INTERNATIONAL DOCTORAL
SCHOOL OF THE USC

Antonio J.
Cortés Montoya

PhD Thesis

Challenges on the road to sustainability in strategic primary sectors

Santiago de Compostela, 2022

Doctoral Programme in Chemical and Environmental Engineering

TESE DE DOUTORAMENTO

Challenges on the road to sustainability in strategic primary sectors

Antonio José Cortés Montoya

ESCOLA DE DOUTORAMENTO INTERNACIONAL DA UNIVERSIDADE DE SANTIAGO DE COMPOSTELA

PROGRAMA DE DOUTORAMENTO EN ENXEÑARIA QUÍMICA E AMBIENTAL

Santiago de Compostela

ANO 2022

DECLARACIÓN DO AUTOR/A DA TESE

D./Dna. **Antonio José Cortés Montoya**

Título da tese: **Challenges on the road to sustainability in strategic primary sectors**

Presento a miña tese, seguindo o procedemento axeitado ao Regulamento, e declaro que:

1) A tese abarca os resultados da elaboración do meu traballo.
2) De ser o caso, na tese faise referencia ás colaboracións que tivo este traballo.
3) Confirmo que a tese non incorre en ningún tipo de plaxio doutros autores nin de traballos presentados por min para a obtención doutros títulos.
4) A tese é a versión definitiva presentada para a súa defensa e coincide a versión impresa coa presentada en formato electrónico

E comprométome a presentar o Compromiso Documental de Supervisión no caso de que o orixinal non estea na Escola.

En **Santiago de Compostela**, **20 de Xullo de 2022**.

Sinatura electrónica

AUTORIZACIÓN DO DIRECTOR / TITOR DA TESE

Challenges on the road to sustainability in strategic primary sectors

D. Gumersindo Feijoo Costa
Dna. Maria Teresa Moreira Vilar

INFORMA/N:

Que a presente tese, correspóndese co traballo realizado por D. **Antonio José Cortés Montoya**, baixo a miña dirección/titorización, e autorizo a súa presentación, considerando que reúne os para requisitos esixidos no Regulamento de Estudos de Doutoramento da USC, e que como director desta non incorre nas causas de abstención establecidas na Lei 40/2015.

De acordo co indicado no Regulamento de Estudos de Doutoramento, declara tamén que a presente tese de doutoramento é idónea para ser defendida en base á modalidade de Monográfica con reproducción de publicaciones, nos que a participación do doutorando foi decisiva para a súa elaboración e as publicacións se axustan ao Plan de Investigación.

En **Santiago de Compostela**, **20 de Xullo de 2022**

Agradecimientos

En primer lugar, me gustaría agradecer al proyecto "*STAR-ProBio* Sustainability Transition Assessment and Research of Bio-based products" (Horizon 2020 no. 727740) y al proyecto "*NEPTUNUS* Water-Energy_Seafood Nexus: Eco-innovation and Circular Economy Strategies in the Atlantic Area" (Interreg Atlantic Area EAPA_576/2018), cuyos recursos y financiación posibilitaron mi completa dedicación a la elaboración de esta tesis. Además, me gustaría mencionar también al Centro Singular de Investigación CRETUS (ED431E 2018/01) por el apoyo institucional que me ha brindado durante todos estos años.

A mis directores de tesis, el profesor Gumersindo Feijoo Costa y la profesora Maria Teresa Moreira, muchísimas gracias por haberme dado la oportunidad de iniciarme en el mundo investigador y haberme permitido trabajar con vosotros en estos años. Sin olvidar por supuesto la dedicación y el apoyo que habéis brindado a mi trabajo, por el respeto a mis sugerencias e ideas y por todo el tiempo y esfuerzo que habéis invertido en apoyar y revisar mi trabajo, sin duda, acabar esta tesis no hubiera sido posible sin vuestra inestimable ayuda.

También me gustaría mencionar a todos los profesores e investigadores con los que he tenido el gusto de colaborar en estos años y, finalmente, muchas gracias a todos mis compañeros del Biogroup, ya sean profesores, investigadores, doctorandos o técnicos, por acompañarme durante estos años. Sin vosotros, completar este trabajo hubiera sido infinitamente más difícil.

Outline of the thesis

Resumo

Nas últimas décadas a presión sobre o planeta terra por parte dos humanos foi aumentando de modo exponencial ata hoxe en que a Terra está a ser explotada moito máis alá das súas capacidades, superando as posibilidades de rexeneración de recursos naturais. O crecemento global da poboación, o aumento dos ingresos e a urbanización unen forzas para representar serios desafíos para os sistemas alimentarios e agrícolas, mentres que os recursos naturais adoitan ser máis limitados para apoiar esa prestación de servizos. Como resultado, a presión sobre os diferentes ecosistemas é enorme e os impactos ambientais causados na terra, o aire e o mar non teñen precedentes. Neste contexto, o sector primario é un dos principais contribuíntes ao cambio climático xa que é responsable de aproximadamente un terzo das emisións globais de Gases de Efecto Invernadoiro (GEI) procedentes de fontes antrópicas. A transformación cara un sector primario sostible require unha multitude de medidas en diversos ámbitos, como o medio ambiente, a economía e a sociedade, para aumentar a súa resiliencia ante futuros eventos adversos ou choques. Para iso, os cambios nos esquemas de produción actuais son unha das ferramentas máis potentes e eficaces. Establecer procesos de valorización para aplicar os principios da economía circular aos sectores produtivos considérase a pedra angular para lograr este obxectivo.

Así, esta tese ten como principal finalidade analizar desde o punto de vista ambiental diferentes estratexias para a implementación da economía circular en sectores primarios relevantes, en concreto o agrogandeiro e o pesqueiro. Para este propósito, este documento dividir en cinco seccións, as cales vanse a explicar a continuación.

A primeira sección está composta polo Capítulo 1 e polo Capítulo 2 e está dirixida a proporcional aos posibles lectores unha visión xeral do estado do arte do problema que se abordará nesta tese, así como as ferramentas de avaliación metodolóxicas empregadas para este fin.

Capítulo 1: Estado do arte.

Este capítulo céntrase non estado de arte do campo de estudo. Así, a situación planetaria actual descríbese dende un punto de vista multidisciplinar. Nun primeiro lugar dende ou punto de vista ambiental, profundando sobre todos nos impactos ambientais derivados do sector primario e, en concreto, a cadea produtiva alimentaria. A continuación descríbese a importancia dous tres subsectores estudados nesta tese, o subsector agrogandeiro, o pesqueiro e o acuícola. Despois enuméranse algúns datos xenéricos sobre a xeración de residuos e a importancia que se lle da actualmente, xa que é un tema altamente tratado desde organismos internacionais como a Comisión Europea, a Axencia de Protección Ambiental dos Estados Unidos o a Organización para a Cooperación e o Desenvolvemento Económico. Despois enuméranse as características dun sector primario sostible e circular, considerando as principais achegas científicas para adoptalo, e como a adopción dun sistema primario sostible e circular contribuiría a avanzar non marco de todos os Obxectivos de Desenvolvemento Sostible. Finalmente neste primeiro capítulo tamén preséntanse os obxectivos e a estrutura xeral desta tese, incluíndo as diferentes seccións e capítulos que a compoñen.

Capítulo 2. Ferramentas de xestión ambiental

Neste segundo capitulo da primeira sección, continuase coa contextualización, para neste caso dar paso á descripción das ferramentas metodolóxicas utilizadas ao longo desta tese. Así pois, nun primeiro lugar detállase a metodoloxía de avaliación de impactos ambientais de Análises de Ciclo de Vida, incluíndo as súas principais etapas como son a definición do obxectivo e alcance, a análise de inventario de ciclo de vida, e a avaliación e interpretación do impacto ambiental. Neste contexto, tamén profundízase na metodoloxía de Análise por Envoltura de Datos, xa que esta servirá de apoio para a estimación da eficiencia de diferentes explotacións agrogandeiras e acuícolas. No último apartado deste capítulo inclúense unha revisión bibliográfica de estudos de ACV publicados en revistas científicas sobre os aspectos tratados nesta tese: diferentes alimentos como carnes, froitas e verduras, produtos lácteos e peixe; así como alternativas de valorización de residuos agrogandeiros.

A segunda sección da tese titúlase “Economía Circular no sector agrícola” e estase formada por tres capítulos (Capítulo 3 a Capítulo 5). A Sección II céntrase na análise ambiental de diferentes alternativas para a valorización completa valorización de residuos agroindustriais procedentes da industria vitiviñícola. Deste xeito, o obxectivo principal é avaliar os impactos ambientais á vez que se busca maximizar a xeración de co-produtos. A información contida nestes capítulos, así como os principais resultados explícanse a continuación.

Capítulo 3. Avaliación integrada da valorización de lías de viño para a obtención de produtos de valor engadido

Este capítulo céntrase no estudo ambiental dun proceso de valorización de leas de viño para producir produtos de valor engadido. Estas leas son esencialmente células mortas restantes do proceso de fermentación e son un dos principais residuos sólidos producidos durante o proceso de vinificación. Neste senso, o capítulo 3 ten como obxectivo avaliar desde unha filosofía de ciclo de vida os impactos ambientais derivados dun proceso de biorefineria na que se producen 4 produtos principais a partir das lías, bioetanol, tartrato de calcio, un extracto rico en antioxidantes e unha fracción sólida rica en proteínas que pode usarse como alimento do gando.

Segundo os resultados obtidos neste capítulo, o principal responsable do impacto do sistema é a produción de vapor para os diferentes procesos de destilado necesarios para separar os produtos, isto é debido a que todas os procesos relacionados coa separación e extracción de compoñentes son físicos, polo que non hai consumo de químicos. Neste sentido, o rendemento ambiental deste sistema compárase con outros procesos có obxectivo de producir antioxidantes a partir de diferentes materiais. A valorización das lías de viño demostra un mellor perfil ambiental ao longo do seu ciclo de vida debido a que non necesita un alto consumo enerxético ou de químicos, reducindo de media un 75% os impactos respecto ao proceso tomado como referencia. Pódese concluír a partir deste capitulo que a biorefineria de co-produtos do viño é un enfoque mais que adecuado para recuperar produtos de valor engadido da cordo cos principios da economía circular, onde os residuos son convertidos en novos materiais, reducindo o impacto durante todo o ciclo de vida. Con todo, aínda hai marxe de mellora, e futuras investigacións deberían enfocarse en optimizar os sistemas de extracción de antioxidantes, por exemplo utilizando sistemas de extracción acuosa de dous etapas, ou ben extracción asistida por ultrasóns ou microondas, reducindo o consumo enerxético do sistema.

Capítulo 4. Desentrañando o impacto ambiental da produción de compostos bioactivos e emenda orgánica a partir do bagazo de uva

En paralelo ao enfoque proposto no anterior capitulo, neste proponse a avaliación ambiental dun proceso de tratamento biolóxico para a conversión de residuos sólidos de bagazo de uva en recursos a través da biooxidación e estabilización dos residuos orgánicos mediante miñocas, denominado vermicompostaxe. Neste roteiro de valorización, prodúcense diferentes produtos de valor engadido con potenciais aplicacións no sector cosmético, alimentario e farmacéutico. O vermicompostaxe é un proceso natural baseado nas interaccións das miñocas (principalmente da especie *Eisenia foetida* ou *Eisenia andrei*) cos microorganismos endóxenos presentes nos residuos como resultado da descomposición da materia orgánica.

Así pois, os principais resultados deste capítulo demostraron que o consumo enerxético para a destilación previa ao proceso de vermicompostaxe e extracción de aceite de sementes debido principalmente á necesaria xeración de vapor de alta temperatura. Doutra banda, compárase este proceso de valorización con outros tratamentos de residuos orgánicos, tendo en conta o impacto ambiental en termos de pegada de carbono e índice de impacto normalizado e o custo operacional. O proceso de vermicompostaxe localízase no cuartil de alto impacto pero baixo custo de operación e, con todo, é relevante ter en conta que se considéranse os beneficios económicos producidos polos diferentes produtos (por exemplo cambiando a unidade funcional a unha baseada en beneficios económicos), o impacto do vermicompostaxe tanto en termos de pegada de carbono como de índice de impacto normalizado é menor que calquera das outras alternativas.

Capítulo 5. Implicacións ambientais da produción de enerxía baseada no biohidrógeno proveniente do reformado con vapor de auga de residuos alcohólicos

A produción de viño xera unha gran cantidade de residuos sólidos, pero tamén líquidos, os cales normalmente son tratados directamente nas destilerías para obter produtos alcohólicos, pero nestes procesos, outras purgas con escaso valor comercial son xeradas. O caso de estudo analizado neste capítulo céntrase en avaliar o potencial deste tipo de correntes residuais para a produción sostible de biohidróxeno como un vector enerxético de alta calidade como unha etapa final na completa valorización de residuos vitivinícolas, unha vez que a extracción de produtos de valor engadido xa non é posible. Deste xeito, o principal obxectivo deste capítulo é analizar desde un punto de vista ambiental os impactos asociados co reformado con vapor de auga de residuos alcohólicos procedentes de destilerías, a corrente de saída deste proceso alimenta unha pila de combustible de óxido sólido de 3kW para producir enerxía. Esta recuperación de enerxía é 100% compatible cos procesos avaliados en capítulos previos, polo que a produción de produtos de valor engadido pode ser complementario con este método de recuperación de enerxía.

De acordo cos resultados obtidos, destácanse as emisións de 350 kg CO_2 eq para a valorización de 1 tonelada de residuos alcohólicos. Comprobando as contribucións relativas, a fabricación da pila de combustible é o maior contribuínte aos impactos ambientais do sistema para case todas as categorías, seguido polo consumo enerxético global do sistema. Se se comparan estes resultados con diferentes alternativas para a produción de biohidróxeno a partir doutras fontes, a pesar de que este proceso de valorización ten uns resultados malos en termos de pegada de carbono e formación de oxidantes fotoquímicos debido ás emisións fuxidas de metano, o seu rendemento ambiental é mellor que calquera dos procesos publicados. É relevante destacar que os resultados obtidos refírense a escala de laboratorio, polo que é necesario seguir avanzando para obter datos a unha escala maior.

A terceira sección da tese titúlase "Economía Circular no sector pesqueiro" e consta de dous capítulos. Así, os capítulos 6 e 7 céntranse na construción dun inventario de ciclo de vida completo da extracción y transformación de vieira mediante métodos tradicionais e no análise ambiental da inclusión dunha valorización completa dos residuos producidos nunha industria conserveira galega, respectivamente. A información contida nestes capítulos, así como os principais resultados explícanse a continuación.

Capítulo 6. Avaliación da sustentabilidade ambiental da Pesqueira costeira de vieira (*Pecten maximus*) en Galicia

Dentro das diferentes especies acuáticas con interese comercial, os bivalvos foron considerados tradicionalmente como unha fonte de proteína animal saudable e de altos niveis de ácidos graxos esenciais, o que provocou un importante aumento da demanda por parte dos consumidores, A vieira (*Pecten maximus*) é unha especie de bivalvo pertencente á familia Pectinidae esencialmente costeira que vive en fondos limpos de area firme, fina ou de grava. Co obxectivo de encher os ocos existentes en termos dun inventario de vida específico e exhaustivo sobre a captura e procesado de vieira, para a realización deste capítulo recolléronse datos de 14 botes artesanais rexistrados no porto de cambados, os cales representan o 67% dos 21 botes rexistrados que se dedican á vieira.

O inventario do ciclo de vida da etapa de pesca abarcou todos os elementos necesarios para o funcionamento do bote, incluíndo un consumo medio de combustible de 123 mL ou 772,7 mg de rede por vieira. Estes valores indican que as redes, aínda que necesitan reparacións e renovacións constantes, representan un consumo moi baixo ao longo do ano en comparación cos elementos principais, como o gasóleo. Considerando a análise ambiental realizada sobre o inventario, demóstrase que a etapa de captura é o principal elemento condutor do impacto ambiental final do produto, mentres que o procesado ten un menor impacto, xa que só destaca o consumo eléctrico da planta de evisceración. Doutra banda, tendo en conta os impactos específicos da etapa de pesca destacan os derivados da produción e consumo de diésel, así como a pintura antiincrustante. Tamén demóstrase que, na combinación de aspectos ambientais e nutricionais, a vieira presenta un dos mellores perfís dentro da categoría de mariscos. O contido en proteínas da vieira é un dos máis altos da categoría de mariscos e peixe, ao nivel dalgunhas carnes como a de vacún ou a de polo; mentres que o impacto ambiental en termos de pegada de carbono está, como era de esperar dado o seu baixo rendemento comestible, á altura doutros moluscos e crustáceos como o percebe e o lagostino.

Capítulo 7. Estratexia multiproduto para mellorar o perfil ambiental da industria conserveira cara á economía circular

Un dos principais retos aos que se enfronta a sociedade é continuar producindo comida suficiente para a poboación. Para conseguir este obxectivo á vez que se loita contra a redución dos impactos ambientais e outros temas, o primeiro paso é demostrar que aplicar os principios de economía circular permiten reducir os impactos reais da cadea de produción alimentaria. Neste sentido, neste capítulo proponse a avaliación do perfil ambiental relacionado coa cadea de valor de conservas de atún nunha conserveira galega mediante a metodoloxía unha perspectiva atribucional, incluíndo todos os procesos produtivos e de valorización dos residuos orgánicos xerados. Así, ademais das latas de atún como principal produto, considéranse como saídas de coproductos envases de paté de atún preparados con outras partes non enlatadas, fariña e aceite de peixe.

Os resultados demostran que os principais sistemas responsables do impacto Ambiental resultaron ser a pesca do atún e o proceso de enlatado primario, resultados que entran dentro dos esperado tendo en conta a experiencia científica en estudos similares noutras conservas de peixe. O sistema avaliado comparouse cun escenario non circular no que só se teñen en conta os impactos do produto principal e a xestión dos residuos orgánicos xerados. Parece coherente pensar que ao eliminar os subsistemas relacionados coa valorización e os seus consumos materiais e enerxéticos, os impactos do sistema global son menores, con todo, se se segue un enfoque de produto, a inclusión de procesos de valorización mellora o perfil ambiental do produto principal, posto que os impactos totais son repartidos entre os diferentes co-produtos. O sistema exposto permite realizar unha primeira aproximación para os obxectivos expostos pola Unión Europea sobre sistemas "cradle-to-cradle", logrando un obxectivo de cero residuo orgánico.

Na cuarta sección da tese, titulada "Análise de ecoeficiencia", proponse e uso de metodoloxía ACV+DEA para comparar a eficiencia de múltiples unidades de produción con características colectivas semellantes. O uso desta metodoloxía evita problemas coas desviacións estándares que xorden cando vos usuarios dá ACV traballan con inventarios promedio. Ademais, este novo enfoque facilita a interpretación dos resultados, así como a verificación dá ecoeficiencia. Deste xeito, aplicouse o método ACV+DEA en dous casos de estudos relacionados cos sectores primarios avaliados nesta tese, a produción de leite (Capítulo 8) e a acuicultura de camaróns (Capítulo 9).

Capítulo 8. Perseguindo o roteiro da ecoeficiencia na produción láctea: o caso da rexión galega

Tendo en conta que a gandería é a principal responsable do consumo e as emisións derivadas da industria alimentaria, parece claro que a procura de procesos máis eficientes e sostibles debe converterse na pedra angular de calquera sistema de produción. Baseándose no enfoque combinado das metodoloxías ACV+DEA, o estudo da sustentabilidade ambiental e a ecoeficiencia non só pode avaliarse dun caso de estudo ou dunha unidade de proceso individual, senón que poden establecerse puntos de referencia para sistemas similares que se comportan de forma diferente. Deste xeito, é posible aprender daqueles sistemas que poden representar as mellores prácticas. Este capítulo avanza nesta dirección integrando ambas as metodoloxías de análises no cálculo de indicadores ambientais asociados á produción de leite para un amplo grupo de explotacións, preto de 100 unidades de decisión, principalmente de pequeno tamaño (entre 10 e 500 cabezas de gando) ao longo dos municipios galegos de Santa Comba, Lalín e Rodeiro.

A produción de penso (principalmente concentrado e millo e herba na granxa), así como as emisións directas de CH_4 e N_2O identificáronse como os procesos críticos do sistema. A pegada de carbono para unha granxa media estimouse en 1,33 kg de CO_2 por kg de leite corrixida en graxa e proteína (FPCM), un valor que está dentro do rango atopado en estudos similares, mentres que a pegada hídrica segundo a norma ISO 14046 é de 52,5 L por kg de FPCM. Identificáronse 21 explotacións leiteiras como eficientes, e a puntuación media de eficiencia das explotacións ineficientes foi de 0,58 sobre 1. A partir da comparación dos niveis actuais de funcionamento cos niveis obxectivo, foi posible cuantificar reducións medias de ata o 53% para os niveis de consumo de materiais, o que se traduce en reducións medias de impacto do 49% nos indicadores de pegada de carbono e do 55% nos de pegada de auga. Este capítulo mostra como o sector lácteo galego debe abordar os obxectivos de desenvolvemento sostible, especialmente os establecidos na Axenda 2030 para lograr unha mellora constante e unha produción sostible e eficiente.

Capítulo 9. Avaliación da ecoeficiencia da produción acuícola de camaróns en México

Considerando que só no estado mexicano de Sonora hai máis de 200 explotacións acuícolas, a análise deste sector require unha análise harmonizado, tendo en conta non só un enfoque de ciclo de vida, senón tamén o DEA. Esta capítulo céntrase na aplicación da metodoloxía ACV+DEA para avaliar a ecoeficiencia de 38 granxas de produción semi-intensiva de camaróns localizadas en Sonora.

Os resultados do ACV mostraron que a xestión dos pensos e o consumo de electricidade son os principais puntos críticos en case todas as categorías de impacto. Avaliáronse outras accións de mellora, a substitución da fariña de trigo por grans secos de destilería con solubles (DDGS) deu lugar a reducións do impacto ambiental que oscilaban entre o 2% e o 57%, segundo a categoría de impacto. Doutra banda, avaliouse a instalación de paneis fotovoltaicos na zona, buscando un cambio cara a unha produción de enerxía menos intensiva en carbono.

A aplicación conxunta de ACV+DEA proporciona un enfoque exhaustivo que permitiu distinguir as explotacións operativamente ineficientes e, aínda que só 5 de 38 consideráronse plenamente eficientes, a eficiencia media da mostra foi de 0,79 sobre 1. As reducións esperadas no consumo de materiais foron significativas, dando lugar a reducións estimadas do 3,6% ao 69,9% no índice de impacto normalizado en función da explotación avaliada. En conclusión, o potencial da acuicultura para satisfacer a demanda de produtos do mar móstrase como unha excelente oportunidade para contribuír á nutrición saudable da poboación, prestando ao mesmo tempo atención á conservación dos recursos mariños. Dado que a principal prioridade é o uso de alternativas ambientalmente sostibles, este capítulo pode considerarse unha guía útil para os xestores das granxas de camaróns, especialmente en México.

Finalmente, a Sección V consta do **Capítulo 10** que leva por título "Resultados xerais e conclusións dá tese". Como ou seu nome indica, esta sección recolle vos principais resultados obtidos dous lagos do documento á vez que enumera perspectivas futuras para a adopción dun sector primario máis circular e sostible. Así, a valorización de residuos para a produción de coproductos é unha das ferramentas mais poderosas para combater o cambio climático e a xeración descontrolada de residuos, xunto con outras pedras angulares como as melloras tecnolóxicas que permitan a valorización enerxética e a redución do consumo de materias primas. Ao mesmo tempo, a consecución destes obxectivos representa tamén unha gran oportunidade para avanzar cara unha auténtica sociedade circular, na que vos residuos son convertidos de forma continuada en recursos.

Neste contexto, as metodoloxías sobre a análise de impacto ambiental aplicadas nesta tese demostráronse como útiles para este propósito, tamén en combinación con outros instrumentos complementarios para a integración de aspectos da eficiencia operacional.

List of acronyms

CF	Carbon Footprint
DEA	Data Envelopment Analysis
DMU	Decision-Making Unit
ERA	Environmental Risk Assessment
EEA	European Environmental Agency
GHG	Greenhouse Gas
FPCM	Fat- and protein- Corrected Milk
FU	Functional Unit
FUI	Fuel Use Intensity
IOA	Input-Output Analysis
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
MFA	Material Flow Analysis
MSW	Municipal Solid Waste
PSA	Pressure Swing Adsorption
SMR	Steam Methane Reforming
SOFC	Solid Oxide Fuel Cell
SS	Subsystem
WF	Water Footprint
WFN	Water Footprint Network
WGS	Water Gas Shift

USC
UNIVERSIDADE
DE SANTIAGO
DE COMPOSTELA

SECTION I

CONTEXTUALISATION

Chapter 1

State-of-the-art

Summary

Our planet is currently near the limit of its capacity, far exceeding the regenerative capacity of natural resources. These conditions together with the exponential rate of population growth highlight the finite limits of resources and the generation of waste and air emissions. As a result, the pressure on different ecosystems is huge and the environmental impacts caused on land, air, and sea are uncertain. These burdens include climate change, and its increasingly unpredictable consequences.

In terms of resources and energy consumption, the primary sector is one of the largest industrial sectors and has been identified as a major player in climate change. Food production accounts for one quarter of the global anthropogenic GHG emissions. But also around 50% of the world's habitable land is used for agriculture, while more than two-thirds of global freshwater withdrawals are used for agriculture and around 80% of ocean and freshwater eutrophication is also caused by the agri-food sector.

In this context, the transformation towards a sustainable primary sector requires a multitude of measures in various domains such as environment, economy, and society to increase their resilience to future adverse events or shocks. To this end, changes in current production schemes are one of the most powerful and effective tools. To establish valorisation processes to apply the circular economy principles to productive sectors is considered as cornerstone for achieving this goal.

OUTLINE

1.1. ENVIRONMENTAL AND SOCIOECONOMIC CONTEXT

1.1.1. Current global situation

The global development model of human activities faces environmental problems of the first magnitude that, far from being solved, persist, and continue to be a deep-rooted evil. It is recognized that the world economy has lost dynamism, that improvements in the quality of life are not closing existing gaps, and that socio-economic development has been achieved with an excessive use of natural resources and serious environmental degradation. It is precisely the balance between socio-economic development and environmental preservation that we are faced with a binomial that is difficult to align.

The development of society has been a story of changing the planet natural systems to support increasingly sophisticated and comfortable ways of life for growing numbers of people. Over the millennia, wilderness areas adapted across the planet to allow settled communities to enjoy a secure supply of food, water, energy, and materials. However, no period has experienced interference with the planet biological machinery as markedly as in the second half of the twentieth century (Carpenter et al., 2009). To avoid discontinuities and disruptions in the biosphere, its ecosystems, societies and economies, changes and innovations in human behaviour, technology, governance and values are necessary (Will et al., 2018). This is illustrated by the demonstrated steady increase of CO_2 in the Earth's atmosphere (Peters et al., 2020) or the accumulation of hundreds of millions of tons of plastic waste (Jambeck et al., 2015). These examples clearly show that the producing societies of the industrial economy must not only rethink their value chains, products, raw materials, manufacturing methodologies and product life cycles, but also incorporate sustainability into products from the moment they are conceived (Dyllick and Hockerts, 2002).

These conditions together with the exponential rate of population growth (Dong et al., 2018), highlight the finite limits of resources and the generation of waste and air emissions (Schaltegger, 2018). Consequently, how to cope with the demands of current and future society to ensure quality of life but also environmental preservation is an increasingly pressing issue (Leal Filho et al., 2019). For example, the Earth Overshoot Day, the day when humanity´s annual demands on resources exceed the regenerative capacity of those resources (Figure 1.1). To determine the date of Earth Overshoot Day for each year, Global Footprint Network calculates the number of days of that year that Earth's biocapacity suffices to provide for humanity's Ecological Footprint. Earth Overshoot Day is computed by dividing the planet's biocapacity (the amount of ecological resources Earth is able to generate that year), by humanity's Ecological Footprint (humanity's demand for that year).

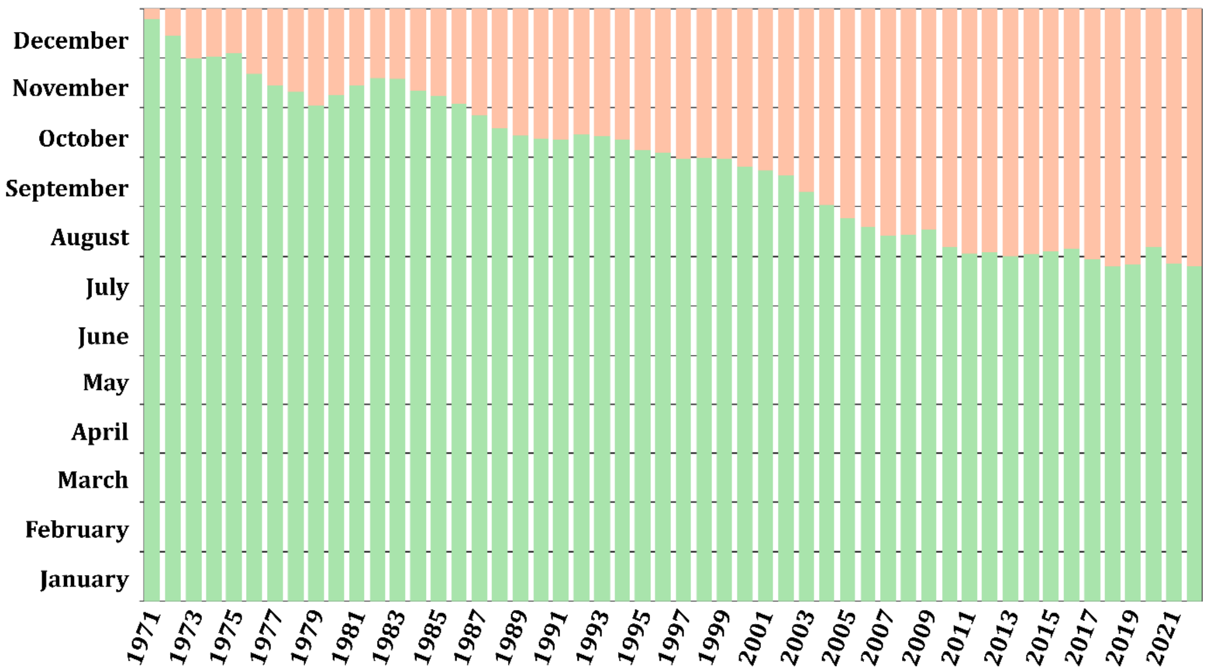


Figure 1.1. Earth Overshoot Day from 1971 to 2022 (Earth Overshoot Day, 2022).

Since the time this parameter began to be calculated in 1970, the Earth Overshoot Day has advanced more than 5 months (Lee et al., 2021), which means that currently, from a global point of view, it requires the equivalent of 1.75 planets to sustain us (Gills and Morgan, 2021). However, it is important to consider the great difference between countries, while poor countries as Angola, Bangladesh or Cameroon do not reach their overshot day, other richer countries as Spain, Sweeden or Luxembourg reach it on May 12th, April 3rd, and February 14th, respectively.

The exceptional confinement situation resulting from COVID-19 due to an acute respiratory infection of unknown origin in Wuhan, China, demonstrated that humans have the potential to reverse this depletion situation, at least for a short period of time (Lokhandwala and Gautam, 2020). The quarantine imposed due to COVID-19 was unprecedented in human history, as all markets were closed, public meetings were banned, travel restrictions were imposed and productive activity was largely halted, resulting in serious consequences for the global economy and the stock market (Khan et al., 2021). On the positive side, overall reductions were observed in air pollution and energy consumption (Wang and Su, 2020), as well as in $PM_{2.5}$, NO_2 and CO emissions (Kumari and Toshniwal, 2020). However, if the focus is on another environmental aspect, such as waste production and management, the intensification of single-use products and panic buying increased the production and consumption of plastic products, thus thwarting efforts to reduce this type of pollution (Sarkodie and Owusu, 2021). Currently, due to the consequences that, for more than 2 years, the pandemic has had on global productive activity and more recently the war in Ukraine has led to a situation of increasing instability in which rising prices and resource availability is the changing reality of everyday life, it seems that the precarious balance between our strong demand for goods and services and their production is increasingly instable, as evidenced by the shortage of electronic components for the automotive industry (Wu et al., 2021), high fertilisers prices (Barbieri et al., 2022) or the unstoppable escalation in energy, gas and fuel prices (Mišík, 2022). It is precisely in this context that we must consider how to change our production and consumption model in which the linear economic model based on the massive use of resources and the management of the waste generated is reaching its physical limits.

1.1.2. Food supply chain, environment, and climate change

During the second half of the 20th century, there was unprecedented growth in global crop production, driven in part by the recent availability of nitrogen fertilisers (Erisman et al., 2008) coupled with unprecedented population growth. Despite increased production capacity, the degree of humanity's susceptibility to a severe global food crisis in the 21st century is a matter of growing uncertainty (D'Odorico et al., 2018). Today, human society faces the challenge of providing food for a continuously growing population while resisting the effects of climate change and widespread degradation of natural resources (Bretschger and Pittel, 2020). Indeed, all projections indicate that at least a 70% increase in food production will be needed to meet food demand by 2050 (FAO, 2012), when the world's population will reach between 9.4 and 10.2 billion (Boretti and Rosa, 2019). Global food production will have to increase substantially to keep up with this growing demand and, at the same time, reduce its environmental impact (Hunter et al., 2017). Mitigation strategies to reduce GHG emissions must be combined with adaptation strategies that aim to reduce negative impacts and exploit beneficial opportunities under climate change (Anderson et al., 2020).

In terms of resources and energy consumption, the primary sector is one of the largest industrial sectors (Nemecek et al., 2016) and has therefore been identified as a major player in climate change. Food production amounts to around 13.7 billion metric tons of CO_2 eq (Poore and Nemecek, 2018), which account for 24% of global anthropogenic GHG emissions (Figure 1.2). It is therefore a significant contributor to climate change and, at the same time, is affected by it (Parajuli et al., 2019). As for every degree of global temperature increase, wheat yields are expected to decrease by 4-6% (Asseng et al., 2015; Liu et al., 2016), temperatures are expected to have a similar impact on maize productivity, and areas producing 56% of the global maize are expected to experience a decrease in yields by the end of the century (Bassu et al., 2014; Pugh et al., 2016). Around 50% of the world's habitable land is used for agriculture, while more than two-thirds of global freshwater withdrawals are used for agriculture (Alexander et al., 2017); and around 80% of ocean and freshwater eutrophication is also caused by the agri-food sector (Ritchie et al., 2018). The impact on pollinators, such as bees, is already under great pressure from habitat loss and intensive agriculture (Kehrberger and Holzschuh, 2019) and so are the effects of global warming on pests and diseases in livestock (Haile, 2020).

Delving deeper into the key drivers of this high environmental impact, in addition to the intrinsic impacts of food production itself, other "avoidable" impacts play an important role, such as environmental burdens related to food packaging and distribution worldwide (Yokokawa et al., 2018).

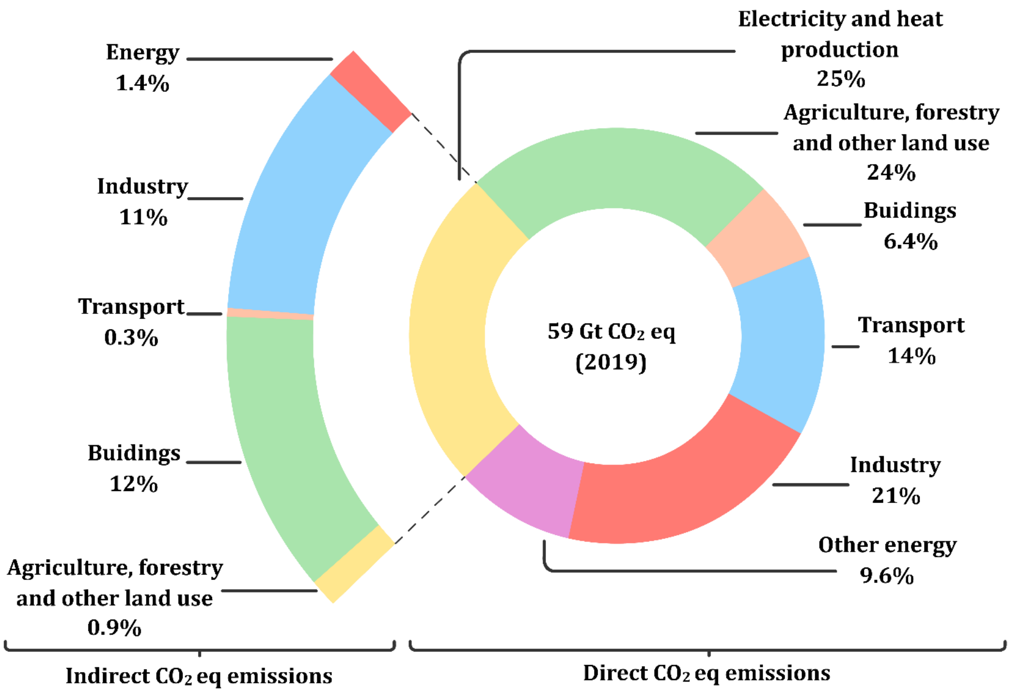


Figure 1.2. Total anthropogenic GHG emissions by economic sectors. Adapted from IPCC (2022).

The food supply chain consists of various activities such as procurement, production, transport, processing, distribution, consumption, and disposal (Davis et al., 2021). According to the European Commission, the food supply chain links the actors involved in the production, collection, processing, and distribution of food, including the consumer. The consumer may buy the food directly or may receive it through various steps of raw material processing, verification of safety standards, packaging, transportation, and other value-added processes (Chen et al., 2020). They are intricate networks consisting of companies of different sizes, such as small and medium-sized individual farmers, companies involved in transport, processing and marketing, as well as multinational companies in the retail sector (Kumar et al., 2022).

Among the most alarming environmental situations facing the food supply chain are the simultaneous increase in demand and depletion of water sources (Atallah et al., 2014), as well as the production of pre-consumer and post-consumer food waste (Alexander et al., 2017), and the energy demand to meet the needs of the entire food industry (Crippa et al., 2021). In this sense, it is important to assess environmental impacts from a broad approach, analysing different environmental scenarios, including climate change, water consumption, particulate emissions, ozone depletion, land use and raw material depletion, among others. In short, sustainable development is not possible without sustainable management of the food supply chain.

This thesis deals with different strategies for the environmental assessment of different sectors of great relevance, especially in Spain. First of all, the wine sector is presented, the Spanish grape growing and wine production enjoys an outstanding global position, being the world leader in vineyard surface, third producer and first exporter in volume (Muñoz et al., 2021). Its extensive value chain represents an important weight in the national economy, generating a total Gross Value Added of more than 23,700 million euros per year, equivalent to 2.2% of the national value (Lorenzo et al., 2018). One of the main objectives of the wine sector's roadmap is to adapt to a more environmentally friendly production that meets the challenges of climate change. Moreover, it should be borne in mind that the wine sector has an important tractor effect on many sectors, in particular the tourism sector where it has an essential role to play in increasing value. Furthermore, the very nature of viticulture makes it an activity that favors the fixation of population in rural areas and contributes to meeting the demographic challenge. The waste collected during the grape harvest and the winery stage as raw materials in the circular economy strategy is considered as a very representative example for its valorisation, an array of products could be generated from winery waste including antimicrobial compounds, food additives, biofuels, functional food, dietary supplement, nutraceuticals, medical remedies, animal feed and cosmetics (Ahmad et al., 2020; Portilla Rivera et al., 2021; Soceanu et al., 2021).

Another representative example of staple food is milk production. Dairy products are key to nutrition and health as they are nutrient-rich foods and provide energy and a large amount of protein and micronutrients, including calcium, magnesium, selenium, riboflavin, and vitamins B5 and B12. Milk ranks among the top five agricultural commodities in terms of both quantity and value (Bir et al., 2019). The dairy sector is growing rapidly: today, world milk production exceed 800 million tonnes and is projected to increase by 177 million tonnes by 2025, with an average growth rate of 1.8% per year per annum in the next 10 years (Kozłowski et al., 2019). In this context and taking into account the high environmental impact of the dairy sector, which is responsible for around 4% of anthropogenic GHGs emissions (de Léis et al., 2015), 10% of the global anthropogenic eutrophication and 6% of potential acidification (Gerber et al., 2011); it seems clear to think that the sustainable production, processing and consumption of milk and dairy products benefits people and the planet and helps achieve the Sustainable Development Goals.

Global fish production is estimated to have reached about 179 million tonnes in 2018 (FAO, 2020), with a total first sale value estimated at USD 401 billion. Of the overall total, 156 million tonnes were used for human consumption, equivalent to an estimated annual supply of 20.5 kg per capita. Fish products are a very important source of protein and an important part of a healthy diet. This, whilst average worldwide fish and seafood consumption was 20.3 kg/person·year in 2018, the EU average consumption was 21.6 kg/person·year. Within EU, fish consumption is led by Portugal and Spain, with 56.7 and 43.4 kg/person·year, respectively. The long-term trend in total global capture fisheries has been relatively stable since the late-1980s, with catches generally fluctuating between 86 million tonnes and 93 million tonnes per year. However, the latest date published in FAO (2020), showed that in 2018, total global capture fisheries production reached the highest level ever recorded at 96.4 million tonnes. This increase was mostly driven by marine capture fisheries, whose production increased from 81.2 million tonnes in 2017 to 84.4 million tonnes in 2018. Per nation, China accounted for about 15% of total global captures. The top seven capture producers (China, Indonesia, Peru, India, the Russian Federation, USA, and Vietnam) accounted for almost 50% of total global capture production. Small-scale fisheries and aquaculture make critical contributions to development in the areas of employment, with over 41 million people worldwide working in fish production. In this sense, As a sector that draws livelihoods from the oceans, has close connection to land and sea, and is socially and culturally embedded in the communities (Weeratunge et al., 2014), small-scale fisheries are not only one of the key actors in the governance for ocean sustainability, but can also play significant role in achieving the SDG1 - Food security, the SDG2 - Reduced poverty, he SDG3 - Community wellbeing, the SDG5 - Gender equality, and the SDG8 - Economic growth (Said and Chuenpagdee, 2019).

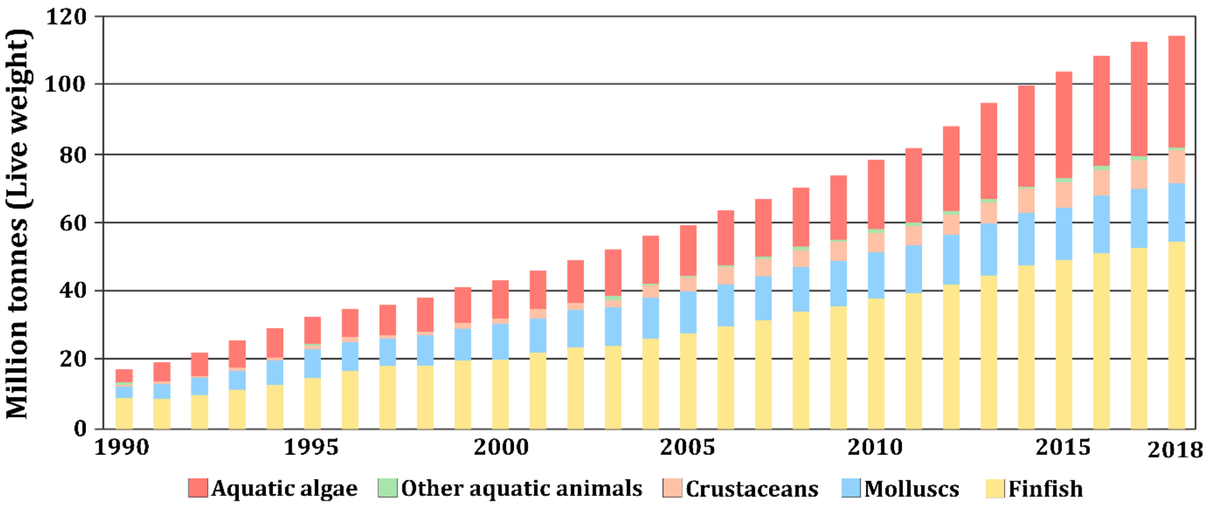


Figure 1.3. World aquaculture production of aquatic animals and algae, 1990-2018. Adapted from FAO (2020).

Finally, the aquaculture sector is the fastest growing food production sector in the world, with an average annual growth rate of 5.3% during the period 2001-2018. The latest data published by FAO (FAO, 2020) shows that world aquaculture production attained another all-time record high of 114.5 million tonnes in live weight in 2018 with a total farmgate sale value of USD 263.3 billion. The total production consisted of 82.1 million tonnes of aquatic animals and 32.4 million tonnes of aquatic algae (Figure 1.3).

The development of this activity is mainly taking place in developing countries, in fact world aquaculture production of farmed aquatic animals has been dominated by Asia, with an 89% share in the last two decades (Ahmed et al., 2019). Among major producing countries, Egypt, Chile, India, Indonesia, Vietnam, Bangladesh and Norway have consolidated their share in world production (FAO, 2020). In terms of production value, the white shrimp is in first place, followed by Atlantic salmon, Chilean carp, catfish and tilapia (Ahmed and Thompson, 2019).

1.1.3. Waste generation and the environment

According to the Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste, waste can be defined as any substance or object which the holder discards, intends to discard, or is obliged to discard. This legislation also establishes a waste hierarchy, i.e., an order of priority in waste prevention and management legislation and policy: (i) prevention; (ii) preparation for re-use; (iii) recycling; (iv) other recovery; and (v) disposal.

Waste management has become an issue of global concern due to its environmental implications and the high costs associated with waste management (Marshall and Farahbakhsh, 2013). The generation of Municipal Solid Waste (MSW) has increased considerably in recent years due to rapid urban population growth (Goorhuis, 2014). In fact, according to the latest data published by Eurostat, the total waste generated in the EU-28 by all economic activities and households amounted to more than 2.3 billion tonnes in 2018 (Eurostat, 2021). Data published in this database indicate an increase in the amount of waste recovered, in other words recycled, used for backfilling, or incinerated with energy recovery from 870 million tonnes in 2004 to 1,184 million tonnes in 2018. However, the amount of waste subject to disposal only decreased by 4.2% from 1,027 million tonnes in 2004 to 984 million tonnes in 2018 (Eurostat, 2021).

Waste management has been identified by the European Commission, the US Environmental Protection Agency and the Organisation for Economic Co-operation and Development, among other institutions, as a key issue for achieving a resource-efficient society and for achieving a sustainable economy. Improved waste management practices are needed to strengthen material recycling, close essential material loops, and also recover energy from waste, while ensuring that toxic substances are not released into the environment. This essentially means moving from a linear 'extraction-use-disposal' economic model to a more circular one (see more detail in Section 1.2), where waste from different sources is converted into raw materials for other activities, fostering what is often referred to as an industrial ecology, a smart economy, a green economy or a circular economy (Kumar et al., 2021).

However, to obtain raw materials or energy from waste streams and closing cycles, they need to be collected, transported, sorted, and finally processed, through recycling or energy recovery processes, which involve the consumption of additional water, energy, and materials as well. From an environmental point of view, raw materials, energy consumption and emissions during waste collection and treatment must be compared with the levels consumed and emitted during recycling or energy recovery processes. Therefore, a waste management strategy should only be considered efficient if the amount of resources used for waste management itself is less than the amount of energy and materials that can eventually be recovered from the waste. Under this premise, efficient planning of integrated solid waste management systems requires accounting for the full set of environmental effects associated with the entire life cycle of solid waste (Tonini et al., 2018), considering a holistic approach. In this sense, LCA is widely accepted as the best tool to provide such a global and extended view of the system, taking into account all the processes involved and their interactions with the economy (through the reintroduction of recycled materials or energy). recovered from the waste management system), as well as considering a full set of environmental effects, such as climate change, ozone layer depletion, acidification, or eutrophication.

1.2. TOWARDS A CIRCULAR AND SUSTAINABLE PRIMARY SECTOR

The transformation towards a sustainable food system requires a multitude of measures in various domains such as environment, economy and society to increase their resilience to future adverse events or shocks such as the recent COVID-19 pandemic (Fan et al., 2021) or the war in Ukraine (Kaiser, 2022). Although there is no universally agreed definition of what a sustainable and circular system is, there is broad agreement on what the outcomes of a sustainable food system should be. A sustainable, circular food system can be considered as a food system that provides food security and nutrition for all in such a way that the economic, social and environmental basis for generating food security and nutrition for future generations is not compromised (Herrero et al., 2020; Santibanez Gonzalez et al., 2019).

The transition to a circular and sustainable food system requires systemic change, in which regions play a key role. However, research on case studies at the regional level remains underexplored. More in-depth discussions on the opportunities and challenges presented by circularity are required to enable the realisation of circularity and sustainability in real-world systems (Vanhamäki et al., 2020).

1.2.1. Circular economy

Currently our productive sector is clearly dominated by a consumption system based on a linear take-made-dispose model (Ng et al., 2019), as detailed in Figure 1.4. Manufacturing industries obtain their resources from the natural environment through extraction and mining to make products and components (Kirchherr et al., 2017). After that, consumers would buy their products and components and dispose of them when they have reached the end of their useful life (Camilleri, 2019; Hao et al., 2020). However, too often both manufacturing and consumer waste is dumped in landfills or incinerated (Camilleri, 2020). Given that all projections indicate that the world's population is expected to continue to grow (Marques et al., 2019) and therefore the demand for natural resources will increase proportionally (Dong et al., 2018); there is a need to shift from the dominant linear take-make-dispose consumption system to a circular economy model that provides a new vision for solving sustainability challenges (Burke et al., 2021).

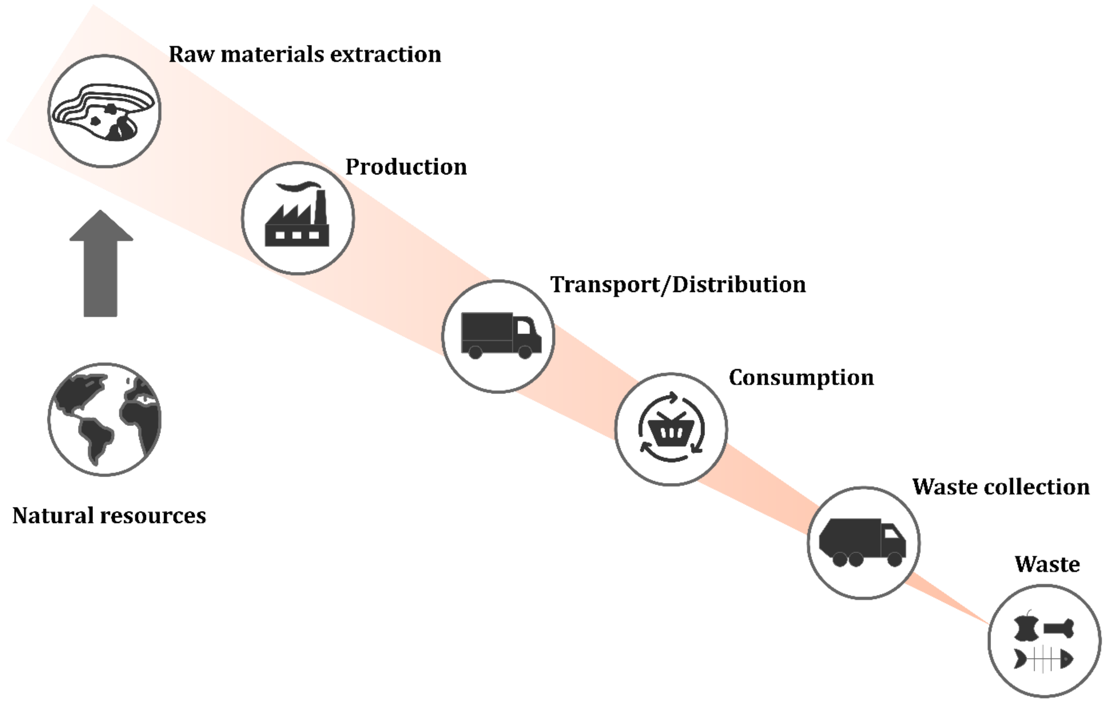


Figure 1.4. Linear model of consumption based on "take-make-dispose."

Thus, the concept of Circular Economy was born, as a circular model of consumption, where raw materials are extracted from natural resources but after consumption, the waste generated is revalued into a resource and reintegrated into the consumption system through recycling or reuse (Figure 1.5).

Although the concept of circular economy has evolved in recent years and there is no single definition (Neves and Marques, 2022), a widely accepted definition can be found in literature (Kirchherr et al., 2017), where circular economy is defined as "an economic system that replaces the "end-of-life" concept with reduction, alternative reuse, recycling and recovery of materials in production, distribution and consumption processes". It is clear that change requires that resources are used as efficiently as possible, with the best indicators of economic and environmental sustainability. In this sense, the management of surplus and waste streams must be addressed in the paradigm shift driven by the Circular Economy, what was once just waste can now have real value and we must find ways to make use of streams that were previously simply useless (Bigdeloo et al., 2021). This idea has become increasingly important, not only in academia but also in the political, economic, business, and social spheres. As defined in the literature, a circular economy is a positive continuous development cycle that preserves and enhances natural capital, optimises resource returns and minimises system risks by managing finite stocks and renewable flows (Halog and Anieke, 2021). It seems clear that applying the circular economy concept encourages environmental protection and social prosperity (Jawahir and Bradley, 2016), ensuring multiple mechanisms of generating new value that are separate from the exploitation of limited natural resources (Sverko Grdic et al., 2020).

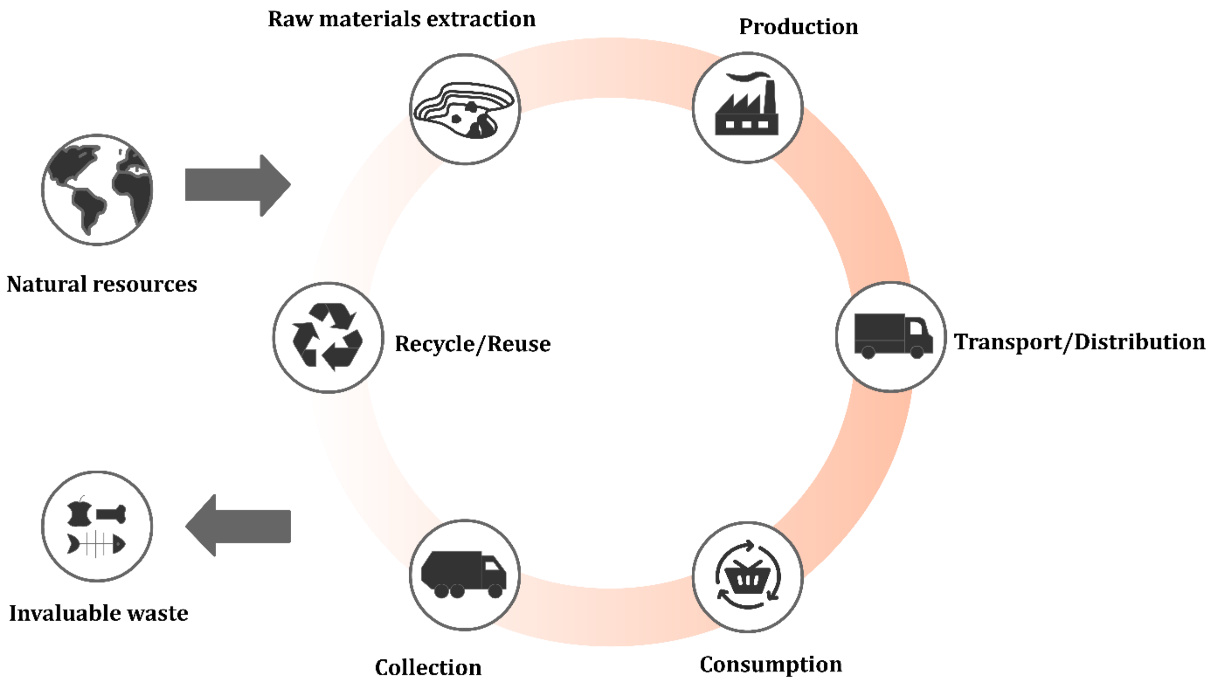


Figure 1.5. Concept of Circular economy, adapted from Sauvé et al., (2016).

Based on the concept and the principles that have been defined, the following key characteristics of a circular economy can be established (EEA, 2017): (i) Reduced inputs and reduced use of natural resources; (ii) Increased sharing of energy, renewable and recyclable resources, replacing non-renewable resources with renewable resources at sustainable supply levels; (iii) Reducing emissions throughout the entire materials cycle, through the use of fewer raw materials and sustainable sourcing of raw materials; (iv) Minimising losses of materials and waste, by limiting the amount of waste incinerated and landfilled; (v) Maintaining the value of products, components and materials in the economy, by extending their useful life and maintaining the value of products in use. In this transition towards a circular and sustainable economy, the engagement of all levels of society seems to be an essential aspect (Jawahir and Bradley, 2016). In this sense, social concern about the environmental problems arising from the massive consumption of natural resources materialised in the action plan of the United Nations 2030 agenda, which consisted of 17 Sustainable Development Goals (SDGs). These goals seek to address major environmental, but also social and economic challenges worldwide (Sachs et al., 2019). In this sense, there seems to be a strong link between the circular economy and the SDGs. The circular economy can be seen as a tool that can be used by different countries, social actors, and institutions to achieve some SDGs. In fact, the 2015 EU Action Plan for the Circular Economy indicated that it was intended to contribute to the achievement of some SDGs, specifically Goal 12, related to sustainable consumption and production patterns (Rodriguez-Anton et al., 2019).

1.2.2. Sustainable Development Goals

The Sustainable Development Goals (SDGs) are an action plan provided by the United Nations as part of its 2030 agenda for sustainable development for all countries, including rich and poor, to promote prosperity and protect the environment (United Nations, 2021). This plan recognises that ending poverty requires strategies that promote economic growth and address a range of social needs, such as education, health, social protection, and employment opportunities, while going hand in hand with protecting the environment and addressing climate change, providing a comprehensive vision for a sustainable and equitable society. Figure 1.6 shows the 17 SDGs included in the United Nations 2030 Agenda for Sustainable Development.

Figure 1.6. Sustainable Development Goals included in the 2030 Agenda for Sustainable Development of the United Nations, adapted from United Nations (2015).

A set of motivations for business when considering engagement with the SDGs can be identified: those that positively drive growth, by providing opportunities for innovation, market development, cost saving, efficiency, and brand building. Another set of motivations are what might be called potential growth 'limiters', where the SDGs outline risks to be managed and negative impacts to be reduced, mitigated, or eliminated. If not managed properly, these might negatively impact growth.

- **Innovation & market development:** The SDGs essentially highlight and address the huge gaps in development that exist across the globe – whether it be lack of access to finance, clean water, food, or education. These 'development' gaps also represent unmet market needs. As governments direct their policy and resource towards meeting these needs, businesses can benefit from analysing which of these may present opportunities for innovating new products, services, and business models. None of these challenges are "new news" and as such there are numerous examples of companies already creating value through addressing these unmet needs.

- **Efficiency & cost savings:** Many of the SDG targets aim to tackle the pressures on the environmental system and encourage economic growth within planetary limits. The impact of limited natural resources will increasingly be felt by businesses through rising costs and volatile supply. The impacts of climate change will exacerbate this effect. Doing more with less will become an imperative for all business, helping to save costs in the short-term and reduce risk in the long-term.

- **Reputation management:** Some SDGs clearly point to the elimination of the negative impacts that result from the activities of the private sector: pollution, environmental degradation, bribery, corruption, forced labour, child labour. Tackling these issues is vital to building and maintaining trust with key stakeholders and maintaining a company's licence to operate. Neglect of these issues, whether in your own operations or further along the value chain can strike a damaging blow at reputation, sometimes undermining the viability of a business.

- **Risk reduction:** The SDGs aim to tackle many issues which pose significant risks to 'business as usual' over a longer time frame. These might be risks in the supply chain or financial, regulatory or technological risks, to name a few. Businesses can minimise their vulnerabilities by understanding the way the Goals impact on their sector and value chain. The SDGs will also increasingly shape the regulatory environment for business. Businesses that respond proactively will increase their resilience in a challenging operational and regulatory context.

Farming, fisheries & plantations, listed under the sector Consumer Goods, will be expected to play a large role in SDG 2 to achieve zero hunger and SDG 14 to protect the marine environment. Focusing on food production, sustainable food systems not only help end hunger but can also help the world make critical progress on the 17 Sustainable Development Goals (Lemaire and Limbourg, 2019).

Focusing on the sector Consumer Goods, which includes sub-sectors as automobiles, food & beverage, personal & household goods, farming & fisheries and clothing & footwear, this sector feeds almost all other sectors further down the economic value chain. With the focus on extraction and production of raw materials, these companies have a high impact on the natural environment. The sector also often operates in areas of conflict or jurisdictions where there is poor enforcement of labour, environmental or business integrity standards. It is therefore exposed to greater risk from malpractice with regards to bribery & corruption, child labour, poor health & safety, worker exploitation, as well as pollution. The potential growth limiters for this sector can be characterised as: (i) Innovation & market development: New products and serviced that address environment & health challenges, directly connected with SDGs 1, 8, 9 and 12; (ii) Efficiency & cost savings: Operational efficiency can significantly reduce costs, linked with SDGs 6, 7, 9 and 12; (iii) Reputation management: Manage potential negative social and environmental impacts associated with the sector as health effect of chemicals, labour rights, child labour or pollution, directly linked with SDGs 3, 6, 8, 14, 15 and 16 and; (iv) Risk reduction: Promote sustainability management of depleting natural resources, connected with SDGs 3, 12 and 15.

