition has changed during the assessed period due to the implementation of more restrictive EU directives

(European Commission, 1998; 2009a) (i.e. Directive 98/70/EC and successive amendments) regarding sulfur and lead content, increasing the final impact for both processes. In addition to this, the renewal of the agricultural machinery fleet —from the regional census (Xunta de Galicia, 2014b)— was taken into account since emissions factors vary depending on the year of manufacture (EMEP, 2013).

Primary data gathering for the LCI was carried out throughout winegrowers and experts from the EVEGA. Data concerning fertilization intensity was obtained from the Yearbook of Agri-food Statistics (MAGRAMA, 2010) and adapted to vineyard requirements —lower than other crops— following the recommendations made by researchers from EVEGA. Only NPK fertilizers were used to meet vineyard requirements in terms of mineral fertilization, whilst two types of organic fertilizers were used depending on vineyard age. The rationale behind this approach is linked, on the one hand, to the fact that farmers, mainly those that own the oldest vineyards, use residues from other crops and/or livestock bedding for vineyards fertilization. On the other hand, new vineyards are fertilized spreading commercially-pelletized or crumbly compost. The data for commercial compost were adapted from Zhong et al. (2013).

Data for pesticides were addressed depending on the vineyards age —related to vineyards management— and the inventory year. Firstly, old vineyards were only treated with sulfur, Bordeaux broth (i.e., quick lime and copper sulfate), and unspecific pesticides (i.e. following ecoinvent® classification) (Nemecek et al., 2007). This is due to the subsistence farming practices in the area 20-25 years ago, when pesticides were sprayed using a wheelbarrow or knapsack sprayers in most cases, as well as bellows for sulfur powder spreading. Secondly, new vineyards were treated with sulfur, Bordeaux broth, copper-based pesticides, and other organic compounds during the first decade. However, it should be noted that pesticide use is superior in new vineyards than in old ones, according to the interviews and the comments provided by researchers at EVEGA. Finally, pesticide types and consumption were considered identical for both types of vineyards (i.e. new and old) during the second decade of the assessed period.

Table 7.1. Life cycle inventory for main inputs per hectare devoted to vineyard (kg/ha)

Year	Diesel	Gasoline	Mineral Fertilizers
1990	163.35	15.10	30.85
1991	165.90	13.80	30.70
1992	169.52	12.50	28.45
1993	171.47	11.22	24.25
1994	174.36	9.94	20.10
1995	178.33	8.69	18.47
1996	268.05	16.75	23.23
1997	325.08	23.47	20.70
1998	310.31	18.82	22.57
1999	344.52	19.82	24.47
2000	306.79	15.72	25.67
2001	351.41	13.50	23.27
2002	331.59	12.78	20.97
2003	311.73	12.09	24.87
2004	291.84	11.41	22.57
2005	316.44	11.47	19.63
2006	336.94	11.51	21.10
2007	357.46	11.53	21.93
2008	357.65	11.22	16.70
2009	317.10	10.28	16.90

The consumption of pesticides for LCI implementation was obtained through direct interviews with winegrowers and researchers at EVEGAL, as well as from previous LCA inventories and guidelines (Vázquez-Rowe et al., 2012a; Villanueva-Rey et al., 2014). Nevertheless, it should be highlighted that on-field pesticide emissions were disregarded since they do not account for environmental impacts in terms of GHG emissions.

Trellis and materials used for vine training were also considered, taking into account the changes throughout the 20 years of assessment. The area under study has shown variations regarding the materials used, as well as in terms of planting density and frame. Hence, the inventory brings to light these changes for new vineyards, which tend to introduce new materials. The materials considered for trellis were granite, concrete, and slate for posts, whilst iron was considered for wire. The share of each material depends, as other inventory elements, on the vineyards age. For instance, in 1990, 20% of vineyards used wooden trellis, and 80% used other materials, mainly granite (64%). However, traditional woody trellis only represented 5% of the vineyards by 2009, while new materials, such as slate (7%) or cement (32%), showed important increases throughout the timeframe under analysis, according to data provided by the Ribeiro appellation (Ribeiro appellation headquarters, personal communication, January 2014). Secondary and background data required to complement the primary data gathered in this case study were mainly obtained from ecoinvent database, version 3.01 (Weidema et al., 2013).

7.2.5 Life Cycle Impact Assessment

Despite the LCA perspective used for this case study, one single impact category was computed in the assessment: climate change potential (CC). For the computation of this impact category, as recommended by the ILCD guidelines (Hauschild et al., 2013), the IPCC 2007 GWP assessment method was selected (Frischknecht et al., 2007). The main reason for the selection of CC exclusively is linked to the objectives of the case study to monitor the role of LUCs in the viticulture sector. Despite the fact that LUCs do not only generate environmental impacts in terms of GHG emissions, the latter have become the main source of preoccupation among policy makers at regional scales. The software used to compute the LCIA results was SimaPro 8.01 (PRè-Product Ecology Consultants, 2014).

7.3 Results

7.3.1 Analysis of land use changes

The results for LUCs show variable gains and losses depending on the land use category. Table 7.2 presents the LUCs matrix following the IPCC guidelines for the area under study during the period assessed. Firstly, forest land has shown a considerable increase in its surface by 15%, mainly due to afforestation from croplands and grassland. In addition, wetlands have also increased their surface by 925 ha, mainly from other lands. In contrast, croplands and grasslands have decreased their surface by 31% and 17%, respectively, mainly towards new forest lands. Finally, other lands have shown a decrease in their surface of 30%, mainly due to changes to wetlands.

Despite the fact that LUCs for the entire region were identified, the study, as aforementioned, aims at monitoring and interpreting LUCs related to viticulture. Therefore, carbon emissions/storage were only considered for those LUCs that imply the existence of vineyards either in the past (1980s) or in the present. Figure 7.2 shows the results for LUCs taking only into account vineyards during the 1990-2009 period (complete data are available in Table C.8 of Appendix). Vineyard losses were towards forest lands and other lands, 1,613 and 384 ha, respectively, whilst gains were also identified from forest lands. The main reasons behind these LUCs can be directly linked to socioeconomic and demographic changes during the 1980s and 1990s (vineyards abandonment) and planting of new vineyards from clearing forests, where land prices were cheaper and plots were larger.

Table 7.2. Land Use Changes (LUCs) matrix for the whole Ribeiro county area. Data in hectares

Final \ Initial	F	G	C	W	S	O	*Final sum*
F	**10,475**	2,636	2,036			154	***15,302***
G	1,845	**1,997**	283			30	***4,155***
C	654	318	**2,707**			120	***3,799***
W	73	10	69			777	***929***
S							***0***
O	245	90	402			**104**	***841***
Initial sum	***13,292***	***5,052***	***5,497***	***0***	***0***	***1,185***	*25,026*

Note: F = Forest land; G= Grassland; C = Cropland; W = Wetlands; S = Settlements; O = Other land

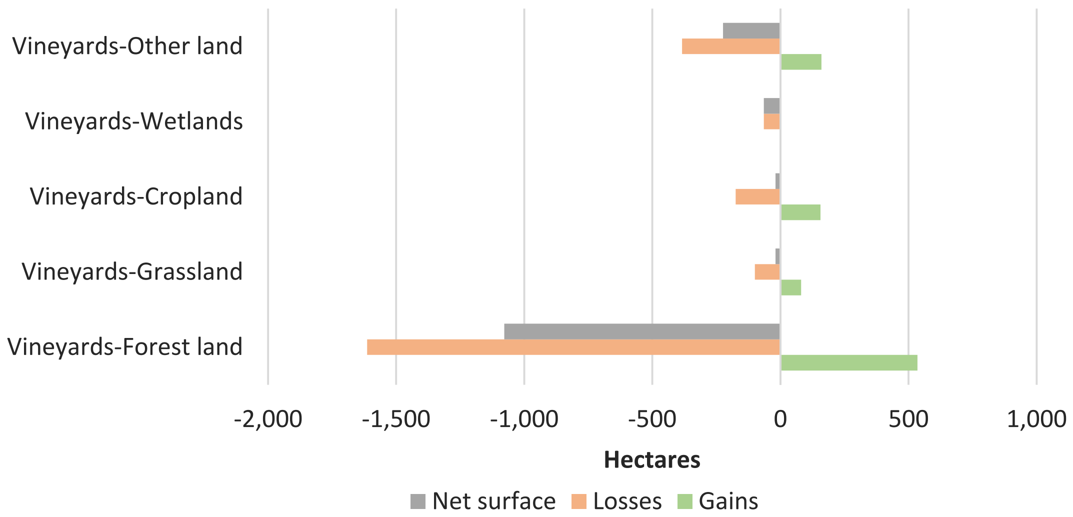

Figure 7.2. Land use changes (LUCs) related to vineyards in the "Ribeiro" appellation in the period 1990-2009

7.3.2 GHG emissions due to land use changes

Figure 7.3 presents the GHG emissions in terms of CO_2 eq per land use category during the period assessed. The complete data disaggregated by compartment: biomass, soil, forest fires and fertilizers, can be consulted in Appendix.

The land categories Forest and Forest in transition resulted to be a potential CO_2 sink during the period evaluated, regardless the negative impacts linked to forest fires, wood and firewood extractions. Forest growth and soil carbon storage derived from vineyards were the main contributors to carbon storage. During the period assessed, variable potential capacity for carbon storage was observed since carbon storage of forest, which was forest previously, is higher than for forest in transition —LUCs from vineyards. In other words, the carbon fixation of forests tends to decrease over time, despite the fact that the final surface area is larger (IPCC, 2006; Rounsevell and Reay, 2009).

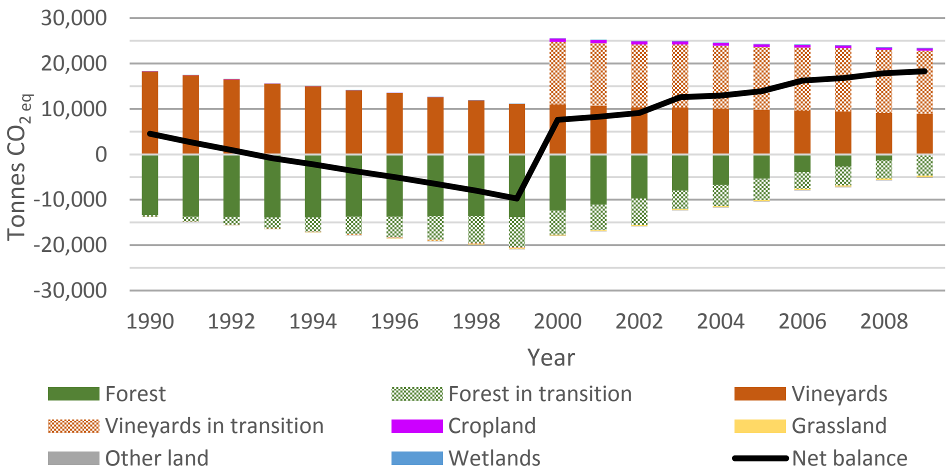

Figure 7.3. GHG emissions related to Land Use Changes (LUCs) and net balance during 1990-2009 period

Vineyards were identified, as initially expected, as important sources of carbon emission. Biomass removal —leaves, branches and pruning— and SOC emissions derived from tillage appeared to be relevant in terms of carbon losses and, therefore, GHG emissions (Table C.9 of Appendix). Furthermore, it should be noted that this category also includes other emissions from fertilizers, such as dinitrogen oxide. Given the fact that vineyards in transition imply deforestation, carbon losses are significantly noteworthy due to forest clearing and changes in soil management. Additionally, carbon fixation of woody biomass —new vines— does not offset carbon losses due to deforestation. Other converted lands to vineyards: grassland and other land are negligible in terms of either carbon losses or storage, when compared to forest related dynamics.

Regarding croplands, two different trends were observed. During the first decade cropland was converted to vineyards, implying that carbon losses were related to cropland fertilization and changes in soil management. During the second decade, the trend was inverted to cropland conversion from vineyards. Consequently, carbon emissions from vineyards removal and GHG emissions derived from fertilization were the main sources of environmental impact.

Nevertheless, it should be highlighted that these emissions were partially mitigated thanks to carbon storage in soil due to changes in management.

The conversion from vineyards to wetlands, other land and grassland implies carbon emissions from biomass removal and carbon storage in soil. However, it should be noted that these conversions are almost negligible in terms of GHG emissions as can be observed in Table C.8 of Appendix.

Finally, once the final carbon balance was obtained, the net GHG emissions presented two different trends throughout the timeframe assessed (Figure 7.3). For the first decade, the studied area becomes a carbon sink due to, among other motives, afforestation of land previously destined to vineyards. For instance, 1999 presented the peak in carbon mitigation, a year in which nearly 9,742 tonnes of CO_2 eq were stored in the area under assessment. In contrast, the tendency inverts by year 2000, when the appellation starts clearing forest land for the expansion of new vineyards. Despite the fact that in other areas of the appellation the land destined to viticulture continues to descend, in 2000 approximately 7,600 tonnes of CO_2 eq were emitted to the atmosphere due to LUCs, a number that increases up to 18,270 tonnes by 2009.

7.3.3 Annual LCA results linked to viticulture operational activities

To complement the GHG emissions linked to LUCs in the appellation, an annual LCA was conducted for vineyard field operations, as shown in Figure 7.4. Production and combustion of fossil fuels to power machinery on field were responsible for the highest relative contribution of GHG emissions during the 1994-2009 period, accounting for up to 68% of the impacts in year 2008. In fact, it is worth noting that the lowest absolute values for fossil fuels were observed in the early 1990s, with the lowest value observed in 1992 (500 kg CO_2 eq per ha). Interestingly, the amount of fossil fuel used per ha has increased ever since, reaching levels ranging from 650-780 kg CO_2 eq per ha in the period 1997-2009.

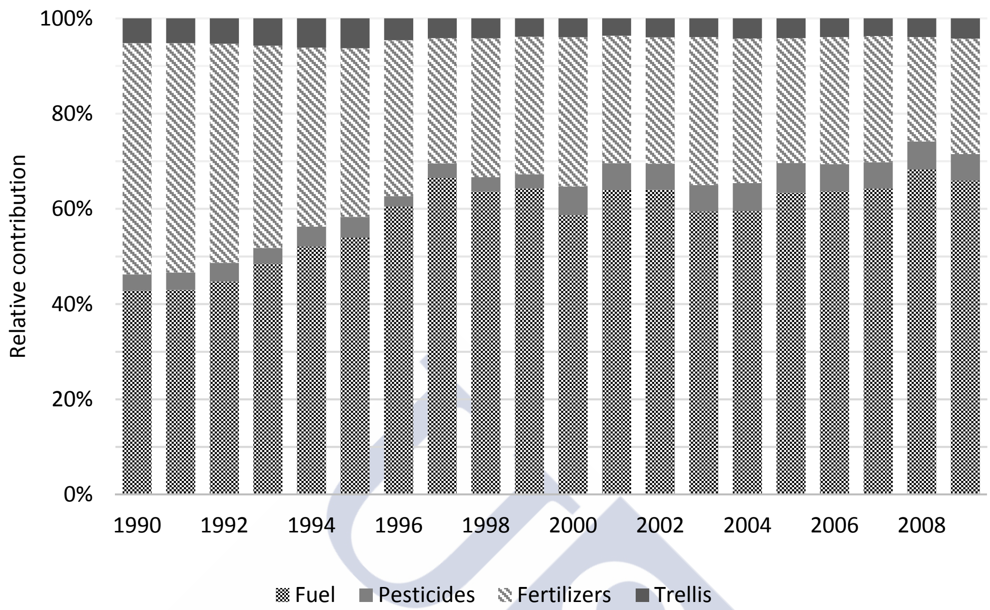

Figure 7.4. GHG emissions relative contribution for vineyard field operations during the 1990-2009 period

In contrast, production of organic and mineral fertilizers[2] was the main source of emissions in the period 1990-1993, accounting for 50% of the impact in 1990, especially due to the production of mineral fertilizers. Fertilizers production accounted for up to 602 kg of CO_2 eq per ha in 1990 and gradually decreased until a record low of 248 kg CO_2 eq per ha in 2008.

Regarding plan protection agents (i.e., pesticides), and given the fact that these have suffered important changes —from simple compounds (powdered sulphur and Bordeaux broth) to organic and more complex compounds— during this period, the annual LCA shows these changes due to the increasing impact contribution over the years. Finally, the impact contribution of trellis remains steady during years (*circa* 5%). More detailed results regarding the different operational activities can be consulted in Table C.10 of Appendix.

[2] It should be noted that GHG emissions due to fertilization were computed as part of the emissions linked to LUCs, following the IPCC guidelines (IPCC, 2006).

The overall GHG emissions per year, including LUCs, as shown in Table C.10 of Appendix, ranged from -697 kg CO_2 eq per ha in 1999 to a record high in 2008: 4,240 kg CO_2 eq per ha. However, if LUCs are removed from the lump sum, it can be observed that annual values range between 900 kg CO_2 eq (1995) and 1,239 kg CO_2 eq per ha (1990).

7.4 Discussion

7.4.1 Identification of the main driving forces related to LUCs

The use of GIS software allowed the identification of the dynamics between land use categories. However, a limitation of using this software in the current case study was that it cannot offer disaggregated annual data. Hence, the combined use of GIS and statistical data shed light on LUCs on an annual basis, allowing more accurate data regarding land occupation per year. Thus, the main trend that was observed was that land destined to viticulture was reduced in benefit of afforestation, especially for the period between 1990 and 1999. The main reasons behind these reductions can be linked to the demographic crisis that most rural areas in the region of Galicia have been suffering for the past 30-40 years, due to population ageing and emigration of an important part of the workforce to coastal areas or more prosperous regions within Spain or Europe (Arroyo, 2003). For instance, the population of the county (Ribeiro, capital Ribadavia, 42º17'16"N, 8º08'33"W) decreased from roughly 25,000 inhabitants in late 1980s to 18,329 inhabitants in 2011, when the last population census was carried out (Xunta de Galicia, 2014b). While similar patterns have been identified for other European countries (Alcantara et al., 2012; Gerard et al., 2010; Lieskovský et al., 2013), the situation in this area is persistent as vineyards abandonment remained steady in the 2000s, implying that these phenomenon is still affecting LUCs.

Nonetheless, despite these vineyard losses, new vineyards started to be established starting in year 2000. The main reason for this expansion was the change in viticulture patterns, shifting from small-holding subsistence viticulture to the proliferation of medium and large size wineries in certain areas. In addition, it should be noted that these new vineyards were derived from deforestation rather than from acquiring land that was already destined to viticulture and, therefore, important losses of carbon from soil and biomass were identified in this process.

The creation of new vineyards from forest land was the consequence of a series of socioeconomic and policy making factors. Firstly, the feasibility to buy large plots —avoiding the traditional smallholding pattern of vineyards— allowed creating new land for viticulture with better machinery access for professional winegrowers (Yáñez et al., 2005). Secondly, the price of forest land was lower than that of vineyards or croplands. However, it should be noted that the costs derived from deforestation and land preparation are higher in this case. Finally, public subsidies, purchase-sale of planting rights, uprooting of old or abandoned vineyards and the boom of new wineries, triggered the entry of investors mainly from outside the wine producing regions (Bravo and Puga, 2014). Additionally, some vineyards were reconverted (i.e. changing planting density and frame) for the sake of increasing high quality regardless of having lower yield. During this reconversion of vineyards, neighbouring land plots destined to growing potatoes, maize, bean or wheat were added since in the past, it was very common to combine vineyards and other crops as subsistence farming.

Regarding other LUCs, new wetlands were created as a consequence of river bank and hydraulic interventions since some vineyards were flooded along the river —the “Ribeiro” appellation is crossed by the Miño and Avia rivers. Furthermore, vineyard losses towards other types of land (i.e. unproductive and shrub lands) occurred due to abandonment and, therefore, the natural progress of ecological succession. For the case of LUCs involving grassland, as in the case of forest converted to new vineyards, the feasibility to buy large plots and low land prices triggered this land conversion, whilst new grassland was created from vineyards to enlarge those that already existed in the area.

7.4.2 Greenhouse gas emissions from LUCs

The main source of GHG emissions was related to vineyards and vineyards in transition (ranging from 8,919 to 13,784 tonnes CO_2 eq) during the assessed period. The soil carbon losses (i.e., tillage and LUCs after deforestation), the biomass removal from forest to establish new vineyards and on-field emissions derived from fertilization triggered carbon emissions. Other carbon losses, such as pruning waste and leaves, were also taken into account for GHG emissions calculation (MAGRAMA, 2011). However, it should be highlighted that new vineyards, given their perennial and woody crop characteristics, are considered a carbon sink due to their growth during the first 10 years (MAGRAMA, 2011). In addition, it should also be noted that pruning waste is traditionally burned or, in recent years, crushed on-field. When vineyards and vineyards in transition are studied separately, new vineyards imply higher carbon losses —derived from forest clearing— than existing vineyards, despite the latter occupying twice the available surface.

Forest and forest in transition were identified as carbon sinks thanks to biomass growth and carbon storage in soil. However, the growing rate and, therefore, the biomass carbon storage of forest in transition were lower than the remaining forest lands. Consequently, forest in transition did not offset the carbon losses derived from deforestation in spite of representing a larger surface area. Croplands (i.e. maize, bean, wheat, rye, potato and fodder) substituting vineyards resulted to be a carbon source due to biomass removal when clearing. In addition, cropland fertilization related GHG emissions and crop waste were almost negligible in comparison with carbon losses from vine removal.

Other lands and grassland converted from vineyards were identified as a carbon sink due to increasing SOC storage and due to the lack of tillage and other soil management operations. Given the lack of data regarding the living biomass removal, carbon losses were allocated over a period of 20 years (IPCC, 2006; MAGRAMA, 2011) rather than in one single year, since the described dynamics does not imply vine uprooting (e.g. croplands in transition).

Finally, the net GHG emissions balance shows two different patterns depending on the assessed period. During the first decade, a carbon sink due to the afforestation from vineyards abandonment can be observed. Other studies which have dealt with LUCs and GHG emissions found similar trends derived from farmlands abandonment and, therefore, the increasing of carbon sequestration in new forest lands (Rounsevell and Reay, 2009). In contrast, during the second decade carbon emissions derived from deforestation to establish new vineyards was the main tendency.

7.4.3 Environmental hotspots linked to viticulture operational activities

GHG emissions derived from viticulture are dominated by fuel production, on-field emissions and the production of fertilizers, in a similar way to other LCA analysing grape production (Vázquez-Rowe et al., 2012a,b; Neto et al., 2012; Point et al., 2012, Gazulla et al., 2010; Rugani et al., 2013). In addition, as stated above, the inventory shows the evolution of the farming techniques, pesticides and machinery in the region over the years so that the increasing impact contribution of fuel highlights a high level of mechanization year after year (Xunta de Galicia, 2014b), even though the overall vineyards surface was getting smaller.

Emissions related to the production of fertilizers were reduced through time, as seen in Table C.10 of Appendix, although a common characteristic was that the production of mineral fertilizer was in all cases the main potential impact. Two main drivers were responsible for these reductions. On the one hand, decreasing area destined to viticulture reduced the need for fertilizers. On the other hand, professionalization of winegrowers allowed the implementation of improved practices regarding fertilization, pruning, tillage and pesticides, including the use of record books which register all fertilizer and pesticide interventions. All these improvements have allowed an important decrease in fertilizers use (from 31 kg of N as mineral fertilizer per ha to 16 kg N/ha).

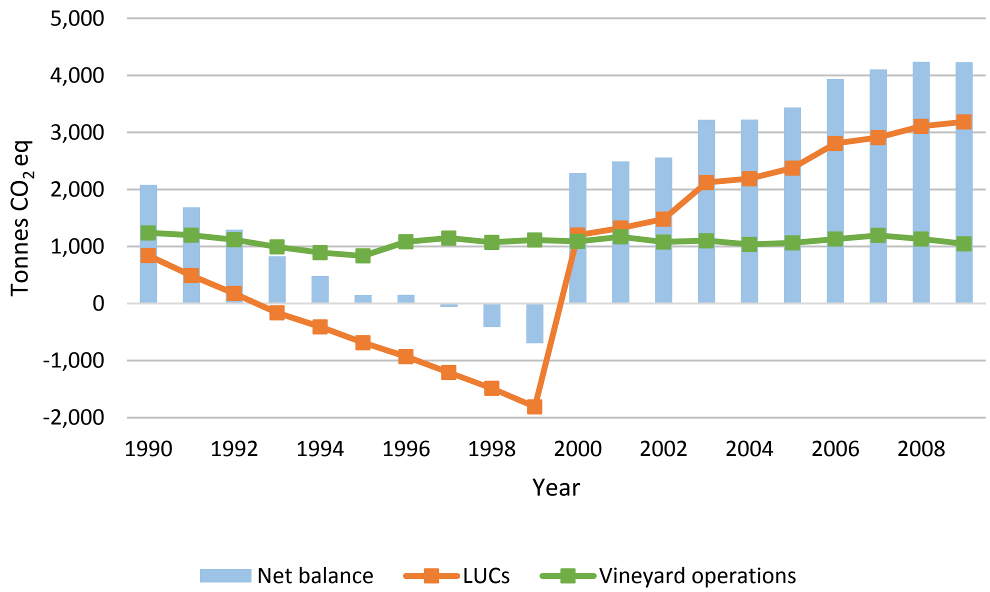

Figure 7.5. Annual GHG emissions from vineyards operations and LUCs

When GHG emissions derived from LUCs and vineyard operations are linked —net balance—, different patterns can be observed (Figure 7.5). On the one hand, despite the fact that emissions from vineyard operations remained in a steady range of around 1,000 tonnes CO_2 eq, the LUCs were able to offset them during the late 1990s by the creation of new forest lands. On the other hand, the creation of new vineyards, mainly from forest lands, increased dramatically GHG emissions during 2000-2009 period.

7.4.4 Mapping the spatial differentiation of GHG emissions

The results of this study are intended to be of utility for farmers to understand the benefits and risks in terms of climate change potential, not only linked to direct operational activities on field, but also related to the carbon net balance of changing vineyard landscapes. Therefore, beyond additional criteria, such as solar radiation, soil characteristics or socioeconomic factors, the GHG emissions consequences of vineyard landscaping should be present in the decision process of wineries and winegrowers.

In addition, these results intend to be of support for policy makers in order to guide future legislative actions to mitigate GHG emissions due to the lack of planning in vineyard landscaping in winegrowing regions. The results presented prove that a higher level of land management and farm stewardship could lead to the maximization of carbon stocks in vineyard landscapes, by implementing legislative actions to avoid the conversion of forest lands with high potential of carbon storage (Jackson et al., 2003). For instance, a study conducted by Williams et al. (2011) showed that even old vines with high aboveground carbon content were only capable of retaining 25% of carbon as compared to surrounding wooded areas.

In fact, Figure 7.6 represents the carbon net balance that would occur in the appellation if new vineyards had not been created in deforested areas. In this scenario, the change in net balance tendency in the late 1990s is still present but the increase in GHG emissions is less dramatic. Therefore, rather than a net emission of approximately 18,270 t of CO_2 eq in 2009, the emissions would be slightly below 5,000 t of CO_2 eq thanks to the avoidance of undesirable LUCs.

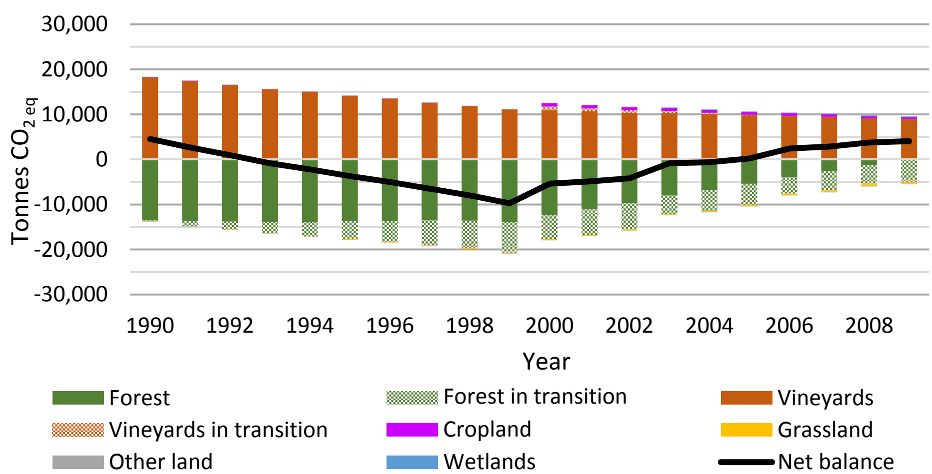

Figure 7.6. LUCs related GHG emissions if land use policies had banned afforestation for new vineyards in the period assessed

In a similar way, Figure 7.7 shows the average annual net carbon loss or gain in a specific area of the appellation based on the cumulative GHG emissions, allowing to identify which areas have had a negative performance in recent years. In other words, the results presented in Figure 7.7 represent the spatialization of the lump sum GHG emissions shown in Figure 7.3. Therefore, while the latter aims at quantifying the magnitude of the environmental impacts, the former provides stakeholders with mapped information regarding vineyard landscaping consequences throughout the appellation.

7.4.5 Main limitations and constraints

Despite the utility of the results presented, a series of constraints were identified when developing the case study that is worth highlighting for the sake of transparency and reproducibility of the results. Given the fact that the maps from the CORINE Land Cover project (EEA, 2014) do not cover both assessed periods, a set of digitalized maps, which dealt with land occupation of Spain during the 1980s and 2000s were used to obtain the LUCs related data. Additionally, the lack of more detailed data for carbon storage/emissions, the decomposition of organic matter, as well as parcel specific inventories for forest lands made impossible to use the Tier 3 level for GHG emissions following the IPCC guidelines. However, it should be noted that specific SOC data, forest species breakdown, living biomass removals (i.e. forest fires, wood extractions, etc.) were used with a considerable level of detail.

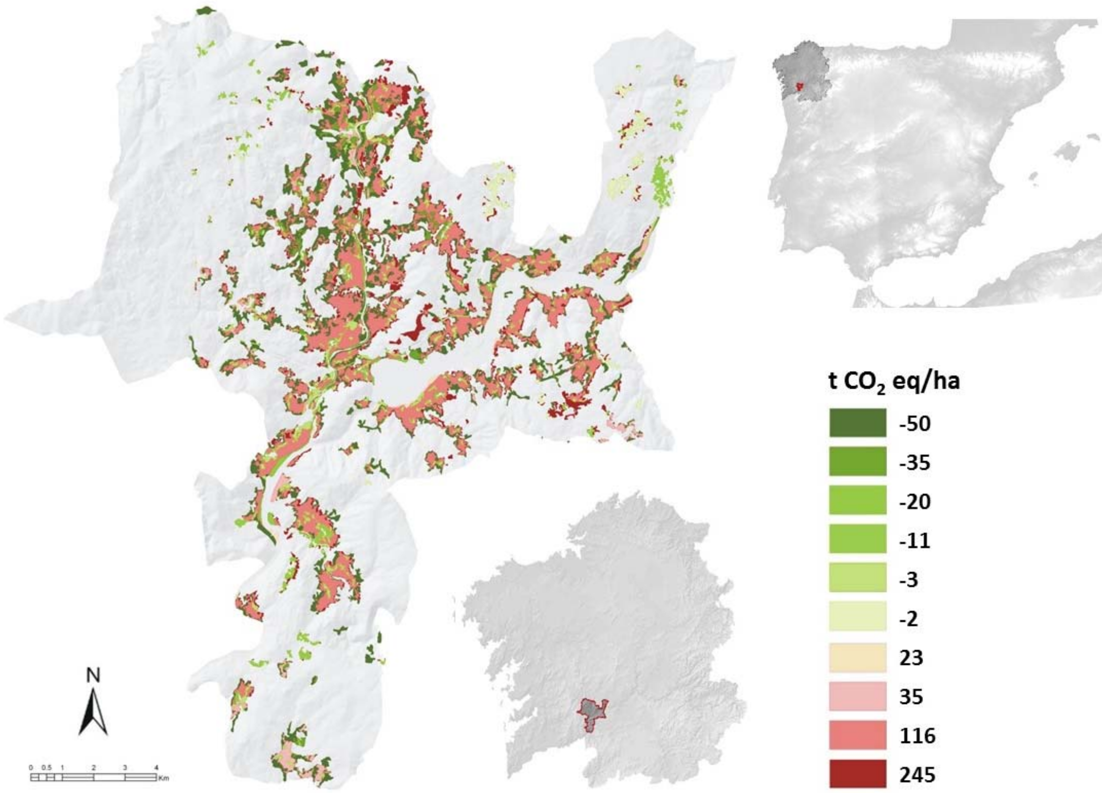

Figure 7.7. Spatial mapping of GHG emissions across the "Ribeiro" appellation in the 1990-2009 period

In a similar way, the LCA of viticulture operational activities was carried out with several limitations due to the difficulties when it came to gathering inventory data for the 1990s, as well as LCA database background processes. Therefore, despite the LCI being implemented throughout interviews to winegrowers and researchers, and based on previous LCA studies in the region (Vázquez-Rowe et al., 2012a; Villanueva-Rey et al., 2013), data gaps were more persistent in the early years of the 1990s, leading to a greater number of assumptions when constructing the datasets. Nonetheless, as aforementioned, the inventory was able to reflect the evolution of machinery, quality and composition of fossil fuels, plant protection agents, trellis and fertilization over the years.

7.5 Conclusions

The GHG emissions linked to viticulture in LCA studies have been recently questioned due to the omission of the carbon balance of LUCs in the GHG balance of winegrowing operations (Rugani et al., 2013). According to the results presented in the current case study, this balance may be of particular importance if new vineyards are being planted in areas that have previously had other uses such as forestry and vice versa. In fact, based on the two-step tendency in terms of changing vineyard landscapes in the Ribeiro appellation, both carbon emissions and carbon sink processes occur in different time periods. Consequently, the carbon net balance due to biogenic carbon should not be neglected in future viticulture studies, as recommended in the ISO 14067 for carbon footprinting and the revision of the PAS 2050 guidelines for agricultural products (BSI, 2011, 2012; ISO, 2013).

While this case study only focused on a specific appellation in NW Spain for which numerous primary data were available, it is important to bear in mind that similar processes occur throughout appellations in the Old World and in the New World. Moreover, as a consequence of climate change, extreme events are expected to intensify in many viticulture areas in the world (Wilbanks and Kates, 2010). Heavy rainfall and flood, droughts and forest fires may be just some of the climatic threats that wine appellations will suffer in the 21st century (Jones et al., 2005; Mira de Orduña, 2010). These scenarios would translate, in many cases, as forecasted by Anderson et al. (2008), in the relocation of vineyards within existing regions and between regions in order to maintain the climatic conditions, as well as the standard quality criteria, that have always been linked to the cultivation of a specific wine product. Consequently, the analysis of LUCs in vineyard landscaping may become a key factor to avoid net carbon emissions in these processes. In fact, the development of specific legislative frameworks, in a similar way to the

Renewable Energy Directive (RES-D 2009) that the European Commission passed to tackle the propagation of unwanted LUCs due to the proliferation of bioenergy crops, would be desirable to prevent changes in vineyard landscaping from being detrimental in efforts to reduce GHG emissions (Lange, 2011).

In addition, a series of measures that do not require relocation strategies can be divided into two main blocks. On the one hand, certain studies have brought about the important carbon fixation in soil of shifting from conventional to organic viticulture (Venkat, 2012), although conversion in many appellations will depend on the willingness of consumers to trigger this market niche. On the other hand, operational activities in the vineyards, which have been the main focus of attention of the existing LCA literature on viticulture systems still remains an integral part in the mitigation of GHG emissions. These improvement actions include, but are not limited to: i) alterations in the trellis systems (Diffenbaugh et al., 2011); ii) the use of low energy-intensive fertilizers (Vázquez-Rowe et al., 2012a); or iii) the optimization of fossil fuel-based interventions in the vineyards.

Based on these results, it can be concluded that the elongation of the system boundaries in viticulture LCA studies to include biogenic CO_2 may allow stakeholders in the wine sector to construct feasible long-term GHG emissions mitigation strategies on both a local and regional/continental scale.

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SECTION III

LIFE CYCLE ASSESSMENT APPLIED TO SEAFOOD

Chapter 8

Protein energy return on investment ratio (EROI) for Spanish seafood products[1]

Summary

Life Cycle Assessment (LCA) has developed into a useful methodology to assess energy consumption of fishing fleets and their derived seafood products, as well as the associated environmental burdens. In this study, however, life cycle inventory data are used to provide a dimensionless ratio between energy inputs and the energy provided by the fish: the edible protein energy return on investment (ep-EROI). The main objective was to perform a critical comparison of seafood products landed in Galicia (NW Spain) in terms of ep-EROI. The combination of energy return on investment (EROI) with LCA, the latter having standardized mechanisms regarding data acquisition and system boundary delimitation, allowed a reduction of uncertainties in EROI estimations. Results allow a deeper understanding of the energy efficiency in the Galician fishing sector, showing that small pelagic species present the highest ep-EROI values if captured using specific fishing techniques.

[1] Vázquez-Rowe, I., Villanueva-Rey, P., Moreira, M.T., Feijoo, G., 2014. Edible Protein Energy Return on Investment Ratio (ep-EROI) for Spanish Seafood Products. AMBIO 43, 381–394.

Index

8.1 Introduction

Current fishing practices depend in many fisheries on large expenditures of energy, mainly fossil fuels. In fact, Tyedmers et al. (2005) have revealed that approximately 1.2% of world oil is used to fuel fishing fleets throughout the world. While fuel dependence implies an important economic limitation in most fisheries, it is also a matter of concern from an environmental perspective, due to the derived high greenhouse gas (GHG) emissions (Fredga and Mäler, 2010).

These findings have recently fueled the debate around the environmental evaluation of fishing systems (Pelletier et al., 2007; Ford et al., 2012; Vázquez-Rowe et al., 2012a). In fact, until recent years environmental indicators in fisheries were limited to evaluating the stock of a particular species or group of species, focusing on the direct impacts of biomass removal from fisheries (de Haes et al., 2002; Langlois, 2012). However, the development of a series of environmental management tools in the past few years has increased the environmental aspects that are examined in these production systems, such as climate change, ozone depletion, eco-toxicity or eutrophication, impact categories included in the Life Cycle Assessment (LCA) methodology (Pelletier et al., 2007; Vázquez-Rowe et al., 2012a; Avadí and Fréon, 2013). In this Chapter, the use of life cycle thinking is proposed to nourish and improve the consistency of a commonly used assessment method for energy analysis: energy return on investment (EROI).

EROI is a term that flourished in the early 1970s and gained importance due to the fuel crisis in the 70s and 80s (Hall, 1972; Gupta and Hall, 2011). The most common use of EROI is within the energy sector in order to determine the energy that is returned from an energy-collecting process as compared to the energy that is required to provide this energy (Gupta and Hall, 2011). In addition, broad comparisons of protein sources can be performed by calculating a dimensionless ratio of the edible protein energy content of an animal relative to the total industrial energy expended in its production/acquisition: the edible protein energy return on investment (ep-EROI) ratio (Tyedmers, 2000; Hall, 2011).

Pelletier et al. (2011) have highlighted the benefits of calculating EROI approaches to monitor energy return in food systems, since these rely on high inputs of non-renewable resources. More specifically, the use of ep-EROI in seafood products has shown to be highly relevant when examining the fishery stage, which in many cases may be highly energy intensive (Tyedmers, 2001; 2004). In fact, ep-EROI provides a policy relevant approach to monitor the underlying energy sources used in fisheries in a way that markets cannot (Murphy and Hall, 2010).

The coupling of these two methods (i.e. LCA and EROI) pursues a reduction in the uncertainties linked to the two main limitations of EROI: data quality and the definition of the system boundaries (Murphy and Hall, 2010). Additionally, the life cycle approach suggested allows a more robust comparison in terms of EROI between production systems as compared to more simplistic estimations such as fuel use intensity (FUI), since the latter does not consider the energy requirements of many fishery operations that rely on other sources of energy (e.g. electricity, fishing net and ice production). Hence, the proposed methodology aims at providing not only an intra-assessment of fishing species between fisheries, but also an inter-assessment with other sources of protein (i.e., meat or aquaculture products) and food products in general (Tyedmers, 2004; Pimentel et al., 2007; Pelletier, 2008).

Consequently, the main aim of this study was to perform a critical comparison of a set of recently examined seafood products within the fishing fleet in Galicia in terms of ep-EROI. The use of inventory data collected from LCA studies, as well as the estimation of cumulative energy demand (CED) through this life cycle perspective tool to calculate ep-EROI values is expected to strengthen the validity of results. More specifically, the partial objectives of the study were to: i) calculate the CED following LCA methodology of a set of 24 different fishing species captured with 3 different fishing techniques; ii) determine the ep-EROI —based on life cycle thinking— linked to the extraction of the different species; and iii) perform a critical assessment of the similarities and differences between species and fishing techniques in terms of EROI to understand and rank their energy efficiency.

8.2 Materials and Methods

8.2.1 Cumulative energy demand (CED) calculation

The current research originated out of the interest in assessing the environmental profile of a set of fishing fleets in Galicia (NW Spain) using LCA. Hence, the available data were collected from a series of existing publications in the field of seafood LCA (Vázquez-Rowe et al., 2010a, 2011a, 2012b, 2012c). A total of 98 vessels, representing a significant proportion of their specific fleets, were included in the present study, as can be observed in Table 8.1. Primary data were gathered through the use of personal questionnaires filled out by Galician skippers. These included a broad range of operational issues, such as diesel consumption, use of raw materials (e.g. bait), material provision (fishing nets, anti-fouling paints, etc.) and caught species, whilst the ecoinvent® v2.2 database was used for background processes (Frischknecht et al., 2007).

LCA and its life cycle perspective provide useful data and results for EROI calculation. A group of studies have highlighted two important flaws when it comes to EROI estimation: setting the system boundaries to analyze the energy requirements of the assessed system and the fact that EROI is highly dependent on monetary data (Hall, 2011). However, its combination with LCA, which has standardized mechanisms regarding data acquisition and system boundary delimitation, allows reducing these uncertainties.

Table 8.1. Selected Galician fishing fleet samples for ep-EROI calculation

	F1	F2	F3	F4	F5	F6	F7
Sample size	30	24	9	12	9	5	9
Percentage over total (%)	18.2	23.8	14.3	20.7	33.33	6.4	24.3
Year of inventory	2008	2008	2008	2008	2009	2009	2000-2004
Total landings (tons)	12,597	16,056	3,769	3,416	5,000	668	72,000
Total captures (tons)	12,998	27,750	6,657	3,473	6,213	727	N/D

F1 = coastal purse seining; F2 = coastal trawling; F3 = offshore trawling; F4 = offshore long lining (Northern Stock); F5 = trawling (Mauritania); F6= offshore long lining Azores; F7= tuna purse seining. N/D= no data available.

In particular, one single impact category, CED, was selected from LCA for its combined use with ep-EROI (Figure 8.1), in order to calculate the energy requirements of the selected production systems (VDI-Richtlinien, 1997). SimaPro 7.3 was the software chosen for computing the results (Prè-Product Ecology Consultants, 2011).

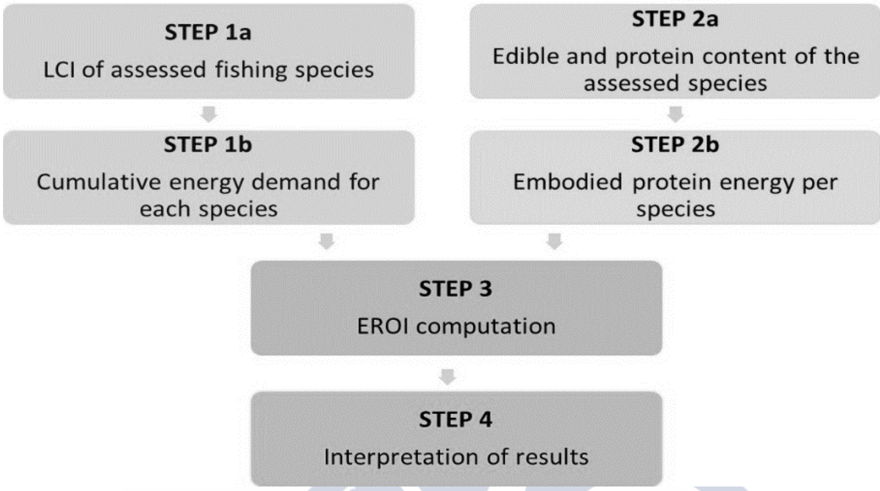

Figure 8.1.Graphical representation of the selected methodology

The system under study in the LCA analysis included the different stages considered for fish extraction performed by the fishing fleets selected (Figure 8.2). Hence, the products were followed from the production of the supply materials to the landing of fish at a Galician port, constituting what in LCA is named a "cradle to gate" perspective (Guinée et al., 2001). Further sections of the supply chain, such as processing or were disregarded due to the lack of data availability for all the supply chains.

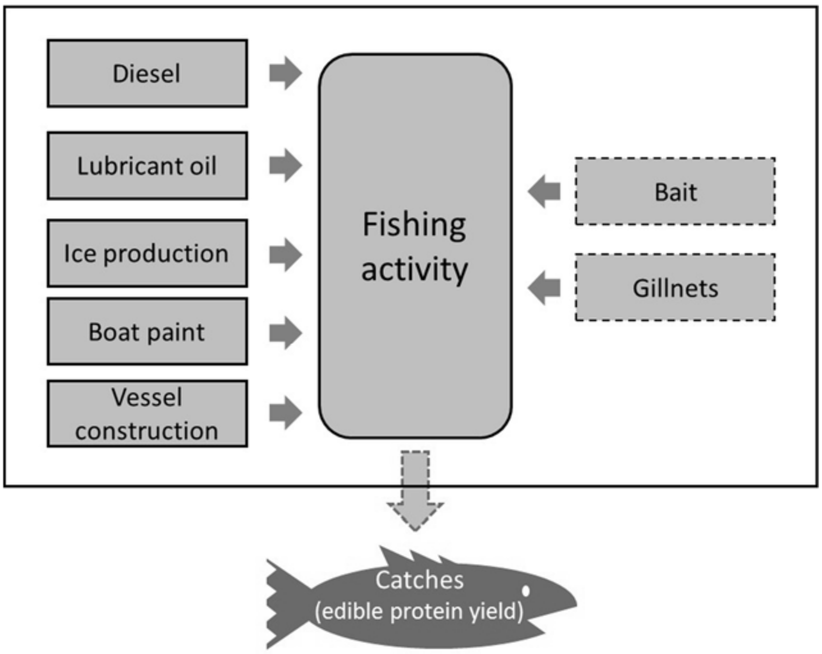

Figure 8.2. Graphical representation of the system boundaries for the coupled LCA+EROI approach. NOTE= grey rectangular boxes represent operational inputs of the fishing fleets assessed; dotted boxes represent operational inputs that are not common to all fishing fleets

While the use of LCA to calculate the total industrial energy expended in the extraction of fish products is considered highly precise, since it embraces not only the direct expenditures, but also the background processes, LCA studies usually exclude human labor from the assessment (Nebel et al., 2006; Rugani et al., 2012). However, recent literature has started questioning this perspective, based on the assumption that human activities constitute an integral part of production systems, since part of these systems are sustained by human life (Rugani et al., 2012). However, given the lack of data, human labor activities, unlike in other EROI calculation approaches available in literature, were excluded from the present system.

The main function of the analysis is the estimation of the potential edible protein energy return on investment of the main fish species landed in Galician ports. The FU considered for each of the different marine species in the different fishing fleets was 1 ton of landed fish at port. Mass allocation was considered (Pelletier and Tyedmers, 2011; Svanes et al., 2011). The rationale behind this decision, rather than assuming an economic allocation perspective, was based on the fact that the different fishing fleets evaluated operate in multi-species fisheries in which the catch of one species does not imply an increased strategic value over the rest of the landed species. Moreover, the use of alternative biophysical allocation approaches, such as energy density, was discarded due to the system boundaries delimited in this study, since they only consider the round weight landing of the fishing species at port without any on-land processing. Hence, the calculations provided in this study assume that all the edible content of the analyzed systems will be used for human nutritional purposes.

8.2.2 Quantification of the Energy Output

The embodied energy was calculated based on the maximum edible content of the inventoried species and the protein content per 100 g of edible portion per FU. The specific values for each of the species were retrieved from a database developed and provided by the School of Resources and Environmental Studies (SRES) at Dalhousie University (Peter Tyedmers, personal communication). This database included the mean edible content and the protein content for 21 of the 25 marine species included in the assessment. Data for megrim and anglerfish, which were not available from this database, were retrieved from a national Spanish database (MAGRAMA, 1998). Moreover, data for forkbeard (*Phycis* spp.) and blackbelly rosefish (*Helicolenus dactylopterus*), two species that were landed by the offshore long lining fleet (NEAFC[2], ICES Divisions VIIIa, b, d and VII) were not included due to lack of available data regarding these two parameters. Finally, it should be noted that the energy content of protein was assumed to be 16.73 MJ/kg of protein (FAO, 1985).

[2] The North Atlantic Fisheries Commission (NEAFC) is one of the most abundant fishing areas worldwide. Its embraces from southern tip of Greenland, east to the Barents Sea, and south to Portugal.

It should be noted that the system boundaries for the calculation of the output energy were fixed in the same terms as that of the CED calculation (Murphy and Hall, 2010). Therefore, the assumption was made that all the edible protein energy content in the species assessed when landed at the fishing ports is finally delivered for human consumption, disregarding the potential food wastes that may occur in the on-land supply chains (FAO, 2011). Discards, which are discussed in more detail in the discussion, were excluded from the computation of the results as recommended by PAS2050-2:2012 (BSI, 2012).

8.2.3 Edible protein content energy return on investment (ep-EROI) calculation

Finally, ep-EROI estimation was accomplished through calculating the coefficient between the protein energy output of the selected marine species and the energy inputs linked to fish extraction (CED impact category), following the formula:

[1] *ep-EROI= (energy inputs/energy outputs)* (adapted from Tyedmers, 2000; Hall et al., 2009)

8.3 Results

As abovementioned, a total of 23 different species, belonging to seven different industrial Galician fishing fleets, were assessed in the current study, as can be observed in Table 8.2.