1.3. THESIS OUTLINE: OBJECTIVES AND STRUCTURE

The main objective of this PhD thesis is to assess the sustainability of some alternatives based on the circular economy applied to the agricultural and fisheries sectors from an environmental point of view. In addition, the eco-efficiency of industrial processes for milk and shrimp production has also been evaluated. With all this, this document has been structured in five main sections with their respective chapters, as shown in Figure 1.7.

Section I: Contextualisation. This section aims to provide potential readers with an overview of the state of the art of the problem addressed in this thesis, as well as the environmental assessment tools used for this purpose. In this regard, **Chapter 1** focuses on the state of the art of the field, including the problems that society must face in the future to provide food security for a continuously growing population and the potential of the primary sector to seek solutions based on the circular economy. **Chapter 2** presents the environmental assessment tools used for sustainability analysis of circular economy-based solutions for primary sectors, including the Life Cycle Assessment (LCA) methodology, the Data Envelopment Analysis (DEA) assessment tool and a methodology combining both to assess the eco-efficiency of industrial operations.

Section II: Circular economy in the agricultural sector. This section focuses on the study of the environmental sustainability of waste recovery in the wine sector. In **Chapter 3**, a biorefinery configuration for the valorisation of wine lees, producing different value-added products is evaluated. Then, in **Chapter 4**, a different approach to valorise grape marc by vermicomposting to produce different outputs was considered, including a nutrient-rich biofertiliser, an extract rich in polyphenols and fatty acids-rich oil rich. **Chapter 5** aims to go one step further by valorising the alcoholic waste streams, which can come as outputs from the previous chapters, so that through steam reforming and a Solid Oxide Fuell Cell (SOFC), clean and sustainable electricity can be produced.

SECTION I - CONTEXTUALISATION

Chapter 1 - State-of-the-art
Chapter 2 - Environmental management tools

SECTION II - CIRCULAR ECONOMY IN THE AGRICULTURAL SECTOR

Chapter 3 - Integrated evaluation of wine lees valorisation to produce value-added products
Chapter 4 - Unravelling the environmental impacts of bioactive compounds and organic amendment from grape marc
Chapter 5 - Environmental implications of biohydrogen-based energy production from steam reforming of alcoholic waste

SECTION III - CIRCULAR ECONOMY IN THE FISHERIES SECTOR

Chapter 6 - Evaluation of the environmental sustainability of the inshore great scallop (*Pecten maximus*) fishery in Galicia
Chapter 7 - Multi-product strategy to enhance the environmental profile of the canning industry towards circular economy

SECTION IV - ECO-EFFICIENCY ASSESSMENT

Chapter 8 - Pursuing the route to eco-efficiency in dairy production: the case of Galician area
Chapter 9 - Eco-efficiency assessment of shrimp aquaculture production in Mexico

SECTION V - GENERAL CONCLUSIONS

Chapter 10 - General findings and conclusions of the thesis

Figure 1.7. Outline of the doctoral thesis.

Section III: Circular economy in the fisheries sector. In this section environmental management in the fisheries sector is considered as a policy or strategy that improves the environmental profile of the productive sector. Environmental indicators are used to design improvement actions and to try to take advantage of more products for a greater valorisation of flows with potential to create added value. Two case studies of the fishing sector were evaluated form an environmental point of view. In this way, **Chapter 6** is the first environmental study of the scallop (*Peten maximus*) fishery in the Ría de Arousa and its subsequent processing in the port of Cambados through the construction of a detailed inventory with primary data. In second place, **Chapter 7** is focused on the environmental profile of a canned product, which represents a clear example of circular economy, since co-products from previous stages are traditionally used to produce other outputs.

Section IV: Eco-efficiency assessment. After having studied different case studies in two primary sectors, Data Envelopment Analysis was used to assess the eco-efficiency of industrial exploitations of each sector: dairy and aquaculture. In these productive sectors in which there are numerous and different process units, ranking them allows to identify better and worse alternatives, so that the operational practices of the best systems can serve as a reference for others, making it possible to identify which actions and strategies can dynamically improve the general process. For this purpose, **Chapter 8** evaluates the eco-efficiency of Galician milk production by assessing more than 100

Galician dairy farms, also calculating the carbon and water footprints of each of them to give a global perspective of the whole Galician sector. On the other hand, **Chapter 9** is focused on shrimp aquaculture production in Mexico, a sector that is still under development and has much room for improvement.

Section V: General conclusions. Finally, **Chapter 10** summarizes all the general findings, results, and conclusions of the thesis.

1.4. REFERENCES

Ahmad, B., Yadav, V., Yadav, A., Rahman, M.U., Yuan, W.Z., Li, Z., Wang, X., 2020. Integrated biorefinery approach to valorize winery waste: A review from waste to energy perspectives. Sci. Total Environ. 719, 137315. https://doi.org/10.1016/j.scitotenv.2020.137315

Ahmed, N., Thompson, S., 2019. The blue dimensions of aquaculture: A global synthesis. Sci. Total Environ. 652, 851–861. https://doi.org/10.1016/j.scitotenv.2018.10.163

Ahmed, N., Thompson, S., Glaser, M., 2019. Global Aquaculture Productivity, Environmental Sustainability, and Climate Change Adaptability. Environ. Manage. 63, 159–172. https://doi.org/10.1007/s00267-018-1117-3

Alexander, P., Brown, C., Arneth, A., Finnigan, J., Moran, D., Rounsevell, M.D.A., 2017. Losses, inefficiencies and waste in the global food system. Agric. Syst. 153, 190–200. https://doi.org/10.1016/j.agsy.2017.01.014

Anderson, R., Bayer, P.E., Edwards, D., 2020. Climate change and the need for agricultural adaptation. Curr. Opin. Plant Biol. 56, 197–202. https://doi.org/10.1016/j.pbi.2019.12.006

Asseng, S., Ewert, F., Martre, P., Rötter, R.P., Lobell, D.B., Cammarano, D., Kimball, B.A., Ottman, M.J., Wall, G.W., White, J.W., Reynolds, M.P., Alderman, P.D., Prasad, P.V. V, Aggarwal, P.K., Anothai, J., Basso, B., Biernath, C., Challinor, A.J., De Sanctis, G., Doltra, J., Fereres, E., Garcia-Vila, M., Gayler, S., Hoogenboom, G., Hunt, L.A., Izaurralde, R.C., Jabloun, M., Jones, C.D., Kersebaum, K.C., Koehler, A.-K., Müller, C., Naresh Kumar, S., Nendel, C., O'Leary, G., Olesen, J.E., Palosuo, T., Priesack, E., Eyshi Rezaei, E., Ruane, A.C., Semenov, M.A., Shcherbak, I., Stöckle, C., Stratonovitch, P., Streck, T., Supit, I., Tao, F., Thorburn, P.J., Waha, K., Wang, E., Wallach, D., Wolf, J., Zhao, Z., Zhu, Y., 2015. Rising temperatures reduce global wheat production. Nat. Clim. Chang. 5, 143–147. https://doi.org/10.1038/nclimate2470

Atallah, S.S., Gómez, M.I., Björkman, T., 2014. Localization effects for a fresh vegetable product supply chain: Broccoli in the eastern United States. Food Policy 49, 151–159. https://doi.org/10.1016/j.foodpol.2014.07.005

Barbieri, P., MacDonald, G.K., Bernard de Raymond, A., Nesme, T., 2022. Food system resilience to phosphorus shortages on a telecoupled planet. Nat. Sustain. 5, 114–122. https://doi.org/10.1038/s41893-021-00816-1

Bassu, S., Brisson, N., Durand, J.-L., Boote, K., Lizaso, J., Jones, J.W., Rosenzweig, C., Ruane, A.C., Adam, M., Baron, C., Basso, B., Biernath, C., Boogaard, H., Conijn, S., Corbeels, M., Deryng, D., De Sanctis, G., Gayler, S., Grassini, P., Hatfield, J., Hoek, S., Izaurralde, C., Jongschaap, R., Kemanian, A.R., Kersebaum, K.C., Kim, S.-H., Kumar, N.S., Makowski, D., Müller, C., Nendel, C., Priesack, E., Pravia, M.V., Sau, F., Shcherbak, I., Tao, F., Teixeira, E., Timlin, D., Waha, K., 2014. How do various maize crop models vary in their responses to climate change factors? Glob. Chang. Biol. 20, 2301–2320. https://doi.org/10.1111/gcb.12520

Bigdeloo, M., Teymourian, T., Kowsari, E., Ramakrishna, S., Ehsani, A., 2021. Sustainability and Circular Economy of Food Wastes: Waste Reduction Strategies, Higher Recycling Methods, and Improved Valorization. Mater. Circ. Econ. 3, 3. https://doi.org/10.1007/s42824-021-00017-3

Bir, C., Widmar, N.O., Wolf, C., Delgado, M.S., 2019. Traditional attributes moo-ve over for some consumer segments: Relative ranking of fluid milk attributes. Appetite 134, 162–171. https://doi.org/10.1016/j.appet.2018.12.007

Boretti, A., Rosa, L., 2019. Reassessing the projections of the World Water Development Report. npj Clean Water 2, 15. https://doi.org/10.1038/s41545-019-0039-9

Bretschger, L., Pittel, K., 2020. Twenty Key Challenges in Environmental and Resource Economics. Environ. Resour. Econ. 77, 725–750. https://doi.org/10.1007/s10640-020-00516-y

Burke, H., Zhang, A., Wang, J.X., 2021. Integrating product design and supply chain management for a circular economy. Prod. Plan. Control 1–17. https://doi.org/10.1080/09537287.2021.1983063

Camilleri, M.A., 2020. European environment policy for the circular economy: Implications for business and industry stakeholders. Sustain. Dev. 28, 1804–1812. https://doi.org/10.1002/sd.2113

Camilleri, M.A., 2019. The circular economy's closed loop and product service systems for sustainable development: A review and appraisal. Sustain. Dev. 27, 530–536. https://doi.org/10.1002/sd.1909

Carpenter, S.R., Mooney, H.A., Agard, J., Capistrano, D., DeFries, R.S., Díaz, S., Dietz, T., Duraiappah, A.K., Oteng-Yeboah, A., Pereira, H.M., Perrings, C., Reid, W. V., Sarukhan, J., Scholes, R.J., Whyte, A., 2009. Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. Proc. Natl. Acad. Sci. 106, 1305–1312. https://doi.org/10.1073/pnas.0808772106

Crippa, M., Solazzo, E., Guizzardi, D., Tubiello, F.N., Leip, A., 2021. Food system are responsible for a third of global GHG emissions. Nat. Food. https://doi.org/10.1038/s43016-021-00225-9

D'Odorico, P., Davis, K.F., Rosa, L., Carr, J.A., Chiarelli, D., Dell'Angelo, J., Gephart, J., MacDonald, G.K., Seekell, D.A., Suweis, S., Rulli, M.C., 2018. The Global Food-Energy-Water Nexus. Rev. Geophys. 56, 456–531. https://doi.org/10.1029/2017RG000591

de Léis, C.M., Cherubini, E., Ruviaro, C.F., da Silva, V.P., Lampert, V. do N., Spies, A., Soares, S.R., 2015. Carbon footprint of milk production in Brazil: a comparative case study. Int. J. Life Cycle Assess. 20, 46–60. https://doi.org/10.1007/s11367-014-0813-3

Dong, K., Hochman, G., Zhang, Y., Sun, R., Li, H., Liao, H., 2018. CO_2 emissions, economic and population growth, and renewable energy: Empirical evidence across regions. Energy Econ. 75, 180–192. https://doi.org/10.1016/j.eneco.2018.08.017

Dyllick, T., Hockerts, K., 2002. Beyond the business case for corporate sustainability. Bus. Strateg. Environ. 11, 130–141. https://doi.org/10.1002/bse.323

Earth Overshoot Day, 2022. Earth Overshoot Day. URL: https://www.overshootday.org/

EEA, 2017. Circylar by design. Products in the circular economy. EEA report nº6/2017.

Erisman, J.W., Sutton, M.A., Galloway, J., Klimont, Z., Winiwarter, W., 2008. How a century of ammonia synthesis changed the world. Nat. Geosci. 1, 636–639. https://doi.org/10.1038/ngeo325

Eurostat, 2021. Waste statistics. Waste generation, 2018. URL: https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Waste_statistics#Total_waste_generation

Fan, S., Teng, P., Chew, P., Smith, G., Copeland, L., 2021. Food system resilience and COVID-19 – Lessons from the Asian experience. Glob. Food Sec. 28, 100501. https://doi.org/10.1016/j.gfs.2021.100501

FAO, 2020. The State of World Fisheries and Aquaculture - Sustainability in action. Food and Agriculture Organization of the United Nations, Rome.

Gerber, P., Vellinga, T., Opio, C., Steinfeld, H., 2011. Productivity gains and greenhouse gas emissions intensity in dairy systems. Livest. Sci. 139, 100–108. https://doi.org/10.1016/j.livsci.2011.03.012

Gills, B., Morgan, J., 2021. Economics and climate emergency. Globalizations 18, 1071–1086. https://doi.org/10.1080/14747731.2020.1841527

Goorhuis, M., 2014. Developments in Collection of Municipal Solid Waste, in: Worrell, E., Reuter, M.A. (Eds.), Handbook of Recycling: State-of-the-Art for Practitioners, Analysts, and Scientists. Elsevier Inc., pp. 405–417. https://doi.org/10.1016/B978-0-12-396459-5.00026-X

Halog, A., Anieke, S., 2021. A Review of Circular Economy Studies in Developed Countries and Its Potential Adoption in Developing Countries. Circ. Econ. Sustain. 1, 209–230. https://doi.org/10.1007/s43615-021-00017-0

Hao, Y., Wang, Y., Wu, Q., Sun, S., Wang, W., Cui, M., 2020. What affects residents' participation in the circular economy for sustainable development? Evidence from China. Sustain. Dev. 28, 1251–1268. https://doi.org/10.1002/sd.2074

Herrero, M., Thornton, P.K., Mason-D'Croz, D., Palmer, J., Benton, T.G., Bodirsky, B.L., Bogard, J.R., Hall, A., Lee, B., Nyborg, K., Pradhan, P., Bonnett, G.D., Bryan, B.A., Campbell, B.M., Christensen, S., Clark, M., Cook, M.T., de Boer, I.J.M., Downs, C., Dizyee, K., Folberth, C., Godde, C.M., Gerber, J.S., Grundy, M., Havlik, P., Jarvis, A., King, R., Loboguerrero, A.M., Lopes, M.A., McIntyre, C.L., Naylor, R., Navarro, J., Obersteiner, M., Parodi, A., Peoples, M.B., Pikaar, I., Popp, A., Rockström, J., Robertson, M.J., Smith, P., Stehfest, E., Swain, S.M., Valin, H., van Wijk, M., van Zanten, H.H.E., Vermeulen, S., Vervoort, J., West, P.C., 2020. Innovation can accelerate the transition towards a sustainable food system. Nat. Food 1, 266–272. https://doi.org/10.1038/s43016-020-0074-1

Hunter, M.C., Smith, R.G., Schipanski, M.E., Atwood, L.W., Mortensen, D.A., 2017. Agriculture in 2050: Recalibrating Targets for Sustainable Intensification. Bioscience 67, 386–391. https://doi.org/10.1093/biosci/bix010

IPCC, 2022. Climate change 2022. Mitigation of Climate Change. Summary for Policymakers.

Jambeck, J.R., Geyer, R., Wilcox, C., Siegler, T.R., Perryman, M., Andrady, A., Narayan, R., Law, K.L., 2015. Plastic waste inputs from land into the ocean. Science (80-.). 347, 768–771. https://doi.org/10.1126/science.1260352

Jawahir, I.S., Bradley, R., 2016. Technological Elements of Circular Economy and the Principles of 6R-Based Closed-loop Material Flow in Sustainable Manufacturing. Procedia CIRP 40, 103–108. https://doi.org/10.1016/j.procir.2016.01.067

Kaiser, M., 2022. A Personal Editorial from the Editor-in-Chief: Food Ethics in Times of War. Food Ethics 7, 9. https://doi.org/10.1007/s41055-022-00103-5

Kehrberger, S., Holzschuh, A., 2019. Warmer temperatures advance flowering in a spring plant more strongly than emergence of two solitary spring bee species. PLoS One 14, e0218824. https://doi.org/10.1371/journal.pone.0218824

Khan, I., Shah, D., Shah, S.S., 2021. COVID-19 pandemic and its positive impacts on environment: an updated review. Int. J. Environ. Sci. Technol. 18, 521–530. https://doi.org/10.1007/s13762-020-03021-3

Kirchherr, J., Reike, D., Hekkert, M., 2017. Conceptualizing the circular economy: An analysis of 114 definitions. Resour. Conserv. Recycl. 127, 221–232. https://doi.org/10.1016/j.resconrec.2017.09.005

Kozłowski, K., Pietrzykowski, M., Czekała, W., Dach, J., Kowalczyk-Juśko, A., Jóźwiakowski, K., Brzoski, M., 2019. Energetic and economic analysis of biogas plant with using the dairy industry waste. Energy 183, 1023–1031. https://doi.org/10.1016/j.energy.2019.06.179

Kumar, A., Mangla, S.K., Kumar, P., 2022. An integrated literature review on sustainable food supply chains: Exploring research themes and future directions. Sci. Total Environ. 821, 153411. https://doi.org/10.1016/j.scitotenv.2022.153411

Kumar, Rakesh, Verma, A., Shome, A., Sinha, R., Sinha, S., Jha, P.K., Kumar, Ritesh, Kumar, P., Shubham, Das, S., Sharma, P., Vara Prasad, P. V, 2021. Impacts of Plastic Pollution on Ecosystem Services, Sustainable Development Goals, and Need to Focus on Circular Economy and Policy Interventions. Sustain. . https://doi.org/10.3390/su13179963

Kumari, P., Toshniwal, D., 2020. Impact of lockdown on air quality over major cities across the globe during COVID-19 pandemic. Urban Clim. 34, 100719. https://doi.org/10.1016/j.uclim.2020.100719

Leal Filho, W., Tripathi, S.K., Andrade Guerra, J.B.S.O.D., Giné-Garriga, R., Orlovic Lovren, V., Willats, J., 2019. Using the sustainable development goals towards a better understanding of sustainability challenges. Int. J. Sustain. Dev. World Ecol. 26, 179–190. https://doi.org/10.1080/13504509.2018.1505674

Lee, Y.-J., Chai, L., Wu, P.-S., 2021. Taiwan's ecological footprint and overshoot day. Sci. Rep. 11, 15068. https://doi.org/10.1038/s41598-021-94540-7

Lemaire, A., Limbourg, S., 2019. How can food loss and waste management achieve sustainable development goals? J. Clean. Prod. 234, 1221–1234. https://doi.org/10.1016/j.jclepro.2019.06.226

Liu, B., Asseng, S., Müller, C., Ewert, F., Elliott, J., Lobell, D.B., Martre, P., Ruane, A.C., Wallach, D., Jones, J.W., Rosenzweig, C., Aggarwal, P.K., Alderman, P.D., Anothai, J., Basso, B., Biernath, C., Cammarano, D., Challinor, A., Deryng, D., Sanctis, G.D., Doltra, J., Fereres, E., Folberth, C., Garcia-Vila, M., Gayler, S., Hoogenboom, G., Hunt, L.A., Izaurralde, R.C., Jabloun, M., Jones, C.D., Kersebaum, K.C., Kimball, B.A., Koehler, A.-K., Kumar, S.N., Nendel, C., O'Leary, G.J., Olesen, J.E., Ottman, M.J., Palosuo, T., Prasad, P.V.V., Priesack, E., Pugh, T.A.M., Reynolds, M., Rezaei, E.E., Rötter, R.P., Schmid, E., Semenov, M.A., Shcherbak, I., Stehfest, E., Stöckle, C.O., Stratonovitch, P., Streck, T., Supit, I., Tao, F., Thorburn, P., Waha, K., Wall, G.W., Wang, E., White, J.W., Wolf, J., Zhao, Z., Zhu, Y., 2016. Similar estimates of temperature impacts on global wheat yield by three independent methods. Nat. Clim. Chang. 6, 1130–1136. https://doi.org/10.1038/nclimate3115

Lokhandwala, S., Gautam, P., 2020. Indirect impact of COVID-19 on environment: A brief study in Indian context. Environ. Res. 188, 109807. https://doi.org/10.1016/j.envres.2020.109807

Lorenzo, J.R.F., Rubio, M.T.M., Garcés, S.A., 2018. The competitive advantage in business, capabilities and strategy. What general performance factors are found in the Spanish wine industry? Wine Econ. Policy 7, 94–108. https://doi.org/10.1016/j.wep.2018.04.001

Marques, A., Martins, I.S., Kastner, T., Plutzar, C., Theurl, M.C., Eisenmenger, N., Huijbregts, M.A.J., Wood, R., Stadler, K., Bruckner, M., Canelas, J., Hilbers, J.P., Tukker, A., Erb, K., Pereira, H.M., 2019. Increasing impacts of land use on biodiversity and carbon sequestration driven by population and economic growth. Nat. Ecol. Evol. 3, 628–637. https://doi.org/10.1038/s41559-019-0824-3

Marshall, R.E., Farahbakhsh, K., 2013. Systems approaches to integrated solid waste management in developing countries. Waste Manag. 33, 988–1003. https://doi.org/10.1016/j.wasman.2012.12.023

Mišík, M., 2022. The EU needs to improve its external energy security. Energy Policy 165, 112930. https://doi.org/10.1016/j.enpol.2022.112930

Muñoz, R.M., Fernández, M. V, Salinero, Y., 2021. Sustainability, Corporate Social

Responsibility, and Performance in the Spanish Wine Sector. Sustain. . https://doi.org/10.3390/su13010007

Nemecek, T., Jungbluth, N., Milà i Canals, L., Schenck, R., 2016. Environmental impacts of food consumption and nutrition: where are we and what is next? Int. J. Life Cycle Assess. 21, 607–620. https://doi.org/10.1007/s11367-016-1071-3

Neves, S.A., Marques, A.C., 2022. Drivers and barriers in the transition from a linear economy to a circular economy. J. Clean. Prod. 341, 130865. https://doi.org/10.1016/j.jclepro.2022.130865

Ng, K.S., Yang, A., Yakovleva, N., 2019. Sustainable waste management through synergistic utilisation of commercial and domestic organic waste for efficient resource recovery and valorisation in the UK. J. Clean. Prod. 227, 248–262. https://doi.org/10.1016/j.jclepro.2019.04.136

Parajuli, R., Thoma, G., Matlock, M.D., 2019. Environmental sustainability of fruit and vegetable production supply chains in the face of climate change: A review. Sci. Total Environ. 650, 2863–2879. https://doi.org/10.1016/j.scitotenv.2018.10.019

Peters, G.P., Andrew, R.M., Canadell, J.G., Friedlingstein, P., Jackson, R.B., Korsbakken, J.I., Le Quéré, C., Peregon, A., 2020. Carbon dioxide emissions continue to grow amidst slowly emerging climate policies. Nat. Clim. Chang. 10, 3–6. https://doi.org/10.1038/s41558-019-0659-6

Poore, J., Nemecek, T., 2018. Reducing food's environmental impacts through producers and consumers. Science (80). 360, 987–992. https://doi.org/10.1126/science.aaq0216

Portilla Rivera, O.M., Saavedra Leos, M.D., Solis, V.E., Domínguez, J.M., 2021. Recent trends on the valorization of winemaking industry wastes. Curr. Opin. Green Sustain. Chem. 27, 100415. https://doi.org/10.1016/j.cogsc.2020.100415

Pugh, T.A.M., Müller, C., Elliott, J., Deryng, D., Folberth, C., Olin, S., Schmid, E., Arneth, A., 2016. Climate analogues suggest limited potential for intensification of production on current croplands under climate change. Nat. Commun. 7, 12608. https://doi.org/10.1038/ncomms12608

Ritchie, H., Reay, D.S., Higgins, P., 2018. The impact of global dietary guidelines on climate change. Glob. Environ. Chang. 49, 46–55. https://doi.org/10.1016/j.gloenvcha.2018.02.005

Rodriguez-Anton, J.M., Rubio-Andrada, L., Celemín-Pedroche, M.S., Alonso-Almeida, M.D.M., 2019. Analysis of the relations between circular economy and sustainable development goals. Int. J. Sustain. Dev. World Ecol. 26, 708–720. https://doi.org/10.1080/13504509.2019.1666754

Sachs, J.D., Schmidt-Traub, G., Mazzucato, M., Messner, D., Nakicenovic, N., Rockström, J., 2019. Six Transformations to achieve the Sustainable Development Goals. Nat. Sustain. 2, 805–814. https://doi.org/10.1038/s41893-019-0352-9

Said, A., Chuenpagdee, R., 2019. Aligning the sustainable development goals to the small-scale fisheries guidelines: A case for EU fisheries governance. Mar. Policy 107, 103599. https://doi.org/10.1016/j.marpol.2019.103599

Santibanez Gonzalez, E.D.R., Koh, L., Leung, J., 2019. Towards a circular economy production system: trends and challenges for operations management. Int. J. Prod. Res. 57, 7209–7218. https://doi.org/10.1080/00207543.2019.1656844

Sarkodie, S.A., Owusu, P.A., 2021. Impact of COVID-19 pandemic on waste management. Environ. Dev. Sustain. 23, 7951–7960. https://doi.org/10.1007/s10668-020-00956-y

Sauvé, S., Bernard, S., Sloan, P., 2016. Environmental sciences, sustainable development and circular economy: Alternative concepts for trans-disciplinary research. Environ. Dev. 17, 48–56. https://doi.org/10.1016/j.envdev.2015.09.002

Schaltegger, S., 2018. Linking Environmental Management Accounting: A Reflection on (Missing) Links to Sustainability and Planetary Boundaries. Soc. Environ. Account. J. 38, 19–29. https://doi.org/10.1080/0969160X.2017.1395351

Soceanu, A., Dobrinas, S., Sirbu, A., Manea, N., Popescu, V., 2021. Economic aspects of waste recovery in the wine industry. A multidisciplinary approach. Sci. Total Environ. 759, 143543. https://doi.org/10.1016/j.scitotenv.2020.143543

Sverko Grdic, Z., Krstinic Nizic, M., Rudan, E., 2020. Circular Economy Concept in the Context of Economic Development in EU Countries. Sustain. . https://doi.org/10.3390/su12073060

Tonini, D., Albizzati, P.F., Astrup, T.F., 2018. Environmental impacts of food waste: Learnings and challenges from a case study on UK. Waste Manag. 76, 744–766. https://doi.org/10.1016/j.wasman.2018.03.032

United Nations, 2021. The Sustainable Development Goals report 2021.

United Nations, 2015. Transforming our world: the 2030 Agenda for Sustainable Development. A/RES/70/1. UN General Assembly. https://doi.org/10.1007/s13398-014-0173-7.2

Vanhamäki, S., Virtanen, M., Luste, S., Manskinen, K., 2020. Transition towards a circular economy at a regional level: A case study on closing biological loops. Resour. Conserv. Recycl. 156, 104716. https://doi.org/10.1016/j.resconrec.2020.104716

Wang, Q., Su, M., 2020. A preliminary assessment of the impact of COVID-19 on environment–A case study of China. Sci. Total Environ. 728, 138915. https://doi.org/10.1016/j.scitotenv.2020.138915

Weeratunge, N., Béné, C., Siriwardane, R., Charles, A., Johnson, D., Allison, E.H., Nayak, P.K., Badjeck, M.-C., 2014. Small-scale fisheries through the wellbeing lens. Fish Fish. 15, 255–279. https://doi.org/10.1111/faf.12016

Will, S., Johan, R., Katherine, R., Timothy, M., Carl, F., Diana, L., Colin, P., Anthony, D., Sarah, E., Michel, C., Jonathan, F., Ingo, F., Steven, J., Marten, S., Ricarda, W., Joachim, S.H., 2018. Trajectories of the Earth System in the Anthropocene. Proc. Natl. Acad. Sci. 115, 8252–8259. https://doi.org/10.1073/pnas.1810141115

Wu, X., Zhang, C., Du, W., 2021. An Analysis on the Crisis of "Chips shortage" in Automobile Industry—Based on the Double Influence of COVID-19 and Trade Friction. J. Phys. Conf. Ser. 1971, 12100. https://doi.org/10.1088/1742-6596/1971/1/012100

Yokokawa, N., Kikuchi-Uehara, E., Sugiyama, H., Hirao, M., 2018. Framework for analyzing the effects of packaging on food loss reduction by considering consumer behavior. J. Clean. Prod. 174, 26–34. https://doi.org/10.1016/j.jclepro.2017.10.242

Chapter 2

Environmental management tools

Summary

Sustainable development is a priority issue for society and its institutions. In a context of continuous socioeconomic and technological development, together with demographic growth and the intensification of anthropogenic activities, it is necessary to define new production and consumption patterns, together with the developing of new processes to reduce raw materials consumption. A major goal for the primary sector is to identify and understand the interactions and feedbacks among human and natural processes that regulate the biosphere and consequently its capacity for providing favourable conditions for complex human societies.

Consequently, a wide set of environmental methodologies have been developed in recent years to bring together environmental protection, economic development, and social welfare. The main objective of this chapter is therefore to provide an overview of the main methodological tools available for the environmental analysis and sustainability assessment of innovative waste valorisation strategies and application of the circular economy to strategic primary sectors (mostly agriculture and fisheries).

In this sense, special attention has been paid to the use of Life Cycle Assessment (LCA), as it is considered the best option to implement in a wide variety of products and processes, while becoming the most widely accepted tool in the scientific community. Additionally, the use of Data Envelopment Analysis (DEA) allows the estimation of efficient practices in similar production systems, so the combination of LCA and DEA provides quantitative life-cycle-based benchmarks tat orientate the performance towards environmental sustainability.

OUTLINE

2.1. SUSTAINABLE DEVELOPMENT

The concept of sustainability has always been an important topic of debate among the scientific community due to the growing awareness of the impact of human activity on the environment and the possible ecological crisis in the near future (Stanitsas et al., 2021). Sustainable development was defined as a concept by the Brundtland Commission report entitled "Our Common Future" in 1987 and formed the basis of the United Nations Conference on Environment and Development held in Rio de Janeiro in 1992. According to this report, Sustainable Development is defined as "the development that meets the needs of the present without jeopardizing the ability of future generations to meet their own needs" (WCED, 1987). his concept of sustainable development implies several relevant facts: the Earth's natural resources are limited and increasingly used and consumed mainly by a minority of people, which creates intra- and intergenerational inequality (Sikdar, 2004). Recently, a new re-definition of this concept was stated by Griggs et al. (2013): "the development that meets the needs of the present while safeguarding Earth´s life-support system, on which the welfare of current and future generations depends". Considering this definition, two new priorities are highlighted: the protection of Earth´s life-support system and poverty reduction.

In summary, there is an international consensus about the three main pillars that integrate the sustainability development: economic development, environmental protection and social participation, and about the close interaction between them (Purvis et al., 2019). The three pillars should be addressed not only from an individual point of view but also including the inter-relations among them. As presented in Figure 2.1, the integration of each pair of pillars is considered a partial state of sustainability, while sustainability only can be in the centre of the diagram, represented by the convergence of the three dimensions. A different approach is that where the economy is a subsystem of society, which is, in turn, a subsystem of the biosphere or environment (Figure 2.1). Hence, a major goal for the global primary sector is to identify and understand the interactions and feedbacks among human and natural processes that regulate the biosphere (i.e., environment) and consequently its capacity for providing favourable conditions for complex human societies (i.e., society) and for long-term human prosperity (i.e., economy) (Castro et al., 2019).

Although this concept of sustainable development is more than three decades old, its importance has only grown over this time, and it remains one of the greatest challenges facing humanity in the face of the climate emergency we are facing (United Nations Environment Program, 2021), that is the reason why sustainable development is now on global political and business agendas (European Commission, 2011).

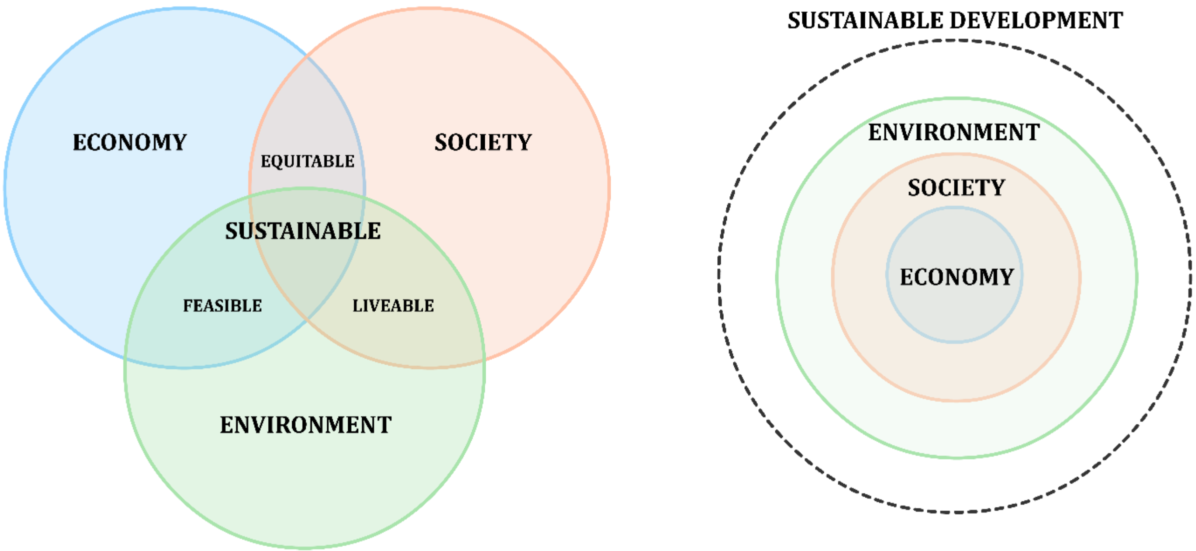


Figure 2.1. Conceptual framework of a Sustainable Development.: Typical representation of the three pillars of sustainability (left); Alternative Sustainable Development conceptual model (right).

In this context, the importance of quantifying the environmental impacts and resource consumption over the entire life cycle of products and processes arises. In this sense, the Life Cycle Thinking (LCT) has become as a key complementary approach to modern environmental policies and decision-making, involving both government and business support (European Commission, 2011). LCT focuses on looking beyond traditional manufacturing processes to recognise the relevance of some potential impacts in all the stage of a product or process life cycle, from resources extraction to end-of-life management (Mesa-Alvarez and Ligthart, 2021). LCT can be particularly useful in making sustainability decisions by taking into account the entire life cycle of a product or process, as focusing on one stage or another can lead to erroneous or unbalanced outcomes (Pelletier, 2015). Moreover, LCT seeks to reduce the emissions to the environment and the consumption of resources associated with a product as well as to improve its socio-economic performance throughout its life cycle (Pelletier, 2015; Petit-Boix et al., 2017). In this way, it may facilitate relationships between the economic, social and environmental dimensions (European Commission, 2010).

Following this perspective, a wide set of environmental management tools have been developed in recent years with the aim of better understanding the impacts that are caused on natural environments and, consequently, minimising the environmental burdens associated with a given product, process, or service. In recent decades, there has been a strong proliferation of environmental management tools: Environmental Risk Assessment (ERA), Input-Output Analysis (IOA), Material Flow Analysis (MFA), among others. However, the use of Life Cycle Assessment (LCA) appears as the best option to implement in a wide variety of products and processes, while becoming the most widely accepted tool in the scientific community.

2.2. LIFE CYCLE ASSESSMENT

LCA is an internationally standardised technique for assessing the environmental impacts associated with a product by compiling an inventory of relevant inputs and outputs of a product system, evaluating the potential environmental impacts associated with those inputs and outputs and interpreting the results of the inventory analysis and impact assessment phases in relation to the objective of the study (ISO 14040, 14044). LCA allows the analysis of the entire life cycle of the products or service under study, including all the stages from raw material extraction to processing, transportation, manufacture, retailing, distribution, consumption, re-use, recycling, and end-of-life. When a study covers all these stages, the study is said to have a cradle-to-grave perspective. However, in practice, most LCAs omit certain stages of the life cycle and only assess the potential impacts of different phases, using perspectives of cradle-to-gate, gate-to-gate, gate-to-cradle or gate-to-grave. Among the wide range of applications of this methodology, the following can be highlighted: (i) The identification of opportunities to improve the environmental profile of products or services; (ii) The selection of important indicators to report environmental performance; (iii) The provision of valuable information to decision-makers in a wide range of institutions, with the aim of influencing strategic planning, priority setting or process design; (iv) Marketing purposes, such as the introduction of eco-labelling for a specific product or to provide environmental product declarations.

A total of 4 stages can be distinguished in LCA methodology according to the framework established in ISO 14040, 14044, which are depicted in Figure 2.2 and include: (i) identifying the context of the study, its benefits and its limitations; (ii) collecting inventory data for significant energy and material inputs; (iii) evaluating the potential environmental impacts linked to the included inputs/outputs and, finally; (iv) interpreting the results obtained.

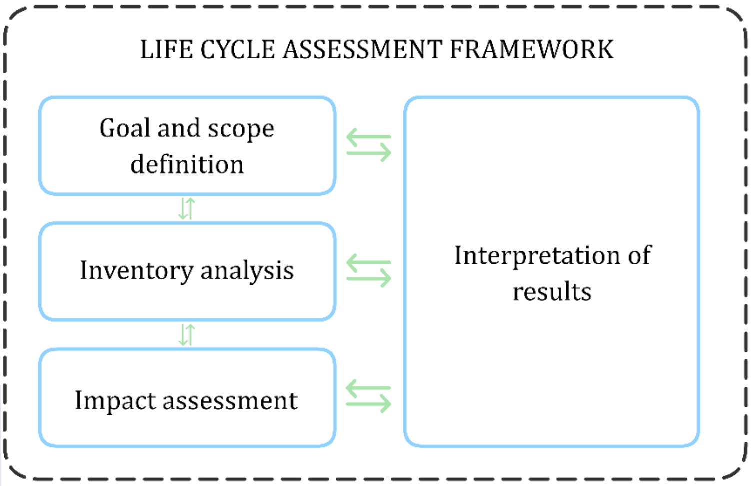


Figure 2.2. Phases of an LCA study (adapted from ISO 14040).

2.2.1. Goal and scope definition

In this first phase of the LCA study, the product or service to be analysed should be defined together with the objectives to be achieved, as well as the decision context (modelling approach) and intended audience and applications of the study. The scope of the system must be also established at this stage, including the system to be studied, its functions and functional unit (FU), the system boundaries and life cycle stages to be covered, the environmental impacts to be calculated, the assessment methods to calculate those environmental impacts, allocation procedures, minimum data quality requirements, assumptions and limitations (ISO 14040).

- **Function and functional unit:** The selection of the function of the system is one of the methodological issues in LCA studies as it is related to the definition of the FU and the system boundaries. The system under study may have several possible functional units, and the one selected as the basis for the analysis will depend on the goal and scope considered. The FU can be defined as the quantified performance of a product system for uses as a reference unit (ISO 14040). The primary purpose of an FU is to provide a reference to which the inputs and outputs are related, it is important to clearly define the FU to ensure comparability of LCA results.

- **System boundaries**: The system boundaries determine the scope of the system that is considered in the LCA. The establishment of the system boundaries depends on several factors, including the application of the study, the assumptions, the level of data detail as well as data limitations. In general, all life cycle stages, unit processes and flows should be considered when establishing the system boundaries, including the raw materials acquisition, inputs and outputs in the main processing, distribution, and transportation, fuel and energy requirements, recovery of used products, waste disposal and other additional operations. Nevertheless, sometimes the stages that are expected not to be significant can be cut-off to focus effort on obtaining more reliable data for the relevant processes. It can be distinguished two types of processes: i) foreground processes that refer to the process required to produce the product under study and ii) background processes that include the processes required to supply energy and materials to the foreground system.

- **Modelling approach**: two main alternatives of modelling are used in LCA: attributional and consequential. The attributional approach: refers to an actual supply chain of the product or service, along with its use and end-of life phases. Thus, it is assumed, that the system under study is embedded into a static Technosphere, which makes it possible to estimate the potential environmental impacts that can be directly attributed to this system (Weidema et al., 2018).

Regarding the consequential approach integrates the supply-chain as it is theoretically expected because of the analysed decision, including the changes derived from the interaction between the system under study and the markets. In this sense, the Technosphere is considered dynamic (Weidema et al., 2020); in other words, consequential LCA "is designed to generate information on the consequences of decisions" (Ekvall et al., 2016). Taking into account that the main objective of this doctoral thesis is the environmental evaluation of circular economy practices per se, not to evaluate the consequences of their application, only attributional approach was considered.

- **Allocation rules:** Most systems do not lead to a single output or are based on the linearity of raw materials inputs and outputs; but rather supply more than one product or even recycle intermediate or discarded products in the process (European Commission, 2010; ISO 14040), which are defined as "multifunctional" systems. Despite this, in most LCA studies, the interest is only meaningful for one of the products, the rest are usually secondary. For this reason, special attention should be paid to the allocation of potential burdens and credits involved in multiple products and/or recycling processes (ISO 14040). In this sense, different approaches can be used to solve this multifunctionality according to ISO (14044): first, allocation should be avoided whenever possible by applying subdivision (to divide the system into different subsystems to collect the inventory data related to each of them) or system expansion (o expand the overall system to include additional functions related to the co-products). When it cannot be avoided, ISO standards propose that the inputs and outputs of the system should be partitioned between the different co-functions/co-products of the system according to some allocation criteria, which can be physical, economic or energetic, while giving priority to the physical relationships (European Commission, 2010; ISO 14044).

2.2.2. Life cycle inventory

Life Cycle Inventory (LCI) is the stage data that involves data collection as well as the calculation of the remaining data required to complete all relevant inputs and outputs of each unit process defined within the system boundary. The LCI data can be divided into primary and secondary data, while the former is produced by the producers of goods and operators of processes and services; the latter is provided by database and represent generic data. Data for each unit process defined in the system boundary should be collected from the system and expressed based on the FU, including energy inputs, raw material inputs, ancillary inputs, products, co-products, waste, emissions to air and discharges to water and soil. It is, therefore, the phase that requires the greatest efforts and resources in a LCA study, involving the collection and modelling of several flows (European Commission, 2010).

2.2.3. Life cycle impact assessment

According to ISO 14044, the life cycle impact assessment phase is the stage of LCA that aims at understanding and evaluating the magnitude and significance of the potential environmental impacts for a product system throughout the life cycle of the product. IN this phase, the LCI data from the previous stage is translated into different impact categories and category indicators related to human health, natural environment, and resource depletion. In this sense, the selection of impacts and methodologies used must be in concordance with the goal and scope defined. The impact assessment may include the iterative process of reviewing the goal and scope of the LCA study to determine if the objectives of the study have been met, or to modify the goal and scope if the assessment indicates that they cannot be achieved.

According to ISO 14044, the three mandatory steps that must include this evaluation are: i) selection of impact categories and characterisation methods, ii) classification and iii) characterisation. Additional steps such as i) normalisation, ii) grouping and iii) weighting can be performed optionally at a later stage.

- **Selection of impact categories and characterisation methods:** The environmental impact categories to be considered in the LCIA, as well as the corresponding characterisation methods, will be defined in the earliest phases, according to the goal and scope of the LCA study. Impact categories are a comprehensive set of environmental issues, and their selection should be connected to specific environmental issues related to the product system under study. Each impact category is quantitatively represented by category indicators expressed each one in a specific unit of measurement calculated according to a selected characterisation model. To date, several common characterisation factors and methods have been developed. They can be classified into midpoint and endpoint level: while the former includes a greater number of impact categories (around 10) and provides more accurate results, the latter focuses on the three areas of protection – human health, environment, resource depletion – commonly used for endpoint assessment. According to the standards, several LCA impact categories are recommended, such as climate change, ozone depletion, human toxicity, photochemical ozone formation, acidification, eutrophication, ecotoxicity, land use and resource depletion. However, each specific study will focus on those categories that are relevant based on the basis of related inputs and outputs (European Commission, 2010).

- **Classification and characterisation:** The classification step consists of the assignment of LCI results to one or more impact categories and/or indicators. Therefore, a cause-effect pathway is used to identify the relationship between the environmental intervention (for instance, the emission of a carbon dioxide) and its potential effects on the environment. In the characterisation step, the LCI results assigned to each impact category are converted to common units using the characterisation factors of the model selected. To this aim, the inventory data of the different flows must be linearly multiplied with the impact factors.

- **Normalisation:** This step estimates the relative magnitude of the results attributed to each category, divided by a selected reference value. The aim of this optional phase seeks to help to better understand the relative relevance of each impact category in the product system under study (ISO 14044).

- **Grouping:** It involves the aggregation of different impact categories into one or several sets, also including the definition of priority rankings according to a hierarchy.

- **Weighting:** This phase is defined as the process of converting the results of different impact categories by using numerical factors based on value-choices related to priority criteria (ISO 14044). It provides a final single impact score, although it is based on subjective knowledge rather than available scientific knowledge, so that different results can be obtained for the same indicator.

- **Life cycle impact assessment mechanisms – the ReCiPe method:** ReCiPe is the most recent and harmonized methodology available in LCA (Goedkoop et al., 2009). This method provides a common framework in which midpoint and endpoint levels can be evaluated and it includes the characterisation models recommended in ILCD handbook. The ReCiPe method comprises eighteen midpoint indicators linked to three additional endpoint indicators (Figure 2.3). As converting midpoints to endpoints can be responsible for greater uncertainties in a LCA study, the ReCiPe method addresses the inherent uncertainties through three different perspectives based on time horizon criteria (Goedkoop et al., 2009; Huijbregts et al., 2016): individualist (I), hierarchist (H) and egalitarian (E). Individualist perspective is based on the short-time interest (20-year horizon), indisputable impacts and technological optimism regarding human adaptation. Hiearchist perspective is based on the most common time frame of the policy principles (100-year horizon), while egalitarian perspective refers to the most precautionary situation, taking into account the longest time horizon of 100,000 years (Goedkoop et al., 2009).

2.2.4. Interpretation of results

Finally, the interpretation phase is carried out based on the combination of the major findings from the previous phases (ISO 14040). Moreover, this phase may incorporate sensitivity, consistency, and uncertainty analyses to ensure the reliability of the results.

In this sense, they are expected to serve as a basis for the conclusions, main findings, and recommendations to decision-makers, in line with the goal and scope of the study.

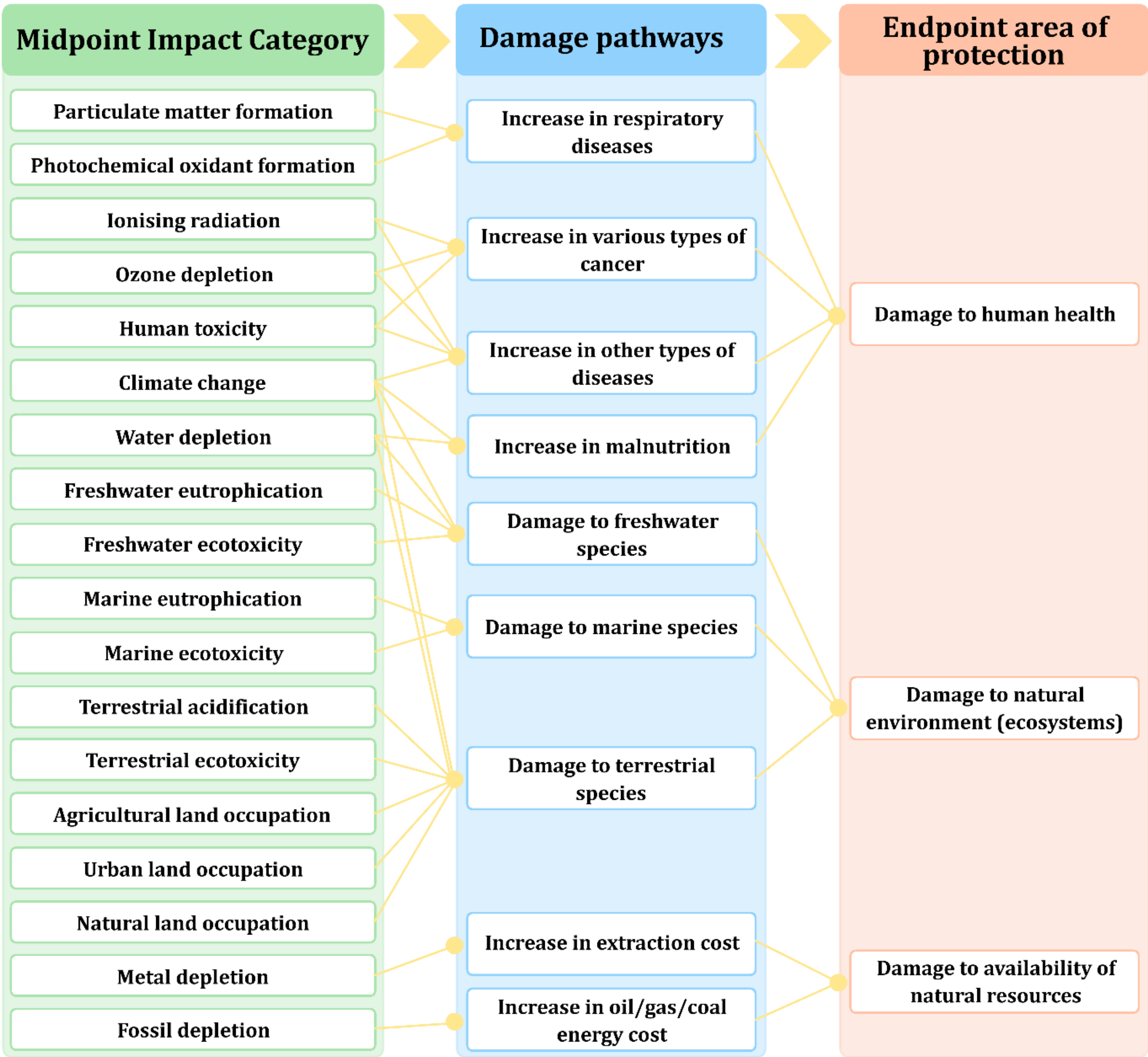


Figure 2.3. Environmental mechanisms in the ReCiPe method (adapted from Huijbregts et al., 2016).

2.3. DATA ENVELOPMENT ANALYSIS

Data Envelopment Analysis (DEA) is a linear programming methodology used to non-parametrically calculate the comparative efficiency of multiple similar entities called Decision Making Units or DMUs (Cooper et al., 2007). A DMU is defined as the entity responsible for the conversion of inputs into outputs and whose performance is the object of assessment. To carry out a DEA, data for inputs and outputs from the different entities must be known. From these data, DEA formulates and solves an optimisation model which facilitates benchmarking the operational performance of each assessed entity, usually resulting in an improved eco-efficiency. Thus, DEA estimates production efficient frontiers for a number of homogenous DMUs; in mathematical terms, these efficient frontiers are said to envelop all units. The region determined by the efficient frontiers is called production possibility set, and the DMUs on the frontiers constitute the reference set. The result for each DMU is an efficiency score and, for those DMUs identified as inefficient, a target operating point.

DEA enables the discrimination of inefficient operating points, therefore promoting feasible technological improvements under the perspective of an efficient operational performance. This is done through the formulation of a model with specific features in terms of metrics (radial or non-radial model), orientation (e.g., input- or output-oriented model), and display of the set of production possibilities (e.g., constant, or variable returns to scale).

2.4. LIFE CYCLE ASSESSMENT + DATA ENVELOPMENT ANALYSIS

Eco-efficiency analysis emerged as a valuable tool towards the target of sustainable development since it connects business and economic goals with environmental objectives, engaging companies in the agenda of sustainable development (Vásquez et al., 2019). According with the World Business Council for Sustainable Development: “eco-efficiency is achieved by the delivery of competitively priced goods and services that satisfy human needs and bring quality of life, while progressively reducing ecological impacts and resource intensity throughout the life-cycle to a level at least in line with the Earth’s estimated carrying capacity" (Koskela and Vehmas, 2012). In this sense, eco-efficiency is based on the concept of creating more value with less impact (Caiado et al., 2017).

Taking into account this definition, the goal is how to reduce environmental impact, improving the value of the system and without compromising productivity. In order to evaluate the environmental efficiency, the joint use of LCA + DEA has been widely applied in recent years. DEA allows to calculate the comparative productive efficiency of multiple DMUs (Cooper et al., 2007) and, given a certain number of inputs and outputs, DEA identifies the efficient DMUs and projects the inefficient DMUs at the efficient frontier, thereby providing target efficient values to convert the inefficient entities into efficient ones

The LCA methodology can be used to determine the indicators from an environmental perspective, in this sense, the combination of LCA and DEA can provide quantitative life-cycle-based benchmarks tat orientate the performance towards environmental sustainability (Vázquez-Rowe and Iribarren, 2015). Generally, when input and output data are available for similar entities, a common solution is to establish an average inventory that includes the average values of the different inputs and outputs. However, the high variability derived with this system can represent a major barrier when conducting the study. An alternative approach to deal with these cases of multiple inventories is to conduct individuals LCAs for each of these inventories, which may represent an improvement in variability, but multiple result can be difficult to interpret. The combination between LCA and DEA for the assessment of a large number of similar entities has become very popular as it avoids the principal problems of computing large datasets as it: i) avoids large standard deviations; ii) facilitates the interpretation of results, and iii) provides additional information useful for improving operational efficiency which can be translated into improved environmental efficiency.

A range of different models is available to run the matrix resulting from the selection of relevant inputs and outputs of the DMUs under study, including variants of the Charnes-Cooper-Rhodes (CCR), the Banker-Charnes-Cooper (BCC) and the assurance region (AR) models. However, the most common method for analysing the eco-efficiency of systems using the LCA+DEA method is the slacks-based measure of efficiency (SBM), several examples are available in literature (Álvarez-Rodríguez et al., 2019; Á. Avadí et al., 2014; Iribarren et al., 2010b, 2015; Lijó et al., 2017; Lorenzo-Toja et al., 2015, 2018; Martín-Gamboa et al., 2018; Vázquez-Rowe et al., 2012c). The SBM model is elastic regarding the calculation of the inefficiencies for the different DMUs since it performs the computation regardless the units of measure used for the different inputs and outputs. With this regard, the input-oriented approach allows to minimise the use of resources and, therefore, an optimisation of operational inputs, while maintaining the number of outputs; on the contrary, the output-oriented approach is focused on the maximisation of outputs (Cooper et al., 2007). Finally, this method also can be performed according to constant or variable returns-to-scale; the former limits the effect of different scales within the eco-efficiency results; although the latter is more suitable for DMUs of the same scale (Cooper et al., 2007).

As specify, DEA is basis on the formulation of a model with specific features in terms of metrics (radial or non-radial model), orientation (e.g., input- or output-oriented model), and display of the set of production possibilities (e.g., constant, or variable returns to scale). In the two cases study performed in this doctoral thesis, the specific non-radial DEA model used is the input-oriented slacks-based measure of efficiency model with constant returns to scale (SBM-I-CRS model), formulated through equations 1-4, according to Iribarren et al. (2010d) and Tone (2011).

$$\Phi_0 = \min\left(1 - \frac{1}{M}\sum_{k=1}^{M}\frac{\sigma_{ko}}{x_{ko}}\right) \quad (1)$$

Subject to:

$$\sum_{j=1}^{N}\lambda_{j0}x_{kj} = x_{k0} - \sigma_{k0}\ \forall\ k \quad (2)$$

$$\sum_{j=1}^{N} \lambda_{j0} y_j = y_0 \tag{3}$$

$$\lambda_{j0} \geq 0 \; \forall \, j, \sigma_{k0} \geq 0 \; \forall \, k \tag{4}$$

Where N: number of DMUs; j: index on the DMU; M: number of inputs; k: index on inputs; x_{kj}: amount of input k demanded by DMU j; y_j: amount of output generated by DMU j; 0: index of the DMU under assessment; (λ_{10},λ_{20},…,λ_{N0}): coefficients of linear combination for assessing DMU 0; σ_{k0}: slack (i.e., potential reduction) in the demand of input k by DMU 0; and Φ_0: efficiency score of DMU 0.

The choice of an input-oriented model aims to reduce inputs and ensure at least the same output (i.e., the same quantity of produced milk or shrimps). Solving the optimisation problem results in the efficiency score (Φ) of each production site. Efficiency scores lead to discriminate between efficient (Φ = 1) and inefficient (Φ < 1) facilities.

2.4.1. Five step LCA + DEA method

To date, this is the most common method to combine the LCA and DEA methodologies, it is recommended to undertake eco-efficiency verification analyses through the quantification of the environmental consequences of operational inefficiencies (Lijó et al., 2017). The five step method was established in Vázquez-Rowe et al. (2010a) for the combined operational and environmental assessment of multiple similar units, the five steps are:

- **Development of the LCI for each DMU**. This stage involves collecting detailed input and output data for all the assessed systems.

- **Life cycle impact assessment for every DMU**. This second steps consists of characterise the environmental profile of the current DMUs from the LCI developed in the first step.

- **Data envelopment analysis**. The DEA matrix, which includes some of the most relevant inputs and outputs of the previous LCI is run. The results of this stage allow to rank the DMUs by their operational efficiency and to determine the measures to be taken to achieve the efficiency objectives. In this way, target DMUs are calculated by applying the inputs-reduction or products-maximisation targets proposed by the methodology. Thus, objectives are calculated by projecting each DMU on the efficient frontier determined by the entire set of analysed DMUs. Thus, operational benchmarking is attained.

- **LCIA of the target DMUs from the new LCI data arising from the third step**. The inventories of the current DMUs are subjected to the changes proposed in step 3 and are environmentally analysed as in step 2. Consequently, the potential environmental impacts associated with the virtual DMUs are determined.

- **Interpretation of results according to eco-efficiency criteria**. The comparison between the potential environmental impacts for the virtual DMUs and those for the current DMUs quantifies the environmental impacts generated by inadequate operational practices (operational inefficiencies). This also estimates the environmental benefits related to good operating practices.

2.4.2. Three step LCA + DEA method

This is an alternative approach that was used in the early stages of the application of this methodology (Lozano et al., 2010). It aims to estimate the environmental impact efficiency and the simultaneous benchmarking of operational and environmental parameters.

The first two steps of this method are coincident with those for the five-step method. However, the third phase comprises a DEA matrix with a higher number of inputs due to the inclusion of the potential environmental impacts determined in the second step as additional inputs. In this sense, the benchmarking results directly estimate targets for both LCI inputs/outputs and the potential environmental impacts. Therefore, unlike the five-step method, this option avoids the environmental characterisation of the target DMUs by implementing environmental impacts as inputs when performing DEA in the third step. For LCA practitioners, this alternative is less interesting, since the DEA is not a method for environmental management, but for operational one.

2.5. LIFE CYCLE ASSESSMENT ON AGRICULTURE AND FISHERIES

Agricultural sector us a very broad field, so it is difficult to determine an exhaustive search on how the LCA methodology has been. Many studies that seek to determine the GHG emissions and carbon footprint of different foodstuffs can be found in literature, such as: i) meat as beef (Clune et al., 2017), chicken (González-García et al., 2014), lamb (Jones et al., 2014) or pork (Noya et al., 2017); ii) fruits and vegetables as asparagus (Vazquez-Rowe et al., 2016), spinach, carrot and zucchini (Clune et al., 2017), avocado (Aguilera et al., 2015a), potato, onion, garlic and tomato (Aguilera et al., 2015b), strawberry (Clune et al., 2017), orange, banana, apple, melon and grape (Aguilera et al., 2015a), or pineapple (Aguilera et al., 2015b); and iii) different dairy products as milk (González-García et al., 2013a; Iribarren et al., 2011; Noya et al., 2018), yogurt (González-García et al., 2013b), butter and cheese (Vergé et al., 2013) or ice-cream (Werner et al., 2014).

Focusing on the specific topic of agricultural waste valorisation, LCA is a useful tool for decision-makers involved in the evaluation of projects that aim to apply circular economy principles on the agricultural sector. As a result, several LCA studies have been published in the last years. According to Yaashikaa et al. (2022), numerous studies have been identified performing environmental assessments of different alternatives for agricultural waste valorisation through the LCA methodology. The main studies that stand out are those in which the aim is to evaluate potential agro-industrial by-products to produce biobased products under economically feasible and eco-friendly conditions. In this sense, some studies about biorefinery products from agro-industrial biomass using different conversion technologies can be found, as corn fibre (Zhang et al., 2021), grape pomace (Rodríguez et al., 2010), corn stover (Buruiana et al., 2014), cotton stalk (Li et al., 2013), sugarcane bagasse (Zhou and Xu, 2019), sweet sorghum bagasse (Su et al., 2020), wheat bran (Tirpanalan et al., 2015), oil palm (Rehman et al., 2021), sawdust (Abdou et al., 2021), spent coffee grounds (López-Linares et al., 2021), chestnut shells (Morales et al., 2018), grass silage (Schwarz et al., 2018), pomegranate peels (Talekar et al., 2018), cheese whey (Alvarez-Guzmán et al., 2020) corn cob molasses (Wang et al., 2010), or bread waste (Gadkari et al., 2021), among others.