Table 8.2. Edible protein energy return on investment (ep-EROI) values for the selected seafood products (CED, energy output and ep-EROI results are referred to the selected FU: 1 tonne of landed fish)

Fishing fleet	Species	Scientific name	CED	Edible meat	Protein content	Energy output (MJ)[a]	ep-EROI (%)
Coastal purse seiners (F1)	Horse mackerel	*Trachurus trachurus*	11,532	52%	0.198	1,722.5	**14.9**
	Atlantic mackerel	*Scomber scombrus*	11,532	61%	0.201	2,051.3	**17.8**
	European pilchard	*Sardina pilchardus*	11,532	62%	0.203	2,105.6	**18.3**
Coastal trawling (F2)	European hake	*Merluccius merluccius*	28,124	53%	0.178	1,578.3	**5.6**
	Horse mackerel	*Trachurus trachurus*	28,124	52%	0.198	1,722.5	**6.1**
	Atlantic mackerel	*Scomber scombrus*	28,124	61%	0.201	2,051.3	**7.3**
	Blue whiting	*Micromesistius poutassou*	28,124	56%	0.174	1,630.2	**5.8**
Offshore trawling (F3)	European hake	*Merluccius merluccius*	117,656	53%	0.178	1,578.3	**1.3**
	Anglerfish	*Lophius budegassa*	117,656	50%	0.149	1,313.3	**1.1**
	Megrim	*Lepidorhombus* spp.	117,656	49%	0.181	1,483.8	**1.3**
	Norway lobster	*Nephrops norvegicus*	117,656	30%	0.188	943.6	**0.8**
Offshore long lining (F4)[b]	European hake	*Merluccius merluccius*	73,471	53%	0.178	1,578.3	**2.1**
	Common ling	*Molva molva*	73,471	65%	0.190	2,066.2	**2.8**
	Conger eel	*Conger conger*	73,471	58%	0.181	1,756.3	**2.4**
	Atlantic pomfret	*Brama brama*	73,471	56%	0.165	1,545.9	**2.1**
Trawling – Mauritania (F5)	Common octopus	*Octopus vulgaris*	95,925	68%	0.173	1,968.1	**2.1**
	European squid	*Loligo vulgaris*	95,925	71%	0.156	2,006.4	**2.1**
	Common cuttlefish	*Sepia officinalis*	95,925	63%	0.179	1,886.6	**2.0**
	Common sole	*Solea solea*	95,925	49%	0.181	1,483.8	**1.5**
	Sand sole	*Pegusa lascaris*	95,925	60%	0.182	1,826.9	**1.9**
	Senegal hake	*Merluccius senegalensis*	95,925	53%	0.163	1,445.3	**1.5**
	Caramote prawn	*Penaeus kerathurus*	95,925	57%	0.204	1,954.9	**2.0**
Offshore long lining – Azores (F6)	Swordfish	*Xiphias glaudius*	77,336	68%	0.180	2,047.8	**2.6**
	Porbeagle	*Lamna nasus*	77,336	51%	0.199	1,697.9	**2.2**
	Blue shark	*Prionace glauca*	77,336	51%	0.174	1,484.6	**1.9**
	Bigeye tuna	*Thunnus obesus*	77,336	57%	0.231	2,202.8	**2.8**

Table 8.2. Edible protein energy return on investment (ep-EROI) values for the selected seafood products (CED, energy output and ep-EROI results are referred to the selected FU: 1 tonne of landed fish) (continuation)

Fishing fleet	Species	Scientific name	CED	Edible meat	Protein content	Energy output (MJ)[a]	ep-EROI (%)
Open-sea tuna purse seining (F7)[c]	Skipjack/Yellowfin[d]	*Thunnus* spp.	19,225	62.5%	0.238	2,488.6	**8.9**
	Skipjack/Yellowfin[e]	*Thunnus* spp.	23,950	62.5%	0.238	2,488.6	**10.4**
	Skipjack/Yellowfin[f]	*Thunnus* spp.	28,085	62.5%	0.238	2,488.6	**12.9**

[a] A generic value of 16.73 MJ per kg of protein was assumed for the energy content of protein (FAO, 1985).
[b] This fleet also lands significant amounts of forkbeard (*Phycis spp.*) and blackbelly rosefish (*Helicolenus dactylopterus*), but were excluded from the assessment due to lack of data regarding edible content and protein content.
[c] A landing ratio of 50% for skipjack and 50% for yellowfin was assumed for the three fleets. Therefore, edible and protein content values represent medium values.
[d] Skipjack and yellowfin caught in the Atlantic Ocean.
[e] Skipjack and yellowfin caught in the Indian Ocean.
[f] Skipjack and yellowfin caught in the Pacific Ocean.

8.3.1 Coastal fishing species

A total of five different fishing species were assessed in the Galician coastal fishery. The results of ep-EROI ranged from 5.6% in the case of European hake (trawling fleet) to 18.3% for European pilchard (purse seining fleet). In fact, all species captured by purse seining vessels showed similar trends in their ep-EROI rate. Similarly, in the trawling fleet the collection of ep-EROI values was relatively small, ranging from 5.6% (European hake) to 7.3% (Atlantic mackerel). It is important to highlight the fact that two pelagic species, Atlantic mackerel and horse mackerel were captured indistinctively by the two fleets. Interestingly, direct comparison between fleets shows an 8.8% decrease for horse mackerel and a 10.5% reduction for Atlantic mackerel when these species are caught with trawl nets. Finally, the average ep-EROI for the evaluated coastal fishing fleets was 9.2%.

8.3.2 Offshore fishing species

Four different offshore fleets were evaluated in the current study. While they are all linked to highly variable characteristics regarding the fishery or the fishing technique, ep-EROI values showed limited variance, ranging from 0.8% (Norway lobster – Northern Stock trawling) to 2.8% (Big eye tuna – Azores long lining fishery). This led to an average ep-EROI of offshore fleets of 1.6%. Nevertheless, despite the highly distinct characteristics of the assessed fishing fleets, a series of parallelisms were set.

In the first place, in the Northern stock the long lining fleet showed higher ep-EROI values than the trawling fleet. In fact, European hake, which was the only species that is captured and landed by both fleets, presented ep-EROI values 62% higher for the long lining fleet. Secondly, if we compare the two long lining fleets, despite the fact that they target completely different species in different geographical areas, the range of values is relatively similar for all the species. Finally, if the two trawling fleets are compared, slightly higher EROI figures are observed for the Mauritanian fleet, which mainly targets cephalopods.

8.3.3 Open-sea fishing species

Three different tuna purse seining fleets were included in this section. The ep-EROI values ranged from 8.9% for the Pacific Ocean tuna captures and 12.9% for tuna extracted in the Indian Ocean. The average value for the three pelagic fleets was 11.2%. For this particular case study, the changes in ep-EROI were mainly linked to the CED of each fishing fleet, since the caught species were in all three fleets the same.

8.3.4 Fishing gear and other correlations

Fishing vessels relying on passive fishing gears were those that showed the highest ep-EROI values regardless of the geographic location. Consequently, species captured by purse seiners showed in most cases values above 10%, substantially higher than coastal trawlers, the following fleet in terms of ep-EROI. However, the other two trawling fleets presented the lowest return values. No apparent correlation was detected between the ep-EROI and the trophic level of the inventoried species.

8.3.5 Galician extractive fishing activity

Based on the ep-EROI values estimated for each of the different fishing fleets, the mean ep-EROI was calculated for the entire Galician extractive fishing sector (Figure 8.3). Nevertheless, due to lack of available data, this approximation was calculated on the basis of industrial fishing fleets in the region and, therefore, excluding the still prominent small-scale fishing fleet. Thus, a total of 7,053 TJ were invested by the industrial fishing fleet to perform their activities in the assessed period, with a return of the edible-protein energy by the captured species of 535.9 TJ. Hence, the average estimate for the ep-EROI value for the Galician industrial fishing fleet was 7.6%.

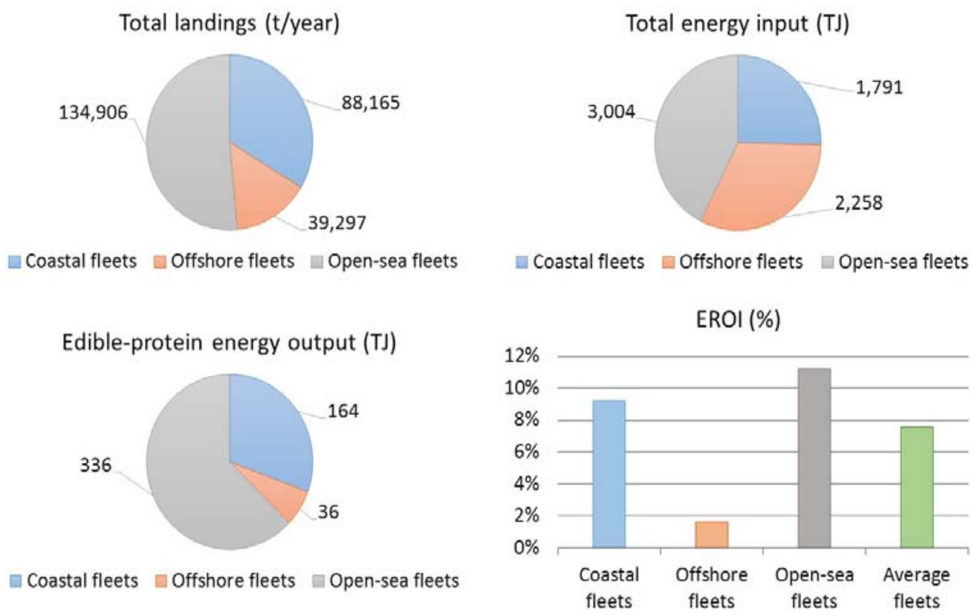

Figure 8.3. Average edible protein EROI values per fishing zone and average edible protein EROI for Galician industrial fishing fleets

In an attempt to provide an average worldwide edible-protein energy return, Tyedmers et al. (2005) calculated a mean value of 8% for world fisheries. However, it must be noted that this estimation was based on the FUI of the assessed fishing fleets. Consequently, the estimate provided by Tyedmers et al. (2005) does not take into consideration the energy consumption linked to a set of background processes. Therefore, despite the worldwide estimate being 0.4% higher than the approximation for the Galician fleet, the fact that non-fuel energy inputs are disregarded for worldwide fisheries would imply that a correction factor using life cycle standardized assumptions would lower this value considerably.

8.4 Discussion

8.4.1 The importance of EROI estimation in Galician fisheries

The analysis of ep-EROI of the assessed species constitutes, as far as we can ascertain, the first study of these characteristics performed for the Galician fishing fleet. In fact, the results allow a deeper understanding of the energy efficiency in the Galician fishing sector, since the use of ep-EROI in combination with LCA provides a more comprehensive approach to the energy requirements of fishing systems than other commonly used indicators, such as FUI or other single issue indicators.

Unfortunately, the presented results represent the state-of-the-art during one year of fish extraction. Hence, the common finding that EROI values tend to decrease through time —law of diminishing returns— could not be tested with this specific sample (Tyedmers, 2001; Pracha and Volk, 2011), but may be used in the future for such purpose when new data are collected. Nevertheless, the results provide valuable information regarding energy requirements of the Galician industrial fishing fleets. First, the revision of the Common Fishery Policy (CFP) by European authorities in 2013 has to be approached as an opportunity not only to improve the EU's management of fishing stocks, but also as a chance to reduce the energy requirements of many fisheries through decision-making. In fact, this perspective would also offer a potential scenario for the reduction of the carbon footprint and other environmental impacts related to European fisheries (Vázquez-Rowe et al., 2011a, 2011b). Next, an important milestone in the development of fishery ep-EROI studies is the fact that, unlike when these first arose in the 1970s, there has been an increasing perception by stakeholders regarding the importance of energy due to increasing fuel prices and more rigid fisheries management regulations. Finally, energy ratios such as EROI have provided important information regarding the imbalances between the increase in human populations and their thirst for energy and the use of natural resources (Pimentel and Pimentel, 2006). Therefore, the global perspective that is provided in this case study for a relevant European fleet will allow future study of time-dependent development of these fisheries in terms of energy investment.

However, the analysis of ep-EROI in this case study must also be performed at a global scale. According to a set of influential publications that have been published in the last 20 years, ecological economists forecast that the so-called "petroleum age" will end in the first few decades of the 21st century, although recent advances in technology and drilling may slightly postpone this deadline (Meng and Bentley, 2008; Almeida and Silva, 2009). This situation will inevitably generate an increase in oil prices driven by a predicted increase in demand, in which the steady increase of renewable fuel production in some Western nations will not compensate the exponential increase in demand in emerging countries, as opposed to dwindling fossil fuel production (Hall and Ko, 2005; Day et al., 2009).

The predicted increase in fossil fuel prices in the following decades will most likely affect the economical sustainability of a traditionally heavily subsidized primary sector in Spain and Europe: fishing fleets (Pauly et al., 2003). Therefore, given that the highest energy expenditures in industrial fisheries are linked to direct energy inputs —75%-90% (Tyedmers, 2004; Vázquez-Rowe et al., 2011b), it seems feasible to think that a change in the energy carrier for industrial fishing fleets will be needed in spite of predicted improvements in technology and energy efficiency (Destouni and Frank, 2010). However, a recent study suggests that the use of alternative energy carriers (i.e. biofuels) for marine transportation would increase the primary energy use to fuel vessels, as well as a series of environmental impacts such as eutrophication or land use impact, despite the reduction in fossil fuel dependence and in climate change impacts (Bengtsson et al., 2012). Moreover, estimations concerning renewable energy production in the short- and mid-term do not indicate that the renewable mix may deliver current energy levels alone (Hirsch et al., 2005; Day et al., 2009).

Based on this reasoning, it seems plausible to predict a scenario in which those fisheries with lowest ep-EROI values will eventually encounter most difficulties to access competitive fuel prices to continue their activities. In fact, the low penetration of energy carrier modifications in the Galician fishing sector suggest that those fleets with the lowest ep-EROI values will run into serious difficulties to maintain their competitiveness. More specifically, recent publications suggest that a major consequence of this future scenario may be the need to rely on small pelagic species for direct human consumption (Pauly et al., 2003), which, as can be seen in Table 8.2, show considerably higher ep-EROI values than any other fishery assessed if captured using specific fishing techniques. Having said this, it is important to mention that small pelagic species tend to show strong annual variations in stock abundance (Pauly et al., 1998; Fréon et al., 2008). Therefore, further fishing pressure on these stocks may increase their vulnerability (Allison et al., 2009).

8.4.2 Comparison between the assessed fishing fleets

As mentioned in section 8.3.4, a high correlation was identified between the use of purse seining gears for seafood extraction and higher ep-EROI rates. While this fact is visible by direct comparison between the different fishing fleets, it gains more relevance when horse mackerel and Atlantic mackerel, two species captured along the Galician coast, are examined in detail. These two species, captured in the same fishing areas by trawlers and purse seiners, not only show important ep-EROI improvements when captured by purse seiners, but this enhanced energy return, as observed in Figure 8.4, is also accompanied by low potential environmental impact values (Vázquez-Rowe et al., 2010a) and lower discards, which also contribute to a reduced depletion of biotic resources (Vázquez-Rowe et al., 2012b).

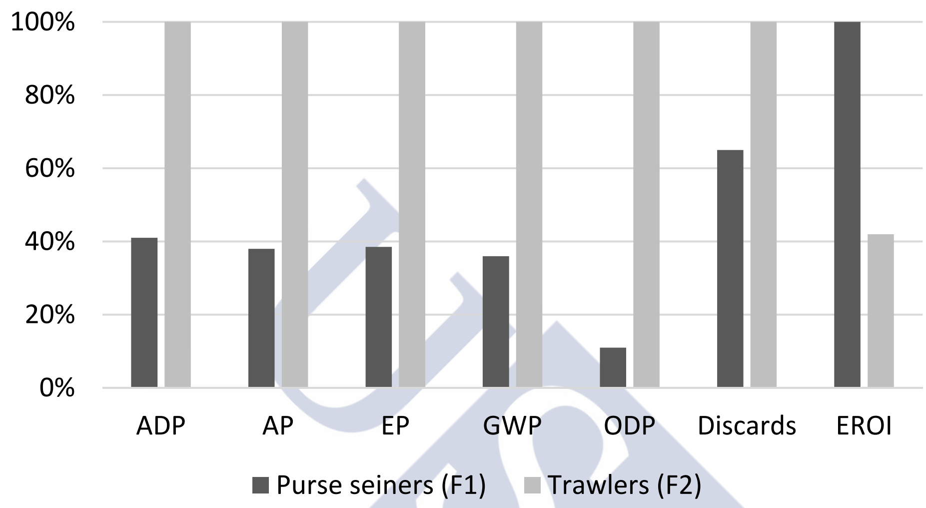

Figure 8.4. Comparison of horse mackerel (Trachurus trachurus) captured by two coastal fishing techniques in Galicia (Source: Vázquez-Rowe et al., 2010, 2012b)

Interestingly, a set of publications using the "LCA+DEA method" approach, which combines environmental LCA with data envelopment analysis (DEA), a linear programming method, suggest that fleets with increased energy demand have a higher operational efficiency and, therefore, less room for minimization of operational inputs and environmental impact reductions (Vázquez-Rowe et al., 2010b, 2011b).

Subsequently, these results exemplify the need for a redefinition of the CFP in terms of balancing the available natural resources with the most convenient fishing techniques in terms of fishing efficiency and environmental performance (Villasante and Sumaila, 2010). Moreover, it should be noted that the lack of standardized social indicators in the sustainability assessment of fisheries is a current constraint in order to attain a comprehensive evaluation of fishing systems, since fishing can be an important source of employment in numerous coastal areas (Cocharne, 2000; Iribarren and Vázquez-Rowe, 2013). Nevertheless, the on-going development of the social life cycle assessment (SLCA) may constitute a promising framework for future studies (Labuschange and Brent, 2006; Jørgensen et al., 2009; Dreyer et al., 2010).

Finally, similar results were also found for the hake fishery in the Northern Stock (Vázquez-Rowe et al., 2011a). However, in this particular case the considerable environmental gains of fishing hake with long lining techniques rather than trawlers was not complemented with an important difference in energy return (Table 8.2).

At this stage it is important to highlight the fact that life cycle perspective studies carried out traditionally account for an attributional perspective, that is, consider production systems as steady-state systems. However, in the last decade consequential life cycle assessment (C-LCA) has developed as a more comprehensive approach in which the consequences of a variation that affect the initial system are analyzed (Schmidt and Weidema, 2008). In fact, in many cases the predicted future change scenarios are computed through a selection of economic models (Dandres et al., 2012). While this particular study has only assessed the energy inputs of industrial fleets from an attributional perspective, the analysis would gain in comprehensiveness if C-LCA were implemented, in order to detect the marginal consequences of fisheries management decisions. Moreover, the validity of a C-LCA approach is also extendable to the computation of EROI, provided that the production system follows a life-cycle inventory perspective, in order to determine the energy returns related to shocks in the structure of these systems.

8.4.3 Contextualization of the results and comparison with other sources of protein

The use of ep-EROI to assess the energy expended in the acquisition of seafood has been developed in many other fisheries. Table 8.3 provides a small summary of available ep-EROI values in literature. For instance, it should be noted that the ep-EROI values for the purse seining fleets included in this study are substantially lower than most of those collected in literature

(Tyedmers, 2001). Nevertheless, many of these fisheries are linked to landings that are destined for fishmeal production, rather than direct human feeding, the latter being the case in the Galician fleets assessed.

Table 8.3. ep-EROI values for other fishing, aquaculture and agriculture food products

FISHING				
Species	**Country**	**Fishing gear**	**Year**	**EROI (%)**
Menhaden	United States	Purse seining	1999	167[a]
Atlantic mackerel	Basque Country (Spain)	Purse seining	2001-2008	68.6[b]
Tuna	Global	Purse seining	2009	14.0[c]
Patagonian grenadier	Galician fleet in Chile	Trawling	2010	10.4[d]
Global fisheries	--	Varied	--	8.0[e]
Tuna	Global	Long lining	2009	5.9[c]
Shrimp	Canada	Trawling	1999	4.1[a]
Swordfish	Canada	Long lining	1999	3.4[a]
AQUACULTURE				
Species	**Country**	**Type**	**Year**	**EROI (%)**
Mussels	Scandinavia	Extensive	N/Sp	10-15[a]
Mussels	Spain	Extensive	2007	6.9[f]
Shrimp	Thailand	Intensive	N/Sp	1.4[g]
LIVESTOCK				
Species	**Country**		**Year**	**EROI (%)**
Chicken	United States		N/Sp	25[h]
Swine	United States		N/Sp	7.1[h]
Pork	Spain		2011	3.9[i]
Beef cattle	United States		N/Sp	2.5[h]
Milk	Spain		2002	7.2[j]
Lamb	United States		N/Sp	1.8[h]

[a]Tyedmers, 2001; [b]Ramos et al., 2011; [c]Parker and Tyedmers, 2012; [d]Vázquez-Rowe et al., 2013; [e]Tyedmers et al., 2005; [f]Iribarren et al., 2010; [g]Troell et al, 2004; [h]Pimentel and Pimentel, 2003; [i]Noya et al., submitted for publication; [j]Hospido et al., 2003.

Galician offshore trawling and long lining fisheries, as far as we could ascertain through a comprehensive set of publications, constituted the fleets with the lowest ep-EROI values, which highly questions the efficiency of these fleets in terms of energy return. This situation highlights the vulnerability of these fleets in a predicted energy-scarce future (Day et al., 2009).

Finally, large-pelagic species caught by large Galician purse seiners around the world show similar ep-EROI values to the average for global tuna purse seining fisheries. Moreover, the ep-EROI of tuna species captured with purse seiners is substantially higher than those landed with long lining or pole and line gears (Parker and Tyedmers, 2012). However, no specific data were available in this case study for the Galician pole and line tuna fishery.

When the computed values in this study are compared to other sources of protein, it is important to highlight that only swine and milk were available sources of protein produced in Galicia, which limits the comparability of the products (Table 8.3). Nevertheless, literature data suggests that livestock products, such as lamb or beef, and intensive aquaculture products, have similar ep-EROI levels to fish species caught with trawling and long lining techniques, while extensive aquaculture products (e.g. mussels) or poultry showed ranges comparable to pelagic species captured with purse seiners.

8.4.4 Methodological choices affecting EROI calculations

Despite the utility of EROI when it comes to compare fleets from an energetic perspective, the methodology assumptions that are taken into account can cause a set of uncertainties that must be considered and discussed. In the first place, all ep-EROI estimates must be seen as overestimates, since the current study only computed the extraction phase of seafood supply chains. Hence, the decrease in ep-EROI when considering the entire supply chain up to consumption will depend on a set of multiple factors, such as the level of processing, storage conditions, transportation or the food waste that is generated throughout the supply chain. The lack of data and the complexity of on-land seafood supply chains impeded the expansion of the system boundaries of the assessment to these stages, such as processing, retailing or consumption. The following Chapters of this dissertation delve into seafood processing and consumption. Nevertheless, a small number of different production systems have been followed to the consumption stage (Figure 8.5). The results show an important decrease in the ep-EROI for those products with medium values (5%-10%), while those with low ep-EROI values show limited reductions due to the high overall contributions of the fishing activities. In fact, all seafood products arriving from industrial fishing fleets available in literature have reported the highest energy inputs taking place in the fishing phase (Vázquez-Rowe et al., 2011b; Ziegler et al., 2011).

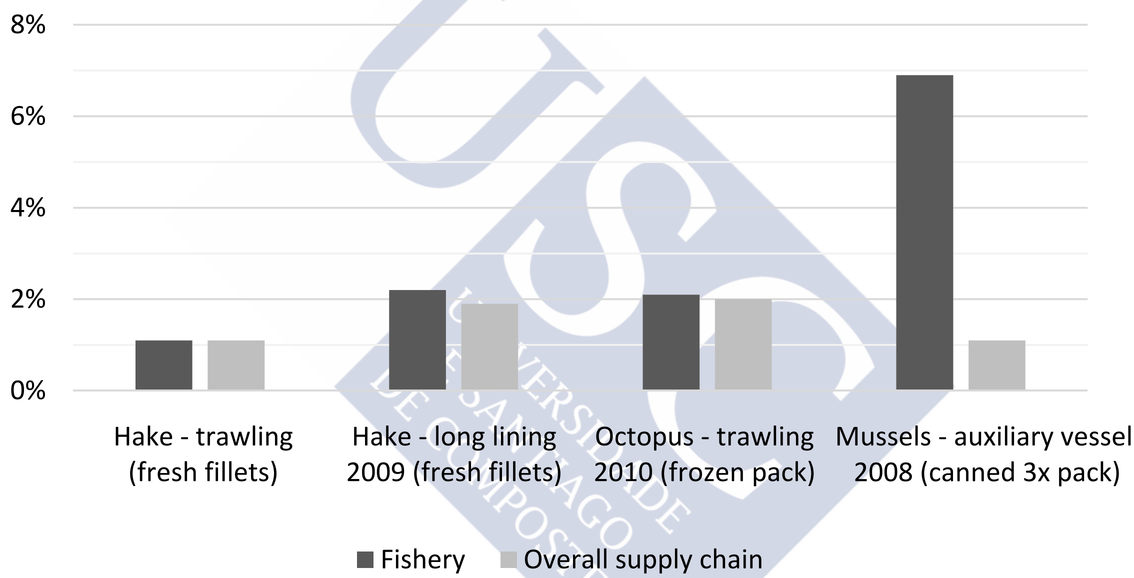

Figure 8.5. ep-EROI of selected species throughout their supply chain. Source: Iribarren et al. (2010); Vázquez-Rowe et al. (2011a); Vázquez-Rowe et al. (2012c)

Secondly, allocation constitutes an important source of result variations in life cycle studies in multispecies fisheries. For instance, as observed in Table 8.4, the variation in EROI results with respect to mass allocation when an economic allocation approach is taken into consideration will vary depending on the differing prices of the landed species. Nevertheless, it should be highlighted that changes in the allocation approach will not change the global EROI results for a single production unit (i.e., fishing vessel or fishing fleet).

A third source of uncertainty that was identified was the high standard deviations in terms of operational inputs observed between individual fishing vessels in some of the fleets assessed. Hence, as can be seen in Table 8.5, the range of ep-EROI values for species caught by coastal purse seiners can show ranges of up to 250% from the best to the worst performing scenario, whereas the range for coastal trawlers was identified as substantially smaller. Having said this, it should be noted that other sources of uncertainty, linked to process energy requirements or illegal and unreported fishing by vessels may also be important sources of uncertainty in reporting ep-EROI values for these fisheries.

Table 8.4. Changes in final ep-EROI results using economic allocation

Fishing species	Fishing gear	ep-EROI (%) Mass allocation	ep-EROI (%) Economic allocation	Difference (%)
European pilchard	Purse seining	18.3	18.3	-0.1
European hake	Trawling (offshore)	1.3	1.7	+30.8
Anglerfish	Trawling (offshore)	1.1	0.9	-18.2
Common octopus	Trawling	2.1	2.1	+0.3

Finally, in terms of the system boundaries of the fishing phase, as shown in Figure 8.1, unutilized captured species (i.e. discards) were not accounted for. On the one hand, this constitutes a logical perspective from an output perspective, since discards return, in many cases dead, to the ocean, so their computation in terms of energy as an output of the system would be deceiving. On the other hand, the large amounts of discards that are produced in fisheries have been confirmed in previous studies for the inventoried fishing fleets (Vázquez-Rowe et al., 2011c, 2012b). In fact, Vázquez-Rowe et al. (2011c) estimated 60,255 tons of discards in the Galician fishing sector in 2008, representing 17% of the total capture, which could translate into approximately 120 TJ of wasted embodied energy. Moreover, it is important to note that many of the species that Galician fishermen reported discarding, such as horse mackerel, are those with higher levels of embodied energy (Vázquez-Rowe et al., 2012b). Consequently, a future improvement in ep-EROI computation may consider the loss of discards and other food wastes throughout the supply chain as a source of inefficiency when calculating the embodied energy of the fish yield, therefore lowering the ep-EROI values of operations with discards. Similarly, this perspective could also be applied in those fleets that use bait for extractive operations.

Table 8.5. Sensitivity analysis for selected species based on the upper and lower standard deviation values of life cycle inventory items

Fishing species	Fishing fleet	ep-EROI (%)	ep-EROI (%) worst scenario	ep-EROI (%) best scenario
Horse mackerel	F1	14.9	10.7	24.5
Pilchard	F1	18.3	13.1	30.0
Atlantic mackerel	F1	17.8	12.8	29.2
European hake	F2	5.6	4.7	7.0
Atlantic mackerel	F2	7.3	6.1	9.1
Horse mackerel	F2	6.1	5.1	7.6
Blue whiting	F2	5.8	4.8	7.2

F1= coastal purse seiners; F2= coastal trawlers.
The standard deviation of the life cycle inventory of the two selected fishing fleets was based on a sample of 30 fishing vessels for F1 and 24 vessels for F2.
The worst scenario of ep-EROI computation considers the upper standard deviation values for operational inputs in the life cycle inventory.
The best scenario of ep-EROI computation considers the lower standard deviation values for operational inputs in the life cycle inventory.

8.4.5 Transfer of technology in the fishing sector

The findings obtained through this study regarding ep-EROI calculation and Galician fleet efficiency were implemented in the registered trademark *pescaenverde®*[3]. This trademark

[3] *pescaenverde®* is a registered trademark by the Spanish Patent and Trademark Office (Guarantee Trademark Number: 3.075.258). For further information: http://www.usc.es/pescaenverde/es

develops software to calculate ep-EROI and Carbon Footprint indicators of fishing fleets, aiming at providing useful information to improve the management of fisheries, by increasing fuel efficiency and reducing GHG emissions related. Additionally, *pescaenverde®* can be a communication tool of the environmental performance of fresh fish to consumers.

8.5 Conclusions

Increasing seafood demand due to human population growth constitutes a main risk in terms of guaranteeing the sustainability of fish stocks. Moreover, current energy consumption patterns will have to be reduced on a mid-term basis, due to depleting fossil fuel resources. Therefore, industrial fishing fleets which are strongly dependent on these two dwindling natural resources are bound to suffer an important reconversion in decades to come.

The use of ep-EROI arises as a useful indicator to monitor the fragile balance between edible energy that humans obtain from the oceans and the energy needed to power the fishing activities. The current study presented a wide range of ep-EROI values for the most representative fishing species captured by industrial fishing vessels in Galicia. While these results constitute the first of its kind for this specific case study, they are intended to provide useful information for the future management of these fisheries. Additionally, the life-cycle perspective given to EROI calculation, through the use of LCA inventories, confers a solid and standardized methodology background to the study, allowing the reproducibility and comparability of the results. Finally, it also permits parallel computation of potential environmental impacts, which adds an environmental sustainability dimension to the evaluation of energy return on investment.

8.6 References

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Chapter 9

Carbon footprint of a multi-ingredient seafood product from a business-to-business perspective[1]

Summary

Greenhouse gas emissions associated with the production system of a wide range of fish products have been analyzed in literature. However, the evaluation of complex multi-ingredient products of marine origin from a carbon footprint perspective has remained unexplored. Therefore, this chapter assessed the environmental profile of hake fish sticks extracted in Chile and processed at a seafood industry near Vigo (NW Spain) for distribution throughout the Spanish market. Despite a brief discussion on the distribution and consumption life cycle stages, result reporting adopted a business-to-business approach, focusing on raw material production and fish stick processing and packaging. In spite of the increased processing stages that a product of these characteristics requires, as well as the on-land production systems for the agricultural products, fishery operations, mainly vessel fuelling and cooling agent leakage, were still responsible for over 50% of total emissions. Additionally, a series of improvement actions were identified in order to reduce the carbon footprint of the product. Finally, the fish sticks, when equated to other comparable substitutable production systems in terms of carbon footprint, energy use and energy return on investment, showed values in the lower range of previously analyzed processes.

[1] Vázquez-Rowe, I., Villanueva-Rey, P., Mallo, J., De la Cerda, J.J., Moreira, M.T., Feijoo, G., 2013. Carbon footprint of a multi-ingredient seafood product from a business-to-business perspective. J. Clean. Prod. 44, 200–210.

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9.1 Introduction

Seafood consumption in Spain is one of the highest on a European Union (EU) and worldwide scale. Most of this seafood is consumed fresh (almost 56%), while 14% is consumed as canned seafood and 30% is sold as frozen or boiled products (Martín-Cerdeño, 2010). Fresh consumption of seafood remains high in Spain, due to a series of factors that include high accessibility to fresh products, thanks to a large fishing fleet, an increasing coastal population and the strong fish tradition that still remains in most households (FROM, 2010). However, statistics also show a decline in fresh seafood consumption in recent years in most Western nations (including Spain) due to changing diet habits (Manrique and Jensen, 2001), giving way to a higher demand for frozen ready meal products (Jensen, 2006; FAO, 2007; Calderón et al., 2010).

Several species of hake constitute the main seafood consumed in Spain, which contrasts with seafood consumption habits in other Western countries, where the main consumed groundfish are cod, halibut or pollock (FAOSTAT, 2011). In 2009, each Spaniard consumed 3.9 kg of hake (Martín-Cerdeño, 2010), mainly consumed fresh or in frozen fillets, with 89% of households consuming fresh hake and 50% frozen hake in 2005 (FROM, 2005). Hence, it is not surprising that the main seafood processing companies in Spain are using hake species to introduce multi-ingredient frozen seafood products, such as fish sticks or breaded or battered fillets. However, European hake (*Merluccius merluccius*), captured in the Northern and Southern stocks in the North-Eastern Atlantic, is still introduced in the market for fresh or frozen consumption. Therefore, most processing companies obtain their hake supply from other Merlucciidae family species, such a Patagonian grenadier (*Macruronus magellanicus*) in Chile or Southern hake (*Merluccius australis*) in Argentina (Villasante et al., 2007).

This chapter aims at assessing the environmental profile of hake fish sticks produced in a processing plant near Vigo (NW Spain) for distribution throughout the Spanish market. More specifically, given the interest that has risen in the Galician fishing sector between skippers, retailers, consumers and other stakeholders with respect to studies that suggest that greenhouse gas (GHG) emissions of the fish extraction and aquaculture sectors in this region account for approximately 3% of total GHG emissions (Iribarren et al., 2010b, 2011), the environmental assessment in this particular study was limited to the estimation of the carbon footprint (CF) of the analyzed product. LCA methodology is implemented in this chapter to evaluate the capture, transport, processing and packaging of the fish sticks, including the assessment of other ingredients used in their elaboration, to detect the main environmental hot spots of the entire system in terms of CF, suggesting possible environmental burden mitigation schemes. Furthermore, results are expected to serve as a reference set for future assessments of seafood-based multi-ingredient products in the seafood processing industry.

9.2 Materials and methods

9.2.1 Methodological framework

Despite LCA limitations when accounting for the biological sustainability in fishing systems, such as stock assessment, its utility in other dimensions of environmental assessment has been employed by LCA practitioners to evaluate an increasing number of seafood products (Pelletier et al., 2007; Vázquez-Rowe et al., 2012a). The estimation of the CF, as previously mentioned, probably constitutes the non-fishery specific aspect (i.e. unrelated directly to stock assessment or ecosystem dynamics) of environmental evaluation of seafood products that is currently drawing most attention (Vázquez-Rowe et al., 2012a; Weidema et al., 2008; Ziegler et al., in press).

Despite the fact that ISO 14067, which provides specific standards for CF implementation, has already been released, it was not available when this study was conducted. Therefore, as many studies on this field, several approaches have been used for carbon foot-printing while awaiting for its release. For instance, the British standards (BSI, 2011) have been used in several food CF publications (Iribarren et al., 2010a, 2010b, 2011). However, in this study the ISO standards for LCA (ISO, 2006a, 2006b) were considered for CF calculation, given the higher flexibility in terms of controversial issues in life cycle perspective studies, such as allocation or inclusion/exclusion of capital goods (Iribarren et al., 2010b), allowing a deeper discussion regarding uncertainties in CF reporting (Ziegler et al., 2013). Moreover, ISO 14040 and 14044 offer a set of robust requirements for carrying out transparent CF calculations (European Commission, 2007), providing a solid basis for the development of other specifications that have arisen exclusively to assess the GHG emissions associated with products and systems, such as PAS 2050:2011, the GHG Protocol Product Life Cycle Accounting and Reporting Standard, and the forthcomings ISO 14067 and ISO 14069 (BSI, 2011; ISO, 2012a, 2012b; Initiative GGP, 2011).

9.2.2 Scope definition and functional unit

The production system included the production stages of the different ingredients (raw materials), their transport to the processing plant in Vigo and finally the processing and packaging phases undergone at the plant (Figure 9.1), constituting a "cradle to gate" analysis (Guinée et al., 2001). The exclusion of two stages: distribution of the products for sale and consumption, is based on the fact that no primary data were available for them. The selected functional unit (FU) for the studied product was set as one package of frozen fish sticks[2]. Acquired data for the study refer to year 2011.

9.2.3 Description of the system under study

The considered production system, as observed in Figure 9.1, is divided into four subsystems. Subsystems I–III are linked to the production and transportation of raw materials to supply the fish stick processing factory. Subsystem IV, on the contrary, deals with the activities and operations developed within the processing factory.

[2] One package contains 10 fish sticks, which correspond to approximately 320 g of edible product. The content of Patagonian grenadier of the fish sticks was 164.5 (50.9%).

Subsystem I

Subsystem I: Fishery. Patagonian grenadier is captured along the Chilean coast by a single fish processing vessel (FAO Area 87; Subarea 87.3 —see Chapter 2). Hake is then processed on board; producing a total of 5 different intermediate products derived from hake catches: fish blocks; fish blocks with SP; hake fillets with skin; headed, gutted and tail off (HGT) fish and fishmeal. The evaluated intermediate products, fish blocks, undergo a series of processing stages on board. In the first place, fish is headed and thereafter passed through a filleting machine. The following stage consists of separating the skin from the flesh using a wheel and a blade. Prior to the formation of the fish blocks, the flesh containing the few remaining bones is removed.

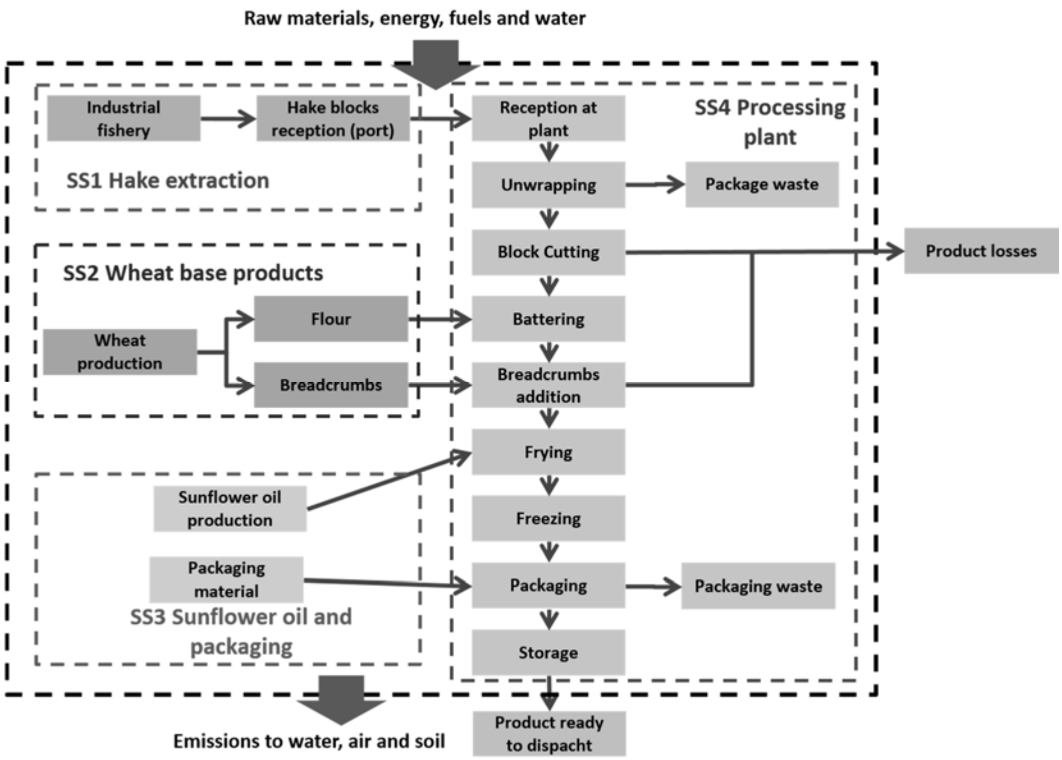

Figure 9.1. System under study for Patagonian grenadier fish blocks converted into fish sticks

Finally, the last processing activity on board implies the packaging and storage of the fish blocks in the cooling chamber. Regarding the organic residues derived from the filleting machines, they are processed to produce fishmeal on board. Fish blocks are landed at a Chilean port (Punta Arenas or Chacabuco) and then marine freighted to the port of Marín (Pontevedra province, Spain). Once they arrive at the port's premises, they are delivered in refrigerated lorries to the processing plant.

Subsystem II

Subsystem II: Wheat products. Flour used for battering and breadcrumbs is obtained from wheat arriving mainly from Ukraine through the port of Santander. Wheat is then grinded at a milling plant in Palencia (Castilla y León autonomous community) in order to produce wheat flour and thereafter it is transported by trucks to an ingredient and food additive processing plant in Vigo (Pontevedra province). At this point, two different products are prepared: breadcrumbs and flour for battering. The composition of both products can be seen in Table 9.1, as well as the origin and proportion of their ingredients.

Subsystem III

Subsystem III: Other raw materials. These include the provision of sunflower oil and other minor food ingredients, as well as the provision of packaging materials used at the processing plant.

Table 9.1. Composition and origin of breadcrumbs and wheat flour mix used in fish stick production

Ingredients	Content (%)	Origin	Previous processing
	Breadcrumbs		
Wheat flour	90	Palencia (Spain)	Grinding at mill
Corn dextrose	<5	Martorell (Spain)	--
Sunflower oil	<5	Jaén (Spain)	Sunflower oil production
Salt	<5	Alicante (Spain)	--
Yeast	<5	Valladolid (Spain)	--
Spices	<5	Denmark	--
	Wheat flour mix		
Wheat flour	80	Palencia (Spain)	Grinding at mill
Wheat starch	<10	The Netherlands	--
Salt	<10	Alicante (Spain)	--
Sunflower oil	<5	Jaén (Spain)	Sunflower oil production
pH regulators (E170i and E341iii)	<10	France	--
Vitamins A and C	<1	Switzerland/China	--

Subsystem IV

Subsystem IV: Processing plant. Wheat derived products arrive from the ingredient plant, which is located in the same industrial park as the processing plant, together with the fish blocks from the port. This subsystem includes all the on land processing stages, starting when the raw seafood product reaches the factory gate up to final storage ready for distribution. All other raw materials are received from the importing points. Subsystem IV can be divided into 9 differentiated phases:

- Reception at plant and storage. The hake blocks are stored in pallets in a cooling chamber on reception for up to 9 months until they are transported to the conveyor belt for processing. The rest of the products are stored in conventional areas.

- Unwrapping. The film surrounding the pallets is removed and the fishing blocks are situated on the conveyor belt where the first machine transports them to the band saw. Packaging residues at this stage are collected for recycling.
- Block cutting. The blocks are then cut into fish sticks using a food processing band saw, which performs three different cuts to each block. Deteriorated or broken blocks are collected for use as animal feed.
- Battering. At this point of the processing chain, the fish sticks are soaked in a mixture of water and the wheat mix from subsystem II.
- Breadcrumb addition. The conveyor belt sends the battered fish sticks to the following stage for breadcrumb addition.
- Industrial frying. Sunflower oil dosage is injected to each fish stick, which allows maintaining unaltered the raw seafood, while sealing the coating.
- Freezing. The fishing sticks undergo a 50 min freezing process at −37 °C in the freezing tunnel.
- Quality control and packaging. It consists of an entirely automatic process in which the fish sticks are introduced in the cardboard box, which has previously been shaped by the molding machine. Thereafter, the catering scale weighs the incoming boxes. At this point of the processing, personnel supervises that all fish boxes contain a total of 10 fish sticks. The following steps entail the closure of the fish stick boxes, as well as a series of packaging and palletizing manoeuvres. Finally, fish stick pallets are transported to the storage area.
- Storage. Once the fish stick boxes are ready for sale, they are stored once again in the cooling chambers, ready for pick up for distribution. The average storage time before depart from the plant is approximately 1 month.

- Ancillary operations. These include a series of complementary activities that are carried out at the plant in order to guarantee the correct functioning of the manufacturing process, such as lighting, air conditioning, the use of cleaning products or wastewater treatment.

9.2.4 Data acquisition

Data acquisition to develop the life cycle inventory (LCI) was obtained mainly through primary sources, provided by the processing company and its supplying companies. A series of questionnaires were sent to the processing company by the authors with detailed enquires regarding the different stages described in Section 9.2.3. The questionnaires included a broad set of operational aspects and capital goods which are detailed in Section 9.2.6. Material production for operational inputs data, such as diesel or plastics were obtained from the ecoinvent® database (Frischknecht et al., 2007a), as well as data regarding transportation of products. Concerning ingredients, data for the production stage of all agricultural products (sunflower oil, wheat, etc.), as well as data for the processing stages of minor ingredients in terms of weight, such as sunflower oil, were obtained from ecoinvent® and modified for a set of specific conditions for the analyzed production system.

9.2.5 Allocation and other assumptions

According to ISO, allocation pursues the segmentation of the input and output flows of a process system between the studied product and those that are left outside the system boundaries (ISO, 2006b). Allocation was considered in several subsystems of the system under study given the multifunctional processes involved. In fact, system expansion was not implemented in this production system due to the lack of alternative production systems.

In the first place, mass allocation was considered for Subsystem I. The rationale behind this allocation choice is based on the fact that only one single species is being captured. In fact, the company reported that an irrelevant amount of other species are extracted, which are processed for animal feed on board. Economic allocation was ruled out due to the fact that the material and energy flows linked to the different co-products derived from the catch are constant (Pelletier et al., 2007). The use of other physical relationships, such as energy content was also dismissed due to the same nature of the extracted product.

In subsystem IV, mass allocation was also taken into consideration when partitioning the inputs and outputs between the different co-products. The fact that the excess or damaged raw materials maintained the same energetic content discarded that option. Economic allocation was also disregarded due to the intermediate characteristics of the co-products (except for the fish sticks under processing), which impeded correct economic quantification of their price (these products pass on to another processing system within the company), and to the fact that their mass weight was very low as compared to the total weight of the incoming raw materials.

Other assumptions taken into account include the fact that on board processing activities are likely to have different energy requirements. The company did not provide disaggregated fuel consumption patterns for the different processing, propulsion and extraction techniques developed by the vessel, which have shown to vary significantly depending on the nature of the fishery/fishing fleet (Ishikawa et al., 1987; Tyedmers, 2001). Therefore, equal energy requirements were considered for the studied product and co-products up to landing at the Chilean port. In subsystem II, the production phase of some ingredients involved in the production of the wheat-based products, such as pH regulators, spices or vitamins, were excluded due to the lack of data. However, given the minimal weight contribution of these raw materials with respect to the final product, no major changes in final values should be expected. Finally, in subsystem IV, the electricity supplied to the cooling chambers was included as ancillary operations given the lack of data availability regarding allocation of the different stored products (i.e. fish blocks and fish sticks).

9.2.6 Life cycle inventory

LCI development implies the collection and computation of the specific data that allow quantifying those inputs and outputs in the production system that contribute to a given impact category, in this case, global warming, including resource usage and emissions related to the analyzed system (ISO, 2006a). Inventory data were divided into four main subsystems, as discussed in Section 9.2.3. A simplified inventory for the most important inputs and outputs can be observed in Table 9.2, Table 9.3, Table 9.4 and Table 9.5. Furthermore, detailed inventories for subsystem IV, dividing inputs and outputs per processing stage, are included in Tables D.1 to D.9 of Appendix.

Table 9.2. Summary of the average inventory data for Subsystem I (data per FU)

Inputs from the technosphere			**Outputs to the technoshere**		
Materials			*Product*		g
Steel	0.96	g	Patagonian grenadier block to SS2	180	g
Trawl net	$8.07·10^{-3}$	g	**Outputs to the environment**		
Anti-fouling	$3.14·10^{-2}$	g	*Emissions to air*		
Boat paint	$9.26·10^{-3}$	g	Diesel		
Cardboard	$2.56·10^{-2}$	g	CO_2	281.00	g
Polyethylene (LDPE)	$8.93·10^{-4}$	g	CO	0.66	g
Film	$1.43·10^{-4}$	g	SO_2	0.89	g
Marine lubricant oil	$9.49·10^{-1}$	g	NO_x	6.00	g
Fossil fuels			NMVOC	0.22	g
Main motor	58.24	g	Methane	$4.44·10^{-3}$	g
Auxilliary motor 1	11.95	g	N_2O	$7.10·10^{-3}$	g
Auxilliary motor 2	11.95	g	Dioxines	$8.87·10^{-12}$	g
Auxilliary motor 3	5.97	g	PAHs	$3.55·10^{-06}$	g
Auxilliary motor 4	0.60	g	Hydrocarbons	$8.87·10^{-10}$	g
Transport			Cooling agent (R404A)	$8.13·10^{-4}$	g
Chile-Spain (transoceanic freighter)	2.05	tkm	Cooling agent (R22)	$7.47·10^{-2}$	g
Port-Processing plant (>16 t lorry)	$9.11·10^{-3}$	tkm			

Table 9.3. Summary of the average inventory data for Subsystem II (data per FU)

Inputs from the technosphere					
Wheat mix (batter)			**Breadcrumbs**		
Raw materials			*Raw materials*		
Wheat flour	17.94	g	Wheat flour	70.2	g
Wheat starch	1.57	g	Sunflower oil	2.34	g
Salt	1.12	g	Salt	1.17	g
Sunflower oil	0.56	g	Corn dextrose	1.56	g
Water	3.77	g	Water	37.67	g
Materials			*Materials*		
Kraft bags	0.18	g	Polypropylene (Big bags)	0.11	g
Inputs from the technosphere					
Wheat mix (batter)			**Breadcrumbs**		
Energy			*Energy*		
Electricity	$2.15 \cdot 10^{-3}$	kWh	Electricity	$5.54 \cdot 10^{-3}$	kWh
Natural gas	$8.99 \cdot 10^{-3}$	kJ	Natural gas	59.71	kJ
Transport			*Transport*		
Wheat flour (>16 t lorry)	$7.50 \cdot 10^{-3}$	tkm	Wheat flour (>16 t lorry)	$3.17 \cdot 10^{-2}$	tkm
Wheat starch (>16 t lorry)	$3.19 \cdot 10^{-3}$	tkm	Sunflower oil (>16 t lorry)	$2.34 \cdot 10^{-3}$	tkm
Salt (>16 t lorry)	$1.12 \cdot 10^{-3}$	tkm	Yeast (>16 t lorry)	$6.60 \cdot 10^{-4}$	tkm
Sunflower oil (>16 t lorry)	$5.11 \cdot 10^{-4}$	tkm	Salt (>16 t lorry)	$1.17 \cdot 10^{-3}$	tkm
Vitamins A and C (>16 t lorry)	$2.22 \cdot 10^{-4}$	tkm	Corn dextrose (>16 t lorry)	$1.75 \cdot 10^{-3}$	tkm
pH regulators (>16 t lorry)	$1.47 \cdot 10^{-3}$	tkm	Spices (>16 t lorry)	$2.20 \cdot 10^{-3}$	tkm
Wheat mix transport	$4.48 \cdot 10^{-5}$	tkm	Breadcrumbs transport	$1.56 \cdot 10^{-4}$	tkm
Outputs to the technosphere					
Products			*Products*		
Wheat mix for batter	22.42	g	Breadcrumbs	78	g
Residues			*Residues*		
Wastewater	0.27	mL	Wastewater	6.47	mL

Table 9.4. Average inventory data for Subsystem III (data per FU)

Inputs from the technosphere					
Packaging			**Sunflower oil**		
Raw materials			*Raw materials*		
Water	64.42	g	Field operations	$6.52 \cdot 10^{-5}$	ha
Kraftliner paper	23.68	g	Pesticides	$1.66 \cdot 10^{-5}$	g
Potato starch	$6.05 \cdot 10^{-4}$	g	*Transport*		g
Energy			Transport (tractor)	$5.27 \cdot 10^{-5}$	tkm
Electricity	$7.14 \cdot 10^{-4}$	kWh	Sunflower oil (>16 t lorry)	$1.44 \cdot 10^{-2}$	tkm
Fuel oil	0.11	g			
Natural gas	12.9	kJ			
Hydropower	1.08	kJ			
Biomass	$9.11 \cdot 10^{-9}$	kJ			
Outputs to the technosphere					
Products			*Products*		
Cardboard (box)	22.98	g	Sunflower oil	11.21	g
Polyethylene-PE (box)	1.17	g			
Butt	0.77	g			
Pallet separators	0.85	g			
Pallet label	2.37	mg			
Retractable polyolefin	1.65	g			
Type C Bag	58.42	mg			
Thermofusion glue	0.6	g			
Blue seal	18.95	mg			
Print boxes	0.96	g			
Automatic stretch	0.21	g			

Table 9.5. Summary of the average inventory data for Subsystem IV (data per FU)

Inputs from the technosphere					
Raw materials			*Electricity*		
Patagonian grenadier block from SS1	180	g	Air conditioning	$1.64 \cdot 10^{-2}$	kWh
Wheat mix (flour)	22.42	g	Illumination	$7.40 \cdot 10^{-3}$	kWh
Breadcrumbs	78	g	Cold chambers	$8.64 \cdot 10^{-2}$	kWh
Sunflower oil	11.21	g	Hydraulic consumption	$9.47 \cdot 10^{-3}$	kWh
Water	52.05	g	Unwrapping	$6.90 \cdot 10^{-5}$	kWh
Materials			Cutting	$9.35 \cdot 10^{-3}$	kWh
NH_3	$1.31 \cdot 10^{-2}$	mg	Battering	$3.66 \cdot 10^{-3}$	kWh
Detergents	0.48	mg	Breadcrumb addition	$1.84 \cdot 10^{-3}$	kWh
Bleach	0.35	mg	Frying	$1.75 \cdot 10^{-3}$	kWh
Caustic soda	49.11	mg	Freezing	$4.83 \cdot 10^{-2}$	kWh
Water	1.03	g	Packaging	$2.71 \cdot 10^{-2}$	kWh
Lubricants	$3.11 \cdot 10^{-3}$	mL	Wastewater treatment	$1.05 \cdot 10^{-2}$	kWh
Packaging					
Cardboard (box)	22.98	g			
Polyethylene (box)	1.17	g			
Retractable polyolefin	1.65	g			
Outputs to the tecnosphere					
Products			*Residues*		
Fish sticks	323.47	g	Excess of batter	3.16	mL
Packaging	25.97	g	Fish block packaging		
Co-products			Cardboard	14.79	g
Damaged fish blocks	15.47	g	Plastics	6.08	g
Excess of breadcrumbs	4.74	g	Packaging (fish stick box)		
			Cardboard	2.48	g

9.2.7 Impact assessment

The IPCC 2001 method was selected for data computation (Frischknecht et al., 2007b). More specifically, the 100 year frame was chosen in order to calculate GHG emissions. Version 7.3 of Simapro was used as the software support to carry out the CF calculations (Prè-Product Ecology Consultants, 2011).

9.3 Results

9.3.1 Carbon footprint results for subsystem I

The total CF in Subsystem I summed up to 501.6 g of CO_2 eq per pack of fish sticks (Table 9.6). The greater part of GHG emissions contribution (96.1%) was linked to on board activities on the fishing vessel, such as emissions from fuel consumption (65.5%), leakage of cooling agents (27.5%) or packaging (2.8%). Post-landing activities, mainly associated with freighting the packaged fish blocks to Spain, on the contrary, represented 3.9% of the GHG emissions.

Table 9.6. Individual process characterization values for Subsystem I per FU (1 package of frozen fish sticks)

Process	Carbon footprint		
	Units	**Value**	**% over total**
On board activities	*g CO_2 eq*	*481.80*	*96.06*
-Vessel fuelling	g CO_2 eq	328.72	65.54
-Cooling agents	g CO_2 eq	137.79	27.48
-Packaging	g CO_2 eq	13.86	2.76
-Other processes	g CO_2 eq	1.43	0.28
Post-landing activities	*g CO_2 eq*	*19.80*	*3.94*
-Marine freight	g CO_2 eq	18.27	3.64
-Road transport	g CO_2 eq	1.53	0.30
TOTAL	**g CO_2 eq**	**501.60**	**100.00**

9.3.2 Carbon footprint results for subsystem II

Wheat mix for batter and breadcrumbs, both wheat-based products used for fish stick processing, represented 3.8% and 12.6% of the total CF of the studies process, respectively. On the one hand, the relative contributions to the production and transport to the plant of the wheat mix were mainly related to the production of ingredients (76.4%), especially wheat flour and wheat starch, as can be observed in Table 9.7. Other important processes in wheat mix production are the use of sodium chloride in the factory premises (13.3%) and the transport of ingredients prior to mix preparation (6.9%). Energy use within the factory premises only constituted 3.4% of the total impact due to the physical characteristics of wheat mix preparation. On the other hand, the CF of breadcrumb production was attributable mainly to wheat flour production and transport to the plant (81.9%), while the contribution of other ingredients was in all cases below 5% (Table 9.7). Finally, energy use in the factory represented 7.4% of the total contribution.

Table 9.7. Individual process characterization values for Subsystem II per FU (1 package of frozen fish sticks)

Carbon footprint of wheat-based products ($g\ CO_2\ eq/FU$)					
Wheat mix for batter			**Breadcrumbs production**		
Process	**Value**	**% over total**	**Process**	**Value**	**% over total**
Production of ingredients	*20.61*	*76.39*	*Production of ingredients*	*77.04*	*84.63*
- Wheat flour	18.01	66.78	- Wheat flour	70.32	77.25
- Wheat starch	1.9	7.03	- Corn dextrose	2.89	318
- Sunflower oil	0.7	2.58	- Sunflower oil	3.73	4.1
Transport of ingredients	*1.87*	*6.92*	- Salt	0.09	0.1
- Wheat flour	1	3.7	*Transport of ingredients*	*7.05*	*7.74*
- Wheat starch	0.43	1.58	- Wheat flour	4.21	4.62
- Other ingredients	0.44	1.64	- Corn dextrose	1.88	2.06
Mix production activities	*4.5*	*16.68*	- Other ingredients	0.96	1.06
- Sodium chlorate	3.58	13.29	*Breadcrumb production*	*6.95*	*7.63*
- Electricity	0.91	3.37	- Electricity	2.33	2.56
- Transport to plant	$5.39 \cdot 10^{-3}$	0.02	- Natural gas (oven)	4.37	4.8
- Others	$2.70 \cdot 10^{-3}$	0.01	- Others	0.25	0.27
TOTAL	26.98	100	TOTAL	91.03	100

9.3.3 Carbon footprint results for subsystem III

GHG emissions relating to other raw materials used for fish stick production only represented 5% of the total CF identified for fish stick production ready for distribution (Table 9.8). These emissions were mainly linked to the provision of packaging materials for the fish sticks container (21.4 g CO_2 eq) and to the production and transport to the plant of sunflower oil for sprinkling in the frying phase of Subsystem IV (14.3 g CO_2 eq).

Table 9.8. Individual process characterization values for Subsystem III per FU (1 package of frozen fish sticks)

Process	Carbon footprint		
	Units	Value	% over total
Sunflower oil			
- Sunflower oil production	g CO_2 eq	13.9	97.48
- Sunflower oil transport	g CO_2 eq	1.36	2.52
TOTAL	g CO_2 eq	14.26	100
Packaging			
- Cardboard (box)	g CO_2 eq	11.27	52.72
- Polyethylene (box)	g CO_2 eq	2.45	11.46
- Butt	g CO_2 eq	0.72	3.37
- Pallet separators	g CO_2 eq	0.79	3.69
- Pallet label	g CO_2 eq	$1.05 \cdot 10^{-3}$	0.01
- Retractable polyolefin	g CO_2 eq	3.26	15.25
- Type C Bag	g CO_2 eq	0.12	0.56
- Thermofusion glue	g CO_2 eq	1.26	5.89
- Blue seal	g CO_2 eq	$3.74 \cdot 10^{-2}$	0.17
- Print boxes	g CO_2 eq	0.9	4.21
- Automatic stretch	g CO_2 eq	0.57	2.67
TOTAL	g CO_2 eq	21.38	100

9.3.4 Carbon footprint results for subsystem IV

The GHG emissions generated in this subsystem add up to a total of 106.96 g CO_2 eq. However, due to the use of damaged fish blocks and excess breadcrumbs as co-products for the production of animal feed, the final balance for CF is of 66.58 g CO_2 eq per box of fish sticks. This value represents 9.2% of the total CF of the analyzed system.

Electricity supply for the different activities developed in the plant showed to be the central source of GHG emissions (93.75 g CO_2 eq). However, most of this impact was concentrated in two specific locations: the freezing of the fish sticks (22%) and the cooling chambers for fish block and fish stick storage, 39% (Figure 9.2).

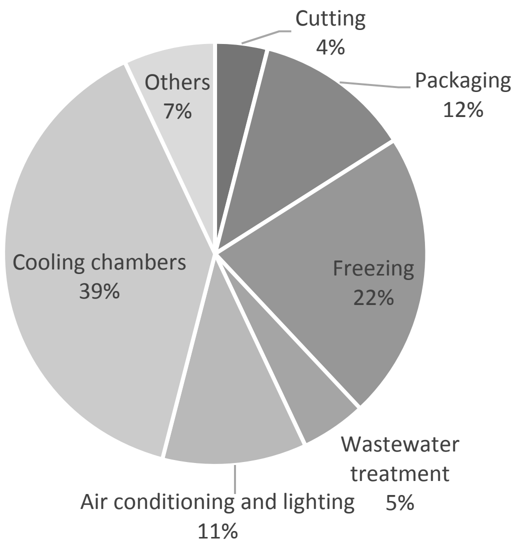

Figure 9.2. Disaggregation of GHG emissions linked to different electricity based activities in subsystem IV

CF reduction due to the reuse of damaged or excess raw materials, whose GHG emissions were reassigned to the corresponding co-products, was divided into two main contributors: damaged fish blocks in the cutting and unwrapping stages (−34.85 g CO_2 eq) and excess breadcrumbs in the breadcrumb addition step (−5.53 g CO_2 eq).

When analyzed by stages, ancillary operations, which include lighting, wastewater treatment or the cooling chambers for storage, were the largest contributors to CF, followed by the freezing stage and packaging (Table 9.9). Other phases of fish stick production, such as cutting, battering or frying, however, showed minimal relative contributions to CF.

9.3.5 Total carbon footprint for fish stick production

The lump sum of the carbon footprint of a package of fish sticks adds up to a total of 716.30 g CO_2. As observed in Figure 9.3, subsystem I is the main contributor (70.0%). The provision of wheat-based products up to the gate of the processing plant accounted for 15.7% of total impacts, while the production, transport and delivery of other raw materials represented 5.0%

of the final CF. Finally, the remaining 9.3% of GHG emissions were due to activities in the processing plant (subsystem IV).

Table 9.9. Individual process characterization values for Subsystem IV per FU (1 package of frozen fish sticks)

Process	Carbon footprint		
	Units	Value	% over total
Storage	*g CO_2 eq*	*1.52*	*2.28*
- Forklift truck (E)	g CO_2 eq	1.11	1.67
- Powered pallet jack (E)	g CO_2 eq	0.08	0.12
- Lubricants	g CO_2 eq	0.33	0.50
Unwrapping	*g CO_2 eq*	*0.00*	*0.00*
Cutting	*g CO_2 eq*	*3.95*	*5.93*
- Processing cutter (E)	g CO_2 eq	1.55	2.33
- Band saw (E)	g CO_2 eq	2.40	3.60
- Others	g CO_2 eq	0.00	0.01
Battering	*g CO_2 eq*	*0.35*	*0.53*
- Batter (E)	g CO_2 eq	0.33	0.50
- Water	g CO_2 eq	0.02	0.03
Breadcrumb addition	*g CO_2 eq*	*0.77*	*1.16*
Frying	*g CO_2 eq*	*0.74*	*1.11*
- Sprinkler (E)	g CO_2 eq	0.74	1.11
Freezing	*g CO_2 eq*	*20.39*	*30.62*
- Freezing tunnel (E)	g CO_2 eq	8.21	12.33
- Frigorific compressors (E)	g CO_2 eq	12.18	18.29
Packaging	*g CO_2 eq*	*11.44*	*17.18*
- Catering scale (E)	g CO_2 eq	0.05	0.07
- Retractable (E)	g CO_2 eq	2.21	3.32
- Sealing machine (E)	g CO_2 eq	0.44	0.66
- Moulding machine (E)	g CO_2 eq	0.30	0.45
- Conveyor belt (E)	g CO_2 eq	0.20	0.30
- Palletizing robots (E)	g CO_2 eq	0.07	0.11
- Packaging robots (E)	g CO_2 eq	2.95	4.43
- Compressed air machines (E)	g CO_2 eq	5.22	7.84

Table 9.9. Individual process characterization values for Subsystem IV per FU (1 package of frozen fish sticks) (continuation)

Process	**Carbon footprint**		
	Units	**Value**	**% over total**
Ancillary operations	*g CO_2 eq*	*56.59*	*85.00*
- Wastewater treatment plant	g CO_2 eq	4.48	6.73
- Emissions cooling agent	g CO_2 eq	11.21	16.84
- Air conditioning	g CO_2 eq	6.92	10.39
- Lighting (storage)	g CO_2 eq	0.02	0.03
- Lighting (manufacture hall)	g CO_2 eq	1.00	1.50
- Lighting (cooling chambers)	g CO_2 eq	2.11	3.17
- Cooling chambers (E)	g CO_2 eq	36.47	54.78
- Hydraulic consumption (E)	g CO_2 eq	4.00	6.01
- Cleaning products	g CO_2 eq	1.59	2.39
Co-products	*g CO_2 eq*	*-40.38*	*-60.65*
- Fish blocks to animal feed	g CO_2 eq	-34.85	-52.34
- Breadcrumbs to animal feed	g CO_2 eq	-5.53	-8.31
TOTAL	**g CO_2 eq**	**66.58**	**100.00**

E= electricity.

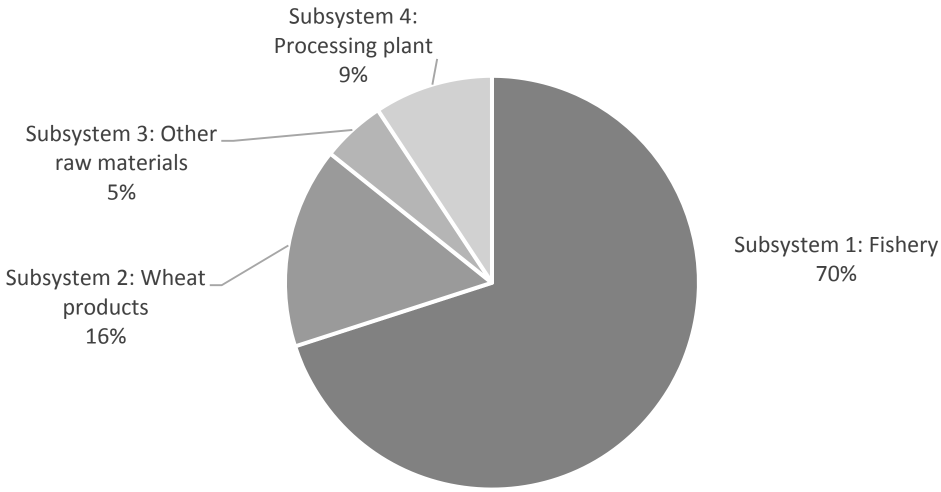

Figure 9.3. Individual CF contribution for each of the considered subsystems (results referred to the FU in g CO_2 eq)

9.4 Discussion

9.4.1 Identification of hot spots

The entire system under study was dominated in terms of CF by subsystem I, due mainly to on board activities by the fishing vessel. In fact, two sole operations accounted for 65.1% of the total CF: emissions linked to fuel consumption (45.9%) and cooling agent leakage from the cooling chambers on board (19.2%). Therefore, it is obvious that the main hot spots of the production system, as reported for many other seafood products, are related to the fishing stage (Eyjolfsdottir et al., 2003; Vázquez-Rowe et al., 2011; Ziegler et al., 2003, 2011). Production of agricultural products for ingredient production or for fish stick processing in the plant was the third highest source of CF in the entire system, representing 14.9% of GHG emissions. Wheat flour and starch appeared as the main contributor within this group of raw materials. However, this is mainly attributable to the higher percentage of wheat used in the final product. Finally, another relevant CF activity was the electricity supply in the processing plant (13.1%). However, it should be noted that the GHG emissions for the processing stage (including packaging) appear to be considerably lower than in the production of other ready-to-consume food products, such as ketchup or canned seafood (Andersson et al., 1998; Hospido et al., 2006; Iribarren et al., 2010c).

9.4.2 Energy use in the fishing systems

The fact that approximately half of the total CF of the product was generated by GHG emissions linked to fuelling the fishing vessel, led us to compare the fuel use intensity (FUI) of this fishery to that of other comparable fishing fleets, as can be seen in Table 9.10.

Table 9.10. Fuel use intensity (FUI) for selected Spanish fishing fleets targeting hake species

Publication	Species	Inventory year	Fishing gear	Fishing area*	FUI (L/tonne)
Current study	Patagonian grenadier	2011	trawling	FAO Area 87 (Chile)	469
Unpublished	European hake	2008	driftnet	Galician coast	123
Vázquez-Rowe et al. (2010)	European hake	2008	trawling	Galician coast	557
Vázquez-Rowe et al. (2011)	European hake	2008	trawling	Northern Stock	2,363
Ramos et al. (2011b)	European hake	2007	trawling	Northern Stock (Basque fleet)	2,255
Vázquez-Rowe et al. (2011)	European hake	2008	long lining	Northern Stock	1,466
Vázquez-Rowe et al. (2012b)	Black hake	2009	trawling	Mauritanian EEZ	1,939

* All fishing fleets are Galician unless otherwise indicated. Patagonian grenadier is landed in Chile and then freighted to Galicia, but FUI only includes fishing vessel operations.
FUI= fuel use intensity; European hake= *Merluccius merluccius*; Black hake= *Merluccius senegalensis*; Patagonian grenadier= *Macruronus magellanicus*.

Data show that the analyzed fishery presents a considerably low FUI as compared to other Spanish fisheries where hake species constitute an important target. In fact, the Patagonian grenadier fishery does not only perform better in terms of FUI than other trawling fisheries, but also when compared to fishing fleets with gears that are usually considered less energy intensive, such as long lining (Tyedmers, 2001; Vázquez-Rowe et al., 2011, 2012c). Therefore, despite the high GHG emissions linked to fuelling the vessel in the current system, the supply of hake species from other fisheries does not appear as a carbon friendly alternative.

Furthermore, when compared on a broader worldwide level, the FUI identified for the current production system appears in the lower range of fuel consumption per unit of catch for trawling vessels (Tyedmers, 2001; Watanabe and Okubo, 1989). Therefore, if we assume that the FUI rate reported for 2011 is maintained through time, results could suggest that stock abundance and the fishing intensity of this fishery are managed in an adequate manner (Tyedmers, 2001). Consequently, as it will be further discussed in Section 9.4.3, considerable reductions in GHG emissions due to fuel burning would only be possible in the frame of technological improvements. However, further analysis on this fishery for a wider timeframe, ideally several fishing seasons, would be needed to confirm this observation (Ramos et al., 2011a).

9.4.3 Identified improvement actions

A reduction in GHG emissions based on energy efficiency in the fishery does not seem a feasible option unless alternative energy carriers become an option for vessel propulsion. In fact, without a shift in the energy strategy in fishing, CF mitigation with respect to fuel consumption will be mainly dependent on three main factors: any technological improvement that reduces fuel burning without affecting the fishing yield of the vessel (Tyedmers, 2001), the ability of the skipper and the crew to use their acquired skills to reduce the amount of fuel consumed per unit of catch (Ruttan and Tyedmers, 2007; Vázquez-Rowe and Tyedmers, 2013) and the natural or human induced changes in stock abundance in a particular fishery (Fréon et al., 2008).

On the contrary, the leakage to the atmosphere of certain cooling agents, such as R22 and R404A, which are both used in the examined production system, could be avoided by introducing refrigerants that lower or do not create impacts in terms of CF, such as carbon dioxide (R744), ammonia (R717) or R134a (Iribarren et al., 2011; Senter Novem, 2002; Ziegler et al., 2013). This mere improvement action would reduce the CF of fish stick production by 19.2%, 578.5 g CO_2 per FU. Additionally, as pointed out by Ziegler et al. (2013), the use of ammonia constitutes a more energy efficiency alternative, which would probably imply lower fuel consumption for refrigeration, reducing even more the CF of the fishing stage.

While the change in refrigeration on board appears as the main source of potential CF reduction for fish stick production, a series of initiatives may help reduce GHG emissions in other stages of the life cycle. For instance, changes in the agricultural production systems of wheat and other major ingredients in fish stick processing or the shift to more renewable energy systems for electricity supply may be beneficial in terms of CF (Thrane et al., 2009).

Regarding electricity supply, however, the processing plant obtains its electricity from the Spanish grid. The electricity production mix used in this study was based on adapting the ecoinvent® electricity mix for Spain to the specific data retrieved from the Spanish government for 2009 (REE, 2009; Spanish Ministry of Industry, 2009). Unfortunately, the Spanish government had not disclosed detailed new values for 2010 and 2011 when this study was conducted. However, despite the growing contribution of renewable energies to the grid in both years, in late 2010 the Spanish government, influenced by the economic crisis the country was suffering with high unemployment levels, announced a strong increase in the subsidies to domestic coal (BOE, 2010). Consequently, and also due to lower contributions by hydroelectric and wind power in 2011, an increase in CO_2 emissions in the electric sector has been forecasted for this year (REE, 2011). Even though this constitutes a notable increase regarding GHG emissions in the Spanish electric sector, an extrapolation of this increase to the production system would only imply an increment of the CF by 3.3%. Nonetheless, this example demonstrates how decision making at governmental level can potentially influence CF reporting (Weidema et al., 2008).

Finally, a potential but also controversial option would be the substitution of Patagonian grenadier by an alternative fish species. As seen in Table 9.11, some alternative production systems may entail important CF reductions.[3] Furthermore, in some nations, the use of pelagic species for fish stick production, such as Atlantic mackerel (*Scomber scombrus*), has become a regular practice, with significant trade quotas (Young's Seafood Limited, 2011). While change in the fish species for fish stick production does not represent a feasible option within the Spanish market, given the social implications it entails, it may be considered as an attractive alternative

[3] The case studies included in Table 9.11 seek a general framework to compare the Patagonian grenadier fish sticks with other seafood production systems. The processes were recalculated in order to provide resembling assumptions to those considered for the production system analyzed in this work.

to fish sticks exported elsewhere, due to the lower attachment of consumers in other countries to fish fingers produced from hake species.

Table 9.11. Comparative table between the CF of fish stick production and other products or scenarios

Publication	Product	Inventory year	Functional Unit	CF ($g\ CO_2/FU$)
Current study	Analyzed fish sticks	2011	1 package (324 g of sticks)	716.3
Scenario A	Atlantic mackerel fish sticks	2011/2007[*]	1 package (324 g of sticks)	284
Iribarren et al. (2010c)	One triple pack of round cans of canned mussels[†]	2007	1 triple pack	4,350
Sund (2009)	Breaded Alaska pollock[‡]	2007	1 kg of breaded pollock	1,250
Sund (2009)	Breaded cod[‡]	2007	1 kg of breaded cod	1,975
Vázquez-Rowe et al. (2011)	Fresh European hake	2008	500 g of fresh hake[**]	1,466
Fulton (2010)	Frozen cod	2009	500 g of frozen cod	350

[*] This scenario was modelled by including the equivalent amount of Atlantic mackerel, substituting Patagonian grenadier.

[†] Each triple pack includes 129 g of mussel flesh plus additional ingredients.

[‡] In the same way as the current study, it does not include retailing or consumption.

[**] CF calculation includes the retailing and consumption stages.

European hake= *Merluccius merluccius*; Patagonian grenadier= *Macruronus magellanicus*; cod= *Gadus morhua*; Alaska pollock= *Theragra chalcogramma*; mussels: *Mytilus galloprovincialis*.

9.4.4 Energy return on investment (EROI) of fish stick production

Previous chapter highlighted the usefulness of reporting the energy efficiency of seafood and agro-food products in terms of the edible protein energy return on investment (ep-EROI), which calculates the ratio between the energy that a given process supplies with respect to the total energy supplied to that process (Cleveland, 2008).In this regard, the use of LCA for energy use identification in EROI appears to be advantageous, given its life cycle perspective, including direct and indirect energy supply.

Therefore, on the one hand, this study used the Cumulative Energy Demand (CED) indicator, as proposed by VDI-Richtlinien (1997) to calculate the energy required to produce a package of fish sticks. On the other hand, assuming that the gross energy content of fish protein is 16.73 MJ/kg (Higgs et al., 1995; Tyedmers, 2000) and obtaining the protein content of the different ingredients used in fish stick production, the edible protein energy of the studied product was calculated (Peter Tyedmers, personal communication).

The ep-EROI value obtained for the examined product was 6%, which is substantially higher than other hake products in literature (see Chapter 8). The characterization values for CED indicator can be seen in Table D.10 of Appendix. Moreover, when the EROI of the studied product was compared to other fish stick processing production systems analyzed in the literature, fish sticks produced with Patagonian grenadier performed up to 2.2% and 35.8% better than cod and pollock fish fingers, respectively, as shown in Figure 9.4 (Sund, 2009). Finally, if the ep-EROI is measured exclusively in terms of the fish extraction stage, results show that Patagonian grenadier extraction implied a return on investment of 10.4%, substantially higher than the average ep-EROI of global fisheries —8% (Tyedmers et al., 2005).

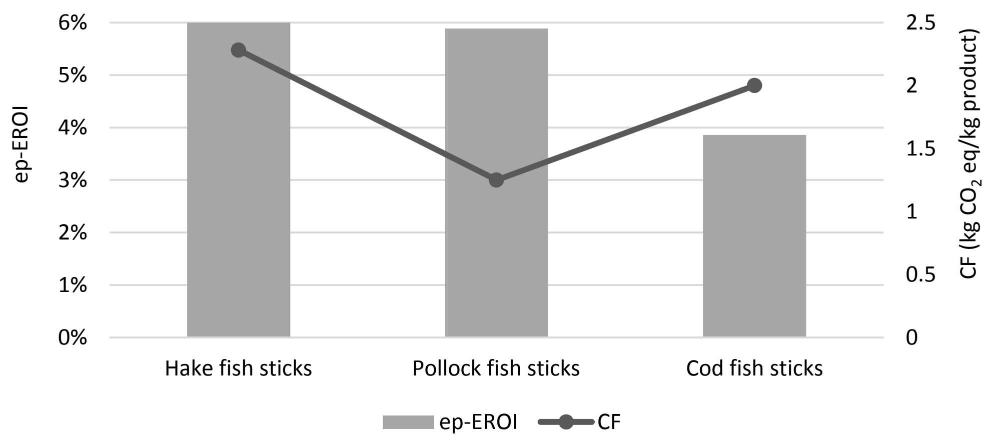

Figure 9.4. EROI and CF values for fish stick processing systems available in literature as compared to hake fish sticks

These results highlight the importance of assumptions and selected approaches when performing LCA based studies (Vázquez-Rowe et al., 2012a). For instance, the use of a mass FU for the studied product led to a CF considerably higher than that of other fish stick products. However, the ep-EROI demonstrated that when an identical portion of fish sticks by weight is considered, hake fish sticks performed better in terms of energy return. This apparent contradiction suggests that hake fish sticks constitute an attractive source of protein supply within multi-ingredient seafood-based products.

9.5 Conclusions

Fish stick production with Patagonian grenadier proved to be a relatively low carbon product. On the one hand, the energy use of this species is outstandingly low when compared to other trawling fleets targeting other hake species, contributing to its low carbon profile. On the other hand, CF values for this product appear in the lower range when compared to similar substitutable products. Therefore, from a GHG emission perspective, the examined fish sticks constitute an attractive alternative with respect to other protein-based food products.

Nevertheless, two similar trends were observed with respect to other previously examined seafood products. In the first place, the energy use of the fishery appears as the major source of GHG emissions. For instance, from an energetic perspective fish stick production presented an EROI of 6%, which despite being relatively positive for a product of these characteristics, demonstrates that fish extraction still presents extremely high energy inputs.

Secondly, the high emissions attributed to cooling agent leakage seem like a relatively easy and low cost improvement action to implement. Hence, while fishing fleets, and consequently seafood products, should benefit from the shift to carbon neutral refrigerant agents within the following years, the reduction of GHG emissions linked to vessel fuelling appears as a major challenge within the sector.

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Chapter 10

The role of consumer purchase and post-purchase decision-making in sustainable seafood consumption[1]

Summary

Sustainable consumption in the food sector is a desirable goal which is often difficult to achieve depending as it does on the interaction of a broad set of factors, such as market prices or consumer preferences. In the current study, a distinction has been made between pre-purchase consumer decision-making, on the one hand, and purchase and post-purchase consumer decision-making, on the other. Purchase and post-purchase decisions are those that affect their actions directly. These include vehicle selection to purchase the products, shelf-time in the household or the cooking method selected. The main goal of this study was to evaluate the environmental profile of a range of scenarios with different decisions taken by consumers. More specifically, the carbon footprint (CF) of an iconic frozen seafood product in Spain: one package of frozen fish sticks of Patagonian grenadier, was modeled for 24 scenarios of fish sticks consumption. Results showed a high variance in the environmental impacts depending on the scenario chosen, proving the high variability in CF that the consumption of frozen seafood can show depending on consumer choices or needs. Additionally, results showed the relative importance of the consumption stage within the entire supply chain in terms of GHG emissions. Hence, important reductions may be achieved merely by changing the behavioral traits when purchasing and consuming food products. Consequently, consumers, if given the correct environmental guidelines through awareness campaigns, can play an active and relevant role in the reduction of the environmental profile of seafood products through behavioral modifications when purchasing and consuming them.

[1] Vázquez-Rowe, I., Villanueva-Rey, P., Moreira, M.T., Feijoo, G., 2013. The role of consumer purchase and post-purchase decision-making in sustainable seafood consumption. A Spanish case study using carbon footprinting. Food Policy 41, 94–102.

Index

10.1 Introduction

In the food sector, as in many other industrial sectors, household consumption patterns, while perhaps not constituting a decisive part of the environmental impact of a specific product, can cause varying impacts depending on the nature of the decisions taken prior to their intake (de Boer et al., 2006). For instance, choosing fresh hake from long liners instead of hake caught by trawlers has shown an outstanding potential environmental benefit for a wide range of environmental dimensions, including climate change and acidification (Vázquez-Rowe et al., 2011). The consumer's decision-making can be divided into two main blocks from an environmental perspective. On the one hand, purchase and post-purchase decision-making, which involves the action of the consumers directly throughout the purchase and consumption process, such as vehicle selection to purchase the products, shelf-time in the household or the cooking method selected. On the other hand, there is pre-purchase decision-making, based on the selection of different options depending on their background environmental profile.

Pre-purchase factors affecting decision-making have been widely analyzed in the literature, mainly through publications that compare the environmental burdens of different dietary choices (Carlsson-Kanyama, 1998; Eshel and Martin, 2006; Franks and Hadingham, 2012) or suggested the implementation of specific carbon calculators (Kim and Neff, 2009; Shirley et al., 2012). For instance, some of these studies reveal that high consumption of animal products in Western countries, including products derived from cows and pigs, significantly contribute to these nations' GHG emissions (Compassion in World Farming, 2007; Garnett, 2011; Deckers, 2012), questioning the sustainability of food consumption patterns (Carlsson-Kanyama, 1998; Carlsson-Kanyama and González, 2009). Other studies have highlighted the high variability in GHG emissions existing between different seafood products (Iribarren et al., 2010a; Iribarren et al., 2011; Ziegler et al., 2013) and fishing patterns (Vázquez-Rowe et al., 2011; Vázquez-Rowe et al., 2013). However, unlike other animal products, certain seafood products, such as mussels or small-pelagic species, show very low emissions in their supply chain (Iribarren et al., 2010b;

Iribarren et al., 2011). Therefore, it seems plausible that pattern changes in the way humans obtain their protein supply will be needed to attain desired GHG emission reductions in the food and beverage sector (McMichael et al., 2007; Deckers, 2012).

However, it may be difficult for the general public to keep track of which elements of the diet have lower environmental profiles, since there are numerous exceptions that can alter the general perception (Coley et al., 2009). In fact, the concept carbon capability has recently been developed to analyze and understand the relation between the abilities of human individuals and their motivations and/or willingness to reduce emissions (Seyfang et al., 2007; Whitmarsh et al., 2011). If linked to consumption patterns, purchase and post-purchase decision-making by consumers would encompass all those behavioral traits that do not depend on the carbon emissions linked to a product prior to its shelf-life, including, therefore, the transportation to a retailing point or the cooking of the purchased products. While some studies have already started analyzing the environmental gains that may be attained through improving purchase and post-purchase consumption decisions (Persson and Bratt, 2001; Coley et al., 2009), this domain remains essentially unexplored.

Seafood represents an important portion of the annual diet intake in Spain, representing approximately 12.9% of their annual spending on food products. Of this total amount, 3.3 kg corresponded to frozen seafood products, such as fish sticks, calamari rings or frozen fish fillets, mainly of hake and cod (FROM, 2010). Concerning consumption patterns, shoppers in Spain have reported an increasing interest in labels on fish products in recent years. In fact, 70.5% of consumers admitted checking the label of seafood products on a regular basis in 2010 (FROM, 2011). While their main concerns were related to expiry dates, ingredients or nutritional values, this situation provides a good starting point for including environmental sustainability data on seafood packages.

Consequently, the inclusion of eco-labels on seafood products in Spain is an achievable goal in years to come, which would be in line with similar trends already observed in other Western nations, such as the United Kingdom (Seafood Choices Alliance, 2007). For instance, Gadema and Oglethorpe (2011), basing their study on an extensive survey targeting UK supermarket consumers, detected a high level of positive attitudes towards eco-labeling (in this specific case, carbon labels). Nevertheless, it is important to take into account that the results from this study also revealed high levels of confusion between customers regarding the interpretation and understanding of these carbon labels. Moreover, given the wide range of decisions consumers may take regarding their final product selection —which do not necessarily have the best environmental profile— it may be more interesting to inform them about how their purchase and post-purchase consumption patterns and choices may influence the final environmental profile of a given product (Garnett, 2011).

While many eco-labels in the seafood sector focus on stock assessment issues, some are starting to include additional environmental parameters, such as discarding or carbon footprint —CF (Thrane et al., 2009). CF is the measure of the potential greenhouse gas (GHG) emissions that are generated by a product, process or service during its life-cycle (BSI, 2011). Hence, it follows the life-cycle perspective of life cycle assessment (LCA) methodology (ISO, 2006a), but limiting its analysis exclusively to the global warming potential impact category. Therefore, in this chapter, the direct environmental decisions made by consumers are modeled in order to: i) track the influence that these decisions may have on the CF of the target product; ii) analyze the different environmental hotspots in each scenario; iii) provide an estimated lump sum of the total reduction in GHG emissions for the sale of this particular product in Spain; and iv) discuss the appropriateness of including consumer best-performing patterns in food product eco-labels. The product selected in this study was one package of frozen fish sticks of Patagonian grenadier —previously analyzed on Chapter 9.

10.2 Materials and Methods

10.2.1 Goal and scope definition

A total of 24 scenarios regarding the consumption of fish sticks were modeled in the current research study. The research undertaken was done in collaboration with Pescanova, the main seafood processing company in Spain and one of the top 10 such companies in the world (Barciela, 2011). While the initial objectives of the study were limited to a business-to-business (B2B) approach (see Chapter 9), in which the system boundaries for the CF estimation of the products was limited to the gate of the processing company, the availability of data sources led to the completion of the entire life-cycle up to consumption in the household (business-to-consumer approach —B2C). Environmental impacts for the fish extraction and processing stages, as well as distribution and retailing were found to be highly stable. However, consumer patterns were identified as highly variable depending on shopping, storage and cooking methods. Hence, the scope of this case study is limited to those activities directly performed by consumers.

Patagonian grenadier (*Macruronus magellanicus*) fish sticks were the product selected due to the variety of cooking options they potentially possess. In fact, the products' package suggests three distinct cooking modes for this specific product (Pescanova, 2012). Therefore, the functional unit (FU), which is the reference unit which the energy and material flows refer to (ISO, 2006b), was set as one package of fish sticks of Patagonian grenadier[2]. The boundaries of the production systems, as seen in Figure 10.1, were set from the moment when the consumer acquired the product at the retailing store (including the travel distance to the purchase point) up to the intake of the fish sticks in the household.

[2] One package of Patagonian grenadier contains approximately 323.5 g of edible product. Consequently, each fish stick weighs on average 32.35 g. The fish content of sticks was approximately 164.5 (50.9%).

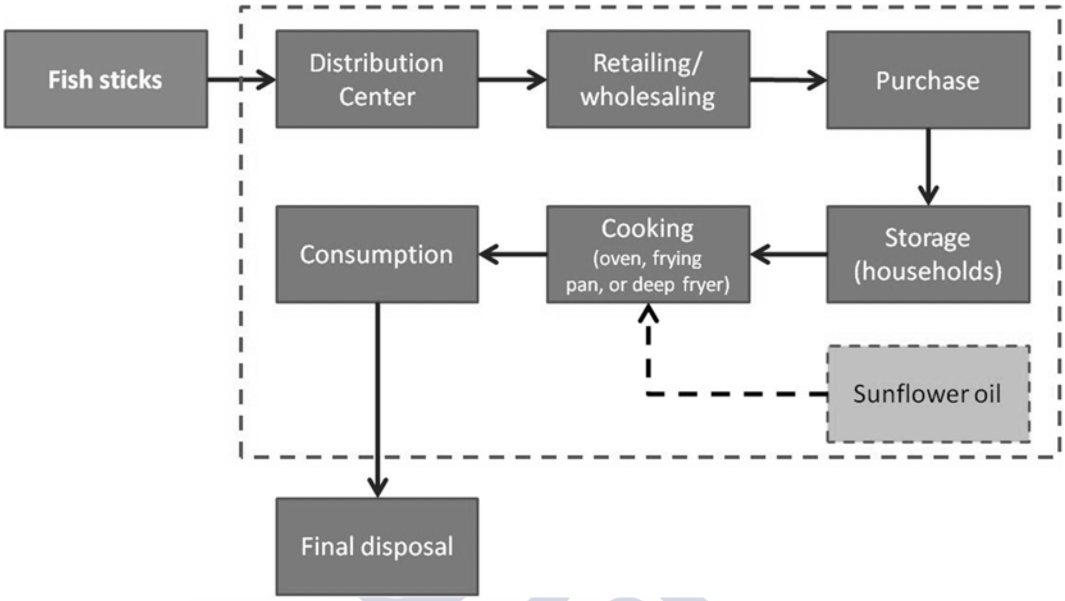

Figure 10.1. System under study for Patagonian grenadier fish sticks consumption

10.2.2 Data acquisition

Data were obtained from a variety of bibliographical sources. Three distinct cooking methods for fish stick preparation were considered: deep frying with sunflower oil, thermal insulation in an oven and conventional frying in a frying pan (with sunflower oil on a gas stove cooker) according to the recommendations of the wholesaler (Pescanova, 2012). Table 10.1 and Table 10.2 present data regarding fish stick cooking, as well as consumer cooking preferences.

Table 10.1. Data considered for fish stick cooking

Cooking mode	Power (W)	Sunflower oil* (mL/FU)	Cooking time (min)
Frying pan	1,750	125	8
Deep fryer	2,000	200	6
Oven	1,300		25

* The value for sunflower oil use refers to an entire use cycle

Table 10.2. Survey regarding preferred cooking methods by fish stick consumers in Spain

Cooking method	Number of respondents	%
Frying pan	1,575	38.9
Deep fryer	866	21.4
Oven	733	18.2
More than one method	870	21.5
Total respondents	*4,044*	*100.0*

Data concerning the storage time of the product in household freezers was based on a database provided by Pescanova (Juan Mallo, Pescanova, personal communication). This database was created from a survey exploring the average storage time of different seafood products. The average consumption time for the fish sticks analyzed was found to be approximately 90 days, although 8.3% of consumers reported consuming them within the first month and 55.6% before 90 days. Furthermore, two different types of freezers were assumed in this study: freezers with an energy efficiency index (I) above 55 named as class A and freezers with an "I" value between 55 and 75 that belong to class B (European Commission, 1994 and European Commission, 2003). Lower efficiency levels were not included due to the low share of the market they had by 2010 (IDAE, 2011).

It is assumed that the transport to the point of retail to buy the fish sticks is made by car if the purchase is made at superstores or large supermarkets and by foot to nearby markets and small shops. Nevertheless, it is important to note, that according to recent values, 77% of Spaniards admitted purchasing frozen seafood products at superstores and large supermarkets, while only 6.4% claimed to acquire these products in small retailer stores and 4.4% in markets (FROM, 2011; MAGRAMA, 2011).

Data related to consumer behavior regarding frozen seafood consumption were retrieved from a set of reports available from the Regulation and Organization Fund for the Fish and Marine Cultures Market (FROM), an organization dependent on the Ministry of Agriculture and Fisheries of the Spanish government (FROM, 2005; FROM, 2011). Finally, data for additional raw materials used in fish stick preparation were limited to sunflower oil use when frying or deep frying. No additional raw materials were considered for thermal insulation cooking in the oven.

Therefore, based on data availability as well as on the different choice options that consumers have when purchasing and consuming this product, 24 different scenarios were modeled from an environmental perspective, which are described in detail in Table 10.3.

Table 10.3. Patagonian grenadier fish stick consumer chains included in the study

Scenario	Cooking mode	Consumer transport	Freezer type	Freezing time (days)	Cooking time (min)	Sunflower oil addition
A1	Deep fryer	Walking	A	30	6	Yes
A2	Deep fryer	Walking	A	90	6	Yes
A3	Deep fryer	Walking	B	30	6	Yes
A4	Deep fryer	Walking	B	90	6	Yes
A5	Deep fryer	Car	A	30	6	Yes
A6	Deep fryer	Car	A	90	6	Yes
A7	Deep fryer	Car	B	30	6	Yes
A8	Deep fryer	Car	B	90	6	Yes
B1	Oven	Walking	A	30	25	–
B2	Oven	Walking	A	90	25	–
B3	Oven	Walking	B	30	25	–
B4	Oven	Walking	B	90	25	–
B5	Oven	Car	A	30	25	–
B6	Oven	Car	A	90	25	–
B7	Oven	Car	B	30	25	–
B8	Oven	Car	B	90	25	–

Table 10.3. Patagonian grenadier fish stick consumer chains included in the study (continuation)

Scenario	Cooking mode	Consumer transport	Freezer type	Freezing time (days)	Cooking time (min)	Sunflower oil addition
C1	Frying pan	Walking	A	30	8	Yes
C2	Frying pan	Walking	A	90	8	Yes
C3	Frying pan	Walking	B	30	8	Yes
C4	Frying pan	Walking	B	90	8	Yes
C5	Frying pan	Car	A	30	8	Yes
C6	Frying pan	Car	A	90	8	Yes
C7	Frying pan	Car	B	30	8	Yes
C8	Frying pan	Car	B	90	8	Yes

10.2.3 Life cycle inventory

The life cycle inventory (LCI) constitutes the second stage to undergo carbon foot-printing, consisting of the collection and computation of the necessary input and output data available for the production system being evaluated (ISO, 2006a). For this CF study, inventory data were obtained mainly in the form of primary data regarding the consumption pattern as well as the cooking preferences from the processing and distribution companies. Background data were retrieved from the ecoinvent® database (Frischknecht et al., 2007). The average inventory data presented in Table 10.4 were allocated to the selected FU.

10.2.4 Allocation and other assumptions

The average distance traveled by consumers driving to the retailing point was assumed to be 2.5 km. This value was taken based on the fact that 30% of car rides in European cities are below 3 km (Dekoster and Schollaert, 2000). Nevertheless, given the high standard deviation found for this value, discussion is provided regarding how changing distances may influence the results. Additionally, 10 other items were taken into consideration when consumers did their shopping. This average value was taken from a report developed by Nielsen (2011) regarding consumption habits in Spain.

Table 10.4. Inventory data for distribution and consumption of fish sticks (data per FU)

Inputs from the technosphere					
Product			*Electricity*		
Fish sticks	323.5	g	Cooling chamber (wholesaling)	$8.64 \cdot 10^{-5}$	kW h
Sunflower oil (frying pan)	49.31	g	Freezer (retailing)	$1.94 \cdot 10^{-2}$	kW h
Sunflower oil (deep fryer)	47.69	g	Freezer (consumer)	$9.36 \cdot 10^{-2}$	kW h
Packaging			Cooking (frying pan)	$1.02 \cdot 10^{-1}$	kW h
Total materials	25.97	g	Cooking (deep fryer)	$2.68 \cdot 10^{-2}$	kW h
Transport			Cooking (oven)	$1.21 \cdot 10^{-1}$	kW h
Lorry >16 t	$3.77 \cdot 10^{-1}$	t km			
Outputs to the technosphere					
Residues					
Cardboard (box)	22.98	g	Sunflower oil	11.21	g
Polyethylene-PE (box)	1.51	g			
Butt	0.77	g			
Automatic stretch	0.21	g			

No allocation was needed for the fish sticks, since the intake of one single product was taken into consideration with no other by-products in the analyzed stage. However, allocation was required for the use of sunflower oil in the deep frying and frying pan scenarios, since it can be reused. Hence, a reuse rate of 10 times was assumed for deep frying, while the reuse ratio for frying in a cooker was assumed to be 5. This lower reuse ratio was considered to avoid the formation of toxic substances, since frying pans do not have a temperature control system (OCU, 2012). The reused amount in the case of deep frying was 2 L, which is the conventional amount of sunflower oil used in fryers. The amount for frying in cookers was set at 500 mL. However, these two values were divided by the reuse ratios considered and described previously (OCU, 2012).

10.2.5 Environmental assessment

The selected scenarios were computed using the IPCC 2001 method (Frischknecht et al., 2007). In particular, the 100 year frame was the option selected in order to calculate GHG emissions.

SimaPro 7.3 was the software selected for method computation (Prè-Product Ecology Consultants, 2011).

The selection of CF as the sole environmental indicator was based on the fact that climate change has developed into an important environmental concern for citizens and stakeholders, becoming a commonly-used impact category in seafood LCA studies (Pelletier et al., 2007; Winther, 2009; Vázquez-Rowe et al., 2012a). Fishery-specific impact categories, which have been included in previous environmental assessment seafood studies (Ziegler et al., 2003; Vázquez-Rowe et al., 2012b), were not considered in this research study given the fact that the same amount of one sole natural resource (Patagonian grenadier) was considered in all the scenarios.

10.3 Results

10.3.1 Environmental impact results for fish stick preparation in a deep fryer

The CF of consumer activities for scenarios in which deep frying was assumed as the cooking option ranged from 301 to 502 g of CO_2 eq depending on the other assumptions that were taken into consideration. Therefore, scenario A1 presented the lowest GHG emissions due to the lack of transportation to the purchasing point and the assumption that the fish sticks were stored for 30 days before consumption. In contrast, scenario A8 presented an increase of 66.8% with respect to scenario A1, due to the inclusion of vehicle emissions, a higher storage time (90 days) and a reduced efficiency rate of the selected freezer. Nevertheless, the use of complementary raw materials (i.e., sunflower oil), was the main contributor in all the scenarios, ranging from 73.4% in scenario A1 to 44% in scenario A8. The contributions of the other elements included in the different scenarios were found to be highly dependent on the specific assumptions. Consequently, in the case of shopping on foot, the relative impact of household storage was substantially higher than when road transport was considered. For instance, household storage ranged from 9.9% in scenario A5 to 37.2 in scenario A4. Finally, cooking activities were considered constant for the 8 scenarios, with relative GHG emissions ranging from 8.4% (A8) to 14.0% (A1).

10.3.2 Environmental impact results for fish stick preparation in an oven

The total CF for oven preparation of fish sticks ranged from 215 to 416 g of CO_2 eq, depending on the scenario selected. This variation was mainly attributable to the transport and storage time, given the lack of raw materials other than the fish sticks themselves. Scenario B1 was the one that presented the lowest GHG emissions, due to the lack of fueled transportation and the low storage time considered. Hence, on the one hand, B1 showed the highest relative impact regarding cooking activities (82.4%) within this group of scenarios. On the other hand, B4 presented the highest relative contribution for household storage (46.8%). Transportation ranged from 19.9% to 27.8% when fueled transportation was considered. Finally, it is important to point out that the scenarios that considered fish stick cooking in the oven showed the lowest average CF, as can be seen in Figure 10.2.

10.3.3 Environmental impact results for fish stick preparation in a frying pan

In general terms, fish sticks prepared in a frying pan using sunflower oil showed the highest environmental impacts, ranging from 356 g CO_2 eq (Scenario C1) to 556 g CO_2 eq (Scenario C8). An increase in storage time and the means of transport selected by customers to purchase the fish sticks were the two factors that accounted for a variation of up to 56.2% in the environmental profile of fish finger consumption in these scenarios. However, the use of sunflower oil in the cooking stage was the main environmental impact in all scenarios, ranging from 39.7% (C8) to 61.0% (C1).

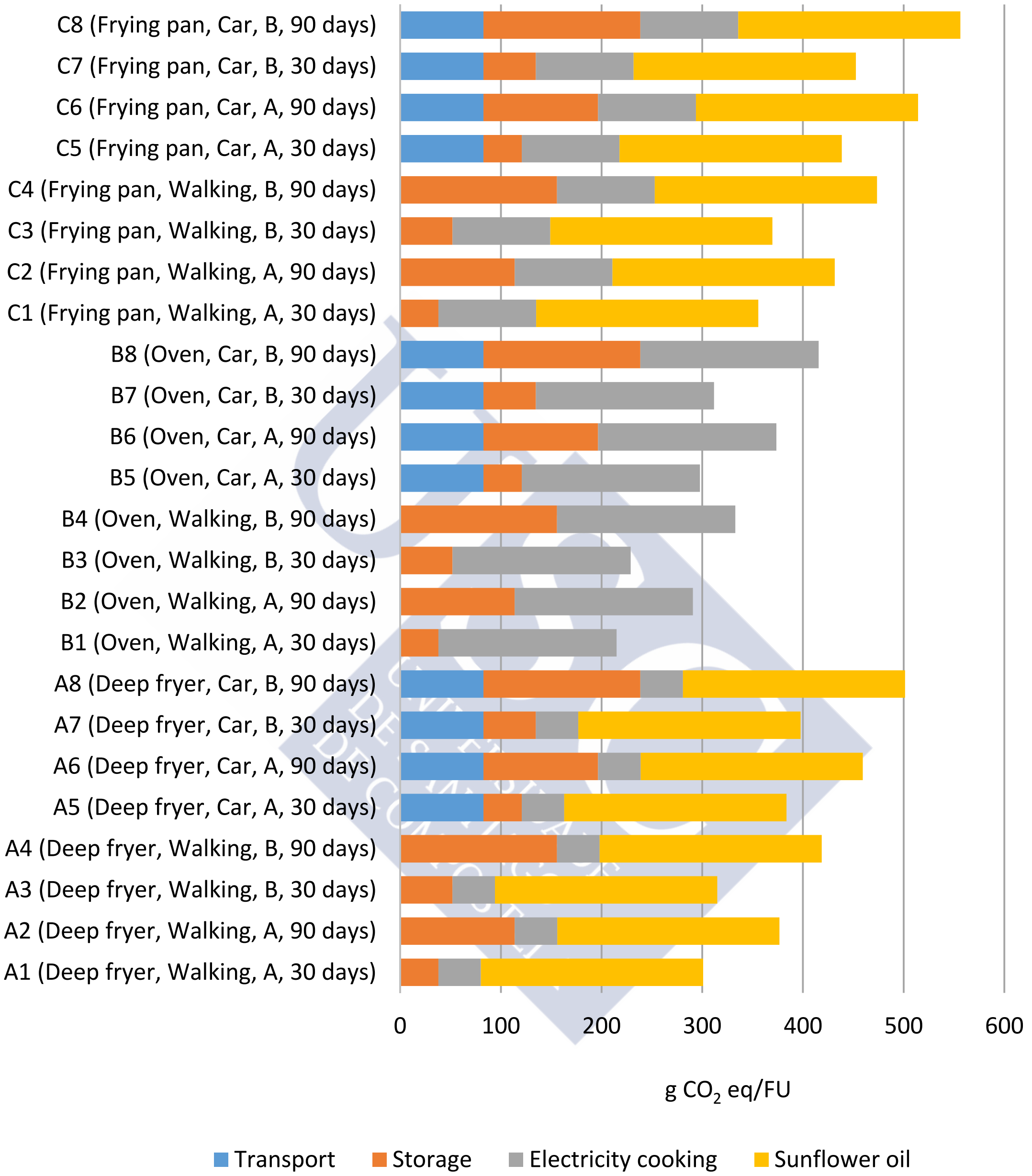

Figure 10.2. Individual carbon footprint (CF) for the selected scenarios per consumption item or activity (data per FU)

10.3.4 Environmental impacts of the entire production system

Taking into account the results obtained in Chapter 9, the CF of the entire production system was considered from a life-cycle approach, embracing all stages of production from the harvesting of raw materials to the final manufacturing stages at the production plant, distribution, wholesaling and retailing, and the purchasing and consumption inventory data included in the current study. Table E.1 of Appendix shows the life cycle inventory for the distribution step.