One of the main important characteristics of agricultural LCA is the use of multiple functional units. The most commonly used are final products in mass/volume terms (kg or L), but also energy or protein content in food products (kJ), area (ha), or even units of livestock. Although the use of LCA in the agri-food industries is rapidly increasing, there are considerable inconsistencies existing among the studies, for example in methodological aspects such as the aforementioned choice of FU, but also in the system boundaries and in the used impact method. Since LCA results directly depend on the functional unit choice, it seems consistent that the next step in conducting LCA studies is a normalisation in the selection of the FU. Therefore, for future LCA studies on agri-food products, there might be a choice of functional unit for studies on food products, that is the production of a specific amount of food, 1 kg for example, allowing a correct communication to consumers on the environmental impact of agri-food.

The application of LCA methodology to determine the environmental impacts of fish catches, farming (aquaculture), and processing started in the mid-2000s. A long list of LCA seafood studies on diverse pelagic species such as horse mackerel (Vázquez-Rowe et al., 2010b), Peruvian and Cantabrian anchovy (Fréon et al., 2014; Laso et al., 2018), carp (Hornborg and Främberg, 2020) or Atlantic mackerel (Ramos et al., 2011) have been reported. Demersal species such as hake (Avadí et al., 2018; Vázquez-Rowe et al., 2011), cod (Svanes et al., 2011; Ziegler et al., 2013) or octopus (Vázquez-Rowe et al., 2012b), crustacean species such as prawns (Farmery et al., 2015; Medeiros et al., 2017), lobster (Driscoll et al., 2015) or goose barnacle (Vázquez-Rowe et al., 2013a) can be found on bibliography. Regarding aquaculture, different studies on mussels (Iribarren et al., 2010c, 2010b; Lourguioui et al., 2017; Tamburini et al., 2020), oysters (Ray et al., 2019; Tamburini et al., 2019), turbot (Iribarren et al., 2012), tilapia (Yacout et al., 2016), trout (Samuel-Fitwi et al., 2013), seabass and seabream (Abdou et al., 2017) or salmon farming (Philis et al., 2021, 2019) can be highlighted. However, it is also important to mention that traditionally not only fishing or farming activities have been evaluated, but also the production of different fish- and seafood-based products such as fish sticks (Vázquez-Rowe et al., 2013b), fishmeal and fish oil (Fréon et al., 2017) and canned products (Almeida et al., 2015; A. Avadí et al., 2014; Avadí et al., 2015; Iribarren et al., 2010a; Laso et al., 2017; Vázquez-Rowe et al., 2014) or the certification for eco-labelling (Vázquez-Rowe et al., 2016). Review articles on fishing (Avadí and Fréon, 2013; Vázquez-Rowe et al., 2012a; Vélez-Henao et al., 2021), aquaculture (Bohnes et al., 2019; Bohnes and Laurent, 2019; Henriksson et al., 2012; Philis et al., 2019) and processing (Gómez et al., 2019; Ruiz-Salmón et al., 2021) stages have also been published in bibliography. Not only the detailed LCA studies for specific species or stages, but also some information on fuel use and GHG emissions related to fisheries can be obtained from some studies at different scales, from small fleets to a global scale (Driscoll and Tyedmers, 2010; Greer et al., 2019; Kitts et al., 2008; Mohan, 2018; Parker et al., 2018, 2015; Parker and Tyedmers, 2015).

The reviewed studies focused mainly on European and American operations, mostly in the Atlantic Ocean and North Sea, but occasionally in African waters and the other oceans waters, during the last decade (2011–2021), demonstrating the need to expand further LCA studies applied not only to different species and fishing gears, but also other global fishing fleets.

The most common pattern found in the studies is the fact that fishing operations are the main contributor to environmental impacts during the extraction phase. Construction and EOL phases were generally roughly considered or directly omitted from analyses, due to the demonstrated claim that those phases generate negligible impacts in comparison to use and maintenance. Few studies dealing with construction, too partially in most cases, indicate the dominant importance of metals on environmental impacts related to toxicity to humans and the environment. All studies reviewed deal with emissions to air due to combustion of fossil fuels during the use phase. Some of the studies discussed the impacts due to vessel maintenance, emphasising the use of antifouling and refrigerants and allocating great importance to these substances as contributors to impacts.

Despite the fact that existing fisheries LCA studies are difficult to contrast due to a general lack of detail and standardisation, valuable conclusions can be mined from available literature, concluding that fuel consumption, use of antifouling paints and associated release of substances are key contributors to environmental impacts.

2.6. REFERENCES

Abdou, K., Aubin, J., Romdhane, M.S., Le Loc'h, F., Lasram, F.B.R., 2017. Environmental assessment of seabass (*Dicentrarchus labrax*) and seabream (*Sparus aurata*) farming from a life cycle perspective: A case study of a Tunisian aquaculture farm. Aquaculture 471, 204–212. https://doi.org/10.1016/j.aquaculture.2017.01.019

Abdou, M., Marcati, A., Pons, A., Vial, C., 2021. Modeling and simulation of a sawdust mixture-based integrated biorefinery plant producing bioethanol. Bioresour. Technol. 325, 124650. https://doi.org/10.1016/j.biortech.2020.124650

Aguilera, E., Guzmán, G., Alonso, A., 2015a. Greenhouse gas emissions from conventional and organic cropping systems in Spain. II. Fruit tree orchards. Agron. Sustain. Dev. 35, 725–737. https://doi.org/10.1007/s13593-014-0265-y

Aguilera, E., Guzmán, G., Alonso, A., 2015b. Greenhouse gas emissions from conventional and organic cropping systems in Spain. I. Herbaceous crops. Agron. Sustain. Dev. 35, 713–724. https://doi.org/10.1007/s13593-014-0267-9

Almeida, C., Vaz, S., Ziegler, F., 2015. Environmental Life Cycle Assessment of a Canned Sardine Product from Portugal. J. Ind. Ecol. 19, 607–617. https://doi.org/10.1111/jiec.12219

Alvarez-Guzmán, C.L., Balderas-Hernández, V.E., De Leon-Rodriguez, A., 2020. Coproduction of hydrogen, ethanol and 2,3-butanediol from agro-industrial residues by the Antarctic psychrophilic GA0F bacterium. Int. J. Hydrogen Energy 45, 26179–26187. https://doi.org/10.1016/j.ijhydene.2020.02.105

Álvarez-Rodríguez, C., Martín-Gamboa, M., Iribarren, D., 2019. Combined use of Data Envelopment Analysis and Life Cycle Assessment for operational and environmental benchmarking in the service sector: A case study of grocery stores. Sci. Total Environ. 667, 799–808. https://doi.org/10.1016/j.scitotenv.2019.02.433

Avadí, A., Adrien, R., Aramayo, V., Fréon, P., 2018. Environmental assessment of the Peruvian industrial hake fishery with LCA. Int. J. Life Cycle Assess. 23, 1126–1140. https://doi.org/10.1007/s11367-017-1364-1

Avadí, A., Bolaños, C., Sandoval, I., Ycaza, C., 2015. Life cycle assessment of Ecuadorian processed tuna. Int. J. Life Cycle Assess. 20, 1415–1428. https://doi.org/10.1007/s11367-015-0943-2

Avadí, A., Fréon, P., 2013. Life cycle assessment of fisheries: A review for fisheries scientists and managers. Fish. Res. 143, 21–38. https://doi.org/10.1016/j.fishres.2013.01.006

Avadí, A., Fréon, P., Quispe, I., 2014. Environmental assessment of Peruvian anchoveta food products: Is less refined better? Int. J. Life Cycle Assess. 19, 1276–1293. https://doi.org/10.1007/s11367-014-0737-y

Avadí, Á., Vázquez-Rowe, I., Fréon, P., 2014. Eco-efficiency assessment of the Peruvian anchoveta steel and wooden fleets using the LCA+DEA framework. J. Clean. Prod. 70, 118–131. https://doi.org/10.1016/j.jclepro.2014.01.047

Bohnes, F.A., Hauschild, M.Z., Schlundt, J., Laurent, A., 2019. Life cycle assessments of aquaculture systems: a critical review of reported findings with recommendations for policy and system development. Rev. Aquac. 11, 1061–1079. https://doi.org/10.1111/raq.12280

Bohnes, F.A., Laurent, A., 2019. LCA of aquaculture systems: methodological issues and potential improvements. Int. J. Life Cycle Assess. 24, 324–337. https://doi.org/10.1007/s11367-018-1517-x

Buruiana, C.-T., Vizireanu, C., Garrote, G., Parajó, J.C., 2014. Optimization of corn stover biorefinery for coproduction of oligomers and second generation bioethanol using non-isothermal autohydrolysis. Ind. Crops Prod. 54, 32–39. https://doi.org/10.1016/j.indcrop.2014.01.003

Caiado, R.G.G., de Freitas Dias, R., Mattos, L.V., Quelhas, O.L.G., Leal Filho, W., 2017. Towards sustainable development through the perspective of eco-efficiency - A systematic literature review. J. Clean. Prod. 165, 890–904. https://doi.org/10.1016/j.jclepro.2017.07.166

Castro, A.J., López-Rodríguez, M.D., Giagnocavo, C., Gimenez, M., Céspedes, L., La Calle, A., Gallardo, M., Pumares, P., Cabello, J., Rodríguez, E., Uclés, D., Parra, S., Casas, J., Rodríguez, F., Fernandez-Prados, J.S., Alba-Patiño, D., Expósito-Granados, M., Murillo-López, B.E., Vasquez, L.M., Valera, D.L., 2019. Six collective challenges for sustainability of Almería greenhouse horticulture. Int. J. Environ. Res. Public Health 16. https://doi.org/10.3390/ijerph16214097

Clune, S., Crossin, E., Verghese, K., 2017. Systematic review of greenhouse gas emissions for different fresh food categories. J. Clean. Prod. 140, 766–783. https://doi.org/10.1016/j.jclepro.2016.04.082

Cooper, W.W., Seiford, L.M., Tone, K., 2007. Data Envelopment Analysis: A compresensive text with models, applications, references and DEA-Solver software. Springer, New York.

Driscoll, J., Boyd, C., Tyedmers, P., 2015. Life cycle assessment of the Maine and southwest Nova Scotia lobster industries. Fish. Res. 172, 385–400. https://doi.org/10.1016/j.fishres.2015.08.007

Driscoll, J., Tyedmers, P., 2010. Fuel use and greenhouse gas emission implications of fisheries management: the case of the new england atlantic herring fishery. Mar. Policy 34, 353–359. https://doi.org/10.1016/j.marpol.2009.08.005

Ekvall, T., Azapagic, A., Finnveden, G., Rydberg, T., Weidema, B.P., Zamagni, A., 2016. Attributional and consequential LCA in the ILCD handbook. Int. J. Life Cycle Assess. 21, 293–296. https://doi.org/10.1007/s11367-015-1026-0

European Commission, 2011. International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance. Publications Office of the European Union. https://doi.org/10.2788/38479

European Commission, 2010. Making sustinable consumption and production a reality. A guide for business and policy makers to Life Cycle Thinking and Assessment. Publications Office of the European Union. https://doi.org/10.2779/91521

Farmery, A., Gardner, C., Green, B.S., Jennings, S., Watson, R., 2015. Life cycle assessment of wild capture prawns: Expanding sustainability considerations in the Australian Northern Prawn Fishery. J. Clean. Prod. 87, 96–104. https://doi.org/10.1016/j.jclepro.2014.10.063

Fréon, P., Avadí, A., Vinatea Chavez, R.A., Iriarte Ahón, F., 2014. Life cycle assessment of the Peruvian industrial anchoveta fleet: Boundary setting in life cycle inventory analyses of complex and plural means of production. Int. J. Life Cycle Assess. 19, 1068–1086. https://doi.org/10.1007/s11367-014-0716-3

Fréon, P., Durand, H., Avadí, A., Huaranca, S., Orozco Moreyra, R., 2017. Life cycle assessment of three Peruvian fishmeal plants: Toward a cleaner production. J. Clean. Prod. 145, 50–63. https://doi.org/10.1016/j.jclepro.2017.01.036

Gadkari, S., Kumar, D., Qin, Z., Ki Lin, C.S., Kumar, V., 2021. Life cycle analysis of fermentative production of succinic acid from bread waste. Waste Manag. 126, 861–871. https://doi.org/10.1016/j.wasman.2021.04.013

Goedkoop, M., Heijungs, R., Huijbregts, M., Schryver, A., De Struijs, J., Van Zelm, R., 2009. ReCiPe 2008, a Life Cycle Impact Assessment Method Which Comprises Harmonised Category Indicators at the Midpoint and the Endpoint Level.

Gómez, B., Munekata, P.E.S., Gavahian, M., Barba, F.J., Martí-Quijal, F.J., Bolumar, T., Campagnol, P.C.B., Tomasevic, I., Lorenzo, J.M., 2019. Application of pulsed electric fields in meat and fish processing industries: An overview. Food Res. Int. 123, 95–105. https://doi.org/10.1016/j.foodres.2019.04.047

González-García, S., Castanheira, É.G., Dias, A.C., Arroja, L., 2013a. Using Life Cycle Assessment methodology to assess UHT milk production in Portugal. Sci. Total Environ. 442, 225–234. https://doi.org/10.1016/j.scitotenv.2012.10.035

González-García, S., Castanheira, É.G., Dias, A.C., Arroja, L., 2013b. Environmental life cycle assessment of a dairy product: The yoghurt. Int. J. Life Cycle Assess. 18, 796–811. https://doi.org/10.1007/s11367-012-0522-8

González-García, S., Gomez-Fernández, Z., Dias, A.C., Feijoo, G., Moreira, M.T., Arroja, L., 2014. Life Cycle Assessment of broiler chicken production: A Portuguese case study. J. Clean. Prod. 74, 125–134. https://doi.org/10.1016/j.jclepro.2014.03.067

Greer, K., Zeller, D., Woroniak, J., Coulter, A., Winchester, M., Palomares, M.L.D., Pauly, D., 2019. Global trends in carbon dioxide (CO_2) emissions from fuel combustion in marine fisheries from 1950 to 2016. Mar. Policy 107, 103382. https://doi.org/10.1016/j.marpol.2018.12.001

Griggs, D., Stafford-Smith, M., Gaffney, O., Rockström, J., Ohman, M.C., Shyamsundar, P., Steffen, W., Glaser, G., Kanie, N., Noble, I., 2013. Policy: Sustainable development goals for people and planet. Nature 495, 305–307. https://doi.org/10.1038/495305a

Henriksson, P.J.G., Guinée, J.B., Kleijn, R., de Snoo, G.R., 2012. Life cycle assessment of aquaculture systems—a review of methodologies. Int. J. Life Cycle Assess. 17, 304–313. https://doi.org/10.1007/s11367-011-0369-4

Hornborg, S., Främberg, A., 2020. Carp (Cyprinidae) Fisheries in Swedish Lakes: A Combined Environmental Assessment Approach to Evaluate Data-limited Freshwater Fish Resources as Food. Environ. Manage. 65, 232–242. https://doi.org/10.1007/s00267-019-01241-z

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M.D.M., Hollander, A., Zijp, M., van Zelm, R., 2016. ReCiPe 2016 v1.1. A harmonized life cycle impact assessment method at midpoint and endpoint level. The Netherlands.

Iribarren, D., Hospido, A., Moreira, M.T., Feijoo, G., 2011. Benchmarking environmental and operational parameters through eco-efficiency criteria for dairy farms. Sci. Total Environ. 409, 1786–1798. https://doi.org/10.1016/j.scitotenv.2011.02.013

Iribarren, D., Hospido, A., Moreira, M.T., Feijoo, G., 2010a. Carbon footprint of canned mussels from a business-to-consumer approach. A starting point for mussel processors and policy makers. Environ. Sci. Policy 13, 509–521. https://doi.org/10.1016/j.envsci.2010.05.003

Iribarren, D., Marvuglia, A., Hild, P., Guiton, M., Popovici, E., Benetto, E., 2015. Life cycle assessment and data envelopment analysis approach for the selection of building components according to their environmental impact efficiency: A case study for external walls. J. Clean. Prod. 87, 707–716. https://doi.org/10.1016/j.jclepro.2014.10.073

Iribarren, D., Moreira, M.T., Feijoo, G., 2012. Life cycle assessment of aquaculture feed and application to the turbot sector. Int. J. Environ. Res. 6, 837–848. https://doi.org/10.22059/ijer.2012.554

Iribarren, D., Moreira, M.T., Feijoo, G., 2010b. Revisiting the Life Cycle Assessment of mussels from a sectorial perspective. J. Clean. Prod. 18, 101–111. https://doi.org/10.1016/j.jclepro.2009.10.009

Iribarren, D., Moreira, M.T., Feijoo, G., 2010c. Life Cycle Assessment of fresh and canned mussel processing and consumption in Galicia (NW Spain). Resour. Conserv. Recycl. 55, 106–117. https://doi.org/10.1016/j.resconrec.2010.08.001

Iribarren, D., Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2010d. Further potentials in the joint implementation of life cycle assessment and data envelopment analysis. Sci. Total Environ. 408, 5265–5272. https://doi.org/10.1016/j.scitotenv.2010.07.078

ISO, 2006a. ISO 14040 - Environmental Management - Life Cycle Assessment - Principles and Framework.

ISO, 2006b. ISO 14044 - Environmental Management - Life Cycle Assessment - Requirements and Guidelines.

Jones, A.K., Jones, D.L., Cross, P., 2014. The carbon footprint of lamb: Sources of variation and opportunities for mitigation. Agric. Syst. 123, 97–107. https://doi.org/10.1016/j.agsy.2013.09.006

Kitts, A., Schneider, G., Lent, R., 2008. Carbon Footprint of Commercial Fishing In the Northeast United States 1–12.

Koskela, M., Vehmas, J., 2012. Defining Eco-efficiency: A Case Study on the Finnish Forest Industry. Bus. Strateg. Environ. 21, 546–566. https://doi.org/10.1002/bse.741

Laso, J., Margallo, M., Fullana, P., Bala, A., Gazulla, C., Irabien, A., Aldaco, R., 2017. Introducing life cycle thinking to define best available techniques for products:

Application to the anchovy canning industry. J. Clean. Prod. 155, 139–150. https://doi.org/10.1016/j.jclepro.2016.08.040

Laso, J., Vázquez-Rowe, I., Margallo, M., Crujeiras, R.M., Irabien, Á., Aldaco, R., 2018. Life cycle assessment of European anchovy (*Engraulis encrasicolus*) landed by purse seine vessels in northern Spain. Int. J. Life Cycle Assess. 23, 1107–1125. https://doi.org/10.1007/s11367-017-1318-7

Li, Q., Lei, J., Zhang, R., Li, J., Xing, J., Gao, F., Gong, F., Yan, X., Wang, D., Su, Z., Ma, G., 2013. Efficient decolorization and deproteinization using uniform polymer microspheres in the succinic acid biorefinery from bio-waste cotton (*Gossypium hirsutum L.*) stalks. Bioresour. Technol. 135, 604–609. https://doi.org/10.1016/j.biortech.2012.06.101

Lijó, L., Lorenzo-Toja, Y., González-García, S., Bacenetti, J., Negri, M., Moreira, M.T., 2017. Eco-efficiency assessment of farm-scaled biogas plants. Bioresour. Technol. 237, 146–155. https://doi.org/10.1016/j.biortech.2017.01.055

López-Linares, J.C., García-Cubero, M.T., Coca, M., Lucas, S., 2021. A biorefinery approach for the valorization of spent coffee grounds to produce antioxidant compounds and biobutanol. Biomass and Bioenergy 147, 106026. https://doi.org/10.1016/j.biombioe.2021.106026

Lorenzo-Toja, Y., Vázquez-Rowe, I., Chenel, S., Marín-Navarro, D., Moreira, M.T., Feijoo, G., 2015. Eco-efficiency analysis of Spanish WWTPs using the LCA+DEA method. Water Res. 68, 651–666. https://doi.org/10.1016/j.watres.2014.10.040

Lorenzo-Toja, Y., Vázquez-Rowe, I., Marín-Navarro, D., Crujeiras, R.M., Moreira, M.T., Feijoo, G., 2018. Dynamic environmental efficiency assessment for wastewater treatment plants. Int. J. Life Cycle Assess. 23, 357–367. https://doi.org/10.1007/s11367-017-1316-9

Lourguioui, H., Brigolin, D., Boulahdid, M., Pastres, R., 2017. A perspective for reducing environmental impacts of mussel culture in Algeria. Int. J. Life Cycle Assess. 22, 1266–1277. https://doi.org/10.1007/s11367-017-1261-7

Lozano, S., Iribarren, D., Moreira, M.T., Feijoo, G., 2010. Environmental impact efficiency in mussel cultivation. Resour. Conserv. Recycl. 54, 1269–1277. https://doi.org/10.1016/j.resconrec.2010.04.004

Martín-Gamboa, M., Iribarren, D., Dufour, J., 2018. Environmental impact efficiency of natural gas combined cycle power plants: A combined life cycle assessment and dynamic data envelopment analysis approach. Sci. Total Environ. 615, 29–37. https://doi.org/10.1016/j.scitotenv.2017.09.243

Medeiros, M. V., Aubin, J., Camargo, A.F.M., 2017. Life cycle assessment of fish and prawn production: Comparison of monoculture and polyculture freshwater systems in Brazil. J. Clean. Prod. 156, 528–537. https://doi.org/10.1016/j.jclepro.2017.04.059

Mesa-Alvarez, C., Ligthart, T., 2021. A social panorama within the life cycle thinking and the circular economy: a literature review. Int. J. Life Cycle Assess. 26, 2278–2291. https://doi.org/10.1007/s11367-021-01979-x

Mohan, R.R., 2018. Time series GHG emission estimates for residential, commercial, agriculture and fisheries sectors in India. Atmos. Environ. 178, 73–79. https://doi.org/10.1016/j.atmosenv.2018.01.029

Morales, A., Gullón, B., Dávila, I., Eibes, G., Labidi, J., Gullón, P., 2018. Optimization of alkaline pretreatment for the co-production of biopolymer lignin and bioethanol from chestnut shells following a biorefinery approach. Ind. Crops Prod. 124, 582–592. https://doi.org/10.1016/j.indcrop.2018.08.032

Noya, I., Aldea, X., González-García, S., M. Gasol, C., Moreira, M.T., Amores, M.J., Marín, D., Boschmonart-Rives, J., 2017. Environmental assessment of the entire pork value chain in Catalonia – A strategy to work towards Circular Economy. Sci. Total Environ. 589, 122–129. https://doi.org/10.1016/j.scitotenv.2017.02.186

Noya, I., González-García, S., Berzosa, J., Baucells, F., Feijoo, G., Moreira, M.T., 2018. Environmental and water sustainability of milk production in Northeast Spain. Sci. Total Environ. 616–617, 1317–1329. https://doi.org/10.1016/j.scitotenv.2017.10.186

Parker, R.W.R., Blanchard, J.L., Gardner, C., Green, B.S., Hartmann, K., Tyedmers, P.H., Watson, R.A., 2018. Fuel use and greenhouse gas emissions of world fisheries. Nat. Clim. Chang. 8, 333–337. https://doi.org/10.1038/s41558-018-0117-x

Parker, R.W.R., Hartmann, K., Green, B.S., Gardner, C., Watson, R.A., 2015. Environmental and economic dimensions of fuel use in Australian fisheries. J. Clean. Prod. 87, 78–86. https://doi.org/10.1016/j.jclepro.2014.09.081

Parker, R.W.R., Tyedmers, P.H., 2015. Fuel consumption of global fishing fleets: Current understanding and knowledge gaps. Fish Fish. 16, 684–696. https://doi.org/10.1111/faf.12087

Pelletier, N., 2015. Life Cycle Thinking, Measurement and Management for Food System Sustainability. Environ. Sci. Technol. 49, 7515–7519. https://doi.org/10.1021/acs.est.5b00441

Petit-Boix, A., Llorach-Massana, P., Sanjuan-Delmás, D., Sierra-Pérez, J., Vinyes, E., Gabarrell, X., Rieradevall, J., Sanyé-Mengual, E., 2017. Application of life cycle thinking towards sustainable cities: A review. J. Clean. Prod. 166, 939–951. https://doi.org/10.1016/j.jclepro.2017.08.030

Philis, G., Ziegler, F., Gansel, L.C., Jansen, M.D., Gracey, E.O., Stene, A., 2019. Comparing Life Cycle Assessment (LCA) of Salmonid Aquaculture Production Systems: Status and Perspectives. Sustain. . https://doi.org/10.3390/su11092517

Philis, G., Ziegler, F., Jansen, M.D., Gansel, L.C., Hornborg, S., Aas, G.H., Stene, A., 2021. Quantifying environmental impacts of cleaner fish used as sea lice treatments in salmon aquaculture with life cycle assessment. J. Ind. Ecol. 1–14. https://doi.org/10.1111/jiec.13118

Purvis, B., Mao, Y., Robinson, D., 2019. Three pillars of sustainability: in search of conceptual origins. Sustain. Sci. 14, 681–695. https://doi.org/10.1007/s11625-018-0627-5

Ramos, S., Vázquez-Rowe, I., Artetxe, I., Moreira, M.T., Feijoo, G., Zufía, J., 2011. Environmental assessment of the Atlantic mackerel (*Scomber scombrus*) season in the Basque Country. Increasing the timeline delimitation in fishery LCA studies. Int. J. Life Cycle Assess. 16, 599–610. https://doi.org/10.1007/s11367-011-0304-8

Ray, N.E., Maguire, T.J., Al-Haj, A.N., Henning, M.C., Fulweiler, R.W., 2019. Low Greenhouse Gas Emissions from Oyster Aquaculture. Environ. Sci. Technol. 53, 9118–9127. https://doi.org/10.1021/acs.est.9b02965

Rehman, S., Khairul Islam, M., Khalid Khanzada, N., Kyoungjin An, A., Chaiprapat, S., Leu, S.-Y., 2021. Whole sugar 2,3-butanediol fermentation for oil palm empty fruit bunches biorefinery by a newly isolated Klebsiella pneumoniae PM2. Bioresour. Technol. 333, 125206. https://doi.org/10.1016/j.biortech.2021.125206

Rodríguez, L.A., Toro, M.E., Vazquez, F., Correa-Daneri, M.L., Gouiric, S.C., Vallejo, M.D., 2010. Bioethanol production from grape and sugar beet pomaces by solid-state fermentation. Int. J. Hydrogen Energy 35, 5914–5917. https://doi.org/10.1016/j.ijhydene.2009.12.112

Ruiz-Salmón, I., Laso, J., Margallo, M., Villanueva-Rey, P., Rodríguez, E., Quinteiro, P., Dias, A.C., Almeida, C., Nunes, M.L., Marques, A., Cortés, A., Moreira, M.T., Feijoo, G., Loubet, P., Sonnemann, G., Morse, A.P., Cooney, R., Clifford, E., Regueiro, L., Méndez, D., Anglada, C., Noirot, C., Rowan, N., Vázquez-Rowe, I., Aldaco, R., 2021. Life cycle assessment of fish and seafood processed products – A review of methodologies and new challenges. Sci. Total Environ. 761. https://doi.org/10.1016/j.scitotenv.2020.144094

Samuel-Fitwi, B., Nagel, F., Meyer, S., Schroeder, J.P., Schulz, C., 2013. Comparative life cycle assessment (LCA) of raising rainbow trout (*Oncorhynchus mykiss*) in different production systems. Aquac. Eng. 54, 85–92. https://doi.org/10.1016/j.aquaeng.2012.12.002

Schwarz, D., Schoenenwald, A.K.J., Dörrstein, J., Sterba, J., Kahoun, D., Fojtíková, P., Vilímek, J., Schieder, D., Zollfrank, C., Sieber, V., 2018. Biosynthesis of poly-3-hydroxybutyrate from grass silage by a two-stage fermentation process based on an integrated biorefinery concept. Bioresour. Technol. 269, 237–245. https://doi.org/10.1016/j.biortech.2018.08.064

Sikdar, S.K., 2004. Sustainable development and sustainability metrics. AIChe 49, 1928–1932. https://doi.org/10.1002/aic.690490802

Stanitsas, M., Kirytopoulos, K., Leopoulos, V., 2021. Integrating sustainability indicators into project management: The case of construction industry. J. Clean. Prod. 279, 123774. https://doi.org/10.1016/j.jclepro.2020.123774

Su, C., Qi, L., Cai, D., Chen, B., Chen, H., Zhang, C., Si, Z., Wang, Z., Li, G., Qin, P., 2020. Integrated ethanol fermentation and acetone-butanol-ethanol fermentation using sweet sorghum bagasse. Renew. Energy 162, 1125–1131. https://doi.org/10.1016/j.renene.2020.07.119

Svanes, E., Vold, M., Hanssen, O.J., 2011. Environmental assessment of cod (*Gadus morhua*) from autoline fisheries. Int. J. Life Cycle Assess. 16, 611–624.

https://doi.org/10.1007/s11367-011-0298-2

Talekar, S., Patti, A.F., Vijayraghavan, R., Arora, A., 2018. An integrated green biorefinery approach towards simultaneous recovery of pectin and polyphenols coupled with bioethanol production from waste pomegranate peels. Bioresour. Technol. 266, 322–334. https://doi.org/10.1016/j.biortech.2018.06.072

Tamburini, E., Fano, E.A., Castaldelli, G., Turolla, E., 2019. Life cycle assessment of oyster farming in the po delta, Northern Italy. Resources 8, 1–17. https://doi.org/10.3390/resources8040170

Tamburini, E., Turolla, E., Fano, E.A., Castaldelli, G., 2020. Sustainability of Mussel (*Mytilus galloprovincialis*) farming in the Po River delta, northern Italy, based on a life cycle assessment approach. Sustain. 12. https://doi.org/10.3390/su12093814

Tirpanalan, Ö., Reisinger, M., Smerilli, M., Huber, F., Neureiter, M., Kneifel, W., Novalin, S., 2015. Wheat bran biorefinery – An insight into the process chain for the production of lactic acid. Bioresour. Technol. 180, 242–249. https://doi.org/10.1016/j.biortech.2015.01.021

Tone, K., 2011. Slacks-Based measure of efficiency. Int. Ser. Oper. Res. Manag. Sci. 164, 195–209. https://doi.org/10.1007/978-1-4419-6151-8_8

United Nations Environment Program, 2021. Making peace with Nature. A scientific blueprint to tackle the climate, biodiversity and pollution emergencies.

Vásquez, J., Aguirre, S., Fuquene-Retamoso, C.E., Bruno, G., Priarone, P.C., Settineri, L., 2019. A conceptual framework for the eco-efficiency assessment of small- and medium-sized enterprises. J. Clean. Prod. 237, 117660. https://doi.org/10.1016/j.jclepro.2019.117660

Vázquez-Rowe, I., Hospido, A., Moreira, M.T., Feijoo, G., 2012a. Best practices in life cycle assessment implementation in fisheries. Improving and broadening environmental assessment for seafood production systems. Trends Food Sci. Technol. 28, 116–131. https://doi.org/10.1016/j.tifs.2012.07.003

Vázquez-Rowe, I., Iribarren, D., 2015. Review of life-cycle approaches coupled with data envelopment analysis: launching the CFP + DEA method for energy policy making. ScientificWorldJournal. 2015, 813921. https://doi.org/10.1155/2015/813921

Vázquez-Rowe, I., Iribarren, D., Moreira, M.T., Feijoo, G., 2010a. Combined application of life cycle assessment and data envelopment analysis as a methodological approach for the assessment of fisheries. Int. J. Life Cycle Assess. 15, 272–283. https://doi.org/10.1007/s11367-010-0154-9

Vazquez-Rowe, I., Kahhat, R., Quispe, I., Bentín, M., 2016. Environmental profile of green asparagus production in a hyper-arid zone in coastal Peru. J. Clean. Prod. 112, 2505–2517. https://doi.org/10.1016/j.jclepro.2015.09.076

Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2013a. Carbon footprint analysis of goose barnacle (*Pollicipes pollicipes*) collection on the Galician coast (NW Spain). Fish. Res. 143, 191–200. https://doi.org/10.1016/j.fishres.2013.02.009

Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2012b. Environmental assessment of frozen common octopus (*Octopus vulgaris*) captured by Spanish fishing vessels in the Mauritanian EEZ. Mar. Policy 36, 180–188. https://doi.org/10.1016/j.marpol.2011.05.002

Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2011. Life Cycle Assessment of fresh hake fillets captured by the Galician fleet in the Northern Stock. Fish. Res. 110, 128–135. https://doi.org/10.1016/j.fishres.2011.03.022

Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2010b. Life cycle assessment of horse mackerel fisheries in Galicia (NW Spain): Comparative analysis of two major fishing methods. Fish. Res. 106, 517–527. https://doi.org/10.1016/j.fishres.2010.09.027

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. https://doi.org/10.1016/j.scitotenv.2013.12.099

Vázquez-Rowe, I., Villanueva-Rey, P., Iribarren, D., Teresa Moreira, M., Feijoo, G., 2012c. 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. https://doi.org/10.1016/j.jclepro.2011.12.039

Vázquez-Rowe, I., Villanueva-Rey, P., Mallo, J., De La Cerda, J.J., Moreira, M.T., Feijoo, G., 2013b. Carbon footprint of a multi-ingredient seafood product from a business-to-business perspective. J. Clean. Prod. 44, 200–210. https://doi.org/10.1016/j.jclepro.2012.11.049

Vázquez-Rowe, I., Villanueva-Rey, P., Moreira, M.T., Feijoo, G., 2016. Opportunities and challenges of implementing life cycle assessment in seafood certification: a case study for Spain. Int. J. Life Cycle Assess. 21, 451–464. https://doi.org/10.1007/s11367-016-1043-7

Vélez-Henao, J.A., Weinland, F., Reintjes, N., 2021. Life cycle assessment of aquaculture bivalve shellfish production — a critical review of methodological trends. Int. J. Life Cycle Assess. 26, 1943–1958. https://doi.org/10.1007/s11367-021-01978-y

Vergé, X.P.C., Maxime, D., Dyer, J.A., Desjardins, R.L., Arcand, Y., Vanderzaag, A., 2013. Carbon footprint of Canadian dairy products: Calculations and issues. J. Dairy Sci. 96, 6091–6104. https://doi.org/10.3168/jds.2013-6563

Wang, A., Wang, Y., Jiang, T., Li, L., Ma, C., Xu, P., 2010. Production of 2,3-butanediol from corncob molasses, a waste by-product in xylitol production. Appl. Microbiol. Biotechnol. 87, 965–970. https://doi.org/10.1007/s00253-010-2557-8

WCED, 1987. Our common future. World Commission on Environment and Development. Oxford University Press. Oxford (UK).

Weidema, B.P., Pizzol, M., Schmidt, J., Thoma, G., 2018. Attributional or consequential Life Cycle Assessment: A matter of social responsibility. J. Clean. Prod. 174, 305–314. https://doi.org/10.1016/j.jclepro.2017.10.340

Weidema, B.P., Simas, M.S., Schmidt, J., Pizzol, M., Løkke, S., Brancoli, P.L., 2020. Relevance of attributional and consequential information for environmental product labelling. Int. J. Life Cycle Assess. 25, 900–904. https://doi.org/10.1007/s11367-019-01628-4

Werner, L.B., Flysjö, A., Tholstrup, T., 2014. Greenhouse gas emissions of realistic dietary choices in Denmark: The carbon footprint and nutritional value of dairy products. Food Nutr. Res. 58, 1–16. https://doi.org/10.3402/fnr.v58.20687

Yaashikaa, P.R., Senthil Kumar, P., Varjani, S., 2022. Valorization of agro-industrial wastes for biorefinery process and circular bioeconomy: A critical review. Bioresour. Technol. 343, 126126. https://doi.org/10.1016/j.biortech.2021.126126

Yacout, D.M.M., Soliman, N.F., Yacout, M.M., 2016. Comparative life cycle assessment (LCA) of Tilapia in two production systems: semi-intensive and intensive. Int. J. Life Cycle Assess. 21, 806–819. https://doi.org/10.1007/s11367-016-1061-5

Zhang, B., Zhan, B., Bao, J., 2021. Reframing biorefinery processing chain of corn fiber for cellulosic ethanol production. Ind. Crops Prod. 170, 113791. https://doi.org/10.1016/j.indcrop.2021.113791

Zhou, X., Xu, Y., 2019. Integrative process for sugarcane bagasse biorefinery to co-produce xylooligosaccharides and gluconic acid. Bioresour. Technol. 282, 81–87. https://doi.org/10.1016/j.biortech.2019.02.129

Ziegler, F., Winther, U., Hognes, E.S., Emanuelsson, A., Sund, V., Ellingsen, H., 2013. The Carbon Footprint of Norwegian Seafood Products on the Global Seafood Market. J. Ind. Ecol. 17, 103–116. https://doi.org/10.1111/j.1530-9290.2012.00485.x

SECTION II

CIRCULAR ECONOMY IN THE AGRICULTURAL SECTOR

Chapter 3

Integrated evaluation of wine lees valorisation to produce value-added products

Summary

As a symbol of a high-quality product, wine can be considered a reference product within the primary production sector, however, the winemaking process involves the management of waste generated at different stages. This chapter conducts an environmental assessment of the valorisation of wine lees to obtain value-added products from a life-cycle perspective. Lees are essentially the dead yeast cells leftover from the fermentation process and are one of the main solid wastes produced in the wine sector. The proposed process is based on a biorefinery model to produce four value-added products: bioethanol, calcium tartrate, an antioxidant-rich extract, and a protein-rich solid fraction. Most of the activities related to this valorisation scheme are based on physical processes. It was shown that steam consumption is the main hot spot, reaching 85.7% of the impact on Fossil Depletion and 85.3% on Climate Change. Comparing this system with other processes that produce antioxidants from different raw materials, the valorisation of wine lees showed a better environmental profile throughout the entire life cycle, due to the fact that it does not require a large consumption of electricity or chemicals. However, there is still room for improvement, and future research should focus on optimizing the extraction of antioxidants from wine lees using two-stage aqueous systems, ultrasound-assisted extraction, or microwaves.

This chapter, under the consent of the publisher, was redrafted after: Cortés, A., Moreira, M. T., Feijoo, G. (2019). Integrated evaluation of wine lees valorization to produce value-added products. Waste Management, 95, 70-77. DOI: https://doi.org/10.1016/j.wasman.2019.05.056. More information related to this article can be found in Appendix III.

OUTLINE

3.1. INTRODUCTION

Low efficiency in food production and processing has been recognised as a major issue, beyond environmental indicators related to environmental impacts. According to a FAO report about food loss and food waste (FAO, 2011), up to 33% of the food produced for human consumption is lost or wasted along the supply chain. But apart from the food waste, it should be considered all the resources needed for its production, such as water, energy, chemicals, or fuels. In addition, according to data estimated by United Nations (2017), in relation to population growth, world's population is expected to increase to 8.6 billion by 2030, which will probably aggravate the problem of food waste.

Biorefinery is a clear example of the change of paradigm in the framework of circular economy and sustainable development. Biorefineries represent the transition from oil refineries to sustainable systems based on the valorisation of waste flows with the aim of producing value-added compounds such as biogas, electricity, chemical products or biomaterials (Cherubini, 2010). Following this principle, food waste can be valorised through different technologies, such as anaerobic digestion to generate bio-hydrogen and bio-methane (Algapani et al., 2019), co-composting with other types of organic waste for the production of bioenergy and fertilisers (Vico et al., 2018), conversion into animal feed (Makkar, 2018) or recovery of sugars, organic acids, pigments, fibre, proteins, oils, antioxidants and vitamins from food waste (García-Herrera et al., 2010). In this sense, an integrated biorefinery approach is recognised as a noteworthy solution for the valorisation of winery waste (Ky and Teissedre, 2015).

Within the primary sector, wine production is becoming increasingly important as a symbol of a quality product, with a growing influence on exports from producing countries. According to data provided by the International Organization of Vine and Wine, world wine production in 2015 was approximately 280 million hectolitres of 78 million tonnes of grapes (Guerini Filho et al., 2018). World wine production is dominated by Italy, France and Spain, which together account for 48% of total production (OIV, 2018). From this perspective, the wine sector should be taken as a reference point in the EU strategy within the primary production sector (Christ and Burritt, 2013). It is also relevant to mention that in Galicia, the wine sector has a long tradition with different varieties of high oenological quality, as evidenced by mentions of excellence and awards (Vázquez-Rowe et al., 2012b).

The winemaking process comprises an wide sequence of activities (Escribano-Viana et al., 2018), from grape growing, harvesting, fermentation and maturation in the winery to the management of waste generated at each stage of the process. The main effluents from the wine sector are wastewater and organic solid waste (Ruggieri et al., 2009). Solid organic waste includes wine lees and grape marc; wine lees are essentially the dead yeast cells leftover from the fermentation process; while grape marc consists of the seeds, pulp and stacks that reman after pressing the grapes (Figure 3.1). In general, the volume of waste generated is around 20-30% of total wine production, which is a meaningful percentage (Zabaniotou et al., 2018). However, this value is lower than other food industries, where the generated waste can account for up to 60% of the initial products (Notarnicola et al., 2017).

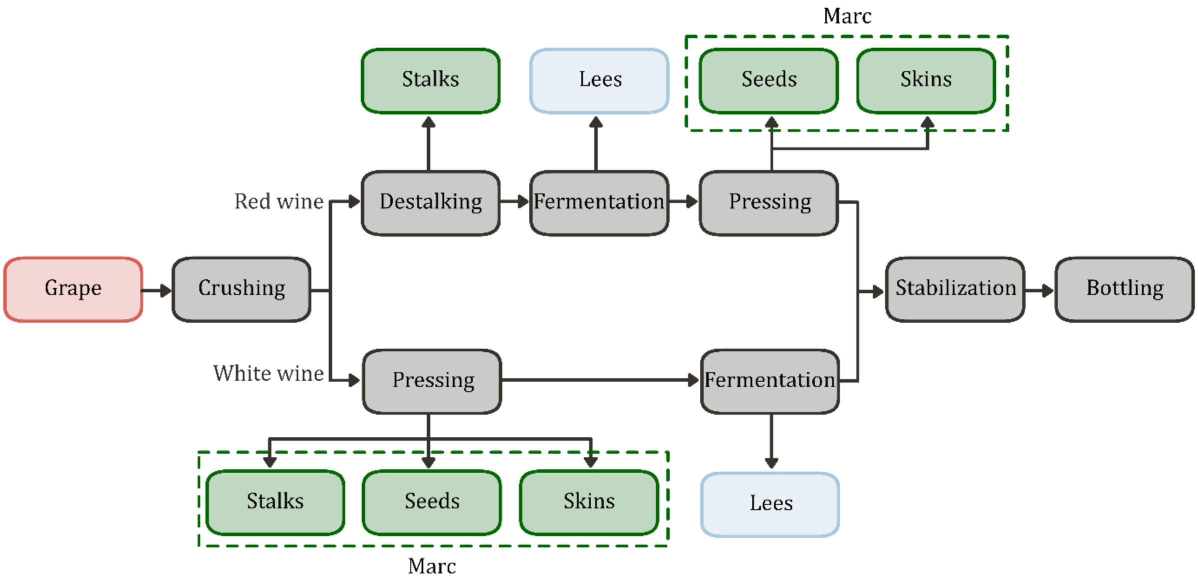


Figure 3.1. Outline of winemaking activities and origin of the generated waste, lees, and marc.

In order to improve the overall efficiency and environmental impact of the winemaking process, actions have been identified that allow for the minimisation, management and effective recovery of waste streams from a circular economy perspective (Musee et al., 2007). The approach of circular economy grants and consolidates the value of each element of the productive chain and deepens the awareness of action, essential to achieve a real change towards sustainability, with efficient use of resources and valorisation of by-products and wastes. In this framework, Gullón et al. (2018) analysed the environmental impacts of different routes for the valorisation of vine shoots. Poveda et al. (2018) proposed revaluing the by-products of winemaking, grape marc and stems as a source of natural preservatives. Nayak et al. (2018) developed a method for recovering polyphenols from exhausted grape marc through activated carbon. Zhang et al. (2017) compared two methods for the valorisation of grape marc, which is the major component of wine production waste, to add value to economic and environmental balance of the overall process; these processes were combustion to produce electricity, and pyrolysis to produce biogas, bio-oil, and bio-char.

The detailed analysis of the wine lees fraction presents high concentrations of macronutrients and polyphenols and low concentrations of micronutrients and heavy metals (Devesa-Rey et al., 2011). Moreover, the presence of other compounds of potential interest such as polyphenols and antioxidants identifies this stream as an ideal candidate to be valorised (Dimou et al., 2015; Kopsahelis et al., 2018; Martinez et al., 2016). These valorisation options can be evaluated according to their environmental performance. Several environmental assessment methods can be found in literature, as material flow accounting, input-output analysis, material intensity analysis or life cycle assessment among others (Vandermeersch et al., 2014). The latter seems to be the best choice since it can consider the full life cycle of the target product or process (cradle-to-gate or cradle-to-grave).

The objective of this chapter seeks to delve into the different strategies for valuing one of the two main winemaking-derived waste, wine lees; proposing the identification of critical points in the environmental profile of the process under study, prior to its development and marketing of the products obtained. The function of the system under study is the use of these wine lees to produce some bio-based products with marketable added value. It is therefore a question of identifying the most suitable process alternatives in a system under development, which may suffer from limitations in terms of data availability, but it may also make it possible to establish a roadmap in the search for viable options from a techno-economic and environmental point of view.

3.2. MATERIALS AND METHODS

3.2.1. Definition of scope and system boundaries

The selection of the functional unit in biorefinery studies is made on the basis of three possible options, i.e. total flow of raw materials, quantity of a single target product or the combination of products (Khoshnevisan et al., 2018). The FU considered in this chapter was 1 tonne of wine lees processed in the winery facility. This feedstock-based FU is consistent with the choice of other similar studies, in which a similar FU was chosen because of their multiple-output nature (Lam et al., 2018; Vaskan et al., 2018). The production plan was evaluated considering all the processes from the production of raw materials to the final products obtained from the wine lees, in such a way that the processes in the winery are analysed, mainly those associated with the production of raw materials, electricity, fertilisers, chemical products and water, as well as the consumption of fuel used in the transport of materials.

The system boundaries were divided into two subsystems: SS1. Wine production and SS2. Processing plant. Figure 3.2 shows the block diagram of the process, identifying the system boundaries, the subsystems and the main inputs and outputs. No infrastructure process was taken into account since the environmental impacts of construction, installation, etc. have been considered negligible during the useful life of the facility. This assumption is a common practice in other life cycle assessment studies of different biorefineries (Jeswani et al., 2015; Karlsson et al., 2014). As far as storage processes, it was considered unnecessary, since only a small warehouse within the facilities is needed to store the wine lees at room temperature, so it is included within the infrastructure processes.

Subsystem 1 was divided into two sections: SS1.1. Viticulture comprises different activities carried out in the agricultural phase of wine production, including fertilisation, field operations or soil management and SS1.2. Vinification includes the processing of grapes in the winery: wine production, bottling and packaging (Vázquez-Rowe et al., 2012b). Wine lees are a co-product generated during this process, which will be further processed in Subsystem 2. It is important to note that the main product of this subsystem is wine; however, since the objective of this chapter is to assess the valorisation of lees, wine production was excluded from the system boundaries. The impacts derived from SS1 were calculated to determine the environmental burdens associated with lees generation.

Subsystem 2 is the industrial process in which wine lees are valorised to produce four added-value products, in this case bioethanol, calcium tartrate, an extract rich in antioxidants and solid fraction rich in protein, which can be marketed with an economic return. Wine lees with a content of 62.9% (w/w) water, 5.7% (w/w) ethanol and 31.4% (w/w) solids (Dimou et al., 2016) fed the biorefinery at room temperature (25°C). The first step is the separation of the liquid and solid fractions by centrifugation of the wine lees in a disc centrifuge. It is possible to recover the residual fraction of ethanol present in the liquid fraction by distillation at 100°C. In this step, the product stream of the bottom contains mainly water and is sent to treatment. The solid fraction is then sent to an extraction tank in which it is mixed with the ethanol recovered in the previous stage. The recovery of ethanol implies a significant reduction in the cost associated with this stage since it is available in the original wine lees (Dimou et al., 2016). A new solid-liquid separation is then conducted in a disc centrifuge so that the liquid fraction is fed to another distillation column in which the antioxidant-rich extract is separated as a bottom product. The removal of water from this stream takes place in an evaporator and in a spray dryer which works with high pressure steam. The solid phase from the first solid-liquid separation is fed to an acidification process, where it is mixed with HCl, and the tartrate salt (insoluble in water) is transformed into tartaric acid (water-soluble). Afterwards this stream is fed into another solid-liquid separation where yeast cells are obtained from the solid fraction. The liquid stream from this stage is mixed with calcium carbonate ($CaCO_3$) and calcium chloride ($CaCl_2$), transforming water-soluble tartaric acid into water-insoluble calcium tartrate. Calcium tartrate is obtained as a solid product after the drying stage.

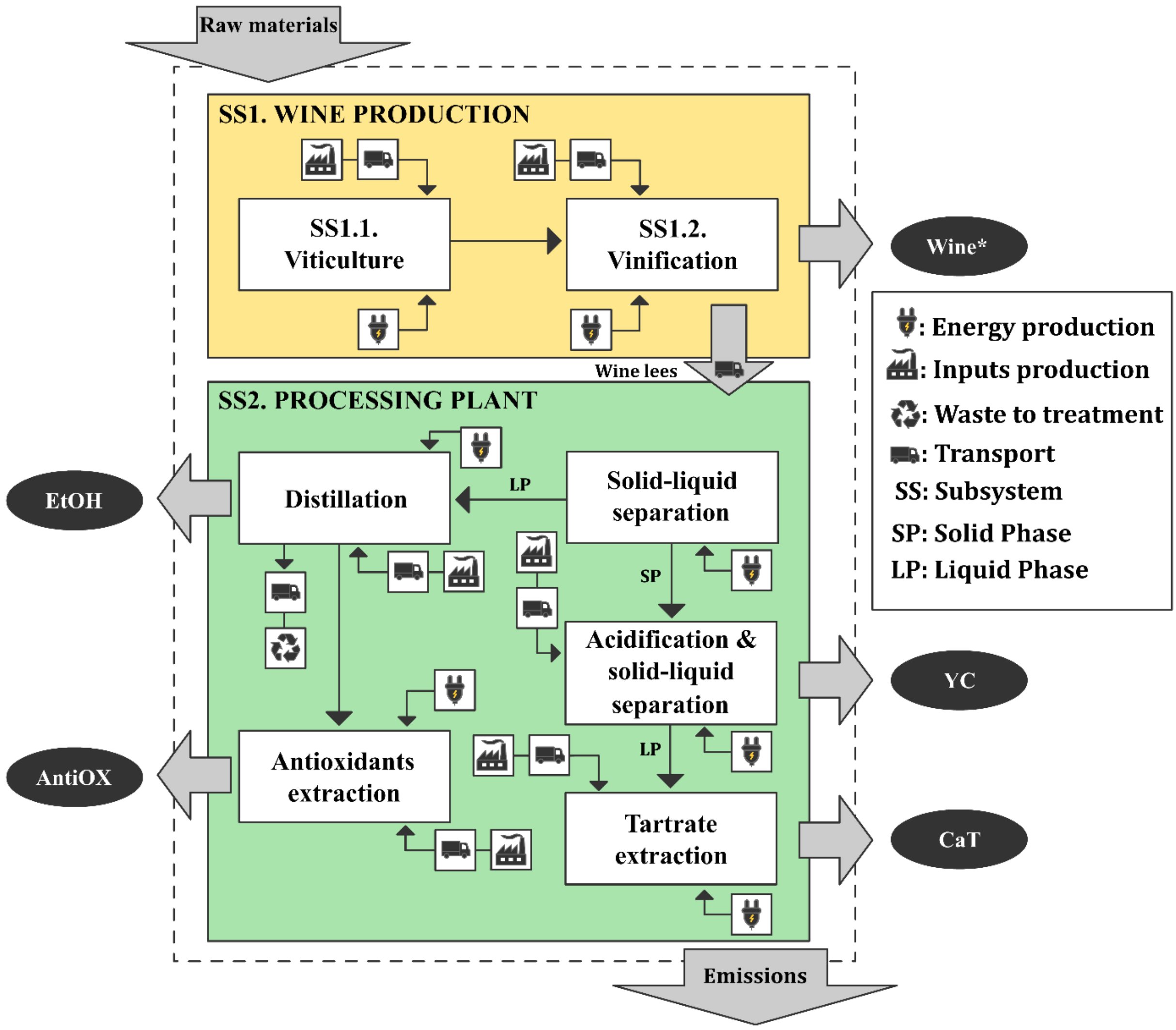


Figure 3.2. System boundaries of the wine lees biorefinery to produce value-added products. Caption: EtOH: Bioethanol; CaT: Calcium tartrate; YC: Yeast cells; AntiOX: Antioxidants.

3.2.2. Data acquisition

It is important to note that all the data used for the assessment were taken from complementary studies that consider all stages of the life cycle of wine production. The data for the construction of the life cycle inventory of SS1. Wine production were extracted from Vázquez-Rowe et al. (2012a), where the vine cultivation and the winery processing stages were analysed while the data corresponding to the valorisation of the wine lees fraction were taken from Dimou et al. (2016), where a biorefinery with a processing capacity of 500 kg/h of wine lees was considered. The environmental assessment was carried out with a common perspective defined on the basis of an identical functional unit for both subsystems: 1 tonne of wine lees. The inventory data for the production of all system inputs from background processes were taken from the Ecoinvent® database. These inputs include the production of the different chemical products necessary for the extraction of calcium tartrate, the electricity consumed in the different stages, the fertilisers for the vine cultivation and any other type of raw material or resource. An average transport distance of 800 km was quantified within continental Europe for chemical products (Pérez-López et al., 2014b) and 100 km of average distance for the transport of the wine lees from the winery to the processing plant were assumed (Hajjaji et al., 2013).

3.2.3. Life cycle inventory

In this chapter, a cradle-to-gate approach has been proposed, including stages from the cultivation of the vine to the activities of the winery and the valorisation of the wine lees. Considering that the economic allocation reflects the function and objective of the production process, which is obviously the marketing of a main product: wine and the associated economic revenues, mass or energy-based allocations were discarded (Rugani et al., 2013).

Since wine lees are a co-product derived from the production of wine, it is necessary to make a cost estimate as a requirement for an economic allocation that allows the environmental impacts corresponding to each fraction to be assessed. Thus, the selling price of a bottle of wine produced after SS1 was compared with the potential selling price of the different products obtained from the valorisation route (extract rich in antioxidants, ethanol, yeast cells and calcium tartrate). In this sense, a market price of 4 € for a 750 mL bottle of wine with designation of origin was considered. Considering that the production of a bottle of this wine generates 11.48 mL of wine lees, the sale of these products generates a profit of 0.022 € and when comparing the contributions of both products in terms of their market value, the economic allocation factors are 99.45% for wine and 0.55% for wine lees.

3.2.4. Life cycle impact assessment: methodology

SimaPro 8.5.2 (PRé Consultants, 2017) has been the software used for the implementation of the Life Cycle Inventory. To analyse the inputs and outputs of the LCI, the Classification and Characterisation guidelines defined by ISO were followed. In this phase, in order to translate the long list of life cycle inventory results into a small number of environmental impact indicators, the ReCiPe Midpoint methodology was used. This method provides a common framework in which both mid-point and end-point indicators can be used (Goedkoop et al., 2009). The characterisation factors reported by the ReCiPe Midpoint methodology were applied, and the potential impact categories evaluated were Climate Change (CC), Ozone Layer Depletion (OD), Terrestrial Acidification (TA), Freshwater Eutrophication (FE), Marine Eutrophication (ME), Human Toxicity (HT), Photochemical Oxidant Formation (POF), Terrestrial Ecotoxicity (TET), Freshwater Ecotoxicity (FET), Marine Ecotoxicity (MET) and Fossil Fuel Depletion (FD).

3.3. RESULTS AND DISCUSSION

3.3.1. Quantitative analysis of inputs and outputs

LCI shows that the consumption of different pesticides is very high in the viticulture stage, which is in line with other studies on different crops (Caldeira et al., 2018; Liang et al., 2019). Furthermore, the consumption of organic fertilisers is also remarkable, leading to relevant nitrate emissions, so the impact on the eutrophication categories is expected to be considerable. As for the winemaking phase, the main inputs are electricity and chemicals such as NaOH and SO_2. In this system, CO_2 emissions due to wine fermentation were calculated, but excluded from the environmental assessment, as was considered as biogenic CO_2. LCI has also allowed to quantify the main inputs and outputs of the wine lees valorisation system, highlighting the consumption of high-pressure steam and the production of four value-added products. Table 3.1 shows the life cycle inventory of the wine lees valorisation system per FU. It is important to note that the inventory tables for subsystems SS1.1. Viticulture and SS1.2. Vinification data are included in Tables A.1 and A.2 of the Appendix I, as they are not the final core in this chapter but influencing the final score of SS2 through burdens allocation.

Table 3.1. Inventory for Subsystem 2. Processing plant per FU.

Inputs from Technosphere			
Materials	**kg**	**Materials**	**m³**
Wine lees	1,000	Cooling water	224.46
Calcium chloride	31.58	**Energy**	**kWh**
Calcium carbonate	31.58	Electricity	129.94
Hydrochloric acid	110.90	**Transport**	**t·km**
Low pressure steam	5,105	Chemicals	139.26
		Wine lees	100.00
Outputs to Technosphere			
Products	**kg**	**Products**	**kg**
Bioethanol	28.22	Calcium tartrate	58.50
Antioxidants	6.78	Yeast cells	241.00
Waste	**m³**		
Wastewater	0.56		

The main product obtained in this process is the extract rich in antioxidants. This extract, as established in Dimou et al. (2016), has a lower total polyphenol content than other studies (26.1 mg of gallic acid equivalent per g of wine lees) as this extraction is not optimised and the product could be even purer. However, it is important to bear in mind that most of the operations carried out are physical, such as solid-liquid extraction or centrifugation, while other studies involve complex operations, such as ultrasound or microwave assisted extraction (Castro-López et al., 2017; Mohammadpour et al., 2019; Pereira et al., 2017), so that the use of chemical products is low, as only 174 kg per tonne of treated lees are consumed.

3.3.2. Analysis of processing plant energy requirements

Electricity and steam consumption reported in this chapter correspond to the operation of the plant and is detailed by equipment in Table 3.2. Although the total electricity consumption is not too high (125 kWh per FU), more than 95% of this consumption corresponds to disc centrifuges, which are used to separate the solid phase from the liquid phase. This separation process is essential to obtain value-added products, as most of the treatments carried out are mainly physical, as stages of distillation, separation, evaporation, and spray drying with compressed hot air.

Table 3.2. Analysis of the electricity, steam, and cooling water consumption of Subsystem 2. Processing plant per FU. Electricity is expressed in kWh, steam in kg and cooling water in m^3.

Unit Operation	Electricity	Unit Operation	Steam	Unit Operation	Cooling water
Disc centrifuges	**119.19**	**Heat exchangers**	**5,104.84**	**Heat exchangers**	**224.46**
CF-101	29.80	E-102	819.44	E-101	33.61
CF-102	29.80	E-104	3994.44	E-103	190.85
CF-103	29.80	E-105	238.19		
CF-104	29.79	E-106	39.58		
Mixing tanks	**4.68**	E-107	13.19		
V-101	2.60				
V-102	1.20				
V-103	0.88				
Blower	**6.06**				
C-101	4.06				
C-102	2.00				

Steam consumption is very high, more than 5 tonnes of steam per tonne of valorised wine lees. In the process, steam is used in distillation columns (units E-102 and E-104) while in the unit E-105, steam is used to evaporate water from the antioxidant-rich stream. Finally, in the E-106 and E-107 exchangers, steam is used to heat the compressed air that will be used in the spray dryers to remove the remaining water from the calcium tartrate and the antioxidant-rich extract. With this consumption, together with the fact that obtaining steam is an activity with high energy requirements (Nieuwlaar et al., 2015), it is possible to anticipate that the environmental impact derived from the use of steam will be important.

3.3.3. Environmental characterisation of wine lees valorisation process

The environmental impacts expressed through different impact categories are presented in the characterisation results, which are summarised in Table 3.3. Specifically, SS1. Wine production is the main contributor to MET, FET, POF, HT, ME, and TA categories. In MET and TET, SS1 represents almost 90% of the total environmental impact, which is related to the treatment of solid waste produced during winemaking. SS1.1 includes different activities of the agricultural phase of wine production such as soil management, field operations or fertilisation. This phase is clearly the hotspot in ME, POF and TA categories with remarkable contributions of 67.9%, 56.8% and 52.7% respectively. It should be noted here that the use of fossil fuels for the operation of machinery such as broadcasters and rotary cultivators and the use of compost for fertilisation cause emissions of nitrogen oxides that affect POF category. Regarding the environmental impacts in the ME category, the application of fertilisers in the agricultural phase of wine production involved the greatest relative impact of this category. These results are in agreement with the results obtained in another study in which an LCA of red wine production from Catalonia was performed (Meneses et al., 2016). Vinification process has a better environmental profile than the viticulture stage although there are no major differences between these two subsystems in HT category. This is due to the fact that the main contributors to the environmental impact in this category are the emissions of heavy metals into the atmosphere derived from the consumption of fossil fuels for the operation of equipment, transport, and steam production.

Table 3.3. Impact assessment results associated with the valorisation of 1 tonne of wine lees.

Impact category	Unit	SS1.1	SS1.2	SS2	Total
CC	kg CO_2 eq	$8.79·10^2$	$3.26·10^2$	$1.33·10^3$	$2.54·10^3$
OD	kg CFC-11 eq	$1.14·10^{-4}$	$2.09·10^{-5}$	$1.52·10^{-4}$	$2.87·10^{-4}$
TA	kg SO_2 eq	6.52	1.00	4.85	12.37
FE	kg P eq	0.15	0.05	0.22	0.42
ME	kg N eq	2.52	1.05	0.14	3.72
HT	kg 1,4-DB eq	$1.85·10^2$	$1.86·10^2$	$2.43·10^2$	$6.14·10^2$
POF	kg NMVOC	4.25	0.68	2.56	7.48
TET	kg 1,4-DB eq	$3.67·10^{-2}$	$8.61·10^{-3}$	$1.08·10^{-1}$	$1.53·10^{-1}$
FET	kg 1,4-DB eq	5.36	57.81	7.73	70.88
MET	kg 1,4-DB eq	5.25	49.77	7.01	62.03
FD	kg oil eq	$1.10·10^2$	46.85	$4.35·10^2$	$5.92·10^2$

In relation to SS2, this subsystem is the main contributor in FD and TET impact categories. It reaches a maximum contribution of 75% in the FD category, associated with the use of natural gas for steam production. In the TET category, it accounts for 70% of the impact due to emissions of heavy metals into the atmosphere linked to the use of fossil fuels, either in transport or in the production of high temperature steam. However, regarding FE, OD and CC categories, the difference between the two subsystems is minimal. Focusing on CC, direct emissions into the atmosphere associated with the fermentation process were quantified in the winemaking process. However, direct CO_2 emissions from SS1 should not be considered as fossil carbon, but as biogenic CO_2. If these emissions were quantified as fossil carbon, SS1 contribution to the CC category would increase to 48.3%, which would mean a global value of 2,570 kg of CO_2 eq per tonne of wine lees.

Apart from the environmental burdens related to SS1, as the main objective of this chapter is to assess the environmental profile of wine lees valorisation, Figure 3.3 shows the relative contribution of each component related with the wine lees biorefinery. The use of high temperature steam is the most impactful component in most of the impact categories. In this sense, steam consumption highlights in FD (85.7%), CC (85.3%), TA (79.4%) and POF (76.7%) categories, all of which are associated with the high consumption of fossil fuels for the production of steam as well as the associated emissions of GHG, SO_2 and NOx.

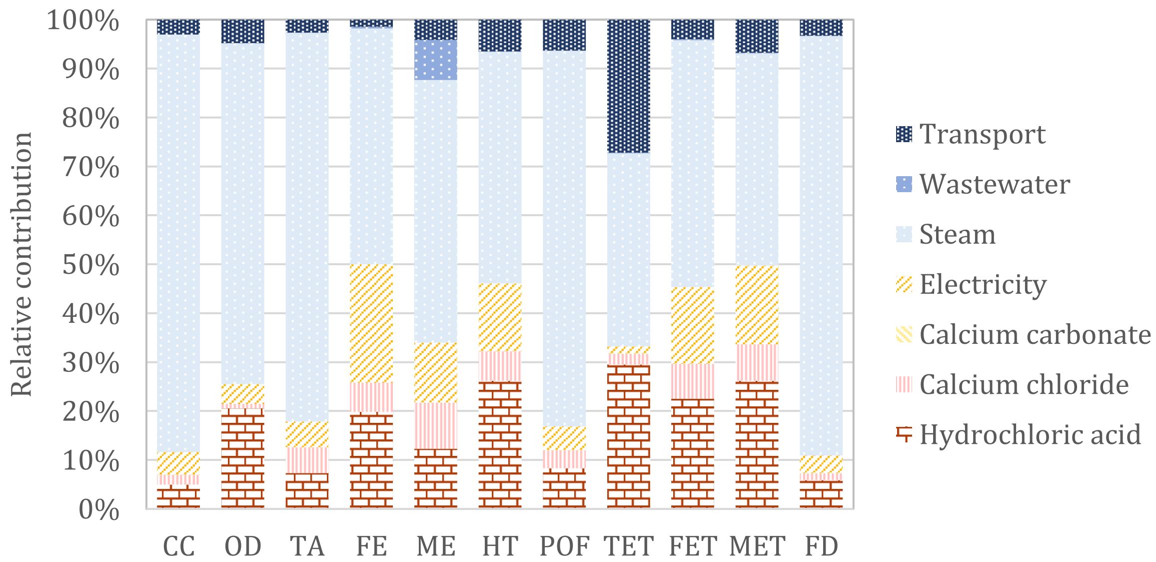


Figure 3.3. Relative contributions (in %) by component in the overall biorefinery production process

HCl becomes the second most contributing element with an average of 16.7%, standing out in the categories of toxicity, either Human (HT) or Ecosystem (TET, MET and FET); due to emissions of heavy metals associated with the production of HCl, H_2SO_4 and Cl_2 compounds. Electricity has a low impact in almost all impact categories, which is attributed to a low-moderate consumption (Table 3.2). The contribution on FE (24.2%), ME (12.1%), HT (13.9%), FET (15.6%) and MET (16.1%) categories are remarkable values. The impact of $CaCl_2$ and $CaCO_3$ are of much lesser importance with average contributions of 4.7% and 0.03% respectively. Transport presents a considerably uniform distribution of environmental impacts in all categories, with contributions almost always lower than 7%, except in the case of the TET category, in which a relative contribution of around 27.2% is reached. This can be explained by the fact that this category gives more weight to emissions to air and soil (which are abundant in transport processes); in contrast, other toxicity categories such as MET and FET are more dependent on water emissions. Wastewater treatment contributions are practically negligible except for a percentage of 8.2% in ME category, due to nitrogen compounds directly discharged into the water.

3.3.4. Sensitivity analysis. Effect of the selection of the allocation factors and Functional Unit on the environmental profile

Considering that the chosen FU was based on a feedstock-processing perspective, the results show what is the environmental performance of the process. If the objective of the analysis were to determine the environmental impacts related to the production of a particular products, it seems coherent to change the FU to a product-based one. Thus, the choice of FU is a critical point in any LCA study as it is a subjective action that must be consistent with the objectives of the study. In this case the function of the system is the treatment of a waste, but this system allows to obtain four different added-value products. The extract rich in antioxidants is the product of greatest interest for its potential applications in the food industry, cosmetics and pharmaceutical industry (Szabo et al., 2018). Therefore, the new FU for analysis was selected as 1 kg of antioxidant-rich extract.

In addition to the choice of the FU, the allocation of impacts is fundamental, especially in multi-production processes. If mass or economic allocation is followed, the impacts assigned to each product are different, so mass and economic allocation factors were calculated by the quantity produced of each element and its potential market price obtained from scientific publications (Table 3.4).

Table 3.4. Computation of the mass and economic allocation factors for SS2. Processing plant.

Product	Production[a]	Market price	Mass allocation	Economic allocation
Bioethanol	28.22 kg	0.67 €/kg[b]	8.4%	1.3%
Calcium tartrate	58.50 kg	4.41 €/kg[c]	17.5%	17.3%
Yeast cells	241.00 kg	0.88 €/kg[c]	72.0%	14.2%
Antioxidants	6.78 kg	147.67 €/kg[d]	2.0%	67.6%

[a] Results per tonne of wine lees.
[b] (Joelsson et al., 2016).
[c] (Dimou et al., 2016).
[d] (Vieira et al., 2013).

These studies show how the market prices of each of the products obtained in the biorefinery varies and how the mass and economic allocations factors fluctuate accordingly. Joelsson et al. (2016) conducted research on the production of biogas and bioethanol from wheat straw impregnated with acetic acid. Beyond the experimental study, they also performed a techno-economic evaluation. In this paper a bioethanol price of 0.57-0.68 €/kg is estimated. In Dimou et al. (2016) a techno-economic evaluation of the wine lees biorefinery was carried out and the market cost for yeast cells and calcium tartrate were estimated as 1 $/kg and 5 $/kg, respectively. These prices can be converted to euros and are equivalent to 0.88 €/kg and 4.41 €/kg, respectively. Finally, in Vieira et al (2013) a chemical and economic evaluation of antioxidants extracted from pulp of *Euterpe edulis* was conducted. The manufacturing costs of the crude extracts obtained in this paper were 165.34 $/kg, which is equivalent to 147.67 €/kg. These manufacturing costs were assumed as the market cost of this extract to obtain an economic return.

The effect of the alternative FU and allocation factors on the environmental profiles is shown in Table 3.5. When allocation factors (mass or economic) are used, a decrease in environmental impact is always observed, as the environmental impact is distributed among the different products. As the amount of the extract rich in antioxidants is low, in the case of mass allocation its environmental impact is small. However, this is not realistic, as it is the product of greatest interest. Consequently, as shown in Table 3.5, the economic allocation distributes the environmental impacts among the products more accurately. In this sense, the production of 1 kg of extract rich in antioxidants is associated with the emission of, among others, 251.3 kg of CO_2 eq in CC category and 58.6 kg of oil eq in FD category.

Table 3.5. Environmental results for each impact category considering the production of 1 kg of antioxidants-rich extract with no allocation, mass, and economic allocation factors.

Impact category	Units	No allocation	Mass allocation	Economic allocation
CC	kg CO_2 eq	374.1	7.6	251.3
OD	kg CFC-11 eq	$4.2·10^{-5}$	$8.6·10^{-7}$	$2.8·10^{-5}$
TA	kg SO_2 eq	1.8	$3.7·10^{-2}$	1.2
FE	kg P eq	$6.2·10^{-2}$	$1.3·10^{-3}$	$4.2·10^{-2}$
ME	kg N eq	$5.5·10^{-1}$	$1.1·10^{-2}$	$3.7·10^{-1}$
HT	kg 1,4-DB eq	90.6	1.8	60.9
POF	kg NMVOC	1.1	$2.2·10^{-2}$	$7.4·10^{-1}$
TET	kg 1,4-DB eq	$2.3·10^{-2}$	$4.6·10^{-4}$	$1.5·10^{-2}$
FET	kg 1,4-DB eq	10.4	$2.1·10^{-1}$	7.0
MET	kg 1,4-DB eq	9.1	$1.9·10^{-1}$	6,15
FD	kg oil eq	87.3	1.8	58.6

3.3.5. Comparison with other methods to obtain antioxidant-rich extracts.

A comparison was made with some processes published in the scientific literature on the basis of an identical FU (1 kg of extract rich in antioxidants) and evaluation methodology (CML 2001 method). Pérez-López et al. (2014a) evaluated the environmental performance of several *Sargassum muticum* macroalgae valorisation strategies. In this study it was considered that the combined extraction of antioxidants and alginates stands out as the most sustainable scenario. Pérez-López et al. (2014b) conducted a life cycle assessment of astaxanthin production on a laboratory and pilot scale. In Papadaki et al. (2017) an evaluation of the life cycle of the recovery of phycocyanin from *Spirulina platensis* cyanobacterium was performed. This study compares six different methods based on Ultrasound-Assisted Extraction to recover phycocyanin. In order to simplify the comparative study, the results were scaled to 100 and represented in Figure 3.4.

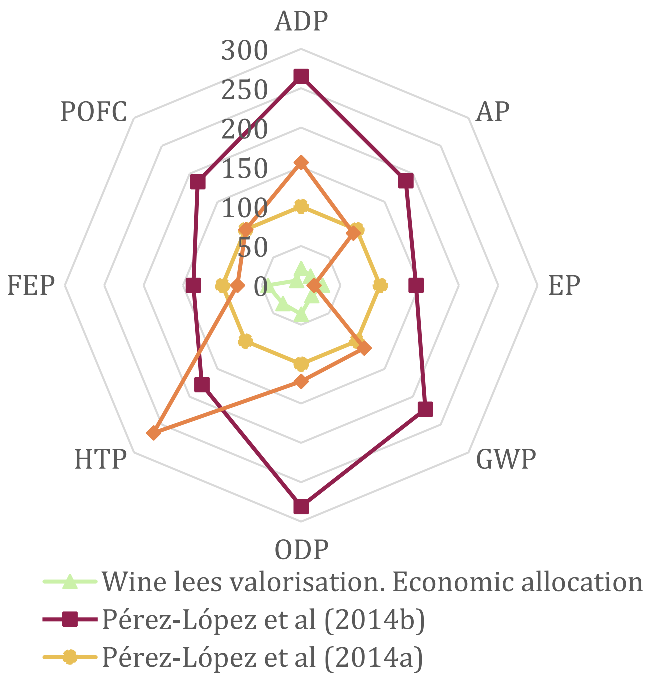


Figure 3.4. Relative environmental profile of the compared valorisation scenarios with the process published in Pérez-López et al., (2014a) as baseline (index = 100)

According to the results depicted in Figure 3.4, the production of antioxidants through the valorisation of wine lees would be the most appropriate route in all the impact categories studied. Except in the case of EP, where the recovery of phycocyanin from a cyanobacterium is the process with the best environmental profile. The production of astaxanthin from microalgae cultivated in photobioreactors on a pilot scale reported the worst environmental profile, with results ranging from 37% worse than the reference (index = 100) in FEP category up to 180% worse in ODP impact category. If wine lees valorisation is compared with the process published by Pérez-López et al. (2014a), the production of antioxidants from the treatment of wine lees presents an environmental profile that is, on average, 75% better in all impact categories. A maximum improvement rate of 92% is reached in the FEP impact category. However, in order to obtain 1 kg of extract rich in antioxidants, it is necessary to treat almost 148 kg of wine lees, while in order to obtain this same amount of extract from the valorisation of *Sargassum muticum*, it is only necessary to process 1.5 kg of biomass. In particular, the treatment of wine lees has better environmental results because most of the operations performed are physical (solid/liquid separations, distillations, evaporations, etc.) and do not involve the large consumption of electricity or chemicals. There is only one determinant consumption in the system, the high temperature steam, while in the rest of the comparative studies, the consumption of electricity and chemicals is relatively high. It can be concluded that, although the total polyphenol content of the extract is lower than other studies and the extraction system would need to be improved to increase the purity of the extract, wine lees valorisation presents the best environmental profile in almost all compared categories.

3.4. CONCLUSIONS

Nowadays special interest is being paid into the valorisation of different waste streams to reduce raw materials consumption. Thus, it has been shown that the integral valorisation of wine lees is a very attractive process to produce value-added products, which is a necessary step in the pursuit of a society based on circular economy and sustainability. It has been proven that steam consumption is an important hotspot in the process, reaching a contribution of 85.7%, 85.3%, 79.4% and 76.7% in FD, CC, TA and POF categories, respectively. Therefore, it will be necessary to reduce this consumption in the future. To achieve this objective, other residues from the winery, such as grape stalks, could be used as raw material to obtain high temperature steam in a boiler, taking this biorefinery scheme to the next level by continuing valorising different winemaking-derived waste. In comparison with other systems that obtain polyphenol-rich antioxidant extracts, it has been demonstrated that the valorisation of wine lees presents the best environmental profile throughout the entire life cycle in almost all the impact categories studied, improving on average 75% in all impact categories with respect to the reference.

It can be concluded that wine waste biorefining is an appropriate way of obtaining products from waste according to the principles of the Circular Economy, where waste is converted into new raw materials. This work shows that the LCA methodology is a useful tool for assessing the environmental impact of wine lees treatment to obtain value-added products. The results of this chapter should be considered in order to develop a more sustainable way of obtaining an antioxidant-rich extract from agricultural residues.

3.5. REFERENCES

Algapani, D.E., Qiao, W., Ricci, M., Bianchi, D., Wandera, S.M., Adani, F., Dong, R., 2019. Bio-hydrogen and bio-methane production from food waste in a two-stage anaerobic digestion process with digestate recirculation. Renew. Energy 130, 1108–1115. https://doi.org/10.1016/j.renene.2018.08.079

Caldeira, C., Quinteiro, P., Castanheira, E., Boulay, A., Dias, A.C., Arroja, L., Freire, F., 2018. Water footprint profile of crop-based vegetable oils and waste cooking oil: Comparing two water scarcity footprint methods. J. Clean. Prod. 195, 1190–1202. https://doi.org/10.1016/j.jclepro.2018.05.221

Castro-López, C., Ventura-Sobrevilla, J.M., González-Hernández, M.D., Rojas, R., Ascacio-Valdés, J.A., Aguilar, C.N., Martínez-Ávila, G.C.G., 2017. Impact of extraction techniques on antioxidant capacities and phytochemical composition of polyphenol-rich extracts. Food Chem. 237, 1139–1148. https://doi.org/10.1016/j.foodchem.2017.06.032

Cherubini, F., 2010. The biorefinery concept: Using biomass instead of oil for producing energy and chemicals. Energy Convers. Manag. 51, 1412–1421. https://doi.org/10.1016/j.enconman.2010.01.015

Christ, K.L., Burritt, R.L., 2013. Critical environmental concerns in wine production: An integrative review. J. Clean. Prod. 53, 232–242. https://doi.org/10.1016/j.jclepro.2013.04.007

Devesa-Rey, R., Vecino, X., Varela-Alende, J.L., Barral, M.T., Cruz, J.M., Moldes, A.B., 2011. Valorization of winery waste vs. the costs of not recycling. Waste Manag. 31, 2327–2335. https://doi.org/10.1016/j.wasman.2011.06.001

Dimou, C., Kopsahelis, N., Papadaki, A., Papanikolaou, S., Kookos, I.K., Mandala, I., Koutinas, A.A., 2015. Wine lees valorization: Biorefinery development including production of a generic fermentation feedstock employed for poly(3-hydroxybutyrate) synthesis. Food Res. Int. 73, 81–87. https://doi.org/10.1016/j.foodres.2015.02.020

Dimou, C., Vlysidis, A., Kopsahelis, N., Papanikolaou, S., Koutinas, A.A., Kookos, I.K., 2016. Techno-economic evaluation of wine lees refining for the production of value-added products. Biochem. Eng. J. 116, 157–165. https://doi.org/10.1016/j.bej.2016.09.004

Escribano-Viana, R., Portu, J., Garijo, P., Gutiérrez, A.R., Santamaría, P., López-Alfaro, I., López, R., González-Arenzana, L., 2018. Evaluating a preventive biological control agent applied on grapevines against Botrytis cinerea and its influence on winemaking. J. Sci. Food Agric. 98, 4517–4526. https://doi.org/10.1002/jsfa.8977

European Comission, 2011. A Roadmap for moving to a competitive low carbon economy in 2050.

FAO, 2011. Global food losses and food waste - Extent, causes and prevention. Rome.

García-Herrera, P., Sánchez-Mata, M.C., Cámara, M., 2010. Nutritional characterization of tomato fiber as a useful ingredient for food industry. Innov. Food Sci. Emerg. Technol. 11, 707–711. https://doi.org/10.1016/j.ifset.2010.07.005

Goedkoop, M., Heijungs, R., Huijbrets, M., de Schryver, A., Struijs, J., Van Zelm, R., 2009. ReCiPe 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. Report I: Characterisation.

Guerini Filho, M., Lumi, M., Hasan, C., Marder, M., Leite, L.C.S., Konrad, O., 2018. Energy recovery from wine sector wastes: A study about the biogas generation potential in a vineyard from Rio Grande do Sul, Brazil. Sustain. Energy Technol. Assessments 29, 44–49. https://doi.org/10.1016/j.seta.2018.06.006

Gullón, P., Gullón, B., Dávila, I., Labidi, J., González-García, S., 2018. Comparative environmental Life Cycle Assessment of integral revalorization of vine shoots from a biorefinery perspective. Sci. Total Environ. 624, 225–240. https://doi.org/10.1016/j.scitotenv.2017.12.036

Hajjaji, N., Pons, M.N., Renaudin, V., Houas, A., 2013. Comparative life cycle assessment of eight alternatives for hydrogen production from renewable and fossil feedstock. J. Clean. Prod. 44, 177–189. https://doi.org/10.1016/j.jclepro.2012.11.043

Jeswani, H.K., Falano, T., Azapagic, A., 2015. Life cycle environmental sustainability of lignocellulosic ethanol produced in integrated thermo-chemical biorefineries. Biofuels, Bioprod. Biorefining 9, 661–676. https://doi.org/10.1002/bbb

Joelsson, E., Dienes, D., Kovacs, K., Galbe, M., Wallberg, O., 2016. Combined production of biogas and ethanol at high solids loading from wheat straw impregnated with acetic acid: experimental study and techno-economic evaluation. Sustain. Chem. Process. 4, 14. https://doi.org/10.1186/s40508-016-0058-5

Karlsson, H., Börjesson, P., Hansson, P.A., Ahlgren, S., 2014. Ethanol production in biorefineries using lignocellulosic feedstock - GHG performance, energy balance and implications of life cycle calculation methodology. J. Clean. Prod. 83, 420–427. https://doi.org/10.1016/j.jclepro.2014.07.029

Khoshnevisan, B., Rafiee, S., Tabatabaei, M., Ghanavati, H., Mohtasebi, S.S., Rahimi, V., Shafiei, M., Angelidaki, I., Karimi, K., 2018. Life cycle assessment of castor-based biorefinery: a well to wheel LCA. Int. J. Life Cycle Assess. 23, 1788–1805. https://doi.org/10.1007/s11367-017-1383-y

Kopsahelis, N., Dimou, C., Papadaki, A., Xenopoulos, E., Kyraleou, M., Kallithraka, S., Kotseridis, Y., Papanikolaou, S., Koutinas, A.A., 2018. Refining of wine lees and cheese whey for the production of microbial oil, polyphenol-rich extracts and value-added co-products. J. Chem. Technol. Biotechnol. 93, 257–268. https://doi.org/10.1002/jctb.5348

Ky, I., Teissedre, P.L., 2015. Characterisation of Mediterranean grape pomace seed and skin extracts: Polyphenolic content and antioxidant activity. Molecules 20, 2190–2207. https://doi.org/10.3390/molecules20022190

Lam, C.M., Yu, I.K.M., Hsu, S.C., Tsang, D.C.W., 2018. Life-cycle assessment on food waste valorisation to value-added products. J. Clean. Prod. 199, 840–848. https://doi.org/10.1016/j.jclepro.2018.07.199

Liang, L., Wang, Y., Ridoutt, B.G., Lal, R., Wang, D., Wu, W., Wang, L., Zhao, G., 2019. Agricultural subsidies assessment of cropping system from environmental and economic perspectives in North China based on LCA. Ecol. Indic. 96, 351–360. https://doi.org/10.1016/j.ecolind.2018.09.017

Makkar, H.P.S., 2018. Review: Feed demand landscape and implications of food-not feed strategy for food security and climate change. Animal 12, 1744–1754. https://doi.org/10.1017/S175173111700324X

Martinez, G.A., Rebecchi, S., Decorti, D., Domingos, J.M.B., Natolino, A., Del Rio, D., Bertin, L., Da Porto, C., Fava, F., 2016. Towards multi-purpose biorefinery platforms for the valorisation of red grape pomace: production of polyphenols, volatile fatty acids, polyhydroxyalkanoates and biogas. Green Chem. 18, 261–270. https://doi.org/10.1039/c5gc01558h

Meneses, M., Torres, C.M., Castells, F., 2016. Sensitivity analysis in a life cycle assessment of an aged red wine production from Catalonia, Spain. Sci. Total Environ. 562, 571–579. https://doi.org/10.1016/j.scitotenv.2016.04.083

Mohammadpour, H., Sadrameli, S.M., Eslami, F., Asoodeh, A., 2019. Optimization of ultrasound-assisted extraction of Moringa peregrina oil with response surface methodology and comparison with Soxhlet method. Ind. Crops Prod. 131, 106–116. https://doi.org/10.1016/j.indcrop.2019.01.030

Musee, N., Lorenzen, L., Aldrich, C., 2007. Cellar waste minimization in the wine industry: a systems approach. J. Clean. Prod. 15, 417–431. https://doi.org/10.1016/j.jclepro.2005.11.004

Nayak, A., Bhushan, B., Rodriguez-Turienzo, L., 2018. Recovery of polyphenols onto porous carbons developed from exhausted grape pomace: A sustainable approach for the treatment of wine wastewaters. Water Res. 145, 741–756. https://doi.org/10.1016/j.watres.2018.09.017

Nieuwlaar, E., Roes, A.L., Patel, M.K., 2015. Final energy requirements of steam for use in environmental life cycle assessment. J. Ind. Ecol. 20, 828–836. https://doi.org/10.1111/jiec.12300

Notarnicola, B., Tassielli, G., Renzulli, P.A., Castellani, V., Sala, S., 2017. Environmental impacts of food consumption in Europe. J. Clean. Prod. 140, 753–765. https://doi.org/10.1016/j.jclepro.2016.06.080

OIV, 2018. State of the Vitiviniculture World Market, April 2018, International Organisation of Vine and Wine.

Papadaki, S., Kyriakopoulou, K., Tzovenis, I., Krokida, M., 2017. Environmental impact of phycocyanin recovery from Spirulina platensis cyanobacterium. Innov. Food Sci. Emerg. Technol. 44, 217–223. https://doi.org/10.1016/j.ifset.2017.02.014

Pereira, P., Cebola, M.J., Oliveira, M.C., Bernardo Gil, M.G., 2017. Antioxidant capacity and identification of bioactive compounds of Myrtus communis L. extract obtained by ultrasound-assisted extraction. J. Food Sci. Technol. 54, 4362–4369. https://doi.org/10.1007/s13197-017-2907-y

Pérez-López, P., Balboa, E.M., González-García, S., Domínguez, H., Feijoo, G., Moreira, M.T., 2014a. Comparative environmental assessment of valorization strategies of the invasive macroalgae *Sargassum muticum*. Bioresour. Technol. 161, 137–148. https://doi.org/10.1016/j.biortech.2014.03.013

Pérez-López, P., González-García, S., Jeffryes, C., Agathos, S.N., McHugh, E., Walsh, D., Murray, P., Moane, S., Feijoo, G., Moreira, M.T., 2014b. Life cycle assessment of the production of the red antioxidant carotenoid astaxanthin by microalgae: from lab to pilot scale. J. Clean. Prod. 64, 332–344. https://doi.org/10.1016/j.jclepro.2013.07.011

Poveda, J.M., Loarce, L., Alarcón, M., Díaz-Maroto, M.C., Alañón, M.E., 2018. Revalorization of winery by-products as source of natural preservatives obtained by means of green extraction techniques. Ind. Crops Prod. 112, 617–625. https://doi.org/10.1016/j.indcrop.2017.12.063

PRé Consultants, 2017. SimaPro Database Manual (No. Methods Library). The Netherlands.

Roy, P., Nei, D., Orikasa, T., Xu, Q., Okadome, H., Nakamura, N., Shiina, T., 2009. A review of life cycle assessment (LCA) on some food products. J. Food Eng. 90, 1–10. https://doi.org/10.1016/j.jfoodeng.2008.06.016

Rugani, B., Vázquez-Rowe, I., Benedetto, G., Benetto, E., 2013. A comprehensive review of carbon footprint analysis as an extended environmental indicator in the wine sector. J. Clean. Prod. 54, 61–77. https://doi.org/10.1016/j.jclepro.2013.04.036

Ruggieri, L., Cadena, E., Martínez-Blanco, J., Gasol, C.M., Rieradevall, J., Gabarrell, X., Gea, T., Sort, X., Sánchez, A., 2009. Recovery of organic wastes in the Spanish wine industry. Technical, economic and environmental analyses of the composting process. J. Clean. Prod. 17, 830–838. https://doi.org/10.1016/j.jclepro.2008.12.005

Szabo, K., Cătoi, A.-F., Vodnar, D.C., 2018. Bioactive compounds extracted from tomato processing by-products as a source of valuable nutrients. Plant Foods Hum. Nutr. 73, 268–277. https://doi.org/10.1007/s11130-018-0691-0

United Nations, 2017. World population prospects: The 2017 Revision, key findings and advance tables.

Vandermeersch, T., Alvarenga, R.A.F., Ragaert, P., Dewulf, J., 2014. Environmental sustainability assessment of food waste valorization options. Resour. Conserv. Recycl. 87, 57–64. https://doi.org/10.1016/j.resconrec.2014.03.008

Vaskan, P., Pachón, E.R., Gnansounou, E., 2018. Techno-economic and life-cycle assessments of biorefineries based on palm empty fruit bunches in Brazil. J. Clean. Prod. 172, 3655–3668. https://doi.org/10.1016/j.jclepro.2017.07.218

Vázquez-Rowe, I., Villanueva-Rey, P., Iribarren, D., Moreira, M.T., Feijoo, G., 2012a. 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. https://doi.org/10.1016/j.jclepro.2011.12.039

Vázquez-Rowe, I., Villanueva-Rey, P., Moreira, M.T., Feijoo, G., 2012b. Environmental analysis of Ribeiro wine from a timeline perspective: Harvest year matters when reporting environmental impacts. J. Environ. Manage. 98, 73–83. https://doi.org/10.1016/j.jenvman.2011.12.009

Vico, A., Pérez-Murcia, M.D., Bustamante, M.A., Agulló, E., Marhuenda-Egea, F.C., Sáez, J.A., Paredes, C., Pérez-Espinosa, A., Moral, R., 2018. Valorization of date palm (*Phoenix dactylifera L.*) pruning biomass by co-composting with urban and agri-food sludge. J. Environ. Manage. 226, 408–415. https://doi.org/10.1016/j.jenvman.2018.08.035

Vieira, G.S., Cavalcanti, R.N., Meireles, M.A.A., Hubinger, M.D., 2013. Chemical and economic evaluation of natural antioxidant extracts obtained by ultrasound-assisted and agitated bed extraction from jussara pulp (*Euterpe edulis*). J. Food Eng. 119, 196–204. https://doi.org/10.1016/j.jfoodeng.2013.05.030

Zabaniotou, A., Kamaterou, P., Pavlou, A., Panayiotou, C., 2018. Sustainable bioeconomy transitions: Targeting value capture by integrating pyrolysis in a winery waste biorefinery. J. Clean. Prod. 172, 3387–3397. https://doi.org/10.1016/j.jclepro.2017.11.077

Zhang, N., Hoadley, A., Patel, J., Lim, S., Li, C., 2017. Sustainable options for the utilization of solid residues from wine production. Waste Manag. 60, 173–183. https://doi.org/10.1016/j.wasman.2017.01.006

USC
UNIVERSIDADE
DE SANTIAGO
DE COMPOSTELA

Chapter 4

Unravelling the environmental impacts of bioactive compounds and organic amendment from grape marc

Summary

In a society that produces large amounts of solid waste, the search for new methods of valorisation has led to the development of techniques that make possible to obtain new products from waste. In the case of bio-waste, vermicomposting is a method of converting solid organic waste into resources through bio-oxidation and stabilization of the organic waste by earthworms. Not all organic waste management processes end in incineration or anaerobic digestion. The possibility of transforming a waste rich in organic matter into vermicompost arises as an alternative to produce a fertiliser with excellent properties. The purpose of this chapter is to establish the environmental impacts of a complete route for the valorisation of grape marc through vermicomposting. In this valorisation route, different value-added products are produced with potential application in the cosmetic, food and pharmaceutical sectors. The main findings from this chapter reported that the energy requirement of the distillation process is an important hot spot of the process. Although the valorisation route has poor results in terms of the two environmental indicators (carbon footprint and normalised impact index), when economic revenues were included in this analysis, its environmental performance was better than other alternatives for bio-waste recovery.

This chapter, under the consent of the publisher, was redrafted after: Cortés, A., Moreira, M. T., Domínguez, J., Lores, M., Feijoo, G. (2020). Unraveling the environmental impacts of bioactive compounds and organic amendment from grape marc. Journal of Environmental Management, 272, 111066. DOI: https://doi.org/10.1016/j.jenvman.2020.111066. More information related to this article can be found in Appendix III.

OUTLINE

4.1. INTRODUCTION

Food waste problem requires research on new processes to achieve the complete valorisation of food waste and public initiatives to change consumers consumption patterns and disposal behaviours (Kibler et al., 2018). It is demonstrated that reducing landfilling in favour of increased recycling of some types of materials such as glass, paper, plastic and metals leads to lower energy demand and environmental impacts (Eriksson et al., 2005). In the case of biowaste, biological treatment such as composting, vermicomposting or anaerobic digestion appear to be suitable options (European Comission, 2008).

Poor waste management involves not only altering the different environmental compartments, but also contributing to problems of global impact. In relative terms, inadequate management of the organic fraction of agricultural residues is a major contributor to global environmental impacts, such as climate change, freshwater pollution and nutrient accumulation (Weidner et al., 2019). On the contrary, the adequate treatment of the organic waste fraction can reduce the environmental impact provided that the organic fraction of the waste stream is recovered in order to produce biogas that can be used as bioenergy or biofertilisers to replace those of chemical origin (Komakech et al., 2015).

As stated in Chapter 3, wine lees and grape marc are the two main winemaking-derived solid waste. They are generated at different times of the winemaking process, while lees are the dead yeast cells remaining from the fermentation process, grape marc is the residue after grape pressing. This fraction of the grape can be considered a valuable source of polyphenols since it contains around 70% of the phenolic compounds of the grape, which could be extracted in a safe and sustainable way (Poveda et al., 2018) since only a small part of the phytochemicals applied during cultivation is transferred from the grape to the wine (Mazza, 1995). The interest in extracting and exploiting the polyphenols present in this type of waste lies in their potential use and application in a wide range of sectors, such as cosmetics, food and pharmaceuticals (Fontana et al., 2013). The current management of wine residues is still in the early stages of development, so it has focused on its application as an organic soil amendment (Domínguez et al., 2017). In small geographic areas with a high burden of agricultural activities, the inappropriate disposal of this material has led to the release of excessive amounts of polyphenols to soils. Phenolic compounds are responsible for the phytotoxic activity of grape marc, so this problem need to be monitored as it can cause inhibition problems for plant growth (Barbera et al., 2013). These agronomic problems associated with the application of grapes to soil could be minimised by stabilising them through different organic decomposition processes as composting or vermicomposting (Gómez-Brandón et al., 2011).

Vermicomposting is a natural process based on the interactions of earthworms (mainly of the species *Eisenia foetida* or *Eisenia andrei*) with the endogenous microorganisms present in the waste as a result of the decomposition of organic matter (Lleó et al., 2013). By varying the operational conditions of the process, it is possible to modify the physical and biochemical properties of the final product (Domínguez et al., 2010). Beyond the enzymatic transformations attributed to earthworms, there is a significant improvement in oxygen concentration, which favours aerobic composting of the waste, resulting in low greenhouse gas emissions (Nigussie et al., 2016). The final product obtained is vermicompost or earthworm humus, which has a stable, homogeneous, and fine particle size appearance. Vermicompost is also a nutrient-rich, peat-like material characterised by high porosity, high water-holding capacity, and low C:N ratio (Domínguez et al., 2014). Residual organic matter tends to humidify, polymerise and polycondense. As a result, the levels of humic acids and, to a lesser extent, fulvic acids increase (by 20-60% compared to those present in the starting materials), affecting the chemical and structural characteristics of the organic matter (Gómez-Brandón et al., 2019). This is why the final product has high water retention capacity and nutrient content (Chen et al., 2018). Vermicomposting is considered a green and clean technology (Karmegam et al., 2019) with moderately low investment and maintenance costs and low energy consumption.

According to a quantitative perspective of impact assessment, the LCA methodology has been used to assess and compare the impact of different waste disposal scenarios, including composting, landfilling and incineration. Several researches have used LCA to analyse the environmental implications of organic waste composting (Saer et al., 2013; ten Hoeve et al., 2019), incineration (Abuşoğlu et al., 2017; Dong et al., 2018; Tong et al., 2018) or landfilling (Buratti et al., 2015; Henriksen et al., 2018). However, only a few LCA studies have analysed the environmental implications of vermicomposting food waste. Within these studies, 2 research works have been published that can be considered as references of great interest for this study. Komakech et al. (2015) and Komakech et al. (2016) compared the environmental performance of different management alternatives based on anaerobic digestion, composting and vermicomposting for food waste and animal manure, but only global warming potential and eutrophication potential categories were considered in both studies. Tedesco et al. (2019) evaluated the life cycle impact of the bioconversion of fruit and vegetable waste into earthworm meal from a "cradle-to-gate" perspective. The main product obtained from vermicomposting are the worms themselves, while in the present study, the worms are mere tools which are used to valorise agricultural waste into some value-added products.

The objective of this chapter is to evaluate the environmental impacts associated with the valorisation of grape marc through vermicomposting using an LCA approach, identifying the stages and the processes that make the greatest contribution to the environmental burdens. Therefore, the system under study converts waste into usable materials following a circular economy approach. The function of the system is to achieve short-term stabilisation of grape marc, obtaining four main outputs: a nutrients-rich biofertiliser, marketable brandy spirit, an extract rich in polyphenols and oil rich in fatty acids.

4.2. MATERIALS AND METHODS

4.2.1. Definition of goal and scope

There are three possible options in the selection of the FU, that is, based on the quantification of a single target product, the total flow of raw materials or the combination of different products (Khoshnevisan et al., 2018). In order to represent the function of the system and to be consistent with the multiple-output nature of the process, it seems correct to select a feedstock-based FU. In this sense, the treatment of 1 tonne of grape marc was chosen as FU.

4.2.2. Description of the overall system and system boundaries

The study was performed through a “cradle-to-gate” perspective, from the extraction of raw materials up to the point when the different products are ready to leave the facilities. The feedstock for the process, as already mentioned, is residual grape marc supplied by different warehouses located at a maximum distance of 130 km from the location of the vermicomposting facilities. The production scheme was evaluated considering all the processes from the production of raw materials to the final products obtained from grape marc. Specifically, the system under study us divided into three subsystems, which are detailed below in Figure 4.1: SS1. Distillation, SS2. Seed oil extraction and SS3. Vermicomposting. It is considered that the production of grape marc as co-product associated with the winemaking process and capital goods are outside the system boundaries.

Subsystem 1. Distillation: Distillation of grape marc to obtain different spirits is an activity traditionally used in local wineries that seek to obtain value-added products from waste. Grape marc is the perfect feedstock to produce brandy spirits named as “orujo” by simple distillation. In this chapter, steam distillation has been considered because it is widely used in large facilities. The use of steam and cooling water to heat and cool the grape marc and the brandy, respectively, have been considered. In addition, the production of wastewater during the distillation process has been considered. In this case the distillation efficiency is relatively high, obtaining 25 L of Brandy per every 200 kg of processed marc. In this subsystem, a large part of the exhausted marc that is obtained as co-product is directed to a grape seed oil extraction process (Subsystem 2), while the rest of the exhausted marc is mixed with fresh marc and is transported by lorry to SS3, in which further operations that allow obtaining an extract rich in polyphenols and an organic fertiliser called vermicompost are carried out. It is important to note that no consideration has been given to transporting these fractions from winery to distillation unit since this type of operation is usually carried out in the same place. However, transportation of the outputs of the distillation unit to the rest of subsystems by lorry and car have been considered.

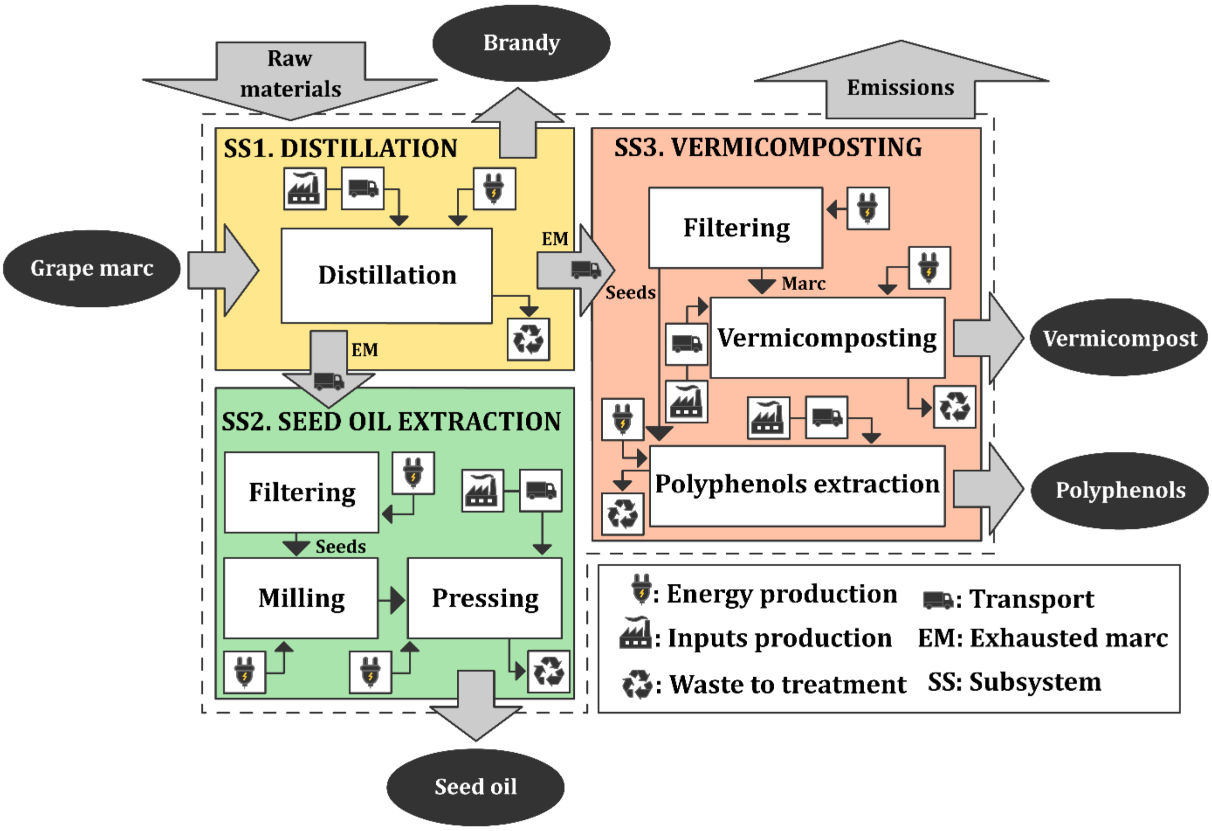


Figure 4.1. Valorisation scheme of grape marc targeting seed oil, brandy, vermicompost and a polyphenol-rich extract.

Subsystem 2. Seed oil extraction: The exhausted marc is subjected to a filtering treatment, in which seeds are separated from the rest of the material. This exhausted marc without seeds is a waste and is sent to landfill for disposal, although it could be considered as a co-product of oil extraction and used to obtain other value-added products, as for energetic or feed purposes. The seeds are feed into a disk crusher, where a fine seed paste is obtained. The paste is pumped into a press, where the grape seed oil is obtained by crushing the seeds. This oil has a good market value due to its high content in vitamin E and linolenic acid. This process is especially interesting since the operations carried out at this subsystem are physical and the consumption of chemicals is scarcely required, only cleaning agents.

Subsystem 3. Vermicomposting: The mixture of fresh and exhausted grape marc from SS1 is taken to a filter similar to the one used in SS2, in which seeds are separated from the grape marc. The quantity of seeds that can be separated has been assumed as 15% of the total grape marc weight. These seeds are led to a pressurised solvent extraction that allows the recovery of a polyphenols-rich extract. The use of sand as dispersant and methanol (65%) in water as solvent have been considered (Álvarez-Casas et al., 2014). In the other route, grape marc separated from the seeds was stored at 4ºC until use. The grape marc was processed in pilot-scale vermireactors with a surface area of 3 m^2 held in a greenhouse in the University of Vigo with no temperature control and the earthworm species *Eisenia andrei* (commonly known as red worm) was used. Vermicomposting system described by Domínguez et al. (2017) was considered. At the beginning of the trial, the vermireactor contained a layer of 12 cm of vermicompost as a bed for the earthworms. Then, successive layers of grape marc were placed through time, for processing by the earthworms. In this way, earthworms are always located in superficial layers of the reactor, while vermicompost is deposited in the lower layers of the reactor. Thus, the reactor was filled in successive layers until a batch is completed in about 12 weeks. At this time, the vermireactor allows the treatment of 600 kg grape marc to obtain over 240 kg vermicompost ready to be used as a high-quality organic fertiliser. During the duration of the trial, it is not necessary the use of additional chemicals or materials. With the aim of preventing the drying of the bed, the vermireactor was watered daily and the leachate was collected and sent to treatment, collecting about 10-12 L leachate per batch. The use of electric sieve and grinder to reduce the particle size of the vermicomposting is also necessary. The electric consumption was estimated considering the average use time and the power of the equipment. Polyphenols extraction was carried out before vermicomposting since, as reported in Domínguez et al. (2016), the amount of polyphenols is reduced by almost one half in a period of only 14 days and by the time period of 42 days, the decrease is about 98% of the initial amount. In the end, two main products are obtained from this subsystem, a nutrient-rich, microbiologically active organic amendment known as vermicompost and a polyphenols-rich extract.

4.2.3. Inventory analysis, data acquisition and allocation approach

The quality of the data handled in the elaboration of the life cycle inventory is especially relevant to ensure the reliability of the study. Therefore, the collection of inventory data requires primary data (typical of real systems under study) or secondary data (those complementary to the main process such as electricity, raw materials, water and fuel). In this chapter, most of the data related to the system correspond to primary data, while those relating to the background system (water, electricity, fuel and chemicals) were taken from the Ecoinvent® v3.5 database.

Regarding the distillation system, the data published in Dimou et al. (2016) has been used. From this work, the data on cooling water consumption, low pressure steam and wastewater generation have been adapted to the characteristics of this study. As for the seed oil extraction subsystem, material and energy consumption has been obtained from Rinaldi et al. (2014), where the evaluation of the life cycle of the production of extra virgin olive oil in Italy is carried out. The total amount of oil obtained from the grape seeds has been estimated based on the study of Fiori et al. (2014), where it was considered that grape seeds contain oil in the range of 8-16% depending on the crop and the harvest year. In this chapter, 10% kg-oil per kg-seeds was considered.

With respect to vermicomposting, primary data were obtained from the pilot-scale vermireactors. The managed data covered the identification of operational aspects of the inventoried reactor, such as the consumption of resources (water, energy, fuel...), waste management or the use of machinery. Direct emissions related to vermicomposting were estimated based on the emission factors taken from different secondary sources. Emissions of methane (CH_4), ammonia (NH_3) and dinitrogen monoxide (N_2O) due to earthworm activity were adapted from Komakech et al. (2015) considering the characteristics of the vermireactors. Non-methane volatile organic compounds (NMVOCs) emissions were adapted from Lleó et al. (2013). Products and residues of the grapevine cultivation contain biogenic carbon from captured carbon dioxide (CO_2) during crop growth. Although CO_2 emissions were calculated, these emissions were not included as they were considered as biogenic CO_2.

Table 4.1. Inventory data of the valorisation scheme for grape marc per FU.

Inputs from Technosphere			
Materials	**kg**	**Materials**	**m^3**
Grape marc	1,000	Water	1.46
Low pressure steam	1,036.60	Cooling water	10.54
Sand	52.94	**Transport**	**t·km**
Methanol	2	Lorry	50.59
Vinyl polychloride	0.12	Car	22.94
Polyethylene	0.14	**Energy**	**kWh**
Cleaning product	$1.65 \cdot 10^{-3}$	Electricity	123.66
Outputs to Technosphere			
Products	**kg**	**Waste**	**kg**
Vermicompost	240	Exhausted marc	289.71
Polyphenols-rich extract	2.43		**L**
Seed oil	4.79	Wastewater	441.35
Brandy	58.82		
Outputs to Environment			
Emissions to air	**kg**	**Emissions to air**	**kg**
NH_3	0.26	CH_4	$2.73 \cdot 10^{-2}$
N_2O	$7.43 \cdot 10^{-3}$	NMVOC	$1.24 \cdot 10^{-2}$

The data necessary to model the extraction of polyphenols from the seeds were obtained from secondary sources. A Pressure Solvent Extraction (PSE) was considered

(Álvarez-Casas et al., 2014) and material consumption of this stage was established based on the extrapolation of laboratory data to a pilot-scale trial considering the primary experimental results as the basis for the analysis. Marine sand was considered as dispersant, and the amount of sand was estimated considering a ratio seeds/solvent of 2/1 (w/w). Methanol 65% was considered as extracting solvent considering a solid/liquid ratio of 1/40 (w/v), as detailed in Dimitrov et al. (2019). Total electricity consumption was estimated from Pradal et al. (2016), taking into account that the methanol content in the solvent (% vol.) and the extraction duration are similar to those selected for the extraction of polyphenols from seeds. Though there may be other ways of polyphenol extraction from grape seeds, this system has been chosen due to its applicability was demonstrated by the analysis of bagasse samples from wineries in Galicia (Álvarez-Casas et al., 2014). A summary of data managed for the complete valorisation of grape marc is displayed in Table 4.1.

The system under assessment is a multi-outputs system where more than one product is obtained. No allocation criteria were considered since a feedstock-based FU was selected, however, if it were necessary to identify the impacts for each product, it is advisable to apply the economic allocation criterion, since the outputs are produced in very different amounts to avoid attributing an unbalanced impact. Therefore, the mass and economic allocation factors were calculated by considering the quantity of each product as well as the market price obtained from scientific publications and websites of specialised companies in the sector. Table 4.2 reports the market price considered for the different added value products as well as the mass and economic allocation factors.

Table 4.2. Computation of allocation factors based on economic and mass allocation approach.

Product	Production (kg)	Market price (€/kg)	Mass allocation	Economic allocation
Vermicompost	240	1.2[a]	78%	11%
Polyphenols-rich extract	2.43	147.67[b]	1%	14%
Seed oil	4.79	300[c]	2%	56%
Brandy	58.82	8.57[d]	19%	19%

[a] Ecocelta (2019)
[b] Vieira et al. (2013)
[c] Le petit jardin (2019)
[d] MAPA (2018)

4.2.4. Life cycle impact analysis: methodology

The software SimaPro 9.0 (PRé Consultants, 2017) was used for the computational implementation of the inventories. The methodology considered to express the environmental impacts was ReCiPe 2016 v1.1 in a hierarchist perspective with the following impact categories at midpoint level (Huijbregts et al., 2016): Global Warming (GW), Stratospheric Ozone Layer Depletion (SOD), Ozone Formation (OF), Terrestrial

Acidification (TA), Freshwater Eutrophication (FE), Marine Eutrophication (ME), Human Toxicity (HT), Terrestrial Ecotoxicity (TET), Freshwater Ecotoxicity (FET), Marine Ecotoxicity (MET) and Fossil Resource Scarcity (FRS).

4.3. RESULTS AND DISCUSSION

4.3.1. Environmental performance of grape marc valorisation to obtain bioactive compounds and organic amendment

The environmental assessment was carried out from a cradle-to-gate, excluding from the analysis the production of the raw material (grape marc). The environmental impacts according to the characterisation phase are reported in Table 4.3. Most environmental burdens are allocated to oil, as the price is very high and, therefore, the economic allocation factor is also high. However, environmental impacts assigned to vermicompost production are much lower. For example, in the case of GW category, the production of 1 kg vermicompost only involves the emission of approximately 200 g CO_2 eq.

It is important to highlight other benefits derived from the use of vermicompost as organic fertiliser in substitution of other more consolidated alternatives such as the use of peat or compost as a soil amendment. The vermicompost produced during the process can be used in vineyards as an organic fertiliser. In fact, due to the chemical characteristics of vermicompost (20.2 ± 1.3 g/kg nitrogen and 2.1 ± 0.1 g/kg phosphorous, among other nutrients), the quantity produced per batch (240 kg) provides the equivalent amount of nitrogen present in 346.3 kg of peat. So, if vermicompost use as organic fertiliser is taken into account, environmental benefits could be calculated by comparing its environmental performance with the impacts derived from the production of different organic fertiliser such as peat or compost.

Table 4.3. Impact assessment results associated with the different products obtained in the process per FU.

	Unit	Vermicompost	Polyphenols	Seed oil	Brandy	Total
GW	kg CO_2 eq	48.9	61.0	244.2	85.7	439.7
SOD	kg CFC11 eq	$2.18 \cdot 10^{-5}$	$2.72 \cdot 10^{-5}$	$1.09 \cdot 10^{-4}$	$3.82 \cdot 10^{-5}$	$1.96 \cdot 10^{-4}$
OF	kg NO_X eq	0.1	0.1	0.4	0.1	0.7
TA	kg SO_2 eq	0.2	0.2	1.0	0.3	1.8
FE	kg P eq	$7.81 \cdot 10^{-3}$	$9.73 \cdot 10^{-3}$	$3.90 \cdot 10^{-2}$	$1.37 \cdot 10^{-2}$	$7.02 \cdot 10^{-2}$
ME	kg N eq	$8.38 \cdot 10^{-4}$	$1.04 \cdot 10^{-3}$	$4.18 \cdot 10^{-3}$	$1.47 \cdot 10^{-3}$	$7.53 \cdot 10^{-3}$
HT	kg 1,4-DCB	0.9	1.1	4.6	1.6	8.2
TET	kg 1,4-DCB	136.8	170.4	682.5	239.4	1229.1
FET	kg 1,4-DCB	1.3	1.6	6.4	2.2	11.5
MET	kg 1,4-DCB	1.8	2.2	9.0	3.1	16.2
FRS	kg oil eq	14.8	18.5	74.0	26.0	133.3

Beyond the comparative performance as soil amendment, it is relevant to identify other benefits associated to preservation of biodiversity and improved resilience of the crops against pests. This enriched-microbial environment provides macro and micro-nutrients to the soil and avoids the extensive use of pesticides, two major consequences that should not be ignored. Direct consequences of the use of vermicompost as a soil amendment are attributed to improved germination, growth, flowering and fruit production for a wide range of plant species, such as trees, horticultural crops and aromatic, medicinal and ornamental plants (Lazcano and Domínguez, 2011).

According to the results, most of the environmental burdens derived from the valorisation strategy are related to the distillation unit (SS1), as displayed in Figure 4.2. This subsystem, along with subsystem 3, are responsible for more than 80% of the environmental burdens in all impact categories, except for FET and MET. Subsystem 1 can be highlighted in categories GW (74.7%), TET (74.7%) and FRS (73.7%). In relation to subsystem 2, it is the main contributor in MET and FET categories, which are highly sensitive to both waste and wastewater treatment. On the contrary, in GW, TA and ME, the environmental burdens related with this subsystem are minimal, with an average of 2.6%.

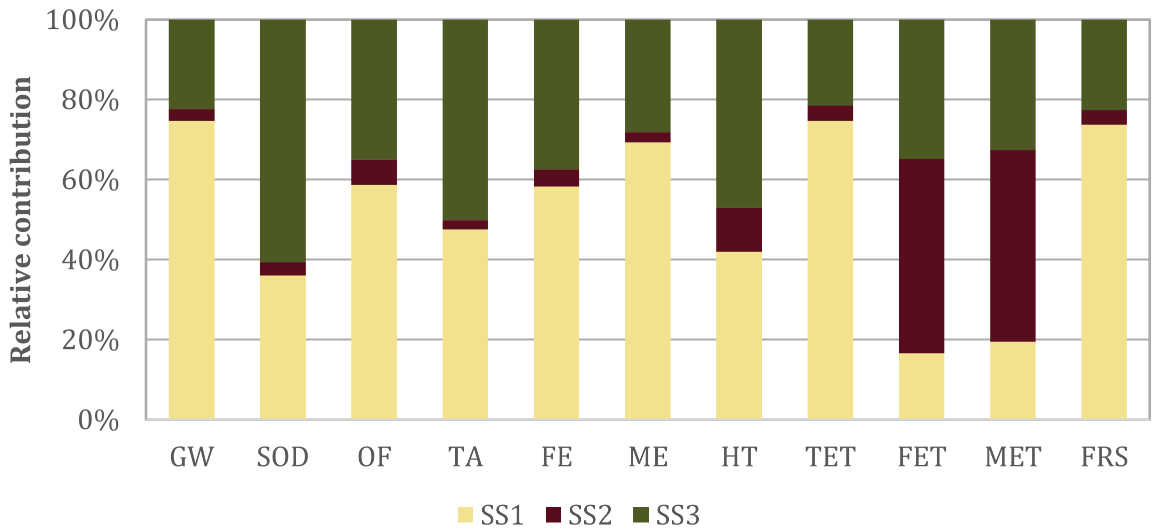


Figure 4.2. Contribution of the different subsystems (SS1-SS3) to the environmental impacts associated with the valorisation of grape marc.

Thus, subsystem 3 presents environmental impact values lower than 40% in all impact categories, except in SOD, TA and HT categories (60.6%, 50.1% and 47.1% respectively). This is mainly due to nitrogen-based gas emissions during the vermicomposting stage, mostly ammonium and dinitrogen monoxide, which have high characterisation factors in these impact categories. Focusing on GW, the environmental burdens of this category are assigned to SS1, mainly associated with the combustion of fossil fuels to obtain the steam required for the distillation of grape marc. Direct emissions into the atmosphere associated with the vermicomposting process were quantified in SS3; however, most of these emissions were substances as ammonium that has no impact in this category, in addition, the production of N_2O and CH_4, with high characterisation factors in this category, is minimal. Direct CO_2 emissions from vermicomposting should not be considered as fossil carbon, but as biogenic CO_2, so they were not included in the inventory analysis. Determining the environmental impacts per activity involved in the valorisation process is useful to pinpoint the "hot spots" of the process. In this way, Figure 4.3 displays the distribution of environmental burdens per activity in the valorisation of grape marc.

Regarding the activities associated with these impacts, steam consumption is the most impacting activity in almost all impact categories (Figure 4.3). Consequently, steam consumption is the main hotspot within the entire valorisation process and should have, therefore, the highest priority for process improvement from an environmental point of view. Omitting FET and MET categories, steam consumption exhibits global contributions ranging from 35.7% in SOD to 74.6% in TET and GW. Regarding GW category, steam production stands out for GHG, SO_2 and NO_X emissions associated with the combustion of fossil fuels.

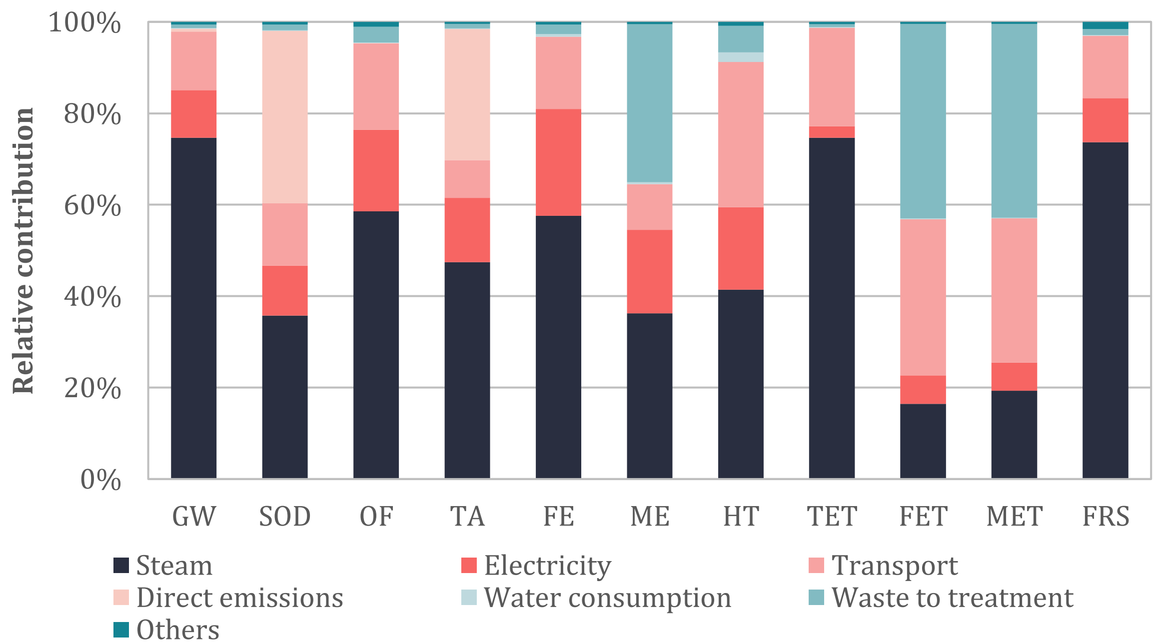


Figure 4.3. Relative environmental impact of the different elements involved in the valorisation of grape marc per FU.

With respect to TET category, steam consumption is again, the main contributor, due to the emission of heavy metals into the air derived from the combustion of fossil fuels. It seems to be consistent that steam consumption was the most contributing process also in FRS, as it is an activity with high energy requirements. In relation to ME, FET and MET categories, the high contribution of waste treatment is remarkable (34.6%, 42.6% and 42.4% respectively), corresponding to the environmental impacts arising from the landfill treatment of waste generated during the production of seed oil in SS2. It is important to note that the information relating to the treatment of waste in landfills has been taken from the Ecoinvent® database, where a significant quantity of metals is emitted to water and air. High concentrations of heavy metals, especially Cu and Zn, are responsible of the impacts observed in these two categories. It is especially noteworthy that electricity consumption has a low impact on almost all impact categories. The rationale behind this evidence is attributed to a low consumption of electricity, reaching a maximum contribution of 23.4% in FE. The contribution of transport is similar in all categories, with no substantial differences highlighted. Toxicity group was the most affected by transportation activities. Specifically, the categories of HT (31.7%), FET (34.2%), MET (31.5%) and TET (21.6%), because of heavy metals emissions into the atmosphere from fuel consumption. As for the environmental impacts related to water consumption, the contribution is practically insignificant, below 0.6% in all the impact categories considered, except for HT, where it reaches the maximum contribution of 2.4%. The rest of the inventoried inputs have almost no impact, so they have been unified in the "others" category, which presents an average contribution lower than 1%.

4.3.2. Comparative assessment with biowaste treatment practices

It is important to note that in this section different biowaste treatment practices have been compared. In this sense, the SS1 subsystem (Distillation) was kept constant in all cases, varying the treatment of the exhausted marc from this subsystem, in order to compare all process on equal terms. This combination of distillation and the different biowaste treatments has been decided based on the fact that grape marc distillation to produce brandy spirits is a practice widely distributed in wineries around the world. The treatment of 1 tonne of biowaste were maintained as functional unit. The chosen treatments were landfilling, anaerobic digestion, incineration, and composting, according to the datasets included in Ecoinvent. Detailed information on the different treatments after the baseline scenario is summarised in Table 4.4.