The final CF value, if an average of the 24 scenarios modeled in the present study is considered, was found to be 1,207.5 g CO_2 eq (Figure 10.3). Roughly 32% corresponded to the consumption stage. If the best performing scenario (B1) for the consumption stage is taken into account, the global CF of the entire system is found to be 13.9% lower. In this particular case, the relative contribution of the consumption phase is 20.7%. Finally, if the worst performing scenario (C8) is analyzed, the global CF sums up to 1,381 g CO_2 eq: 14.3% higher than the average scenario. The relative contribution of consumer activities in this scenario increases to 40.3%.

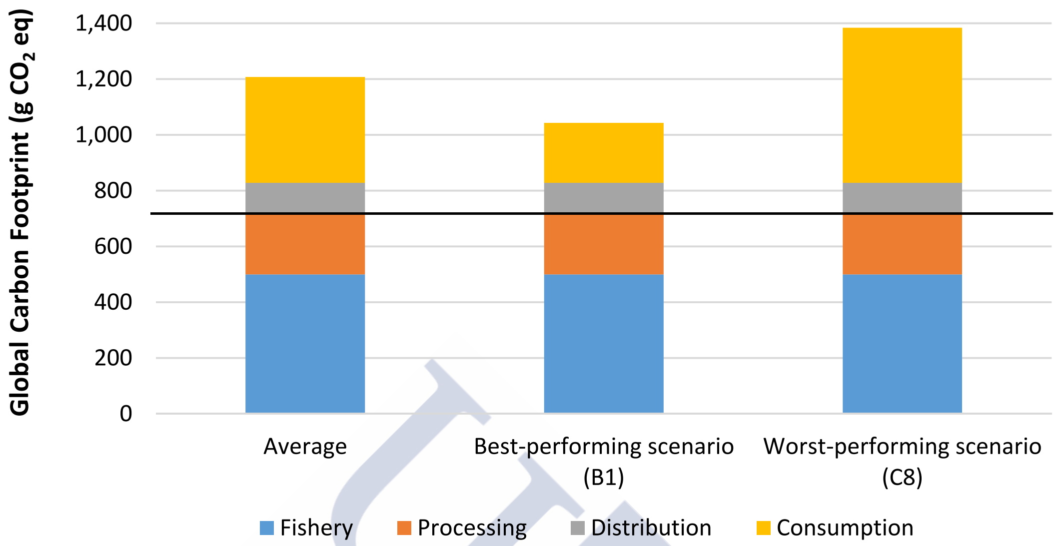

Figure 10.3. Global carbon footprint (CF) for fish stick consumption considering their entire life-cycle. Selected scenarios (data per FU)

10.4 Discussion

10.4.1 Consumption phase relevance in frozen seafood products in terms of CF

Results obtained in this research not only prove the high variability in CF that the consumption of frozen seafood can present depending on consumer behavior or needs, but also highlight the relative importance of the consumption stage within the entire supply chain in terms of GHG emissions. In fact, if the best-performing scenario is assumed, this stage accounts for roughly 20% of the total impact, ahead of the distribution and processing stages. Furthermore, if the worst-performing scenario is taken into account, consumption becomes the most GHG emissions intensive stage.

These results are relevant when considering that seafood is an important source of protein worldwide, accounting for 10% of average protein intake, which in the case of Spain increases up to approximately one third of total protein intake (Lombán González; Millán Gómez, 1998;

Tyedmers, 2004). Many studies have highlighted the high CF related to the production and processing of protein-based food products (Weber and Matthews, 2008; González et al., 2011; Virtanen et al., 2011). However, the results shown prove that, at least in some cases, consumer choices determine a significant proportion of the final emissions.

10.4.2 Contrasting the final values

If the average annual frozen seafood per capita in Spain is assumed to consist entirely of Patagonian grenadier fish sticks, an average Spaniard would consume a total of 10.2 packages of Patagonian grenadier fish sticks per year. Based on the results reported by Muñoz et al. (2010), the average Spanish diet accounted for an annual emission of 2,100 kg of CO_2 eq per person, of which 300 kg corresponded to the supply chain stages assessed here (consumer purchase and household consumption). Consequently, assuming that this average diet value includes the highest emitting scenario modeled in the current study, a reduction of 1.2% could be achieved in the diet of an individual just by improving the behavioral traits when purchasing and consuming frozen seafood (if the consumption stage is considered exclusively). While this amount may seem insignificant at first sight, it is important to note that an average Spanish diet includes roughly 873 kg of food (MAGRAMA, 2006), whereas the analysis is performed on one sole food segment: frozen seafood. In fact, the quantification of consumer abilities to reduce GHG emissions linked to their diet may be an interesting future perspective to monitor the influence of their purchase and post-purchase patterns on GHG emissions in the food sector. Nevertheless, it is important to note the high level of uncertainty that the extrapolation provided above may have, given the different methodological assumptions taken in the two studies.

In order to compare with other sources of protein, the consumption stage GHG emissions for meat products collected by Roy et al. (2012) for the Japanese market were computed and compared to the results obtained in the current study. For this comparison, 1 kg of fish sticks was compared to the consumption of 1 kg of different animal products rendered from cows and bulls (so-called 'beef'), chickens (so-called 'chicken') and pigs (so-called 'pork') (Table 10.5). The results show a similar trend in environmental impacts in terms of the consumption phase for all four products. However, due to the low GHG emissions linked to the production chain of fish sticks, the relative impact of the consumption stage is considerably higher (on average 52%) than for chicken (9.7%) or beef (1.6%).

Table 10.5. Comparison of the fish stick production system and meat production and consumption in Japan in terms of carbon footprint (CF). Data reported per 1 kg of fish flesh or meat

Product	CF per kg of meat kg CO_2	CF per 100 g of protein kg CO_2	Impact of the consumption stage %	g CO_2 eq/kg meat
Fish sticks	4.4	2.2	52	1,300–3,500
Chicken*	6	2.7	9.7	570
Pork*	6.9	3.33	8.4	5,802†
Beef*	35.6	24.25	1.63	5,802†

* Roy et al. (2012); † Roy et al. (2012) assumed the same CF values for the consumption stage of beef and pork.

Consequently, these results suggest that the importance of the consumption stage in protein food products is strongly dependent on the higher or lower environmental profile of the production and supply chain. In fact, Iribarren et al. (2010a, 2011), in a study that encompassed over 100 different fishing species landed by Galician fishing fleets, demonstrated that the extraction stage of seafood products can range from approximately 0.8 kg CO_2 eq. per kg of landed fish for some pelagic species such as horse mackerel (*Trachurus trachurus*) to 28.3 kg CO_2/kg fish in the case of Norway lobster (*Nephrops norvegicus*).

10.4.3 The importance of the consumption stage in environmental management

The results of this study not only certify the importance of the consumption stage in the global CF assessment of a seafood product, as seen in Figure 10.3, but also show that consumers, if given the correct environmental guidelines through awareness campaigns, can play an active and relevant role in the reduction of the environmental profile of seafood products through behavioral modifications when purchasing and consuming these products (Tukker et al., 2010; Caeiro et al., 2012).

In fact, a wide range of studies have analyzed consumer response to variations in energy and environmental policies to reduce GHG emissions (Brannlund and Nordströom, 2004), assessing the potential influence of different actors, such as governments, NGOs or companies (Bocken and Allwood, 2012). For instance, prior studies have highlighted several options to influence consumers to improve their environmental behavior, such as the implementation of social norms (Goldstein et al., 2008), incentives, social engagement (Carrico et al., 2011) or predetermining a specific context for consumers to take certain decisions —i.e., choice editing (Thaler and Sunstein, 2009). Other policies, such as the inclusion of Pigovian taxes (e.g., carbon tax), may also be implemented. For example, frozen seafood products could be taxed due to the expected extra shelf-time of these products in the household, increasing the CF of the consumption stage with respect to fresh seafood products. However, in the case of frozen fish sticks and other seafood products, it does not seem feasible to tax consumers, since the actions for which they are directly responsible only involve a minor proportion of the total impact. Additionally, the proposed taxation system in this particular case study would not be consistent with other stages of the supply chain. Several studies demonstrate the environmental benefits of transporting frozen seafood products (Vázquez-Rowe et al., 2012b; Ziegler et al., 2013), rather than delivering these products fresh. Consequently, even though the taxation of long shelf-time products at the consumer stage would seem desirable, if we consider that products with these characteristics are freighted in by slow means of transport, implying a considerable reduction in GHG emissions throughout the entire supply chain (Vázquez-Rowe et al., 2012b), it no longer seems appropriate to tax consumers.

Moreover, the results suggest that despite recent studies recommending that Western diets should shift to less animal protein-based products in order to decarbonize them (Garnett, 2011; González et al., 2011) and also for health reasons (Stehfest et al., 2009), additional reductions can be obtained through shifts in the way consumers handle food products once purchased (Baiocchi et al., 2010).

This should trigger the debate concerning what is more appropriate to inform consumers of: the environmental impacts associated with the production and supply chain of seafood products, which they cannot control directly and can only make a purchase decision about (product choice, brand, amount, etc.), influenced by other factors besides environmental awareness, or the different post-purchase consumption options that may contribute to reducing the global environmental impact of the product if consumed responsibly. Therefore, from a carbon eco-labeling perspective it may be more interesting to start to inform the customer on how the CF of a product can be reduced at the consumption stage through responsible consumption patterns, rather than provide CF values of the production change that the consumer cannot alter anyhow. In fact, this issue gains importance whenever the energy efficiency, and therefore the GHG emissions, of a specific fishery or fishing fleet are considered. The fishing stage in seafood products usually accounts for the greater proportion of overall emissions throughout the supply chain (Vázquez-Rowe et al., 2012a; Ziegler et al., 2013). However, a low carbon profile at this stage may not reflect a sustainable stock, since the direct effects on the targeted stocks, such as stock abundance or seafloor impact, are not accounted for (Brécard et al., 2009; Karlsen et al., 2012), which hinders the environmental relevance of a unique carbon label.

Additionally, it must be noted that in recent years a series of reports have highlighted the problem of renaming and misreporting in the seafood industry, which undoubtedly may create barriers in terms of GHG accounting and reporting (Jacquet and Pauly, 2008; Miller and Mariani, 2010; García-Vázquez et al., 2012). These practices, which imply an increased difficulty in terms of product traceability, are performed on a worldwide level in order to improve marketing or to deceive customers (Jacquet and Pauly, 2008). For instance, García-Vázquez et al. (2012) identified that approximately 30% of the hake fillets commercialized in the Greek and Spanish markets were genetically identified as belonging to a different species. Similar trends have been estimated in other studies for cod and haddock (Miller and Mariani, 2010) or sharks (Barbuto et al., 2010).

These studies not only suggest the relative success of the strict policies that regulate seafood eco-labeling in the European Union (Miller and Mariani, 2010), but also reflect the high level of illegal fishing and the shortage of catches in some popular consumed species (Jacquet and Pauly, 2008). Furthermore, the patterns observed in these studies, if channeled through the media, may create disaffection in society regarding labeling of seafood products and, therefore, reverting the trends observed in the last decade (FROM, 2011). Consequently, while it is not the intention of this study to suggest the elimination of seafood eco-labeling as understood to date, since it would simply lead to an increase in the risks regarding irresponsible behavior throughout the production chain of seafood, the relevance of the consumer stage in a major environmental dimension (i.e., CF) may provide an interesting opportunity for stakeholders to pursue a more direct involvement of customers in seafood environmental sustainability. Therefore, this may prevent a loss of interest or confidence in these mechanisms by buyers (van Amstel et al., 2008).

Nevertheless, it should be noted that the inclusion of eco-labels that seek consumer participation through their individual practices should only be implemented if notable environmental improvements can be achieved through this additional information. Otherwise, an indiscriminate use of labels could give rise to a market saturation of these items due to green-washing, priming an increase in competitive advantages in the seafood market rather than providing realistic environmental profiles and, therefore, inhibiting their benefits (Potts and Haward, 2007; van Amstel et al., 2008; Karlsen et al., 2012).

10.5 Conclusions

The environmental impacts that consumer behavior has on climate change are assessed in the current study. The selected product was linked to the consumption of frozen seafood. Results proved a high variability in the CF of frozen seafood products depending on the behavioral patterns the buyers manifest. Therefore, the inclusion of this stage in future seafood CF studies is recommended, in order to provide a complete evaluation of the products analyzed. Moreover, results suggest that substantial reductions in GHG emissions can be achieved without altering the diet patterns of the population.

Furthermore, the results presented raise the question of whether eco-labeling of seafood products should shift in life cycle environmental aspects, such as CF, in order to seek direct involvement of customers to attain significant environmental reductions. Given that an important sector of consumers values the use of eco-labels on seafood products, despite the fact that recent studies question the legitimacy and transparency of some of these labels, the inclusion of purchase and post-purchase decision-making hints for consumers on eco-labels may further improve their acceptance.

Nevertheless, while the use of carbon foot-printing has proved to be a suitable mechanism to reach the general public, allowing life cycle management to penetrate into wider sectors of society through eco-labeling and other awareness techniques, it remains unknown whether its proliferation in the food sector may hinder the visibility of other important environmental dimensions.

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Chapter 11

Life Cycle Assessment of European pilchard (*Sardina pilchardus*) consumption[1]

Summary

European pilchard or sardines (*Sardina pilchardus*) are an attractive raw material to extract from Iberian waters, since they constitute a cheap source of protein and they are a popular product among consumers. This has led to a wide range of final products available for consumers to purchase based on this single raw material. Therefore, this study presents a cross-product environmental assessment using Life Cycle Assessment of three different final products based on sardine landings: canned sardines, fresh sardines and sardine as bait to catch European hake. In addition, the products were followed throughout their entire life cycle, considering different cooking methods for each final product. Results showed high variability in environmental impacts, not only between the three final products, but also when one single product was cooked in different ways, highlighting the importance that the consumption phase and other post-landing stages may have on the final environmental profile of seafood. Results are then analyzed regarding relevant limitations and uncertainties, as well as in terms of consumer and policy implications.

[1]Vázquez-Rowe, I., Villanueva-Rey, P., Hospido, A., Moreira, M.T., Feijoo, G., 2014. Life cycle assessment of European pilchard (*Sardina pilchardus*) consumption. A case study for Galicia (NW Spain). Sci. Total Environ. 475, 48–60.

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11.1 Introduction

As remarked in Chapter 2, Spain has traditionally been an important fishing nation, not only from an extraction and processing perspective, constituting the main fishing country in Europe, but also from a seafood consumption and trade approach. For instance, the per capita consumption of seafood in Spain is the second highest in the European Union (EU), second only to Portugal (Maneiro, 2015).

Galicia has historically been an important actor within seafood extraction and trade. Its seafood industry became buoyant in the 13th and 14th centuries thanks to the abundant sardine (*Sardina pilchardus*) stocks available off its coast (Ferreira-Priegue, 1998). Although there has always been an important demand for fresh sardine consumption, the arrival of new preservation techniques in the 19th century led to the development of the canning industry (Santos-Castroviejo, 1998). Sardine canning had become one of the main drivers of the Galician economy by the late 19th century. In fact, the growth of the sardine canning industry was coincident with a period of low landings of this species in Brittany, a situation that favored international trade (Fernández-Casanova, 1998).

Currently, sardines are still an attractive seafood product in Galician and Spanish households. However, due to the great natural fluctuation it presents and to the overexploitation of the stock, which has led to fishing moratoria in the sardine stock off the coast of Portugal[2] and Galicia, other pelagic species such as horse mackerel (*Trachurus trachurus*) or Atlantic mackerel (*Scomber scombrus*) have also become popular alternatives (Cruz Ferreiro and Noguera Méndez, 1999). Nevertheless, sardines are still widely consumed fresh or canned with a ratio of 2.1 kg/(person·year) in Spain, placing them only behind tuna and hake products in annual

[2] The sardine purse seining fishery in Portugal was certified as sustainable on 14th January 2010 by the Marine Stewardship Council (MSC). However, on 12th January 2012 MSC decided to suspend the fishery certificate, considering that it no longer met the organizations principles and criteria, given the low spawning stock biomass – SSB, 62% lower than the historic mean, and the high fishing mortality in 2011. The certificate was reinstated on 24th January 2013 (MSC, 2013).

consumption (Martín-Cerdeño, 2010). Fresh sardines are consumed mainly in the summer months (June to early October) when their fat content is higher, improving their taste and aroma (Zlatanos and Laskaridis, 2007). In fact, *San Xoán* festivity at the end of June, which coincides with other mid-summer commemorations throughout Europe, has people gathered around bonfires to eat sardines and drink wine in Galicia and in the North of Portugal (Johnson, 2007).

Sardines captured from October to late spring are mainly destined to the canning industry or to bait. For instance, in 2003 a total of 15,229 tonnes of sardine and sardinella were used for seafood canning in Spain, creating economic revenue of approximately 87 million euros (ANFACO-CECOPESCA, 2014).

The alleged overexploitation of the sardine stock in the fishery off the coast of Portugal and NW Spain has forced many canning companies to import sardine from other countries or *Sardina sardina* from the Mediterranean Sea (Xunta de Galicia, 2005). Nevertheless, Galician consumers and many companies as part of their market strategy still prefer using local captures (i.e. *Sardina pilchardus*).

The main objective of the present study was to determine the environmental profile of the entire life-cycle of selected seafood products derived from European pilchard (i.e. sardine) extraction by Galician purse seiners from an LCA perspective, in order to detect the main hot spots in the production systems. Furthermore, given that seafood products derived from sardine captures in Galicia are consumed in several manners, a cross-product assessment is provided based on an equal amount of protein supply in order to understand the potential differences in environmental impact when an identical raw material (i.e. sardines) is processed or consumed in different ways. Finally, a set of alternative scenarios are given in order to determine how the environmental burdens associated with the production systems can be minimized. These goals aim to constitute a guideline for decision-making in the seafood canning and baiting industries, a reference point for future programs to reduce environmental impacts in this sector and a first step to harmonize the environmental profile of food products based on their protein supply to human populations. Therefore, despite the raw material perspective provided in this study, it may also constitute an interesting study for consumers to understand how the optimization of nutrients of food products may also determine the environmental impacts of dietary patterns (González et al., 2011).

11.2 Materials and Methods

11.2.1 Goal and scope definition. Functional unit

The function of the production system under analysis is principally that of nourishing the human population, due to the variety of positive aspects that have been linked to the consumption of fish in general (Budtz-Jørgensen et al., 2007; Sabinsky et al., 2012) and, more specifically, of consuming seafood products derived from the fishing of one single raw material: sardines caught by Galician purse seiners (Cardenia et al., 2013). Therefore, the functional unit (FU) selected was set as the amount of protein (17.26 g) supplied by one can of sardines (85.0 g) in olive oil produced by a Galician canning industry with an annual production of approximately 850 tonnes —Scenario A (hereafter referred to as Sc. A). This FU selection is based on the fact that the main advantage that human populations obtain from fisheries is the protein supply, which constitutes an essential part of human diets (Pimentel and Pimentel, 2003; Tyedmers, 2004; Pelletier et al., 2011; FAO, 2012). For the alternative scenarios, equal amounts of final protein content for human consumption were fixed (i.e. 17.26 g of protein). On the one hand, for the fresh consumption scenario (Scenario B —hereafter Sc. B) a total of 137.1 g of round sardine were assumed, since it is the equivalent amount of protein content delivered to the consumer. On the other hand, for the bait scenario (Scenario C —hereafter Sc. C), the consumer does not directly intake the sardine individuals, since they are previously used as bait to catch another species. Hence, for this case study, the equivalent protein content for European hake (*Merluccius merluccius*) caught by long liners in the Northern stock was modelled; i.e. 183.0 g of hake caught by using a total of 75.2 g of sardine as bait. Table 11.1 and Table 11.2 show detailed data regarding products analyzed and protein content.

Table 11.1. Edible Meat Fraction, Fillet Yield and Protein Content of selected fish species (source: Prof. Peter Tyedmers, personal communication)

Fish species		Edible meat fraction (%)	Fillet yield (%)	Protein content (%)
English name	***Scientific name***			
European hake	*Merluccius merluccius*	53	41	17.8
European pilchard	*Sardina pilchardus*	62	50	20.3

Table 11.2. Edible composition of the final selected seafood products

Product	**Unit**	**Scenario A**	**Scenario B**		**Scenario C**	
		Canned sardines	**Fried sardines (using olive oil and flour)**	**Sardine BBQ**	**Fried hake**	**Boiled hake**
European pilchard	g	85.00	137.10	137.10	--	--
European hake	g	--	--	--	183.00	183.00
Olive oil	mL	32.06	9.60	--	13.73	--
Salt	g	--	0.66	0.66	0.46	0.46

The production system, as observed in Figure 11.1, starts with the extraction of the raw material (sardines) by Galician purse seining vessels, includes landing and auction of the seafood, transportation and the different product chains associated with the three scenarios defined, and ends with the final consumption of the correspondent seafood product, including household consumption, end of life processes and human excretion.

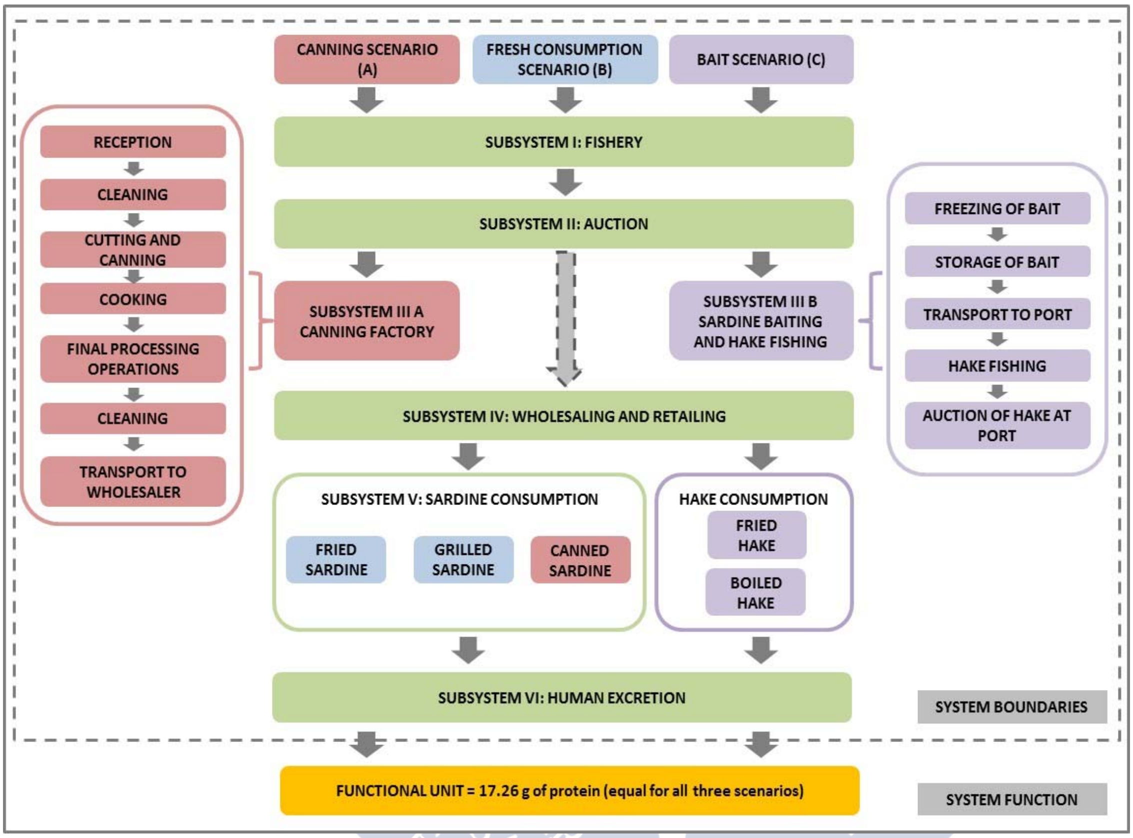

Figure 11.1. Block diagram of the analyzed system per scenario

11.2.2 System description

As presented above (Figure 11.1), the production system is divided into 6 different subsystems for scenarios A and C, while Sc. B (i.e. fresh sardine consumption) is comprised of 5 different subsystems, since there was no processing stage considered after sardine landing at port. The minor processing activities that take place in this scenario are included in the wholesaling (i.e. packaging) and retailing and consumption stages (i.e. different levels of cleaning and cutting).

Subsystem I, common to all three scenarios, includes the extraction of sardines by purse seiners in the coastal fishery in Galicia (ICES Divisions VIIId and IXa) and the landing of the catch at a Galician port. Subsystem II, also common to the three scenarios, encompasses the auction of the catches at a Galician port, as well as the transport of the sardines to the wholesaling/processing point. As aforementioned, subsystem III only applies to Scs. A and C. For Sc A (i.e. SS3.A), it consists of the set of processes that are developed at the canning factory:

- Reception and storage (SS3.A1). Discharging operations from the trucks and storage of the catch in cooling chambers. Other raw materials are stored in conventional areas.
- Cleaning (SS3.A2). This stage consists of the cleaning and salting of the sardines to maintain sardine quality. These operations facilitate handling by the personnel.
- Cutting and canning (SS3.A3). First, any undesired sections of the sardines are cut away and discarded. The boxes containing the sardines are placed manually on the conveyor belt to facilitate handling by the personnel in charge of heading, cutting and eviscerating the individual fish. Thereafter, sardines are washed individually to ensure their quality, and introduced manually in cans. Each can contains three sardines.
- Cooking (SS3.A4). The raw material is subject to thermal treatment at atmospheric pressure. The cans are placed on a conveyor belt and enter the cooker. The cooking stage lasts approximately one hour.
- Final processing operations (SS3.A5). Olive oil and salt are added to the cans through automated dispensers prior to sealing. Thereafter, the cans are cleaned with hot water (50°C) and a small dose of detergent to eliminate dirt. Finally, sterilization, a thermal treatment to destroy or inhibit any microorganisms that may proliferate, is carried out.
- Packaging (SS3.A6). The sardine cans are automatically wrapped in individual cardboard boxes and then introduced in larger boxes that are later filmed, palletized and stored at room temperature.

- Ancillary operations and transport to the regional distribution center —RDC (SS3.A7). Once the pallets are stored, they are transported in trucks to the distribution center, situated 35 km away from the canning factory.

A more detailed description of the generic operations linked to the Galician seafood canning industry can be consulted in Bello-Bugallo et al. (2013).

In Subsystem III of Sc. C (i.e. SS3.B), the sardine was assumed to be transported to a freezing company where the load is frozen and stored for future use as bait. Thereafter, the bait is distributed to the fishing vessels that require it for their operations. In this particular case study, the bait was assumed to be used by long lining vessels in the Northern Stock whose main target species is European hake. Furthermore, operations linked to hake fishing in the Northern Stock, as well as landing of the final product and auction operations, were included within this subsystem.

Subsystem IV includes the wholesaling and retailing operations of each of the selected alternative scenarios, such as transportation, electricity and storage operations. Retailing was considered throughout the Galician territory. However, Scenarios B and C do not comprise wholesaling, since the main retailers purchase the catch at the port. Subsystem V embraces the consumption operations by household consumers in all three scenarios, as well as the end-of-life (EOL) operations. Finally, subsystem VI covers the activities related to impacts from human excretion (e.g. wastewater treatment, sludge spreading or detergent), based on the assumptions performed by Muñoz et al. (2008).

11.2.3 Data acquisition

The data used for the fishery (Subsystem I) were obtained from a prior study that analyzed the Galician purse seining fleet from an LCA perspective (Vázquez-Rowe et al., 2010), where a total of 30 vessels targeting sardine were considered when carrying out the average inventory.

Data regarding the landing and auction stage were obtained from the auction at the port in Vigo (Autoridade Portuaria de Vigo, personal communication, September 2010). However, it is important to note that only a portion of the sardines caught by the inventoried vessels are landed in Vigo, whereas another important proportion is landed in smaller ports such as Ribeira, Sada or Portosín (Figure 11.2). Therefore, a scale factor may be partially influencing these data, an issue that is not accounted for in the inventory provided.

For the processing stage, inventory data for the canning factory (SS3.A) were mainly primary data provided by a canning factory located in Cabo de Cruz (A Coruña province; 42°36′55″N 8°53′8.62″O), where approximately 95% of the production consists of sardine canning. For SS3.B, operations regarding sardine use for bait in long-liners were obtained from a representative baiting company located in Foz (Lugo province; 43°33′0″N 7°18′0″W). Moreover, fishing operations of long liners were retrieved through personal questionnaires with skippers. A total of 12 different skippers were interviewed, representing roughly 25% of the entire fishing fleet. These data had previously been reported by Vázquez-Rowe et al. (2011a).

The inventory data used for the remaining subsystems were retrieved mainly from bibliographical sources (e.g. olive oil — Papadakis, 2006; Salome and Ioppolo, 2012) and from personal interviews with fishmongers (Praza de Abastos de Santiago de Compostela, personal comm., January 2013). More details on the data acquisition sources and assumptions for these subsystems are available in Section 11.2.6. Finally, background data relating to energy and raw materials, such as olive oil or salt, provision and disposal scenarios were obtained from the ecoinvent® database v2.2 (Frischknecht et al., 2007).

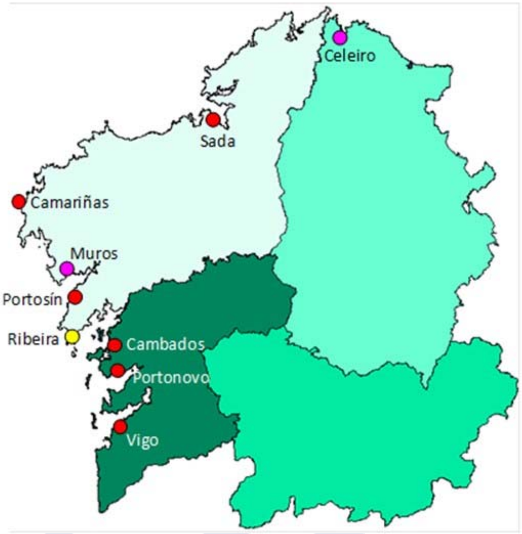

Figure 11.2. Map showing the fishing ports at which the inventoried vessels are based. Red ports refer to purse seining vessels; violet ports refer to trawling vessels; yellow ports refer to both trawling and purse seining vessels being inventoried

11.2.4 Life cycle inventory

The life cycle inventory (LCI) provides the collection and computation of the data to quantify the considered inputs and outputs in the production system, in order to calculate their contribution to the final environmental impacts (ISO, 2006a). For this particular study, the inventory data were divided into the six main subsystems, as shown in section 11.2.3 (Figure 11.1). Detailed inventories per subsystem for the computed scenarios are shown below (Table 11.3 to Table 11.9). Additionally, inventory data for Subsystem VI: human excretion can be consulted in Table F.1 of Appendix.

Table 11.3. Average inventory data for Subsystem I: fishery. Data reported per FU = 17.26 g of protein

	Units	Scenario A			Scenario B	Scenario C
		Economic revenue allocation	***Economic savings allocation***	***Energy allocation***		
		Inputs				
Diesel	g	39.89	33.91	29.14	24.13	13.24
Steel	g	0.61	0.52	0.45	0.37	0.20
Seine net	g	2.31	1.97	1.69	1.40	0.77
Anti-fouling	mg	82.73	70.32	60.43	50.04	27.45
Ice	g	72.76	61.85	53.14	44.01	24.14
		Outputs				
Sardines (live weight)	g	226.67	192.67	165.56	137.10	75.20
CO_2	g	126.48	107.51	92.38	76.50	41.96
SO_2	g	0.41	0.35	0.30	0.25	0.14
NO_x	g	2.95	2.50	2.15	1.78	0.98
R22	mg	5.28	4.49	3.86	3.19	1.75

Table 11.4. Average inventory data for Subsystem II: port auction. Data reported per FU = 17.26 g of protein

	Units	Scenario A			Scenario B	Scenario C
		Economic revenue allocation	***Economic savings allocation***	***Energy allocation***		
		Inputs				
Sardines (live weight)	g	226.67	192.67	165.56	137.10	75.20
Pallets	p	$2.25\cdot10^{-4}$	$1.91\cdot10^{-4}$	1.64·10-4	$1.36\cdot10^{-4}$	$7.45\cdot10^{-5}$
Polystyrene	mg	319.80	271.83	233.58	193.43	106.10
Detergent	mg	10.54	8.96	7.70	6.38	3.50
Fish boxes	mg	207.34	176.24	151.44	125.41	68.79
Electricity	kWh	$3.51\cdot10^{-3}$	$2.98\cdot10^{-3}$	2.56·10-3	$2.13\cdot10^{-3}$	$1.17\cdot10^{-3}$
		Outputs				
Sardines (live weight)	g	226.67	192.67	165.56	137.10	75.20

Table 11.5. Inventory data for Subsystem III.A: canning factory. Data reported per FU = 17.26 g of protein

	Subsystem section	Units	Scenario A		
			Economic revenue allocation	***Economic savings allocation***	***Energy allocation***
		Inputs			
Electricity					
Elevator	SS3.A1	kWh	$3.32\cdot10^{-4}$	$2.82\cdot10^{-4}$	$2.42\cdot10^{-4}$
Elevator	SS3.A2	kWh	$9.97\cdot10^{-4}$	$8.47\cdot10^{-4}$	$7.28\cdot10^{-4}$
Pump	SS3.A3	kWh	$5.05\cdot10^{-3}$	$4.29\cdot10^{-3}$	$3.69\cdot10^{-3}$
Conveyor belt	SS3.A3	kWh	$5.05\cdot10^{-3}$	$4.29\cdot10^{-3}$	$3.69\cdot10^{-3}$
Conveyor belt	SS3.A4	kWh	$1.40\cdot10^{-3}$	$1.40\cdot10^{-3}$	$1.40\cdot10^{-3}$
Boiler	SS3.A4	kWh	$1.97\cdot10^{-3}$	$1.97\cdot10^{-3}$	$1.97\cdot10^{-3}$
Sealing	SS3.A5	kWh	$5.33\cdot10^{-4}$	$5.33\cdot10^{-4}$	$5.33\cdot10^{-4}$
Cleaning pump	SS3.A5	kWh	$3.93\cdot10^{-3}$	$3.93\cdot10^{-3}$	$3.93\cdot10^{-3}$
Autoclave	SS3.A5	kWh	$1.03\cdot10^{-2}$	$1.03\cdot10^{-2}$	$1.03\cdot10^{-2}$
Antioxidant cleaning pump	SS3.A5	kWh	$2.29\cdot10^{-3}$	$2.29\cdot10^{-3}$	$2.29\cdot10^{-3}$
Cleaning pump	SS3.A5	kWh	$1.99\cdot10^{-3}$	$1.99\cdot10^{-3}$	$1.99\cdot10^{-3}$
Conveyor belt	SS3.A5	kWh	$3.37\cdot10^{-3}$	$3.37\cdot10^{-3}$	$3.37\cdot10^{-3}$
Packaging	SS3.A6	kWh	$1.68\cdot10^{-3}$	$1.68\cdot10^{-3}$	$1.68\cdot10^{-3}$
Packaging robots	SS3.A6	kWh	$1.90\cdot10^{-5}$	$1.90\cdot10^{-5}$	$1.90\cdot10^{-5}$
Cutting wheel	SS3.A6	kWh	$1.98\cdot10^{-3}$	$1.98\cdot10^{-3}$	$1.98\cdot10^{-3}$
Elevator	SS3.A6	kWh	$6.64\cdot10^{-4}$	$6.64\cdot10^{-4}$	$6.64\cdot10^{-4}$
Ancillary operations	SS3.A7	kWh	$8.97\cdot10^{-3}$	8.97·10-^{3}	$8.97\cdot10^{-3}$
Transportation					
Salt (>16 t lorry)	SS3.A2	tkm	$3.30\cdot10^{-6}$	$2.81\cdot10^{-6}$	$2.41\cdot10^{-6}$
Tin (>16 t lorry)	SS3.A3	tkm	$1.00\cdot10^{-3}$	$8.50\cdot10^{-4}$	$7.30\cdot10^{-4}$
Lids (>16 t lorry)	SS3.A5	tkm	$1.00\cdot10^{-3}$	$1.00\cdot10^{-3}$	$1.00\cdot10^{-3}$
Salt (>16 t lorry)	SS3.A5	tkm	$7.00\cdot10^{-7}$	$7.00\cdot10^{-7}$	$7.00\cdot10^{-7}$
Olive oil (>16 t lorry)	SS3.A5	tkm	$3.66\cdot10^{-2}$	3.66·10-2	$3.66\cdot10^{-2}$
Cardboard box (>16 t lorry)	SS3.A6	tkm	$3.36\cdot10^{-3}$	$3.36\cdot10^{-3}$	$3.36\cdot10^{-3}$
Glue (>16 t lorry)	SS3.A6	tkm	$2.76\cdot10^{-3}$	$2.76\cdot10^{-3}$	$2.76\cdot10^{-3}$

Table 11.5. Inventory data for Subsystem III.A: canning factory. Data reported per FU = 17.26 g of protein (continuation)

	Subsystem section	Units	Scenario A		
			Economic revenue allocation	***Economic savings allocation***	***Energy allocation***
		Inputs			
Transportation					
Adhesive tape (>16 t lorry)	SS3.A6	tkm	$3.60 \cdot 10^{-6}$	$3.60 \cdot 10^{-6}$	$3.60 \cdot 10^{-6}$
Tin plates (>16 t lorry)	SS3.A6	tkm	$4.48 \cdot 10^{-3}$	$4.48 \cdot 10^{-3}$	$4.48 \cdot 10^{-3}$
Film	SS3.A6	tkm	$2.51 \cdot 10^{-6}$	$2.51 \cdot 10^{-6}$	$2.51 \cdot 10^{-6}$
Bleach	SS3.A7	tkm	$9.22 \cdot 10^{-5}$	$9.22 \cdot 10^{-5}$	$9.22 \cdot 10^{-5}$
Lubricant oil	SS3.A7	tkm	$3.89 \cdot 10^{-6}$	$3.89 \cdot 10^{-6}$	$3.89 \cdot 10^{-6}$
Detergent	SS3.A7	tkm	$7.18 \cdot 10^{-6}$	$7.18 \cdot 10^{-6}$	$7.18 \cdot 10^{-6}$
Other inputs					
Sodium chlorate	SS3.A2	mg	38.00	32.30	27.76
Tin	SS3.A2	g	19.50	16.58	14.24
Sodium chlorate	SS3.A5	mg	8.00	8.00	8.00
Tin	SS3.A5	g	15.10	15.10	15.10
Olive oil	SS3.A5	g	35.00	35.00	35.00
Tap water	SS3.A5	g	19.09	19.09	19.09
Tap water	SS3.A5	g	497.29	497.29	497.29
Tap water	SS3.A5	g	114.56	114.56	114.56
Tap water	SS3.A7	g	1245.7	1245.7	1245.7
Packaging					
Copolymer (ethylene vinyl acetate)	SS3.A6	g	4.00	4.00	4.00
Corrugated board	SS3.A6	g	19.99	19.99	19.99
Core board	SS3.A6	g	15.00	15.00	15.00
Polypropylene	SS3.A6	g	0.10	0.10	0.10
Packaging film	SS3.A6	g	0.07	0.07	0.07
EUR-flat pallet	SS3.A6	p	0.04	0.04	0.04

Table 11.5. Inventory data for Subsystem III.A: canning factory. Data reported per FU = 17.26 g of protein (continuation)

	Subsystem section	Units	Scenario A		
			Economic revenue allocation	***Economic savings allocation***	***Energy allocation***
			Inputs		
Ancillary operations					
Light fuel oil	SS3.A7	g	276.81	276.81	276.81
Sodium hypochlorite	SS3.A7	mg	77.00	77.00	77.00
Soap	SS3.A7	g	206.00	206.00	206.00
Lubricating oil	SS3.A7	g	46.00	46.00	46.00
			Outputs		
European pilchard can	SS3.A6	p	1	1	1
Co-products					
Biomass to fish meal (cutting)	SS3.A3	g	90.67	90.67	90.67
Biomass to fish meal	SS3.A4	g	51.00	51.00	51.00
Wastes to treatment					
Film residues	SS3.A6	mg	70.00	70.00	70.00
Salt residues	SS3.A3	mg	34.00	28.90	24.83

SS3.A1= Reception and storage; SS3.A2= Cleaning; SS3.A3= Cutting and canning; SS3.A4= Cooking; SS3.A5= Final processing operations; SS3.A6= Packaging; SS3.A7= Ancillary operations and transport to the regional distribution center.

Table 11.6. Inventory data for Subsystem III.B: baiting. Data reported per FU = 17.26 g of protein

	Units	**Scenario C**
Inputs		
Sardines (live weight)	g	75.20
Paperboard	g	1.25
Polyethylene (LDPE)	g	2.50
Detergent	mg	76.62
Energy		
Electric energy	kWh	0.03
Transport		
Fresh sardine to wholesale	kg·km	5.39
Frozen sardine to long liner loading	kg·km	9.36
Outputs		
Products		
Bait (European pilchard)	g	75.20

Table 11.7. Average inventory data for Subsystem III.B: hake fishery. Data reported per FU = 17.26 g of protein

	Units	Scenario C
Inputs		
Materials and fuels		
Diesel	g	238.9
Steel	g	2.57
Ice	g	117.7
Boat paint	g	0.12
Anti-fouling paint	g	0.32
Marine lubricant oil	g	1.02
Outputs		
Products		
European hake	g	183.0
Discards		
Discarded fish	g	38.59
Emissions to the atmosphere		
CO_2	g	757.3
SO_2	g	2.39
VOC	g	0.57
NO_x	g	17.20
CO	g	1.77
Emissions to the ocean		
Xylene	mg	31.38
Sea nine 211	mg	3.19
Ethylbenzene	mg	7.46
Wastes to treatment		
Plastic to recycling	g	0.32
Cardboard to recycling	g	0.52
Plastic to landfill	g	1.20
Cardboard to landfill	g	0.24

Table 11.8. Inventory data for Subsystem IV: wholesaling and retailing. Data reported per FU = 17.26 g of protein

	Units	Scenario A			Scenario B	Scenario C
		Economic revenue allocation	***Economic savings allocation***	***Energy allocation***		
Inputs						
Sardines (fresh)	g	--	--	--	137.10	--
Sardines (canned)	g	85.00	85.00	85.00		
European hake	g	--	--	--	--	183.00
Materials						
Polyethylene (HDPE)	g	0.10	0.10	0.10		3.55
Energy						
Electric energy	kWh	0.01	0.01	0.01	0.02	0.03
Transport						
Lorry transport, to wholesale	tkm	0.20	0.20	0.20	0.07	0.11
Van transport, to retailer	kg·km	12.45	12.45	12.45	7.34	10.13
Outputs						
Sardines (live weight)	g	85.00	85.00	85.00	137.10	183.00

Table 11.9. Inventory data for Subsystem V: consumption phase and end-of-life (EOL). Data reported per FU = 17.26 g of protein

	Unit	**Fried sardines (using olive oil and flour)**	**Sardine BBQ**	**Canned sardines**	**Fried hake**	**Boiled hake**
			Inputs			
Fried sardine	g	137.10	--	--	--	--
Grilled sardine	g	--	137.10	--	--	--
Canned sardine	g	--	--	85.00	--	--
Fried hake	g	--	--	--	183.00	--
Boiled hake	g	--	--	--	--	183.00
Olive oil	mL	9.60	--	--	13.73	--
Salt	g	0.66	0.66	--	0.46	0.46
Flour	g	4.66	--	--	4.58	--
Water	mL	--	--	--	--	183.0
Wood	g	--	0.14	--	--	--
Electricity (cooking)	kWh	0.52	--	--	0.14	0.12
Electricity (storage)	kWh	$4.44 \cdot 10^{-4}$	$4.44 \cdot 10^{-4}$	--	$7.41 \cdot 10^{-4}$	$7.41 \cdot 10^{-4}$
Consumer transport	km	0.14	0.14	0.25	0.18	0.18
			Outputs			
Residues						
Tin (recycling)	g	--	--	29.34	--	--
Tin (municipal solid waste)	g	--	--	5.26	--	--
Cardboard (recycling)	g	--	--	11.10	--	--
Cardboard	g	--	--	3.90		--
Biodiesel	mL	--	--	--	4.12	--
COD (wastewater treatment plant)	g COD	--	--	101.0	19.42	--

11.2.5 Allocation and other assumptions

Allocation constitutes a crucial discussion point in most multispecies fisheries, as proved by the on-going debate in literature (Pelletier and Tyedmers, 2011; Weinzettel, 2011; Vázquez-Rowe et al., 2012). Two different allocation hotspots were identified in the production systems analyzed. On the one hand, allocation was necessary in Subsystem I due to the multispecies nature of the fishing stage. Mass-based allocation was the selected approach according to the recommendations gathered in ISO 14044 in order to account for some type of physical relationship between the co-products (ISO, 2006b). In addition, other allocation alternatives were disregarded due to the similar economic value of the landed species (all small-pelagic fish), their similar protein content (Peter Tyedmers, personal communication, 2011) and final consumption patterns, as in previous LCA studies of small-pelagic seafood (Vázquez-Rowe et al., 2010). On the other hand, in Scenario A an important portion of the live weight of the sardines is finally disposed of as an intermediary residue during the canning process (mainly animal fat and offal). These residues are collected, free of charge, by a specialized residue management company for use in the turbot aquaculture industry. Three different allocation approaches were considered and compared (Table 11.10): i) a first approach based on economic allocation; given that all the economic revenue of the factory is provided by the sardine cans, all the environmental impacts are assigned to this product; ii) a second perspective that also uses economic allocation, but includes the economic savings linked to the free service by the fish meal company. Hence, 85% of the environmental impact was assigned to the canned sardines, while the remaining 15% was allocated to the offal and fat residues; and iii) a final approach based on energy allocation, taking into account the fact that the final use of the residues derives in a certain amount of food available to nourish humans (Figure 11.3). In the latter approach, the environmental burdens were allocated based on the relation between the final protein content of the canned sardines and that of the turbot fed on fish meal, which was produced partially by canning organic residues (Table 11.10). For this perspective, a set of assumptions were considered regarding reduction and turbot feeding (section 11.2.6 delves into assumptions made along the study) leading to a final allocation of 73% of environmental impacts for the canned sardines (Iribarren et al., 2010b, 2012; Peter Tyedmers, personal communication, 2011; Leticia Regueiro, personal communication, 2012). Finally, a mass allocation approach was disregarded, since this would imply that the main product of the production system would be assigned lower environmental impacts than the residues (Ayer et al., 2007; Pelletier and Tyedmers, 2011).

Table 11.10. Allocation values in Scenario A for the selected allocation perspectives

	Economic revenue	**Economic savings**	**Energy (protein content)**
Canned sardines	100%	85%	73.04%
Sardine residues (fats and offal)	0%	15%	26.96%

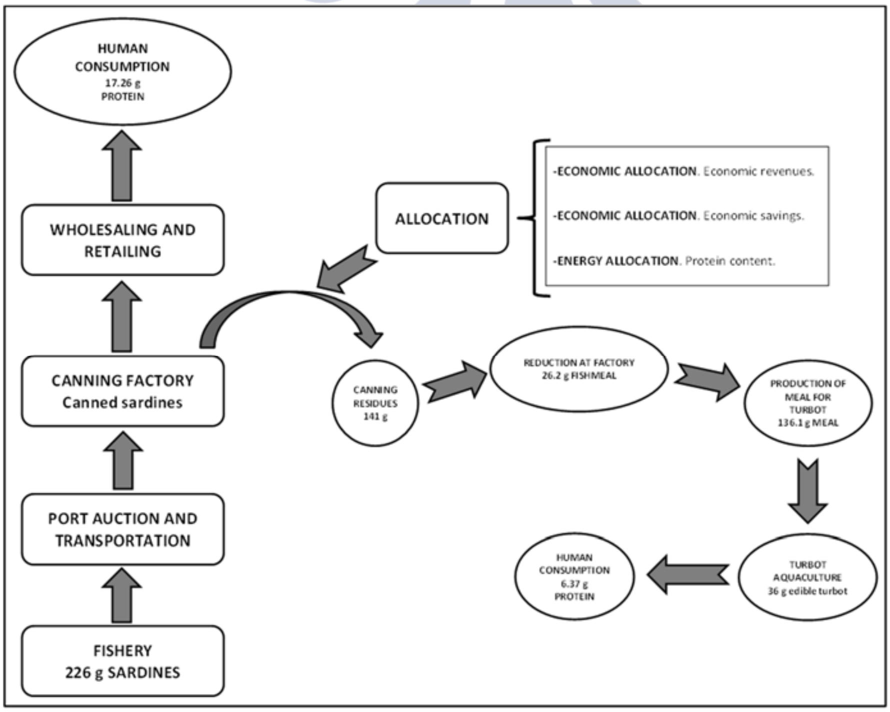

Figure 11.3. Schematic representation of the allocation approaches considered in Scenario A

11.2.6 Assumptions

Organic waste

Organic wastes linked to offal, heading and cutting are collected at the canning factory for their use in the production of feed for aquaculture products in the region. In this study, we assumed that these residues are used for the production of feed for turbot aquaculture plants. Hence, for each 226 g of European pilchard entering the canning factory, 141 g become residues (Figure 11.3). These residues are then transported to a reduction factory within the region to produce fishmeal. A conversion rate of 186 kg of fishmeal per tonne of canning residues was assumed, based on direct enquiry at CONRESA, a fishmeal factory in Ribeira, Spain (Leticia Regueiro, personal communication).

Fishmeal arriving from sardine residues is then mixed with other feed components to provide nourishment in turbot aquaculture. The contacted stakeholders affirmed that the proportion of fishmeal from sardine residues is roughly 20% of the total (Iribarren et al., 2012; Leticia Regueiro, personal communication).

The final feed to deliver to the turbot aquaculture plant (136.1 g) allows the nourishment of 36 g of edible turbot (Iribarren et al., 2012), which based on the edible content of turbot (41%) and its protein content per 100 g (17.7 g), implying a final 6.37 g of protein that humans finally consume from the residues linked to the production of one single can of European pilchard (Peter Tyedmers, personal communication).

Transport to retailing centre

For all scenarios and final consumed products, transport by bus was assumed. A total purchase of 2 kg of European pilchard or European hake, depending on the scenario, was considered. The travelled distance by public transportation was assumed to be 2 km.

Storage of products at household

The storage of the seafood products in the household was assumed to be a 24 hour period in an A Energy Class refrigerator. A volume of approximately 3 L was taken into consideration for the introduction of 2 kg of fish into the refrigerator. It should be noted that these values were then rescaled in the LCI to the amount of seafood consumed in each scenario.

Cooking of the seafood products

The cooking time for each of the different consumed seafood products is shown in Table 11.11. These time periods were used to assign the amount of electricity consumed for the induction burners, the smoke extractor and the light. However, it should be noted that the smoke extractor was only working at full power for fried sardine, whereas for fried and boiled European hake, medium power was assumed.

Table 11.11. Cooking times for the selected final seafood products consumed in an average household. Time reported per FU

Scenario	Seafood product	Unit	Time
Scenario A	Canned sardine	min/FU	0
Scenario B	Fried sardine	min/FU	2.74
	Barbeque	min/FU	N/Ap*
Scenario C	Boiled hake	min/FU	1.83
	Fried hake	min/FU	1.37

*N/Ap= not applicable

- Scenario A. Canned sardines. It was assumed that the can of sardines is opened by the consumer and eaten directly from the can. The olive oil contained in the can is drained in the sink in the kitchen, although a small portion always remains covering the sardines. We assumed that the entire amount of sardines is ingested by the consumer. Therefore, no organic wastes of European pilchard are generated in this stage. The disposal of the tin and cardboard, as well as the COD originated from the drainage of the olive oil were included in the LCI as waste outflows (see Table 11.8). Data for the recycling of tin were obtained from Ecoacero (Ecoacero, 2013), assuming a recycling rate of 84.8% for 2011. Data for recycling paper were obtained from Repacar (REPACAR, 2013). A recovery rate of 73.9% was assumed for 2011. Finally, COD computation was based on the conversion factor calculated by Hospido et al. (2006).

- Scenario B. Fried sardines. Electricity consumption for this final edible product includes the energy consumed by the smoke extractor: 0.125 kW and the cooking light: 0.028 kW (IKEA, 2013), as well as the induction burners: 9,000 W (Bosch, 2013). The generated residues include the non-edible organic waste from the European pilchard (roughly 38% of the live weight of the sardines), as well as flour and oil covering these non-edible portions. For the olive oil use in the frying stage, we assumed that 36% of this oil was recovered, according to the Ministry of Agriculture, Food and Environment (MAGRAMA, 2013). The remaining part of the olive oil was assumed to be drained down the kitchen's sink. Therefore, it was computed as COD as suggested by Hospido et al. (2006).

- Scenario B. Barbeque of sardines. Very low consumption of inventory items (see Table 11.8 in Section S4). Slat was used and we assumed that 2 kg of wood are needed to prepare 2 kg of sardines.

- Scenario C. Fried hake. Inventory inputs (i.e. flour and olive oil) for frying European hake are considerably lower than for frying sardines due to the different total frying surface (i.e. it is not the same to batter 20 sardines as to batter 2-4 hake fillets).

- Scenario C. Boiled hake. Energy consumption for cooking is lower as compared to fried hake, but not for the preparation of the hake fillets, since induction burners are needed in this stage.

Human excretion

For human excretion, as shown in Table F.1 of Appendix, a disaggregation in terms of food products (i.e. flour, fish, oil) was needed in terms of macronutrients: proteins, carbohydrates, lipids, water, fiber, etc. Data for this disaggregation were obtained from a database developed and provided by Professor Peter Tyedmers, from the School of Resources and Environmental Studies (SRES) at Dalhousie University (Peter Tyedmers, personal communication) and from the online database of an important Spanish supermarket (EROSKI, 2013). Once these data were gathered, the model suggested by Muñoz et al. (2008) was applied. In addition to the characteristics of the macronutrients, the model takes into consideration the amount of time and the type of intake to perform an allocation of different inputs, such as toilet paper, electricity and water to wash hands, soap or towels. Finally, the model also considers the inputs required for wastewater treatment.

11.2.7 Life cycle impact assessment

The analysis of the midpoint environmental impacts, as well as endpoint damages was performed by using ReCiPe (Goedkoop et al., 2009). To do so, Simapro version 7.3 was the software selected to carry out the LCA calculations (Prè-Product Ecology Consultants, 2011).

11.3 Results

11.3.1 The supply chain of canned sardine consumption – Scenario A

The total environmental impacts using ReCiPe endpoint for this production system ranged from 0.52 to 0.58 Pt depending on the allocation method used to assign the canned and residual sardine biomass (Figure 11.4; see also Table F.2 of Appendix). Therefore, the global environmental profile of canned sardines only showed variations of up to 11% due to the methodological allocation choices. The main impact categories that contributed to the total burdens were mainly those linked to resources (mineral and fossil depletion – MD and FD), climate change (CC) and particulate matter formation (PMF) within the human health categories and, finally, in terms of impacts on the ecosystem, agricultural land occupation (ALO) and to a lesser extent, natural land transformation (NLT).

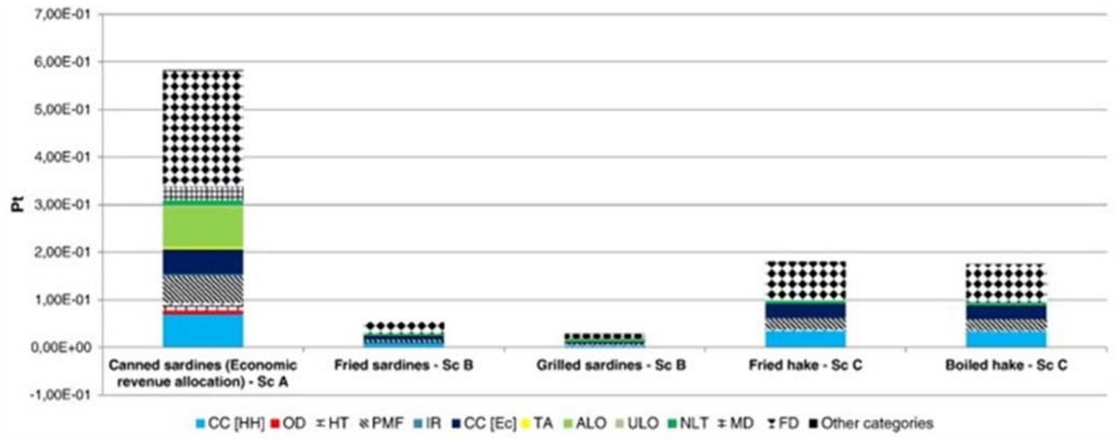

Figure 11.4. Endpoint single score environmental impact results for the selected scenarios (Results reported per FU= 17.26 g of protein for human consumption). NOTE: CC [HH]= climate change –human health; CC [Ec]= climate change – ecosystems; OD= ozone depletion; HT= human toxicity; PMF= particulate matter formation; IR= ionizing radiation; TA= terrestrial acidification; ALO= agricultural land occupation; ULO= urban land occupation; MD= metal depletion; FD= fossil depletion; Other categories= photochemical oxidant formation, freshwater eutrophication, terrestrial eco-toxicity, freshwater eco-toxicity and marine eco-toxicity

Concerning subsystem contributions, these were dominated by the canning stage, which accounted for approximately 91% of the environmental impact, while the fishery only represented 4% of global impacts and the other 6% is shared by the remaining stages throughout the supply chain. The specific contributions per subsystem for the midpoint impact categories can be seen in Figure 11.5 (see Table 11.12) for characterization results per scenario and Tables F.3 and F.4 of Appendix for environmental impacts per subsystem). The fishery stage shows the highest relative contributions in Human Health (HH) impact categories, such as ozone depletion —OD (38%) or photochemical oxidant formation —POF (18%). The canning stage in Figure 11.4 is subdivided into the processes linked to the production of olive oil that is used as filling liquid and the remaining canning operations. On the one hand, the production and transportation of olive oil constitutes the highest contributor to environmental impacts for some Ecosystems (Ec) impact categories, such as terrestrial eco-toxicity —TET (77%) and marine eco-toxicity –MET (15%), as well as water depletion —WD (85%). On the other hand, the remaining activities linked to the canning process entail the major environmental burdens in those environmental impacts (i.e. CC, PMF, FD or ALO) with the highest contributions to the final endpoint results (see Figure 11.4). More specifically, both the production of tin for the cans and the use of electricity and fuel, were the main contributors to these impacts.

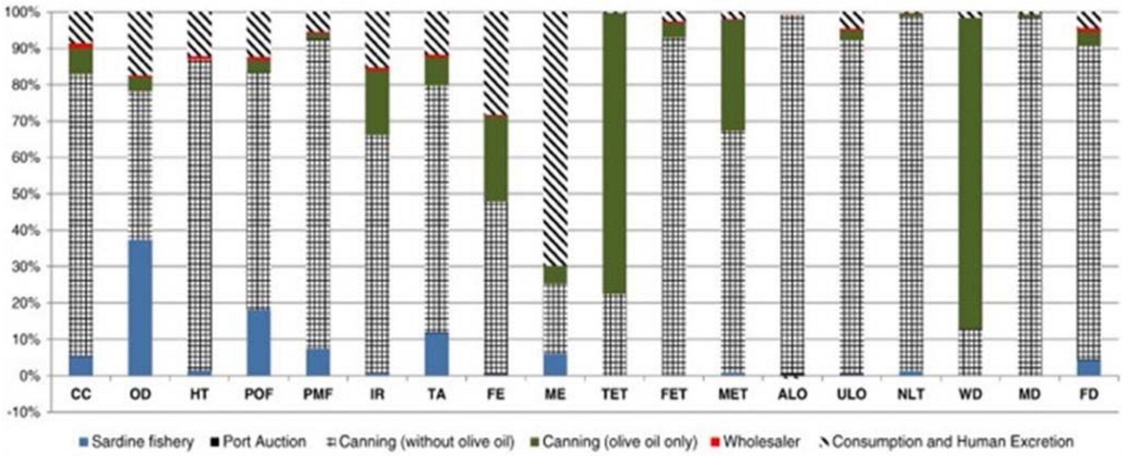

Figure 11.5. Relative subsystem contributions for the midpoint characterization environmental impact results for Scenario A. Results refer to the economic revenue allocation perspective. Results reported per FU. NOTE: CC = climate change; OD = ozone depletion; HT = human toxicity; POF = photochemical oxidant formation; PMF = particulate matter formation; IR = ionizing radiation; TA = terrestrial acidification; FE = freshwater eutrophication; ME = marine eutrophication; TET = terrestrial eco-toxicity; FE = freshwater eco-toxicity; ME = marine eco-toxicity; ALO = agricultural land occupation; ULO = urban land occupation; WD = water depletion; MD = metal depletion; FD = fossil depletion.

Table 11.12. Total environmental impacts per scenario (ReCiPe midpoint)

Impact categories	Unit	Scenario A			Scenario B		Scenario C	
		Economic revenue allocation	Economic savings allocation	Energy allocation	Fried sardines	Grilled sardines	Fried hake	Boiled hake
CC	kg CO_2 eq	3.36	3.09	2.88	$4.71\cdot10^{-1}$	$2.06\cdot10^{-1}$	1.44	1.38
OD	kg CFC-11 eq	$7.48\cdot10^{-7}$	$7.32\cdot10^{-7}$	$7.19\cdot10^{-7}$	$2.01\cdot10^{-7}$	$1.75\cdot10^{-7}$	$6.65\cdot10^{-6}$	$6.64\cdot10^{-6}$
HT	kg 1,4-DB eq	1.71	1.54	1.40	$1.73\cdot10^{-1}$	$9.89\cdot10^{-2}$	$1.80\cdot10^{-1}$	$1.39\cdot10^{-1}$
POF	kg NMVOC	$1.78\cdot10^{-2}$	$1.69\cdot10^{-2}$	$1.62\cdot10^{-2}$	$3.57\cdot10^{-3}$	$2.74\cdot10^{-3}$	$2.09\cdot10^{-2}$	$2.06\cdot10^{-2}$
PMF	kg PM_{10} eq	$1.20\cdot10^{-2}$	$1.08\cdot10^{-2}$	$9.83\cdot10^{-3}$	$1.22\cdot10^{-3}$	$8.61\cdot10^{-4}$	$5.87\cdot10^{-3}$	$5.72\cdot10^{-3}$
IR	kg U235 eq	$9.18\cdot10^{-1}$	$8.56\cdot10^{-1}$	$8.06\cdot10^{-1}$	$1.57\cdot10^{-1}$	$1.62\cdot10^{-2}$	$1.07\cdot10^{-1}$	$7.33\cdot10^{-2}$
TA	kg SO_2 eq	$1.98\cdot10^{-2}$	$1.85\cdot10^{-2}$	$1.75\cdot10^{-2}$	$3.36\cdot10^{-3}$	$1.86\cdot10^{-3}$	$1.53\cdot10^{-2}$	$1.48\cdot10^{-2}$
FE	kg P eq	$2.36\cdot10^{-3}$	$2.23\cdot10^{-3}$	$2.13\cdot10^{-3}$	$4.00\cdot10^{-4}$	$2.31\cdot10^{-5}$	$2.26\cdot10^{-4}$	$6.73\cdot10^{-5}$
ME	kg N eq	$1.94\cdot10^{-3}$	$1.91\cdot10^{-2}$	$1.89\cdot10^{-2}$	$2.87\cdot10^{-3}$	$9.07\cdot10^{-4}$	$3.53\cdot10^{-3}$	$9.14\cdot10^{-4}$
TET	kg 1,4-DB eq	$7.60\cdot10^{-2}$	$7.59\cdot10^{-2}$	$7.59\cdot10^{-2}$	$-7.21\cdot10^{-4}$	$5.42\cdot10^{-4}$	$-1.81\cdot10^{-3}$	$2.06\cdot10^{-4}$
FET	kg 1,4-DB eq	$1.41\cdot10^{-1}$	$1.23\cdot10^{-1}$	$1.08\cdot10^{-1}$	$2.78\cdot10^{-2}$	$1.75\cdot10^{-3}$	$6.68\cdot10^{-3}$	$3.58\cdot10^{-3}$
MET	kg 1,4-DB eq	$2.02\cdot10^{-1}$	$1.82\cdot10^{-1}$	$1.67\cdot10^{-1}$	$6.06\cdot10^{-3}$	$2.61\cdot10^{-3}$	$4.97\cdot10^{-2}$	$4.87\cdot10^{-4}$
ALO	m^2a	3.94	3.93	3.92	$3.02\cdot10^{-2}$	$1.06\cdot10^{-1}$	$2.36\cdot10^{-2}$	$3.50\cdot10^{-2}$
ULO	m^2a	$7.57\cdot10^{-2}$	$7.19\cdot10^{-2}$	$6.88\cdot10^{-2}$	$2.78\cdot10^{-3}$	$1.95\cdot10^{-3}$	$3.72\cdot10^{-3}$	$3.15\cdot10^{-3}$
NLT	m^2	$5.92\cdot10^{-3}$	$5.89\cdot10^{-3}$	$5.85\cdot10^{-3}$	$1.10\cdot10^{-4}$	$7.83\cdot10^{-5}$	$5.13\cdot10^{-4}$	$5.08\cdot10^{-4}$
WD	m^3	$4.89\cdot10^{-1}$	$4.87\cdot10^{-1}$	$4.85\cdot10^{-1}$	$1.85\cdot10^{-1}$	$3.34\cdot10^{-3}$	$2.64\cdot10^{-2}$	$5.40\cdot10^{-3}$
MD	kg Fe eq	6.10	5.20	4.49	$3.33\cdot10^{-2}$	$3.99\cdot10^{-3}$	$2.57\cdot10^{-2}$	$1.10\cdot10^{-2}$
FD	kg oil eq	1.36	1.28	1.22	$1.27\cdot10^{-1}$	$5.54\cdot10^{-2}$	$3.90\cdot10^{-1}$	$3.78\cdot10^{-1}$

CC= climate change; OD= ozone depletion; HT= human toxicity; POF= photochemical oxidant formation; PMF= particulate matter formation; IR= ionizing radiation; TA= terrestrial acidification; FE= freshwater eutrophication; ME= marine eutrophication; TET= terrestrial eco-toxicity; FET= freshwater eco-toxicity; MET= marine eco-toxicity; ALO= agricultural land occupation; ULO= urban land occupation; NLT= natural land transformation; WD= water depletion; MD= mineral depletion; FD= fossil depletion.

11.3.2 The supply chain of fresh sardine consumption – Scenario B

Fresh consumption of sardines in Galicia was found to be the scenario with the lowest environmental impact concerning conventional impact categories (Figure 11.4): 90-95% lower than Scenario A and 70-83% lower than Scenario C. However, strong disparities in the final environmental profile of fresh sardine consumption were identified depending on the cooking methods employed. Hence, sardines prepared on a barbeque presented a final endpoint value of $2.93 \cdot 10^{-2}$ Pt: 44.7% lower than for fried sardines. In the former, the main carrier of the environmental burdens was the fishery stage (45% —see Figure 11.6), followed by the consumption stage (24%) and the wholesaling and retailing phase (15%). In the latter, the main carrier of the impacts was the consumption stage (57%), due to the high energy intensity in the cooking process and to the use of other ingredients (i.e. olive oil and flour) for the preparation of the sardines. The fishery stage, in contrast, only represented 25% of the total environmental burdens. In terms of environmental impact categories, FD, CC and PMF were the leading contributors regardless the cooking methods selected, although slight differences were observed. However, it should be noted that wood consumption for barbequed sardines implied higher relative environmental impacts in terms of ALO and NLT (approximately 10% of the total each).

The sardine fishery subsystem was the main contributor to the environmental impact in the three impact categories (CC, PMF and FD) that represent 62% of the endpoint environmental impact when the sardines are barbequed. The relative midpoint contributions for these categories ranged from 53% (CC) to 64% (FD). In addition, this stage was also the main contributor to other impact categories, such as OD (96%) or marine eutrophication (79%). Port auction contributions were only relevant in the ALO category, representing 10% of the total impact. Wholesaling and retailing impacts represented over 35% of the impacts in categories such as ionizing radiation (IR), FET or MD, but most importantly in CC (19%) and FD (17%). Finally, consumption and human excretion operations (i.e. subsystems V and IV) represented 13% of the total environmental burdens for FD and 27% for CC.

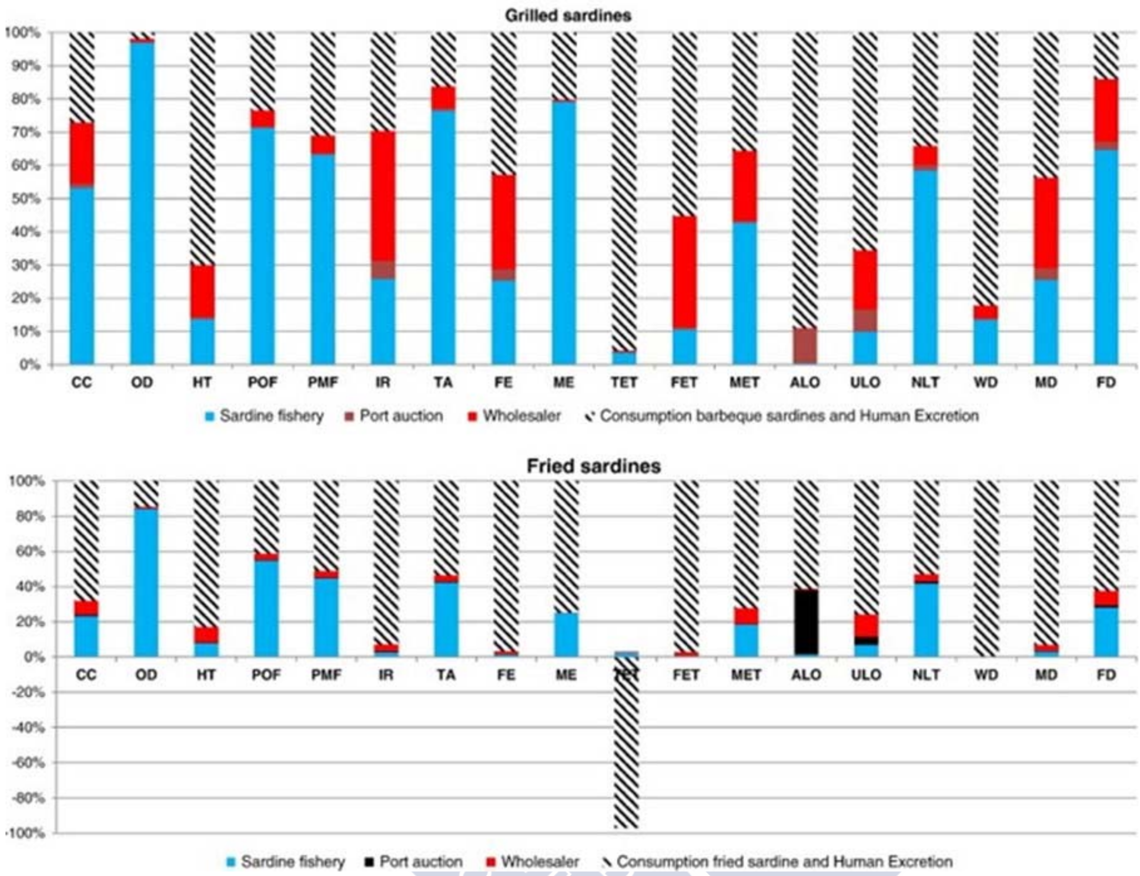

Figure 11.6. Relative subsystem contributions for the midpoint characterization environmental impact results for Scenario B. Data reported per FU. NOTE: CC = climate change; OD= ozone depletion; HT= human toxicity; POF= photochemical oxidant formation; PMF= particulate matter formation; IR= ionizing radiation; TA= terrestrial acidification; FE= freshwater eutrophication; ME= marine eutrophication; TET= terrestrial eco-toxicity; FE= freshwater eco-toxicity; ME= marine eco-toxicity; ALO= agricultural land occupation; ULO= urban land occupation; WD= water depletion; MD= metal depletion; FD= fossil depletion

When these results are compared to those for sardines that are fried in the household, a higher contribution of the consumption and human excretion stage can be observed throughout all impact categories, including those with higher weighted relevance, such as FD (62%), CC (65%) or POF (42%). This increased contribution in these subsystems is due mainly to the use of olive oil for cooking (SS5), electric consumption in the cooking stage (SS5) and the disposal of olive oil and human excretion emissions in SS6.

11.3.3 Sardines for the bait industry to catch European hake – Scenario C

The storage of frozen sardines and their use by long liners who fish European hake in the Northern stock show $1.80 \cdot 10^{-1}$ Pt (if hake is fried) and $1.75 \cdot 10^{-1}$ Pt (if hake is boiled), due mainly to the fuel use intensity (FUI) of the long liners and, to a lesser extent, to the use of fossil fuels in the sardine fishery and the terrestrial stages. The consumption stage in this scenario, despite producing higher environmental burdens when European hake is fried rather than boiled, accounted for less than 10% of the impacts. This overall environmental impact is approximately 70% lower than Scenario A (revenues allocation), but still substantially higher than the impacts identified for Scenario B. The main impact categories that contributed to the total endpoint single score results, as shown in Figure 11.4, were FD (46%), CC (36%) and PMF (13%) when hake is boiled. Only minor changes are observed whenever hake is fried due to the additional input of olive oil.

Subsystem contributions (Figure 11.7) are dominated by the capture stage of hake by the long liners, ranging from 36% for freshwater eco-toxicity (FET) to 98% (OD), except for ALO, in which relative contributions from this stage were below 5%. Nevertheless, relative contributions of this subsystem to the four most important impact categories in this scenario were in all cases above 80% regardless of the cooking option. The subsystem linked to the capture of sardines by the purse seiners shows relative impacts in all cases below 10%. The port auction contributions are negligible in all impact categories, except for ALO (82%) and urban land occupation —ULO (8%). Wholesaling and retailing activities were found to be relevant in terms of ULO and FET. However, their contribution was minimal in the key categories in this scenario. Finally, the consumption subsystem presented the highest relative contributions for a total seven impact categories when the hake is fried, primarily linked to the use of olive oil and energy consumption. While negligible for PMF, consumption also showed substantial relative impacts for FD and, to a lesser extent, CC. Nevertheless, its highest relative contribution was linked to MD and WD, as well as to toxicity impact categories.

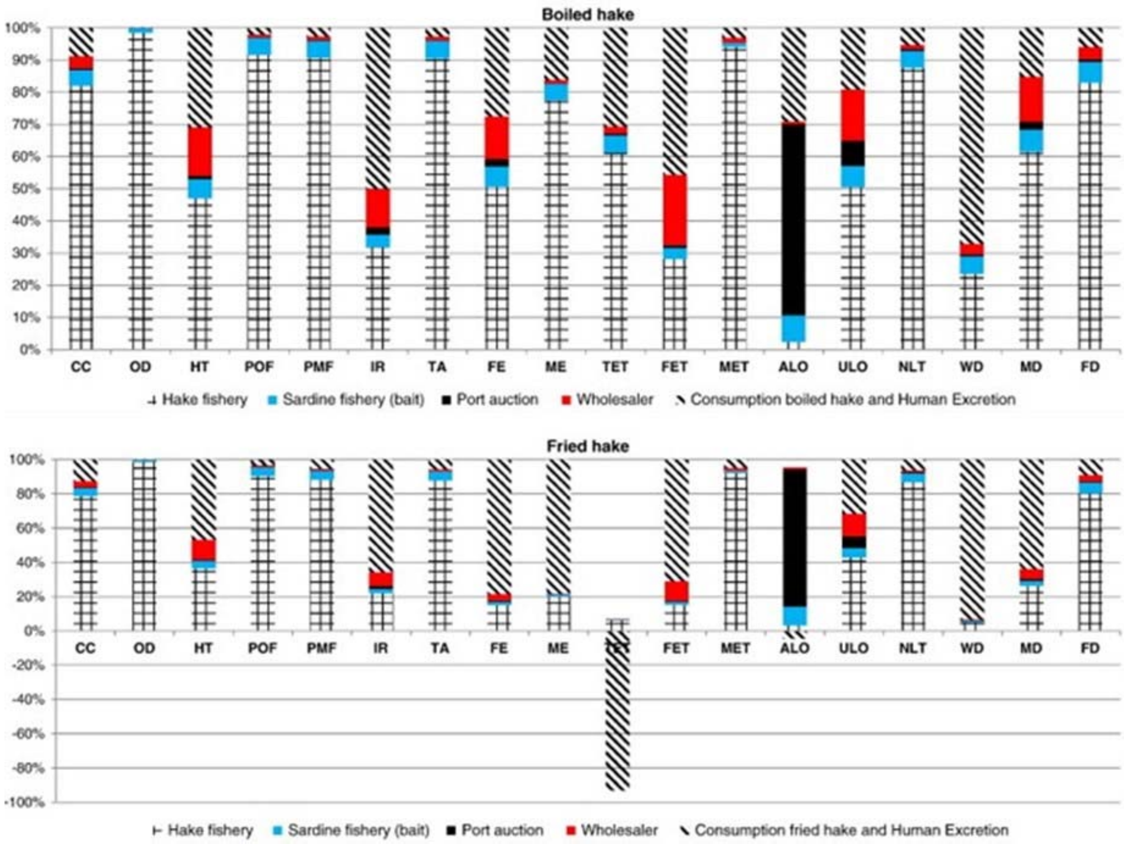

Figure 11.7. Relative subsystem contributions for the midpoint characterization environmental impact results for Scenario C. Data reported per FU. NOTE: CC = climate change; OD= ozone depletion; HT= human toxicity; POF= photochemical oxidant formation; PMF= particulate matter formation; IR= ionizing radiation; TA= terrestrial acidification; FE= freshwater eutrophication; ME= marine eutrophication; TET= terrestrial eco-toxicity; FE= freshwater eco-toxicity; ME= marine eco-toxicity; ALO= agricultural land occupation; ULO= urban land occupation; WD= water depletion; MD= metal depletion; FD= fossil depletion

11.4 Discussion

11.4.1 Identification of hot spots

The hot spots identified throughout the life cycle of the three scenarios that have been assessed are closely linked to the characteristics of each scenario. For instance, environmental impacts in Scenario A, as highlighted in section 11.3.1, are strongly influenced by the canning stage. More specifically, consumption of fuel-oil, packaging items (e.g. corrugated board) and olive oil, and especially tin for canning, were important carriers of environmental impact. Tin for canning, in fact, is the main factor responsible for CC, land occupation and FD impacts due to its high energy and land intensive extraction process. However, it should be noted that the inventory for tin was retrieved directly from the ecoinvent® database, in which they point out that the degree of uncertainty for this mineral is substantial, since tin recycling rates are not considered (Classen et al., 2007). Nevertheless, it should be noted that current recycling rates of tin in Spain were taken into account in the EOL outputs (SS5).

Notwithstanding, the environmental burdens associated with the extraction, processing and transportation of tin are high enough for an alternative raw material to be considered. Hospido et al. (2006) suggested the use of plastic bags instead of tin for tuna packaging, with environmental reductions of up to 50% in terms of GHG emissions and acidification in the entire canning stage. However, this alternative, which has already been implemented by some important tuna processing companies in Spain (Calvo, 2013), does not seem feasible in the case of sardine canning, due to the fact that sardines, unlike tuna which is canned in the form of a processed mass of white meat (García-Arias et al., 2004), are canned conserving their fishlike shape and, therefore, the aesthetics of the final processed product may be considerably hampered by the use of plastic bags. Therefore, the use of other plastic formats or the use of glass jars, the latter showing a higher reuse potential by consumers in households (Barr et al., 2001), may be an interesting future alternative to sardine canning. However, the environmental benefits of glass recycling would depend to a large degree on the number of reuse cycles before final disposal of the glass jar (Mata and Costa, 2001).

Another important hot spot in Sc. A is the production and transport of olive oil from the south of Spain. However, the environmental burdens of these activities are atomized between several operations, including the use of pesticides or fertilizers in the agricultural phase, the refining process and transport activities (Notarnicola et al., 2004; Salomone and Ioppolo, 2012).

Therefore, the operational improvements in this part of the life cycle will be closely linked to an integrated optimization of activities (Salomone and Ioppolo, 2012; Iraldo et al., 2014).

A final hot spot for the production of canned sardines is also common to the other two scenarios: diesel production and consumption to propel the fishing vessels. As addressed in the Results section, the scenario with the highest environmental impacts related to the FUI was Sc. C, due to the double fishery scheme of the production system, with most environmental impacts linked to the long lining fishery. Scenarios A and B, in contrast, show lower total impacts for fuelling due to the relatively low FUI of the purse seining vessels. Nevertheless, it should be noted that the purse seining fleet examined shows a considerably high FUI and, therefore, derived environmental impacts, as compared to other purse seining fleets which also target small pelagic species analyzed in the literature (Vázquez-Rowe et al., 2010; Ramos et al., 2011). Consequently, a reduction in environmental burdens associated with vessel propulsion should be explored through adequate optimization plans or by future changes in the energy carrier (Bengtsson et al., 2012). Finally, it should be noted that only sardines captured by Galician purse seiners were considered in the assessment. Despite the fact that this segment represents the highest proportion of sardines consumed in Galicia, other gears and fishing areas provide a minor share of the supply. Hence, future research should focus on exploring the differences in environmental impact related to a variable geographical and technical supply of the main raw material (i.e. European pilchard).

11.4.2 Comparative analysis for the three selected scenarios

Comparison between the scenarios, as well as with previous studies in the available literature, must be performed with care, given the different market niches that the different products may have, as well as certain differences in their characteristics, such as the shelf life or the food wastes they generate. Nevertheless, the endpoint single score environmental impacts provided in Figure 11.4 show a clear differentiation between the three scenarios. While the three different allocation approaches delivered reduced impact variations for Sc A, scenarios B and C show relevant environmental benefits as compared to the former.

When compared to other results available in peer-reviewed literature, it is important to note that there are three specific issues that hamper direct comparability. In the first place, as pointed out in several review papers (Vázquez-Rowe et al., 2012; Avadí and Fréon, 2013), most studies have used assessment methods other than ReCiPe and, therefore, the equivalence of impact categories or assessment units is far from complete. Secondly, the use of endpoint indicators for weighting is a novel aspect of the current manuscript within seafood LCA, since previous studies based their assessment on the use of commonly-used impact categories which in most cases did not represent a fully-available assessment method (Vázquez-Rowe et al., 2012). Finally, a third issue is the energy content orientation of the current study, rather than adopting an intermediate (e.g. landed raw fish) or final seafood (e.g. dish portion or canned amount) product based on mass. This final aspect, while quite insignificant from an endogenous LCA perspective, has important connotations from a dissemination approach, since a protein content FU is aimed directly at optimizing the use of seafood and understanding the environmental benefits or drawbacks of a series of seafood consumption patterns that will determine the final sustainability of human diets.

Nevertheless, the midpoint values obtained in this study can be partially confronted with previous studies. On the one hand, the canning scenarios show lower environmental impacts than those obtained by Iribarren et al. (2010a) for the canning of mussels. When these results are converted to the FU considered in the current study (i.e. 17.26 g of protein), the GHG emissions linked to mussel canning up to the household were 3.90 kg CO_2 eq: 16% higher when compared to the 3.36 kg CO_2 eq for sardine canning (Scenario A —economic revenues). On the other hand, the fresh seafood products analyzed in scenarios B and C were confronted to other seafood products available in literature, using Ziegler et al. (2013) for comparison. The FU for Ziegler and colleagues was one kilogram of edible product delivered to the wholesaler. Hence, this FU was converted to 17.26 g of protein and the results for scenarios B and C were calculated excluding the wholesaling, retailing and consumption stages in order to attain the maximum comparability, although the assessment methods used were still different. The results, shown in Table 11.13, suggest that fresh sardine consumption in Galicia shows similar GHGs emissions to other pelagic fish used for human consumption, while the protein content provided by hake from long liners embraces a very high environmental impact when compared to the same amount of protein delivered by other demersal species in the Norwegian fisheries.

Table 11.13. Comparative table between the GHG emissions obtained in Scenarios B and C and other products available in literature. All results rescaled to the same functional unit – FU (i.e. 17.26 g of protein) and to the same system boundaries (fishery stage and distribution up to wholesaler)

Publication	Product	Landing country	Delivery area	Protein content (g/100 g of edible fish)	GHG emissions (g CO_2 eq)
Current	Hake, fresh, gutted, head on	Galicia (Spain)	Galicia (truck)	17.8	1,256.9
Current	Sardine, fresh, round	Galicia (Spain)	Galicia (truck)	20.3	140.88
Ziegler et al. (2013)	Cod, fresh, gutted, head on	Norway	Paris (truck)	17.8	351.03
Ziegler et al. (2013)	Haddock, fresh, gutted, head on	Norway	London (truck)	18.4	348.94
Ziegler et al. (2013)	Mackerel, frozen round	Norway	Moscow (bulk boat and train)	18.4	92.86

11.4.3 The importance of the cooking stage in the environmental impacts of seafood

In previous chapter, it was highlighted the importance that the consumer stage had on the overall carbon footprint of a commonly consumed frozen seafood product in Spain. The results presented in the current study confirm this importance in other seafood presentations such as fresh or canned products, not only in terms of GHG emissions, but also regarding a comprehensive assessment of the environmental impacts. In fact, consumer behavior appears to be important on two levels. On the one hand, regarding the product selection at the retailer's, consumers may choose products that provide the same amount of protein and have the same initial raw material provision (i.e. European pilchard), but with strong environmental impact variations, as high as 90%. On the other hand, once a given product is purchased, the cooking method selected can imply an environmental reduction of up to 45% in Sc. B (low environmental profile) and of 3% in Sc. C (high environmental profile).

Finally, the results obtained in the current study also highlight the importance of post-landing processes, not only for products with a low-energy intensive fishery stage or with a limited degree of processing (i.e. fresh sardines), but also for more elaborate products such as canned seafood. For instance, the total contribution of post-landing processes in Sc. A was approximately 96% (economic revenue allocation), 75% for fried sardines from Sc. B (55% if prepared on a barbeque), and 27% for fried hake in Sc. C (25% for boiled hake). However, while the consumption and human excretion stages were critical within these values for Sc. B (37%-65% depending on the cooking method), these values were as low as 5% for Sc. A.

11.4.4 The role of seafood products in the food pyramid. Environmental perspective

Seafood, including the intake of the two species that are examined in this case study, constitutes an important source of protein supply for humans. Current worldwide figures indicate that approximately 15% of animal protein consumed by humans is linked to the intake of seafood (Smith et al., 2010; FAO, 2012). In addition, many seafood products constitute the main source of omega-3 fatty acids and other micronutrients, which trigger the importance of the presence of these products in the human diet (Hibbeln et al., 2007). Many studies have highlighted, however, the environmental risks associated with seafood consumption, ranging from health issues, such as heavy metal contents, to overfishing or impacts on marine ecosystems (FAO, 2012; Olmedo et al., 2013).

Moreover, seafood is considered to be a relatively high impact product in terms of environmental profile as compared to other food products (Barilla Center, 2010). In fact, the Barilla Center for Food and Nutrition has developed the so-called double pyramid, in which they demonstrate that the classical dietary pyramid can be inversely complemented with an environmental pyramid in which products from vegetal origin, which they recommend to be more abundant in the human diet, show a better environmental profile than animal-based products, which they recommend to be consumed more sparingly (Ciati and Ruini, 2012). However, this double pyramid provides very generic data that in the case of seafood does not account for the very significant differences in environmental impact that can be identified between products. For instance, in the case study examined the three different uses of sardines demonstrate that the way in which consumers choose to consume one initial raw material products can have important consequences on the final environmental impacts. Therefore, changes in consumption patterns of seafood can imply important reductions in environmental impacts and may allow reductions in the dependency of human communities for terrestrial animal protein sources, which have repeatedly proved to be the highest sources of environmental impact in human diets together with a limited number of seafood products (Garnett, 2011; González et al., 2011; Nijdam et al., 2012). These changes in patterns may include, but are not limited to, a higher consumption of lower trophic level species (i.e. sardines, but also anchoveta, mackerel or menhaden), relieving the stress on high trophic level marine species, livestock and, ultimately, monoculture feed crops, such as soy or corn, which have been shown to be an important threat to biodiversity (Padilla et al., 2012).

Finally, it is important to point out that since the final objective of food products (including seafood) is to nourish human communities worldwide with the added aim of guaranteeing balanced diets at low environmental cost, it seems feasible that the development of FUs and allocation perspectives that focus on the final return on investment of food products (e.g. protein content, caloric content, etc.), should be further explored in agro-food LCA studies.

11.4.5 Sources of uncertainty

A future improvement of the study would include a monthly or seasonal disaggregation of the sardine landed. Since the time of the year has an important effect on the final consumption pattern of the sardine catch, this division would reveal whether there are seasonal variations in the environmental profile of sardine extraction. Moreover, previous studies have proved the interannual variation of the environmental impacts of pelagic species to be substantial (Ramos et al., 2011). Unfortunately, detailed data were not available and, therefore, temporal analysis was left beyond the scope of this study.

An additional source of uncertainty that has been repeatedly disregarded in seafood LCA studies is the consideration of food wastes that are generated beyond the discards and offal wastes that are characteristic of the fishery stage (Kelleher, 2005; Vázquez-Rowe et al., 2011b). A recent FAO report (Gustavsson et al., 2011) points out that up to 36% of the total edible content of seafood is wasted in European nations throughout the supply chain (see Table 11.14). Therefore, Table 11.14 includes a recalculation of the endpoint environmental impact results for the scenarios modelled in this study. Only food wastes occurring in post-harvest activities were included (26.5%, which is converted to 29.25% if landed seafood is considered exclusively), therefore, excluding discards.

Inclusion of seafood waste occurring in terrestrial activities within the system boundaries would imply that the FU should be rescaled to 24.40 g of protein to allow the final consumer to obtain 17.26 g of protein intake. Results show a 42% increase in endpoint environmental impacts for all scenarios, due to the fact that the same food/waste ratio was assumed for all the different scenarios. However, this homogenous increase is not realistic. The shelf life of canned sardines is substantially higher than that of fresh products, such as fresh sardines or hake (Buzby et al., 2009). Hence, it is presumable that the food waste/ratio for the latter should be considerably higher, thus reducing the environmental gap observed in Figure 11.4.

Table 11.14. Total environmental impacts for the rescaled scenarios considering food waste (ReCiPe endpoint). FU= 21.37 g of protein

Production system	Unit	Value	% increase with respect to the scenarios without food waste computation
Scenario A (economic revenues)	Pt	$8.23 \cdot 10^{-1}$	
Scenario A (economic savings)	Pt	$7.76 \cdot 10^{-1}$	
Scenario A (energy allocation)	Pt	$7.41 \cdot 10^{-1}$	
Scenario B (Fried sardines)	Pt	$7.49 \cdot 10^{-2}$	+41.67%
Scenario B (Sardines in a BBQ)	Pt	$4.14 \cdot 10^{-2}$	
Scenario C (Boiled hake)	Pt	$2.48 \cdot 10^{-1}$	
Scenario C (Fried hake)	Pt	$2.55 \cdot 10^{-1}$	

11.5 Conclusions

Despite demographics and climate change, it seems that there is still enough food available at a worldwide level to feed human communities (Godfray et al., 2010). However, environmental impacts due to anthropogenic activities are a major threat to food security in the upcoming decades. Given the important role of the food sector in worldwide environmental burdens (over 15% in developed countries in terms of GHG emissions), understanding the specific environmental impacts of alternative food supply chains appears to be a key step in the minimization of these burdens (Garnett, 2008 and 2011).

The results of this study suggest that the way in which consumers decide to intake a certain product may strongly influence the environmental impacts derived from this process. In fact, final seafood products with minor processing stages showed environmental benefits of up to 95% as compared to more elaborate methods. In addition, cooking methods of identical purchased products showed relevant environmental disparities, as already highlighted in Chapter 10. However, it should be noted that a higher degree of data quality regarding food waste in post-landing activities would be needed in order to analyze how the variable food/waste ratios for final products with different shelf life affects final environmental profiles.

While this study focuses exclusively on a specific seafood product (i.e. European pilchard as a raw material), future research should focus on the importance of this phenomenon in other protein and non-protein food products, in order to determine upcoming food policies. For instance, the confirmation of this tendency throughout important portions of the food pyramid would advocate an increase of consumer-oriented environmental sustainability campaigns to raise awareness regarding the important role that individual decisions can play in reducing the environmental profile of diet patterns.

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Chapter 12

LCA of sodium alginate extraction from seaweed[1]

Summary

Brown seaweeds are rich in high valuable compounds such as alginate. Kelps —which grow along the West coast of the United States— present a high ratio of sodium alginate within their cell walls. This fact has become this seaweed of interest to extract alginate at industrial level. This chapter is focused on assessing the environmental profile of the extraction process —from wild harvesting to lab extraction— of sodium alginate from *Nereocystis luetkeana*. The application of Life Cycle Assessment methodology to this process allowed to detect the main hot spots and thereby the inventory items responsible for the environmental impact. Additionally, several scenarios and improvement actions were considered in order to reduce the environmental burdens related to the extraction process.

[1]Research developed in collaboration with the Department of Plant Sciences, College of Agriculture and Natural Resources of the University of Wyoming (USA).

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12.1 Introduction

Seaweeds have been harvested along coast line for different purposes, dating the first evidences some thousands of years ago (Naylor and FAO, 1976). Potential uses of harvested seaweeds depend on the type, being used either as food for human consumption, feed for aquaculture sector, or fertilizers due to their nutrients content.

During the last decades, industry and scientific community are focusing their research on valorizing seaweeds and algae to find new compounds and products, which may be used for medical and pharmaceutical purposes (Balboa et al., 2013). Therefore, given the link between seafood and seaweeds in terms of their marine origin, harvest methods (using boats in some cases) and processing, the production of high value compounds from seaweed was included in this dissertation in order to cover a wider scope of marine origin products.

Seaweeds are classified in three groups attending to their pigments (i.e. color): brown, red, and green. Kelps are large brown marine algae that belong to the order Laminariales, where up to 30 different genera such as Macrocystis, Nereocystis, or Laminaria are included. Additionally, kelps are featured for forming large forests in clean and clear waters, creating ecosystems for other organisms (e.g. other seaweed, fish, etc.) and for their high growth rate (Springer et al., 2007).

These seaweeds are commonly used —as other macro-algae— to obtain different products, being traditionally harvested to supply raw materials for food and colloid industries all over the world since mid-20th (McHugh, 2003). Thus, the first industry related to Kelp harvesting was established in Norway, where new harvesting techniques and vessels (e.g. seaweed trawler) were developed to harvest large amounts of algae per day (Vea and Ask, 2010). Additionally, along harvesting industry, a new business emerged—an ancillary industry focused on Kelp processing located in the ports where seaweeds were landed.

Nonetheless, due to the quick development of this industry, environmental problems raised really soon because of uncontrolled harvesting (Vasquez, 1995; Rotheman et al., 2006; Vea and

Ask, 2010). In this sense, Norway legislated on this subject to avoid the overexploitation of this resource and reduce impact on marine ecosystems. To do so, following the recommendations made by the scientific community, a management scheme was elaborated. This scheme implied to harvest an area every 4 or 5 years to assure re-growth and natural regeneration of stocks (Svendsen, 1972).

The main product obtained from Kelps is alginate, a compound widely used by pharmaceutical and food industries because of its features: viscosity, gel and film formation capacity (FMC Corporation, 2015). For instance, the use of alginate is greatly extended among dentists and dermatologists, or even in the haute cuisine (Pelayo, 2008; Ser, 2014). Also, it has been used as supporting material to tissue growth at lab aiming at transplanting of damaged tissues (Wang et al., 2003) or as a potential substitute of plastics (Paslier, 2014).

Given the great potential uses of alginate, there is an increasing interest to establish new industries in coast zones where Kelps grow. Thus, *Nereocystis luetkeana* (hereafter Nereocystis), which grow along the West coast of USA, is a potential raw material for alginate extraction, since it has been traditionally harvested for other purposes such as mariculture feed, human consumption or agricultural fertilization. Nereocystis has been harvested in California, where a large industry related can be found, and Oregon through lease rights since the 80s (Springer et al., 2007). However, the State of Washington does not allow Nereocystis harvesting for commercial purposes, being only authorized the recreational harvest through game and hunting licenses (Springer et al., 2007).

During the last 3-5 years, some projects are planning to establish in the State of Washington, under the umbrella of scientific community, an industry around Nereocystis (i.e. harvesting and processing), aiming at boosting economically depressed coast towns. To do this, in a first phase, a pilot plant will be created to optimize the extraction process, focusing on alginate (sodium alginate) extraction, as well as other high value compounds from seaweed. This pilot plant would be built in Laramie (Wyoming) as a spin-off of the University of Wyoming (Laramie). Once the process is completely optimized, the project will be moved to the coast in a second phase to build an industrial processing plant by the seashore. This pilot plant would not only employ people from little coastal towns, but also offer a reconversion alternative for fishermen communities. In fact, seaweed harvesting would be carried out by fishermen from local towns exclusively, since social issues is one of the pillars of the project, whose goal is to get sustainability through its three pillars: social, economic, and environmental (World Commission on Environment and Development, 1987).

The study focuses on the environmental profile of sodium alginate extraction, using Nereocystis harvested in the State of Washington waters and delivered to Laramie for processing, throughout life cycle assessment (LCA). The application of LCA methodology to macro-algae and micro-algae or extraction processes is really new. In fact, there is no too many bibliographic references on the subject, especially for complex processes such as those aiming at obtaining bioactive compounds and drugs (Clarens et al., 2010; Pérez-López et al., 2014a). Nonetheless, LCA of algae processes related to bio-fuel production are more common (Curtis and Kreider, 2010; Singh, 2010; Stephenson, 2010) to benchmark the results with conventional energy sources.

12.2 Materials and Methods

12.2.1 Scope definition and functional unit

This study aims at performing LCA of sodium alginate extraction from Nereocystis to be used for the medical and pharmaceutical industry. The extraction method assessed was the alkaline extraction (González-López et al., 2012) (Figure 12.1), which is the most commonly used by alginate industry. As abovementioned, Nereocystis was harvested in the State of Washington waters and then delivered by road to Laramie. The extraction of alginate took place in the facilities of the Plant Science Department, College of Agriculture and Natural Resources of the University of Wyoming.

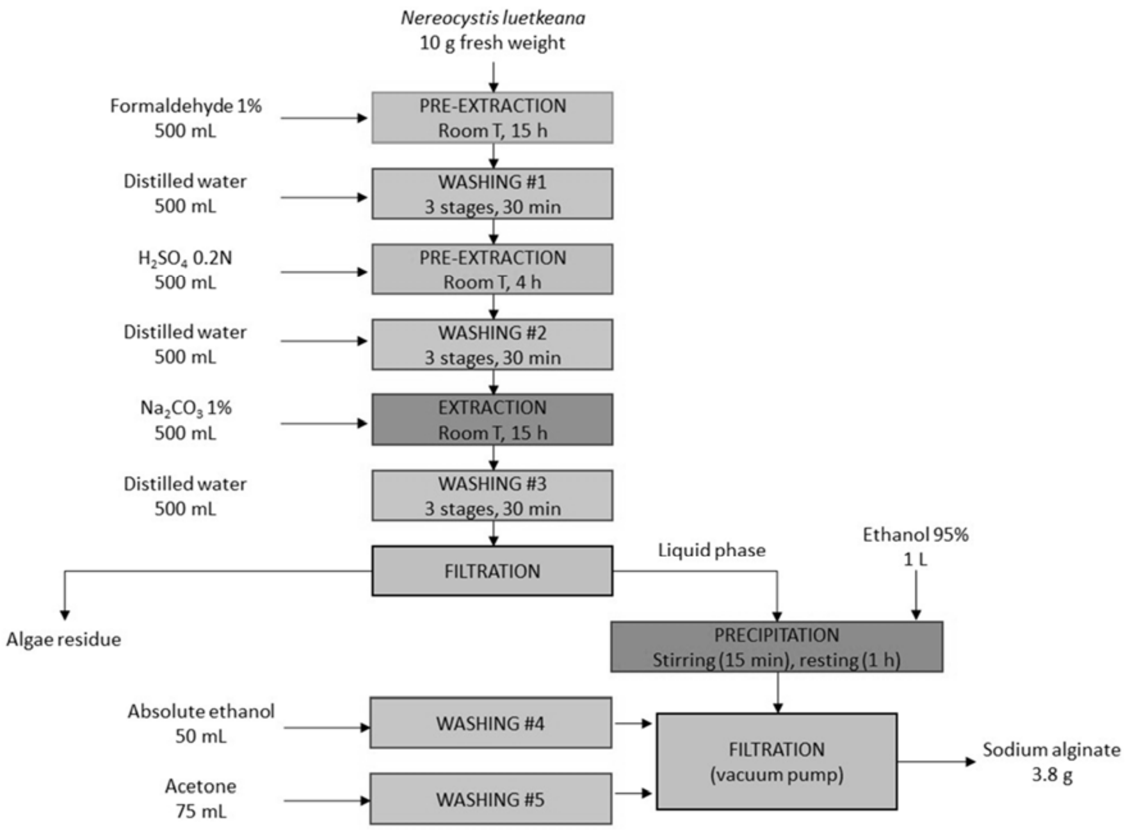

Figure 12.1. Flow diagram of alkaline extraction in the sodium alginate process. Adapted from González-López et al. (2012)

The function of the system is the extraction of sodium alginate from seaweed, so the functional unit (FU) selected was 1 kg of extracted sodium alginate. The selection of this FU was based on the necessity of establishing a commercial FU (the project aims at obtaining high quality alginate with commercial purposes), rather than other FUs commonly used in extraction processes based on the amount of alginate per extraction, product obtained by batch, etc. (Pérez-López et al., 2014a).

12.2.2 System description

The system under study was divided into two subsystems: harvesting of seaweed from the natural environment (SS1) and alginate extraction in lab (SS2). The subsystems and processes included within the system boundaries are shown in Figure 12.2.

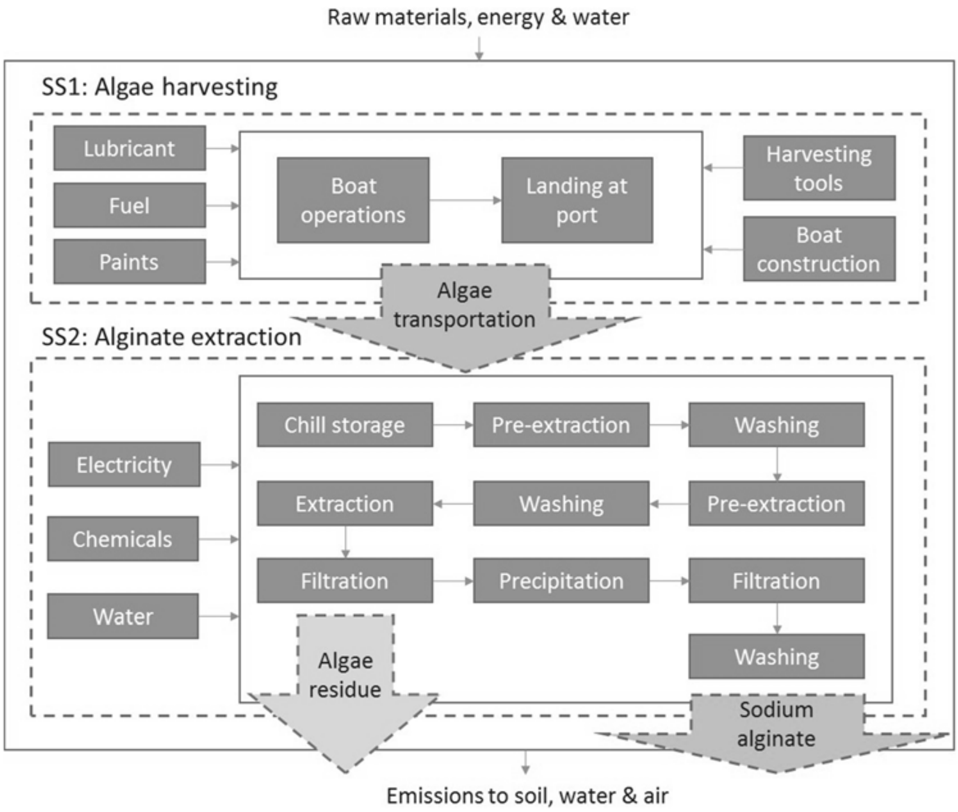

Figure 12.2. System boundaries of the study

The SS1 includes the boat operations —small boats— during harvesting, seaweed landing at port, and transportation by road in a refrigerated truck (in order to maintain the best conditions) to Laramie (circa 2,200 km). Once seaweed is received, it is stored in a refrigerator at 10-15°C, where it remains until being processed. Additionally, seaweeds are sprayed with marine water once per day to maintain humidity.