Table 4.4. List of Ecoinvent database processes considered for end-of-life treatments.

Treatment	Ecoinvent database process
Landfilling	Inert waste {Europe without Switzerland} treatment of inert waste, sanitary landfill
Anaerobic digestion	Biowaste {RoW} | treatment of biowaste by anaerobic digestion
Incineration	Biowaste {GLO} | treatment of biowaste, municipal incineration
Composting	Biowaste {RoW} | treatment of biowaste, industrial composting

Operational costs of the different scenarios were estimated based on different scientific publications. The operating costs of landfill and composting were taken from a study focused on the optimal design of the windrow composting system (Vigneswaran et al., 2016). The estimation of costs of anaerobic digestion and incineration was performed from a model that optimises different waste treatments (Münster et al., 2015). Finally, as far as vermicomposting is concerned, Komakech et al. (2016) was used as the calculation base to estimate the economic cost of the vermicomposting process. With all this information, an environmental-economic comparison of the different biowaste treatment practices was carried out. The environmental burdens of each scenario were calculated by analysing the corresponding Ecoinvent process while vermicomposting scenario corresponds to the system boundaries presented in this chapter. The results of this comparative study have been presented in terms of two indicators: carbon footprint and the normalised impact index of the ReCiPe methodology. The normalised impact index reflects the results of environmental burdens in the form of different impact categories, offering a global view of the environmental performance of the process. Figures 4.4 and 4.5 display the environmental impact in terms of carbon footprint (kg CO_2 eq) and normalised impact index (pts); and the operational costs (€/tonne) of the different scenarios present in the study.

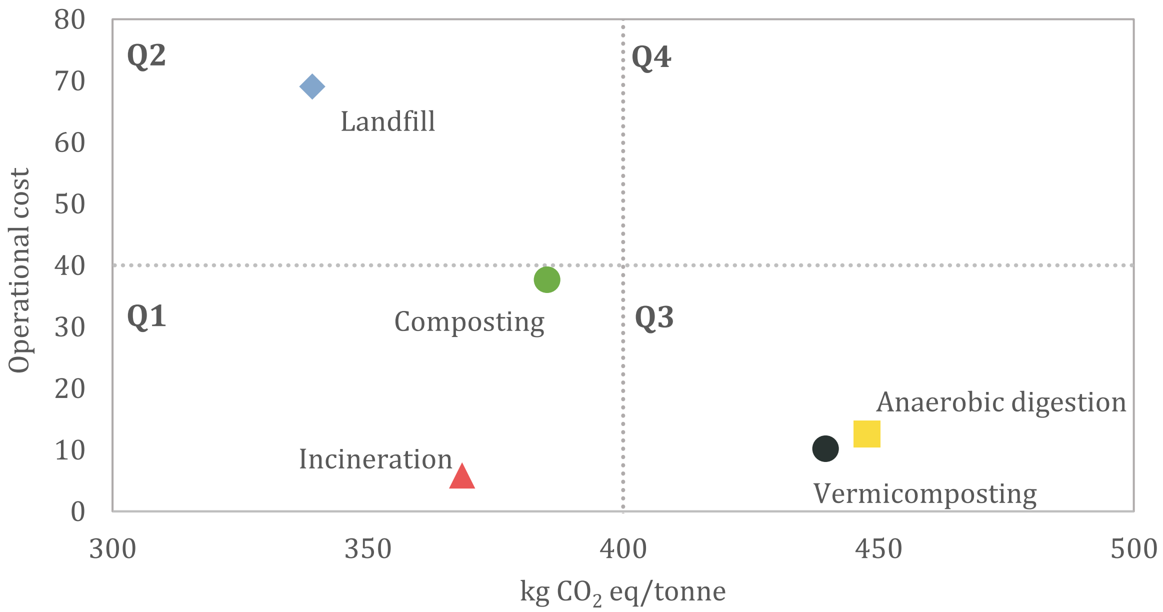


Figure 4.4. Comparative results related to different valorisation process per FU in terms of carbon footprint.

Figure 4.4 presents the GW impact and the operational costs of all the alternatives considered in the study. In terms of carbon footprint, anaerobic digestion presents the worst environmental performance, due to the direct emissions of GHG as methane. However, anaerobic digestion presents a low operational cost of about 12 € per tonne of waste. On the contrary, landfilling is located in the second quadrant and presents the lowest environmental burdens of all the alternatives studied thanks to low GHG emissions when this process is compared with any of the other scenarios. However, operation costs derived from landfill are the highest of all the alternatives, since it is not possible to obtain revenues from the sale of a product with a market value that allows reducing the operation costs. Composting is in the first quadrant, but very close to the second, mainly due to poor economic results. On the other hand, the other alternatives (incineration and vermicomposting) are situated in the first and third quadrant respectively, which correspond to low operational costs and low or medium environmental impact. It is quite relevant that biological treatments present the worst environmental results in terms of carbon footprint, mainly due to the GHG emissions generated in the fermentation and anaerobic digestion processes. However, these processes produce value-added products (biogas, compost, vermicompost...) which would improve the environmental profile if they were considered.

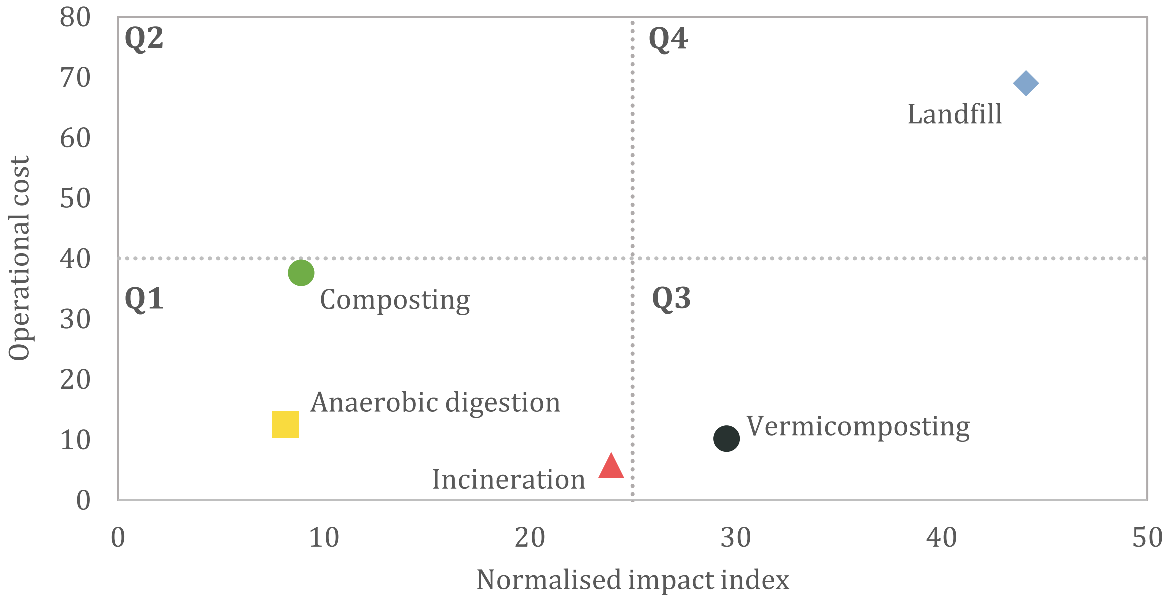


Figure 4.5. Comparative results related to different valorisation process per FU in terms of normalised impact index of the ReCiPe methodology.

Figure 4.5 shows the environmental impact in terms of the normalised impact index of the ReCiPe methodology. This approach provides a global view of the impacts generated within the process in a single value that facilitates the communication of the results. Thus, the calculation of environmental performance is not limited to a single impact category. The same importance is given to other categories that are normally ignored in relation to the carbon footprint, such as ecotoxicity, acidification or eutrophication.

According to the results represented in Figure 4.5, landfilling and vermicomposting scenarios reported the worst environmental profiles. In contrast to using the carbon footprint as the indicator of environmental impact, the alternative with the worst environmental profile is landfill, as ecotoxicity and human toxicity categories include heavy metals pollution. As for anaerobic digestion, which presented the worst environmental profile in previous graph, it has the lowest environmental impact value in this case. Vermicomposting presents a relatively high environmental impact in both indicators (carbon footprint and normalised impact index). However, the multi-product nature of the vermicomposting process must be considered, in the next section an additional analysis that considers the outputs of the different processes is carried out.

4.3.3. Environmental implications of switching from mass-based FU to a benefit-based one.

The results shown in section 4.3.2 were related to a functional unit based on the amount of biowaste treated: 1 tonne of grape marc. This functional unit is useful when analysing valorisation systems where multiple by-products are obtained as it corresponds to the amount of waste to be treated. However, the quantity of valuable by-products, which also have different market prices, means that the potential revenue obtained per alternative is variable and depends on the technology. Thus, in addition to the environmental characterisation of the process, it is important considering the production of value-added products that have certain environmental benefits which come from the environmental credits produced by not manufacturing these products that consume raw materials and energy. The selection of an economic-based Functional Unit has been discussed in previous studies where different biorefinery-based system have been assessed (Budzinski and Nitzsche, 2016; González-García et al., 2018; Pérez-López et al., 2014)

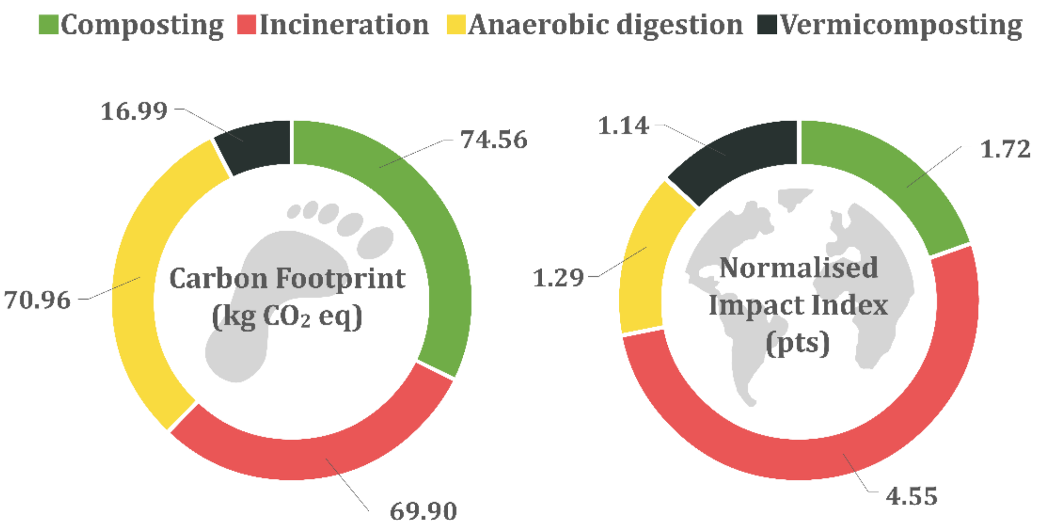


Figure 4.6. Comparative environmental impacts in terms of the normalised impact index (pts) and the carbon footprint (kg CO_2 eq) considering 100 € of revenue as FU.

To consider the market price of all outputs produced, an alternative functional unit based on the economic benefit expected in each scenario was chosen. The alternative functional unit proposed for this section is the generation of 100 € of economic revenue from the sale of the different outputs. The landfill scenario was not included in this comparative analysis since no outputs with market value was considered. Figure 4.6 shows the main environmental indicators in terms of Normalised Impact Index and Carbon Footprint considering 100 € of economic revenue as a functional unit. Different results can be obtained if a mass-based FU or an economy-based FU is chosen. According to the results, as previously reported in Figures 4.4 and 4.5, vermicomposting involved low impact in terms of the two selected indicators. In this case, it had the lowest environmental impact in both cases (1.14 pts and 16.99 kg CO_2 eq). This can be explained by the fact that vermicomposting can be considered as a biorefinery-based process, from which several added-value products can be obtained.

The incineration scenario maintains a performance similar to that of the previous analysis, in terms of carbon footprint presents a relatively low impact (69.90 kg CO_2 eq). However, when the rest of the impact categories considered in the study are incorporated, the impact increases, being the alternative with the worst environmental profile in terms of the normalised impact index (1.29 pts). Anaerobic digestion presented the worst environmental profile in terms of carbon footprint (more than 450 kg CO_2 eq per tonne of biowaste) due to methane emissions, however, in this analysis, when considering the benefits provided by biogas, the carbon footprint of this alternative is almost equal to the alternatives of composting and incineration. In terms of the carbon footprint, composting shows the worst environmental behaviour (74.56 kg CO_2 eq), mainly due to the low market price of compost and the amount of GHGs emissions during the process. It has been shown that the use of an environmental indicator which assesses the complete profile of the process (normalised impact index) and not only a specific aspect (carbon footprint) is appropriate. In this way, not a single environmental aspect is enhanced, as shown in Figure 4.4, where the landfill presented the lowest environmental impact in terms of carbon footprint, but the most shocking profile when the normalised impact index was evaluated. In addition, if a global vision of the different alternatives is considered (both waste treatment and production of added-value products), vermicompost is proven as the best alternative to biowaste treatment.

4.4. CONCLUSIONS

In recent years, there is a growing interest in the exploitation of the waste generated by the wine industry. This chapter has shown that grape marc is a feedstock with the capacity to produce a wide range of value-added products, which represents a great opportunity for the wine sector in the future. Furthermore, it has been proven that vermicomposting is an innovative and environmentally sustainable valorisation treatment. Using the LCA method, it has been demonstrated that the energy needs of the distillation process are an important hotspot of the process. Based on the results obtained in this chapter, it would be interesting to analyse, in future research, a scenario in which most fossil energy sources would be replaced by renewable energy sources.

If economic allocation factors are considered, the environmental burdens of the process can be distributed among the different products, which corresponds to 200 g CO_2 eq per kg of vermicompost. The comparative analysis between the end-of-life treatments has shown that, although vermicomposting presents some poor results in terms of carbon footprint and normalised impact index, its environmental performance is better than the other alternatives when economic revenues are included in the analysis. This study provides relevant information in the basic design of a patent on which the process has been developed on a commercial scale and can contribute to the development of the process, not only from an environmental but also from an economic point of view.

4.5. REFERENCES

Abuşoğlu, A., Özahi, E., İhsan Kutlar, A., Al-jaf, H., 2017. Life cycle assessment (LCA) of digested sewage sludge incineration for heat and power production. J. Clean. Prod. 142, 1684–1692. https://doi.org/10.1016/j.jclepro.2016.11.121

Álvarez-Casas, M., García-Jares, C., Llompart, M., Lores, M., 2014. Effect of experimental parameters in the pressurized solvent extraction of polyphenolic compounds from white grape marc. Food Chem. 157, 524–532. https://doi.org/10.1016/j.foodchem.2014.02.078

Barbera, A.C., Maucieri, C., Cavallaro, V., Ioppolo, A., Spagna, G., 2013. Effects of spreading olive mill wastewater on soil properties and crops, a review. Agric. Water Manag. 119, 43–53. https://doi.org/10.1016/j.agwat.2012.12.009

Budzinski, M., Nitzsche, R., 2016. Comparative economic and environmental assessment of four beech wood based biorefinery concepts. Bioresour. Technol. 216, 613–621. https://doi.org/10.1016/j.biortech.2016.05.111

Buratti, C., Barbanera, M., Testarmata, F., Fantozzi, F., 2015. Life Cycle Assessment of organic waste management strategies: An Italian case study. J. Clean. Prod. 89, 125–136. https://doi.org/10.1016/j.jclepro.2014.11.012

Chen, Y., Chang, S.K.C., Chen, J., Zhang, Q., Yu, H., 2018. Characterization of microbial community succession during vermicomposting of medicinal herbal residues. Bioresour. Technol. 249, 542–549. https://doi.org/10.1016/j.biortech.2017.10.021

Dimitrov, K., Pradal, D., Vauchel, P., Baouche, B., Nikov, I., Dhulster, P., 2019. Modeling and optimization of extraction and energy consumption during ultrasound-assisted extraction of antioxidant polyphenols from pomegranate peels. Environ. Prog. Sustain. Energy 1–7. https://doi.org/10.1002/ep.13148

Dimou, C., Vlysidis, A., Kopsahelis, N., Papanikolaou, S., Koutinas, A.A., Kookos, I.K., 2016. Techno-economic evaluation of wine lees refining for the production of value-added products. Biochem. Eng. J. 116, 157–165. https://doi.org/10.1016/j.bej.2016.09.004

Domínguez, J., Aira, M., Gómez-Brandón, M., 2010. Vermicomposting: Earthworms enhance the work of microbes, in: Insam, H., Frankie-Whittle, I., Goberna, M. (Eds.), Microbes at Work. Springer Berlin Heidelberg, pp. 1–329. https://doi.org/10.1007/978-3-642-04043-6

Domínguez, J., Martínez-Cordeiro, H., Álvarez-casas, M., Lores, M., 2014. Vermicomposting grape marc yields high quality organic biofertiliser and bioactive polyphenols. Waste Manag. Res. 32, 1235–1240. https://doi.org/10.1177/0734242X14555805

Domínguez, J., Martínez-Cordeiro, H., Lores, M., 2016. Earthworms and Grape Marc: Simultaneous Production of a High-Quality Biofertilizer and Bioactive-Rich Seeds, in: Morata, A., Loira, I. (Eds.), Grape and Wine Biotechnology. https://doi.org/10.5772/64751

Domínguez, J., Sanchez-Hernandez, J.C., Lores, M., 2017. Vermicomposting of Winemaking By-Products, in: Handbook of Grape Processing By-Products: Sustainable Solutions. pp. 55–78. https://doi.org/10.1016/B978-0-12-809870-7.00003-X

Dong, J., Tang, Y., Nzihou, A., Chi, Y., Weiss-Hortala, E., Ni, M., 2018. Life cycle assessment of pyrolysis, gasification and incineration waste-to-energy technologies: Theoretical analysis and case study of commercial plants. Sci. Total Environ. 626, 744–753. https://doi.org/10.1016/j.scitotenv.2018.01.151

Ecocelta, 2019. Ecocelta, biofertilizante y gestión ambiental. http://ecocelta.com/es/ (accessed 5.22.19).

Eriksson, O., Reich, M.C., Frostell, B., Björklund, A., Assefa, G., Sundqvist, J.O., Granath, J., Baky, A., Thyselius, L., 2005. Municipal solid waste management from a systems perspective. J. Clean. Prod. 13, 241–252. https://doi.org/10.1016/j.jclepro.2004.02.018

Escribano-Viana, R., Portu, J., Garijo, P., Gutiérrez, A.R., Santamaría, P., López-Alfaro, I., López, R., González-Arenzana, L., 2018. Evaluating a preventive biological control agent applied on grapevines against Botrytis cinerea and its influence on winemaking. J. Sci. Food Agric. 98, 4517–4526. https://doi.org/10.1002/jsfa.8977

European Comission, 2008. Green paper on the management of bio-waste in the European Union.

Eurostat, 2019. Generation of waste by waste category, hazardousness and NACE Rev. 2 activity. https://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=env_wasgen&lang=en (accessed 1.1.20).

Fiori, L., Lavelli, V., Duba, K.S., Sri Harsha, P.S.C., Mohamed, H. Ben, Guella, G., 2014. Supercritical CO_2 extraction of oil from seeds of six grape cultivars: Modeling of mass transfer kinetics and evaluation of lipid profiles and tocol contents. J. Supercrit. Fluids 94, 71–80. https://doi.org/10.1016/j.supflu.2014.06.021

Gómez-Brandón, M., Lazcano, C., Lores, M., Domínguez, J., 2011. Short-term stabilization of grape marc through earthworms. J. Hazard. Mater. 187, 291–295. https://doi.org/10.1016/j.jhazmat.2011.01.011

Gómez-Brandón, M., Lores, M., Insam, H., Domínguez, J., 2019. Strategies for recycling and valorization of grape marc. Crit. Rev. Biotechnol. 39, 437–450. https://doi.org/10.1080/07388551.2018.1555514

González-García, S., Gullón, B., Moreira, M.T., 2018. Environmental assessment of biorefinery processes for the valorization of lignocellulosic wastes into oligosaccharides. J. Clean. Prod. 172, 4066–4073. https://doi.org/10.1016/j.jclepro.2017.02.164

Goorhuis, M., 2014. Developments in Collection of Municipal Solid Waste, in: Worrell, E., Reuter, M.A. (Eds.), Handbook of Recycling: State-of-the-Art for Practitioners, Analysts, and Scientists. Elsevier Inc., pp. 405–417. https://doi.org/10.1016/B978-0-12-396459-5.00026-X

Guerini Filho, M., Lumi, M., Hasan, C., Marder, M., Leite, L.C.S., Konrad, O., 2018. Energy recovery from wine sector wastes: A study about the biogas generation potential in a vineyard from Rio Grande do Sul, Brazil. Sustain. Energy Technol. Assessments 29, 44–49. https://doi.org/10.1016/j.seta.2018.06.006

Henriksen, T., Astrup, T.F., Damgaard, A., 2018. Linking data choices and context specificity in Life Cycle Assessment of waste treatment technologies: A landfill case study. J. Ind. Ecol. 22, 1039–1049. https://doi.org/10.1111/jiec.12709

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M.D.M., Hollander, A., Zijp, M., van Zelm, R., 2016. ReCiPe 2016 v1.1. A harmonized life cycle impact assessment method at midpoint and endpoint level. The Netherlands.

Karmegam, N., Vijayan, P., Prakash, M., John Paul, J.A., 2019. Vermicomposting of paper industry sludge with cowdung and green manure plants using *Eisenia fetida*: A viable option for cleaner and enriched vermicompost production. J. Clean. Prod. 228, 718–728. https://doi.org/10.1016/j.jclepro.2019.04.313

Khoshnevisan, B., Rafiee, S., Tabatabaei, M., Ghanavati, H., Mohtasebi, S.S., Rahimi, V., Shafiei, M., Angelidaki, I., Karimi, K., 2018. Life cycle assessment of castor-based biorefinery: a well to wheel LCA. Int. J. Life Cycle Assess. 23, 1788–1805. https://doi.org/10.1007/s11367-017-1383-y

Kibler, K.M., Reinhart, D., Hawkins, C., Motlagh, A.M., Wright, J., 2018. Food waste and the food-energy-water nexus: A review of food waste management alternatives. Waste Manag. 74, 52–62. https://doi.org/10.1016/j.wasman.2018.01.014

Komakech, A.J., Sundberg, C., Jönsson, H., Vinnerås, B., 2015. Life cycle assessment of biodegradable waste treatment systems for sub-Saharan African cities. Resour. Conserv. Recycl. 99, 100–110. https://doi.org/10.1016/j.resconrec.2015.03.006

Komakech, A.J., Zurbrügg, C., Miito, G.J., Wanyama, J., Vinnerås, B., 2016. Environmental impact from vermicomposting of organic waste in Kampala, Uganda. J. Environ. Manage. 181, 395–402. https://doi.org/10.1016/j.jenvman.2016.06.028

Lazcano, C., Domínguez, J., 2011. The use of vermicompost in sustainable agriculture : Impact on plant growth, in: Soil Nutrients. pp. 1–23.

Le petit jardin, 2019. Le petir jardin - cosmética natural. https://le-petitjardin.com/ (accessed 5.22.19).

Lleó, T., Albacete, E., Barrena, R., Font, X., Artola, A., Sánchez, A., 2013. Home and vermicomposting as sustainable options for biowaste management. J. Clean. Prod. 47, 70–76. https://doi.org/10.1016/j.jclepro.2012.08.011

MAPA, 2018. Base de datos de consumo en hogares. https://www.mapa.gob.es (accessed 5.22.19).

Marshall, R.E., Farahbakhsh, K., 2013. Systems approaches to integrated solid waste management in developing countries. Waste Manag. 33, 988–1003. https://doi.org/10.1016/j.wasman.2012.12.023

Mazza, G., 1995. Anthocyanins in grapes and grape products. Crit. Rev. Food Sci. Nutr. 35, 341–371. https://doi.org/10.1080/10408399509527704

Münster, M., Ravn, H., Hedegaard, K., Juul, N., Ljunggren Söderman, M., 2015. Economic and environmental optimization of waste treatment. Waste Manag. 38, 486–495. https://doi.org/10.1016/j.wasman.2014.12.005

Notarnicola, B., Tassielli, G., Renzulli, P.A., Castellani, V., Sala, S., 2017. Environmental impacts of food consumption in Europe. J. Clean. Prod. 140, 753–765. https://doi.org/10.1016/j.jclepro.2016.06.080

Pérez-López, P., Balboa, E.M., González-García, S., Domínguez, H., Feijoo, G., Moreira, M.T., 2014. Comparative environmental assessment of valorization strategies of the invasive macroalgae *Sargassum muticum*. Bioresour. Technol. 161, 137–148. https://doi.org/10.1016/j.biortech.2014.03.013

Poveda, J.M., Loarce, L., Alarcón, M., Díaz-Maroto, M.C., Alañón, M.E., 2018. Revalorization of winery by-products as source of natural preservatives obtained by means of green extraction techniques. Ind. Crops Prod. 112, 617–625. https://doi.org/10.1016/j.indcrop.2017.12.063

Pradal, D., Vauchel, P., Decossin, S., Dhulster, P., Dimitrov, K., 2016. Kinetics of ultrasound-assisted extraction of antioxidant polyphenols from food by-products: Extraction and energy consumption optimization. Ultrason. Sonochem. 32, 137–146. https://doi.org/10.1016/j.ultsonch.2016.03.001

PRé Consultants, 2017. SimaPro Database Manual (No. Methods Library). The Netherlands.

Rinaldi, S., Barbanera, M., Lascaro, E., 2014. Assessment of carbon footprint and energy performance of the extra virgin olive oil chain in Umbria, Italy. Sci. Total Environ. 482–483, 71–79. https://doi.org/10.1016/j.scitotenv.2014.02.104

Saer, A., Lansing, S., Davitt, N.H., Graves, R.E., 2013. Life cycle assessment of a food waste composting system: Environmental impact hotspots. J. Clean. Prod. 52, 234–244. https://doi.org/10.1016/j.jclepro.2013.03.022

Tedesco, D.E.A., Conti, C., Lovarelli, D., Biazzi, E., Bacenetti, J., 2019. Bioconversion of fruit and vegetable waste into earthworms as a new protein source: The environmental impact of earthworm meal production. Sci. Total Environ. 683, 690–698. https://doi.org/10.1016/j.scitotenv.2019.05.226

ten Hoeve, M., Bruun, S., Jensen, L.S., Christensen, T.H., Scheutz, C., 2019. Life cycle assessment of garden waste management options including long-term emissions after land application. Waste Manag. 86, 54–66. https://doi.org/10.1016/j.wasman.2019.01.005

Tong, H., Shen, Y., Zhang, J., Wang, C.H., Ge, T.S., Tong, Y.W., 2018. A comparative life cycle assessment on four waste-to-energy scenarios for food waste generated in eateries. Appl. Energy 225, 1143–1157. https://doi.org/10.1016/j.apenergy.2018.05.062

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. https://doi.org/10.1016/j.jenvman.2011.12.009

Vieira, G.S., Cavalcanti, R.N., Meireles, M.A.A., Hubinger, M.D., 2013. Chemical and economic evaluation of natural antioxidant extracts obtained by ultrasound-assisted and agitated bed extraction from jussara pulp (*Euterpe edulis*). J. Food Eng. 119, 196–204. https://doi.org/10.1016/j.jfoodeng.2013.05.030

Vigneswaran, S., Kandasamy, J., Johir, M.A.H., 2016. Sustainable operation of composting in solid waste management. Procedia Environ. Sci. 35, 408–415. https://doi.org/10.1016/j.proenv.2016.07.022

Weidner, T., Yang, A., Hamm, M.W., 2019. Consolidating the current knowledge on urban agriculture in productive urban food systems: Learnings, gaps and outlook. J. Clean. Prod. 209, 1637–1655. https://doi.org/10.1016/j.jclepro.2018.11.004

Zabaniotou, A., Kamaterou, P., Pavlou, A., Panayiotou, C., 2018. Sustainable bioeconomy transitions: Targeting value capture by integrating pyrolysis in a winery waste biorefinery. J. Clean. Prod. 172, 3387–3397. https://doi.org/10.1016/j.jclepro.2017.11.077

Chapter 5

Environmental implications of biohydrogen-based energy production from steam reforming of alcoholic waste

Summary

Taking into account that the world´s energy demand is increasing and also that our energy system is still mainly based on fossil fuels, a paradigm shifts towards clean energy production based on available renewable resources is needed. Thus, in recent years, numerous alternatives to the use of traditional fossil fuels have been proposed, such as the production of biofuels, bio-alcohols, biohydrogen or any type of renewable energy. In particular, hydrogen is a high-quality energy carrier that can be used with great efficiency and is expected to acquire a great importance in the next generation of fuels. However, the environmental profile of hydrogen-based energy systems is as "clean" or "dirty" depending on the source of the hydrogen. This chapter aims to analyse the potential environmental impacts associated with the steam reforming of alcoholic waste from distilleries to produce hydrogen, which can be valorised to produce clean electricity. The main findings from this chapter reported that the global environmental profile is better than other alternatives more common as sanitary landfill or incineration. In terms of some impact categories as Abiotic and Ozone Depletion, Acidification and Eutrophication, steam reforming of alcoholic waste showed better profiles than other processes that produce hydrogen from diverse feedstocks.

This chapter, under the consent of the publisher, was redrafted after: Cortés, A., Feijoo, G., Chica, A., Da Costa-Serra, J. F., Moreira, M. T. (2019). Environmental implications of biohydrogen based energy production from steam reforming of alcoholic waste. Industrial Crops & Products, 138, 111465. DOI: https://doi.org/10.1016/j.indcrop.2019.111465. More information related to this article can be found in Appendix III.

OUTLINE

5.1. INTRODUCTION

In recent years, numerous alternatives to the use of traditional fossil fuels have been proposed, such as the production of biofuels, bio-alcohols, hydrogen or any type of renewable energy (Balat, 2011). In particular, biomass is one of the renewable energy sources that has experienced strong growth in recent years, due to its global availability and diversity (Spiridon et al., 2016; Tian et al., 2018). Focusing on the different types of fuels, hydrogen is a high quality energy carrier that can be used with high efficiency (Frolov et al., 2013) and is expected to acquire great importance in next generation fuels (Alipour-Moghadam et al., 2014). This fact, together with declining fossil fuel reserves, steadily rising prices and increasing pollution make hydrogen a very attractive product for meeting global energy demand (Khaodee et al., 2011).

However, the environmental profile of hydrogen-based energy systems is as "clean" or "dirty" depending on the scheme of conversion (Rabenstein and Hacker, 2008). The traditional schemes producing H_2 from natural gas are a major source of CO_2, with emissions of approximately 10-12 kg of CO_2 per kg of H_2 (Spath and Mann, 2001). Traditional plants produce hydrogen by catalytic steam reforming of natural gas, which is a mature technology and is the pathway by which most hydrogen is produced today. Because of this, reducing CO_2 emissions associated with hydrogen production would result in a considerable reduction of pollution (Salkuyeh et al., 2018).

In this sense, fuel cells technology and the use of hydrogen are proposed as one of the most promising environmental solutions in relation to the reduction of global emissions (Díaz Alvarado and Gracia, 2010). Fuel cells are devices that electrochemically convert chemical energy from fuels into electricity (Morales et al., 2010). Among the different types of fuel cells, the Solid Oxide Fuel Cell (SOFC) is the most efficient, due to its high operating temperatures and the fact that it is not poisoned with CO (Hernández and Kafarov, 2009). When this type of battery is used, an efficiency around 50% can be obtained (Strazza et al., 2015); in addition, an efficiency of 70% can be achieved if cogeneration system is used (Strazza et al., 2010).

Hydrogen production from renewable sources such as poplar (Susmozas et al., 2016) or willow wood (González-García et al., 2012) have been investigated as the first actions to achieve a significant reduction of environmental impacts (Salkuyeh et al., 2018). Other elements have been evaluated as potential sources of biohydrogen, such as sugar cane (Halleux et al., 2008), sweet potato (Costa et al., 2018), sorghum (Aguilar-Sánchez et al., 2018) or sugar beet (Luo et al., 2009), however, these sources are also food, which may represent a major controversy in the production of biohydrogen on industrial scale if it clashed with the production of food for human consumption. Hydrogen can be obtained from different feedstocks through steam reforming (Braga et al., 2016; López et al., 2019; Zheng et al., 2019), autothermal reforming (Khila et al., 2017; Spallina et al., 2018; Xue et al., 2017) and aqueous phase reforming (Coronado et al., 2018; Esteve-Adell et al., 2017;

García et al., 2018), among them, steam reforming is the option which presents the highest conversion efficiency, around 70%. It is not surprising that this option is, at this moment, the most common as almost 90% of H_2 is produced by natural gas reforming.

Steam reforming of natural gas is the most popular method for producing commercial hydrogen that currently covers about 50% of global hydrogen demand (Anzelmo et al., 2018) and is sometimes referred to as steam methane reforming (SMR). Steam reforming is an endothermic process based on the reaction of gas with steam at high temperature and moderate pressure. In this way, the chemical reaction taking place leads to hydrogen and carbon dioxide (Reaction 1):

$$CH_3CH_2OH + H_2O \rightarrow 2CO_2 + 6H_2 \qquad \Delta H_r = 174 \text{ kJ mol}^{-1} \qquad (1)$$

However, depending on the reaction mixture and operating conditions in the reactor, another route can be followed, producing undesirable products (Ni et al., 2007), such as carbon monoxide (Reaction 2), methane (Reaction 3) or ethylene (Reaction 4):

$$CH_3CH_2OH + H_2O \rightarrow 2CO + 4H_2 \qquad \Delta H_r = 256 \text{ kJ mol}^{-1} \qquad (2)$$
$$CH_3CH_2OH \rightarrow CO + CH_4 + H_2 \qquad \Delta H_r = 50 \text{ kJ mol}^{-1} \qquad (3)$$
$$CH_3CH_2OH \rightarrow C_2H_4 + H_2O \qquad \Delta H_r = 46 \text{ kJ mol}^{-1} \qquad (4)$$

Once the process is complete, the output stream must undergo purification treatment to avoid the presence of by-products such as methane and carbon monoxide. The removal of CO is an important step because it normally poisons the catalyst in fuel cells, that is why CO is removed first by the Water Gas Shift (WGS) reaction (Reaction 5). WGS is an exothermic and reversible reaction usually used in industry to produce high purity hydrogen (Alamolhoda et al., 2019). Usually, 90% of the CO outflowing from the steam reforming reactor can be converted to CO_2 (Rossetti et al., 2015b).

$$CO + H_2O \rightarrow H_2 + CO_2 \qquad \Delta H_r = -41 \text{ kJ mol}^{-1} \qquad (5)$$

Following this stage, the Pressure Swing Adsorption (PSA) process separates hydrogen from the rest of the components of the gas stream with 85% efficiency, obtaining H_2 with 99% purity (Susmozas et al., 2013), and whose energy content is usually higher than that of the natural gas used for reforming.

The implementation of other choices of hydrogen production can be considered from alternative raw materials, such as alcohols (Rossetti et al., 2015a). In addition to steam reforming of ethanol, studies have been published on steam reforming of different types of alcohol with the aim of producing hydrogen. Some of these alcohols are butanol (Kumar et al., 2018), propanol (Wang et al., 2015), methanol (Tian et al., 2017) or glycerol (Menezes et al., 2018) but, even so, the use of ethanol for this purpose offers the best opportunity to produce hydrogen from renewable sources (Ramírez and Homs, 2008), especially if ethanol is produced as a co-product from different processes. Specifically, alcoholic waste from the wine industry represents an attractive raw material due to 65%

of world wine production is managed by European winegrowers, mostly small and medium-sized wineries, according to the Comité Européen des Enterprises Vins (CEEV, 2016).

As discussed in Chapters 3 and 4, wine production generates large amounts of solid and liquid wastes, with a serious impact on the environment when they are not adequately treated. The liquid wastes are normally processed in distilleries to obtain purified alcohols, but in these processes, non-commercial purge streams containing impurities separated from good quality alcohols are generated. Thus, the process here analysed aims to evaluate the potential of these by-products and their valorisation activities as a sustainable way to produce hydrogen. In this chapter, a method for energy recovery is proposed as a final step in the complete valorisation of viticulture waste, once the extraction of value-added products is no longer possible.

The main objective of this chapter is to analyse the potential environmental impacts associated with the steam reforming of alcoholic waste from distilleries. Quantifying the consumption of material and energy resources during the life cycle makes it possible to estimate potential changes and emissions to the environment. The main product of the process is hydrogen (H_2), along with a certain amount of carbon monoxide (CO), carbon dioxide (CO_2), methane (CH_4) and ethylene (C_2H_4), which accounts for a proportion lower than 30%. This output stream is used to produce electricity in a 3 kW SOFC. This energy recovery method is compatible with the previously evaluated processes, as it can be adapted to the outflow of the distillation subsystems of Chapters 3 and 4 to treat the liquid waste streams. In this way, the production of value-added products can be complemented by this novel energy recovery system.

5.2. MATERIALS AND METHODS

5.2.1. Definition of goal and scope

The Life Cycle Analysis methodology has been considered as a fundamental tool in the analysis of the environmental profile associated with the steam reforming of alcoholic waste from distilleries. In distilleries, alcohol can be extracted from some wines that cannot be marketed. During this distillation process, an ethanol-rich fraction is obtained, but also a residual fraction that remains in the distiller's tail, which is the residue used in this study.

Figure 5.1 presents the block diagram of the process, identifying the system boundaries, the different subsystems considered and the main inputs and outputs of the system. No infrastructure process was considered in the evaluation, since the environmental impacts per process unit, from installation, construction, decommissioning, infrastructure, machinery, etc., have been considered negligible during the lifetime of this type of facilities. This has been a common practice in other life cycle assessment studies of biorefineries (Jeswani et al., 2015; Karlsson et al., 2014). However, this study has taken into account the manufacture of the catalyst and the SOFC phase, due to the fact that their useful life is clearly shorter than that of large installations.

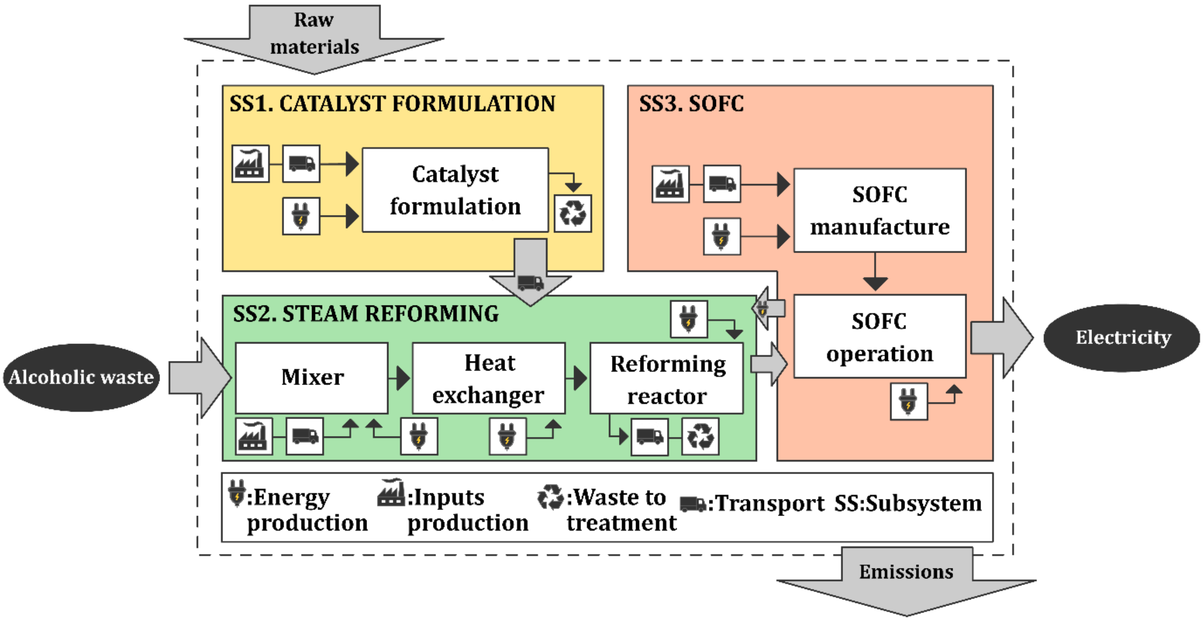


Figure 5.1. System boundaries of the reforming system for the valorisation of the alcoholic waste.

The system boundaries include the process units that are the direct object of this study, but also the production of all the elements necessary for the proper operation of the process. For this purpose, three subsystems were considered, which are detailed below:

Subsystem 1. Catalyst formulation: This subsystem considers all the materials necessary for the manufacture of the catalyst used in the reforming reactor (Menor et al., 2017). The catalyst is composed of a sepiolite base with Nickel (15% weight) and Lanthanum (1% weight). Its estimated useful life is 20 months, with periodic regeneration every 4 months. The transport of the catalyst to the plant was also considered, taking 10 km as an average distance.

Subsystem 2. Steam reforming: This process includes all the inputs needed to perform the steam reforming process. These inputs are mainly electricity, water and alcohol residues from distilleries. The transport of the alcoholic residues to the plant is not included, as this type of installation is designed as an auxiliary process in the distillery. The waste produced in this subsystem is mainly the spent catalyst at the end of its useful life and 100 km was considered as the average transport to landfill (Hajjaji et al., 2013).

Subsystem 3. SOFC: This subsystem includes the net production of electricity in the SOFC using the SS2 gas stream as feed. At the exit of this subsystem, CO_2 and H_2O emissions are derived from the reactions taking place with CO and CH_4, C_2H_4 and H_2 inside the SOFC. The electricity produced is fed into the grid. The SOFC works at 600°C and produces a large amount of heat, as represented in Figure 5.2. This heat is redirected to the system and used to heat the stream entering the reforming reactor. This subsystem includes also the SOFC manufacturing stage. Gas stream purifying processes are not included because SOFC are not poisoned by the presence of CO (Hernández and Kafarov, 2009). It is important to note that the electrodes of this type of battery are catalytic, so they are relatively stable and are not consumed (Fragiacomo et al., 2018).

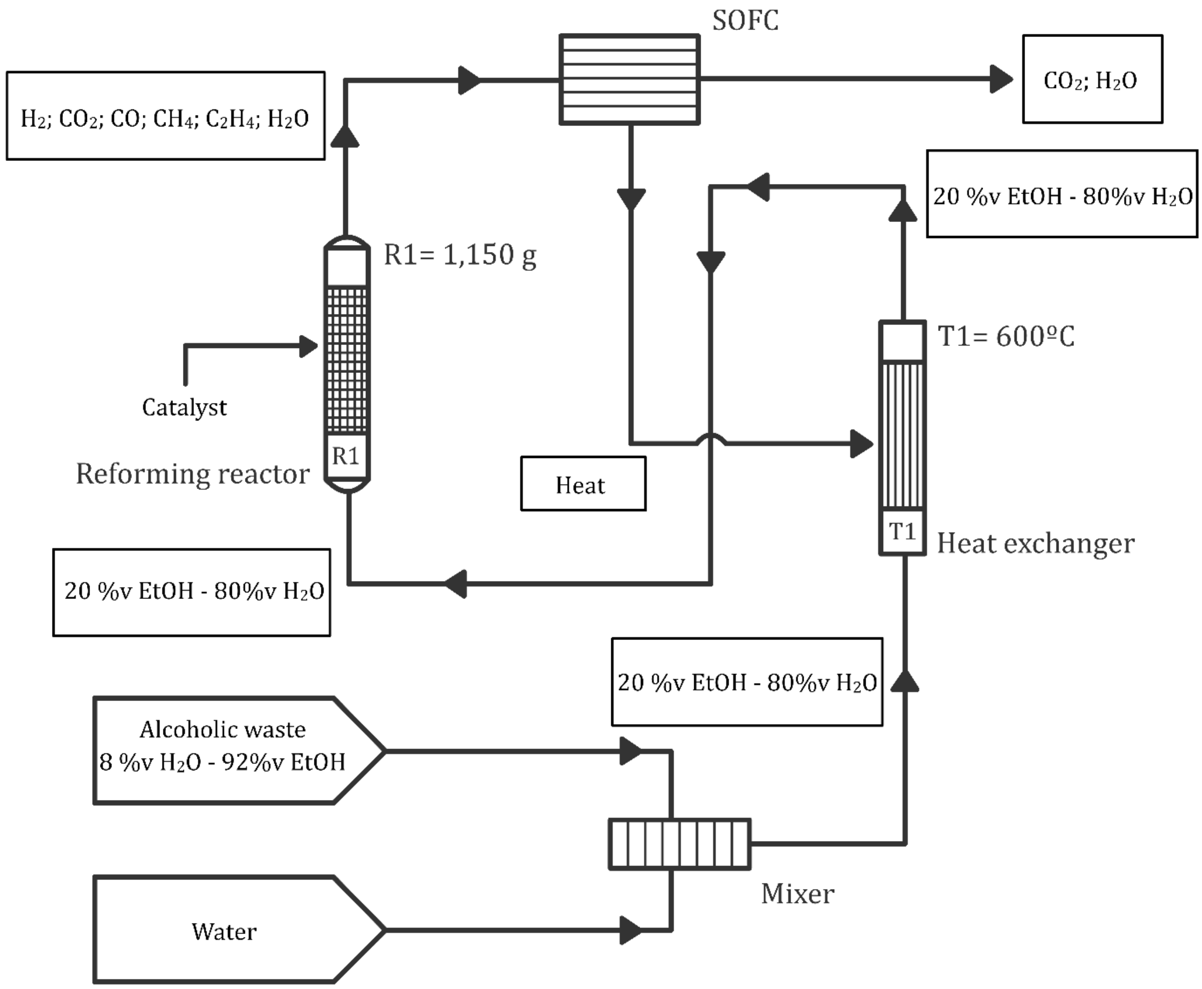


Figure 5.2. Detailed scheme of Subsystems 2 and 3, showing the composition of the different streams and their temperature.

As the basis for the calculation or Functional Unit, the treatment of 1 tonne of alcoholic waste in the facility was chosen, which will be taken as a reference for all the inputs and products of the system as well as the emissions, energy consumption and transport associated with this process (ISO, 2006a, 2006b). The selection of this FU was made considering that the main objective of the system is waste valorisation, so it seems reasonable to choose a feedstock-based functional unit to evaluate the environmental performance of the system. The description of the steam reforming process is presented in detail in Figure 5.2. The waste from the distillery enters the plant with approximately

92% v/v ethanol, to which water is added to reach 80% v/v water and 20% v/v ethanol. The steam-to-carbon molar ratio (S/C ratio) used in most cases is 3:1 (Jeon et al., 2018), but in this case a steam-to-carbon ratio of 6:1 was considered. This steam-to-carbon ratio increases the production of CO_2 and H_2 and reduces the formation of the undesirable products mentioned in Equations 2-4 but increases the heat necessary to vaporise the water/ethanol mixture. However, since this heat is supplied by the residual heat produced in the SOFC, the use of additional fuel is not necessary. The stream passes through a heat exchanger where it is heated up to 600ºC, the temperature required for the inlet stream of the reforming reactor. The output of the reactor is a gas stream comprised by H_2, CO_2, CO, CH_4, H_2O and a minor proportion of C_2H_4. This stream is fed to the SOCF, where heat and electricity are produced. The high temperature stream is recycled in the system to heat the water and ethanol feed, reducing energy consumption. The electricity produced is fed into the grid.

5.2.2. Data acquisition and life cycle inventory

In the case of SS1, catalyst manufacturing data were obtained from a previous study about the formulation of a sepiolite-based catalyst with the addition of Ni (Menor et al., 2017). In relation to SS2, all the information on the consumption of the process comes from primary data, except electricity consumption, which was assumed 1.35 kWh per kg H_2 (Susmozas et al., 2013). Finally, the inventory of Subsystem 3 was obtained from various scientific publications (Strazza et al., 2015, 2010), where several assessment of the environmental profile of SOFC systems with different electrical power (230, 100 and 20 kW) were conducted (Lee et al., 2015; Strazza et al., 2015, 2010). The different life cycle inventories published in the above-mentioned manuscripts were adapted to the characteristics of the SOFC used in this chapter. Table 5.1 presents the life cycle inventory of the three subsystems considered in this chapter.

5.2.3. Life cycle impact analysis: methodology

Simapro 8.5.2 (PRé Consultants, 2017) has been the software used for the implementation of the Life Cycle Inventory. To analyse the inputs and outputs of the Life Cycle Inventory, the Classification and Characterisation guidelines defined by ISO 14040 and ISO 14044 (ISO, 2006a, 2006b) were followed. The environmental results have been presented in terms of the following impact categories of the ReCiPe methodology: Climate Change (CC), Ozone Layer Depletion (OD), Terrestrial Acidification (TA), Freshwater Eutrophication (FE), Marine Eutrophication (ME), Human Toxicity (HT), Photochemical Oxidant Formation (POF), Particulate Matter Formation (PMF), Terrestrial Ecotoxicity (TET), Freshwater Ecotoxicity (FET), Marine Ecotoxicity (MET) and Fossil Fuel Depletion (FD).

Table 5.1. Life cycle inventory for the three subsystems considered in the valorisation of alcoholic waste.

SS1. Catalyst formulation			
Inputs from Technosphere			
Materials	**kg**	**Energy**	**kWh**
Milli Q water	2.63	Electricity	0.72
$Ni(NO_3)_2·6H_2O$	$6.05·10^{-2}$	**Transport**	**kg·km**
$La(NO_3)_3·6H_2O$	$2.54·10^{-3}$	Catalyst to plant	8.15
Sepiolite	$6.84·10^{-2}$		
Outputs to Technosphere			
Products	**kg**	**Waste**	**L**
Catalyst to SS2	$8.15·10^{-2}$	Wastewater	2.63
SS2. Steam reforming			
Inputs from Technosphere			
Materials	**kg**	**Energy**	**MWh**
Water	4,470	Electricity	0.25
Alcoholic waste	1,000	**Transport**	**kg·km**
Catalyst from SS1	$8.15·10^{-2}$	Catalyst waste to landfill	8.15
Outputs to Technosphere			
Products	**kg**	**Waste**	**kg**
Gas stream to SS3	5,509	Catalyst waste to landfill	$8.15·10^{-2}$
SS3. SOFC			
Inputs from Technosphere			
Materials	**kg**	**Materials**	**kg**
$LaMnO_3$	10.05	Polyethylene glycol	0.03
$LaCrO_3$	0.15	Dibutyl phthalate	0.03
Zirconium chloride	1.96	Water, deionised	2.32
Yttrium chloride	0.29	Nitric acid	13.45
Zirconium oxide	0.18	Chlorine	1.61
Nickel oxide	0.18	Petroleum coke	0.26
Nickel	$2.36·10^{-4}$	Urea, as N	6.61
Ethanol	0.44	Gas stream from SS2	5,509
Polyvinyl butyral	0.07	**Transport**	**kg·km**
Energy	**MJ**	Freight rail	31.92
Heating, natural gas	50.76	Lorry > 16t	5.67
Spray drying, natural gas	560.71	Transoceanic freight	202.09
Outputs to Technosphere		**Outputs to Nature**	
Products	**MWh**	**Emissions**	**kg**
Electricity	0.18	CO_2, biogenic	1,800
		CO_2	33.10
		H_2O	5810

5.3. RESULTS AND DISCUSSION

5.3.1. Environmental hotspots linked to the production from steam reforming of alcoholic waste

The environmental characterisation of the alcoholic waste steam reforming resulted in the impacts reported in Table 5.2.

Table 5.2. Inventory data for the subsystems considered in the study per FU.

Impact category	Units	SS1	SS2	SS3	Total
CC	kg CO_2 eq	33.08	109.91	207.79	350.78
OD	kg CFC-11 eq	$5.98 \cdot 10^{-6}$	$1.40 \cdot 10^{-5}$	$4.52 \cdot 10^{-5}$	$6.52 \cdot 10^{-5}$
TA	kg SO_2 eq	0.22	0.67	1.32	2.21
FE	kg P eq	$1.41 \cdot 10^{-2}$	$3.09 \cdot 10^{-2}$	$7.56 \cdot 10^{-2}$	$1.21 \cdot 10^{-1}$
ME	kg N eq	$6.08 \cdot 10^{-2}$	$2.16 \cdot 10^{-2}$	$4.23 \cdot 10^{-2}$	$1.25 \cdot 10^{-1}$
HT	kg 1,4-DB eq	17.11	24.57	122.23	163.91
POF	kg NMVOC	0.10	0.35	0.62	1.07
PMF	kg PM10 eq	0.10	0.24	0.66	1.00
TET	kg 1,4-DB eq	$1.04 \cdot 10^{-2}$	$3.15 \cdot 10^{-3}$	$8.05 \cdot 10^{-2}$	$9.41 \cdot 10^{-2}$
FET	kg 1,4-DB eq	0.76	1.59	4.67	7.01
MET	kg 1,4-DB eq	0.61	1.42	3.68	5.71
FD	kg oil eq	11.65	30.11	89.78	131.55

The SOFC is the main contributor to the Climate Change category, since this subsystem comprises several processes necessary for the manufacture of the main components. In SS3, direct emissions into the atmosphere are quantified, directly from the alcoholic stream, residue produced from grape fermentation. Therefore, CO_2 emissions from SS3 should not be considered as fossil carbon, but as biogenic CO_2. This decision agrees with a study on the production of hydrogen and electricity by reforming supercritical water from bio-glycerol feedstock (Galera and Gutiérrez-Ortiz, 2015). If CO_2 was not considered biogenic, direct CO_2 emissions would represent almost 60% of environmental impact in the climate change category. Therefore, the final impact on this category would be 2,146 kg CO_2 eq per tonne of alcoholic waste processed at the facility.

The Ozone Layer Depletion category has fairly low values in all subsystems. Freshwater and Marine Eutrophication show similar behavior, as they are influenced by the same type of substances. The impact value in these categories is relatively similar in all subsystems, which makes the final impact value low. Human Toxicity and Fossil Depletion present relatively high levels compared to the rest of the impact categories. The reason is the formulation of catalysts and the manufacture of SOFC, since heavy metals and hazardous substances are required that cause a high impact in the HT category. As for the FD category, its value is related to the high consumption of fossil fuels associated with the formulation of catalysts and the manufacture of SOFC.

The specific contribution of each subsystem to the impacts calculated for the system is shown in Figure 5.3. The manufacture and operation of the SOFC (SS3) was found to dominate almost all impact categories, with percentages ranging from 57.7% in Photochemical Oxidant Formation to 85.6% in Terrestrial Ecotoxicity. Additionally, the steam reforming subsystem (SS2) shows a significant contribution to Climate Change (31,3%), Terrestrial Acidification (30,4%) and Photochemical Oxidant Formation (32,9%). The remaining subsystem (SS1) accounted for a contribution percentage around 10%, except for ME, where the contribution was higher than 45%.

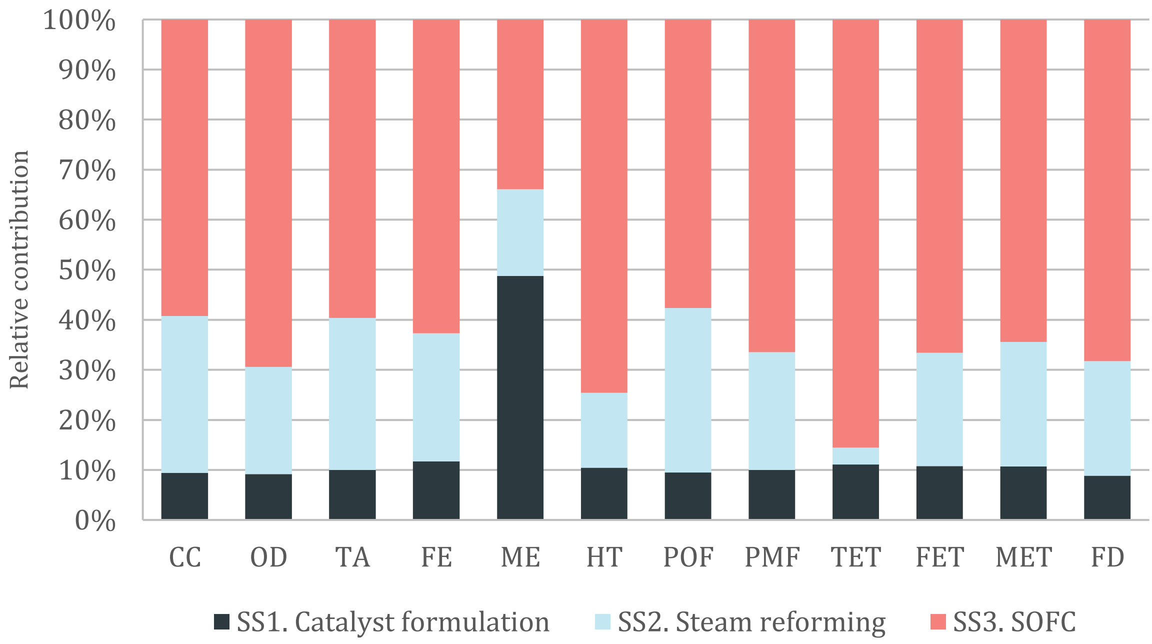


Figure 5.3. Relative contribution (%) of the different subsystems to the total environmental impact.

In order to highlight the processes with the highest environmental impact on the life cycle performance of the system, the individual contributions to the impact are broken down in Figure 5.4. These results show that SOFC manufacturing is the major contribution in almost all impact categories, except for ME. Therefore, the manufacture of SOFC is the main hotspot of the system and must have the highest priority in the improvement actions from the environmental point of view.

The second largest contributor to the total environmental impact is electricity consumption, with contributions percentage ranging from 15% in HT to 32.9% in POF. If electricity production is considered, its contribution to environmental impact decreases significantly to 4.2% and 9.1% in HT and POF, respectively. The formulation of catalysts presented a uniform distribution of environmental impacts in all categories, with contributions always below 10.6%. The consumption of natural gas to heat the process is only responsible for a maximum of 9.4% in CC and 8.5% in FD, but in the rest of the impact categories, their contributions never exceed 6%. The rest of the elements (transport, water consumption and waste treatment) contributed practically irrelevantly to the environmental impact.

Wastewater treatment is the main contributor to the ME impact, due to the high amount of nitrogen-based compounds such as nitrites and nitrates that are discharged in the treated effluent. This may explain, as seen in Figure 5.3, why the main contributor to the ME impact category is SS1, due to the wastewater generated during the formulation of the catalyst.

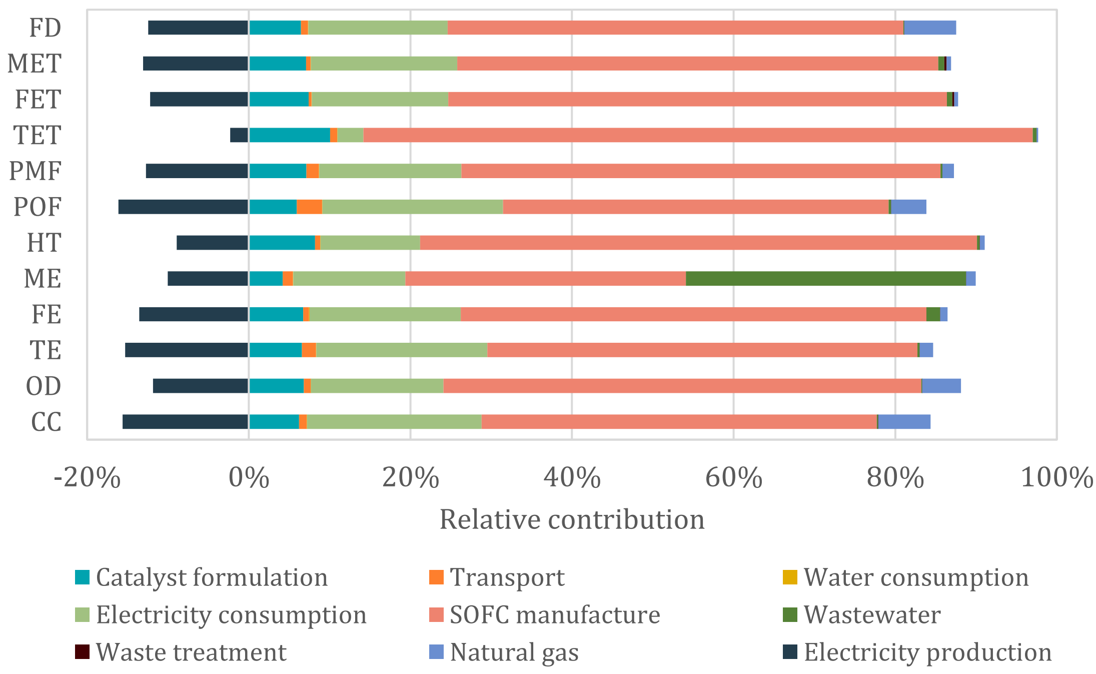


Figure 5.4. Relative contribution (%) of the components of alcoholic waste steam reforming to the overall impact.