Figure 12.1 shows the stages carried out at lab during alginate extraction (SS2). The extraction process aims at obtaining powder of sodium alginate. With this purpose, the alginate naturally present in algae cell walls as calcium, magnesium, or sodium salts is solubilized —only sodium salts get soluble— to be subsequently precipitated by adding ethanol. Figure 12.3 offers pictures of the extraction process. A detailed description of these stages is described below:

- Cutting: Nereocystis is washed, using tap water to remove solids and chopped, obtaining pieces of roughly 0.5 cm.
- Soaking with formaldehyde: chopped algae is soaked with formaldehyde 1% for 15 h to remove pigments.
- Washing #1: after 15 h, formaldehyde is removed and the remaining solids of algae are washed with distilled water three times during 30 min.
- Pre-extraction: a solution of sulfuric acid 0.2 N is added, remaining soaked during 4 h. This acid pre-extraction aims at making alginate more soluble in the alkaline solution.
- Washing #2: after 4 h, the liquid solution is removed, and solids are washed with distilled water —three times during 30 min.
- Alkaline extraction: after washing, a solution of sodium carbonate 1% is added to algae solids, remaining at room temperature during 15 h.
- Filtration #1: after 15 h, the algae solids in the solution of sodium carbonate are filtrated, obtaining a solid phase —algae residue— and a liquid phase, which contains solubilized sodium alginate.
- Precipitation: a solution of ethanol 95% is added to the liquid phase to precipitate sodium alginate. This solution is stirred for 15 min and then it rests for one hour.
- Filtration #2: precipitated sodium alginate is filtrated with the aid of a vacuum pump.

- Washing #3: precipitated sodium alginate is washed with absolute ethanol.
- Washing #4: precipitated sodium alginate is washed with acetone.
- Drying: precipitated sodium alginate is removed from filter, dried at room temperature and then stored in a desiccator. However, it should be noted the fact that along filtration alginate is washed with absolute ethanol and acetone so that water is almost completely removed from alginate.

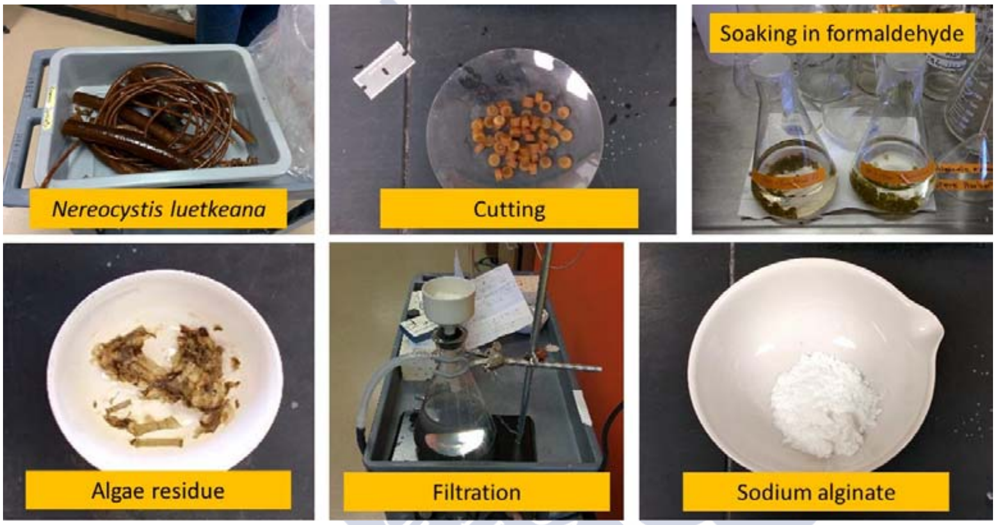

Figure 12.3. Pictures taken during the extraction process

The extraction process was carried out using 10 g (fresh weigh) of Nereocystis per extraction and obtaining an average of 3.8 g of sodium alginate per extraction. However, it should be noted the fact that other extraction methods and configurations (Hernández-Carmona et al., 1999) will be tested in order to get the best extraction method in terms of alginate yield and economical cost.

12.2.3 Data acquisition

Life cycle inventory (LCI) requires the collection of reliable data to assure the quality of the assessment. Data for the subsystems assessed were collected from different sources. Data regarding seaweed harvesting (SS1) were obtained following the assumptions made by Pérez-López et al. (2014b), which deals with different options to valorize an invasive macro-algae in Galicia (NW Spain) (*Sargassum muticum*). Thus, given the similar features of both seaweed (*Sargassum muticum* and *Nereocystis luetkeana*) in terms of harvest methods, inventory implementation follows its procedures for boat maintenance, fuel consumption, boat paints, etc. In this regard, materials for hull and engine were increased by 25 and 50% respectively, taking into account a life span of 30 and 15 years. Regarding emissions, fuel consumption emission factors were obtained from EMEP-Corinair (2009), whereas emissions to marine waters derived from anti-fouling and boat paints were estimated following Hospido and Tyedmers (2005) assumptions, considering that 67% is emitted to water.

Furthermore, data for sodium alginate extraction (SS2), chemicals and water consumption, were obtained from experimental data during extraction process. Additionally, a solvent recovery of 90% was assumed for ethanol (Pérez-López et al., 2014b). The recovery of other chemicals such as sulfuric acid or acetone was not considered due to the lower amount, in comparison with ethanol, required for extraction. In this sense, whilst *circa* 280 L of ethanol are consumed to extract 1 kg of sodium alginate, only 20 L of acetone or 13.2 L of sulfuric acid are needed. Electricity consumptions were estimated taking into account equipment specifications (power) and the duration of each stage.

Background data for chemicals production, materials and electricity were obtained from the ecoinvent® database (Frischknecht et al., 2007). In addition, Wyoming electricity production mix was modeled for year 2014 (EIA, 2015). This assumption was based on the negative net interstate flow —electricity production is higher than consumption— of Wyoming (EIA, 2015) so that electricity consumed during extraction process would come from Wyoming's grid exclusively.

12.2.4 Allocation procedure

Following ISO guidelines, allocation pursues the segmentation of the input and output flows of a process system between the studied product and those that are left outside the system boundaries (ISO, 2006b). In this study, allocation was not necessary since only one product is obtained after extraction —sodium alginate. However, as other studies on the field (Pérez-López et al., 2014b), seaweed residue could be a potential fertilizer. Therefore, this sub-product could be valorized as fertilizer, being included in the process as an avoided product —using its equivalent as fertilizer (Pérez-López et al., 2014b).

12.2.5 Life cycle inventory

LCI entails the collection of specific data that allows quantifying those inputs and outputs of the system under study (ISO, 2006b). Inventory data for the inputs and outputs of harvesting and extraction process are shown in Tables 12.1 12.2. It should be noted that inventory was performed for average data obtained throughout extractions.

Table 12.1. Life cycle inventory for seaweed harvesting (SS1). Data per functional unit (FU: 1 kg of extracted sodium alginate)

Inputs from technosphere			**Outputs to the technosphere**		
Materials			*Products*		
Glass fiber	34.38	g	Seaweed	2.63	kg
Steel	10.32	g			
Anti-fouling	21.89	g			
Lubricant	27.31	g			
Paint	5.51	g			
Polyethylene	16.95	g			
Nylon	9.33	g			
Transportation					
Refrigerated truck	5,621	kg·km			
Energy					
Diesel	12.23	kg			
Outputs to the environment					
Emissions to air			*Emissions to water*		
CO_2	38.74	kg	Xylene	0.13	g
SO_2	24.46	g	Cobalt	0.00	g
NMVOC	361.72	g	Copper	4.54	g
Methane	2.20	g	Zinc	2.05	g
NO_x	287.08	g	Xylene	1.82	g
CO	90.50	g			

Table 12.2. Life cycle inventory for sodium alginate extraction (SS2). Data per functional unit (FU: 1 kg of extracted sodium alginate)

Inputs from technosphere		
Materials		
Seaweed	2.63	kg
Formaldehyde	1.07	kg
Sulfuric acid	24.21	kg
Sodium carbonate	1.32	kg
Ethanol	21.44	kg
Acetone	15.61	kg
Deionized water	643.43	kg
Ice	65.79	kg
Energy		
Electricity	34.10	kWh
Outputs to the technosphere		
Products		
Sodium alginate	1.00	kg
Avoided products		
Fertilizer (Nitrogen fertilizer)	27.61	g

12.2.6 Impact assessment

The environmental profile of the extraction of alginate from seaweed was carried out through the CML baseline v2.05 method (Guinee et al., 2001). The environmental profile was obtained using the impact categories used for previous studies of algae LCA (Pérez-López et al., 2014a, b, c; Lardon et al., 2009; Collet et al., 2011): abiotic depletion potential (ADP), acidification potential (AP), eutrophication potential (EP), global warming potential (GWP), ozone layer depletion potential (OLD), human toxicity (HTP), fresh water aquatic eco-toxicity (FTP), marine aquatic eco-toxicity (MEP), terrestrial eco-toxicity (TEP) and photochemical oxidants formation potential (POFP). The software Simapro v8.1 (PRè-Product Ecology Consultants, 2014) was used to implement the inventories and impact assessment.

12.3 Results

Table 12.3 shows the environmental profile per FU (1 kg of extracted sodium alginate) for the impact categories abovementioned. Sodium alginate extraction process (SS2) presents a higher impact than harvesting (SS1) in terms of relative contribution, ranging from 51% (ODP) to 93% (TEP and POFP) and accounting for 83% on average.

Table 12.3. Environmental profile for the whole process and each subsystem. Data per functional unit (FU: 1 kg of extracted sodium alginate)

Impact categories	**Units**	**SS1**	**SS2**	**Total**
Abiotic depletion (ADP)	kg Sbeq	0.12	1.28	1.40
Acidification (AP)	kg SO_2eq	0.26	0.89	1.15
Eutrophication (EP)	kg PO_4^{3-}eq	0.05	0.14	0.19
Global warming (GWP)	kg CO_2eq	18.64	106.24	124.88
Ozone layer depletion (ODP)	kg CFC-11 eq	$2.17 \cdot 10^{-6}$	$2.23 \cdot 10^{-6}$	$4.40 \cdot 10^{-6}$
Human toxicity (HTP)	kg 1,4-DB eq	3.77	31.33	35.10
Fresh water aquatic ecotoxicity (FEP)	kg 1,4-DB eq	3.00	17.84	20.84
Marine aquatic ecotoxicity (MEP)	kg 1,4-DB eq	$3.71 \cdot 10^{3}$	$3.89 \cdot 10^{4}$	$4.27 \cdot 10^{4}$
Terrestrial ecotoxicity (TEP)	kg 1,4-DB eq	0.03	0.32	0.35
Photochemical oxidation (POFP)	kg C_2H_4eq	0.01	0.06	0.07

SS1: seaweed harvesting; SS2: extraction process

Figure 12.4 displays the relative contribution of the SS1 subsystem. The main responsible for impact for all impact categories, except for FEP, was diesel production and consumption, ranging from 99% for AP to 46% for MEP. It should be remarked the contribution of the process named as “others” —which includes emissions to water derived from boat paints and the manufacture of the tools used to harvest the algae, as well as its final disposal as residue— to the impact categories of FEP and MEP, accounting for 72% and 14%, respectively. Additionally, transportation presented significant impact contribution for toxicity categories, ranging from

11% for HTP to 31% for MEP. Anti-fouling paint production involved low relative contribution (almost negligible); however, it accounted for 9% and 8% for HTP and MEP, respectively. Remaining inventory elements, boat construction, and boat paint and lubricant production, were negligible in terms of impact contribution.

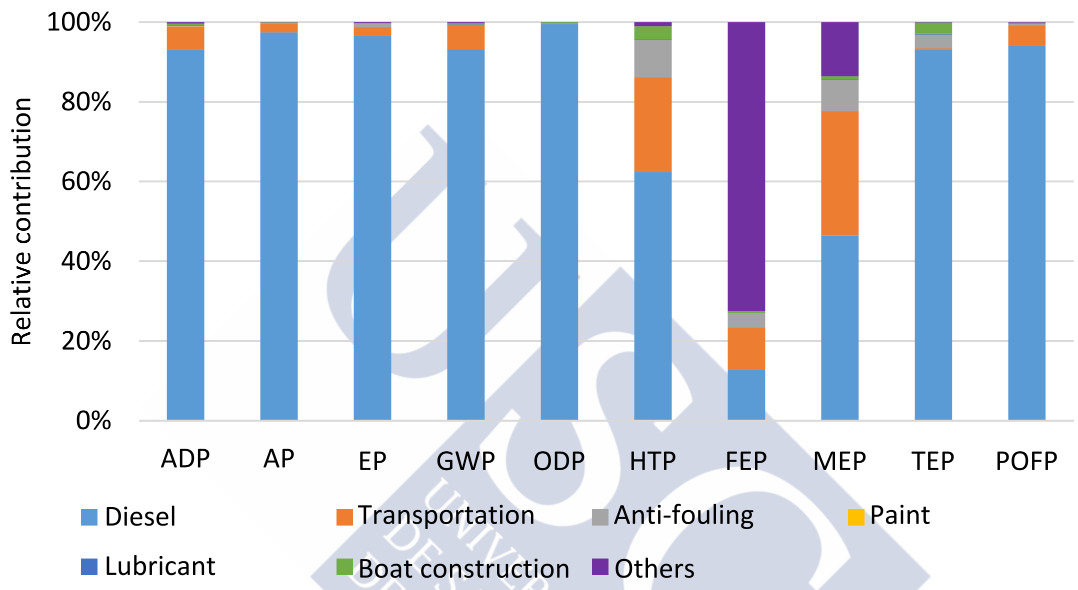

Figure 12.4. Relative contribution to the environmental profile of the SS1 (seaweed harvesting)

Furthermore, Figure 12.5 shows the environmental profile for SS2 subsystem. The environmental burden was dominated in most impact categories by electricity consumption, mainly for EP, GWP and toxicity categories. Thus, the impact contribution of electricity ranged from 11% to 70% for ODP and MEP, respectively. The following process in terms of impact contribution was ethanol production, which presented a remarkable impact for POFP (49%), ADP (36%), ODP (35%), and TEP (25%).

Regarding other solvents used the during extraction process (i.e. acetone, sulfuric acid, formaldehyde and sodium carbonate), sulfuric acid presented higher impact contributions — especially for AP (43%), TEP (25%), and POFP (24%)— than the other solvents, except for the impact categories ADP and GWP where acetone accounted for 37% and 32% respectively. For the remaining inventory items (formaldehyde, sodium carbonate, distilled water and ice), the impact contribution was relatively low (less than 10%) in comparison to the other processes—except for ODP, where the contribution of distilled water and ice production was 17% and 10% respectively.

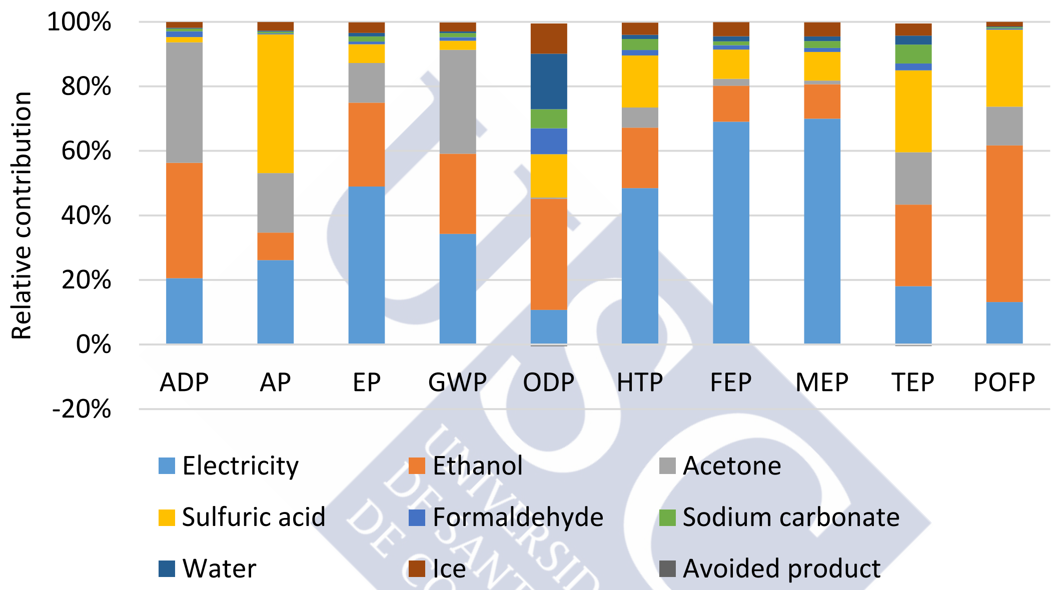

Figure 12.5. Relative contribution to the environmental profile of the SS2 (extraction process)

Regarding environmental gains due to seaweed residue valorization as fertilizer, as can be seen in Figure 12.5, the potential impact reduction was negligible for all impact categories, accounting by less than 1%.

12.4 Discussion

12.4.1 Identification of the main hot spots

Despite the low impact contribution of SS1 to the whole process, the impact derived from diesel production and related emissions should be noted. As fishing LCA studies, fuel use was the input responsible for most environmental burdens (Avadí and Freón, 2013), being necessary a breakdown of fuel in order to understand the vessel activities that could be of interest to reduce its consumption (Vázquez-Rowe et al., 2012). Additionally, the results obtained for SS1 were similar to other studies on this field (Pérez-López et al., 2014b), ranging from 10 to 15%, except for ODP (49%).

Electricity consumption was the main responsible for most impact categories in SS2. This should not be strange due to the fact that lab equipment is generally oversized, increasing their electricity consumption. Thus, focusing on electricity requirements per stage (Figure 12.6), the vacuum pump used for filtering the precipitated alginate was the device with the highest electricity consumption: 58% in share, followed by the two stages of stirring with 25% and 17%, stirring before precipitation and sodium carbonate preparation, respectively. Chilling storage was almost negligible in terms of electricity consumption —seaweed was stored, on average, during one week until been processed.

Delving into the impact categories to which electricity presented a significant relative contribution (AP, EP, GWP, and toxicity categories), the electricity generation mix of Wyoming —which highly depends on coal— played a relevant role on the aforementioned categories due to the background process of coal burning. However, the fact that Wyoming is the largest coal producer within the United States —accounting by 40% in 2013 (EIA, 2015)— may explain its dependency on coal burning to electricity production, representing *circa* 90% of the production currently (EIA, 2015).

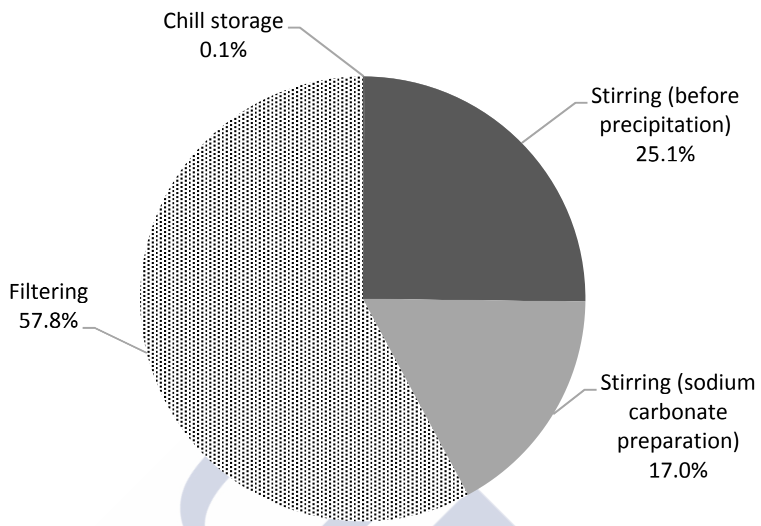

Figure 12.6. Electricity consumption breakdown for the extraction process (SS2)

Furthermore, the chemicals used during extraction, specially ethanol and acetone, showed significant impact for the categories AP, GWP, and POFP. The main reasons of this impact was due to the large amount consumed —mainly ethanol, despite the fact that 90% is recovered after extraction. Also, it should be highlighted the impact contribution of the sulfuric acid for the impact categories of AP, TEP, and POFP. The underlying reasons for impacts derived from chemical manufacture processes were due to the high demand of energy and other inputs during their production.

12.4.2 Improvement actions identified

LCA methodology allows identifying the main impact sources of a given process. A set of improvements actions —focused on those processes previously identified as hot spots— were evaluated in order to reduce the environmental burdens related to sodium alginate extraction. Firstly, two scenarios regarding electricity production mix were evaluated in order to reduce the impact related to electricity consumption. Thus, whereas scenario 1 (Sc 1) uses the United States electricity mix instead of Wyoming ones, scenario 2 (Sc 2) increases the renewable energy generated —through wind generators— in Wyoming's mix (up to 20% of the total production rather than the current 10%), and, at the same time, takes to lower values the electricity obtained from coal burning (79% instead of 89%).

Secondly, two scenarios involving yield for both subsystems were conducted. Thus, scenario 3 (Sc 3) deals with the harvesting process (SS1) and, rather than using recreational boats for harvesting, larger vessels —which are specifically engineered for seaweed harvesting (Vea and Ask, 2011) — were modelled in order to reduce the amount of inputs needed per kg of harvested seaweed —e.g. 25 L of fuel/ ton of seaweed, similar values to those obtained for some inshore fishing fleets (Vázquez-Rowe, 2012). Furthermore, scenario 4 (Sc 4) increases the amount of sodium alginate obtained during extraction process (SS2) —from 3.8 g to 4.18 g. The rationale behind this scenario was the different amounts of sodium alginate obtained after extraction, ranging from 2.5 to 5.8 g.

Thirdly, a combined scenario (Sc 5) for scenarios 3 and 4 was evaluated due to the fact that both are feasible to implement. Scenarios that imply changes on electricity mix were not combined since both are more complex to carry out —especially for scenario 2— because of the difficulties that would imply to modify the energy policy of a State as Wyoming, which highly depends on coal for electricity generation, or to change the location of the projected pilot plant (Wyoming). In this regard, using the United States electricity mix would entail to move the projected pilot plant to another state, which, at the same time, should have a positive net interstate flow (i.e. lower electricity production than consumption).

Finally, a scenario without ethanol recovering (Sc 6) was conducted, aiming at highlighting the importance of solvents recovering in this kind of processes (Pérez-López et al., 2014b; Raymond et al., 2010). Table 12.4 depicts the environmental performance of the evaluated scenarios, using the current study as base scenario. Additionally, Figure 12.7 shows the relative environmental profile of each scenario. As abovementioned, environmental burden related to electricity consumption was the main impact contributor for the impact categories of AP, EP, GWP and toxicity ones. In this sense, scenario 1, which uses the USA electricity mix, would reduce the impact from 2% for POFP to 18% for FEP and MEP, obtaining an average impact reduction of 10%. It should be remarked that the environmental impact would be increased by 10% for ODP when this scenario is implemented. Furthermore, scenario 2, which implies changes in Wyoming electricity mix increasing electricity obtained through wind generators up to 20% at the expenses of coal burning, the impact reduction would range from roughly 1% for ODP to 7% to MEP, with an average reduction of 3.5%.

Table 12.4. Impact results for the evaluated scenarios. Data per functional unit (FU: 1 kg of extracted sodium alginate)

Impact categories	Units	Base scenario	Sc 1	Sc 2	Sc 3	Sc 4	Sc 5	Sc 6
ADP	kg Sbeq	1.40	1.33	1.37	1.28	1.27	1.16	5.53
AP	kg SO_2eq	1.15	1.11	1.13	0.90	1.06	0.82	1.84
EP	kg PO_4^{3-}eq	0.19	0.17	0.18	0.14	0.18	0.13	0.52
GWP	kg CO_2eq	124.88	113.92	120.89	106.39	114.26	97.12	362.85
ODP	kg CFC-11 eq	$4.40 \cdot 10^{-6}$	$4.83 \cdot 10^{-6}$	$4.38 \cdot 10^{-6}$	$2.27 \cdot 10^{-6}$	$4.18 \cdot 10^{-6}$	$2.05 \cdot 10^{-6}$	$1.13 \cdot 10^{-5}$
HTP	kg 1,4-DB eq	35.10	31.73	33.65	31.33	31.97	29.15	88.01
FEP	kg 1,4-DB eq	20.84	17.09	19.50	17.88	19.05	16.43	38.79
MEP	kg 1,4-DB eq	$4.27 \cdot 10^{4}$	$3.48 \cdot 10^{4}$	$3.97 \cdot 10^{4}$	$3.90 \cdot 10^{4}$	$3.88 \cdot 10^{4}$	$3.63 \cdot 10^{4}$	$8.00 \cdot 10^{4}$
TEP	kg 1,4-DB eq	0.35	0.34	0.35	0.32	0.31	0.29	1.07
POFP	kg C_2H_4eq	0.07	0.07	0.07	0.06	0.06	0.06	0.35

Base scenario: current study; Sc 1: USA electricity mix; Sc 2: 20% electricity production in Wyoming comes from wind generators; Sc 3: increase harvesting yield; Sc 4: increase extraction yield; Sc 5: combination of Sc 3 and Sc 4; Sc 6: no ethanol recovering

The effect of harvesting and extraction yield is conducted in scenarios 3 and 4, respectively. Firstly, scenario 3 —which increases harvesting yield for SS1— would present significant reductions for all impact categories, ranging from 8% for ADP, TEP, and POFP to 45% for ODP, and obtaining an average decrease of 17%. The main reason behind this reduction was due to the fuel consumption reduction during harvesting since, as aforementioned, its consumption was lowered to inshore fishing vessels performing similarly (Vázquez-Rowe et al., 2010). Secondly, scenario 4 involves increasing the yield of alginate extraction by 10% and thereby acts over the SS2 subsystem. The environmental impact reduction obtained for this scenario would range from 7% to EP to 9% for ADP, HTP, FEP, MEP, TEP, and POFP, with an average reduction of 8%. Finally, the combined scenario for scenarios 3 and 4 (Sc 5) was found to be the best performing scenario (Figure 12.7), obtaining an impact decrease which ranges from 15% for MEP to 50% ODP, with an average reduction of 24%.

When the system is analyzed without ethanol recovering (scenario 6), the environmental burden was dramatically increased for all impact categories, ranging from 60% for AP to 405% for POFP, obtaining an average increasing of 182%. Accordingly, solvents as ethanol should be recovered to the largest extent for this kind of processes since impact reduction and, therefore final environmental gains are remarkable for all impact categories (Pérez-López et al., 204b).

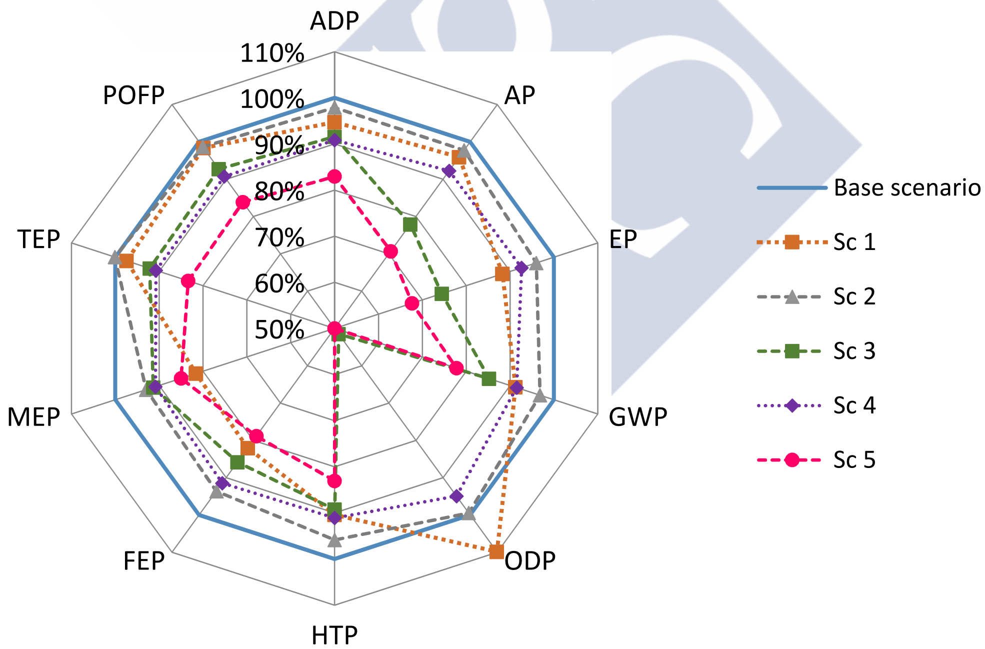

Figure 12.7. Relative environmental profile of the scenarios evaluated. Note: scenario 6 was not included in the chart in order to maintain a better display of environmental gains

12.5 Conclusions

When processes that imply a valorization of a natural resource to obtain high quality or valuable compounds, especially for those that are under development, a thorough analysis of the aspects involved is needed (i.e. social, economic, and environmental). The use of LCA to assess the environmental profile of sodium alginate extraction has proven to be useful in order to identify the processes responsible for impact. The results obtained together with the improvement actions evaluated should be considered for the sake of the optimal valorization of natural resources.

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SECTION IV

CONCLUSIONS

Chapter 13

General conclusions

The main aim of this doctoral thesis was to assess two key primary sectors within the Galician industrial structure through the application of LCA and other complementary tools. The wine-growing and winemaking of the two most important —in terms of production and revenues— PDOs in Galicia: "Rías Baixas" and "Ribeiro"; and the fishing and seafood processing industry were evaluated. Additionally, seaweed processing was included as part of fishing and seafood processing due to their similarities based on wild harvest/extraction from nature and subsequently processing stages. General conclusions obtained in sections II and III are detailed below.

Section II

The chapters regarding winegrowing offer detailed inventory for the assessed PDOs during several years (e.g. 2007, 2008, 2010 and 2011). Additionally, apart from conventional wine-growing techniques, other techniques —"more environmental friendly"— such as biodynamic were assessed. From the results obtained for the "Ribeiro" appellation it can be highlighted:

- When both wine-growing and winemaking stages were assessed, the environmental burdens were higher for viticulture stages. In this regard, inventory items such as fertilizers, diesel production and on field emissions were the main responsible for impact. In addition, the use of pesticides with lower toxicity should be considered to reduce environmental impacts. For vinification processes, bottling and packaging processes presented the highest contribution, where glass manufacture was the main responsible for impact.

- When results were reported in a temporal basis, it allowed identifying the environmental performance between years. In fact, it was proposed for being used to report environmental performance of wine based on individual impacts for each harvest year.

- Moreover, a set of improvement actions were conducted in order to improve the environmental performance: i) change the bottle design using plastic bottles rather than glass ones; ii) use of wood for trellis system; iii) change the fertilizer supply chain in order to reduce transportation; iv) reduce the number of interventions on vineyards, which implies the reduction of machinery use to reduce diesel consumption; and vi) fertilizers and pesticides dosage control, as well as an exhaustive record of fertilizers and pesticides applied to vineyards, respecting the safety periods prescribed by pesticide manufacturers.

- The environmental burdens of other agricultural techniques such as biodynamic has shown to be lower than conventional ones. The main reasons for this strong decrease for the biodynamic holding was related to the large reduction in diesel consumption (80%), due to the lower application of plant protection products and fertilizers as well as the introduction of manual work rather than mechanized activities in the vineyards. However, the production yield of the biodynamic holding was lower and, therefore, the impact for land competition resulted to be higher. Additionally, human labor impacts were also higher for the biodynamic site due to the introduction of manual work rather than mechanized activities.

- Traditionally, the GHG emissions linked to viticulture in LCA studies have been focused on agriculture operations whilst emissions derived from LUCs have been disregarded from system limits. Thus, the “Ribeiro” appellation was assessed as a whole in order to evaluate the GHG emissions derived from vineyard operations and LUCs over 20 years, taking into account the technological improvements and changes in the management of plant protection agents, fertilizers, and energy sources for machinery. The results obtained showed the relevance of carbon emissions/fixation derived from LUCs depending on the previous LUs, and how the social and demographic patterns could trigger the LUCs patterns over the years.

- The GHG emissions —vineyard operations and LUCs— for the whole "Ribeiro" appellation showed two different tendencies in different time period. Therefore, the GHGs from LUCs should not be neglected in future viticulture studies and appellations.

In addition to the application of LCA to wine sector, the combined five-step LCA+DEA methodology has proved to be a useful tool for wine-growing. A total of 40 vine-growing exploitations in the "Rías Baixas" appellation were analyzed in order to determine the level of operational efficiency of each producer. The main insights obtained from this study were:

- A high percentage of the vineyards (60%) were found to operate efficiently. In this way, a series of differences were identified between the vine-growing farms depending on their production size. Vine-growers with low production exploitations presented an average efficiency of 79%, lower than intermediate production vineyards (83%) and high production sites (86%). These differences in the average scores are mainly linked to operational differences.

- The results led to average reduction levels of up to 30% per material input, which translated into environmental gains that ranged from 28% to 39% depending on the selected impact category. Additionally, input minimization was considered from an economic point of view, in order to determine to what extent operational optimization may translate into economic benefits for vine-growers. Therefore, the inefficient vineyards assessed in this study would obtain on average an additional economic benefit of 10% with respect to the sale price for grapes.

Section III

Concerning fishing and seafood processing sectors, the combination of energy return on investment (EROI) along LCA methodology allowed a reduction of uncertainties in EROI estimations. The key conclusions obtained from the assessment of Galician fleet through ep-EROI were:

- The indicator ep-EROI has shown to be useful to report the energy efficiency for fishing sector, where small pelagic species presented the highest ep-EROI values if captured using specific fishing techniques.
- Based on the ep-EROI values estimated for each of the different fishing fleets, the mean ep-EROI for the entire Galician extractive fishing sector was 7.6%.
- By fishing fleet, the higher ep-EROI was obtained for open sea fleets (11.2%), followed by coastal fleets (9.2%), whereas the worst results were obtained for the offshore fleets (1.6%). Furthermore, fishing vessels relying on passive fishing gears were those that showed the highest ep-EROI values, whilst trawling fleets presented the lowest return values.
- ep-EROI figures took lower values when the entire supply chain was considered for its calculation due to the addition of more stages —which required energy supply— to the process.
- Allocation procedure and standard deviations of operational inputs were found to be a source on uncertainty that could vary final ep-EROI values.

CF has proved to be a useful tool for the assessment of the life cycle GHG emissions associated with a given product or process. Thus, the calculation of CF of seafood products supply chain such as hake fish sticks had remained unexplored because of the inclusion of other ingredients from agricultural systems and the omission of the consumption phase. The main conclusions obtained from the whole supply chain of hake fish sticks were the following:

- Fishery operations were responsible for over 50% of total emissions, highlighting the importance of fuel and cooling agent leakage, followed by the production of other ingredients such as wheat base products and sunflower oil.
- A reduction in GHG emissions based on energy efficiency in the fishery does not seem feasible, but CF could be lowered by introducing refrigerant agents which did not create impacts in terms of GHGs.
- A set of 24 scenarios of fish sticks consumption were evaluated based on the different cooking methods and consumer choices. When the whole supply chain was analyzed, the final CF was found to account by 32% on average, ranging from 20% (best-performing scenario) to 40% (worst-performing scenario).
- Results proved a high variability in the CF of frozen seafood products depending on the behavioral patterns the buyers manifest, suggesting that substantial reductions in GHG emissions can be achieved without altering the diet patterns of the population.

The application of LCA to the European pilchard supply chain has shown to be useful to identify the environmental profile of several products —cross-product: canned sardines, fresh sardines, and European hake caught using sardine as bait— based on an equal amount of protein. The LCA of the derived seafood products derived from sardine has led to the following general conclusions:

- The results of this study suggested that the way in which consumers decide to intake a certain product may strongly influence the environmental impacts derived from this process.

- Cooking methods of identical purchased products showed relevant environmental disparities.
- Final seafood products with minor processing stages showed environmental benefits of up to 95% as compared to more elaborate methods.
- The main hot spots identified for the three assessed scenarios were directly linked to the characteristics of them. In this regard, consumption of fuel-oil, packaging items (e.g. corrugated board) and olive oil, and especially tin for canning, were important carriers of environmental impact. Additionally, environmental burdens related to fuel consumption were higher for European hake fishing due to the double fishery scheme for this scenario.

Finally, LCA was applied to other marine related processes such as seaweed wild harvesting and processing. Thus, sodium alginate extraction from a brown seaweed (*Nereocystis luetkeana*) was evaluated, obtaining the following conclusions:

- Extraction processes was responsible for most environmental burdens, highlighting the contribution of electricity consumed by lab equipment, followed by chemicals consumed during extraction process —mainly ethanol.
- Impact derived from seaweed harvesting, as fishing LCA studies, was dominated by fuel consumption.
- A set of improvement actions were evaluated to lower the environmental burdens. Given the fact that a change on the electricity mix would be difficult to put into practice, the increase of both harvesting and extraction yield are more feasible scenarios, obtaining a significant impact reduction.
- Ethanol recovering has shown to be a key issue for this kind of processes.

ADDITIONAL CONTENTS

Appendix
Complementary data

Section A presents data regarding relative contributions for wine production in the "Ribeiro" appellation.

Table A.1. Relative contribution (%) to ADP and AP for the different subsystems in the assessed period

ADP	2007	2008	2009	2010	AP	2007	2008	2009	2010
Grape production					*Grape production*				
Infrastructure	23.3	23.2	24.6	23.2	Infrastructure	10.0	10.0	10.4	10.0
Diesel (P and C)	32.4	32.9	29.9	32.9	Diesel (P and C)	24.8	25.2	22.6	25.3
Compost (P and T)	35.8	35.5	37.9	35.6	Compost (P and T)	61.1	60.9	63.7	60.8
Pesticide (P)	6.9	6.9	6.2	6.9	Pesticide (P)	3.4	3.4	2.9	3.4
Electricity	1.2	1.1	1.0	1.0	Electricity	0.5	0.4	0.3	0.3
Field operations	0.3	0.3	0.3	0.3	Field operations	0.1	0.1	0.1	0.1
Other	0.1	0.1	0.1	0.1	Other	0.1	0.0	0.0	0.1
Total	100.0	100.0	100.0	100.0	Total	100.0	100.0	100.0	100.0
Wine processing					*Wine processing*				
Electricity	95.6	95.1	94.5	94.3	Electricity	98.0	97.0	96.4	96.0
Wastewater treatment	0.3	0.4	0.4	0.5	Wastewater treatment	0.6	0.9	1.1	1.3
Propane (P)	1.5	1.8	2.0	1.9	Propane (P)	0.4	0.6	0.8	0.8
Water (tap)	0.5	0.5	0.6	0.7	Water (tap)	0.3	0.4	0.5	0.6
Ethylene glycol	1.2	1.3	1.5	1.5	Ethylene glycol	0.3	0.4	0.5	0.5
Other	0.9	0.9	1.0	1.1	Other	0.4	0.7	0.7	0.8
Total	100.0	100.0	100.0	100.0	Total	100.0	100.0	100.0	100.0
Bottling and packaging					*Bottling and packaging*				
Bottle (P)	78.0				Bottle (P)	86.8			
Bottle (T to winery)	8.0				Bottle (T to winery)	5.9			
Cork (P and T)	2.5				Cork (P and T)	2.1			
Bottle accessories	1.6				Bottle accessories	1.5			
Packaging	9.9				Packaging	3.7			
Total	100.0				Total	100.0			

ADP= abiotic depletion potential; AP= acidification potential; P= production; C= consumption; T= transport.

Table A.2. Relative contribution (%) to EP and GWP for the different subsystems in the assessed period

EP	2007	2008	2009	2010	GWP	2007	2008	2009	2010
Grape production					*Grape production*				
Infrastructure	2.0	1.9	1.9	2.0	Infrastructure	7.3	7.2	7.4	7.2
Diesel (P and C)	10.5	10.5	9.1	10.9	Diesel (P and C)	15.4	15.6	13.7	15.7
Emissions (fertilizers)	48.1	49.4	50.6	47.3	Emissions (fertilizers)	7.0	7.0	7.2	7.0
Compost (P and T)	35.8	34.8	35.4	36.3	Compost (P and T)	67.5	67.4	69.2	67.3
Pesticide (P)	3.2	3.1	2.7	3.3	Pesticide (P)	2.3	2.3	2.0	2.3
Electricity	0.3	0.2	0.1	0.1	Electricity	0.4	0.4	0.4	0.4
Field operations	0.1	0.1	0.1	0.1	Field operations	0.1	0.1	0.1	0.1
Total	100.0	100.0	100.0	100.0	Total	100.0	100.0	100.0	100.0
Wine processing					*Wine processing*				
Electricity	85.3	79.4	76.2	73.1	Electricity	96.5	96.1	95.6	95.4
Wastewater treatment	12.8	17.9	20.6	23.5	Wastewater treatment	0.6	0.7	0.7	0.9
Propane (P)	0.3	0.4	0.4	0.4	Propane (P)	0.3	0.3	0.4	0.4
Water (tap)	0.9	1.2	1.4	1.6	Water (tap)	0.5	0.6	0.7	0.8
Ethylene glycol	0.6	0.8	1.0	0.9	Ethylene glycol	0.6	0.7	0.8	0.8
Winery emissions	0.0	0.0	0.0	0.0	Winery emissions	1.0	1.1	1.3	1.2
Other	0.1	0.3	0.4	0.5	Other	0.5	0.5	0.5	0.5
Total	100.0	100.0	100.0	100.0	Total	100.0	100.0	100.0	100.0
Bottling and packaging					*Bottling and packaging*				
Bottle (P)	76.2				Bottle (P)	79.2			
Bottle (T to winery)	8.8				Bottle (T to winery)	8.5			
Cork (P and T)	1.7				Cork (P and T)	1.7			
Bottle accessories	2.1				Bottle accessories	1.8			
Packaging	10.1				Packaging	8.6			
Waste treatment	1.1				Waste treatment	0.2			
Total	100.0				Total	100.0			

EP= eutrophication potential; GWP= global warming potential; P= production; C= consumption; T= transport.

Table A.3. Relative contribution (%) to ODP and POFP for the different subsystems in the assessed period

ODP	2007	2008	2009	2010	POFP	2007	2008	2009	2010
Grape production					*Grape production*				
Infrastructure	0.2	0.2	0.3	0.2	Infrastructure	27.9	27.8	28.7	27.8
Diesel (P and C)	17.7	18.1	17.3	18.0	Diesel (P and C)	16.6	17.1	15.0	17.0
Compost (P and T)	22.5	22.4	25.2	22.4	Compost (P and T)	50.2	50.0	51.8	50.1
Pesticide (P)	59.2	58.9	56.7	59.0	Pesticide (P)	4.7	4.7	4.1	4.7
Electricity	0.3	0.3	0.3	0.3	Electricity	0.5	0.3	0.3	0.3
Field operations	0.1	0.1	0.1	0.1	Field operations	0.1	0.1	0.1	0.1
Total	100.0	100.0	100.0	100.0	Total	100.0	100.0	100.0	100.0
Wine processing					*Wine processing*				
Electricity	93.8	94.5	94.1	93.6	Electricity	56.0	47.9	43.1	38.1
Wastewater treatment	1.1	1.0	1.0	1.2	Wastewater treatment	0.4	0.4	0.5	0.5
Propane (P)	3.5	3.2	3.5	3.4	Propane (P)	0.3	0.4	0.5	0.4
Winery emissions	0.0	0.0	0.0	0.0	Winery emissions	42.4	50.0	54.7	59.7
Other	1.6	1.3	1.4	1.8	Other	0.9	1.3	1.2	1.3
Total	100.0	100.0	100.0	100.0	Total	100.0	100.0	100.0	100.0
Bottling and packaging					*Bottling and packaging*				
Bottle (P)	76.2				Bottle (P)	77.7			
Bottle (T to winery)	12.1				Bottle (T to winery)	4.2			
Cork (P and T)	2.7				Cork (P and T)	6.8			
Bottle accessories	3.3				Bottle accessories	4.3			
Packaging	5.7				Packaging	6.8			
Waste treatment	0.0				Waste treatment	0.2			
Total	100.0				Total	100.0			

ODP= ozone layer depletion potential; POFP= photochemical oxidant formation potential; P= production; C= consumption; T= transport.

Table A.4. Relative contribution (%) to LC and aquatic eco-toxicity (Etox) for the different subsystems in the assessed period

LC	2007	2008	2009	2010	Etox	2007	2008	2009	2010
Grape production					*Grape production*				
Pesticide emissions	0.0	0.0	0.0	0.0	Pesticide (P)	0.6	0.6	0.4	0.5
Infrastructure	22.7	22.7	22.9	22.8	Compost (P and T)	0.9	0.9	0.8	0.8
Diesel (P and C)	1.4	1.5	1.3	1.5	Terbuthylazine emissions	53.2	52.7	57.4	57.5
Compost (P and T)	70.9	70.9	71.5	71.0	Folpet emissions	44.4	44.0	39.7	39.2
Pesticide (P)	2.9	2.9	2.5	2.9	Mancozeb emissions	0.6	0.6	0.5	0.7
Field operations	1.4	1.4	1.4	1.4	Fludioxinil emissions	0.1	0.9	0.9	0.9
Electricity	0.5	0.4	0.4	0.4	--	--	--	--	--
Other	0.2	0.2	0.0	0.0	Other	0.3	0.3	0.3	0.4
Total	100.0	100.0	100.0	100.0	Total	100.0	100.0	100.0	100.0
Wine processing					*Wine processing*				
Electricity	95.7	95.0	94.1	93.2	Electricity	89.6	86.0	84.5	82.6
Water (tap)	2.6	3.0	3.5	4.2	Wastewater treatment	6.7	9.0	9.9	11.5
Other	1.7	2.0	2.4	2.6	Other	3.7	5.0	5.6	5.9
Total	100.0	100.0	100.0	100.0	Total	100.0	100.0	100.0	100.0
Bottling and packaging					*Bottling and packaging*				
Bottle (P)	41.6				Bottle (P)	66.9			
Pallet (P)	38.8				Bottle (T to winery)	7.8			
Corrugated board	13.6				Packaging	19.4			
Other	6.0				Other	5.9			
Total	100.0				Total	100.0			

LC= land competition; Etox= eco-toxicty; P= production; C= consumption; T= transport.

Section B presents data regarding synthetic pesticides emissions —atmosphere and water— calculated through PestLCI for the conventional site (CV) and characterization values, in terms of human labor (HL), for the two modelled viticulture sites: biodynamic (BD) and conventional (CV).

Table B.1. Inventory of the emissions derived from pesticides use in the conventional farm

	To the environment		
Emissions to the atmosphere	Unit	2010	2011
Cymoxanil	mg	9.12	7.20
Cyprodinil	mg	28.50	22.49
Fludioxinil	mg	19.00	14.99
Folpet	mg	117.9	92.96
Fosetyl-Al	mg	508.7	323.5
Glyphosate	mg	248.0	195.8
Iprovalicarb	mg	36.15	21.41
Mancozeb	mg	304.3	180.1
Metalaxyl	mg	7.30	5.76
Terbuthylazine	mg	239.7	189.9
Triadimenol	mg	12.05	9.51
Emissions to the water			
Cymoxanil	mg	1.4E-12	0.00
Cyprodinil	mg	1.40	2.2E-68
Fludioxinil	mg	2.87	1.8E-05
Folpet	mg	4.4E-03	0.00
Fosetyl-Al	mg	0.00	0.00
Glyphosate	mg	10.00	4.6E-82
Iprovalicarb	mg	0.12	6.1E-211
Mancozeb	mg	1.42	1.2E-204
Metalaxyl	mg	0.70	2.0E-28
Terbuthylazine	mg	29.10	1.8E-16
Triadimenol	mg	1.49	6.6E-17

Table B.2. Characterization results per impact category for human labor (%) for the conventional (CV) cultivation site

Impact category	**Unit**	**Human Labor (HL)**	**Total**	**% HL**
Climate Change	g CO2 eq	5.60	379.3	1.48
Human Toxicity	g 1,4-DB eq	0.02	160.7	0.10
Photochemical Oxidant Formation	g NMVOC	1.35E-2	4.92	0.27
Particulate Matter Formation	g PM10 eq	1.30E-2	1.71	0.76
Terrestrial Acidification	g SO_2 eq	3.89E-2	4.78	0.81
Freshwater Eutrophication	g P eq	5.19E-4	0.11	0.48
Marine Eutrophication	g N eq	1.97E-2	3.42	0.57
Terrestrial Eco-toxicity	g 1,4 DB eq	8.28E-3	4.87E-2	14.5
Freshwater Eco-toxicity	g 1,4 DB eq	6.74E-2	2.44	2.69
Marine Eco-toxicity	g 1,4 DB eq	5.26E-3	2.16	0.25
Agricultural Land Occupation	m^2a	1.89	1177	0.16
Urban Land Occupation	m^2a	4.77E-2	1.58	2.92
Mineral Depletion	g Fe eq	0.25	127.7	0.20
Fossil Depletion	g oil eq	4.21	112.8	3.60

Table B.3. Characterization results per impact category for human labor (%) for the biodynamic (BD) cultivation site

Impact category	**Unit**	**Human labour (HL)**	**Total**	**% HL**
Climate Change	g CO_2 eq	13.40	109.8	12.2
Human Toxicity	g 1,4 DB eq	0.40	57.70	0.69
Photochemical oxidant formation	g NMVOC	3.21E-2	1.19	2.70
Particulate Matter Formation	g PM10 eq	3.11E-2	0.42	7.44
Terrestrial Acidification	g SO_2 eq	9.28E-2	0.97	9.58
Freshwater Eutrophication	g N eq	1.24E-3	0.03	4.02
Marine Eutrophication	g P eq	4.71E-2	0.11	0.44
Terrestrial Eco-toxicity	g 1,4 DB eq	1.98E-2	0.04	53.8
Freshwater Eco-toxicity	g 1,4 DB eq	0.16	0.93	17.2
Marine Eco-toxicity	g 1,4 DB eq	1.28E-2	0.91	1.41
Agricultural Land Occupation	m^2a	4.25E-3	2.45	0.18
Urban Land Occupation	m^2a	1.14E-3	0.29	0.04
Mineral Depletion	g Fe eq	0.60	22.71	2.64
Fossil Depletion	g oil eq	10.00	40.45	24.8

Section C offers complementary data for LUC dynamic, GHG emissions (derived from LUCs and vineyard operations) through assessed period, and main equations used to carry out the study.

Equation C.1. $$\Delta \boldsymbol{C}_{LU} = \Delta \boldsymbol{C}_{\boldsymbol{Biomass}} + \Delta \boldsymbol{C}_{\boldsymbol{Dead\ organic\ Matter}} + \Delta \boldsymbol{C}_{\boldsymbol{Mineral\ Soils}}$$

Where:
ΔC_{LU} = Annual change in carbon stocks from Land Use.
$\Delta C_{Biomass}$ = Annual change in carbon stocks from living biomass.
$\Delta C_{Dead\ organic\ Matter}$ = Annual change in carbon stocks in dead organic. This study excludes this compartment from carbon stocks calculation.
$\Delta C_{Mineral\ Soils}$= Annual change in carbon stocks in mineral soils.