5.3.2. Sensitivity analysis. Comparison with traditional waste treatment systems

In order to compare the environmental characterisation results of some alternative waste treatments to steam reforming, a sensitivity analysis was performed. The methods selected for this analysis were landfill and incineration. It is important to note that the inventory data for incineration and landfill were taken from the Ecoinvent® database. Figure 5.5 depicts the environmental performance of the alternative treatments for the alcoholic waste considered. As noted, the steam reforming scenario potentially implied a more acceptable environmental profile than the other scenarios, except for OD, TA, FET and MET categories. It can be concluded that steam reforming reduces GHG emissions by 33% compared to incineration and by 30% compared to landfill. Steam reforming is the largest contributor to OD impact for the emission of harmful gases to the stratospheric ozone layer during background operations such as electricity generation or chemical production. Regarding TA and FE, steam reforming presents the worst environmental performance. The consumption of Ni-based compounds in the manufacture of SOFC and some processes derived from the extraction of lanthanum are the responsible processes of the poor performance in TA and FE, respectively. With respect to TET, steam reforming has worse results, but when considering the three ecotoxicity categories (TTE, MET and FET), the environmental impact of steam reforming is lower, improving by 95.8% compared to incineration and by 97.8% compared to landfill.

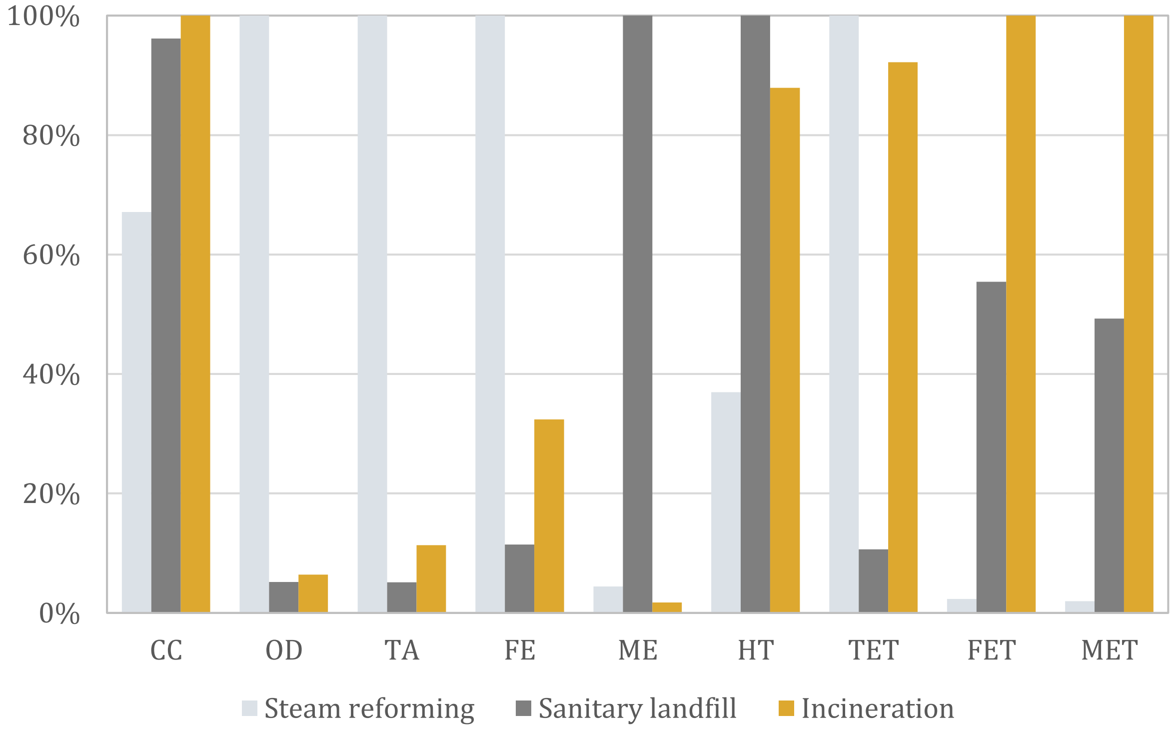


Figure 5.5. Comparative environmental profile of the alternative treatments for alcoholic waste.

5.3.3. Comparison with different hydrogen production alternatives

In addition to the results obtained above considering the basic scheme shown in Figure 3.1, a comparison was made with some processes published in the scientific literature. To be consistent with the different studies considered, the FU was changed to 1 kg of hydrogen produced in the plant with 99.9 vol% purity by steam reforming. The studies selected for comparison were the following reports: Hajjaji et al. (2016, 2013); Khila et al. (2016); Susmozas et al. (2016, 2015, 2013).

With the aim of comparing the environmental performance of the alcoholic waste steam reforming system with different alternatives, it was necessary to consider a new system setup, since the original configuration did not include a purification system downstream the reforming reactor (Figure 5.1). For this reason, it was considered that the WGS process removes carbon monoxide and process a small amount of additional hydrogen. Additionally, in a COPROX reactor the remaining CO can be further reduced to CO_2 in the presence of oxygen. Finally, the PSA process separates H_2 from the rest of the gases in the stream, obtaining H_2 with 99% purity. In summary, the SOFC subsystem was removed, and two additional subsystems were added to purify the hydrogen stream and to provide cooling water, as can be seen in Figure 5.6.

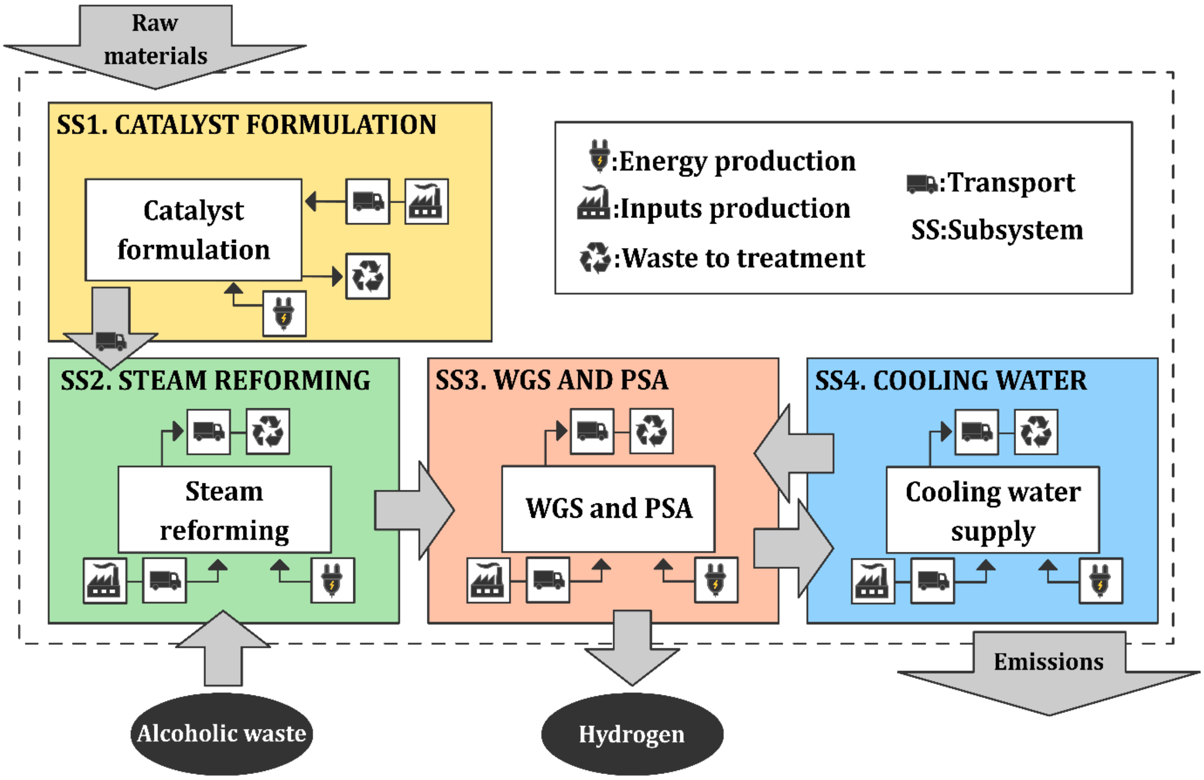


Figure 5.6. New flowchart considered to compare the steam reforming of alcoholic waste other published studies.

The following processes were considered for comparison: SMR-H_2: Steam reforming of methane obtained from natural gas (Susmozas et al., 2013). PG-H_2: Poplar biomass gasification. The system includes the cultivation of poplar and its transport to the plant. Once in the plant are included all the operations necessary to obtain hydrogen and the production of electricity from the steam produced in the system (Susmozas et al., 2013). PG&C-H_2: Gasification of poplar biomass, as mentioned above, but includes carbon fixation during the cultivation stage (Susmozas et al., 2016). GSR-H_2: Glycerol reforming, obtained as a co-product of biodiesel production by transesterification of rapeseed oil. Carbon sequestration during oil production is considered (Susmozas et al., 2015). BSR-H_2: Biofuel reforming, including also carbon fixation produced during the growth of biomass used for biofuel (Susmozas et al., 2015). SBR-H_2: Steam reforming of bioethanol, including bioethanol production (Hajjaji et al., 2016). BAR-H_2: Autothermal reforming of bioethanol (Khila et al., 2016). For comparison, the original life cycle inventories were modified. The WGS data were obtained from Compagnoni et al. (2017). All the necessary data to measure the inputs and outputs in SS3. WGS and PSA and SS4. Cooling water supply were obtained from (Susmozas et al., 2015, 2013). As a result of these calculations, inventory data of WGS, PSA and Cooling water supply can be found in Table 5.5.

Table 5.3. New life cycle inventory to compare the steam reforming of alcoholic wastes with other published processes.

Inputs from Technosphere			
Materials	**kg**	**Energy**	**kWh**
Alcoholic waste	5.42	Electricity	1.36
Water	24.21	**Transport**	**t·km**
SR catalyst	$4.41\cdot10^{-4}$	Road	0.48
WGS catalyst	$1.11\cdot10^{-3}$		
Outputs to the Technosphere			
Products	**kg**	**Emissions**	**kg**
H_2	1.00	CO_2	8.16
Waste	**kg**	CH_4	0.45
SR catalyst to landfill	$1.11\cdot10^{-3}$	C_2H_4	$9.41\cdot10^{-3}$
WGS catalyst to landfill	$4.41\cdot10^{-4}$	CO	1.97
Wastewater from WGS	4.61		

The comparison between the present study and other published processes is possible because the environmental performance of the different studies was published in some LCA studies with a methodological framework consistent with this chapter. The results of the comparison between steam reforming of alcoholic residues and other related processes are presented in terms of the impact categories of the CML methodology: Global warming potential (GWP – kg CO_2 eq), Depletion of abiotic resources (ADP – kg Sb eq), Ozone layer depletion (ODP – kg CFC-11 eq), Photochemical oxidation (POFP – kg C_2H_4 eq), Acidification potential (AP – kg SO_2 eq) and Eutrophication potential (EP – kg PO_4^{3-}). The selection of this assessment method was made for methodological standardisation, as this impact method, which differs significantly from ReCiPe in some impact categories, was used in all cases selected for comparison.

The environmental impacts of hydrogen production systems are displayed in Figure 5.7. For example, alcoholic waste steam reforming shows the best results in terms of ADP, ODP, AP and EP, but performs worse in GWP and has the worst result in POFC.

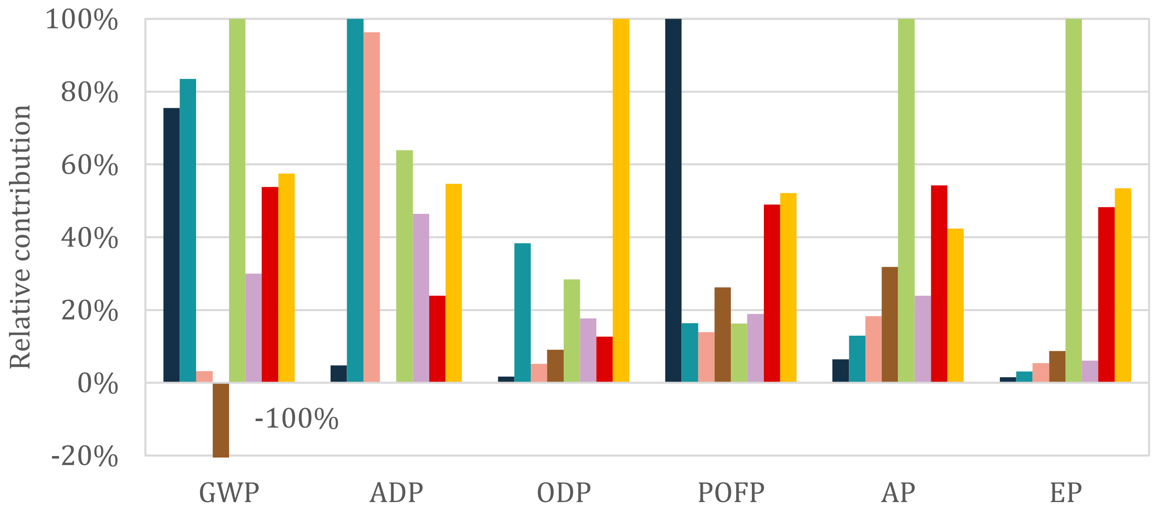


Figure 5.7. Comparison (in %) of different reforming processes to obtain hydrogen. Caption: Dark Blue: Steam reforming of alcoholic waste; Light Blue: SMR-H_2; Pink: PG-H_2; Brown: PG&C-H_2; Green: GSR-H_2; Purple: BSR-H_2; Red: SBR-H_2; Yellow: BAR-H_2.

Steam reforming of alcoholic residues has the highest value in POF, due to direct emissions of CH_4, which occur in relatively high quantities during steam reforming. However, this process performs well in terms of ODP with a value around 2% of BAR-H_2, which is the process with the worst environmental performance in this category. The total GHG emissions of the system are estimated at approximately 9.55 kg CO_2 eq per kg of H_2 produced, this value is relatively higher than that of other technologies but is considerably lower than that of a conventional H_2 production system (SMR-H_2). Approximately 90% of these emissions are attributed to direct methane emissions from the reforming reactor, as CH_4 is 21 times more likely to affect GWP over a 100-year period, according to IPCC. The lowest value in this category corresponds to PG&G-H_2 as this process considers CO_2 capture during biomass cultivation. This explains the importance of system boundaries in an LCA study since PG&G-H_2 covers from biomass cultivation to hydrogen production with CO_2 capture. However, in this chapter, the system boundaries encompass from alcoholic residues entering the plant to the production of electricity, so carbon sequestration during biomass cultivation was not considered.

5.4. CONCLUSIONS

From a life cycle perspective, the results suggest that this type of energy systems that produce hydrogen from alcoholic waste through steam reforming present a good environmental performance. Overall, steam reforming of this type of alcoholic waste for energy production could play a significant role in future energy systems.

The SOFC is the main contributor to environmental impact in most impact categories. Analysing the different processes, the manufacture of SOFC is the process with the greatest environmental impact in all impact categories except in ME, where wastewater treatment is the main contributor. It can be concluded that extending the lifetime of these devices can be an important element in reducing the environmental impacts of these systems. The sensitivity analysis shows the promising performance of this waste treatment, since the treatment of 1 tonne of alcoholic waste produces 351 kg of CO_2 eq, this result is 33% and 30% better than incineration and sanitary landfill respectively.

Comparative analysis has been useful to compare this process with others related to the production of hydrogen from different raw materials. Although steam reforming has some poor results in GWP and POF due to methane emissions, its environmental performance is generally better than other processes published elsewhere. However, these results have been obtained by evaluating the data taken on a laboratory scale, and more studies on a larger scale will be needed in the future to determine a more accurate estimate of the actual environmental profile of the process. It is clear that the combination of this technology together with the other systems discussed in Chapters 3 and 4 is a good start in the search for the integral valorisation of winemaking waste.

5.5. REFERENCES

Aguilar-Sánchez, P., Navarro-Pineda, F.S., Sacramento-Rivero, J.C., Barahona-Pérez, L.F., 2018. Life-cycle assessment of bioethanol production from sweet sorghum stalks cultivated in the state of Yucatan, Mexico. Clean Technol. Environ. Policy 20, 1685–1696. https://doi.org/10.1007/s10098-017-1480-4

Alamolhoda, S., Vitale, G., Hassan, A., Nassar, N.N., Almao, P.P., 2019. Synergetic effects of cerium and nickel in Ce-Ni-MFI catalysts on low-temperature water-gas shift reaction. Fuel 237, 361–372. https://doi.org/10.1016/j.fuel.2018.09.096

Alipour-Moghadam, R., Yusup, S., Azlina, W., Nehzati, S., Tavasoli, A., 2014. Investigation on syngas production via biomass conversion through the integration of pyrolysis and air-steam gasification processes. Energy Convers. Manag. 87, 670–675. https://doi.org/10.1016/j.enconman.2014.07.065

Anzelmo, B., Wilcox, J., Liguori, S., 2018. Hydrogen production via natural gas steam reforming in a Pd-Au membrane reactor. Investigation of reaction temperature and GHSV effects and long-term stability. J. Memb. Sci. 568, 113–120. https://doi.org/10.1016/j.memsci.2018.09.054

Balat, M., 2011. Production of bioethanol from lignocellulosic materials via the biochemical pathway: A review. Energy Convers. Manag. 52, 858–875. https://doi.org/10.1016/j.enconman.2010.08.013

Braga, A.H., Sodré, E.R., Batista, J., Santos, O., Paula, C.M. De, Maria, J., Bueno, C., 2016. Steam reforming of acetone over Ni- and Co-based catalysts : Effect of the composition of reactants and catalysts on reaction pathways. Appl. Catal. B, Environ. 195, 16–28. https://doi.org/10.1016/j.apcatb.2016.04.047

Comité Européen des Enterprises Vins, 2016. European Wine: a solid pillar of the European Union economy. Brussels.

Compagnoni, M., Mostafavi, E., Tripodi, A., Mahinpey, N., Rossetti, I., 2017. Techno-economic Analysis of a Bioethanol to Hydrogen Centralized Plant. Energy and Fuels 31, 12988–12996. https://doi.org/10.1021/acs.energyfuels.7b02434

Coronado, I., Pitínová, M., Karinen, R., Reinikainen, M., Puurunen, R.L., Lehtonen, J., 2018. Aqueous-phase reforming of Fischer-Tropsch alcohols over nickel-based catalysts to produce hydrogen: Product distribution and reaction pathways. Appl. Catal. A Gen. 567, 112–121. https://doi.org/10.1016/j.apcata.2018.09.013

Costa, D., Jesus, J., Virgínio e Silva, J., Silveira, M., 2018. Life Cycle Assessment of Bioethanol Production from Sweet Potato (*Ipomoea batatas L.*) in an Experimental Plant. Bioenergy Res. 11, 715–725. https://doi.org/10.1007/s12155-018-9932-1

Da Costa-Serra, J.F., Chica, A., 2018. Catalysts based on Co-Birnessite and Co-Todorokite for the efficient production of hydrogen by ethanol steam reforming. Int. J. Hydrogen Energy 43, 16859–16865. https://doi.org/10.1016/j.ijhydene.2017.12.114

Díaz Alvarado, F., Gracia, F., 2010. Steam reforming of ethanol for hydrogen production: Thermodynamic analysis including different carbon deposits representation. Int. J. Hydrogen Energy 165, 649–657. https://doi.org/10.1016/j.cej.2010.09.051

Esteve-Adell, I., Crapart, B., Primo, A., Serp, P., Garcia, H., 2017. Aqueous phase reforming of glycerol using doped graphenes as metal-free catalysts. Green Chem. 19, 3061–3068. https://doi.org/10.1039/c7gc01058c

Fragiacomo, P., de Lorenzo, G., Corigliano, O., 2018. Performance Analysis of a Solid Oxide Fuel Cell-Gasifier Integrated System in Co-Trigenerative Arrangement. J. Energy Resour. Technol. 140, 092001 (1–9). https://doi.org/10.1115/1.4039872

Frolov, S.M., Medvedev, S.N., Basevich, V.Y., Frolov, F.S., 2013. Self-ignition of hydrocarbon-hydrogen-air mixtures. Int. J. Hydrogen Energy 38, 4177–4184. https://doi.org/10.1016/j.ijhydene.2013.01.075

Galera, S., Gutiérrez-Ortiz, F.J., 2015. Life cycle assessment of hydrogen and power production by supercritical water reforming of glycerol. Energy Convers. Manag. 96, 637–645. https://doi.org/10.1016/j.enconman.2015.03.031

García, L., Valiente, A., Oliva, M., Ruiz, J., Arauzo, J., 2018. Influence of operating variables on the aqueous-phase reforming of glycerol over a Ni/Al coprecipitated catalyst. Int. J. Hydrogen Energy 43, 20392–20407. https://doi.org/10.1016/j.ijhydene.2018.09.119

González-García, S., Iribarren, D., Susmozas, A., Dufour, J., Murphy, R.J., 2012. Life cycle assessment of two alternative bioenergy systems involving *Salix spp*. biomass: Bioethanol production and power generation. Appl. Energy 95, 111–122. https://doi.org/10.1016/j.apenergy.2012.02.022

Hajjaji, N., Khila, Z., Baccar, I., Pons, M.N., 2016. A thermo-environmental study of hydrogen production from the steam reforming of bioethanol. J. Energy Storage 7, 204–219. https://doi.org/10.1016/j.est.2016.06.010

Hajjaji, N., Pons, M.N., Renaudin, V., Houas, A., 2013. Comparative life cycle assessment of eight alternatives for hydrogen production from renewable and fossil feedstock. J. Clean. Prod. 44, 177–189. https://doi.org/10.1016/j.jclepro.2012.11.043

Halleux, H., Lassaux, S., Renzoni, R., Germain, A., 2008. Comparative Life Cycle Assessment of Two Biofuels. Ethanol from Sugar Beet and Rapeseed Methyl Ester. Int. J. Life Cycle Assess. 13, 184–190. https://doi.org/10.1065/lca2008.03.382

Haryanto, A., Fernando, S., Murali, N., Adhikari, S., 2005. Current status of hydrogen production techniques by steam reforming of ethanol: A review. Energy & Fuels 19, 2098–2106. https://doi.org/10.1021/ef0500538

Hernández, L., Kafarov, V., 2009. Use of bioethanol for sustainable electrical energy production. Int. J. Hydrogen Energy 34, 7041–7050. https://doi.org/10.1016/j.ijhydene.2008.07.089

ISO, 2006a. ISO 14040 - Environmental Management - Life Cycle Assessment - Principles and Framework.

ISO, 2006b. ISO 14044 - Environmental Management - Life Cycle Assessment - Requirements and Guidelines.

Jeon, J., Nam, S., Ko, C.H., 2018. Rapid evaluation of coke resistance in catalysts for methane reforming using low steam-to-carbon ratio. Catal. Today 309, 140–146. https://doi.org/10.1016/j.cattod.2017.08.051

Jeswani, H.K., Falano, T., Azapagic, A., 2015. Life cycle environmental sustainability of lignocellulosic ethanol produced in integrated thermo-chemical biorefineries. Biofuels, Bioprod. Biorefining 9, 661–676. https://doi.org/10.1002/bbb

Karlsson, H., Börjesson, P., Hansson, P.A., Ahlgren, S., 2014. Ethanol production in biorefineries using lignocellulosic feedstock - GHG performance, energy balance and implications of life cycle calculation methodology. J. Clean. Prod. 83, 420–427. https://doi.org/10.1016/j.jclepro.2014.07.029

Khaodee, W., Wongsakulphasatch, S., Kiatkittipong, W., Arpornwichanop, A., Laosiripojana, N., Assabumrungrat, S., 2011. Selection of appropriate primary fuel for hydrogen production for different fuel cell types: Comparison between decomposition and steam reforming. Int. J. Hydrogen Energy 36, 7696–7706. https://doi.org/10.1016/j.ijhydene.2011.03.123

Khila, Z., Baccar, I., Jemel, I., Hajjaji, N., 2017. Thermo-environmental life cycle assessment of hydrogen production by autothermal reforming of bioethanol. Energy Sustain. Dev. 37, 66–78. https://doi.org/10.1016/j.esd.2016.12.003

Khila, Z., Baccar, I., Jemel, I., Houas, A., Hajjaji, N., 2016. Energetic, exergetic and environmental life cycle assessment analyses as tools for optimization of hydrogen production by autothermal reforming of bioethanol. Int. J. Hydrogen Energy 41, 17723–17739. https://doi.org/10.1016/j.ijhydene.2016.07.225

Kumar, B., Kumar, Shashi, Sinha, S., Kumar, Surendra, 2018. Utilization of acetone-butanol-ethanol-water mixture obtained from biomass fermentation as renewable feedstock for hydrogen production via steam reforming: Thermodynamic and energy analyses. Bioresour. Technol. 261, 385–393. https://doi.org/10.1016/j.biortech.2018.04.035

Lee, Y.D., Ahn, K.Y., Morosuk, T., Tsatsaronis, G., 2015. Environmental impact assessment of a solid-oxide fuel-cell-based combined-heat-and-power-generation system. Energy 79, 455–466. https://doi.org/10.1016/j.energy.2014.11.035

López, E.R., Dorado, F., de Lucas-Consuegra, A., 2019. Electrochemical promotion for hydrogen production via ethanol steam reforming reaction. Appl. Catal. B Environ. 243, 355–364. https://doi.org/10.1016/j.apcatb.2018.10.062

Luo, L., van der Voet, E., Huppes, G., 2009. Life cycle assessment and life cycle costing of bioethanol from sugarcane in Brazil. Renew. Sustain. Energy Rev. 13, 1613–1619. https://doi.org/10.1016/j.rser.2008.09.024

Menezes, J., Manfro, R., Souza, M., 2018. Hydrogen production from glycerol steam reforming over nickel catalysts supported on alumina and niobia: Deactivation process, effect of reaction conditions and kinetic modeling. Int. J. Hydrogen Energy 43, 15064–15082. https://doi.org/10.1016/j.ijhydene.2018.06.048

Menor, M., Sayas, S., Chica, A., 2017. Natural sepiolite promoted with Ni as new and efficient catalyst for the sustainable production of hydrogen by steam reforming of the biodiesel by-products glycerol. Fuel 193, 351–358. https://doi.org/10.1016/j.fuel.2016.12.068

Morales, M., Roa, J.J., Capdevila, X.G., Segarra, M., Piñol, S., 2010. Mechanical properties at the nanometer scale of GDC and YSZ used as electrolytes for solid oxide fuel cells. Acta Mater. 58, 2504–2509. https://doi.org/10.1016/j.actamat.2009.12.036

Ni, M., Leung, D.Y.C., Leung, M.K.H., 2007. A review on reforming bio-ethanol for hydrogen production. Int. J. Hydrogen Energy 32, 3238–3247. https://doi.org/10.1016/j.ijhydene.2007.04.038

PRé Consultants, 2017. SimaPro Database Manual (No. Methods Library). The Netherlands.

Rabenstein, G., Hacker, V., 2008. Hydrogen for fuel cells from ethanol by steam-reforming, partial-oxidation and combined auto-thermal reforming: A thermodynamic analysis. J. Power Sources 185, 1293–1304. https://doi.org/10.1016/j.jpowsour.2008.08.010

Ramírez, P., Homs, N., 2008. Use of biofuels to produce hydrogen (reformation processes). Chem. Soc. Rev. 37, 2459–2467. https://doi.org/10.1039/b712181b

Reyes-Valle, C., Villanueva-Perales, A.L., Vidal-Barrero, F., Ollero, P., 2015. Integrated economic and life cycle assessment of thermochemical production of bioethanol to reduce production cost by exploiting excess of greenhouse gas savings. Appl. Energy 148, 466–475. https://doi.org/10.1016/j.apenergy.2015.03.113

Rossetti, I., Compagnoni, M., Torli, M., 2015a. Process simulation and optimization of H_2 production from ethanol steam reforming and its use in fuel cells. 1. Thermodynamic and kinetic analysis. Chem. Eng. J. 281, 1024–1035. https://doi.org/10.1016/j.cej.2015.08.045

Rossetti, I., Compagnoni, M., Torli, M., 2015b. Process simulation and optimization of H_2 production from ethanol steam reforming and its use in fuel cells. 2. Process analysis and optimization. Chem. Eng. J. 281, 1036–1044. https://doi.org/10.1016/j.cej.2015.08.045

Salkuyeh, Y.K., Saville, B.A., MacLean, H.L., 2018. Techno-economic analysis and life cycle assessment of hydrogen production from different biomass gasification processes. Int. J. Hydrogen Energy 43, 9514–9528. https://doi.org/10.1016/j.ijhydene.2018.04.024

Spallina, V., Matturro, G., Ruocco, C., Meloni, E., Palma, V., Fernandez, E., Melendez, J., Pacheco-Tanaka, A.D., Viviente-Sole, J.L., van Sint-Annaland, M., Gallucci, F., 2018. Direct route from ethanol to pure hydrogen through autothermal reforming in a membrane reactor: Experimental demonstration, reactor modelling and design. Energy 143, 666–681. https://doi.org/10.1016/j.energy.2017.11.031

Spath, P.L., Mann, M.K., 2001. Life cycle assessment of hydrogen production via natural gas steam reforming, National Renewable Energy Laboratory.

Spiridon, I., Darie-Nita, R.N., Hitruc, G.E., Ludwiczak, J., Cianga, I.A., Niculaua, M., 2016. New opportunities to valorize biomass wastes into green materials. J. Clean. Prod. 133, 235–242. https://doi.org/10.1016/j.jclepro.2016.05.143

Strazza, C., Del Borghi, A., Costamagna, P., Gallo, M., Brignole, E., Girdinio, P., 2015. Life Cycle Assessment and Life Cycle Costing of a SOFC system for distributed power generation. Energy Convers. Manag. 100, 64–77. https://doi.org/10.1016/j.enconman.2015.04.068

Strazza, C., Del Borghi, A., Costamagna, P., Traverso, A., Santin, M., 2010. Comparative LCA of methanol-fuelled SOFCs as auxiliary power systems on-board ships. Appl. Energy 87, 1670–1678. https://doi.org/10.1016/j.apenergy.2009.10.012

Susmozas, A., Iribarren, D., Dufour, J., 2015. Assessing the Life-Cycle Performance of Hydrogen Production via Biofuel Reforming in Europe. Resources 4, 398–411. https://doi.org/10.3390/resources4020398

Susmozas, A., Iribarren, D., Dufour, J., 2013. Life-cycle performance of indirect biomass gasification as a green alternative to steam methane reforming for hydrogen production. Int. J. Hydrogen Energy 38, 9961–9972. https://doi.org/10.1016/j.ijhydene.2013.06.012

Susmozas, A., Iribarren, D., Zapp, P., Linβen, J., Dufour, J., 2016. Life-cycle performance of hydrogen production via indirect biomass gasification with CO_2 capture. Int. J. Hydrogen Energy 41, 19484–19491. https://doi.org/10.1016/j.ijhydene.2016.02.053

Tian, J., Ke, Y., Kong, G., Tan, M., Wang, Y., Lin, J., Zhou, W., Wan, S., 2017. A novel structured PdZnAl/Cu fiber catalyst for methanol steam reforming in microreactor. Renew. Energy 113, 30–42. https://doi.org/10.1016/j.renene.2017.04.070

Tian, Y., Zhou, X., Lin, S., Ji, X., Bai, J., Xu, M., 2018. Syngas production from air-steam gasification of biomass with natural catalysts. Sci. Total Environ. 645, 518–523. https://doi.org/10.1016/j.scitotenv.2018.07.071

Uusitalo, V., Väisänen, S., Inkeri, E., Soukka, R., 2017. Potential for greenhouse gas emission reductions using surplus electricity in hydrogen, methane and methanol production via electrolysis. Energy Convers. Manag. 134, 125–134. https://doi.org/10.1016/j.enconman.2016.12.031

Valente, A., Iribarren, D., Gálvez-Martos, J.L., Dufour, J., 2019. Robust eco-efficiency assessment of hydrogen from biomass gasification as an alternative to conventional hydrogen: A life-cycle study with and without external costs. Sci. Total Environ. 650, 1465–1475. https://doi.org/10.1016/j.scitotenv.2018.09.089

Wang, M., Au, C.T., Lai, S.Y., 2015. H_2 production from catalytic steam reforming of n-propanol over ruthenium and ruthenium-nickel bimetallic catalysts supported on ceria-alumina oxides with different ceria loadings. Int. J. Hydrogen Energy 40, 13926–13935. https://doi.org/10.1016/j.ijhydene.2015.07.162

Xue, Z., Shen, Y., Zhu, S., Li, P., Zeng, Y., Xi, Z., Cai, Y., 2017. Autothermal reforming of ethyl acetate for hydrogen production over $Ni_3La_7O_y/Al_2O_3$ catalyst. Energy Convers. Manag. 146, 34–42. https://doi.org/10.1016/j.enconman.2017.05.018

Zheng, T., Zhou, W., Yu, W., Ke, Y., Liu, Y., Liu, R., San Hui, K., 2019. Methanol steam reforming performance optimisation of cylindrical microreactor for hydrogen production utilising error backpropagation and genetic algorithm. Chem. Eng. J. 357, 641–654. https://doi.org/10.1016/j.cej.2018.09.129

SECTION III

CIRCULAR ECONOMY IN THE FISHERIES SECTOR

Chapter 6

Evaluation of the environmental sustainability of the inshore great scallop (*Pecten maximus*) fishery in Galicia

Summary

Great scallop (*Pecten maximus*) is a bivalve species belonging to the *Pectinidae* family that is deeply linked to the Way of Saint James, becoming a gourmet product of the Galician cuisine. The goal of this chapter was to fill the gaps that currently exist in a specific inventory dedicated to the capture and processing of great scallop, as well as to assess the environmental burdens derived from this system. The construction of a comprehensive life cycle inventory was one of the fundamental stages, gathering primary information from questionnaires filled out by 14 skippers and the manager of the evisceration plant in the port of Cambados. Diesel consumption of the fishing boats was identified as the major contributor to the environmental burdens. Furthermore, electricity consumption on the evisceration plant also stood out as a determinant element in the environmental profile of the product. The integrated GHG emission/protein content correlation placed the scallop in the quadrant of high protein content (the highest of the fisheries), but also high environmental impact (mainly due to the high FUI and low edible yield).

This chapter, under the consent of the publisher, was redrafted after: Cortés, A., González-García, S., Franco-Uría, A., Moreira, M. T., Feijoo, G. (2021). Evaluation of the environmental sustainability of the inshore great scallop (Pecten maximus) fishery in Galicia. Journal of Industrial Ecology, 1-13. DOI: https://doi.org/10.1111/jiec.13153. More information related to this article can be found in Appendix III.

OUTLINE

6.1. INTRODUCTION

Within the different aquatic species, bivalves have traditionally been considered a source of healthy animal protein and high levels of essential fatty acids, which has led to a significant increase in consumer demand. In fact, it is estimated that around 25% of the seafood consumed in Spain in 2018 is canned, fresh or frozen molluscs, of which a significant percentage correspond to bivalve species (MAPA, 2021). Due to their excellent organoleptic qualities, the consumption of bivalves has traditionally been associated with products of high commercial value, representing a gourmet product, as is the case of oysters, scallops and clams.

Great scallop (*Pecten maximus*) is a bivalve species that belongs to the *Pectinidae* family, commonly referred as "scallop". Great scallop is essentially a coastal species that lives on clean firm sand, fine or sandy gravel bottoms (Andrew R Brand, 2006), which feeds mainly on phytoplankton, algae and organic particles in suspension. This species is characterised by its wide geographical distribution along the European Atlantic coastline from Spain to Norway (Andrew R. Brand, 2006). Scallops have been commercially landed in Europe for over 100 years, but modern dredge fisheries really began to develop in the 1950s and 1960s around the coasts of the British Isles and France (Duncan et al., 2016). Since then, landings of *P. maximus* have remained constant, accounting for about 67% of total European scallop landings in 2013 (European Commission, 2020). Given that this species is not well managed in much of its fishing area, and coupled with the significant increase in fishing effort, there is growing concern about the long-term environmental sustainability of this fishery. In fact, *P. maximus* fisheries in Europe are now almost all fully exploited or overexploited, becoming dependent on fishing catch limits. With the natural variability of scallop catches, this has led to instability of supplies for this fishery (Duncan et al., 2016).

If European landing data are compared with the Spanish scenario, significant divergences are observed. In 2018, 116 tonnes of great scallops were caught (MAPA, 2020a), which were entirely captured in the Galician "rias" (Figure 6.1). The Galician rias are complex ecosystems with a unique biodiversity, quality and abundance of marine resources, as demonstrated by the fishing and shellfish farming tradition in the region (Picado et al., 2016). The reason behind the low catches lies in the strict control that is followed in Galicia to respect closed seasons and catch per boat ratios. In fact, for the scallop fishery the following regulations apply to ensure a continuous supply over the years and avoid the depletion of scallop stocks: (i) The gears, tackle, tools, implements, equipment and techniques permitted for the professional extraction of live marine resources are regulated by the DECREE 15/2011 of January 28; (ii) The minimum size of various fishery products in the Autonomous Community of Galicia, including scallops are regulated by the ORDER of July 27, 2012; and (iii) The closed season is established every year in the general plan of shellfish exploitation, published by the Regional Government. The corresponding plan for this article is the ORDER of December 20, 2018.

Thus, the scarcity of fresh scallops in Spanish markets, together with the nutritional quality of all bivalve species, makes the Galician great scallop highly appreciated, becoming a gastronomic reference and a gourmet product of the Galician cuisine. In fact, the market Price of a Galician great scallop can reach around 5-6 euros per unit in an average Spanish market (typically sold including the shell), depending on the size of the product.

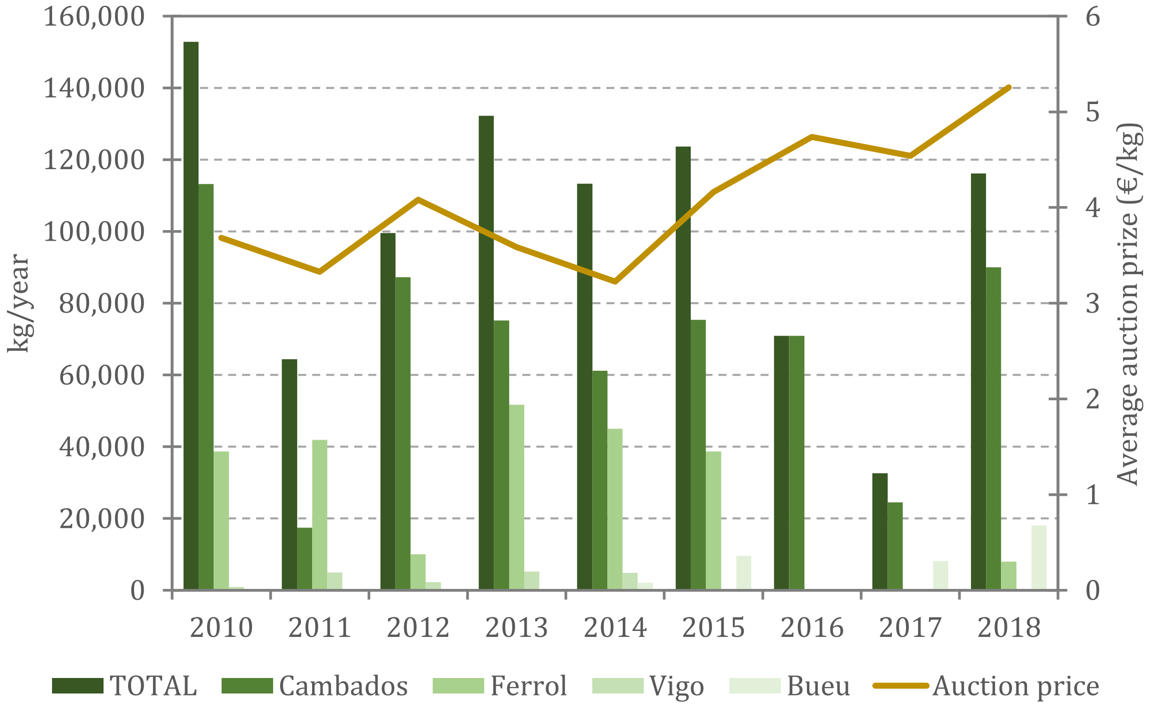


Figure 6.1. Catches of Atlantic scallop during the years 2010-2018 in different Galician ports. The red line represents the average auction price. Source: (Xunta de Galicia, 2020).

Taking into account the growing global demand for fishery products, both from catches and aquaculture (FAO, 2020), as well as the high environmental costs of fishing, it is increasingly essential to assess the environmental burdens of the fisheries and aquaculture sectors. The Life Cycle Assessment methodology (ISO 14040, 14044) can be used to support decision-making in fisheries by identifying critical points to reduce their environmental impacts or by comparing several alternative systems (Ruiz-Salmón et al., 2021). The application of LCA methodology to determine the environmental impacts of fish catches, farming (aquaculture), and processing started in the mid-2000s. It is important also to highlight that not only fishing activities and catches have been traditionally evaluated, but also the production of processed seafood products.

This chapter aims to analyse the environmental impacts related to the capture, landing and processing of scallops by the Galician fleet in the "Ría de Arousa" through the LCA methodology. Beyond this objective, it is necessary to continue to raise awareness among stakeholders and consumers about the environmental impact of different products and services. It is especially interesting to know the environmental implications of this gourmet product, which is not only a reference of tasty delicacy, but also a symbol in the traditional popularity of the Way of Saint James. Preserving the traditional values associated with Galician gastronomic culture is in itself a long-term objective, and the fact that this work sets out to understand the environmental profile associated with the capture, landing and processing of scallops may demonstrate the potency of the Galician fishing sector. Finally, a comparison of the environmental and nutritional quality in terms of greenhouse gas emissions and protein content with respect to other widely consumed foods is provided.

6.2. MATERIAL AND METHODS

6.2.1. Definition of goal and scope. Functional Unit

Due to the lack of a specific inventory dedicated to the capture and processing of Galician great scallops, the objective of this LCA study is to fill those gaps, providing valuable information to assess the environmental burdens associated with capture and processing operations related to the extraction of great scallops in Galician waters. The scope of this chapter focuses on all the stages required for the extraction and processing of great scallops. Waste treatment operations were taken into account within the system boundaries, corresponding to a cradle-to-gate analysis (Guinée et al., 2001)

The Functional Unit chosen to analyse the capture and processing of the great scallop is based on a product-oriented approach (1 raw eviscerated frozen scallop that left the processing plant ready for the market). This FU contains 139.5 g of great scallop, 3.3 g of plastic film and 5 g of plastic label. Due to the waste generated during the processing stage, 139.5 g of final eviscerated scallop correspond to 155 g of landed scallop. The edible meat of the scallop is 20.5 g, which corresponds to 13.2% of the gross weight of the scallop. This value is in line with Tyedmers (2004), where it is stated that the abductor muscle in scallops generally represents around 10-12% of the live weight of the animal. It is important to note the reason for selecting this FU. During the months of December to March, while the fishing season is open, scallops are sold fresh, while the rest of the year, scallops caught during these months are sold frozen. Since most of the Galician scallops are sold frozen, it was decided to analyse this format of ready-to-sell product because it is more representative.

6.2.2. Description of fishing and post-harvesting operations

The ideal season for scallop fishing is mainly during the winter months and therefore, the scallop season in Galician waters runs from December to February/March. Twenty-one trawlers with a scallop fishing license operate in the port of Cambados, which in 2018 reached a total of 90,029 kg of great scallop. Hence, the fishing and processing system evaluated consists of two subsystems which are SS1 – Vessel operations and SS2 – Post-harvesting operations. Figure 6.2 shows the subsystems and process steps included within the system boundaries.

The vessel operations subsystem includes all activities that are carried out until the boat arrives at the port of Cambados, where all the catches of the day are landed. The assessed fleet operates in waters within the Ria de Arousa between 5-9 miles operating at a speed of 2-2.5 knots. The fleet is composed entirely of small-scale boats with an average size of 10,7 m in length, 3-4 m in beam and an average gross tonnage of 7.9. It is important to note that once the scallop season is over, vessels may engage in other traditional fisheries, being the following the main gears and in brackets the target species: “Xeito” (pilchard), “Miños” (spider crab), “Vetas” (mackerel, pout), “Bou de vara” (queen scallop, Velvet swim crab, spider crab) and “Bou de man” (cuttlefish, octopus).

In order to start the fishing period, the Fishermen´s Association must submit a catch and processing plan together with an authorised company (in charge of the evisceration processes) to the Regional Government for approval. The Cambados Fishermen´s Association has a plan in place to ensure the sustainability of the fishery and the commercial value of the product based on two pillars: (1) a minimum size of 115 mm, (2) a maximum daily catch for the entire fleet of 3,000 kg, a quota that is shared proportionally among all fishermen in the fleet.

This fishing activity is the only bottom trawling gear allowed in the Galician small-scale fisheries, restricted to in-shore water of the Ria de Arousa and is not allowed in the other Galician rias (Outeiro et al., 2020). Great scallop fishery is carried out using a dredge (Figure 6.3), which is made of steel and the net is made of polyethylene with a mesh width of about 100 mm and a total weight of 2 kg. Other characteristics of the fleet are two days of rest per week and a maximum power set at 500 hp. Although it is well documented the impact that some types of toothed dredges can cause on other species living in or on the seabed due to the effect of their long teeth (Hinz et al., 2012; Stewart and Howarth, 2016), it is important to note that this particular fishing gear does not dredge the ground, but slides parallel to the bottom, remaining open up to 4-5 cm thanks to the speed of the vessel, reducing the impact caused on the seabed. The “sweep chain” is made up of a series of metal teeth inclined inwards, which allows the dredge to pass obstacles without dragging them.

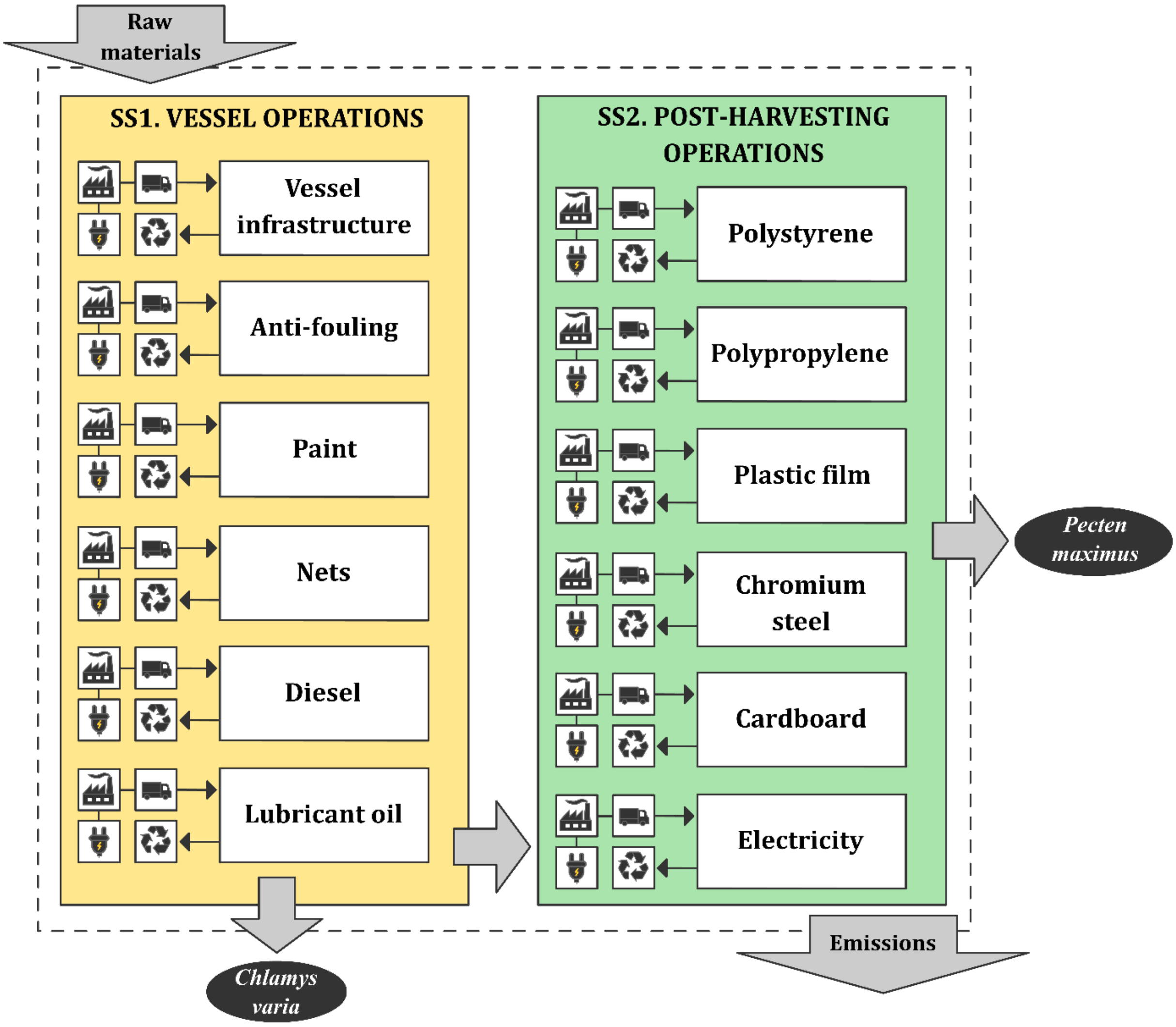


Figure 6.2. Flow chart of scallop fishing and processing. Legend: Black: Subsystems; Cream: Inputs; Yellow: Emissions and waste generated; Green: Products and co-products.

It is documented that scallop fisheries in general are relatively target-species specific (Duncan et al., 2016). In fact, a study conducted on the queen scallop trawl fishery (Duncan, 2009), indicated a relatively low level (3.4%) of by-catch while Boyle and Thompson (2012) reported similar general trends (7.4%), but highlighting the species variability in queen scallop trawl by-catch (Duncan et al., 2016). It is noteworthy that the majority of by-catch is discarded damaged, dying, or dead (Aldous et al., 2013; Jenkins and Brand, 2001; Stewart and Howarth, 2016). In this chapter, the conclusion on the high selectivity of the fishery has been based on three elements: (1) the unanimous opinion of the fishermen on the cleanliness of this fishing gear, including those who did not belong to that fleet; (2) the follow-up on landings in port; (3) the port authorities verify that the fishing gear does nor discharge other species by-catch, which is punishable by an administrative sanction. With all this information, and due to the lack of official data or statistics, discards were not quantified. It is relevant to mention that variegated scallops

(*Chlamys varia*) are also caught, representing around 10% of the total catch, which is considered as a co-product.

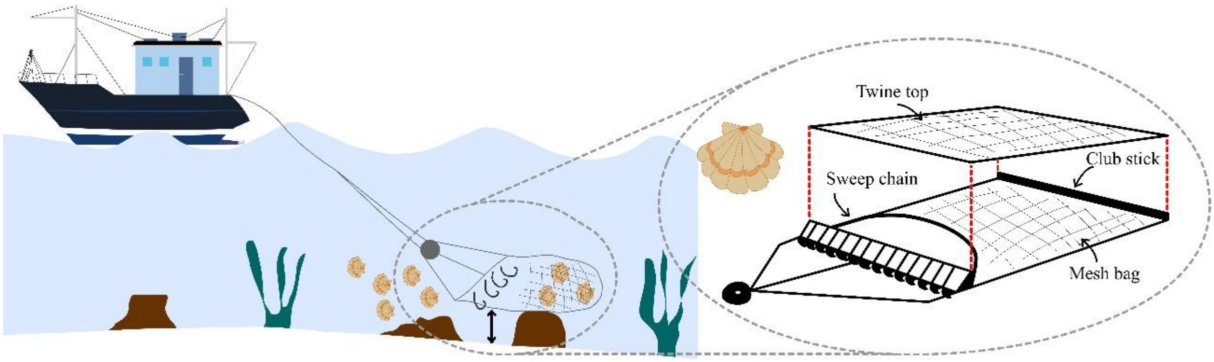


Figure 6.3. Simplified diagram of scallop fishery operations. On the right, the dredge used is shown in detail.

Subsystem 2 – Post-harvesting operations starts once the boats arrive at the port and the great scallops are taken to the processing plant of the Association of fishermen of the Port of Cambados, where they are kept for a full day in propylene drums with clean seawater for filtration. Once filtered, great scallops are taken to the evisceration area, where the workers in charge remove the hepatopancreas and soft tissues, maintaining the abductor muscle and gonads. Each scallop is then individually wrapped in plastic film, labelled and stored in cardboard boxes for year-round frozen distribution. It is important to note that scallops are vacuum packed in their original shell, since traditionally the scallop is baked in its shell.

The evisceration process is carried out by hand with a knife, highlighting the non-consumption of chemicals, additives or other elements, only the consumption of electricity and cleaning and protection materials for workers (knives, gloves, etc.) are noteworthy.

6.2.3. Data collection

Data acquisition is the most relevant step in an LCA study since the quality of the life cycle inventory data directly influences the quality and representativeness of the environmental results. In this chapter, a considerable effort was made to acquire data from primary sources to obtain reliable results. The data used for the Subsystem 1. Vessel operations were obtained from a set of 14 artisanal boats registered in the Port of Cambados, representing 67% of the 21 boats that make up the entire fleet in this town.

A series of questionnaires fulfilled by fishermen provided the primary information of the life cycle inventory. These questionnaires included the most relevant operational parameters necessary to carry out the environmental analysis, such as the distance to the fishing area, trips made per day and months dedicated to the maintenance of the boat, as well as the direct material consumption in the boat (diesel, anti-fouling, paint, lubricant oil, nets, etc.). An example of these questionnaires is included in Table B.1 of the Appendix

I. Different aspects directly related to the boat construction (weight and material of the boat, dimensions, lifetime, etc.) were also considered to build the life cycle inventory. Although this chapter used primary information to determine the consumption of materials related to fishing, it was necessary to use secondary data from scientific studies and the database for the background system. In this way, the Ecoinvent database v3.5 (Moreno Ruiz et al., 2018) was used as the main source of secondary data for the background system.

The questionnaires showed that the boats are sending to the docks for maintenance for 1-2 months per year, so paint and anti-fouling were considered important inputs in vessel operations as in previous research (Vázquez-Rowe et al., 2011; Villanueva-Rey et al., 2018). Data regarding the composition of the main paints and anti-fouling agents were taken from Vázquez-Rowe et al. (2010). Regarding the nets, as in Vázquez-Rowe et al. (2010), the composition based on nylon and Pb was taken into account, although the dimensions and weight provided by the questionnaires were considered. The annual consumption of nets was increased by 25% to take into account the potential replacement due to net losses at sea. This value was estimated as the maximum replacement ratio due to net losses during fishing according to the information provided by fishermen and net menders. Finally, the release of Pb into the sea due to net use was also estimated.

With respect to the boat construction, to establish the consumption of materials, the lifetime and the total weight of the boat were considered. In this way the "consumption" of the boat per year was estimated, using the Ecoinvent database to consider the necessary materials for the construction of an average small-size boat. These materials include wood (71%), steel (26%), plastics (2%), Pb (0.3%), other metals (0.3%), epoxy resin (0.02%) and other elements.

It is important to note that the time dedicated to the scallop campaign has been considered, as this type of small-scale vessel operates all year in different small-scale fisheries. In some cases, the questionnaires collected data directly related to the scallop fishery, but in other cases, the data obtained were related to annual consumption, so a temporal disambiguation was necessary. The direct gaseous emissions from diesel combustion were taken from Ecoinvent, considering the EEA (2013) emission factors. Finally, bilge water was also included within the system boundaries.

Data acquisition to develop the life cycle inventory of the post-harvesting operations (Subsystem 2) was obtained mainly through primary sources. The information was provided by the manager of the processing plant located in the Port of Cambados. The information included a wide set of operational and capital goods aspects related to the different stages described in Section 8.2.2, which included the main material and energy consumption of the plant. Secondary data taken from the Ecoinvent database was used for the background processes involved in the production of operational inputs such as electricity, plastics, or packaging material. The consumption of materials includes the months of plant activity (mainly from December 2019 to March 2020), while the electricity consumption refers to the whole year, since the scallops are stored throughout the year in the cold room before they are marketed.

6.2.4. Co-product allocation strategies

The recommendations of the ISO standards give priority to the division of the unit process into sub-processes or the extension of the system boundaries to include additional co-product functions as opposed to the application of allocation factors. The scallop fishery includes the capture of a small amount of variegated scallop, which is also highly valued by Galician gastronomy, with a good market niche, so the allocation of environmental loads between the two products is required. When accounting the total catch ratios, 90.0% mass allocation is considered for great scallop, while if the wholesale prices are considered, an economic allocation factor of 87.9% is achieved, as detailed in Table 6.1. Therefore, because of the small difference in whether one method or the other is used, mass allocation was considered the most appropriate approach for this case study. This selection was based on the fact that the use of mass allocation enables reducing the uncertainty caused by fish prices volatility (Vázquez-Rowe et al., 2011).

Table 6.1. Mass and economic allocation factors for scallop fishery

Species	Landings (kg)	Mass allocation	Value (€/kg)	Economic allocation
Pecten maximus	90,029	90.0%	5.47	87.9%
Chlamys varia	10,013	10.0%	6.79	12.1%

6.2.5. Life cycle impact assessment: methodology

A wide range of environmental indicators have been used in this chapter to establish the environmental impact of great scallop fishing and processing. In this sense, the life cycle impact assessment step was carried out using the ReCiPe 2016 v1.1 methodology in a hierarchist perspective at midpoint level (Huijbregts et al., 2016) in terms of the following impact categories: Global Warming (GW) and Stratospheric Ozone Depletion (SOD) to establish the impacts on the atmosphere and the ozone layer related to gaseous emissions; Freshwater Eutrophication (FE), Marine Eutrophication (ME), Freshwater Ecotoxicity (FET) and Marine Ecotoxicity (MET) to quantify the impacts on fresh and marine water since the Galician rías correspond to fluvial-marine transition ecosystems; and Fuel Resource Scarcity (FRS) to establish a link with fuel consumption in the boats, as it is proven as one main hotspots in fishing. SimaPro v9.0 (PRé Consultants, 2017) was the software used to conduct the computational implementation of the life cycle inventories.

6.2.6. Uncertainty analysis: Monte Carlo simulation

When managing multiple life cycle inventories, the common procedure is based on the definition of an average inventory data. The use of these average values involves the handling of standard deviations and, consequently, data quality problems. The different types of uncertainty include those related to the choice of scenarios (e.g., choice of functional unit or allocation methods), those related to the LCA model (e.g., uncertainties of characterisation factors), and uncertainty related to parameters (e.g., measurement inaccuracies or variability resulting from horizontal averaging) (Huijbregts, 2002). In this chapter, the focus has been mainly on data uncertainty due to variability caused by using an average life cycle inventory from several boats. To assess the uncertainty of the average inventory data, the Monte Carlo method was used. For simplicity, the normal distribution was assumed to be the probability distribution of the life cycle inventory, so it was necessary to characterise all the inputs data with their mean and standard deviation. The Monte Carlo analysis was performed using the Monte Carlo module of the SimaPro v9.0 software on background data (processes from the Ecoinvent database v3.5). The number of iterations was set to 1000 at a 95% significance level (Longo et al., 2017).

6.2.7. GHG emission/protein content correlation

In order to place the environmental and nutritional aspects of great scallops in the context of an average diet, the environmental performance of this product in terms of its carbon footprint and the protein content has been compared with that of other widely consumed food (seafood, meat, dairy products and fruits and vegetables).

The carbon footprint was chosen as environmental indicator because it is a widespread element that enjoys high consumer recognition (Laurent et al., 2012). It is important to point out that the ready-to-eat product was considered for the analysis, i.e., eviscerated great scallops, harvested fruit or seafood landed at the port and all products are considered in a cradle-to-gate approach, excluding the production of packaging, retail, transport and consumption stages. The carbon footprint values were reported for 1 kg of edible weight for all the assessed foodstuffs. For this purpose and taking into account the objective of this chapter, the values given per unit of live weight were translated into the edible yield using different species-factors for edible yield from different sources: (i) FAO (1989) for fish; (ii) Ruiz-Torralba et al (2018) for fruits; (iii) Clune et al (2017) for meat; and (iv) for cases where specific values were not available, generic data collected in Hartikainen et al (2018) were used. At this point, it is important to note that a 100% edible part was considered for dairy products.

To introduce nutritional quality, the protein content in grams per 100 g of edible product was considered, obtained from the Spanish Agency of Food Security and Nutrition (AESAN, 2018). Protein content was chosen as an indicator of nutritional quality since many nutrient density models indicate that protein should be encouraged. Furthermore, it is demonstrated that protein has the strongest positive correlation with the level of GHG emissions linked to the 19 main macronutrients (van Dooren et al., 2017).

6.3. RESULTS AND DISCUSSION

6.3.1. Quantitative analysis of inputs and outputs

The life cycle inventory of the fishing stage encompassed all the necessary elements for vessel operation, including an average fuel consumption of 123 mL or 772.7 mg of net per scallop. These values indicate that the nets, although they need constant repairs and renovations, represent a very low consumption throughout the year compared to the main elements such as diesel. Based on the average weight of each boat (7.9 tonnes) and the average lifespan (38 years), given that on average, boats dedicated 3 months per year on scallop fishery and taking into account the total scallop captures in the season and the mass allocation factors, the "consumption" of infrastructure was calculated as 1.84 g of vessel per scallop meanwhile consumption of 132.1, 337.1 and 891.5 mg of anti-fouling, paint, and lubricant oil, respectively, were calculated for maintenance operations.

As for the inventory of the processing subsystem, the low consumption of materials stands out. Only the consumption of plastic film is noteworthy, since each scallop is individually packed with 8.3 g of plastic film. It is also important to highlight the consumption of electricity, which was around 90,000 kWh in 2018 for the operation of the freezers where the scallops are stored. The life cycle inventories calculated for the two considered subsystems are summarised in Table 6.2.