Equation C.2. $$\Delta \boldsymbol{C}_{\boldsymbol{Biomass}} = \Delta \boldsymbol{C}_{\boldsymbol{Growth}} - \Delta \boldsymbol{C}_{\boldsymbol{Losses}}$$

Where:
ΔC_{Growth} = Annual increase in carbon stocks due to biomass growth.
ΔC_{Losses} = Annual decrease in carbon stocks due to biomass loss (i.e. wood, firewood, harvesting, and forest fires)

Equation C.3. $$\Delta \boldsymbol{C}_{\boldsymbol{Mineral\ Soils}} = \frac{(\boldsymbol{SOC}_{\boldsymbol{After\ LUC}} - \boldsymbol{SOC}_{\boldsymbol{Before\ LUC}})}{\boldsymbol{Time}}$$

Where:
$SOC_{After\ LUC}$= Soil organic carbon under current state.
$SOC_{Before\ LUC}$ = Soil organic carbon under previous state.
$Time$ = Default time 20 years.

Equation C.4. $$\boldsymbol{SOC} = \boldsymbol{SOC}_{\boldsymbol{ref}} \cdot \boldsymbol{F}_{\boldsymbol{LU}} \cdot \boldsymbol{F}_{\boldsymbol{MG}} \cdot \boldsymbol{F}_{\boldsymbol{I}}$$

Where:
SOC_{ref} = Reference carbon stock unmanaged.
F_{LU} = Stock change factor for land use or land use change type.
F_{MG}= Stock change factor due to management regime.
F_I = Stock change factor for input of organic matter.

Equation C.5. $$\Delta \boldsymbol{C}_{\boldsymbol{Land\ Conversion\ Biomass}} = \Delta \boldsymbol{C}_{\boldsymbol{Growth}} + \Delta \boldsymbol{C}_{\boldsymbol{Conversion}} - \Delta \boldsymbol{C}_{\boldsymbol{Losses}}$$

Where:
$\Delta C_{Land\ Conversion\ Biomass}$ = Annual change in carbon living biomass due to land conversion.
ΔC_{Growth} = Annual increase in carbon stocks due to new living biomass growth.
$\Delta C_{Conversion}$= annual change in carbon stocks in living biomass due to land conversion.
ΔC_{Losses} = Annual decrease in carbon stocks due to biomass losses (i.e. wood, firewood, harvesting, and forest fires)

Equation C.6. $$\Delta \boldsymbol{C}_{\boldsymbol{Conversion}} = (\boldsymbol{B}_{\boldsymbol{After\ LUC}} - \boldsymbol{B}_{\boldsymbol{Before\ LUC}}) \cdot \boldsymbol{CF}$$

Where:
$B_{After\ LUC}$ = Biomass stocks after land conversion.
$B_{Before\ LUC}$ = Biomass stocks before land conversion.
CF= Carbon fraction of dry matter.

Table C.7. Life cycle inventory during assessed period (FU: 5,380 ha)

		Year									
	Units	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**
Diesel	t	700.25	684.47	672.09	652.19	635.10	620.84	890.01	1,027.0	930.35	977.39
Gasoline	t	64.74	56.92	49.56	42.66	36.22	30.24	55.61	74.13	56.41	56.24
Mineral Fertilizers	t	132.25	126.66	112.80	92.24	73.21	64.29	77.14	65.40	67.66	69.41
Organic fertilizers	t	4,286.9	4,125.8	3,964.7	3,803.6	3,642.5	3,481.4	3,320.3	3,159.2	2,998.1	2,837.0
Pesticides											
Glyphosate	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Bordeaux Broth	t	68.92	50.17	80.29	32.20	59.39	57.69	41.64	47.71	40.78	42.02
Sulfur	t	514.43	493.44	478.25	449.80	437.10	417.77	204.29	234.44	199.74	204.98
Copper 25%	t	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.29	0.30	0.36
[sulfonyl]urea-compounds	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
[thio]carbamate compounds	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Acetamide-anillide compounds	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Dithiocarbamate compounds	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pyretroid-compounds	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Benzo[thia]diazole compounds	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitrile-compounds	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cyclic N-compounds	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Folpet	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Phtalamide-compounds	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Triazine compounds	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pesticide unspecified	t	0.04	0.08	0.12	0.21	0.13	0.24	5.68	6.67	6.47	6.87
Trellis											
Granite	t	31.58	30.34	29.09	27.84	26.57	25.31	23.88	22.53	21.08	19.82
Slate	t	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.20	0.22	0.23
Concrete	t	2.61	2.70	2.78	2.83	2.86	2.87	2.81	2.78	2.74	2.62
Iron	t	0.00	0.00	0.02	0.08	0.23	0.40	0.93	1.33	1.79	2.12
Wood	t	14.16	12.82	11.55	10.34	9.20	8.12	7.11	6.17	5.29	4.48

Table C.7. Life cycle inventory during assessed period (FU: 5,380 ha) (continuation)

		Year									
	Units	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**
Diesel	t	876.7	1011.4	961.2	910.1	858.0	936.9	1004.5	1073.0	1081.0	964.9
Gasoline	t	44.93	38.87	37.05	35.28	33.55	33.95	34.31	34.62	33.92	31.29
Mineral Fertilizers	t	73.35	66.97	60.78	72.60	66.35	58.13	62.90	65.84	50.47	51.43
Organic fertilizers	t	2,857.6	2,878.2	2,898.8	2,919.4	2,940.0	2,960.6	2,981.2	3,001.8	3,022.4	3,043.0
Pesticides											
Glyphosate	t	7.04	7.04	7.14	7.19	7.22	7.29	7.35	7.39	7.44	7.50
Bordeaux Broth	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Sulfur	t	25.72	25.73	21.74	26.27	26.41	26.65	22.36	27.02	27.20	22.82
Copper 25%	t	30.29	30.30	24.64	30.94	31.10	31.38	25.34	31.81	32.03	25.86
[sulfonyl]urea-compounds	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
[thio]carbamate compounds	t	1.28	1.28	1.04	1.30	1.31	1.32	1.07	1.34	1.35	1.09
Acetamide-anillide compounds	t	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.22	0.22	0.22
Dithiocarbamate compounds	t	8.58	8.59	6.53	8.77	8.81	8.89	6.71	9.01	9.08	6.85
Pyretroid-compounds	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Benzo[thia]diazole compounds	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitrile-compounds	t	1.34	1.34	1.36	1.37	1.38	1.39	1.40	1.41	1.42	1.43
Cyclic N-compounds	t	0.34	0.34	0.00	0.35	0.35	0.35	0.00	0.36	0.36	0.00
Folpet	t	14.35	14.35	11.72	14.66	14.73	14.86	12.06	15.07	15.17	12.31
Phtalamide-compounds	t	3.32	3.32	3.37	3.39	3.41	3.44	3.47	3.49	3.51	3.54
Triazine compounds	t	6.80	6.80	6.90	6.95	6.98	7.05	7.10	7.14	7.19	7.24
Pesticide unspecified	t	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Trellis											
Granite	t	19.48	19.01	18.50	18.33	17.80	17.58	17.16	16.75	16.26	15.34
Slate	t	0.26	0.28	0.31	0.33	0.59	0.83	1.07	1.30	1.52	1.73
Concrete	t	2.70	2.76	2.78	2.78	2.75	2.63	2.58	2.59	2.55	2.55
Iron	t	2.87	3.72	4.83	5.93	7.42	8.74	10.15	11.27	12.84	15.17
Wood	t	4.18	3.89	3.61	3.33	3.07	2.81	2.55	2.31	2.08	1.85

Table C.8. Complete LUCs vineyards related during assessed period. Data is presented in hectares

LUCs	Year																				
	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
Remaining forest	535	535	535	535	535	535	535	535	535	535	535	482	428	375	321	268	214	161	107	54	0
Forest in transition	0	161	323	484	645	807	968	1,129	1,290	1,452	1,613	1,613	1,613	1,613	1,613	1,613	1,613	1,613	1,613	1,613	1,613
Vineyards-Forest	*0*	*161*	*323*	*484*	*645*	*807*	*968*	*1,129*	*1,290*	*1,452*	*1,613*	*1,613*	*1,613*	*1,613*	*1,613*	*1,613*	*1,613*	*1,613*	*1,613*	*1,613*	*1,613*
Remaining vineyards	4,448	4,259	4,071	3,882	3,693	3,504	3,316	3,127	2,938	2,749	2,561	2,516	2,471	2,426	2,381	2,336	2,291	2,246	2,201	2,156	2,111
Vineyards in transition	0	28	55	83	111	138	166	194	221	249	277	342	408	473	539	604	670	735	801	866	932
Forest-Vineyards	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*54*	*107*	*161*	*214*	*268*	*321*	*375*	*428*	*482*	*535*
Grassland-Vineyard	*0*	*4*	*8*	*12*	*16*	*20*	*24*	*28*	*32*	*36*	*41*	*45*	*49*	*53*	*57*	*61*	*65*	*69*	*73*	*77*	*81*
Cropland-Vineyard	*0*	*16*	*31*	*47*	*62*	*78*	*94*	*109*	*125*	*140*	*156*	*156*	*156*	*156*	*156*	*156*	*156*	*156*	*156*	*156*	*156*
Other land-Vineyard	*0*	*8*	*16*	*24*	*32*	*40*	*48*	*56*	*64*	*72*	*80*	*88*	*96*	*104*	*112*	*120*	*128*	*136*	*144*	*152*	*160*
Remaining grassland	81	77	73	69	65	61	57	53	49	45	41	36	32	28	24	20	16	12	8	4	0
Grassland in transition	0	5	10	15	20	25	30	35	40	45	50	55	60	65	70	75	80	85	90	95	100
Vineyard-Grassland	*0*	*5*	*10*	*15*	*20*	*25*	*30*	*35*	*40*	*45*	*50*	*55*	*60*	*65*	*70*	*75*	*80*	*85*	*90*	*95*	*100*
Remaining cropland	156	140	125	109	94	78	62	47	31	16	0	0	0	0	0	0	0	0	0	0	0
Cropland in transition	0	0	0	0	0	0	0	0	0	0	0	18	35	53	70	88	105	123	140	158	175
Vineyard-Cropland	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*18*	*35*	*53*	*70*	*88*	*105*	*123*	*140*	*158*	*175*
Remaining other land	160	152	144	136	128	120	112	104	96	88	80	72	64	56	48	40	32	24	16	8	0
Other land in transition	0	19	38	58	77	96	115	134	154	173	192	211	230	250	269	288	307	326	346	365	384
Vineyard-Other land	*0*	*19*	*38*	*58*	*77*	*96*	*115*	*134*	*154*	*173*	*192*	*211*	*230*	*250*	*269*	*288*	*307*	*326*	*346*	*365*	*384*
Remaining wetlands	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Wetlands in transition	0	3	7	10	13	16	20	23	26	29	33	36	39	42	46	49	52	55	59	62	65
Vineyard-Wetlands	*0*	*3*	*7*	*10*	*13*	*16*	*20*	*23*	*26*	*29*	*33*	*36*	*39*	*42*	*46*	*49*	*52*	*55*	*59*	*62*	*65*
TOTAL	5,380	5,380	5,380	5,380	5,380	5,380	5,380	5,380	5,380	5,380	5,380	5,380	5,380	5,380	5,380	5,380	5,380	5,380	5,380	5,380	5,380

Table C.9. GHGs breakdown related to LUCs during assessed period. Negative values mean carbon sink/fixation, whilst positive values mean carbon source/emissions

		1990	1991	1992	1993	1994	1995	1996	1997	1998	1999
Remaining forest	Biomass	-1.35E+04	-1.34E+04	-1.37E+04	-1.38E+04	-1.39E+04	-1.39E+04	-1.38E+04	-1.37E+04	-1.36E+04	-1.36E+04
	Soil	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	Forest fires	1.53E+01	1.77E+01	2.80E+00	1.84E+00	8.24E-01	1.03E+00	1.07E+01	3.05E+00	6.43E+00	7.98E+00
Vineyard-Forest land	Biomass	0.00E+00	7.11E+01	-2.13E+02	-4.98E+02	-7.82E+02	-1.07E+03	-1.35E+03	-1.63E+03	-1.92E+03	-2.20E+03
	Soil	0.00E+00	-4.11E+02	-8.23E+02	-1.23E+03	-1.65E+03	-2.06E+03	-2.47E+03	-2.88E+03	-3.29E+03	-3.70E+03
	Forest fires	0.00E+00	3.76E-02	1.19E-02	1.17E-02	6.99E-03	1.09E-02	1.36E-01	4.52E-02	1.09E-01	1.52E-01
Remaining vineyards	Biomass	1.27E+04	1.22E+04	1.16E+04	1.11E+04	1.06E+04	1.00E+04	9.48E+03	8.94E+03	8.40E+03	7.86E+03
	Soil	4.10E+03	3.93E+03	3.75E+03	3.58E+03	3.41E+03	3.23E+03	3.06E+03	2.88E+03	2.71E+03	2.54E+03
	Fertilizers	1.00E+03	2.11E+03	2.01E+03	1.83E+03	1.60E+03	1.73E+03	1.56E+03	1.69E+03	1.48E+03	1.46E+03
Forest-Vineyards	Biomass	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	Soil	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	Fertilizers	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cropland-Vineyard	Biomass	0.00E+00	-7.04E+01	-1.36E+02	-2.07E+02	-2.73E+02	-3.43E+02	-4.14E+02	-4.80E+02	-5.50E+02	-6.16E+02
	Soil	0.00E+00	4.71E+01	9.13E+01	1.38E+02	1.83E+02	2.30E+02	2.77E+02	3.21E+02	3.68E+02	4.12E+02
	Fertilizers	0.00E+00	1.14E+01	2.21E+01	3.24E+01	4.00E+01	5.51E+01	6.41E+01	8.23E+01	8.95E+01	1.04E+02
Grassland-Vineyard	Biomass	0.00E+00	2.20E+00	-1.54E+01	-3.30E+01	-5.06E+01	-6.82E+01	-8.58E+01	-1.03E+02	-1.21E+02	-1.39E+02
	Soil	0.00E+00	1.29E+01	2.58E+01	3.87E+01	5.16E+01	6.45E+01	7.74E+01	9.03E+01	1.03E+02	1.16E+02
	Fertilizers	0.00E+00	2.33E+00	4.65E+00	6.84E+00	8.78E+00	1.05E+01	1.24E+01	1.52E+01	1.70E+01	1.94E+01
Other land-Vineyard	Biomass	0.00E+00	-3.52E+01	-7.04E+01	-1.06E+02	-1.41E+02	-1.76E+02	-2.11E+02	-2.46E+02	-2.82E+02	-3.17E+02
	Soil	0.00E+00	1.57E+01	3.14E+01	4.71E+01	6.27E+01	7.84E+01	9.41E+01	1.10E+02	1.25E+02	1.41E+02
	Fertilizers	0.00E+00	4.94E+00	9.85E+00	1.42E+01	1.76E+01	2.44E+01	2.81E+01	3.69E+01	3.96E+01	4.67E+01
Remaining cropland	Biomass	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	Soil	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	Fertilizers	7.58E+01	6.95E+01	6.14E+01	5.09E+01	3.99E+01	3.78E+01	2.84E+01	2.48E+01	1.65E+01	9.44E+00
Vineyard-Copland	Biomass	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	Soil	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	Fertilizers	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Vineyard-Grassland	Biomass	0.00E+00	1.10E+01	2.20E+01	3.30E+01	1.10E+01	1.10E+01	1.10E+01	1.10E+01	1.10E+01	1.10E+01
	Soil	0.00E+00	-1.78E+01	-3.56E+01	-5.34E+01	-7.12E+01	-8.90E+01	-1.07E+02	-1.25E+02	-1.42E+02	-1.60E+02
Vineyard-Other land	Biomass	0.00E+00	4.22E+01	8.45E+01	1.27E+02	1.69E+02	2.11E+02	2.53E+02	2.96E+02	3.38E+02	3.80E+02
	Soil	0.00E+00	-4.90E+01	-9.79E+01	-1.47E+02	-1.96E+02	-2.45E+02	-2.94E+02	-3.43E+02	-3.92E+02	-4.41E+02
Vineyard-Wetlands	Biomass	0.00E+00	7.15E+00	1.43E+01	2.15E+01	2.86E+01	3.58E+01	4.29E+01	5.01E+01	5.72E+01	6.44E+01
	Soil	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00

Table C.9. GHGs breakdown related to LUCs during assessed period. Negative values mean carbon sink/fixation, whilst positive values mean carbon source/emissions (continuation)

		2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
Remaining forest	Biomass	-1.24E+04	-1.11E+04	-9.76E+03	-8.03E+03	-6.76E+03	-5.41E+03	-3.95E+03	-2.68E+03	-1.34E+03	0.00E+00
	Soil	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	Forest fires	4.85E+00	1.81E+00	1.24E+01	3.06E+00	1.87E+01	1.91E+01	1.74E+01	3.39E-01	2.56E-01	0.00E+00
Vineyard-Forest land	Biomass	-1.18E+03	-1.49E+03	-1.70E+03	7.56E+01	-5.33E+02	-5.52E+02	4.69E+02	-3.99E+01	1.33E+02	-5.82E+02
	Soil	-4.11E+03	-4.11E+03	-4.11E+03	-4.11E+03	-4.11E+03	-4.11E+03	-4.11E+03	-4.11E+03	-4.11E+03	-4.11E+03
	Forest fires	1.14E-01	4.80E-02	3.75E-01	1.08E-01	7.93E-01	1.01E+00	1.23E+00	3.59E-02	5.44E-02	1.68E-01
Remaining vineyards	Biomass	7.19E+03	7.07E+03	6.94E+03	6.81E+03	6.68E+03	6.55E+03	6.42E+03	6.29E+03	6.17E+03	6.04E+03
	Soil	2.32E+03	2.28E+03	2.24E+03	2.20E+03	2.15E+03	2.11E+03	2.07E+03	2.03E+03	1.99E+03	1.95E+03
	Fertilizers	1.45E+03	1.32E+03	1.22E+03	1.33E+03	1.23E+03	1.11E+03	1.13E+03	1.14E+03	9.48E+02	9.35E+02
Forest-Vineyards	Biomass	1.26E+04	1.24E+04	1.22E+04	1.19E+04	1.17E+04	1.15E+04	1.12E+04	1.10E+04	1.08E+04	1.05E+04
	Soil	2.28E+02	4.57E+02	6.85E+02	9.14E+02	1.14E+03	1.37E+03	1.60E+03	1.83E+03	2.06E+03	2.28E+03
	Fertilizers	4.94E+01	9.48E+01	1.36E+02	1.95E+02	2.34E+02	2.66E+02	3.19E+02	3.70E+02	3.78E+02	4.21E+02
Cropland-Vineyard	Biomass	4.46E+02	4.46E+02	4.46E+02	4.46E+02	4.46E+02	4.46E+02	4.46E+02	4.46E+02	4.46E+02	4.46E+02
	Soil	4.59E+02	4.59E+02	4.59E+02	4.59E+02	4.59E+02	4.59E+02	4.59E+02	4.59E+02	4.59E+02	4.59E+02
	Fertilizers	1.24E+02	1.18E+02	1.12E+02	1.22E+02	1.16E+02	1.09E+02	1.13E+02	1.15E+02	1.02E+02	1.03E+02
Grassland-Vineyard	Biomass	-1.78E+02	-1.96E+02	-2.13E+02	-2.31E+02	-2.49E+02	-2.66E+02	-2.84E+02	-3.01E+02	-3.19E+02	-3.37E+02
	Soil	1.45E+02	1.58E+02	1.71E+02	1.84E+02	1.97E+02	2.10E+02	2.22E+02	2.35E+02	2.48E+02	2.61E+02
	Fertilizers	2.50E+01	2.66E+01	2.82E+01	3.14E+01	3.29E+01	3.41E+01	3.67E+01	3.92E+01	3.92E+01	4.14E+01
Other land-Vineyard	Biomass	-3.87E+02	-4.22E+02	-4.58E+02	-4.93E+02	-5.28E+02	-5.63E+02	-5.98E+02	-6.34E+02	-6.69E+02	-7.04E+02
	Soil	1.73E+02	1.88E+02	2.04E+02	2.20E+02	2.35E+02	2.51E+02	2.67E+02	2.82E+02	2.98E+02	3.14E+02
	Fertilizers	6.12E+01	6.33E+01	6.48E+01	7.66E+01	7.78E+01	7.72E+01	8.51E+01	9.19E+01	8.47E+01	8.97E+01
Remaining cropland	Biomass	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	Soil	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	Fertilizers	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Vineyard-Copland	Biomass	7.65E+02	7.65E+02	7.65E+02	7.65E+02	7.65E+02	7.65E+02	7.65E+02	7.65E+02	7.65E+02	7.65E+02
	Soil	-4.21E+01	-8.42E+01	-1.26E+02	-1.68E+02	-2.10E+02	-2.53E+02	-2.95E+02	-3.37E+02	-3.79E+02	-4.21E+02
	Fertilizers	1.42E+01	2.92E+01	4.05E+01	6.08E+01	7.51E+01	9.01E+01	1.06E+02	1.21E+02	1.15E+02	1.40E+02
Vineyard-Grassland	Biomass	1.10E+01	1.10E+01	1.10E+01	1.10E+01	1.10E+01	1.10E+01	1.10E+01	1.10E+01	1.10E+01	1.10E+01
	Soil	-1.96E+02	-2.14E+02	-2.31E+02	-2.49E+02	-2.67E+02	-2.85E+02	-3.03E+02	-3.20E+02	-3.38E+02	-3.56E+02
Vineyard-Other land	Biomass	4.65E+02	5.07E+02	5.49E+02	5.91E+02	6.34E+02	6.76E+02	7.18E+02	7.60E+02	8.03E+02	8.45E+02
	Soil	-5.39E+02	-5.88E+02	-6.37E+02	-6.86E+02	-7.35E+02	-7.84E+02	-8.33E+02	-8.82E+02	-9.30E+02	-9.79E+02
Vineyard-Wetlands	Biomass	7.87E+01	8.58E+01	9.30E+01	1.00E+02	1.07E+02	1.14E+02	1.22E+02	1.29E+02	1.36E+02	1.43E+02
	Soil	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00

Table C.10. GHG emissions derived from vineyard operations and LUCs. Data in kg CO_2 eq per ha

Year	Fuel	Pesticides	Fertilizers	Trellis	LUC	***Total***
1990	531.12	41.93	601.77	64.28	841.64	*2,080.73*
1991	514.44	43.97	576.97	62.36	491.71	*1,689.45*
1992	500.44	44.20	514.55	59.79	177.25	*1,296.23*
1993	481.59	33.12	421.70	57.05	-162.94	*830.52*
1994	465.03	37.65	335.83	54.79	-407.62	*485.68*
1995	450.75	36.38	295.87	52.53	-685.69	*149.84*
1996	655.32	23.51	354.89	49.70	-929.18	*154.24*
1997	763.54	35.82	302.11	47.64	-1,209.48	*-60.37*
1998	683.76	32.04	312.99	45.01	-1,486.80	*-413.00*
1999	716.09	33.36	321.58	43.00	-1,810.78	*-696.76*
2000	638.23	66.03	340.89	43.08	1,200.93	*2,289.16*
2001	748.19	66.03	313.33	42.66	1,323.98	*2,494.18*
2002	690.75	59.00	286.63	42.89	1,480.48	*2,559.75*
2003	654.14	61.56	341.78	43.23	2,122.68	*3,223.39*
2004	616.93	61.56	314.81	43.75	2,189.41	*3,226.46*
2005	673.51	66.99	278.88	44.25	2,374.91	*3,438.54*
2006	720.58	64.30	302.03	44.24	2,807.43	*3,938.58*
2007	768.02	66.49	316.81	44.53	2,910.59	*4,106.45*
2008	772.99	66.94	248.40	44.50	3,106.69	*4,239.54*
2009	690.77	58.58	254.17	44.43	3,185.32	*4,233.27*

Section D offers data regarding fish sticks: detailed life cycle inventory for manufacture and characterization results for Cumulative Energy Demand (CED).

Table D.1. Detailed life cycle inventory for the reception at plant and storage step (data per FU)

INPUTS FROM THE TECHNOSPHERE		
Products		
Patagonian grenadier block from SS1	180	g
Materials		
Lubricants (Forklift truck)	8.77E-4	mL
Lubricants (Powered pallet jack)	4.82E-4	mL
Electricity		
Forklift truck	3.33E-5	kWh
Powered pallet jack	2.37E-6	kWh
OUTPUTS TO THE TECHNOSPHERE		
Products		
Patagonian grenadier block	180	g

Table D.2. Detailed life cycle inventory for the unwrapping step (data per FU)

INPUTS FROM THE TECHNOSPHERE		
Products		
Patagonian grenadier block from SS1	180	g
Materials		
Lubricants (Forklift truck)	8.77E-4	mL
Electricity		
Forklift truck	3.33E-5	kWh
OUTPUTS TO THE TECHNOSPHERE		
Products		
Patagonian grenadier block	177.93	g
Co-products		
Damaged or broken blocks	2.07	g
Residues		
Cardboard	13.51	g
Polyethylene (PE)	0.47	g
Other plastics	4.09	g

Table D.3. Detailed life cycle inventory for the cutting step (data per FU)

INPUTS FROM THE TECHNOSPHERE		
Products		
Patagonian grenadier block from SS1	177.93	g
Materials		
Lubricants (Band saw)	8.77E-4	mL
Electricity		
Processing cutter	3.66E-3	kWh
Band saw	5.68E-3	kWh
Powered pallet jack	2.37E-6	kWh
OUTPUTS TO THE TECHNOSPHERE		
Products		
Patagonian grenadier block	164.53	g
Co-products		
Damaged blocks	13.40	g

Table D.4. Detailed life cycle inventory for the battering step (data per FU)

INPUTS FROM THE TECHNOSPHERE		
Products		
Patagonian grenadier block from SS1	164.53	g
Wheat mix (flour)	22.42	g
Water for battering	52.05	g
Electricity		
Mixer	3.66E-3	kWh
OUTPUTS TO THE TECHNOSPHERE		
Products		
Patagonian grenadier fish sticks	238.99	g
Residues		
Excess batter	3.16	mL

Table D.5. Detailed life cycle inventory for the breadcrumb addition step (data per FU)

INPUTS FROM THE TECHNOSPHERE		
Products		
Patagonian grenadier fish sticks	238.99	g
Breadcrumbs	78.00	g
Electricity		
Coating machine	1.84E-3	kWh
OUTPUTS TO THE TECHNOSPHERE		
Products		
Patagonian grenadier fish sticks	312.26	g
Co-products		
Breadcrumbs	4.74	g

Table D.6. Detailed life cycle inventory for the industrial frying step (data per FU)

INPUTS FROM THE TECHNOSPHERE		
Products		
Patagonian grenadier fish sticks	312.26	g
Sunflower oil	11.21	g
Electricity		
Sprinkler	1,75E-3	kWh
Transport		
Sunflower oil (>16t lorry)	1.44E-2	tkm
OUTPUTS TO THE TECHNOSPHERE		
Products		
Patagonian grenadier fish sticks	323.47	g

Table D.7. Detailed life cycle inventory for the freezing step (data per FU)

INPUTS FROM THE TECHNOSPHERE		
Products		
Patagonian grenadier fish sticks	323.47	g
Electricity		
Freezing tunnel	1.94E-2	kWh
Frigorific compressor A	1.75E-2	kWh
Frigorific compressor B	1.14E-2	kWh
OUTPUTS TO THE TECHNOSPHERE		
Products		
Patagonian grenadier fish sticks	323.47	g

Table D.8. Detailed life cycle inventory for the packaging step (data per FU)

INPUTS FROM THE TECHNOSPHERE		
Products		
Patagonian grenadier fish sticks	323.47	g
Packaging materials		
Cardboard (box)	22.98	g
Polyethylene-PE (box)	1.17	g
Butt	0.77	g
Pallet separators	0.85	g
Pallet label	2.37	mg
Retractable polyolefin	1.65	g
Type C Bag	58.42	mg
Thermofusion glue	0.60	g
Blue seal	18.95	mg
Print boxes	0.96	g
Automatic stretch	0.21	g
Electricity		
Catering scale	1.18E-4	kWh
Retractable	5.24E-3	kWh
Sealing machine	1.05E-3	kWh

Table D.8. Detailed life cycle inventory for the packaging step (data per FU) (continuation)

INPUTS FROM THE TECHNOSPHERE		
Electricity		
Moulding machine	6.99E-4	kWh
Conveyor belt	4.66E-4	kWh
Palletizing robots	1.75E-4	kWh
Packaging robots	6.99E-3	kWh
Compressed air machine A	6.12E-3	kWh
Compressed air machine B	2.56E-3	kWh
Compressed air machine C	3.67E-3	kWh
OUTPUTS TO THE TECHNOSPHERE		
Products		
Patagonian grenadier fish sticks	323.47	g
Packaging	25.97	g
Residues		
Cardboard (box)	2.48	g
Plastics (box)	0.13	g
Retractable	0.30	g
Dividers	0.15	g
Butt	0.19	g
Automatic stretcher	52.11	mg

Table D.9. Detailed life cycle inventory regarding ancillary operations and activities in the processing plant (data per FU)

INPUTS FROM THE TECHNOSPHERE		
Materials		
NH_3	1.31E-2	mg
Detergents	0.48	mg
Bleach	0.35	mg
Caustic soda	49.11	mg
Water	1.03	g
Lubricants	3.11E-3	mL
Electricity		
Air conditioning	1.64E-2	kWh
Illumination	7.40E-3	kWh
Cold chambers	8.64E-2	kWh
Hydraulic consumption	9.47E-3	kWh
Wastewater treatment	1.05E-2	kWh

Table D.10. Characterization values for Cumulative Energy Demand of fish stick production (data per FU).

Cumulative energy demand (CED) (KJ/FU)		
Energy type	*Value*	*Unit*
Renewable, water	146.06	KJ
Renewable, wind, solar, geothermal	137.07	KJ
Renewable, biomass	1197.26	KJ
Non-renewable, biomass	0.65	KJ
Non-renewable, nuclear	948.30	KJ
Non-renewable, fossil	7576.80	KJ
Total	10,006.13	KJ

Section E presents data regarding fish sticks distribution and wholesaling for consumption assessment.

Table E.1. Life cycle inventory data for the distribution step (wholesaling and retailing)

INPUTS FROM THE TECHNOSPHERE		
Products		
Patagonian grenadier fish sticks	323.47	g
Packaging	25.97	g
Electricity		
Cold chamber (distribution centre)	8.63E-02	kWh
Freezer (wholesale and retail)	1.93E-02	kWh
Transport		
Patagonian grenadier fish sticks	3.77E-01	tkm
OUTPUTS TO THE TECHNOSPHERE		
Products		
Patagonian grenadier fish sticks	323.47	g
Packaging	21.55	g
Residues		
Cardboard (dividers)	1.28	g
Plastics	1.51	g

Section F shows detailed inventory and characterization data for sardines supply chain.

Table F.1. Inventory data for Subsystem VI: human excretion. Data reported per FU

	Unit	Fried sardines (using olive oil and flour)	Grilled sardines	Canned sardines	Fried hake	Boiled hake
Inputs from the technosphere						
Human excretion						
Fried sardine	g	93.30	--	--	--	--
Grilled sardine	g	--	85.00	--	--	--
Canned sardine	g	--	--	85.00	--	--
Fried hake	g	--	--	--	104.00	--
Boiled hake	g	--	--	--	--	97.00
Tissue paper	mg	746.3	682.9	653.9	835.5	768.9
Water	L	2.35	2.15	2.06	2.63	2.42
Soap	mg	621.9	569.1	544.9	696.3	640.8
Detergent	mg	34.98	32.01	30.65	39.17	36.04
Electricity[1]	kWh	2.22E-3	2.03E-3	1.94E-3	2.48E-3	2.28E-3
Wastewater treatment						
Sewer grid infrastructure	mg	5.20E-4	4.80E-4	4.60E-4	5.70E-4	5.40E-4
Wastewater treatment plant infrastructure	mg	1.40E-5	1.30E-5	1.20E-5	1.50E-5	1.40E-5
Ferric chloride	mg	0.66	0.60	0.57	0.72	0.68
Ferrous sulfate	mg	0.48	0.44	0.42	0.52	0.49
Aluminium sulfate	mg	0.13	0.12	0.11	0.14	0.13
Electricity[1]	kWh/m^3	3.40E-3	3.10E-3	3.00E-3	3.10E-3	3.00E-3
Slurry spreading						
Truck	kgkm	0.29	0.25	0.24	0.24	0.21
Slurry spreading	cm^3	13.93	12.20	11.48	11.46	10.44

Table F.1. Inventory data for Subsystem VI: human excretion. Data reported per FU (continuation)

	Unit	Fried sardines (using olive oil and flour)	Grilled sardines	Canned sardines	Fried hake	Boiled hake
		Outputs to the technosphere				
Outputs from wastewater treatment						
Toilet paper	g	0.75	0.68	0.65	0.82	0.77
Sludge to other disposal routes	g 37% dry mass	0.65	0.57	0.54	0.53	0.49
Emissions to air from human excretion						
Carbon dioxide	g	84.31	66.48	57.43	53.88	42.22
Methane	mg	26.53	21.13	18.42	17.40	13.89
Emissions to air from wastewater treatment						
Carbon dioxide	g	2.20	1.91	1.78	1.77	1.59
Nitrous oxide	mg	1.77	1.70	1.70	1.77	1.75
Emissions to air from biogas incineration						
Carbon dioxide	g	3.56	3.09	2.88	2.87	2.58
Carbon monoxide	mg	5.75	4.99	4.65	4.63	4.17
NMVOC	mg	8.58E-2	7.44E-2	6.95E-2	6.90E-2	6.22E-2
Methane	mg	18.88	16.37	15.28	15.19	13.68
Nitrogen dioxide	mg	31.51	30.22	30.21	31.43	31.04
Ammonia	mg	3.50	3.36	3.36	3.49	3.45
Nitrous oxide	mg	2.33	2.24	2.24	2.33	2.30
Emissions to air from sludge application to soil						
Ammonia	mg	28.11	26.95	26.95	28.03	27.68
Nitrous oxide	mg	3.33	3.19	3.19	3.32	3.28
Carbon dioxide	g	1.75	1.52	1.42	1.41	1.27
Emissions to water from wastewater treatment						
N-total	g	2.18	2.09	2.09	2.17	2.14
TOC	g	0.28	0.24	0.23	0.22	0.20
BOD	g	0.37	0.32	0.30	0.30	0.27

Table F.1. Inventory data for Subsystem VI: human excretion. Data reported per FU (continuation)

	Unit	Fried sardines (using olive oil and flour)	Grilled sardines	Canned sardines	Fried hake	Boiled hake
Outputs to the technosphere						
Emissions to water from wastewater treatment						
COD	mg	1.03	0.90	0.84	0.84	0.76
SO_4^{2-}	g	1.24	1.19	1.19	1.23	1.22
P-total	mg	0.55	0.50	0.48	0.60	0.57
Chlorine (Cl)	mg	0.43	0.39	0.38	0.47	0.44
Emissions to water from sludge application to soil						
PO_4^{3-} to groundwater	mg	9.15E-4	8.37E-4	8.01E-4	1.00E-3	9.42E-4
PO_4^{3-} to river water	mg	3.22E-3	2.94E-3	2.82E-3	3.51E-3	3.31E-3
Emissions to soil from sludge application to soil						
TOC	mg	41.62	36.10	33.70	33.49	30.16
Aluminium (Al)	mg	2.05E-2	1.87E-2	1.79E-2	2.24E-2	2.11E-2
Iron (Fe)	mg	0.40	0.37	0.35	0.44	0.41

[1] The assumed electricity grid was the average electricity mix for Spain in 2011.

Table F.2. Total environmental impacts per impact category for the selected scenarios (ReCiPe endpoint). Data reported per FU

Impact category	Unit	Scenario A			Scenario B		Scenario C	
		Economic revenue allocation	***Economic savings allocation***	***Energy allocation***	***Fried sardines***	***Sardines in a BBQ***	***Boiled hake***	***Fried hake***
CC [HH]	Pt	6.78E-2	6.23E-2	5.79E-2	9.97E-3	4.45E-3	3.37E-2	3.49E-2
OD	Pt	4.31E-3	4.31E-3	4.31E-3	1.03E-5	9.26E-6	3.58E-4	3.59E-4
HT	Pt	1.73E-2	1.55E-2	1.41E-2	1.81E-3	1.04E-3	2.42E-3	2.85E-3
POF	Pt	3.03E-5	2.97E-5	2.93E-5	2.08E-6	1.60E-6	1.24E-5	1.26E-5
PMF	Pt	5.85E-2	5.38E-2	5.01E-2	4.75E-3	3.38E-3	2.32E-2	2.38E-2
IR	Pt	1.91E-3	1.89E-3	1.88E-3	3.85E-5	4.18E-6	2.74E-5	3.56E-5
CC [Ec]	Pt	5.47E-2	4.99E-2	4.60E-2	8.68E-3	3.87E-3	2.94E-2	3.04E-2
TA	Pt	2.46E-3	2.44E-3	2.43E-3	4.49E-5	2.50E-5	2.07E-4	2.13E-4
FE	Pt	5.10E-4	4.97E-4	4.87E-4	4.05E-5	2.40E-6	1.08E-5	2.68E-5
TET	Pt	3.29E-4	3.17E-4	3.06E-4	-2.10E-4	1.58E-4	6.57E-5	-5.24E-4
FET	Pt	8.15E-5	7.05E-5	6.18E-5	1.66E-5	1.05E-6	4.23E-6	6.07E-6
MET	Pt	1.53E-5	1.53E-5	1.53E-5	1.12E-8	4.83E-9	1.36E-8	1.53E-8
ALO	Pt	8.72E-2	8.70E-2	8.69E-2	8.52E-4	2.74E-3	9.48E-4	4.43E-4
ULO	Pt	4.07E-3	3.90E-3	3.76E-3	1.29E-4	9.26E-5	2.38E-4	2.63E-4
NLT	Pt	1.04E-2	1.03E-2	1.02E-2	3.55E-3	3.11E-3	4.94E-3	5.24E-3
MD	Pt	2.62E-2	2.55E-2	2.49E-2	2.69E-5	3.47E-6	1.43E-5	2.61E-5
FD	Pt	2.46E-1	2.32E-1	2.20E-1	2.33E-2	1.04E-2	7.97E-2	8.19E-2
TOTAL	***Pt***	***5.82E-1***	***5.49E-1***	***5.24E-1***	***5.30E-2***	***2.93E-2***	***1.75E-1***	***1.80E-1***

CC [HH]= climate change –human health; CC [Ec]= climate change – ecosystems; OD= ozone depletion; HT= human toxicity; POF= photochemical oxidant formation; PMF= particulate matter formation; IR= ionizing radiation; TA= terrestrial acidification; FE= freshwater eutrophication; TE= terrestrial eco-toxicity; FE= freshwater eco-toxicity; ME= marine eco-toxicity; ALO= agricultural land occupation; ULO= urban land occupation; MD= metal depletion; FD= fossil depletion.
Pt= millipoints.

Table F.3. Total environmental impacts per life-cycle stage for Scenario A (ReCiPe midpoint). Results refer to the economic revenue allocation perspective. Data reported per FU

Impact category	Unit	SS1	SS2	SS3		SS4	SS5	SS6
		Sardine fishery	Port auction	Canning (without olive oil)	Canning (olive oil only)	Wholesaling and retailing	Consumption	Human Excretion
CC	kg CO_2 eq	1.81E-1	4.75E-3	2.60E0	2.44E-1	4.18E-2	2.79E-1	9.44E-3
OD	kg CFC-11 eq	2.80E-7	2.08E-10	3.05E-7	2.99E-8	2.58E-9	1.30E-7	3.68E-10
HT	kg 1,4-DB eq	2.22E-2	1.14E-3	1.46E0	1.55E-3	1.59E-2	2.05E-1	2.16E-3
POF	kg NMVOC	3.22E-3	2.30E-5	1.16E-2	6.27E-4	1.49E-4	2.14E-3	5.02E-5
PMF	kg PM10 eq	8.98E-4	8.78E-6	1.02E-2	2.14E-4	5.03E-5	6.29E-4	1.66E-5
IR	kg U235 eq	6.91E-3	1.43E-3	5.99E-1	1.64E-1	6.74E-3	1.37E-1	2.35E-3
TA	kg SO_2 eq	2.35E-3	2.60E-5	1.34E-2	1.57E-3	1.35E-4	2.28E-3	4.05E-5
FE	kg P eq	9.63E-6	1.34E-6	1.12E-3	5.58E-4	6.87E-6	6.65E-4	2.37E-6
ME	kg N eq	1.19E-3	9.25E-7	3.72E-3	9.63E-4	1.10E-5	1.35E-2	1.99E-5
TET	kg 1,4-DB eq	3.18E-5	8.88E-7	1.71E-2	5.87E-2	3.67E-6	9.13E-5	4.44E-5
FET	kg 1,4-DB eq	3.03E-4	2.41E-5	1.31E-1	5.78E-3	5.95E-4	3.58E-3	5.68E-5
MET	kg 1,4-DB eq	1.84E-3	2.41E-5	1.34E-1	6.19E-2	5.65E-4	3.65E-3	4.47E-5
ALO	m^2a	7.76E-4	1.82E-2	3.94E0	5.26E-3	2.21E-4	-3.84E-2	7.68E-3
ULO	m^2a	3.17E-4	2.16E-4	6.92E-2	2.16E-3	4.06E-3	3.26E-3	1.69E-4
NLT	m^2	7.56E-5	2.20E-6	5.78E-3	5.08E-5	5.90E-6	-1.40E-6	1.34E-5
WD	m^3	7.42E-4	2.79E-5	6.21E-2	4.18E-1	1.43E-4	5.03E-3	2.50E-3
MD	kg Fe eq	1.68E-3	2.34E-4	6.01E0	4.83E-2	1.24E-3	3.23E-2	2.75E-4
FD	kg oil eq	5.92E-3	2.22E-3	1.17E0	5.31E-2	1.24E-2	5.92E-2	1.39E-3

CC= climate change; OD= ozone depletion; HT= human toxicity; POF= photochemical oxidant formation; PMF= particulate matter formation; IR= ionizing radiation; TA= terrestrial acidification; FE= freshwater eutrophication; ME= marine eutrophication; TE= terrestrial eco-toxicity; FE= freshwater eco-toxicity; ME= marine eco-toxicity; ALO= agricultural land occupation; ULO= urban land occupation; WD= water depletion; MD= metal depletion; FD= fossil depletion.

Table F.4. Total environmental impacts per life-cycle stage for Scenario A (ReCiPe endpoint). Results refer to the economic revenue allocation perspective. Data reported per FU

Impact category	Unit	SS1	SS2	SS3		SS4	SS5	SS6
		Sardine fishery	**Port auction**	**Canning (without olive oil)**	**Canning (only olive oil)**	**Wholesaling and retailing**	**Consumption**	**Human Excretion**
CC [HH]	Pt	3.77E-3	9.89E-5	5.23E-2	5.08E-3	4.79E-4	5.81E-3	1.97E-4
OD	Pt	1.49E-5	7.93E-9	1.19E-5	4.27E-3	9.50E-8	5.18E-6	1.41E-8
HT	Pt	2.32E-4	1.19E-5	1.48E-2	1.25E-6	4.09E-5	2.13E-3	2.25E-5
POF	Pt	1.87E-6	1.34E-8	7.22E-6	1.98E-5	7.18E-8	1.24E-6	2.91E-8
PMF	Pt	3.48E-3	3.40E-5	5.23E-2	6.15E-5	1.34E-4	2.43E-3	6.41E-5
IR	Pt	1.69E-6	3.49E-7	1.64E-4	1.71E-3	7.71E-7	3.35E-5	5.75E-7
CC [Ec]	Pt	3.29E-3	8.62E-5	4.56E-2	9.08E-7	4.18E-4	5.06E-3	1.72E-4
TA	Pt	3.13E-5	3.45E-7	2.42E-4	2.15E-3	1.06E-6	3.03E-5	5.39E-7
FE	Pt	9.71E-7	1.35E-7	1.21E-4	3.21E-4	3.14E-7	6.70E-5	2.40E-7
TET	Pt	9.28E-6	2.58E-7	1.68E-4	1.12E-4	7.76E-7	2.66E-5	1.30E-5
FET	Pt	1.80E-7	1.43E-8	7.68E-5	2.25E-6	4.70E-8	2.13E-6	3.39E-8
MET	Pt	3.37E-9	4.41E-11	2.46E-7	1.51E-5	1.62E-10	6.70E-9	8.20E-11
ALO	Pt	1.99E-5	4.68E-4	8.73E-2	1.40E-4	1.93E-6	-9.92E-4	2.10E-4
ULO	Pt	1.40E-5	9.54E-6	3.77E-3	9.87E-5	1.87E-5	1.44E-4	7.49E-6
NLT	Pt	2.44E-4	8.55E-6	7.31E-3	3.82E-4	2.03E-5	-3.13E-4	2.76E-3
MD	Pt	1.35E-6	1.87E-7	2.28E-2	3.36E-3	1.08E-6	2.59E-5	2.20E-7
FD	Pt	1.06E-2	3.99E-4	2.13E-1	8.56E-3	2.07E-3	1.06E-2	2.51E-4
TOTAL	***Pt***	***2.18E-2***	***1.12E-3***	***5.01E-1***	***2.63E-2***	***3.19E-3***	***2.51E-2***	***3.70E-3***

CC [HH]= climate change –human health; CC [Ec]= climate change – ecosystems; OD= ozone depletion; HT= human toxicity; POF= photochemical oxidant formation; PMF= particulate matter formation; IR= ionizing radiation; TA= terrestrial acidification; FE= freshwater eutrophication; TE= terrestrial eco-toxicity; FE= freshwater eco-toxicity; ME= marine eco-toxicity; ALO= agricultural land occupation; ULO= urban land occupation; MD= metal depletion; FD= fossil depletion.

Table F.5. Total environmental impacts per life-cycle stage for Scenario B (ReCiPe midpoint). Results refer to the consumption of fried sardines. Data reported per FU

		SS1	SS2	SS4	SS5	SS6
Impact category	**Unit**	**Sardine fishery**	**Port auction**	**Wholesaling and retailing**	**Consumption of fried sardine**	**Human excretion**
CC	kg CO_2 eq	1.10E-1	2.88E-3	3.80E-2	3.10E-1	1.09E-2
OD	kg CFC-11 eq	1.69E-7	1.26E-10	1.98E-9	2.92E-8	4.21E-10
HT	kg 1,4-DB eq	1.35E-2	6.94E-4	1.55E-2	1.40E-1	2.47E-3
POF	kg NMVOC	1.95E-3	1.39E-5	1.32E-4	1.42E-3	5.44E-5
PMF	kg PM10 eq	5.44E-4	5.32E-6	4.53E-5	6.03E-4	1.83E-5
IR	kg U235 eq	4.18E-3	8.67E-4	6.33E-3	1.43E-1	2.68E-3
TA	kg SO_2 eq	1.42E-3	1.58E-5	1.24E-4	1.75E-3	4.46E-5
FE	kg P eq	5.83E-6	8.12E-7	6.55E-6	3.84E-4	2.71E-6
ME	kg N eq	7.18E-4	5.60E-7	5.93E-6	2.13E-3	2.16E-5
TET	kg 1,4-DB eq	1.93E-5	5.38E-7	2.99E-6	-7.94E-4	5.07E-5
FET	kg 1,4-DB eq	1.83E-4	1.46E-5	5.86E-4	2.69E-2	6.51E-5
MET	kg 1,4-DB eq	1.11E-3	1.46E-5	5.53E-4	4.32E-3	5.13E-5
ALO	m^2a	4.69E-4	1.10E-2	2.07E-4	9.67E-3	8.77E-3
ULO	m^2a	1.92E-4	1.31E-4	3.48E-4	1.91E-3	1.94E-4
NLT	m^2	4.57E-5	1.33E-6	4.52E-6	4.29E-5	1.52E-5
WD	m^3	4.49E-4	1.69E-5	1.29E-4	1.82E-1	2.86E-3
MD	kg Fe eq	1.02E-3	1.42E-4	1.09E-3	3.07E-2	3.16E-4
FD	kg oil eq	3.58E-2	1.34E-3	1.05E-2	7.79E-2	1.59E-3

CC= climate change; OD= ozone depletion; HT= human toxicity; POF= photochemical oxidant formation; PMF= particulate matter formation; IR= ionizing radiation; TA= terrestrial acidification; FE= freshwater eutrophication; ME= marine eutrophication; TE= terrestrial eco-toxicity; FE= freshwater eco-toxicity; ME= marine eco-toxicity; ALO= agricultural land occupation; ULO= urban land occupation; WD= water depletion; MD= metal depletion; FD= fossil depletion.

Table F.6. Total environmental impacts per life-cycle stage for Scenario B (ReCiPe midpoint). Results refer to the consumption of sardines in a barbeque. Data reported per FU

Impact category	Unit	SS1 Sardine fishery	SS2 Port auction	SS4 Wholesaling and retailing	SS5 Consumption of sardines BBQ	SS6 Human excretion
CC	kg CO_2 eq	1.10E-1	2.88E-3	3.80E-2	4.60E-2	9.86E-3
OD	kg CFC-11 eq	1.69E-7	1.26E-10	1.98E-9	2.82E-9	3.85E-10
HT	kg 1,4-DB eq	1.35E-2	6.94E-4	1.55E-2	6.70E-2	2.26E-3
POF	kg NMVOC	1.95E-3	1.39E-5	1.32E-4	5.89E-4	5.11E-5
PMF	kg PM10 eq	5.44E-4	5.32E-6	4.53E-5	2.50E-4	1.70E-5
IR	kg U235 eq	4.18E-3	8.67E-4	6.33E-3	2.33E-3	2.45E-3
TA	kg SO_2 eq	1.42E-3	1.58E-5	1.24E-4	2.60E-4	4.15E-5
FE	kg P eq	5.83E-6	8.12E-7	6.55E-6	7.40E-6	2.48E-6
ME	kg N eq	7.18E-4	5.60E-7	5.93E-6	1.62E-4	2.03E-5
TET	kg 1,4-DB eq	1.93E-5	5.38E-7	2.99E-6	4.73E-4	4.64E-5
FET	kg 1,4-DB eq	1.83E-4	1.46E-5	5.86E-4	9.06E-4	5.95E-5
MET	kg 1,4-DB eq	1.11E-3	1.46E-5	5.53E-4	8.81E-4	4.69E-5
ALO	m^2a	4.69E-4	1.10E-2	2.07E-4	8.65E-2	8.02E-3
ULO	m^2a	1.92E-4	1.31E-4	3.48E-4	1.10E-3	1.77E-4
NLT	m^2	4.57E-5	1.33E-6	4.52E-6	1.28E-5	1.39E-5
WD	m^3	4.49E-4	1.69E-5	1.29E-4	1.32E-4	2.61E-3
MD	kg Fe eq	1.02E-3	1.42E-4	1.09E-3	1.45E-3	2.91E-4
FD	kg oil eq	3.58E-2	1.34E-3	1.05E-2	6.28E-3	1.45E-3

CC= climate change; OD= ozone depletion; HT= human toxicity; POF= photochemical oxidant formation; PMF= particulate matter formation; IR= ionizing radiation; TA= terrestrial acidification; FE= freshwater eutrophication; ME= marine eutrophication; TE= terrestrial eco-toxicity; FE= freshwater eco-toxicity; ME= marine eco-toxicity; ALO= agricultural land occupation; ULO= urban land occupation; WD= water depletion; MD= metal depletion; FD= fossil depletion.