Life cycle inventory analysis has shown that direct consumption of materials on the boat is a key element in understanding the environmental profile of the final frozen scallop. Fuel consumption is the most important component of the inventory, reaching 111.8 mL of diesel per scallop and a fuel use intensity (FUI) of 721.2 L/tonne. This value is higher than those reported by other authors, i.e. Kitts et al. (2008) and Tyedmers (2004) reported 364 and 350 L/tonne respectively. These values represent less than 50% of those obtained in this chapter, however, these values should be taken with caution, as they refer to fisheries in the late 1990s in North America. Another more recent study reported the FUI of general mollusc fisheries using dredges in North America at 295 L/tonne with a minimum value of 71 L/tonne and a maximum of 361 L/tonne (Parker and Tyedmers, 2015). This same study, however, reported the average for mollusc dredge fisheries in Europe at 525 L/tonne, which is much closer to that reported in this chapter. Finally, in Parker et al. (2018), which is a study on fuel consumption in different fisheries around the world, an average value of 523 L/tonne is reported for all types of demersal molluscs.

Table 6.2. Inventory data for the subsystems considered in this chapter per FU

Subsystem 1. Vessel operations					
Inputs from the Technosphere			**Outputs to the Ocean**		
Materials	**Unit**	**Value**	**Emissions**	**Unit**	**Value**
Diesel	mL	111.8	Lead	mg	37.5
Nets	mg	772.7	Xylene	mg	17.9
Anti-fouling	mg	132.1	Cobalt	µg	348
Boat Paint	mg	337.1	COD	g	1.69
Lubricant oil	mg	891.5	Copper	mg	28.7
Infrastructure	g	1.84	**Outputs to the Atmosphere**		
			Emissions	**Unit**	**Value**
			CO_2	g	355
			SO_2	g	3.3
			NMVOC	mg	361
			NO_X	g	7.7
			CO	mg	806
			Outputs to the Technosphere		
			Products	**Unit**	**Value**
			Pecten maximus	g	155
			Co-products	**Unit**	**Value**
			Chlamys varia	g	17.2
Subsystem 2. Post-harvesting operations					
Inputs from the Technosphere			**Outputs to the Technosphere**		
Materials	**Unit**	**Value**	**Products**	**Unit**	**Value**
Pecten maximus from SS1	g	155	Frozen scallop	g	139.5
Plastic film	g	10.7	Packaging	g	8.3
Corrugated board	g	2.6	**Waste to treatment**	**Unit**	**Value**
Chromium Steel	mg	2.9	Biowaste	g	15.5
Polypropylene	mg	13.6	Mixed plastics to landfill	mg	60.5
Polyethylene	mg	138.2	Steel to recycling	mg	2.9
Energy	**Unit**	**Value**	Polypropylene to recycling	mg	13.6
Electricity	MJ	0.56			

In general, the fuel consumption obtained in this chapter is higher than others reported in previous work on scallop or mollusc fisheries with dredges. This may be representative of the low performance of a fishery with very strict fishing quotas, which makes the combined fuel consumption during the vessel travel to the catch area and the fishing activities inefficient. However, maintaining these fishing quotas is essential to ensure the long-term sustainability of this fishery, so improvement actions should focus on reducing fishing effort (Farmery et al., 2014), i.e. improving fuel efficiency by targeting high-density scallop aggregations. For this purpose, technologies such as multi-beam echosounders or video survey techniques have proved to be a fast and accurate way to map the location of scallop beds (Duncan et al., 2016).

6.3.2. Environmental characterisation of great scallop fishing and processing

The relative distribution of the environmental impacts in the processing stage is shown in Figure 6.4. The final results per FU for the different allocation approaches (mass and economic); as well as a complete breakdown of the results, including the relative contribution to the impact of each item of the life cycle inventory in each impact category can be found in Table B.2 of the Appendix I.

Looking at the full set of environmental results, the fishing stage can be designed as the most burdensome subsystem, as it accounted for most of the impact in MET (98.5%), GW (83.5%), FRS (82.0%), SOD (72.9%), and ME (52.0%). It should be noted that, in the categories related to freshwater, the production of electricity for the operation of the evisceration plant is the main contributor to environmental impact. This is due to the Spanish electricity mix in the Ecoinvent database, which shows a significant percentage of electricity production from coal. The treatment of waste from coal and lignite mining is responsible for the high impact in these impact categories.

Plastic film production presented a considerably uniform distribution of environmental impacts across almost all categories, with contributions between 5% and 10%, except for SOD and MET categories, where it showed no relevant impact (<2%). Corrugated board production achieved a significant impact on ME (ca. 10%), although in the other categories it was not very relevant, reaching even less than 0.5% in GW, MET and FRS. The treatment of the waste generated during the processing is not relevant, except in the SOD category (ca. 5%) and, finally, the influence of other consumables on the environmental profile is irrelevant, below 0.2% in all impact categories. The consumption of electricity in the plant is the only element of the post-harvesting operations subsystem that is relevant to the environmental profile. In fact, this makes sense since great scallops must be stored in freezers for their distribution during the rest of the year. This fact leaves open the option of future improvement actions leading to lower electricity consumption or the search for cleaner production systems to ensure an even lower impact of electricity consumption on the environmental profile.

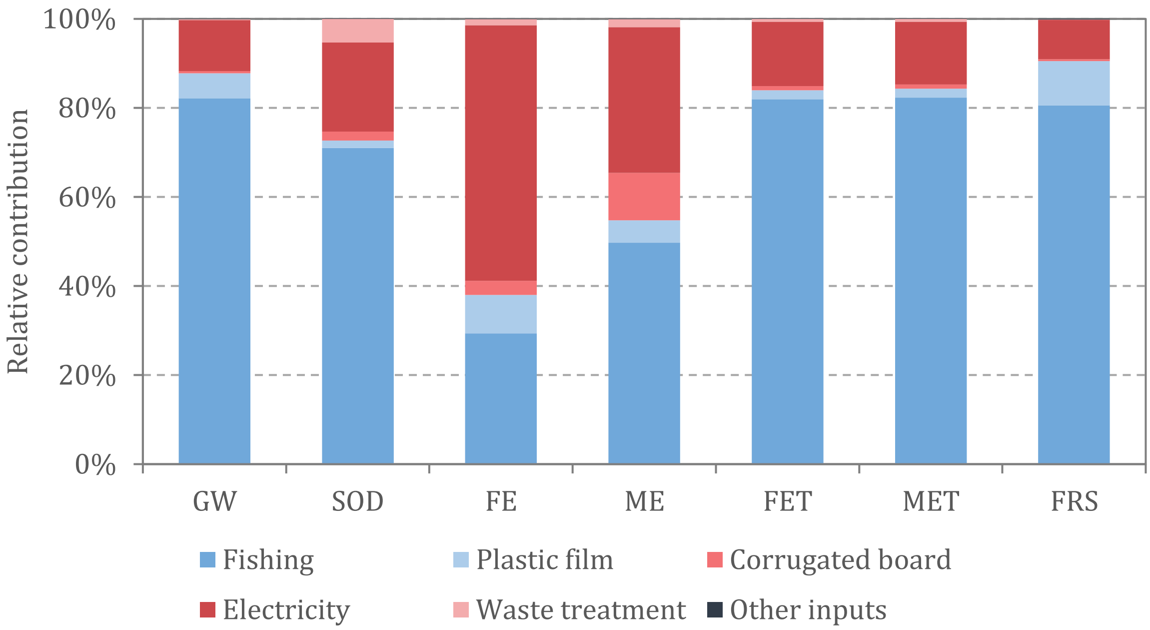


Figure 6.4. Relative contribution of environmental impacts per process involved in the fishing and processing of scallops.

In order to highlight the process with the greatest environmental impact in the life cycle of the system, the individual contributions of the fishing stage were broken down in Figure 6.5. According to the results, there are three main activities that produce most of the environmental impacts. In the first place, diesel production and combustion accounted for most of the impact in GW (97.7%), SOD (92.2%), FRS (98.0%) and FE (55.5%). The great influence of this element in the profile can be explained from the perspective that diesel production presented a high impact on fossil resource scarcity category, while the emission of GHG and other gaseous pollutants to the atmosphere during diesel combustion is behind the high values in GW and SOD, respectively. In the other categories, diesel relative contribution is reduced by the presence of other elements more relevant.

The production and consumption of anti-fouling presented a relevant impact on the ecotoxicity categories (98.9% and 40.5% in MET and FET categories, respectively). The high impact of this element is mainly due to the emission of Cu- and Sn-based emissions into the sea during vessel operation. As for the ME category, the element with the greatest environmental impact is the manufacture and use of nets, which represents 54.7% of the impact in this category, while in the others it is always less than 6%. Regarding vessel construction, as in previous research (Hospido and Tyedmers, 2005; Laso et al., 2018; Vázquez-Rowe et al., 2011), it has been shown not to have a major relevance on the environmental profile due to the long lifetime of this type of vessel. The environmental impact of boat construction is only noteworthy in the categories of FET (19.3%), FE (10.1%) and ME (10.0%) due to the treatment of the waste generated during the production of the raw materials. The element "others", which includes the production and use of boat paint and lubricant oil and the treatment of bilge water, presented a relatively constant contribution around 1-2% in all impact categories.

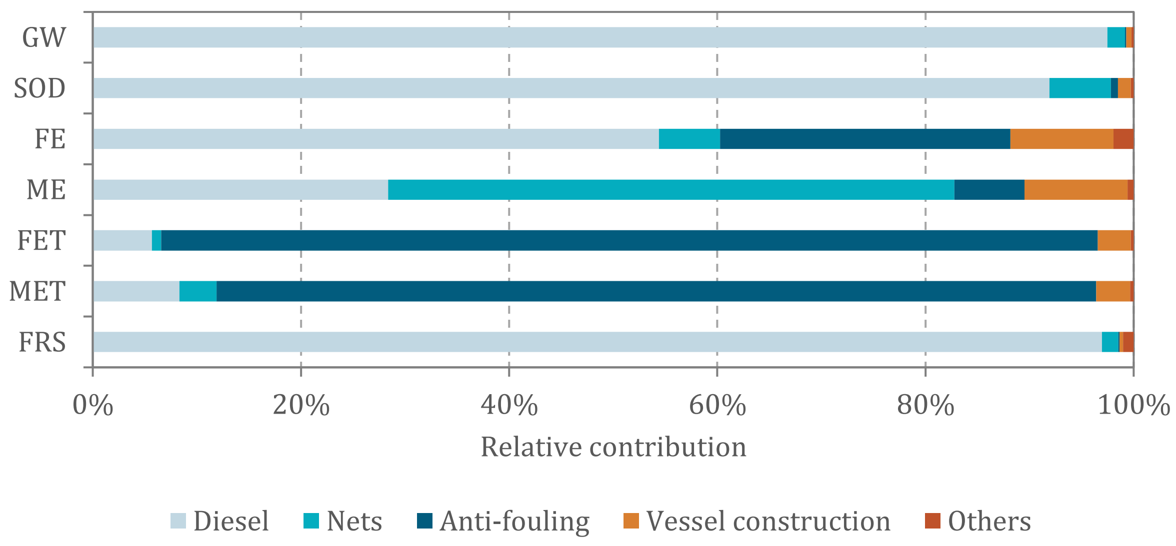


Figure 6.5. Relative contributions (in %) by component in the environmental profile of scallop fishing.

The production and consumption of anti-fouling presented a relevant impact on the ecotoxicity categories (98.9% and 40.5% in MET and FET categories, respectively). The high impact of this element is mainly due to the emission of Cu- and Sn-based emissions into the sea during vessel operation. As for the ME category, the element with the greatest environmental impact is the manufacture and use of nets, which represents 54.7% of the impact in this category, while in the others it is always less than 6%. Regarding vessel construction, as in previous research (Hospido and Tyedmers, 2005; Laso et al., 2018; Vázquez-Rowe et al., 2011), it has been shown not to have a major relevance on the environmental profile due to the long lifetime of this type of vessel. The environmental impact of boat construction is only noteworthy in the categories of FET (19.3%), FE (10.1%) and ME (10.0%) due to the treatment of the waste generated during the production of the raw materials. The element "others", which includes the production and use of boat paint and lubricant oil and the treatment of bilge water, presented a relatively constant contribution around 1-2% in all impact categories.

6.3.3. Uncertainty analysis

As mentioned above, the LCI for the fishing stage was constructed using average data from 14 fishing boats and the background data was taken from the Ecoinvent database. An uncertainty analysis was performed to assess the extent to which the uncertainties of the background data and the deviation from the primary data can influence the environmental results.

The mean values of the impact categories have been represented in a bar chart and the uncertainty margins express the 95% confidence interval (Fantin et al., 2015). Figure 6.6 shows the impact assessment profile per FU with 95% confidence interval.

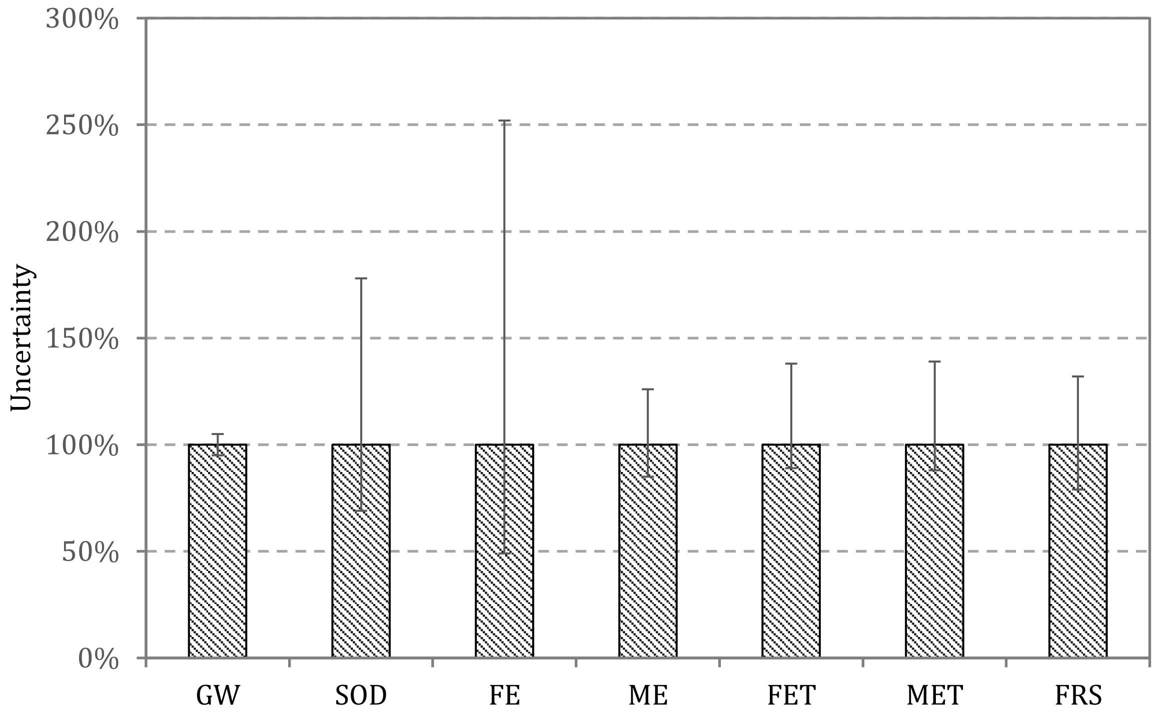


Figure 6.6. Bar-chart of Monte Carlo simulation results for each impact category per FU. The error bars represent the 95% confidence interval. Legend: GW-Global Warming, SOD-Stratospheric Ozone Depletion, FE-Freshwater Eutrophication, ME-Marine Eutrophication, FET-Freshwater Ecotoxicity, MET-marine Ecotoxicity, FRS-Fossil Resources Scarcity.

According to these results, FE and FET showed the highest data variability, while GW and ME showed the lowest. This uncertainty probably comes mainly from the uncertainty values in some Ecoinvent background data that were propagated to the final results, since the variability of the data handled from fishing boats is quite low as they have similar sizes and characteristics. These results are in line with other uncertainty analyses performed on life cycle inventories based on Ecoinvent data (Fantin et al., 2015; Lijó et al., 2017; Longo et al., 2017), which showed that the highest uncertainty was associated with the categories of Freshwater Eutrophication and Ozone Depletion categories while the lowest uncertainty was associated with Global Warming, Acidification and Marine Eutrophication.

6.3.4. GHG emission and protein content in different foodstuffs

Figure 6.7 shows the comparative analysis between the protein content and Carbon Footprint in terms of GHG emissions of different food products. The results were represented according to the average value obtained for the sample, so that the elements with better or worse results can be easily identified. It is important to note that the size of the bubble represents the consumption in Spain in 2018, obtained from the Ministry of Agriculture, Fisheries and Food (MAPA, 2020b).

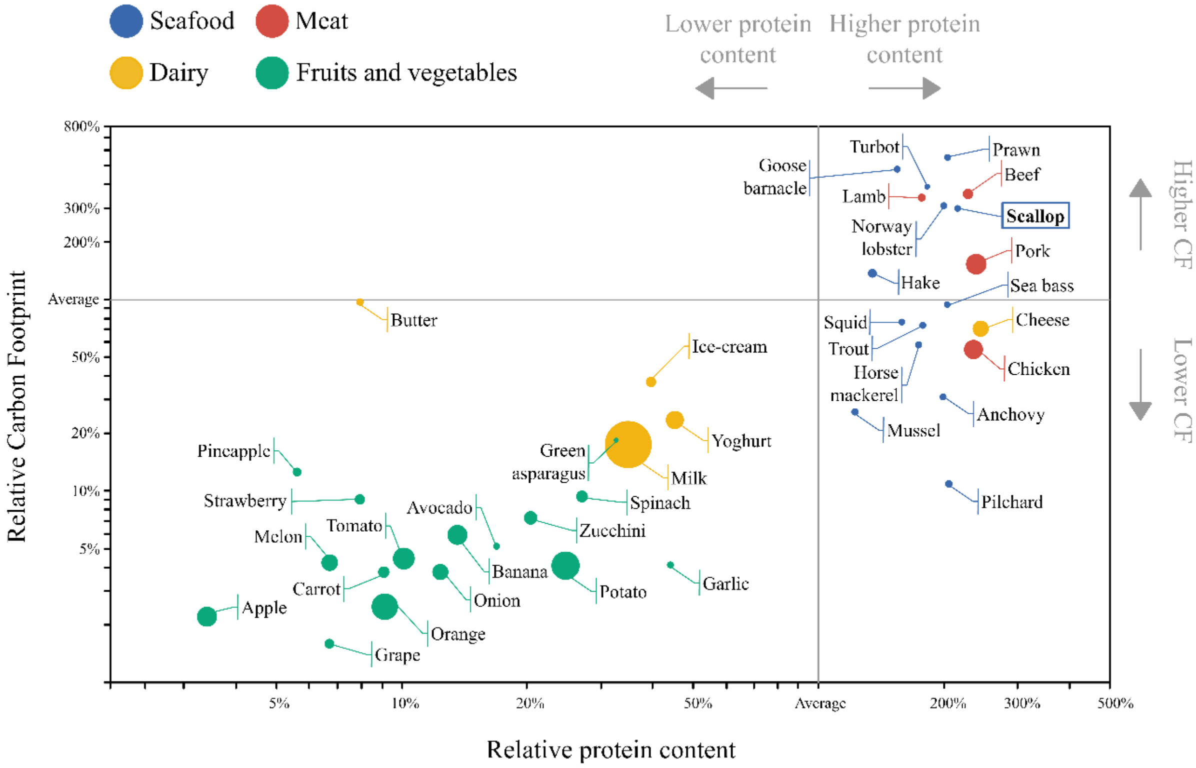


Figure 6.7. Protein content and carbon footprint of different foodstuffs. Log transformed data scaled around average of all the products analysed. The colour of the bubbles represents the different groups. The size of the bubble reflects the consumption in 2018.

According to the obtained results, the analysed food categories can be classified into 3 groups: 1) fruits and vegetables and most dairy products are located in the low-protein and low-CF sector. 2) meat products are placed in the high-protein and high-GHG sector, which makes sense given the high environmental costs linked to meat products. Although chicken is the exception, as it is below the average carbon footprint. 3) seafood is entirely located in the high protein sector but is almost equally divided between high and low emissions. The seafood species with high emissions are mainly molluscs and crustaceans (with low catch ratios) and turbot (farmed in aquaculture facilities), while the low-emissions species include finfish with high catch ratios by purse seine and similar fishing gears (horse mackerel, anchovy, pilchard…) but also mussels farmed in rafts with a very low impact and scallops.

In general, the results obtained match the expected correlation between protein content and the associated GHG emissions (González et al., 2011; van Dooren et al., 2017). Great scallops are in the quadrant of high protein content (the highest of the fisheries and just below meat), but also high environmental impact in terms of carbon footprint. It is important to consider the edible content of great scallop, which is particularly low around

13.2% compared to finfish species, where edible muscle usually constitutes 50-60% of the total weight (Tyedmers, 2004). Therefore, taking these data, the results obtained are in line with those reported by Hallström et al. (2019), where it is reported that crustaceans, flatfishes, scallops and oysters had the highest climate impact among the different fisheries due to a combination of resource-intensive production and/or low edible yield. The Galician great scallop fishery assessed in this chapter has both circumstances: high FUI and low edible yield.

6.4. CONCLUSIONS

In this chapter, an important part of the great scallop trawling fleet from the Port of Cambados has been inventoried, representing 77.5% of the total Galician landings in 2018. This chapter is the first comprehensive life cycle assessment performed on the Galician great scallop fishing fleet, the main results showed that the critical points of the process are the fishing stage and the consumption of electricity in the processing facilities. More specifically, diesel consumption in fishing boats stands out as the critical point.

It has been shown that, in the combination of environmental and nutritional aspects, great scallop presented one of the best profiles within the category of seafood. The protein content of the scallop is one of the highest in the category of seafood, at the level of some meats such as beef or chicken, while the environmental profile in terms of carbon footprint is, as expected given its low edible yield, on a par with other molluscs and crustaceans such as goose barnacle, prawn and Norway lobster.

This work represents an important step forward in the search for sustainability of the Galician fishing sector, which has a great influence on the productive fabric of this region. This chapter has shed light on the determination of material and energy flows of the fishing and processing of great scallop, filling the existing gaps in the inventory data of this species. Finally, future perspectives on the environmental assessment of different scallop fisheries in Europe should aim at providing the environmental burdens of a wide range of fleets, for which this chapter can be used as the first iteration for following studies in the coming years.

6.5. REFERENCES

AESAN, 2018. Spanish Food Composition Database (BEDCA). URL: https://www.bedca.net/ (accessed 9.3.20).

Aldous, D., Brand, A.R., Hall-Spencer, J.M., 2013. MSC Public Certification Report (PCR) for USA Sea Scallop FIshery. Intertek Moody Marine LTD.

Boyle, K., Thompson, S., 2012. Bycatch Survey – Isle of Man Queen Scallop Otter Trawl Fishery Summer 2012. Bangor University Fisheries and Conservation Report No. 19.

Brand, Andrew R, 2006. Chapter 12. Scallop ecology: Distributions and behaviour, in: Shumway, S.E., Parsons, G.J.B.T. (Eds.), Scallops: Biology, Ecology and Aquaculture. Elsevier, pp. 651–744. https://doi.org/10.1016/S0167-9309(06)80039-6

Brand, Andrew R., 2006. Chapter 19. The European scallop fisheries for *Pecten maximus*, *Aequipecten opercularis* and *Mimachlamys varia*, in: Shumway, S.E., Parsons, G.J.B.T. (Eds.), Scallops: Biology, Ecology and Aquaculture. Elsevier, pp. 991–1058. https://doi.org/10.1016/S0167-9309(06)80046-3

Clune, S., Crossin, E., Verghese, K., 2017. Systematic review of greenhouse gas emissions for different fresh food categories. J. Clean. Prod. 140, 766–783. https://doi.org/10.1016/j.jclepro.2016.04.082

Duncan, P.F., 2009. An assessment of bycatch in the Isle of Man queen scallop trawl fishery. Report for the Isle of Man Government Department of Agriculture, Fisheries and Forestry, p. 54.

Duncan, P.F., Brand, A.R., Strand, Ø., Foucher, E., 2016. The European scallop fisheries for *Pecten maximus*, *Aequipecten opercularis*, *Chlamys islandica*, and *Mimachlamys varia*. Dev. Aquac. Fish. Sci. 40, 781–858. https://doi.org/10.1016/B978-0-444-62710-0.00019-5

EEA, 2013. EMEP/EEA air pollutant emission inventory guidebook 2013. Technical guidance to prepare national emission inventories. https://doi.org/10.2800/92722

European Commission, 2020. Facts and figures on the Common Fisheries Policy - 2020 Edition.

Fantin, V., Giuliano, A., Manfredi, M., Ottaviano, G., Stefanova, M., Masoni, P., 2015. Environmental assessment of electricity generation from an Italian anaerobic digestion plant. Biomass and Bioenergy 83, 422–435. https://doi.org/10.1016/j.biombioe.2015.10.015

FAO, 2020. The State of World Fisheries and Aquaculture - Sustainability in action. Food and Agriculture Organization of the United Nations, Rome.

FAO, 1989. Yield and nutritional value of the commercially more important fish species. Food & Agriculture Organization of the United Nations (FAO), Rome.

Farmery, A., Gardner, C., Green, B.S., Jennings, S., 2014. Managing fisheries for environmental performance: The effects of marine resource decision-making on the footprint of seafood. J. Clean. Prod. 64, 368–376. https://doi.org/10.1016/j.jclepro.2013.10.016

González, A.D., Frostell, B., Carlsson-Kanyama, A., 2011. Protein efficiency per unit energy and per unit greenhouse gas emissions: Potential contribution of diet choices to climate change mitigation. Food Policy 36, 562–570. https://doi.org/10.1016/j.foodpol.2011.07.003

Guinée, J.B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Sleeswijk, A.W., Suh, S., de Haes, H., 2001. Life cycle assessment. An operational guide to the ISO standards. Leiden (The Netherlands): Centre of Environmental Science.

Hallström, E., Bergman, K., Mifflin, K., Parker, R., Tyedmers, P., Troell, M., Ziegler, F., 2019. Combined climate and nutritional performance of seafoods. J. Clean. Prod. 230, 402–411. https://doi.org/10.1016/j.jclepro.2019.04.229

Hartikainen, H., Mogensen, L., Svanes, E., Franke, U., 2018. Food waste quantification in primary production – The Nordic countries as a case study. Waste Manag. 71, 502–511. https://doi.org/10.1016/j.wasman.2017.10.026

Hinz, H., Murray, L.G., Malcolm, F.R., Kaiser, M.J., 2012. The environmental impacts of three different queen scallop (*Aequipecten opercularis*) fishing gears. Mar. Environ. Res. 73, 85–95. https://doi.org/10.1016/j.marenvres.2011.11.009

Hospido, A., Tyedmers, P., 2005. Life cycle environmental impacts of Spanish tuna fisheries. Fish. Res. 76, 174–186. https://doi.org/10.1016/j.fishres.2005.05.016

Huijbregts, M., 2002. Uncertainty and variability in environmental life-cycle assessment. Int. J. Life Cycle Assess. 7, 173. https://doi.org/10.1007/BF02994052

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M.D.M., Hollander, A., Zijp, M., van Zelm, R., 2016. ReCiPe 2016 v1.1. A harmonized life cycle impact assessment method at midpoint and endpoint level. The Netherlands.

ISO, 2006a. ISO 14040 - Environmental Management - Life Cycle Assessment - Principles and Framework.

ISO, 2006b. ISO 14044 - Environmental Management - Life Cycle Assessment - Requirements and Guidelines.

Jenkins, S.R., Brand, A.R., 2001. The effect of dredge capture on the escape response of the great scallop, *Pecten maximus (L.)*: Implications for the survival of undersized discards. J. Exp. Mar. Bio. Ecol. 266, 33–50. https://doi.org/10.1016/S0022-0981(01)00345-8

Kitts, A., Schneider, G., Lent, R., 2008. Carbon Footprint of Commercial Fishing In the Northeast United States 1–12.

Laso, J., Vázquez-Rowe, I., Margallo, M., Crujeiras, R.M., Irabien, Á., Aldaco, R., 2018. Life cycle assessment of European anchovy (*Engraulis encrasicolus*) landed by purse seine vessels in northern Spain. Int. J. Life Cycle Assess. 23, 1107–1125. https://doi.org/10.1007/s11367-017-1318-7

Laurent, A., Olsen, S.I., Hauschild, M.Z., 2012. Limitations of carbon footprint as indicator of environmental sustainability. Environ. Sci. Technol. 46, 4100–4108. https://doi.org/10.1021/es204163f

Lijó, L., González-García, S., Bacenetti, J., Moreira, M.T., 2017. The environmental effect of substituting energy crops for food waste as feedstock for biogas production. Energy 137, 1130–1143. https://doi.org/10.1016/j.energy.2017.04.137

Longo, S., Frison, N., Renzi, D., Fatone, F., Hospido, A., 2017. Is SCENA a good approach for side-stream integrated treatment from an environmental and economic point of view? Water Res. 125, 478–489. https://doi.org/10.1016/j.watres.2017.09.006

MAPA, 2021. Fisheries statistics: Products of the fish processing industry. URL: https://www.mapa.gob.es/es/estadistica/temas/estadisticas-pesqueras/industrias-procesado-pescado/productos-procesado-pescado/ (accessed 1.22.21).

MAPA, 2020a. Fisheries statistics: Marine catch and landings statistics. URL: https://www.mapa.gob.es/es/estadistica/temas/estadisticas-pesqueras/pesca-maritima/estadistica-capturas-desembarcos/ (accessed 9.4.20).

MAPA, 2020b. Household consumption database. URL: https://www.mapa.gob.es/app/consumo-en-hogares/ (accessed 9.2.20).

Moreno Ruiz, E., Valsasina, L., Brunner, F., Symeonidis, A., FitzGerald, D., Treyer, K., Bourgault, G., Wernet, G., 2018. Documentation of changes implemented in the Ecoinvent Database v3.5. Ecoinvent, Zurich, Switzerland.

Outeiro, L., Rodríguez-Mendoza, R., Bañón, R., Alonso-Fernández, A., 2020. Influence of aquaculture on fishing strategies: Insights from Galician small-scale fisheries. Aquaculture 521. https://doi.org/10.1016/j.aquaculture.2020.735043

Parker, R.W.R., Blanchard, J.L., Gardner, C., Green, B.S., Hartmann, K., Tyedmers, P.H., Watson, R.A., 2018. Fuel use and greenhouse gas emissions of world fisheries. Nat. Clim. Chang. 8, 333–337. https://doi.org/10.1038/s41558-018-0117-x

Parker, R.W.R., Tyedmers, P.H., 2015. Fuel consumption of global fishing fleets: Current understanding and knowledge gaps. Fish Fish. 16, 684–696. https://doi.org/10.1111/faf.12087

Picado, A., Lorenzo, M.N., Alvarez, I., deCastro, M., Vaz, N., Dias, J.M., 2016. Upwelling and Chl-a spatiotemporal variability along the Galician coast: dependence on circulation weather types. Int. J. Climatol. 36, 3280–3296. https://doi.org/10.1002/joc.4555

PRé Consultants, 2017. SimaPro Database Manual (No. Methods Library). The Netherlands.

Ruiz-Salmón, I., Laso, J., Margallo, M., Villanueva-Rey, P., Rodríguez, E., Quinteiro, P., Dias, A.C., Almeida, C., Nunes, M.L., Marques, A., Cortés, A., Moreira, M.T., Feijoo, G., Loubet, P., Sonnemann, G., Morse, A.P., Cooney, R., Clifford, E., Regueiro, L., Méndez, D., Anglada, C., Noirot, C., Rowan, N., Vázquez-Rowe, I., Aldaco, R., 2021. Life cycle assessment of fish and seafood processed products – A review of methodologies and new challenges. Sci. Total Environ. 761. https://doi.org/10.1016/j.scitotenv.2020.144094

Ruiz-Torralba, A., Guerra-Hernández, E.J., García-Villanova, B., 2018. Antioxidant capacity, polyphenol content and contribution to dietary intake of 52 fruits sold in spain. CYTA - J. Food 16, 1131–1138. https://doi.org/10.1080/19476337.2018.1517828

Stewart, B.D., Howarth, L.M., 2016. Quantifying and Managing the Ecosystem Effects of Scallop Dredge Fisheries, Developments in Aquaculture and Fisheries Science. Elsevier B.V. https://doi.org/10.1016/B978-0-444-62710-0.00018-3

Tyedmers, P., 2004. Fisheries and Energy Use. Encycl. Energy 2, 683–693.

https://doi.org/10.1016/b0-12-176480-x/00204-7

van Dooren, C., Douma, A., Aiking, H., Vellinga, P., 2017. Proposing a Novel Index Reflecting Both Climate Impact and Nutritional Impact of Food Products. Ecol. Econ. 131, 389–398. https://doi.org/10.1016/j.ecolecon.2016.08.029

Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2011. Life Cycle Assessment of fresh hake fillets captured by the Galician fleet in the Northern Stock. Fish. Res. 110, 128–135. https://doi.org/10.1016/j.fishres.2011.03.022

Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2010. Life cycle assessment of horse mackerel fisheries in Galicia (NW Spain): Comparative analysis of two major fishing methods. Fish. Res. 106, 517–527. https://doi.org/10.1016/j.fishres.2010.09.027

Villanueva-Rey, P., Vázquez-Rowe, I., Arias, A., Moreira, M.T., Feijoo, G., 2018. The importance of using life cycle assessment in policy support to determine the sustainability of fishing fleets: a case study for the small-scale xeito fishery in Galicia, Spain. Int. J. Life Cycle Assess. 23, 1091–1106. https://doi.org/10.1007/s11367-017-1402-z

Xunta de Galicia, 2020. Plataforma Tecnolóxica da Pesca. Informes estadísticos. Xunta Galicia. Cons. do Mar. URL: https://www.pescadegalicia.gal/estadisticas/ (accessed 9.15.20).

USC
UNIVERSIDADE
DE SANTIAGO
DE COMPOSTELA

Chapter 7

Multi-product strategy to enhance the environmental profile of the canning industry towards circular economy

Summary

Spain, and in particular Galicia, which is an eminently fishing region characterised by the consumption of large quantities of fish, both fresh and processed, must face the challenge of shifting its seafood productive fabric towards a circular economy. To achieve this objective, the first task is to demonstrate that circular economy principles allow to reduce the environmental impacts associated with seafood production. In this sense, this chapter proposes the environmental evaluation of the skipjack tuna (*Katsuwonus pelamis*) value chain within a canning industry located in Galicia through the LCA methodology from an attributional perspective, including the valorisation processes for biowaste (edible and inedible by-products). Results indicate that the main crucial subsystems of the value chain are tuna fishing and the canning process, as it was expected considering other similar studies on seafood products. Moreover, this specific case study demonstrates that the multi-product strategy applied to the canning sector is environmentally viable. Thus, although the environmental impacts of the entire system are increased by including further valorisation operations, the environmental loads assigned to the main product (canned tuna) decrease compared to the one-product system by assigning environmental burdens to other value-added products.

This chapter, under the consent of the publisher, was redrafted after: Cortés, A., Esteve-Llorens, X., González-García, S., Moreira, M. T., Feijoo, G. (2021). Multi-product strategy to enhance the environmental profile of the canning industry towards circular economy. Science of the Total Environment, 791, 148249. DOI: https://doi.org/10.1016/j.scitotenv.2021.148249. More information related to this article can be found in Appendix III.

OUTLINE

7.1. INTRODUCTION

The situation in the oceans is agonizing, with fish stocks being decimated over the years worldwide (Wilson et al., 2020). The state of the oceans is becoming extremely worrying over time. In 2017, the maximum peak of overfished marine stocks (34.2%) and a minimum of underfished stocks (6.2%) was reached, according to the results published in FAO (2020). In parallel to the increasing rise of overfishing, aquaculture continues to grow steadily, to the point that today fish produced in aquaculture facilities account for 46% of total fish production (FAO, 2020). At this point, an intense debate has started to emerge regarding the long-term sustainability of wild fisheries or whether aquaculture should be chosen as the main fish source (Ruiz-Salmón et al., 2021). The valorisation of waste and discard fractions for the production of fishmeal and fish oil to be used for the formulation of feed for farmed fish also needs to be considered (Fréon et al., 2014b). There is growing evidence that the approach to utilize such fractions: fish bones, viscera, heads and other less desirable parts as raw material for the production of value-added products such as omega-3 acids and collagen, although these alternatives are at a less developed stage for industrial implementation (Laso et al., 2018).

In Spain, a country that has traditionally been an important fishing nation from the point of view of catching, processing and consumption (Vázquez-Rowe et al., 2014), 922,564 tons of fish and seafood were landed in 2018, making Spain the first country in the European Union, both in terms of volume and value, with almost 2,150 million euros (European Commission, 2020a). Spain has also developed an important seafood processing sector, especially smoked, processed and, above all, canned seafood. Domestic canned tuna production leads EU production, accounting for approximately 70% of the total volume (García-del-Hoyo et al., 2017). Specifically, the volume of canned seafood production by Spanish companies reached 353,000 tons in 2018, being Galicia (NW Spain) the leading region at national level, accounting for more than 85% of Spanish production (EUMOFA, 2019).

Galicia´s canning tradition means that it is home to 7 of the 10 largest companies on the Spanish canning sector, including the Top-5 (Ardán, 2018). Moreover, the presence of small and medium-sized enterprises (SMEs) is predominant in the Galician canning sector, with a high percentage of small companies (<50 employees), which represent 66% of the total, even highlighting that 22% of the total number of companies are very small with less than 10 employees (Ardán, 2018). Numerous initiatives and projects for the development of circular economy strategies are already underway in different companies. However, due to the aforementioned characteristics of the Galician canning industry, it is difficult for these initiatives to permeate the market and in the present context the Galician canning sector must face the challenge of the current paradigm shift from a linear economy to a circular economy in which the main objective of companies must be to maximize production through the valorisation of waste (Ciccullo et al., 2021). In this sense, among the actions to be developed to achieve a complete integration of the circular economy within the canning sector, the following stand out: (i) the use of processing techniques with a low environmental impact; (ii) the reduction of the packaging residues; (iii) the valorisation of wastewater flows; and (iv) the accomplishment of the objective of zero biological waste, valorising all biowaste fractions to produce new value-added products. With this in mind, the canning industry in general, and the Galician canning industry in particular, has enormous room for improvement, since a large part of the fish is directly discarded (heads, viscera, bones, etc.), which can open the door to the development of new products (García-Santiago et al., 2020). Circular economy emerges as an opposite solution to the current linear system as a sustainable system where economic growth is decoupled from resources use, through the reduction in the consumption and the recirculation of raw materials (Korhonen et al., 2018). On the main points of the circular economy is the reduction of waste generated throughout the value chain, valorising them as raw materials for the generation of added-value products. However, increasing circularity does not necessarily translate into a direct reduction of environmental impacts (Niero and Kalbar, 2019), which creates a dilemma for decision-makers when selecting adequate circular practices and innovations (Rufí-Salís et al., 2021). The impacts or benefits generated by these circular strategies are often measured through the use of circularity metrics (Corona et al., 2019).

This chapter proposes the environmental evaluation of the skipjack tuna (*Katsuwonus pelamis*) value chain within a canning industry located in Galicia through the LCA methodology. The production line focuses on gourmet products, with high added value, basing its production on local raw materials, with traditional manufacturing methods and using, as far as possible, certified organic ingredients. Thus, the processing plant meets some of the circular economy principles: (i) the fish is caught with traditional techniques in national fishing grounds; (ii) traditional techniques such as cooking in seawater and air-drying are followed; (iii) the primary packaging is made of aluminium, so it is 100% recyclable; (iv) high quality by-products (non-canned edible parts) are used to produce other products; (v) the low-quality by-products are valorised in the form of fishmeal that could be used for animal feed. In this way, the canning plant minimises the consumption of raw materials, minimises transport and follows a multi-product strategy. The main objectives of this chapter are to determine the environmental viability of this approach and to lay the foundations for the way forward for other companies to position themselves in a highly competitive market. The main novelty of this study lies in the fact that the LCA methodology has been used to analyse the environmental impacts of the entire canned tuna value chain, and not only those impacts assigned to the production of the main product. The inclusion of the valorisation processes of residual organic fractions within the system boundaries makes it possible to analyse the product from a broader point of view and opens the door to the identification and evaluation of opportunities for environmental improvement.

7.2. MATERIALS AND METHODS

7.2.1. Defining the goal and scope. Impact assessment methodology

Moving towards a veritable circular economy requires taking small steps to demonstrate the viability of multi-product processes from an environmental point of view. In this sense, this chapter aims to assess the environmental sustainability of the entire canned tuna value chain following the LCA methodology (ISO 14040; 14044) from a attributional perspective. Although the main product is canned tuna, all stages of the value chain were included within the system boundaries, including the manufacture of by-products and the valorisation of organic waste. Thus, the main objective of this chapter is to determine from an environmental point of view whether the production of multiple value-added products is more sustainable than single-product approaches.

A cradle-to-gate approach was considered in the study, that is, considering the extraction of raw materials to produce the required inputs and the manufacture of the products, but not the consumption and final disposal stages. This perspective was assumed since the main objective of the study is to recognise the environmental implications of the production of tuna-based products. The main raw material is skipjack tuna, so fishing and transport to the canning plant, as well as the production and transport of other ingredients and packaging materials, were included in the system boundaries. The Functional Unit considered for assessment was 1 tonne of raw tuna at processing plant gate since it seems consistent to select a feedstock-based FU as the plant is characterised by its multi-product nature. The software SimaPro 9.0 (PRe-Consultants, 2017) was used for the computational implementation of the inventories. The life cycle impact assessment step was carried out using the ReCiPe 2016 v1.1 methodology in a hierarchist perspective at midpoint level (Huijbregts et al., 2017). The environmental burdens were calculated in terms of the following impact categories: Global Warming (GW), Stratospheric Ozone Depletion (SOD), Terrestrial Acidification (TA), Freshwater Eutrophication (FE), Marine Eutrophication (ME), Freshwater Ecotoxicity (FET), Marine Ecotoxicity (MET), Mineral Resources Scarcity (MRS) and Fossil Resources Scarcity (FRS).

7.2.2. Description of the system under study

The value chain associated with canned tuna was divided in 4 different subsystems, as depicted in Figure 7.1. Subsystem 1 is related to the fishing and transportation of tuna as the main raw material to supply the canning plant. Subsystems 2-4 are linked to the different activities and operations that take place within the canning factory. It is important to note that the aim is to use the residual fractions of the process to produce value-added products; however, the treatment of non-recoverable waste and wastewater has been included in all subsystems. All liquid fractions are directly sent to a municipal wastewater treatment plant located close to the site. On the other hand, packaging waste is recycled as far as possible and landfilled or incinerated according to the Spanish profile.

7.2.2.1. Tuna fishery (SS1)

This subsystem includes all operations related to tuna fishery in FAO 34 waters. Tuna fishing is carried out by small vessels belonging to local coastal communities, using traditional and highly selective methods. Thus, the pole-and-line method is used, a method recommended by organisations and research to limit catches, avoid overfishing and minimise discards (Khan et al., 2018). Sardine (*Sardina pilchardus*) is used as bait in this fishery, so sardine fishing by an average purse seine fleet, which is landed in port and processed for bait production was included within the system boundaries of this subsystem as detailed in Vázquez-Rowe et al. (2011). Tuna, once caught, is discharged in the port of Arrecife (Lanzarote-Canary Islands) and transferred by ship to the port of Algeciras (Cadiz-Andalusia), and then transported by road to the canning plant located in Galicia.

7.2.2.2. Canning process (SS2)

This subsystem includes all the operations carried out in the canning plant that are directly related to the main product (canned tuna), from the extraction and production of the raw materials and fuels and the transport of these from their place of origin to the processing plant. This subsystem is composed by the different operations that are performed in the canning plant: the tuna pieces are unloaded and stored in a cooling chamber at reception until they are transported to the processing stage. The rest of the products and ingredients are stored in conventional areas until they are needed. The tuna is then cooked whole and not defrosted. Once cooked, the tuna is cooled in the plant overnight. The next morning, the tuna is cleaned, and the loins are obtained for canning and other by-products for use in other production lines or for valorisation. In this step, it is important to note that for every 1,000 kg of tuna that enters the plant, only 365 kg are canned.

The other parts are sent to valorisation processes, 8 kg of edible by-products are sent to tuna pâté production, while the remaining 627 kg of inedible by-products are used for the production of fishmeal and fish oil. Every 18 g can is filled with 90 g of tuna loins and 30 g of an additive composed of water, salt, and some vegetables. Once filled, the cans are closed, sealed and sterilised in an autoclave, leaving them completely watertight and disinfected. Finally, cans are placed in individual cardboard cases for sale. 10 cases are shrink-wrapped with plastic film and 5 of these packages are placed in cardboard boxes which are stored until their distribution and marketing.

7.2.2.3. Production of tuna pâté (SS3)

By-products tailored for human consumption that are not used for canning (e.g., gut meat or near the tail) are used to produce a tuna and black olive pâté that is marketed by the company. This pâté is composed of tuna (52%), olives (22%), extra virgin olive oil (12%), mashed potatoes (9%) and other minority components such as onion, garlic, black pepper and salt (5%). Once the parts of the tuna to be used for this pâté have been separated, they are directly crushed and mixed with the rest of the components of the pâté since tuna is already cooked. 125 g of the mixture is dosed into a 146 g glass jar, covered with an aluminium lid (8 g) and placed in a steriliser. A 0.5 g adhesive label is then added, and 10 jars are placed in cardboard boxes for storage and distribution.

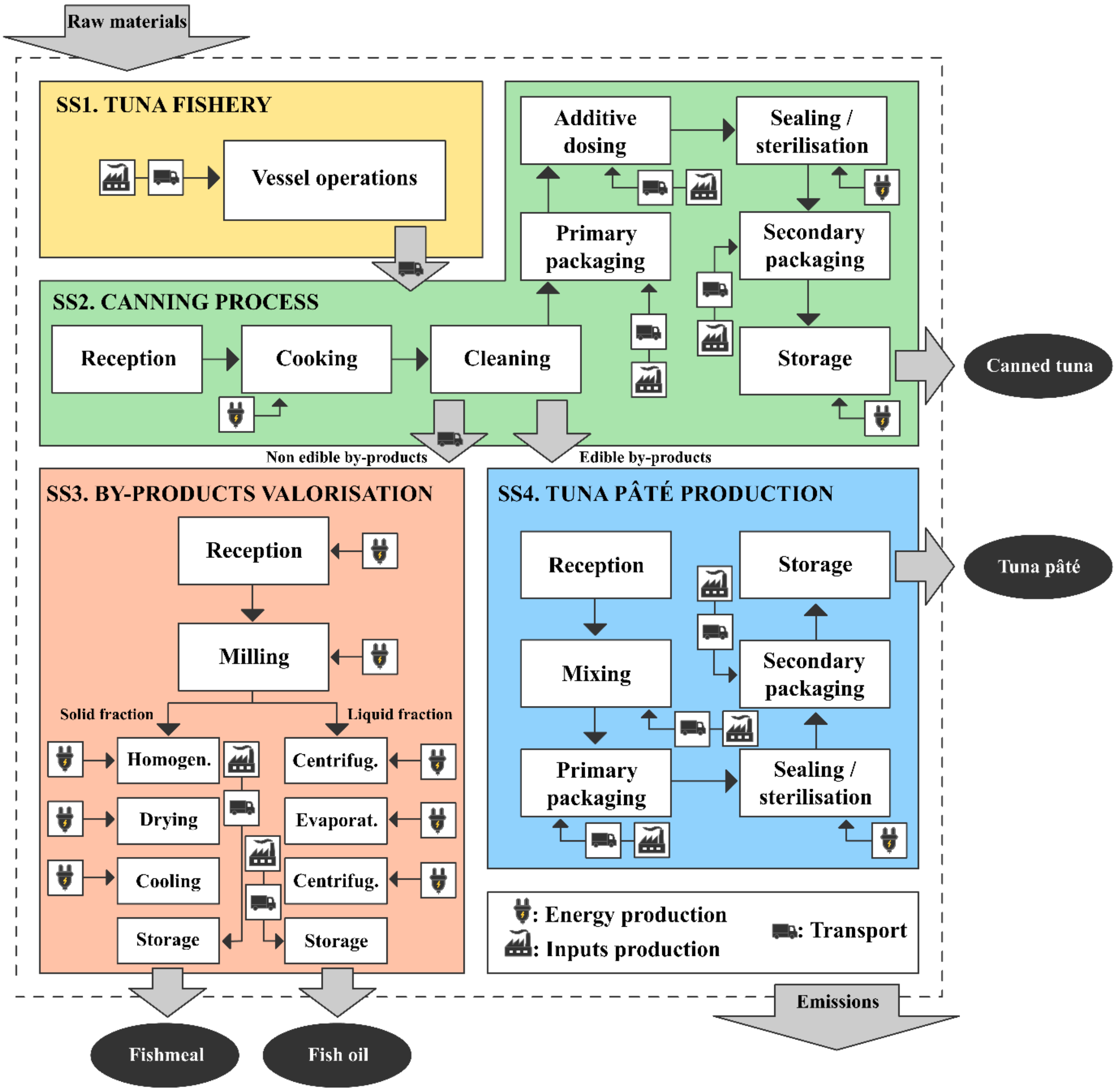


Figure 7.1. System boundaries for the environmental assessment of canned tuna value chain.

7.2.2.4. By-product valorisation (SS4)

The by-products that are not suitable for human consumption (heads, viscera, bones...) are taken to the crushers, where their size is reduced to less than 50 mm, and then to the storage hoppers. Pressing produces a press cake (solid phase) and water and cooking oils (liquid phase). The press cake passes through a homogeniser before being sent to the drying process which is carried out continuously inside the dryers to remove excess water at a temperature of around 100ºC. To continue the process, the temperature of the fishmeal is cooled by means of a sleeve cooler. Subsequently, the fishmeal is subjected to a grinding process in order to obtain a homogeneous product. Finally, the fishmeal is stored in silos, where it is shipped both in bulk and packaged in 25 kg format.

Conversely, with the help of a tricanter, the liquid phase is introduced into a continuous centrifuge, where the liquid is separated from the remaining solids, which are incorporated into the press cake before entering the dryer. At the outlet of the tricanter, the cooking water is concentrated by means of a double effect evaporator, using the exhaust gases from the press cake dehydration. Fish oil undergoes another centrifugation process to remove impurities and humidity before being stored in tanks until it is shipped.

7.2.3. Life cycle inventory, data collection and allocation approach

Data acquisition for the Life Cycle Inventory was mainly obtained through primary sources, provided by the canning company. Company staff filled out a questionnaire that collected all the information related to material and energy consumption of the plant. Data related to tuna fishing and transport (SS1) were obtained from the supplying company, which operates in the port of Arrecife. Details of material consumption were established following key parameters previously established as relevant in previous studies of fisheries for different species (Ramos et al., 2011; Vázquez-Rowe et al., 2011b, 2011a). Primary data on the bait requirements of the fleet were also obtained. In this regard, the sardine (*Sardina pilchardus*) fishery by an average purse seine fleet, which is landed in port and processed for bait production, was included within the system boundaries of this subsystem, as detailed in Vázquez-Rowe et al. (2011).

SS2 and SS3 data were obtained directly from a comprehensive questionnaire completed by cannery workers. This questionnaire detailed both the material and energy consumption of the plant, as well as a description of all the processes carried out in the canning plant. The production of the aluminium can considered the virgin/recycled aluminium ratio as 63/37% (Laso et al., 2017). Although the LCI of the production of the agricultural ingredients were taken from Ecoinvent, the production of Extra Virgin Olive Oil (EVOO) was taken from Laso et al. (2017). The transport of materials was taken into account considering primary information on the place of origin of the different raw materials. By-product valorisation data (SS4) was based on information from the environmental declarations of one of the most important plants belonging to leading Spanish companies in the fishmeal sector. The total production capacity of the plant is 50,000 tonnes/year for an average production of about 14,000 tonnes/year of fishmeal. Thus, primary data related to the generation of fish by-products were used to associate this production line with the rest of the data in this chapter. Is important to note that the life cycle impacts related to the production of the background processes (embodied emissions from raw material production processes) were taken from the Ecoinvent v3.5

database (Moreno Ruiz et al., 2018), considering the information obtained from primary sources.

Table 7.1. Life cycle inventory of the Subsystem 2. Canning process per FU (1 tonne of raw tuna at processing plant).

SUBSYSTEM 2. CANNING PROCESS					
Inputs from the Technosphere					
Materials	**Unit**	**Value**	**Transport**	**Unit**	**Value**
Tuna from SS1	kg	1,000	Tuna	t·km	2,097
Powder onion	g	241.2	Powder onion	kg·km	282.2
Powder garlic	g	160.8	Powder garlic	kg·km	188.1
White pepper	g	80.4	White pepper	kg·km	94.1
Salt	g	602.9	Salt	kg·km	30.1
Aluminium can	kg	73	Aluminium can	t·km	1.2
Bleached board	kg	36.5	Bleached board	t·km	27.0
Plastic film (LDPE)	kg	3.2	Plastic film	kg·km	255.1
Corrugated board	kg	34.5	Corrugated board	kg·km	621.9
Water	L	120.6	Cardboard waste	t·km	2.7
Sea water	L	5,096	Plastic waste	kg·km	347.8
Energy	**Unit**	**Value**			
Electricity	kWh	180.8			
Natural gas	MWh	2.6			
Outputs to the Technosphere					
Products	**Unit**	**Value**	**Waste to treatment**	**Unit**	**Value**
Canned tuna	Amount	4,055	Biowaste	kg	17.3
Co-products	**Unit**	**Value**	Plastic to recycling	kg	2.8
Inedible by-products to SS3	kg	627	Cardboard to recycling	kg	21.5
Edible by-products to SS4	kg	8	Wastewater	m^3	16.2

With all this primary information and some secondary data sources, the LCI was compiled, which involves the collection and computation of specific data that allow quantifying those inputs and outputs in the production system that contribute to a given impact category (Vázquez-Rowe et al., 2013). Table 7.1 contains the complete LCI of SS2, as it could be considered the core of this chapter; however, the detailed LCI of each subsystem considered in the study are shown in Tables E.1, E.2 and E.3 of the Appendix I.

The scenario under assessment is a multi-output system where more than one product is obtained. According to the functional unit chosen (1 tonne of tuna entering the canning industry), no allocation procedure was required. This system is a clear example of system expansion, following the guidelines set out in the ISO standards for dealing with multi-product systems. However, the use of allocation factors seems unavoidable to assess the environmental performance of a particular product; and even more so, within these circular processes, where the objective is precisely to maximise the production of outputs while minimising the consumption of inputs. In this case, although the main objective is to establish the environmental profile of the overall process, the economic values of all products have been considered, in order to relativise the environmental loads of the system in comparison with other food systems. Table 7.2 includes the different market prices considered, as well as total amount of produced outputs. The total production and market prices of canned tuna and tuna pâté were calculated considering only the total weight of the edible product, excluding the weight of the packaging.

Table 7.2. Capacity of end-products per FU and associated market prices.

	Production (kg)	Market price (€/kg)	Data source
Canned tuna	486.6	24.88	Primary information
Tuna pâté	15.4	26.96	Primary information
Fishmeal	172.2	0.55	Mullon et al. (2009)
Fish oil	36.1	0.61	Mullon et al. (2009)

7.3. RESULTS AND DISCUSSION

7.3.1. Environmental performance of the canned tuna value chain

According to the results shown in Figure 7.2, most of the environmental burdens are produced by two subsystems: SS1 on Tuna fishery and SS2 on Canning process. Thus, the combined contribution of these two subsystems accounts for an average of 94% of the impact in all impact categories, with a minimum of 87% in the ME category and a maximum of 99% in TA. The individual contribution of SS1 is variable, ranging from a minimum of 8.6% in ME to a maximum of 89.9% in TA. Regarding the SS2, it is especially relevant in the MRS and ME categories, with 89.9% and 78% of total contributions, respectively. The environmental impact of SS3 on By-products valorisation is fairly constant in all impact categories, always below 1%, except for the FE and ME categories. A contribution of 4% stands out in the ME category due to the treatment of wastewater generated in the valorisation process. Finally, regarding the production of tuna pâté (SS4), the environmental impact is variable and ranges from 0.8% in TA to the maximum reached in ME with 9.5%. This high value is again reached due to the treatment of the wastewater produced in the processing plant.

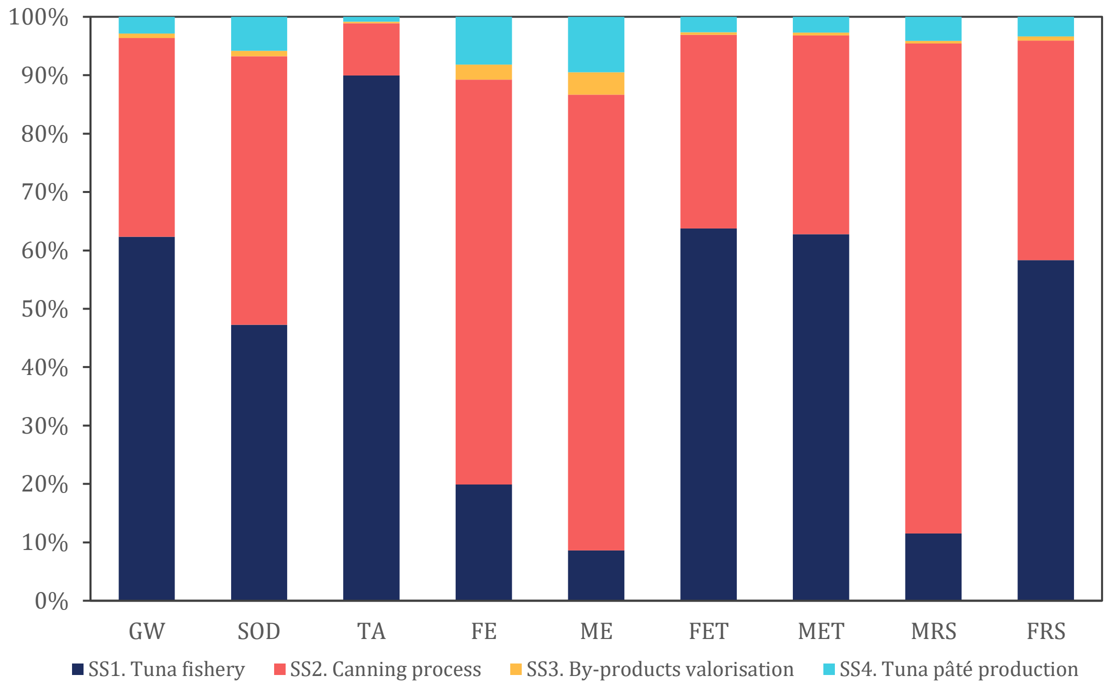


Figure 7.2. Relative contribution to environmental impacts associated with the canned tuna value chain.

Since most of the environmental impact of the entire value chain is produced in SS1 and SS2, Figure 7.3 provides a complete breakdown of the processes involved in these sub-systems. When comparing the pole-and-line tuna fishery in FAO 34 waters with other published studies, similar hotspots are observed: mainly fuel production and consumption, antifouling production and consumption, and GHG emissions of the refrigerant gas (Abdou et al., 2020, 2018; Vázquez-Rowe et al., 2011b), and bait fishing and processing (Vázquez-Rowe et al., 2014). As can be observed in Figure 7.3.a, fuel production and consumption during fishing operations turned out to be one of the main sources of environmental impact, accounting for more than 83% of the total impact in the TA category, 78% in FRS and 65% in GW. This result was expected and is in line with other fishing fleets revised, where direct and indirect fuel emissions were highlighted as the most important carrier of GHG emissions (Avadí et al., 2018; Sandison et al., 2021; Vázquez-Rowe et al., 2014; Villanueva-Rey et al., 2018), as well as other environmental impacts, such as terrestrial acidification (Ziegler et al., 2016). In this article, a Fuel Use Intensity (FUI) of 548.9 L/tonne was obtained for pole-and-line tuna fishing. This result can be compared with different results published in scientific literature; e.g. Miller et al. (2017) estimated the fuel consumption of the Maldivian pole-and-line fleet among different shoal of tuna, varying from 200 L/tonne to almost 600 L/tonne.

It is worth noting that special emphasis is placed on the fact that bait accounts for approximately 15-20% of the amount of fuel consumed in a fishing trip, which coincides with the results obtained in this chapter, since the impacts associated with fishing and bait production reach 71.6% in ME and do not fall below 16% in any impact category.

Pole-and-line tuna fishing methods have not been traditionally studied, so there are only a few studies that quantify the fuel consumption of this type of fishing gear with different target species, however, tuna fishing by different fishing gears has been extensively studied. In this context, Hospido and Tyedmers (2005) quantified the fuel consumption of the Spanish tuna purse seine fishing fleet in the Indian (373 L/tonne), Atlantic (442 L/tonne) and Pacific (442 L/tonne) oceans, setting the framework for the quantification of FUI of other fisheries in the future. Moreover, Parker et al. (2015) achieved a significantly larger sample of vessels to update this result, obtaining on average that the tuna purse seine fishery consumes on average 365 L/tonne.