Table F.7. Total environmental impacts per life-cycle stage for Scenario B (ReCiPe endpoint). Results refer to the consumption of fried sardines. Data reported per FU

Impact category	Unit	SS1	SS2	SS4	SS5	SS6	TOTAL
		Sardine fishery	Port auction	Wholesaling and retailing	Consumption of fried sardines	Human Excretion	
CC [HH]	Pt	2.29E-3	5.99E-5	9.36E-4	6.45E-3	2.26E-4	9.97E-3
OD	Pt	9.01E-6	4.80E-9	1.18E-7	1.17E-6	1.61E-8	1.03E-5
HT	Pt	1.41E-4	7.23E-9	1.72E-4	1.46E-3	2.57E-5	1.81E-3
POF	Pt	1.13E-6	8.09E-9	9.34E-8	8.22E-7	3.16E-8	2.08E-6
PMF	Pt	2.11E-3	2.06E-5	2.12E-4	2.33E-3	7.07E-5	4.74E-3
IR	Pt	1.02E-6	2.11E-7	1.79E-6	3.49E-5	6.55E-7	3.85E-5
CC [Ec]	Pt	1.99E-3	5.23E-5	8.16E-4	5.63E-3	1.97E-4	8.68E-3
TA	Pt	1.89E-5	2.09E-7	1.91E-6	2.32E-5	5.93E-7	4.49E-5
FE	Pt	5.88E-7	8.17E-8	7.33E-7	3.88E-5	2.73E-7	4.05E-5
TET	Pt	5.62E-6	1.56E-7	1.35E-6	-2.33E-4	1.48E-5	-2.11E-4
FET	Pt	1.09E-7	8.69E-9	3.61E-6	1.60E-5	3.88E-8	1.66E-5
MET	Pt	2.04E-9	2.67E-11	1.06E-9	7.93E-9	9.40E-11	1.12E-8
ALO	Pt	1.21E-5	2.84E-4	6.24E-6	3.10E-4	2.40E-4	8.52E-4
ULO	Pt	8.49E-6	5.78E-6	2.19E-5	8.45E-5	8.56E-6	1.29E-4
NLT	Pt	1.48E-4	5.18E-6	2.38E-5	2.20E-4	3.15E-3	3.55E-3
MD	Pt	8.16E-7	1.13E-7	1.15E-6	2.46E-5	2.53E-7	2.69E-5
FD	Pt	6.45E-3	2.42E-4	2.34E-3	1.40E-2	2.86E-4	2.33E-2
TOTAL	***Pt***	***1.32E-2***	***6.78E-4***	***4.53E-3***	***3.04E-2***	***4.22E-3***	***5.30E-2***

CC [HH]= climate change –human health; CC [Ec]= climate change – ecosystems; OD= ozone depletion; HT= human toxicity; POF= photochemical oxidant formation; PMF= particulate matter formation; IR= ionizing radiation; TA= terrestrial acidification; FE= freshwater eutrophication; TE= terrestrial eco-toxicity; FE= freshwater eco-toxicity; ME= marine eco-toxicity; ALO= agricultural land occupation; ULO= urban land occupation; MD= metal depletion; FD= fossil depletion.

Table F.8. Total environmental impacts per life-cycle stage for Scenario B (ReCiPe endpoint). Results refer to the consumption of sardines in a barbeque. Data reported per FU

Impact category	Unit	SS1	SS2	SS4	SS5	SS6	TOTAL
		Sardine fishery	**Port auction**	**Wholesaling and retailing**	**Consumption of sardines (BBQ)**	**Human Excretion**	
CC [HH]	Pt	2.29E-3	5.99E-5	9.36E-4	9.58E-4	2.06E-4	4.45E-3
OD	Pt	9.01E-6	4.80E-9	1.18E-7	1.08E-7	1.48E-8	9.26E-6
HT	Pt	1.41E-4	7.23E-9	1.72E-4	6.97E-4	2.35E-5	1.04E-3
POF	Pt	1.13E-6	8.09E-9	9.34E-8	3.42E-7	2.97E-8	1.60E-6
PMF	Pt	2.11E-3	2.06E-5	2.12E-4	9.67E-4	6.58E-5	3.38E-3
IR	Pt	1.02E-6	2.11E-7	1.79E-6	5.69E-7	5.98E-7	4.18E-6
CC [Ec]	Pt	1.99E-3	5.23E-5	8.16E-4	8.35E-4	1.79E-4	3.87E-3
TA	Pt	1.89E-5	2.09E-7	1.91E-6	3.46E-6	5.52E-7	2.50E-5
FE	Pt	5.88E-7	8.17E-8	7.33E-7	7.48E-7	2.50E-7	2.40E-6
TET	Pt	5.62E-6	1.56E-7	1.35E-6	1.37E-4	1.36E-5	1.58E-4
FET	Pt	1.09E-7	8.69E-9	3.61E-6	5.40E-7	3.55E-8	1.05E-6
MET	Pt	2.04E-9	2.67E-11	1.06E-9	1.62E-9	8.59E-11	4.83E-9
ALO	Pt	1.21E-5	2.84E-4	6.24E-6	2.22E-3	2.20E-4	2.74E-3
ULO	Pt	8.49E-6	5.78E-6	2.19E-5	4.86E-5	7.84E-6	9.26E-5
NLT	Pt	1.48E-4	5.18E-6	2.38E-5	4.77E-5	2.88E-3	3.11E-3
MD	Pt	8.16E-7	1.13E-7	1.15E-6	1.16E-6	2.33E-7	3.47E-6
FD	Pt	6.45E-3	2.42E-4	2.34E-3	1.13E-3	2.62E-4	1.04E-2
TOTAL	***Pt***	***1.32E-2***	***6.78E-4***	***4.53E-3***	***7.05E-3***	***3.86E-3***	***2.93E-2***

CC [HH]= climate change –human health; CC [Ec]= climate change – ecosystems; OD= ozone depletion; HT= human toxicity; POF= photochemical oxidant formation; PMF= particulate matter formation; IR= ionizing radiation; TA= terrestrial acidification; FE= freshwater eutrophication; TE= terrestrial eco-toxicity; FE= freshwater eco-toxicity; ME= marine eco-toxicity; ALO= agricultural land occupation; ULO= urban land occupation; MD= metal depletion; FD= fossil depletion.

Table F.9. Total environmental impacts per life-cycle stage for Scenario C (ReCiPe midpoint). Results refer to the consumption of fried hake. Data reported per FU

Impact category	Unit	SS1	SS2	SS3	SS4	SS5	SS6
		Sardine fishery	**Port auction**	**Hake fishery**	**Wholesaling and retailing**	**Consumption of fried hake**	**Human Excretion**
CC	kg CO_2 eq	6.91E-2	5.42E-3	1.13E0	5.24E-2	1.62E-1	1.93E-2
OD	kg CFC-11 eq	9.34E-8	2.37E-10	6.54E-6	2.90E-9	1.27E-8	4.47E-10
HT	kg 1,4-DB eq	8.24E-3	1.31E-3	6.52E-2	2.10E-2	8.15E-2	2.69E-3
POF	kg NMVOC	1.11E-3	2.63E-5	1.89E-2	1.83E-4	6.88E-4	2.66E-5
PMF	kg PM10 eq	3.10E-4	1.00E-5	5.18E-3	6.28E-5	2.95E-4	1.27E-5
IR	kg U235 eq	2.88E-3	1.63E-3	2.33E-2	8.69E-3	6.76E-2	2.81E-3
TA	kg SO_2 eq	8.13E-4	2.97E-5	1.34E-2	1.70E-4	8.33E-4	2.99E-5
FE	kg P eq	4.26E-6	1.53E-6	3.41E-5	8.92E-6	1.75E-4	2.95E-6
ME	kg N eq	4.73E-5	1.06E-6	7.07E-4	9.97E-6	2.75E-3	1.10E-5
TET	kg 1,4-DB eq	1.15E-5	1.01E-6	1.25E-4	4.39E-6	-2.01E-3	5.67E-5
FET	kg 1,4-DB eq	1.22E-4	2.75E-5	1.01E-3	7.88E-4	4.66E-3	7.11E-5
MET	kg 1,4-DB eq	6.32E-4	2.75E-5	4.58E-2	7.45E-4	2.42E-3	5.56E-5
ALO	m^2a	2.88E-3	2.08E-2	8.17E-4	2.84E-4	-1.10E-2	9.81E-3
ULO	m^2a	2.11E-4	2.46E-4	1.59E-3	5.00E-4	9.59E-4	2.15E-4
NLT	m^2	2.67E-5	2.51E-6	4.45E-4	6.63E-6	1.51E-5	1.70E-5
WD	m^3	2.93E-4	3.18E-5	1.27E-3	1.78E-4	2.15E-2	3.19E-3
MD	kg Fe eq	7.59E-4	2.67E-4	6.75E-3	1.53E-3	1.61E-2	3.44E-4
FD	kg oil eq	2.49E-2	2.53E-3	3.13E-1	1.46E-2	3.36E-2	1.69E-3

CC= climate change; OD= ozone depletion; HT= human toxicity; POF= photochemical oxidant formation; PMF= particulate matter formation; IR= ionizing radiation; TA= terrestrial acidification; FE= freshwater eutrophication; ME= marine eutrophication; TE= terrestrial eco-toxicity; FE= freshwater eco-toxicity; ME= marine eco-toxicity; ALO= agricultural land occupation; ULO= urban land occupation; WD= water depletion; MD= metal depletion; FD= fossil depletion.

Table F.10. Total environmental impacts per life-cycle stage for Scenario C (ReCiPe midpoint). Results refer to the consumption of boiled hake. Data reported per FU

Impact category	Unit	SS1	SS2	SS3	SS4	SS5	SS6
		Sardine fishery	Port auction	Hake fishery	Wholesaling and retailing	Consumption of boiled hake	Human Excretion
CC	kg CO_2 eq	6.91E-2	5.42E-3	1.13E0	5.24E-2	1.04E-1	1.85E-2
OD	kg CFC-11 eq	9.34E-8	2.37E-10	6.54E-6	2.90E-9	6.73E-9	4.16E-10
HT	kg 1,4-DB eq	8.24E-3	1.31E-3	6.52E-2	2.10E-2	4.05E-2	2.49E-3
POF	kg NMVOC	1.11E-3	2.63E-5	1.89E-2	1.83E-4	3.97E-4	2.48E-5
PMF	kg PM10 eq	3.10E-4	1.00E-5	5.18E-3	6.28E-5	1.41E-4	1.18E-5
IR	kg U235 eq	2.88E-3	1.63E-3	2.33E-2	8.69E-3	3.41E-2	2.62E-3
TA	kg SO_2 eq	8.13E-4	2.97E-5	1.34E-2	1.70E-4	3.76E-4	2.79E-5
FE	kg P eq	4.26E-6	1.53E-6	3.41E-5	8.92E-6	1.58E-5	2.74E-6
ME	kg N eq	4.73E-5	1.06E-6	7.07E-4	9.97E-6	1.38E-4	1.02E-5
TET	kg 1,4-DB eq	1.15E-5	1.01E-6	1.25E-4	4.39E-6	1.09E-5	5.22E-5
FET	kg 1,4-DB eq	1.22E-4	2.75E-5	1.01E-3	7.88E-4	1.57E-3	6.60E-5
MET	kg 1,4-DB eq	6.32E-4	2.75E-5	4.58E-2	7.45E-4	1.47E-3	5.17E-5
ALO	m^2a	2.88E-3	2.08E-2	8.17E-4	2.84E-4	1.16E-3	9.03E-3
ULO	m^2a	2.11E-4	2.46E-4	1.59E-3	5.00E-4	4.07E-4	1.98E-4
NLT	m^2	2.67E-5	2.51E-6	4.45E-4	6.63E-6	1.16E-5	1.57E-5
WD	m^3	2.93E-4	3.18E-5	1.27E-3	1.78E-4	6.89E-4	2.94E-3
MD	kg Fe eq	7.59E-4	2.67E-4	6.75E-3	1.53E-3	1.35E-3	3.18E-4
FD	kg oil eq	2.49E-2	2.53E-3	3.13E-1	1.46E-2	2.11E-2	1.57E-3

CC= climate change; OD= ozone depletion; HT= human toxicity; POF= photochemical oxidant formation; PMF= particulate matter formation; IR= ionizing radiation; TA= terrestrial acidification; FE= freshwater eutrophication; ME= marine eutrophication; TET= terrestrial eco-toxicity; FET= freshwater eco-toxicity; MET= marine eco-toxicity; ALO= agricultural land occupation; ULO= urban land occupation; WD= water depletion; MD= metal depletion; FD= fossil depletion.

Table F.11. Total environmental impacts per life-cycle stage for Scenario C – consumption of fried hake (ReCiPe endpoint). Data reported per FU

Impact category	Unit	SS1	SS2	SS3	SS4	SS5	SS6	TOTAL
		Sardine fishery	Port auction	Hake fishery	Wholesaling and retailing	Consumption of fried hake	Human Excretion	
CC [HH]	Pt	1.44E-3	1.13E-4	2.36E-2	5.97E-3	3.37E-3	4.02E-4	3.49E-2
OD	Pt	4.96E-6	9.05E-9	3.53E-4	6.15E-7	5.09E-7	1.72E-8	3.59E-4
HT	Pt	8.58E-5	1.36E-5	6.79E-4	1.19E-3	8.48E-4	2.80E-5	2.85E-3
POF	Pt	6.46E-7	1.52E-8	1.10E-5	5.82E-7	3.99E-7	1.55E-8	1.26E-5
PMF	Pt	1.20E-3	3.88E-5	2.00E-2	1.33E-3	1.14E-3	4.90E-5	2.38E-2
IR	Pt	7.04E-7	3.98E-7	5.70E-6	1.16E-5	1.65E-5	6.85E-7	3.56E-5
CC [Ec]	Pt	1.25E-3	9.84E-5	2.06E-2	5.20E-3	2.94E-3	3.51E-4	3.04E-2
TA	Pt	1.08E-5	3.94E-7	1.78E-4	1.24E-5	1.11E-5	3.98E-7	2.13E-4
FE	Pt	4.29E-7	1.54E-7	3.43E-6	4.91E-6	1.76E-5	2.97E-7	2.68E-5
TET	Pt	3.35E-6	2.95E-7	3.66E-5	6.99E-6	-5.87E-4	1.66E-5	-5.24E-4
FET	Pt	7.29E-8	1.64E-8	6.02E-7	2.56E-6	2.77E-6	4.24E-8	6.07E-6
MET	Pt	1.16E-9	5.04E-11	2.10E-9	7.46E-9	4.44E-9	1.02E-10	1.53E-8
ALO	Pt	7.55E-5	5.34E-4	2.11E-5	4.03E-5	-4.96E-4	2.69E-4	4.43E-4
ULO	Pt	9.33E-6	1.09E-5	7.03E-5	1.21E-4	4.24E-5	9.49E-6	2.63E-4
NLT	Pt	9.30E-5	9.76E-6	1.44E-3	1.22E-4	5.06E-5	3.53E-3	5.24E-3
MD	Pt	6.07E-7	2.13E-7	5.40E-6	6.72E-6	1.29E-5	2.75E-7	2.61E-5
FD	Pt	4.47E-3	4.55E-4	5.63E-2	1.44E-2	6.04E-3	3.03E-4	8.19E-2
TOTAL	***Pt***	***8.65E-3***	***1.27E-3***	***1.23E-1***	***2.84E-2***	***1.34E-2***	***4.95E-3***	***1.80E-1***

CC [HH] = climate change – human health; OD= ozone depletion; HT= human toxicity; POF= photochemical oxidant formation; PMF= particulate matter formation; IR= ionizing radiation; CC [Ec] = climate change – ecosystems; TA= terrestrial acidification; FE= freshwater eutrophication; TET= terrestrial eco-toxicity; FET= freshwater eco-toxicity; MET= marine eco-toxicity; ALO= agricultural land occupation; ULO= urban land occupation; MD= metal depletion; FD= fossil depletion.

Table F.12. Total environmental impacts per life-cycle stage for Scenario C – consumption of boiled hake (ReCiPe endpoint). Data reported per FU

Impact category	Unit	SS1	SS2	SS3	SS4	SS5	SS6	TOTAL
		Sardine fishery	**Port auction**	**Hake fishery**	**Wholesaling and retailing**	**Consumption of boiled hake**	**Human Excretion**	
CC [HH]	Pt	1.44E-3	1.13E-4	2.36E-2	5.97E-3	2.16E-3	3.85E-4	3.37E-2
OD	Pt	4.96E-6	9.05E-9	3.53E-4	6.15E-7	2.55E-7	1.60E-8	3.58E-4
HT	Pt	8.58E-5	1.36E-5	6.79E-4	1.19E-3	4.21E-4	2.59E-5	2.42E-3
POF	Pt	6.46E-7	1.52E-8	1.10E-5	5.82E-7	2.30E-7	1.44E-8	1.24E-5
PMF	Pt	1.20E-3	3.88E-5	2.00E-2	1.33E-3	5.44E-4	4.56E-5	2.32E-2
IR	Pt	7.04E-7	3.98E-7	5.70E-6	1.16E-5	8.33E-6	6.40E-7	2.74E-5
CC [Ec]	Pt	1.25E-3	9.84E-5	2.06E-2	5.20E-3	1.89E-3	3.36E-4	2.94E-2
TA	Pt	1.08E-5	3.94E-7	1.78E-4	1.24E-5	5.00E-6	3.71E-7	2.07E-4
FE	Pt	4.29E-7	1.54E-7	3.43E-6	4.91E-6	1.59E-5	2.77E-7	1.08E-5
TET	Pt	3.35E-6	2.95E-7	3.66E-5	6.99E-6	3.18E-6	1.53E-5	6.57E-5
FET	Pt	7.29E-8	1.64E-8	6.02E-7	2.56E-6	9.35E-7	3.93E-8	4.23E-6
MET	Pt	1.16E-9	5.04E-11	2.10E-9	7.46E-9	2.69E-9	9.48E-11	1.36E-8
ALO	Pt	7.55E-5	5.34E-4	2.11E-5	4.03E-5	2.97E-5	2.47E-4	9.48E-4
ULO	Pt	9.33E-6	1.09E-5	7.03E-5	1.21E-4	1.80E-5	8.75E-6	2.38E-4
NLT	Pt	9.30E-5	9.76E-6	1.44E-3	1.22E-4	3.69E-5	3.24E-3	4.94E-3
MD	Pt	6.07E-7	2.13E-7	5.40E-6	6.72E-6	1.08E-6	2.55E-7	1.43E-5
FD	Pt	4.47E-3	4.55E-4	5.63E-2	1.44E-2	3.79E-3	2.83E-4	7.97E-2
TOTAL	***Pt***	***8.65E-3***	***1.27E-3***	***1.23E-1***	***2.84E-2***	***8.91E-3***	***4.59E-3***	***1.75E-1***

CC [HH] = climate change – human health; OD= ozone depletion; HT= human toxicity; POF= photochemical oxidant formation; PMF= particulate matter formation; IR= ionizing radiation; CC [Ec] = climate change – ecosystems; TA= terrestrial acidification; FE= freshwater eutrophication; TET= terrestrial eco-toxicity; FET= freshwater eco-toxicity; MET= marine eco-toxicity; ALO= agricultural land occupation; ULO= urban land occupation; MD= metal depletion; FD= fossil depletion.

Resumo

Alcanzar a sostibilidade ambiental faise cada vez máis necesario para reducir o impacto ambiental asociado á actividade humana. Debido ao seu enfoque de ciclo de vida para avaliar o perfil ambiental de produtos, procesos e servizos, a Análise de Ciclo de Vida (ACV), unha metodoloxía desenvolvida baixo o paraugas das normas ISO 14040 e 14044, é de gran utilidade na avaliación do impacto ambiental asociado a un proceso dado. Por iso, é a ferramenta de xestión ambiental na que se apoian os estudos levados a cabo nesta tese de doutoramento.

Ademais do ACV, consideráronse outras metodoloxías complementarias para a avaliación do desempeño ambiental. A continuación detállanse as mesmas:

- A Análise por Envoltura de Datos (DEA polas súas siglas en inglés), permite a cuantificación da eficiencia produtiva de entidades similares mediante a comparación do desempeño operacional destas. Utilizouse en combinación co ACV para estimar a eficiencia operacional e os impactos ambientais potenciais.
- A Taxa de Retorno Enerxético en forma de proteína comestible (TRE_{prot}) é un indicador que se propón nesta tese como medida útil da eficiencia enerxética dos produtos alimentarios, ao conxugar o aporte enerxético dos alimentos e a enerxía investida para a obtención dos mesmos.
- Considerouse a metodoloxía proposta polo Panel Intergubernamental para o Cambio Climático (IPCC polas súas siglas en inglés) para a estimación de emisións de gases de efecto invernadoiro (GEIs) derivadas do uso da terra (UT) e cambios do uso da terra (CUT) aplicados ao sector vitícola de toda unha rexión.
- A Pegada de Carbono é unha metodoloxía que resulta de utilidade na comunicación da sostibilidade ambiental dos produtos en base á cuantificación da emisión de GEIs ao longo do seu ciclo de vida.

O obxectivo desta tese é analizar o perfil ambiental dos sectores vitivinícola —na Denominación de Orixe (DO) Ribeiro e DO Rías Baixas— e da pesca e industria de procesado de produtos mariños mediante a metodoloxía de ACV en combinación coas descritas anteriormente. Seleccionáronse estes sectores debido á súa importancia dentro do tecido industrial galego. Ademais, cabe salientar que mentres o sector pesqueiro e procesado de alimentos de produtos mariños xa fora analizado previamente noutros estudos de ACV, o sector do viño non fora estudado para esta rexión. Doutra banda, a cadea de abastecemento de produtos de orixe mariño non se tiña avaliado no seu conxunto, incluíndo as etapas finais de produto procesado e consumo final.

De acordo cos obxectivos propostos, a presente tese de doutoramento dividiuse nas seguintes seccións:

- Introdución aos sectores estudados e as ferramentas de xestión ambiental, en especial preséntase a metodoloxía de ACV (Sección I).
- Catro capítulos relacionados coa aplicación das metodoloxías de ACV, ACV+DEA e uso da terra no sector vitivinícola (Sección II).
- Cinco capítulos relacionados coa pesca e a cadea de abastecemento de peixe e outros produtos de orixe mariña analizados mediante as metodoloxías de ACV, Pegada de Carbono e TRE_{prot} (Sección III).
- Conclusións xerais (Sección IV, Capítulo 13).

Sección I: Introdución

Os Capítulos 1 e 2 desta tese contextualizan os sectores baixo estudo desde un enfoque a nivel mundial ata as particularidades do nivel rexional correspondente ao ámbito galego. Pola súa banda, o Capítulo 3 céntrase na xestión ambiental e na necesidade de alcanzar as metas da sostibilidade, sinalando o ACV como unha das metodoloxías de xestión ambiental con maior recoñecemento a nivel mundial. Para rematar, preséntase o estado da arte dos distintos estudos de ACV realizados nos sectores baixo estudo.

Sección II: Análise de Ciclo de Vida aplicado ao sector vitivinícola

Esta sección céntrase na aplicación de ACV e outras metodoloxías complementarias ao sector vitivinícola na DO Ribeiro e a DO Rías Baixas. O Capítulo 4 analiza o perfil ambiental da produción de viño —viticultura e vinificación— nunha explotación na DO Ribeiro para as colleitas 2007-2010. As principais conclusións derivadas deste estudo detállanse a continuación: i) os potenciais impactos ambientais atópanse asociados ás operacións agrícolas; ii) o uso de fertilizantes merece especial atención debido á súa contribución ao impacto ambiental; iii) entre as operacións de vinificación, a etapa de embotellado presenta un importante impacto potencial debido ao proceso de fabricación do vidro; e iv) o estudo de varias colleitas permitiu identificar as variacións do perfil ambiental para cada unha delas. Ademais, co obxectivo de reducir o impacto final da produción de viño, propuxéronse diferentes accións de mellora no devandito sistema produtivo.

O Capítulo 5 combina a metodoloxías de ACV+DEA analizando un total de 40 explotacións vitícolas na DO Rías Baixas. O obxectivo deste estudo é o de identificar a eficiencia operacional e ambiental de cada explotación. As principais conclusións deste estudo son as seguintes: i) unha elevada porcentaxe —60%— dos viticultores analizados operan de xeito eficiente; ii) atopáronse diferenzas en canto á eficiencia operacional entre os viticultores dependendo do tamaño das explotacións; iii) os resultados obtidos foron proxectados en aras de estimar a redución potencial dos principais consumos, traducíndose nunha importante redución do impacto ambiental; e iv) a redución dos consumos nas explotacións vitícolas levaría a un incremento dos beneficios de ata un 10%.

O Capítulo 6 analiza o perfil ambiental de varias explotacións da DO Ribeiro nos anos 2010 e 2011. As explotacións analizadas caracterízanse por utilizar diferentes técnicas de viticultura: i) biodinámica, que implica un xestión innovadora en canto aos insumos e as intervencións sobre a propia viña; ii) híbrido convencional-biodinámica, que utiliza técnicas propias da biodinámica combinadas con outras da viticultura convencional; e iii) convencional. As principais conclusións obtidas son: i) a viticultura en biodinámica presentou un menor impacto que a convencional; ii) a menor intervención sobre o viñedo foi o principal responsable destas reducións, destacando unha menor dependencia da mecanización das diferentes operacións; iv) a viticultura biodinámica mostrou un menor rendemento por hectárea e unha maior dependencia do traballo manual, o cal penaliza outro tipo de indicadores tales como a ocupación de solo.

O Capítulo 7 profunda na estimación das emisións de GEIs na DO Ribeiro no seu conxunto. Para iso combina as metodoloxías de ACV e de CUT para avaliar as emisións de GEIs durante o período 1990-2009 derivadas das operacións vitícolas e as diferentes dinámicas de CUT atopadas na rexión. Preséntanse as principais conclusións obtidas para este estudo: i) a emisión/fixación de GEIs derivadas de CUT foron relevantes en base aos usos do solo de partida; ii) identificáronse diferentes tendencias en canto a emisións de GEIs en base ao período estudado; iii) os cambios sociais e demográficos observados na área baixo estudo poden ser responsables das tendencias de CUT atopadas; e iv) este estudo foi o primeiro destas características que se levou para unha rexión concreta, e polo tanto, debido aos resultados obtidos, as emisións derivadas do CUT deberían terse en conta para futuros estudos de viño e ACV.

Sección III: Análise de Ciclo de Vida aplicado a produtos mariños

Esta sección analiza o perfil ambiental de diferentes produtos de orixe mariña e a pesca extractiva mediante o uso de ACV, TRE_{prot} e Pegada de Carbono. O Capítulo 8 analiza —dende unha perspectiva de ciclo de vida— a eficiencia enerxética das capturas da flota pesqueira galega introducindo o indicador TRE_{prot}. As principais conclusións obtidas deste estudo son as seguintes: i) o uso deste indicador resultou ser de gran utilidade para comunicar as eficiencias enerxéticas das capturas; ii) as capturas da flota galega no seu conxunto presentan un valor de TRE_{prot} de 7,6%; iii) atopáronse diferentes resultados para este indicador en función dos caladoiros onde opera a flota baixo estudo; iv) o retorno enerxético para produtos procesados foi menor debido á necesidade de incluír etapas que demandan enerxía dentro do proceso; e v) o tipo de asignación levada a cabo durante a etapa de inventario, así como as desviacións estándar das entradas ao sistema, resultaron ser unha fonte de incerteza que poderían influenciar o valor de TRE_{prot}.

O Capítulo 9 analiza a Pegada de Carbono para un produto multi-ingrediente en base a peixe — variñas de pescada rebozadas. Para iso, analizáronse as etapas de pesca do ingrediente principal (pescada) nos caladoiros do Cono Sur en Chile, o procesado a bordo das capturas, o transporte a Galicia e o procesado en planta para a elaboración do produto final. As principais conclusións deste estudo corresponde aos seguintes apartados: i) as operacións de pesca presentan contribucións importantes nas emisións de GEIs, destacando o consumo de diésel e os axentes refrixerantes; ii) a produción doutros ingredientes tales como trigo e aceite de xirasol son relevantes para as emisións de GEIs; e iii) a redución de emisións relacionada coas operacións de pesca centraríanse nun cambio no tipo de axente refrixerante utilizado, posto que unha actuación sobre a redución dos consumos de diésel non sería factible.

O Capítulo 10 analiza a Pegada de Carbono do produto analizado previamente, pero neste caso céntrase nas etapas posteriores dentro da cadea de abastecemento: distribución, venda e consumo, analizando as diferentes opcións de preparado propostas polo fabricante. As principais conclusións obtidas deste estudo son: i) o patrón de consumo deste tipo de produtos demostrou ser relevante na Pegada de Carbono final; ii) as decisións tomadas por parte do consumidor poden implicar unha importante redución final das emisións de GEIs; e iii) entre os diferentes escenarios analizados, o preparado en forno, pese a requirir un maior consumo eléctrico, implica unha menor Pegada de Carbono que as outras formas de preparado debido a que non precisa aceite para o seu preparado.

O Capítulo 11 analiza o perfil ambiental mediante ACV da cadea de fornezo de diferentes produtos derivados da sardiña. Para iso, realizouse un estudo cruzado de varios produtos tomando como base un mesmo aporte proteico, analizando: sardiña en lata, sardiña fresca e pescada capturada usando a sardiña como cebo. Téñense en conta as diferentes etapas dentro da cadea de fornezo de cada produto, desde as operacións de pesca (sardiña e pescada), descarga e poxa en porto, procesado (sardiña en lata) e consumo final (con diferentes opcións de preparado e consumo para cada un dos produtos). As principais conclusións obtidas deste estudo son: i) a forma na que o consumidor decide consumir un certo produto inflúe considerablemente no impacto do proceso; ii) a forma de preparado dun mesmo produto mostrou diferenzas significativas en canto a impacto potencial; iii) os produtos cun menor procesado presentaron menor impacto en comparación con aqueles máis elaborados; e iv) os principais responsables de impacto dos produtos analizados son aqueles relacionados cos procesos involucrados. Neste sentido, o consumo de diésel (especialmente para o caso da pesca de pescada debido ao dobre escenario de pesca —sardiña para cebo e pescada), embalaxe (lata) e o aceite de oliva destacaron pola súa contribución ao impacto final.

Finalmente, o Capítulo 12 analiza o perfil ambiental da obtención de alxinato de sodio a partir de algas pardas en Estados Unidos. Neste caso, incluíronse as etapas desde a extracción do alga do medio natural ata o seu posterior procesado para a extracción de alxinato. As principais conclusións deste estudo se compilan en dous aspectos fundamentais: i) destácase a etapa de extracción de alxinato en laboratorio como a etapa con maior impacto ambiental, fundamentalmente debido ao consumo eléctrico; ii) o consumo de diésel, ao igual que os sistemas de pesca extractiva, foi o principal responsable do impacto da etapa de recolección do alga no mar. Avaliáronse unha serie de accións de mellora co obxectivo de mellorar o perfil ambiental do proceso e a recuperación de etanol resultou ser clave para este tipo de procesos.

Sección IV: Conclusións

Para finalizar, a Sección IV mostra no Capítulo 13 as principais conclusións obtidas ao longo desta tese de doutoramento na que se comprobou que o Análise de Ciclo de Vida é unha ferramenta moi útil na avaliación do perfil ambiental dos sectores estudados.

Resumen

Alcanzar la sostenibilidad ambiental se hace cada vez más necesario para reducir el impacto ambiental asociado a la actividad humana. Debido a su enfoque de ciclo de vida para evaluar el perfil ambiental de productos, procesos y servicios, el Análisis de Ciclo de Vida (ACV), una metodología desarrollada bajo el paraguas de las normas ISO 14040 y 14044, es de gran utilidad en la evaluación del impacto ambiental asociado a un proceso dado. Por ello, es la herramienta de gestión ambiental en la que se apoyan los estudios llevados a cabo en esta tesis doctoral.

Además del ACV, se han considerado otras metodologías complementarias para la evaluación del desempeño ambiental. A continuación se detallan las mismas:

- El Análisis por Envoltura de Datos (DEA por sus siglas en inglés), permite la cuantificación de la eficiencia productiva de entidades similares mediante la comparación del desempeño operacional de estas. Se ha utilizado en combinación con el ACV para estimar la eficiencia operacional y los impactos ambientales potenciales.
- La Tasa de Retorno Energético en forma de proteína comestible (TRE_{prot}) es un indicador que se propone en esta tesis como medida útil de la eficiencia energética de los productos alimenticios, al conjugar el aporte energético de los alimentos y la energía invertida para la obtención de los mismos.
- Se ha considerado la metodología propuesta por el Panel Intergubernamental para el Cambio Climático (IPCC por sus siglas en inglés) para la estimación de emisiones de gases de efecto invernadero (GEIs) derivadas del uso de la tierra (UT) y cambios del uso de la tierra (CUT) aplicados al sector vitícola de toda una región.
- La Huella de Carbono es una metodología que resulta de utilidad en la comunicación de la sostenibilidad ambiental de los productos en base a la cuantificación de la emisión de GEIs a lo largo de su ciclo de vida.

El objetivo de esta tesis es analizar el perfil ambiental de los sectores vitivinícola —en la Denominación de Origen (DO) Ribeiro y DO Rías Baixas— y de la pesca e industria de procesado de productos marinos mediante la metodología de ACV en combinación con las descritas anteriormente. Se han seleccionado estos sectores debido a su importancia dentro del tejido industrial gallego. Además, cabe destacar que mientras el sector pesquero y procesado de alimentos de productos marinos ya había sido analizado previamente en otros estudios de ACV, el sector del vino no había sido estudiado previamente para esta región. Por otro lado, la cadena de suministro de productos de origen marino no se había evaluado en su conjunto, incluyendo las etapas finales de producto procesado y consumo final.

De acuerdo con los objetivos propuestos, la presente tesis doctoral se ha dividido en las siguientes secciones:

- Introducción a los sectores estudiados y las herramientas de gestión ambiental, en especial se presenta la metodología de ACV (Sección I).
- Cuatro capítulos relacionados con la aplicación de las metodologías de ACV, ACV+DEA y uso de la tierra en el sector vitivinícola (Sección II).
- Cinco capítulos relacionados con la pesca y la cadena de suministro de pescado y otros productos de origen marino analizados mediante las metodologías de ACV, Huella de Carbono y TRE_{prot} (Sección III).
- Conclusiones generales (Sección IV, Capítulo 13).

Sección I: Introducción

Los Capítulos 1 y 2 de esta tesis contextualizan los sectores bajo estudio desde un enfoque a nivel mundial hasta las particularidades del nivel regional correspondiente al ámbito gallego. Por su parte, el Capítulo 3 se centra en la gestión ambiental y en la necesidad de alcanzar las metas de la sostenibilidad, señalando el ACV como una de las metodologías de gestión ambiental con mayor reconocimiento a nivel mundial. Por último, se presenta el estado del arte de los distintos estudios de ACV realizados en los sectores bajo estudio.

Sección II: Análisis de Ciclo de Vida aplicado al sector vitivinícola

Esta sección se centra en la aplicación de ACV y otras metodologías complementarias al sector vitivinícola en la DO Ribeiro y la DO Rías Baixas. El Capítulo 4 analiza el perfil ambiental de la producción de vino —viticultura y vinificación— en una explotación en la DO Ribeiro para las cosechas 2007-2010. Las principales conclusiones derivadas de este estudio se detallan a continuación: i) los potenciales impactos ambientales se encuentran asociados a las operaciones agrícolas; ii) el uso de fertilizantes merece especial atención debido a su contribución al impacto ambiental; iii) dentro de las operaciones de vinificación, la etapa de embotellado presenta un importante impacto potencial debido al proceso de fabricación del vidrio; y iv) el estudio de varias cosechas ha permitido identificar las variaciones del perfil ambiental para cada una de ellas. Además, con el objetivo de reducir el impacto final de la producción de vino, se han propuesto diferentes acciones de mejora en dicho sistema productivo.

El Capítulo 5 combina la metodologías de ACV+DEA analizando un total de 40 explotaciones vitícolas en la DO Rías Baixas. El objetivo de este estudio es el de identificar la eficiencia operacional y ambiental de cada explotación. Las principales conclusiones de este estudio son las siguientes: i) un elevado porcentaje —60%— de los viticultores analizados operan de manera eficiente; ii) se han encontrado diferencias en cuanto a la eficiencia operacional entre los viticultores dependiendo del tamaño de las explotaciones; iii) los resultados obtenidos han sido proyectados en aras de estimar la reducción potencial de los principales consumos, traduciéndose en una importante reducción del impacto ambiental; y iv) la reducción de los consumos en las explotaciones vitícolas llevaría a un incremento de los beneficios de hasta un 10%.

El Capítulo 6 analiza el perfil ambiental de varias explotaciones de la DO Ribeiro en los años 2010 y 2011. Las explotaciones analizadas se caracterizan por utilizar diferentes técnicas de viticultura: i) biodinámica, que implica un gestión innovadora en cuanto a los insumos y las intervenciones sobre la propia viña; ii) híbrido convencional-biodinámica, que utiliza técnicas propias de la biodinámica combinadas con otras de la viticultura convencional; y iii) convencional. Las principales conclusiones obtenidas son: i) la viticultura en biodinámica ha presentado un menor impacto que la convencional; ii) la menor intervención sobre el viñedo ha sido el principal responsable de estas reducciones, destacando una menor dependencia de la mecanización de las diferentes operaciones; iv) la viticultura biodinámica ha mostrado un menor rendimiento por hectárea y una mayor dependencia del trabajo manual, lo cual penaliza otro tipo de indicadores tales como la ocupación de suelo.

El Capítulo 7 profundiza en la estimación de las emisiones de GEIs en la DO Ribeiro en su conjunto. Para ello combina las metodologías de ACV y de CUT para evaluar las emisiones de GEIs durante el periodo 1990-2009 derivadas de las operaciones vitícolas y las diferentes dinámicas de CUT encontradas en la región. Se presentan las principales conclusiones obtenidas para este estudio: i) la emisión/fijación de GEIs derivadas de CUT han sido relevantes en base a los usos del suelo de partida; ii) se han identificado diferentes tendencias en cuanto a emisiones de GEIs en base al periodo estudiado; iii) los cambios sociales y demográficos observados en el área bajo estudio pueden ser responsables de las tendencias de CUT encontradas; y iv) este estudio ha sido el primero de estas características que se ha llevado para una región concreta, por tanto, debido a los resultados obtenidos, las emisiones derivadas del CUT deberían tenerse en cuenta para futuros estudios de vino y ACV.

Sección III: Análisis de Ciclo de Vida aplicado a productos marinos

Esta sección analiza el perfil ambiental de diferentes productos de origen marino y la pesca extractiva mediante el uso de ACV, TRE_{prot} y Huella de Carbono. El Capítulo 8 analiza —desde una perspectiva de ciclo de vida— la eficiencia energética de las capturas de la flota pesquera gallega introduciendo el indicador TRE_{prot}. Las principales conclusiones obtenidas de este estudio son las siguientes: i) el uso de este indicador ha resultado ser de gran utilidad para comunicar las eficiencias energéticas de las capturas; ii) las capturas de la flota gallega en su conjunto presentan un valor de TRE_{prot} de 7,6%; iii) se han encontrado diferentes resultados para este indicador en función de los caladeros donde opera la flota bajo estudio; iv) el retorno energético para productos procesados fue menor debido a la necesidad de incluir etapas que demandan energía dentro del proceso; y v) el tipo de asignación llevada a cabo durante la etapa de inventario, así como las desviaciones estándar de las entradas al sistema, han resultado ser una fuente de incertidumbre que podrían influenciar el valor de TRE_{prot}.

El Capítulo 9 analiza la Huella de Carbono para un producto multi-ingrediente en base a pescado —varitas de merluza rebozadas. Para ello, se analizaron las etapas de pesca del ingrediente principal (merluza) en los caladeros del Cono Sur en Chile, el procesado a bordo de las capturas, el transporte a Galicia y el procesado en planta para la elaboración del producto final. Las principales conclusiones de este estudio corresponde a los siguientes apartados: i) las operaciones de pesca presentan contribuciones importantes en las emisiones de GEIs, destacando el consumo de diésel y los agentes refrigerantes; ii) la producción de otros ingredientes tales como trigo y aceite de girasol son relevantes en términos de emisiones de GEIs; y iii) la reducción de emisiones relacionada con las operaciones de pesca se centrarían en un cambio en el tipo de agente refrigerante utilizado, puesto que una actuación sobre la reducción de los consumos de diésel no sería factible.

El Capítulo 10 analiza la Huella de Carbono del producto analizado previamente, pero en este caso se centra en las etapas posteriores dentro de la cadena de suministro: distribución, venta y consumo, analizando las diferentes opciones de preparado propuestas por el fabricante. Las principales conclusiones obtenidas de este estudio son: i) el patrón de consumo de este tipo de productos ha demostrado ser relevante en la Huella de Carbono final; ii) las decisiones tomadas por parte del consumidor pueden implicar una importante reducción final de las emisiones de GEIs; y iii) entre los diferentes escenarios analizados, el preparado en horno, pese a requerir un mayor consumo eléctrico, conlleva una menor Huella de Carbono que las otras formas de preparado debido a que no precisa aceite para su preparado.

El Capítulo 11 analiza el perfil ambiental mediante ACV de la cadena de suministro de diferentes productos derivados de la sardina. Para ello, se realizó un estudio cruzado de varios productos tomando como base un mismo aporte proteico, analizando: sardina en lata, sardina fresca y merluza capturada usando la sardina como cebo. Se han tenido en cuenta las diferentes etapas dentro de la cadena de suministro de cada producto, desde las operaciones de pesca (sardina y merluza), descarga y subasta en puerto, procesado (sardina en lata) y consumo final (con diferentes opciones de preparado y consumo para cada uno de los productos). Las principales conclusiones obtenidas de este estudio son: i) la forma en la que el consumidor decide consumir un cierto producto influye considerablemente en el impacto del proceso; ii) la forma de preparado de un mismo producto mostró diferencias significativas en cuanto a impacto potencial; iii) los productos con un menor procesado presentaron menor impacto en comparación con aquellos más elaborados; y iv) los principales responsables de impacto de los productos analizados son aquellos relacionados con los procesos involucrados. En este sentido, el consumo de diésel (especialmente para el caso de la pesca de merluza debido al doble escenario de pesca —sardina para cebo y merluza), embalaje (lata) y el aceite de oliva han destacado por su contribución al impacto final.

Finalmente, el Capítulo 12 analiza el perfil ambiental de la obtención de alginato de sodio a partir de algas pardas en Estados Unidos. En este caso, se incluyeron las etapas desde la extracción del alga del medio natural hasta su posterior procesado para la extracción de alginato. Las principales conclusiones de este estudio se compilan en dos aspectos fundamentales: i) se destaca la etapa de extracción de alginato en laboratorio como la etapa con mayor impacto ambiental, fundamentalmente debido al consumo eléctrico; ii) el consumo de diésel, al igual que los sistemas de pesca extractiva, ha sido el principal responsable del impacto de la etapa de recolección del alga en el mar. Se han evaluado una serie de acciones de mejora con el objetivo de mejorar el perfil ambiental del proceso y la recuperación de etanol resultó ser clave para este tipo de procesos.

Sección IV: Conclusiones

Para finalizar, la Sección IV muestra en el Capítulo 13 las principales conclusiones obtenidas a lo largo de esta tesis doctoral en la que se ha comprobado que el Análisis de Ciclo de Vida es una herramienta muy útil en la evaluación del perfil ambiental de los sectores estudiados.

Curriculum vitae

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March 2009	**Bachelor of Science in Biology.** Faculty of Biology, University of Santiago de Compostela.
March 2011	**Master of Science in Environmental Engineering**. School of Engineering, University of Santiago de Compostela.
Fall 2015 (expected)	**PhD in Chemical Engineering**. School of Engineering, University of Santiago de Compostela. Tittle: *From Vineyard to Sea. Application of Life Cycle Assessment to Wine and Seafood Sectors.*

LANGUAGES

Spanish	Mother tongue
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English	First Certificate English (FCE). Cambridge University. (July 2013) Regular Classes in the Laramie County Community College. Laramie, Wyoming, United States. (Spring Semester 2014). English as a Second Language (ESL) advance level classes in the Laramie County Community College. (Spring Semester 2014).
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RESEARCH STAYS

College of Agriculture and Natural Resources (University of Wyoming). Laramie, Wyoming. February-May 2014.

PARTICIPATION IN RESEARCH PROJECTS

Sustainability in emerging technologies of wastewater treatment through life cycle assessment and carbon footprint methodologies. Galician Government (PGIDIT09MDS010262PR). Project leader: Gumersindo Feijoo (University of Santiago de Compostela).

Technology based on enzymatic membrane reactors for removing endocrine disruptor compounds. Spanish Government (CTQ2010-20258). Project leader: María Teresa Moreira Vilar (University of Santiago de Compostela).

Creating and testing method for controlling the air quality base on a new biotechnological tool. Use of a devitalized moss clone as passive contaminant sensor (MOSSCLONE).European Union (EU FP7). Project leader: José Angel Fernández Escribano (University of Santiago de Compostela)

Climate Change and aquaculture. MAGRAMA. Ministry for the Environment and Rural and Marine Affairs; Biodiversity Foundation. Project leader: Gumersindo Feijoo (University of Santiago de Compostela)

Zero Emissions City (ZEC). Galician Government (Conecta PEME). Project leader: Gumersindo Feijoo Costa (University of Santiago de Compostela).

Environmental and Energy Optimization of the Logistic Supply Chain (EOSIN). Galician Government (Conecta PEME). Project leader: Gumersindo Feijoo Costa (University of Santiago de Compostela)

Servicizing Policy for Resource Efficient Economy (SPREE). European Union (EU FP7). Project leader: Eugenijus Butkus (The Research Council of Lithuania)

SCHOLARSHIPS AND CONTRACTS

Scholarship of Spanish Ministry of Education and Science for Bachelor of Science in Biology. From course 2002/2003 to 2007/2008.

Scholarship of Spanish Ministry of Education and Science for Master of Science in Environmental Engineering. From course 2009/2010 to 2010/2011.

Scholarship in Conservas Isabel de Galicia S.A. August 2010 - September 2010.

Research contract at the University of Santiago de Compostela. March 2011– December 2014.

COURSES

1.- **Title:** Eco-design as a key tool for innovation in industry (Doctoral course of the Chemical Engineering Program)
Organized by: Department of Chemical Engineering, University of Santiago de Compostela
Place and date: Santiago de Compostela, March 2012

2.- **Title:** SciFinder training (Doctoral course of the Chemical Engineering Program)
Organized by: Department of Chemical Engineering, University of Santiago de Compostela
Place and date: Santiago de Compostela, May 2015

3.- **Title:** Design of biotechnological processes (Doctoral course of the Chemical Engineering Program)
Organized by: Department of Chemical Engineering, University of Santiago de Compostela
Place and date: Santiago de Compostela, July 2014

REGISTERED TRADEMARKS

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Conference: LCA XI
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Participation: Oral presentation
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Place and date: Almería (Spain), October 2013

6.- **Authors:** Villanueva-Rey, P., Vázquez-Rowe, I., Otero, M., Blanco, R., Moreira, M.T., Feijoo, G.
Title: The effect of land use changes on greenhouse gas emissions in the wine sector: a case study for the *Ribeiro* appellation (NW Spain)
Participation: Oral presentation
Conference: SETAC Europe 24th Annual Meeting
Place and date: Basel (Switzerland), May 2014

7.- **Authors:** Villanueva-Rey, P., Vázquez-Rowe, I., Otero, M., Moreira, M.T., Feijoo, G.
Title: Dynamic Life Cycle Assessment of the *Ribeiro* wine appellation (NW Spain) in the period 1989-2009
Participation: Oral presentation
Conference: LCA Food 2014
Place and date: San Francisco (USA), October 2014

8.- **Authors:** Villanueva-Rey, P., Vázquez-Rowe, I., Hospido, A., M., Moreira, M.T., Feijoo, G.
Title: Environmental sustainability pathways based on one single raw material: European pilchard (*Sardina pilchardus*) in NW Spain
Participation: Poster
Conference: LCA Food 2014
Place and date: San Francisco (USA), October 2014

9.- **Authors:** Villanueva-Rey, P., Lorenzo-Toja, Y., González-García, S., Pereira, A., Vence, X., Moreira, M.T., Feijoo, G.
Title: Environmental profile of new milk consumption patterns: vending machines
Participation: Oral presentation
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Place and date: Novi Sad (Serbia), November 2014

BOOK CHAPTERS

1.- Vázquez-Rowe, I., Villanueva-Rey, P., Moreira, M.T., Feijoo, G., 2014. A review on energy use and greenhouse gas emissions derived of worldwide hake fishing. Assessment of product Carbon Footprint in different industrial sectors Vol. II. In: Subramanian Senthilkannan Muthu (ed). Springer.1-29 pp.

2.- Villanueva-Rey, P., Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2014. The use of Carbon Footprint in the wine sector: methodological assumptions. Assessment of product Carbon Footprint in different industrial sectors Vol. II. In: Subramanian Senthilkannan Muthu (ed). Springer. 269-298 pp.

PUBLICATIONS

Published

1.- Vázquez-Rowe, I., Villanueva-Rey, P., Moreira, M.T., Feijoo, G., 2012. Environmental analysis of *Ribeiro* wine from a timeline perspective: Harvest year matters when reporting environmental impacts. Journal of Environmental Management. 98, 73–83.

2.- Vázquez-Rowe, I., Villanueva-Rey, P., Iribarren, D., Teresa Moreira, M., Feijoo, G., 2012. Joint life cycle assessment and data envelopment analysis of grape production for vinification in the *Rías Baixas* appellation (NW Spain). Journal of Cleaner Production 27, 92–102.

3.- Vázquez-Rowe, I., Villanueva-Rey, P., Mallo, J., De la Cerda, J.J., Moreira, M.T., Feijoo, G., 2013. Carbon footprint of a multi-ingredient seafood product from a business-to-business perspective. Journal of Cleaner Production 44, 200–210.

4.- Vázquez-Rowe, I., Villanueva-Rey, P., Moreira, M.T., Feijoo, G., 2013. The role of consumer purchase and post-purchase decision-making in sustainable seafood consumption. A Spanish case study using carbon footprinting. Food Policy 41, 94–102.

5.- Villanueva-Rey, P., Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2014. Comparative life cycle assessment in the wine sector: biodynamic vs. conventional viticulture activities in NW Spain. Journal of Cleaner Production 65, 330–341.

6.- Vázquez-Rowe, I., Villanueva-Rey, P., Moreira, M.T., Feijoo, G., 2014. Edible Protein Energy Return on Investment Ratio (ep-EROI) for Spanish Seafood Products. AMBIO 43, 381–394

7.- Vázquez-Rowe, I., Villanueva-Rey, P., Hospido, A., Moreira, M.T., Feijoo, G., 2014. Life cycle assessment of European pilchard (*Sardina pilchardus*) consumption. A case study for Galicia (NW Spain). Science of The Total Environment 475, 48–60.

8.- Villanueva-Rey, P., Vázquez-Rowe, I., Otero, M., Moreira, M.T., Feijoo, G., 2015. Accounting for time-dependent changes in GHG emissions in the *Ribeiro* appellation (NW Spain): Are land use changes an important driver? Environmental Science & Policy 51, 215–227.

9.- González-García, S., Villanueva-Rey, P., Belo, S., Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., Arroja, L., 2015. Cross-vessel eco-efficiency analysis. A case study for purse seining fishing from North Portugal targeting European pilchard. The International Journal of Life Cycle Assessment 1–14.

Submitted for publication

1.- Vázquez-Rowe, I., Villanueva-Rey, P., Santimillán, J., Ramzy, K., Moreira, M.T., Feijoo, G. Opportunities and challenges of uptaking Life Cycle Assessment in seafood certification: a case study for Spain. Science of the Total Environment.

2.- Noya, I., Villanueva-Rey, P., González-García, S., Fernandez, M.A., Rodriguez, M.R., Moreira, M.T., Feijoo, G. Life cycle assessment of pig production: a case study in Northwest Spain. Agriculture, Ecosystems & Environment.

OTHER INTERESTING DATA

- Reviewer for the following international journals:

  - *International Journal of Environmental Research*
  - *Energy Policy*
  - *Science of the Total Environment*
  - *Fisheries Research*
  - *Turkish Journal of Fisheries and Aquatic Sciences*
  - *Social Science Research*
  - *International Journal of Hydrogen Energy*
  - *International Journal of Life Cycle Assessment*
  - *Journal of Cleaner Production*

- Regular classes in the Laramie County Community College during Spring semester 2014:

  - Introduction to Cultural Anthropology
  - Public Speaking
  - English Composition

- Member of Laramie Morning Toastmasters Club.

- Collaborator with Kiwanis International Club in Laramie, Wyoming.

- Academic work in Science of Biology during course 2007/2008: *Aplicación de un Sistema de detección rápida de focos de contaminación atmosférica mediante biomonitorización con briófitos terrestres.*

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