Specifically, the purse seine fishery for skipjack tuna in Atlantic waters requires a fuel consumption of 445 L/tonne according to their results. On the other hand, Parker and Tyedmers (2015) estimated a fuel consumption of 1,612 L/tonne for large pelagic (mainly tuna) fisheries using longlines and other forms of pole-and-lines. From another, much more generic point of view, Parker et al. (2018) calculated the global CO_2 emissions linked to fuel combustion in fishing vessels, estimating the FUI of pelagic fish (> 30 cm) at 430 L/tonne. In brief, most of the values provided for tuna fishing, except for longlines, are within the same range, which can give the idea that the values obtained in this chapter are close to reality. However, it is important to note that FUI measurements from previous studies may vary considerably depending on both the measurement method and the analysed fishing gear (Parker et al., 2015).

On the other side, the production and consumption of antifouling has been revealed as a differential element in the ecotoxicity categories (74.5% in FET and 71% in MET) due to copper and zinc emissions during the use stage, as demonstrated in previous literature (Avadí and Fréon, 2013). It is also noteworthy the 20% of total fishing impact in MRS category, due to the extraction of bauxite to produce cooper needed for antifouling formulation. The bauxite ore is treated with dilute sulphuric acid over a period of months, dissolving copper to form a weak solution of copper sulphate, from which copper can recovered by electrolysis. Ice production contributes much less to the environmental burdens of the system, its contribution remains almost constant in all categories ranging from a minimum of 0.5% to a maximum of 2%, except in the FET category, where it reaches a maximum of 7%, mainly due to electricity consumption and the high dependence on coal in the Spanish electricity profile. Within the GW category, the presence of R-410A stands out, which is a refrigerant gas widely used in refrigeration machines and whose leakage influences the carbon footprint of the process, as has been shown in previous literature (Vázquez-Rowe et al., 2011b). Finally, the environmental impact of paint consumption is practically negligible, while the treatment of the waste (both solid and liquid) is only remarkable in the ME category due to the treatment of bilge wastewater.

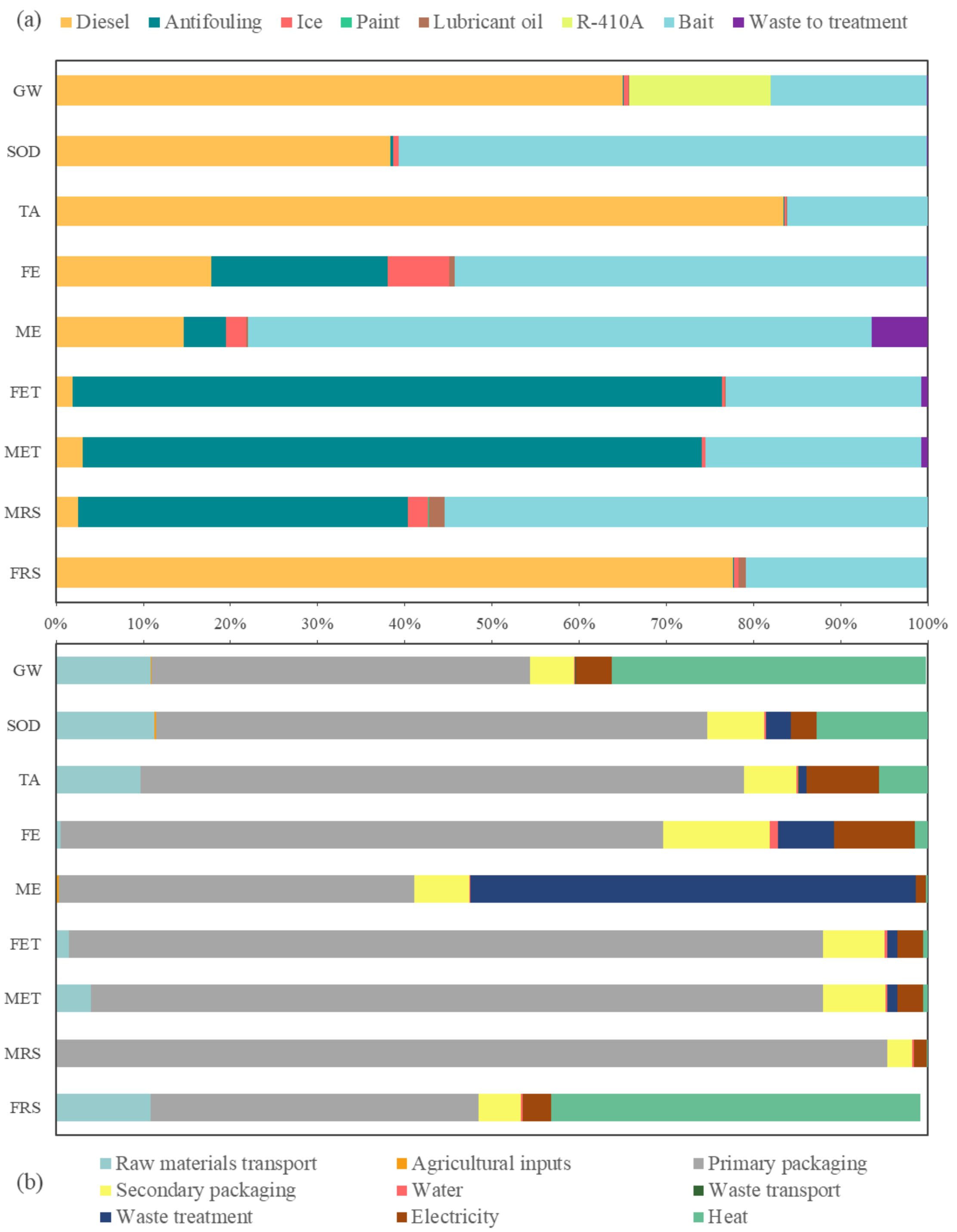


Figure 7.3. Relative contribution to environmental impacts associated with Subsystem 1. Tuna fishery (a) and Subsystem 2. Canning process (b).

According to the results shown in Figure 7.3.b, the largest contribution to the environmental impact in 6 of the 9 impact categories analysed comes from the production of aluminium primary packaging, proving to be one of the most important elements in the production of canned seafood (Almeida et al., 2014; Hospido et al., 2006). In fact, the production of aluminium cans is the main contributor to the MRS category, where it reaches the highest contribution to any impact category (95%), linked to the production of virgin aluminium, which requires a high consumption of bauxite (almost 5 kg per kg of aluminium) (Laso et al., 2017). Heat production by natural gas (for cooking and sterilisation) is also an important contributor to GW (36.2%) and FRS (43%), although its contribution to the other categories is in all cases less than 5%. Wastewater treatment, as in the fisheries subsystem (Figure 7.3.a) is only relevant in the ME category (50%) due to nutrient emissions of the treated effluent, while in the other categories, it hardly exceeds 6%. The production of secondary packaging (cardboard and plastic) presents a relative constant contribution around 5-7%, reaching 12.2% in FE and dropping to 2.9% in MRS. Electricity consumption is almost negligible, only relevant in the FE and TA categories (9.2% and 8.3%, respectively), while in the rest of the categories it hardly exceeds 4%. Finally, the production of the agricultural ingredients for the garnish and seasoning (onion, garlic, pepper, etc.) and the transport of waste to specialised plants have a very low contribution in all impact categories, always below 0.5%. The numerical results related to the FU can be found in the Table S.4 of the supplementary material.

7.3.2. Effect of the allocation strategies on the environmental profile

This chapter has assessed the entire canned tuna value chain from an environmental point of view, from the production of raw materials to the processing of products and co-products and the treatment of waste, trying to move towards the target of 0 bio-waste. This chapter is an example of system expansion to avoid the use of allocation strategies, as prioritised in the ISO standards. However, when the objective of the study is to analyse the environmental impacts associated with a particular product within the value chain, and full segregation of material and energy consumption for each production line is not possible, the use of allocation factors to accurately report environmental burdens seems unavoidable (Ayer et al., 2007). Traditionally, the selection of allocation factors is one of the procedures that generates the least consensus among LCA practitioners. In this regard, different authors have proposed different allocation methods for different case studies. Within the seafood-specific case studies, for example, Thrane (2006) used system expansion strategies to handle the co-product allocation in the LCA of different fish products. On the other side, Ziegler et al. (2003) applied economic allocation factors to calculate the environmental burdens of cod fillets, but also they applied mass allocation considering that price fluctuations may condition the reliability of the results. On this basis and considering that the main objective of the canning industry is the production of marketable products that generate an economic income, the economic allocation was calculated for canned tuna (95.8%), tuna pâté (3.3%), fishmeal (0.7%) and fish oil (0.2%).

When analysing the environmental profile of the main product (canned tuna), one different approach from system expansion corresponds to the exclusion of subsystems SS3 and SS4 from the system boundaries, including within the boundaries of SS2 the organic waste treatment processes corresponding to the co-products (635 kg per 1,000 kg of raw tuna). However, this approach would not be entirely realistic, as currently, the processing of fish co-products into fishmeal and fish oil for feed formulation is a widely used option for the treatment of bio-waste. In any case, Figure 7.4 shows the variation in the environmental profile of the original value chain (Base scenario) compared to the scenario in which only tuna fishing and processing to produced canned tuna is included, but the valorisation of organic waste from the processing is excluded (Non-circular scenario).

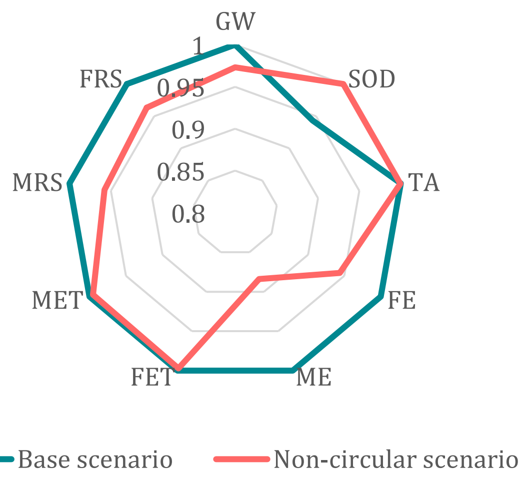


Figure 7.4. Comparative environmental profile of the two alternative canned tuna production scenarios.

It seems obvious that removing SS3 and SS4 with all associated material and energy consumption from the system boundaries will reduce the environmental impact of the entire value chain. Especially relevant is the reduction in the categories ME and FE, where the contribution of the SS4 subsystem was higher due to the environmental burdens of wastewater treatment, as can be seen in Figure 7.2. However, it is remarkable that the impact of the non-circular approach is almost equal to that of the baseline scenario in the categories of MET, FET and TA. Even more remarkable is that the impact on the SOD category is higher in the non-circular scenario, as the emissions from bio-waste treatment are higher than the emissions from the inputs and outputs of SS3 and SS4.

In order to relativise the impacts of the system towards the production of canned tuna, in the case of the baseline scenario it is necessary to apply economic allocation factors, while for the non-circular scenario, the entire environmental burden is allocated to canned tuna, as it is the main output of the system. Considering that only 365 kg out of 1,000 kg are canned and that 90 g of tuna and 30 g of additive are put into each can, the carbon footprint of a can of tuna was quantified as 0.98 kg CO_2 eq in the baseline scenario (considering the economic allocation of all co-products), while in the non-circular scenario it reached 1.01 kg CO_2 eq per can. To give a broader value and not so focused on the carbon footprint, the total environmental impacts were calculated in terms of the

ReCiPe methodology endpoint, ranging between 0.050 and 0.052 pts per can in the baseline scenario and in the non-circular scenario, respectively.

In this way, the comparison between the environmental profile of the system evaluated in Figure 1 (Base scenario) and this new scenario (Non-circular scenario) allow to answer the question: is the application of multi-product strategies environmentally viable when the objective of the assessment is to assess only one of the products, without considering the products avoided and the environmental credits? In the specific case studied and taking into account that the inclusion of SS3 and SS4 has not had much influence on the total impact of the system (less than 6% on average), the distribution of the loads between the 4 different products has made it possible to reduce the life cycle impact of canned tuna, while the environmental performance of the whole value chain is hardly modified. It can be seen that a circular economy approach is feasible and effective and these principles are in line with those mentioned by the European Commission in the Circular Economy action plan (European Commission, 2020b).

7.3.3. Benchmarking with other canned seafood

When comparing with other results available in the peer-reviewed literature, it is important to note that there are some issues that need to be addressed in detail in order to make realistic comparisons (Avadí and Fréon, 2013; Vázquez-Rowe et al., 2012): (i) The life cycle impact methodology should be detailed since, although the categories of different methodologies may be analogous, in many cases the units of measurement are different; (ii) The functional unit selected for the analysis, as well as the edible content of each product, to adequately estimate the associated environmental impact per unit and per quantity of product (e.g. 1 kg); and (iii) the environmental indicator used for comparison (environmental footprint, normalised impact factor, etc.). Taking all this information into account, Table 7.3 shows the detailed results of the comparison between the carbon footprint of different canned seafood products.

It is important to note that Table 7.3 compiles the results with a cradle-to-gate approach, considering the impacts related to fishing, processing and packaging. Thus, in those studies that considered the distribution and consumption stages (Almeida et al., 2015; Laso et al., 2017; Vázquez-Rowe et al., 2014), the environmental burdens corresponding to these stages were not taken into account.

The carbon footprint of canned tuna presents an intermediate value, much lower than the values presented by sardine in olive oil (Vázquez-Rowe et al., 2014) and mussels, mainly due to the production of the primary packaging, as these two cases were packed in tinplate. However, this value is very similar to the associated with the can of sardines with olive oil assessed in Almeida et al. (2015), where similar packaging is used and similar techniques are followed.

Cantabrian anchovy is also packaged in aluminium and, despite having a much lower fish/package ratio than that obtained in this study, they reported only 4.7 kg CO_2/kg. This is because the processing operations are very different, and the catch ratios of this fishery are much higher. In this sense, the low results for Ecuadorian canned tuna (Avadí et al., 2015) and Peruvian anchovy (Avadí et al., 2014) can be explained by lower fuel use in the Ecuadorian and Peruvian fisheries, mainly due to a better catch per unit effort in relation to a higher abundance of the resource (Fréon et al., 2014a). While it is true that in all cases it was concluded that both the fishing stage and the production of primary packaging (tinplate or aluminium) are the main drivers of environmental impacts, and all improvement actions should focus on them.

Table 7.3. Carbon footprint values for other canned seafood products

Product	Assessment method	Product	kg CO_2/kg	Reference
Tuna	ReCiPe 2016	90 g tuna 30 g additive 20 g can 5 g board	8.2	This chapter
Pilchard	CML-IA Baseline	95 g pilchard 35 g olive oil 20 g can	7.5	(Almeida et al., 2015)
Pilchard	ReCiPe 2008	85 g pilchard 35 g olive oil 20 g can	25.2	(Vázquez-Rowe et al., 2014)
Mussels	IPCC 2016	129 g mussels 120 g sauce 81 g can 12.7 g board	17.5	(Iribarren et al., 2010)
Tuna	ReCiPe 2008	n.d.	3.7	(Avadí et al., 2015)
Peruvian anchovy	ReCiPe 2008	n.d.	1.7	(Avadí et al., 2014)
Cantabrian anchovy	ESA with metrics from ICheme 2002	30 g anchovy 20 g olive oil 15 g can 5 g board	4.7	(Laso et al., 2017)

7.4. CONCLUSIONS

This chapter demonstrates that, from a product approach, the inclusion of by-product valorisation processes to address a multi-product strategy improves the environmental profile of the main product. It is a clear example of a system expansion to avoid burden allocation between products when the focus is on the assessment of the entire value chain. When the focus is on the assessment of a single product, the allocation of environmental burdens seems unavoidable.

It has been shown that the fishing and primary processing stages are the most relevant sub-systems within the environmental profile of the canned tuna value chain. The inventory of the fishing stage showed, as previous studies on different fishing fleets, that the impacts of the fishing stage come mainly from the production and consumption of diesel and antifouling. In this case, the importance of the bait used for fishing also stands out, as it requires the fishing and processing of sardine for use as bait. Primary packaging presented the highest environmental impact in the life cycle impacts of canned tuna. Aluminium production, lamination and extrusion had the highest impact in almost all impact categories, as expected for canned products. By-product valorisation processes, both edible and inedible, have proven to have a low impact.

This system has allowed an approximation of the EU target towards a cradle-to-cradle system approach, achieving the goal of zero biowaste. The results show the need to improve the application of the circular economy in the primary sector, converting waste into raw materials to produce new products, minimising the consumption of material and energy resources. In this sense, the application of multi-product strategies has been shown to improve the environmental profile of canned products through the allocation of environmental burdens among the new products, although further analysis from a sustainability point of view is required. Similar studies need to be further applied to specific primary sub-sectors in the future to continue the path towards a more sustainable and circular food system.

7.5. REFERENCES

Abdou, K., Gascuel, D., Aubin, J., Romdhane, M.S., Ben Rais Lasram, F., Le Loc'h, F., 2018. Environmental life cycle assessment of seafood production: A case study of trawler catches in Tunisia. Sci. Total Environ. 610–611, 298–307. https://doi.org/10.1016/j.scitotenv.2017.08.067

Abdou, K., Le Loc'h, F., Gascuel, D., Romdhane, M.S., Aubin, J., Ben Rais Lasram, F., 2020. Combining ecosystem indicators and life cycle assessment for environmental assessment of demersal trawling in Tunisia. Int. J. Life Cycle Assess. 25, 105–119. https://doi.org/10.1007/s11367-019-01651-5

Almeida, C., Vaz, S., Cabral, H., Ziegler, F., 2014. Environmental assessment of sardine (*Sardina pilchardus*) purse seine fishery in Portugal with LCA methodology including biological impact categories. Int. J. Life Cycle Assess. 19, 297–306. https://doi.org/10.1007/s11367-013-0646-5

Almeida, C., Vaz, S., Ziegler, F., 2015. Environmental Life Cycle Assessment of a Canned Sardine Product from Portugal. J. Ind. Ecol. 19, 607–617. https://doi.org/10.1111/jiec.12219

Ardán, 2018. Informe económico y de competitividad 2018. Capítulo 11: El sector de la pesca en Galicia.

Avadí, A., Adrien, R., Aramayo, V., Fréon, P., 2018. Environmental assessment of the

Peruvian industrial hake fishery with LCA. Int. J. Life Cycle Assess. 23, 1126–1140. https://doi.org/10.1007/s11367-017-1364-1

Avadí, A., Bolaños, C., Sandoval, I., Ycaza, C., 2015. Life cycle assessment of Ecuadorian processed tuna. Int. J. Life Cycle Assess. 20, 1415–1428. https://doi.org/10.1007/s11367-015-0943-2

Avadí, A., Fréon, P., 2013. Life cycle assessment of fisheries: A review for fisheries scientists and managers. Fish. Res. 143, 21–38. https://doi.org/10.1016/j.fishres.2013.01.006

Avadí, A., Fréon, P., Quispe, I., 2014. Environmental assessment of Peruvian anchoveta food products: Is less refined better? Int. J. Life Cycle Assess. 19, 1276–1293. https://doi.org/10.1007/s11367-014-0737-y

Ayer, N.W., Tyedmers, P.H., Pelletier, N.L., Sonesson, U., Scholz, A., 2007. Co-product allocation in life cycle assessments of seafood production systems: Review of problems and strategies. Int. J. Life Cycle Assess. 12, 480–487. https://doi.org/10.1065/lca2006.11.284

Ciccullo, F., Cagliano, R., Bartezzaghi, G., Perego, A., 2021. Implementing the circular economy paradigm in the agri-food supply chain: The role of food waste prevention technologies. Resour. Conserv. Recycl. 164, 105114. https://doi.org/10.1016/j.resconrec.2020.105114

Corona, B., Shen, L., Reike, D., Rosales Carreón, J., Worrell, E., 2019. Towards sustainable development through the circular economy—A review and critical assessment on current circularity metrics. Resour. Conserv. Recycl. 151, 104498. https://doi.org/10.1016/j.resconrec.2019.104498

EUMOFA, 2019. The EU fish market, 2019 edition. Directorate-General for Maritime Affairs and Fisheries, Brussels. https://doi.org/10.2771/168390

European Commission, 2020a. Facts and figures on the Common Fisheries Policy - 2020 Edition.

European Commission, 2020b. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the regions. A new Circular Economy Action Plan For a cleaner and more competitive Europe. Brussels.

FAO, 2020. The State of World Fisheries and Aquaculture - Sustainability in action. Food and Agriculture Organization of the United Nations, Rome.

Fréon, P., Avadí, A., Soto, W.M., Negrón, R., 2014a. Environmentally extended comparison table of large- versus small- and medium-scale fisheries: The case of the Peruvian anchoveta fleet. Can. J. Fish. Aquat. Sci. 71, 1459–1474. https://doi.org/10.1139/cjfas-2013-0542

Fréon, P., Sueiro, J.C., Iriarte, F., Miro Evar, O.F., Landa, Y., Mittaine, J.F., Bouchon, M., 2014b. Harvesting for food versus feed: A review of Peruvian fisheries in a global context. Rev. Fish Biol. Fish. 24, 381–398. https://doi.org/10.1007/s11160-013-9336-4

García-del-Hoyo, J.J., Jiménez-Toribio, R., Guillotreau, P., 2017. A demand analysis of the Spanish canned tuna market. Mar. Policy 86, 127–133. https://doi.org/10.1016/j.marpol.2017.09.014

García-Santiago, X., Franco-Uría, A., Antelo, L.T., Vázquez, J.A., Pérez-Martín, R., Moreira, M.T., Feijoo, G., 2020. Eco-efficiency of a marine biorefinery for valorization of cartilaginous fish biomass. J. Ind. Ecol. 1–13. https://doi.org/10.1111/jiec.13066

Hospido, A., Tyedmers, P., 2005. Life cycle environmental impacts of Spanish tuna fisheries. Fish. Res. 76, 174–186. https://doi.org/10.1016/j.fishres.2005.05.016

Hospido, A., Vazquez, M.E., Cuevas, A., Feijoo, G., Moreira, M.T., 2006. Environmental assessment of canned tuna manufacture with a life-cycle perspective. Resour. Conserv. Recycl. 47, 56–72. https://doi.org/10.1016/j.resconrec.2005.10.003

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M., Zijp, M., Hollander, A., van Zelm, R., 2017. ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. Int. J. Life Cycle Assess. 22, 138–147. https://doi.org/10.1007/s11367-016-1246-y

Iribarren, D., Hospido, A., Moreira, M.T., Feijoo, G., 2010. Carbon footprint of canned mussels from a business-to-consumer approach. A starting point for mussel processors and policy makers. Environ. Sci. Policy 13, 509–521. https://doi.org/10.1016/j.envsci.2010.05.003

ISO, 2006a. ISO 14040 - Environmental Management - Life Cycle Assessment - Principles and Framework.

ISO, 2006b. ISO 14044 - Environmental Management - Life Cycle Assessment - Requirements and Guidelines.

Khan, A.M.A., Gray, T.S., Mill, A.C., Polunin, N.V.C., 2018. Impact of a fishing moratorium on a tuna pole-and-line fishery in eastern Indonesia. Mar. Policy 94, 143–149. https://doi.org/10.1016/j.marpol.2018.05.014

Korhonen, J., Honkasalo, A., Seppälä, J., 2018. Circular Economy: The Concept and its Limitations. Ecol. Econ. 143, 37–46. https://doi.org/10.1016/j.ecolecon.2017.06.041

Laso, J., García-Herrero, I., Margallo, M., Vázquez-Rowe, I., Fullana, P., Bala, A., Gazulla, C., Irabien, Á., Aldaco, R., 2018. Finding an economic and environmental balance in value chains based on circular economy thinking: An eco-efficiency methodology applied to the fish canning industry. Resour. Conserv. Recycl. 133, 428–437. https://doi.org/10.1016/j.resconrec.2018.02.004

Laso, J., Margallo, M., Fullana, P., Bala, A., Gazulla, C., Irabien, A., Aldaco, R., 2017. Introducing life cycle thinking to define best available techniques for products: Application to the anchovy canning industry. J. Clean. Prod. 155, 139–150. https://doi.org/10.1016/j.jclepro.2016.08.040

Miller, K.I., Adam, M.S., Baske, A., 2017. Rates of fuel consumption in the Maldivian Pole-and-Line tuna fishery. International Pole & Line Foundation, London and marine research Centre, Maldives.

Moreno Ruiz, E., Valsasina, L., Brunner, F., Symeonidis, A., FitzGerald, D., Treyer, K., Bourgault, G., Wernet, G., 2018. Documentation of changes implemented in the Ecoinvent Database v3.5. Ecoinvent, Zurich, Switzerland.

Mullon, C., Mittaine, J.F., Thébaud, O., Péron, G., Merino, G., Barange, M., 2009. Modeling the global fishmeal and fish oil markets. Nat. Resour. Model. 22, 564–609. https://doi.org/10.1111/j.1939-7445.2009.00053.x

Niero, M., Kalbar, P.P., 2019. Coupling material circularity indicators and life cycle based indicators: A proposal to advance the assessment of circular economy strategies at the product level. Resour. Conserv. Recycl. 140, 305–312. https://doi.org/10.1016/j.resconrec.2018.10.002

Parker, R.W.R., Blanchard, J.L., Gardner, C., Green, B.S., Hartmann, K., Tyedmers, P.H., Watson, R.A., 2018. Fuel use and greenhouse gas emissions of world fisheries. Nat. Clim. Chang. 8, 333–337. https://doi.org/10.1038/s41558-018-0117-x

Parker, R.W.R., Tyedmers, P.H., 2015. Fuel consumption of global fishing fleets: Current understanding and knowledge gaps. Fish Fish. 16, 684–696. https://doi.org/10.1111/faf.12087

Parker, R.W.R., Vázquez-Rowe, I., Tyedmers, P.H., 2015. Fuel performance and carbon footprint of the global purse seine tuna fleet. J. Clean. Prod. 103, 517–524. https://doi.org/10.1016/j.jclepro.2014.05.017

Ramos, S., Vázquez-Rowe, I., Artetxe, I., Moreira, M.T., Feijoo, G., Zufía, J., 2011. Environmental assessment of the Atlantic mackerel (*Scomber scombrus*) season in the Basque Country. Increasing the timeline delimitation in fishery LCA studies. Int. J. Life Cycle Assess. 16, 599–610. https://doi.org/10.1007/s11367-011-0304-8

Rufí-Salís, M., Petit-Boix, A., Villalba, G., Gabarrell, X., Leipold, S., 2021. Combining LCA and circularity assessments in complex production systems: the case of urban agriculture. Resour. Conserv. Recycl. 166, 105359. https://doi.org/10.1016/j.resconrec.2020.105359

Ruiz-Salmón, I., Laso, J., Margallo, M., Villanueva-Rey, P., Rodríguez, E., Quinteiro, P., Dias, A.C., Almeida, C., Nunes, M.L., Marques, A., Cortés, A., Moreira, M.T., Feijoo, G., Loubet, P., Sonnemann, G., Morse, A.P., Cooney, R., Clifford, E., Regueiro, L., Méndez, D., Anglada, C., Noirot, C., Rowan, N., Vázquez-Rowe, I., Aldaco, R., 2021. Life cycle assessment of fish and seafood processed products – A review of methodologies and new challenges. Sci. Total Environ. 761. https://doi.org/10.1016/j.scitotenv.2020.144094

Sandison, F., Hillier, J., Hastings, A., Macdonald, P., Mouat, B., Marshall, C.T., 2021. The environmental impacts of pelagic fish caught by Scottish vessels. Fish. Res. 236, 105850. https://doi.org/10.1016/j.fishres.2020.105850

Vázquez-Rowe, I., Hospido, A., Moreira, M.T., Feijoo, G., 2012. Best practices in life cycle assessment implementation in fisheries. Improving and broadening environmental assessment for seafood production systems. Trends Food Sci. Technol. 28, 116–131.

https://doi.org/10.1016/j.tifs.2012.07.003

Vázquez-Rowe, I., Iribarren, D., Hospido, A., Moreira, M.T., Feijoo, G., 2011a. Computation of operational and environmental benchmarks within selected galician fishing fleets. J. Ind. Ecol. 15, 776–795. https://doi.org/10.1111/j.1530-9290.2011.00360.x

Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2011b. Life Cycle Assessment of fresh hake fillets captured by the Galician fleet in the Northern Stock. Fish. Res. 110, 128–135. https://doi.org/10.1016/j.fishres.2011.03.022

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. https://doi.org/10.1016/j.scitotenv.2013.12.099

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. https://doi.org/10.1016/j.jclepro.2012.11.049

Villanueva-Rey, P., Vázquez-Rowe, I., Arias, A., Moreira, M.T., Feijoo, G., 2018. The importance of using life cycle assessment in policy support to determine the sustainability of fishing fleets: a case study for the small-scale xeito fishery in Galicia, Spain. Int. J. Life Cycle Assess. 23, 1091–1106. https://doi.org/10.1007/s11367-017-1402-z

Wilson, J.R., Bradley, D., Phipps, K., Gleason, M.G., 2020. Beyond protection: Fisheries co-benefits of no-take marine reserves. Mar. Policy 122, 104224. https://doi.org/10.1016/j.marpol.2020.104224

Ziegler, F., Hornborg, S., Green, B.S., Eigaard, O.R., Farmery, A.K., Hammar, L., Hartmann, K., Molander, S., Parker, R.W.R., Skontorp Hognes, E., Vázquez-Rowe, I., Smith, A.D.M., 2016. Expanding the concept of sustainable seafood using Life Cycle Assessment. Fish Fish. 17, 1073–1093. https://doi.org/10.1111/faf.12159

Ziegler, F., Nilsson, P., Mattsson, B., Walther, Y., 2003. Life Cycle Assessment of frozen cod fillets including fishery-specific environmental impacts. Int. J. Life Cycle Assess. 8, 39–47. https://doi.org/10.1007/bf02978747

SECTION IV

ECO-EFFICIENCY ASSESSMENT

Chapter 8

Pursuing the route to eco-efficiency in dairy production: the case of Galician area

Summary

Taking into account that livestock is the main carrier in the consumption and emissions derived from the food industry, it seems clear that the search for more efficient and sustainable processes should become the cornerstone of any production system. The combined use of Life Cycle Assessment (LCA) and Data Envelopment Analysis (DEA) can be an appropriate methodology to assess the eco-efficiency of multiple units and providing targets and benchmarks. Based on this approach, the study of environmental sustainability and eco-efficiency allows to learn from those systems that may represent best practice. This work advances in this direction by integrating both analysis methodologies in the calculation of environmental indicators associated with milk production for a large group of farms, nearly 100 decision-making units, mainly familiar farms (between 10 and 500 livestock heads) throughout the Galician municipalities of Santa Comba, Lalín and Rodeiro. Twenty-one dairy farms were identified as efficient, and the average efficiency score of the inefficient farms was 0.58. Based on the comparison of current operation levels with target levels, it was possible to quantify average reductions of up to 53% for input consumption levels, resulting in average impact reductions of 49% in carbon footprint and 55% in water footprint indicators. This chapter shows how the Galician dairy sector must address sustainable development objectives, especially those established in Agenda 2030 to achieve sustainable improvement.

This chapter, under the consent of the publisher, was redrafted after: Cortés, A., Feijoo, G., Fernández, M., Moreira, M. T. (2021). Pursuing the route to eco-efficiency in dairy production: The case of Galician area. Journal of Cleaner Production, 252, 124861. DOI: https://doi.org/10.1016/j.jclepro.2020.124861. More information related to this article can be found in Appendix III.

OUTLINE

8.1. INTRODUCTION

When referring to the agricultural sector, not only agrarian activities should be included, but also livestock activities, which represent an important proportion of the entire primary sector (Grossi et al., 2019). In fact, the environmental impacts of the food industry are largely driven by livestock production, which accounts for 3-8% of total energy consumption and emits 14.5% of total anthropogenic GHG emissions worldwide (Eurostat, 2020), associated with emissions of nitrous oxide (N_2O) and methane (CH_4) from enteric fermentation, fertilisation activities and manure storage (Aguirre-Villegas et al., 2015). Despite their relevance, the impacts of this sector on other environmental aspects, such as eutrophication, acidification and water scarcity, should not be ignored (González-García et al., 2013). Predictions of the climate impact are hard to quantify, however, it is generally accepted that the agricultural sector is particularly vulnerable to the impact of climate change (McEldowney, 2021). It seems obvious that Climate change will affect the conditions necessary for growing fruit, vegetables, cereals, and livestock and the availability of water will affect the sustainability of farms and human settlements (Grossi et al., 2019).

Today, milk is one of the most widely produced foods in the world (Üçtuğ, 2019), with dairy products being a fundamental pillar of the human diet (Wang et al., 2018). In the context of the European Union, Spain is the seventh largest producer of cow milk, with 5% of the total (Eurostat, 2019). In Spain, the dairy sector is the second most important of all the livestock sectors. The latest data published by the Spanish Agrarian Guarantee Fund (FEGA, 2019) show that the Spanish dairy industry processes more than 7 million m^3 of milk. Galicia, a region in northwest Spain, produces 38% of the national milk production (MAPA, 2019), making it the ninth largest dairy region in Europe, with a remarkable turnover of 800 million euros and more than 25,000 people employed. Given this context, it is desirable to propose strategies for environmental improvement in livestock and milk production.

Among the different methods to evaluate the environmental performance of milk production, Life Cycle Assessment has been applied in recent years for a wide range of production systems in different countries (Baldini et al., 2020; Berton et al., 2020; Djekic et al., 2019; Egas et al., 2020; Escribano et al., 2020; Famiglietti et al., 2019; Knudsen et al., 2019; Woldegebriel et al., 2017). Noya et al. (2018) evaluated the environmental burdens of milk production in facilities of Northeast Spain. Although a wide range of environmental indicators were calculated, the study focused mainly on the water footprint according to the Water Footprint Network (WFN). The capital importance of feed production in the water footprint was demonstrated due to characterisation factors of agricultural products. Baldini et al. (2018) compared the environmental profile of three Italian dairy farms within two different scenarios, estimating the direct gaseous emissions according to the Intergovernmental Panel on Climate Change (IPCC) and European Environmental Agency (EAA) guidelines. Pirlo and Lolli (2019) carried out a different comparison, eight conventional and six organic dairy farms from Italy. This study concluded that conventional production is slightly higher that organic (9,004 vs. 7,736 kg/cow per year, respectively). However, the differences in environmental impacts in terms of GWP, ACP and EUP categories were not significant. Other authors focus their research on establishing the environmental performance of milk production based on a single indicator. Thus, numerous papers on carbon footprint (Finnegan et al., 2017; Horrillo et al., 2020; Laca et al., 2020; Morais et al., 2018; Vida and Tedesco, 2017) or water footprint (Lu et al., 2018; Mekonnen et al., 2019; Payen et al., 2018; Usva et al., 2019) have been published in recent years. All these studies present differences in the selection in the FU, system boundaries, allocation factors. These are precisely the characteristics that make LCA a versatile tool, but whose methodological decisions shall be unified into a comprehensive approach for milk production systems.

Since the use of high-quality data is essential for a study to be transparent and reliable, it is often necessary to collect inventory data from different similar facilities to ensure the representativeness of the data. A common solution for managing a large volume of data is to establish an average. However, the high degree of variability that results from such a system can result in uncertainty. An alternative approach to dealing with these cases is to conduct individual analyses for each inventory. However, this approach makes the results difficult to interpret. It is therefore necessary to use methodologies that allow performance indicators to be determined for the operating system as a whole, considering all facilities. This is how the combined use of the LCA and DEA methodologies came about, which allows for the assessment of the eco-efficiency of similar production systems that enables the environmental and operational assessment of similar production systems.

Taking into account that the chapters 3-7 were focused on the valorisation of winemaking-derived waste and the environmental characterisation of fisheries-related case study and considering that the objective of this doctoral thesis is to propose circular economy-based solutions for strategic primary sectors. It seems appropriate to include a section where the focus is placed on advancing in the look for the eco-efficiency in the other brands of the agricultural and fisheries sectors, livestock and aquaculture.

In this sense, the objective of this chapter focuses on the application of LCA + DEA methodology to a group of 96 dairy farms throughout Galicia to evaluate the eco-efficiency of the Galician dairy sector. Based on this approach, we can advance the study of environmental sustainability and eco-efficiency not only of a case study or an individual process unit, but we can benchmark similar systems that behave significantly differently. In this way it is possible to learn from those systems that may represent best practice. A secondary objective is to establish the "hot-spots" in milk production process by determining two widely used environmental indicators: Carbon and Water Footprint.

8.2. MATERIALS AND METHODS

8.2.1. Definition of the case study

Galician dairy farms are characterised, like all agricultural and livestock farms, by a great variability in the consumption of materials and production models (Aguirre-Villegas et al., 2017). Thus, it is necessary to include as many farms as possible in the analysis so that the sample is characteristic of the Galician dairy sector. Taking this premise as a key element in the analysis, 96 farms distributed throughout Galicia were considered. All the farms studied have an agricultural area around the farm within a 5 km radius to grow mainly maize and grass, which is subsequently stored in silos and used as cattle feed. This agricultural land is managed by the farmers themselves and was included within the system boundaries. In this way, the processes of grass and maize cultivation were modelled considering the use of machinery, the time of use per hectare, the consumption of diesel and other materials, such as fertilisers or agrochemicals. In some cases, dry grass is also cut for hay production. All farms also use concentrate as cattle feed, to a greater or lesser extent. The composition of this feed is variable for dairy cows, dry cows and heifers, but in general it is composed of 30%, 26%, 17% and 12% maize, soybean, rapeseed and barley respectively, in addition to other minor components.

The size of the different farms is variable; the smallest farm is composed of 13 animals with annual production around 20,000 kg of milk, while the largest farm has 520 animals and produces 3,000,000 kg of milk per year. Although milk is the main objective of the farms, meat production should not be neglected. Thus, the production obtained from old cows slaughtered for meat has been considered a co-product of the farms.

In relation to manure management, due to its high amount of nutrients, it is used as an organic fertiliser in the agricultural land. The direct emissions produced during the storage of the manure and its subsequent application to the land have been estimated. Infrastructure related to the farm has not been included, as it has an impact that can be considered insignificant throughout its useful life (Castanheira et al., 2010; de Léis et al., 2015). However, the manufacture of tractors and implements used in crops has been computed within the production of on-farm feed (grass and maize). The main characteristics of each of the farms evaluated (number of animals and production of milk and meat) can be found in Table D.6 of the Appendix I.

8.2.2. LCA methodology

Life Cycle Assessment is a fundamental element as a tool to determine the impacts and give a global vision of the environmental performance of Galician dairy farms. The environmental performance of dairy farms was analysed, and the main "hotspots" of the process were determined using LCA methodology. The methodology followed the principles established in the ISO 14040 and 14044 standards for CF and 14046 standard for WF.

8.2.2.1. Goal and scope definition

The main objective of this chapter is to determine the evolution of eco-efficiency in milk production in Galicia. To this end, the environmental impacts of a model farm will be analysed to determine which elements are the determining factors in the environmental impact and in the eco-efficiency score. The study was carried out under a "cradle-to-gate" perspective, as shown in Figure 8.1, which is a block diagram of an average farm, representative of the set of installations evaluated, in which the system boundaries are identified, as well as the main elements, inputs and outputs. All relevant processes related to milk production, including energy and material consumption during milking and farming activities were considered such as electricity for machinery use and lighting and different cleaning and chemical agents: detergent, sealer, acid solution or disinfectant. In addition, other inputs considered were the production of feed, paper, plastic for silos, containers for chemical products, refrigerant and the management of the waste produced, and transport activities. Gaseous emissions from enteric fermentation, storage of manure and its application as organic fertiliser for crops were estimated.

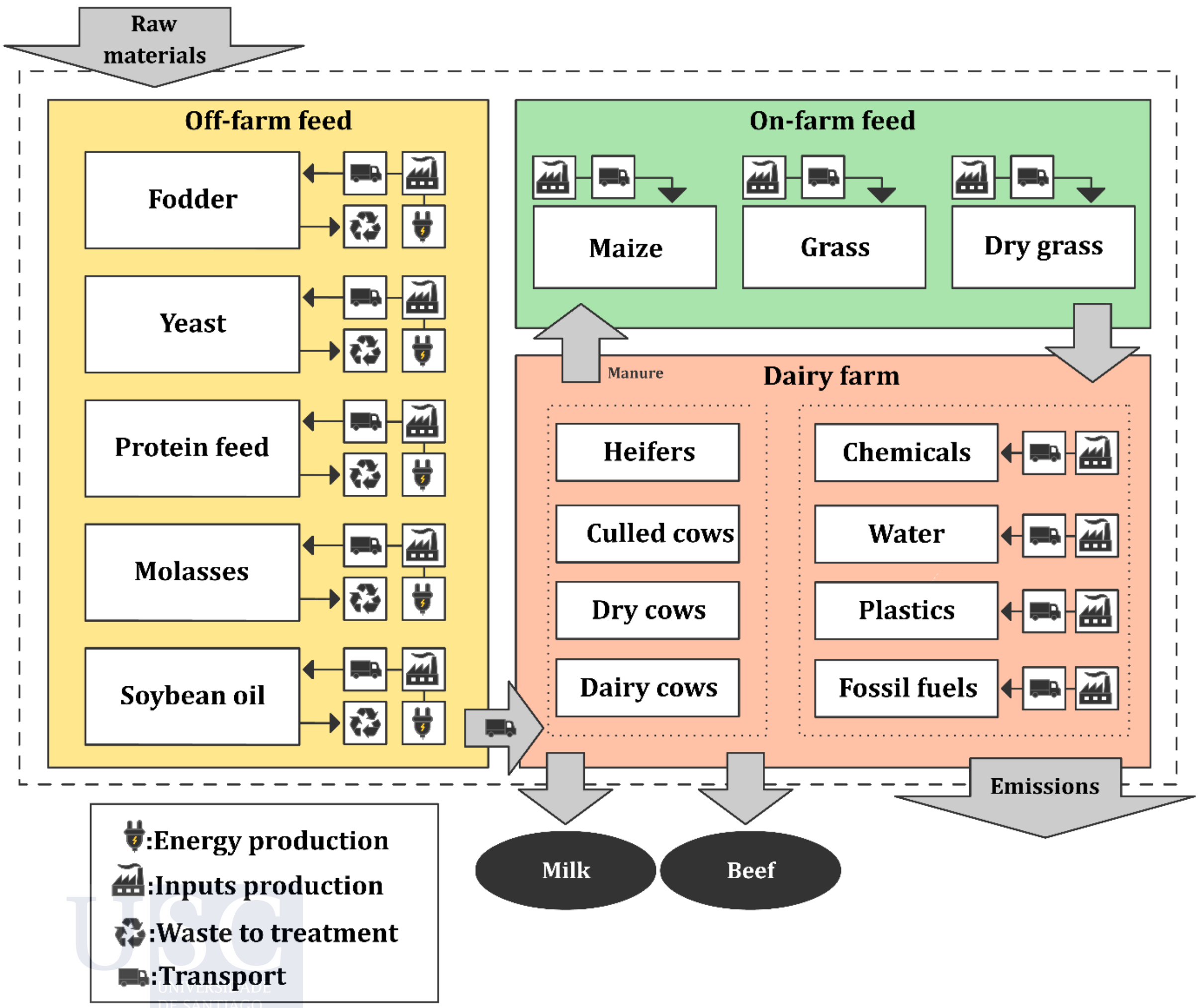


Figure 8.1. System boundaries for the dairy farm model evaluated in this chapter. Legend: T: Transport.

8.2.2.2. Functional Unit and allocation approach

In this chapter, following the IDF (2015) guidelines for the study of agricultural and livestock farms, the quantity of Fat- and Protein- Corrected Milk (FPCM) produced in one year, corresponding to the campaign Apr18/Mar19, has been taken as the Functional Uunit. To convert the raw milk weight to FPCM, Eq. (1) was followed:

$$FPMC\left(\frac{kg}{yr}\right) = P\left(\frac{kg}{yr}\right) * [0.1226 * FC(\%) + 0.0776 * PC(\%) + 0.2534] \quad (1)$$

Where: P: Production; FC: Fat content; PC: Protein content.

In accordance with ISO standards, the allocation of environmental loads should be avoided as much as possible by giving priority to the division of units into subsystems or the expansion of the system boundaries to include other co-production functions. However, since the units assessed are considered to have a multi-output system, allocation is unavoidable. Following the IDF (2015) guidelines, biophysical allocation between the two products produced – milk and meat – has been considered, according to Eq. (2) and Eq. (3):

$$AF_{MILK} = 1 - 6.04 * BMR \quad (2)$$
$$AF_{MEAT} = 1 - AF_{MILK} \quad (3)$$

Where: AF_{MILK} is the allocation factor for milk; BMR is the ratio M_{MEAT}/M_{MILK}; M_{MEAT} is the sum of live weight of all animals sold; and M_{MILK} is the sum of total FPMC.

Table D.2 of the Appendix I shows the economic, mass, and biophysical allocation factors calculated for each farm.

8.2.2.3. Data collection

The quality of the inventory data is a key element in ensuring the accuracy and reproducibility of LCA studies. A consistent environmental assessment requires high quality baseline data. To ensure this data quality, priority should be given to the use of primary sources, minimising as far as possible the use of secondary data from databases and/or similar sources. In this context, most of the information provided in the life cycle inventory was constructed from primary data collected through questionnaires completed by workers. These questionnaires collect information on all relevant aspects of the farm, such as operational characteristics, general data on location and degree of technology used, number of animals in the farm, feed consumption, use of machinery or production of waste, corresponding to the campaign Apr18/Mar19.

The life cycle inventories of the background system (chemicals, fossil fuels, electricity, water...) were taken from the Ecoinvent® database version 3.5, considering the consumption of each element according to the information collected in the questionnaires. In this way, the processes of electricity production (Spanish electricity mix), cleaning agents, fuels, lubricants, fertilisers and pesticides correspond to Ecoinvent inventory data (Althaus et al., 2007; Dones et al., 2007; Hischier, 2007; Spielmann et al., 2007). Regarding livestock feed, two main sources for feed production were considered. Concentrate, which is formulated with the same composition as considered in Iribarren et al. (2011). Thus, a content of 30% maize, 26% soybean, 17% rape meal, 12% barley and 2% wheat were considered, as well as a certain amount of chemicals and additives. The production of the background processes was taken from the Ecoinvent database. Another source of livestock feed is grass and maize grown by farm owners on the surrounding farmland. These productions were modelled individually considering the primary information provided by the farmers. The fuel consumption for the machinery used on the crops was calculated based on the working capacity of the machinery at each stage (h/ha) and the corresponding fuel consumption (L/h). The activities considered in each of the crops have been the typical stages of any cereal crop: organic fertilisation, land clearing, grading, sowing, irrigation, weed control, mineral fertilisation, harvesting and storage (Noya et al., 2015). In addition, direct emissions related to diesel combustion in agricultural machinery during cultivation activities were also estimated from the Ecoinvent database -*Diesel, burned in agricultural machinery*- (Nemecek and Käggi, 2007). In some cases, the own agricultural production does not meet the requirements for feeding livestock. A common practice among Galician farms in this case is to gain surplus production from nearby farms. Given that the production of neighbours can be considered similar, no differentiation was made between the maize or grass produced but feed transport to the farm was taken into account.

In some cases, farmers allow their cattle to graze for a few hours a day. In addition, those farms did not report any material consumption related to those pastures. According to the information provided by the farmers, in any case, these grazing lands do not require any care or consumption of materials. For this reason, no environmental burdens were specifically attributed to grazing land, but animal emissions with grazing feed intake are fully accounted for within the annual per-head emission factors applied.

Finally, emissions of methane (CH_4), dinitrogen monoxide (N_2O) were obtained following the guidelines established by the Intergovernmental Panel on Climate Change (IPCC, 2006). CH_4 emissions from enteric fermentation as well from manure storage and subsequent field application were calculated by combining the Tier 1 method and primary data collected through questionnaires. Direct nitrogen emissions during manure management and soil application were also calculated, following the Tier 1 approximation due to lack of reliable data. Indirect nitrogen emissions in form of NH_3 and NO_3^- were also estimated (Denier van der Gon and Bleeker, 2005). In more detail, Appendix II lists the procedures followed in accordance with the IPCC guidelines for the estimation of gaseous emissions.

Table 8.1. Life cycle inventory data per functional unit (the quantity of FPCM produced during the campaign Apr18/Mar19)

Inputs from Environment			
Raw materials	**L**	**Land**	**ha**
Water	3,110,141	Crops	48
Inputs from Technosphere			
Animal feed	**kg**	**Crops**	**ha**
Concentrate dairy cow	249,363	Maize	21
Concentrate dry cow	6,507	Grass	27
Concentrate heifer	32,422		**kg**
Straw	62,279	Seeds	1,541
Cleaning agents	**L**	**Plastics**	**kg**
Detergent	2334	Silage plastic	807
Acid solution	98	Bottles	137
Disinfectant	112	**Fossil fuels**	**L**
	kg	Lubricant oil	60
Kraft paper	122		**kg**
Sealer	237	Diesel	2,678
Chemicals	**L**	**Energy**	**kWh**
Refrigerant	1	Electricity	27,645
Pesticide	68		**kg**
	kg	Butane	26
Mineral fertiliser	18,29	**Transport**	**t·km**
Calcium carbonate	26,763	Lorry	28,829
Outputs to Environment			
Air emissions	**kg**	**Water emissions**	**kg**
CH_4-enteric fermentation	10,000	NO_3^--manure management	408
CH_4-manure management	2,100	NO_3^--soil management	8756
N_2O-manure management	29		
NH_3-manure management	1,006		
N_2O-soil management	576		
NH_3-soil management	2,161		
Outputs to Technosphere			
Products	**kg**	**Waste to treatment**	**kg**
FPCM	654,441	Plastics to recycling	944
Beef	3,514.30	Paper to recycling	124
Co-products	**m³**	Municipal Solid Waste	201
Manure	2,686		**m³**
		Wastewater	759

8.2.2.4. Life cycle inventory

It is important to highlight the large volume of data handled in this chapter, corresponding to 96 farms. The inventories were classified according to farm size and total milk production. Thus, small farms with a production below 400 m^3 and around 50 livestock head, medium farms between 400 and 1,000 m^3 and 120 livestock heads, and large farms for production above 1,000 m^3 and over 250 livestock heads. In this chapter, the impacts of the life cycle of a simulated farm were evaluated in detail (Table 8.1). This simulated farm corresponds to an average farm of all farms included in the medium size. Medium size farms were chosen for this purpose due to this size is the most numerous within the sample evaluated. However, this life cycle environmental impact analysis was carried out for each of the 96 farms evaluated.

8.2.2.5. Impact assessment

The selected assessment method for the calculation of the environmental impacts of the system was the ReCiPe Midpoint (H) (Huijbregts et al., 2016). In particular, the impact assessment step followed the guidelines established in the ISO standards (ISO 14040, 14044, 14046). ISO 14046 states that, to calculate the water footprint of the system, an environmental study based on ISO 14040 and ISO 14044 standards must be carried out and, in the impact stage, categories related to water consumption must be analysed. Therefore, the environmental results have been presented in terms of Global Warming and Water Consumption impact categories for the estimation of the CF and WF indicators, respectively. The software SimaPro 9.0 was used for the computational implementation of the inventories (PRé Consultants, 2017).

8.2.3. Description and selection of DEA methodology

Based on different models described in the DEA methodology, three of the most used ones were tested for the available dataset: Slacks-Based Measure (SBM), Charnes-Cooper-Rhodes (CCR) and Epsilon-Based Measure (EBM). Finally, a slacks-based measure of efficiency (SBM) model was selected. This model is the most widely used as it allows efficiency scores to be calculated independently of the units of measurement used for the set of inputs and outputs (Tone, 2011). Another feature of this model is that it follows a non-radial approach, assuming conditions of convexity and scalability to obtain the efficient production frontier (Lozano and Gutiérrez, 2011). In addition, the SBM model provides targets to reduce inputs and/or maximise outputs based on the difference with the efficient production frontier established by the model (Lijó et al., 2017). The specific DEA model used in this work was an input-oriented SBM model with constant returns to scale (SBM-I_CRS). The same model as that used by Iribarren et al. (2011) was chosen in order to establish a consistent methodological basis on which to compare the results obtained and establish a time trend. The computational implementation of the DEA matrix in the SBM-I model was performed through the DEA-solver Pro software (Cooper et al., 2007).

8.2.4. LCA + DEA framework

In this chapter, the five-step LCA + DEA method (Vázquez-Rowe et al., 2012) was selected to assess the eco-efficiency of 96 dairy farms, allocating each farm as one DMU. Is important to note that DEA and LCA input/output elements are not the same. The limits of the LCA are broader than the considered in the DEA. Thus, the selection of the elements included in the DEA was based on the importance in the environmental profile. The DEA matrix was completed in order of priority, from those with the greatest impact on the environmental profile to the elements with least influence. A reasonable number of inputs and outputs have been taken into account to allow convergence of the model taking into account the total number of DMUs analysed. The DEA matrix was composed of 7 inputs: i) concentrate (kg), ii) grass silage (kWh), iii) maize silage (kg), iv) electricity (kWh), v) diesel (kg), vi) silage plastic (kg) and vii) water (m^3); and 5 outputs, four of them undesirable and one product: i) CH_4 (kg), ii) N_2O (kg), iii) NH_3 (kg), iv) wastewater (m^3) and v) raw milk (m^3). It is important to note that the direct emissions and the wastewater have been modelled as inputs (Lozano et al., 2009). The complete DEA matrix is shown in Section S4 of the supplementary material.

8.3. RESULTS AND DISCUSSION

8.3.1. Carbon and water footprint of an average medium-size dairy farm

Figure 8.2 shows the distribution of the different elements that contribute to the carbon and water footprints associated with the operation of a dairy farm. The carbon footprint is 1.33 kg CO_2 per kg of FPCM, while the water footprint is 52.5 L per kg of FPCM. To facilitate analysis, some of the inputs were grouped into global elements:

Waste treatment: This category includes both the treatment of solid waste produced on the farm and the treatment of the wastewater generated. Solid waste includes plastic packaging, paper and cardboard waste and municipal solid waste.

Fossil fuels: It includes the production of diesel, lubricating oil and butane. It is important to note that the diesel quantified in this category is different from that used for crops, which is considered in animal feed category. The diesel considered in this category is used for non-feed related activities, such as mixing operations or additional machinery.

On-farm emissions: This element is composed of direct emissions of CH_4, N_2O, NH_3 and NO_3^- directly derived from enteric fermentation, slurry management and soil application. This category also included emissions derived from diesel consumption in different operations than feeding. It is important to differentiate the environmental impacts from production and combustion of diesel. Environmental burdens of diesel production and combustion are quantified in animal feed or fossil fuels categories, depending on diesel use.

Others: It includes the rest of the elements inventoried on the farm that are not included in another category, highlighting the production and use of detergent, acid solution, disinfectant, sealant, plastics, refrigerants, etc.

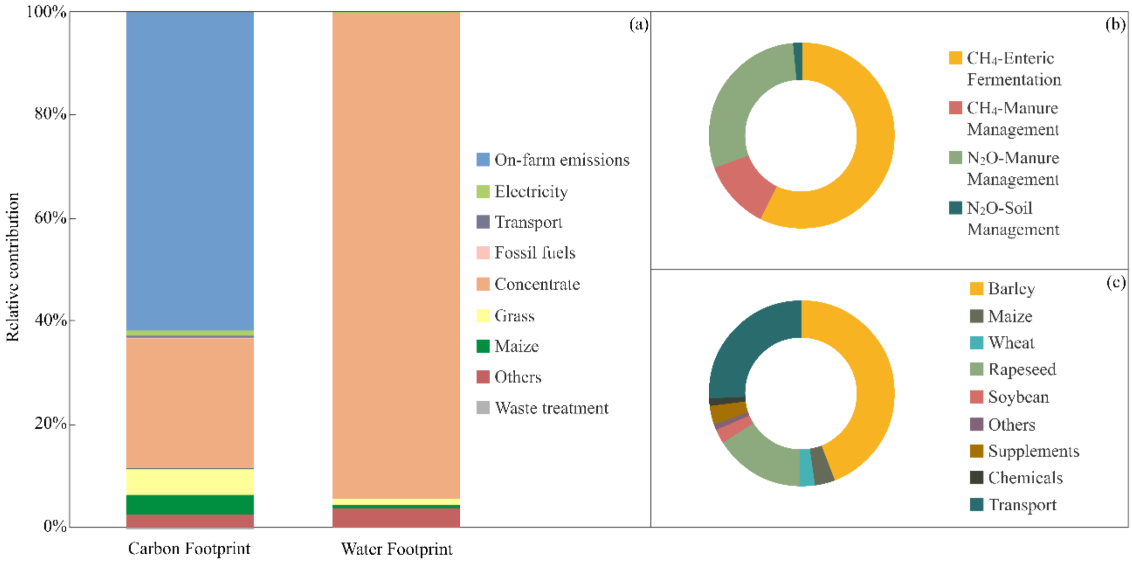


Figure 8.2. Contribution of the most relevant processes in milk production. (a) Environmental profile and distribution of impacts in terms of carbon and water footprint; (b) breakdown of carbon footprint of on-farm emissions and (c) breakdown of water footprint of the concentrate.

Most of the contribution of GHG emissions (64.9%) was linked to on-farm emissions, mainly CH_4 and N_2O, from enteric fermentation and manure management (Figure 8.2.b). In fact, the contribution of enteric fermentation, manure management and feed production stand out in the environmental profile of milk production. This result is in line with other previously published results, which establish these same elements as those with the highest environmental impact in the dairy industry (Famiglietti et al., 2019; Pirlo and Lolli, 2019; Vida and Tedesco, 2017). Other previous studies obtained similar carbon footprint values to those obtained in this chapter, despite small differences in the system boundaries, the allocation factors and the inventory data used. Thus, Noya et al. (2018) obtained a value of 1.32 kg CO_2 eq per kg of FPCM for a similar sized farm located in Catalonia. Similar values were found in a study conducted in the Netherlands, with values of about 1.4 kg CO_2 per kg of FPCM (Thomassen et al., 2008). However, the CF of this farm was higher than the results of 1.02 kg CO_2 eq per kg of FPCM reported by Aguirre-Villegas et al. (2015) or 1.11 kg CO_2 eq per kg FPCM reported by Vida and Tedesco (2017). These studies, despite the subtle differences in the data inventory used, have in common the use of economic or biologic allocation between milk and meat production.

While other studies using other types of allocation obtained significantly different values, de Léis et al. (2015) reported values of 0.78 kg CO_2 eq per kg of Energy Corrected Milk (ECM) using mass allocation while Castanheira et al. (2010) obtained as result 0.72 kg CO_2 eq per kg of raw milk eq with economic-allocation. These different results from different LCA studies can be compared with caution due to the differences between the specific methodologies and assumptions used, although the general principles may be common (Mc Geough et al., 2012). Most of the studies consulted use as FU the production of a certain amount (usually 1 kg) of FPCM, so is possible to carry out direct comparison with most of the studies.

In terms of water footprint, as observed in Figure 8.2.a, the impact is practically focused on feed production (90.7%), which is logical since this element encompasses the production of different crops for animal feed (barley, soybean, maize or rapeseed). This relative contribution is in line with a previous study on the calculation of the water footprint in a dairy farm in Catalonia (Noya et al., 2018), in which feed production was found to account for 99% of the total water footprint.

However, comparing the water footprint is an extremely complex task, as there is no standardised method, as there is for the carbon footprint. Although in Noya et al. (2018), the contribution of feed is similar, the water footprint was quantified according to the Water Footprint Network (WFN), which is a completely different methodology to ISO 14046, so the two absolute values cannot be compared. A similar case was reported in Payen et al. (2018), which analysed two farms located in different regions of New Zealand. A system very similar to this chapter was established, as it included the production of cereals and crops for animal feed, the production of different materials such as fertilisers, pesticides, fuels, etc. However, the abovementioned manuscript reported values of 726 and 537 L per kg FPCM, for the 53 L estimated here. The difference lies mainly in the different methods used, since Payen et al. (2018) use the Available Water Remaining (AWaRe) methodology.

Figure 8.2.c also shows the breakdown of water footprint elements. Most of the environmental impact (70%) comes from the cultivation of agricultural products (mainly barley, maize, wheat, rapeseed and soybean). However, it is the barley crop that has the greatest impact on this indicator, mainly because it has a high irrigation rate (0.75 m^3 per kg product) and because it is the majority component of feed within the agricultural products. While the irrigation rate of wheat is similar (0.71 m^3/kg), the proportion in feed is much lower, and the irrigation in maize is practically negligible (only 0.05 m^3/kg). Another remarkable element is the transport of raw materials (mainly those same agricultural products), by transoceanic freight ship. This fact demonstrates the need for a local feed supply that avoids the massive transport of raw materials and products.

8.3.2. Environmental characterisation of dairy farms

The environmental results obtained for the complete set of farms evaluated are depicted in Figure 8.3. The results are highly variable, ranging from 0.9 to 3.71 kg CO_2 eq per kg FPCM in the case of carbon footprint and from 18.4 to 96.7 L per kg FPCM in terms of water footprint. The average carbon footprint of the complete sample was 1.6 kg CO_2 eq per kg FPCM, a relatively high value, since the DMUs with the worst environmental results were included within the set. The results obtained for DMUs 85, 95 and 64 are noteworthy, with CF values of 3.71, 3.23 and 2.78 kg CO_2 eq per kg FPCM, respectively. The case of DMU 85 is remarkable since it is a farm with certified organic production that does not use concentrate for animal feed. However, the carbon footprint presents poor results when put in perspective with a low milk production. The average CF result is within the range of 1.1-1.7 kg CO_2 eq per kg of milk quantified in Baldini et al. (2018) and Famiglietti et al. (2019). Once again, the high variability in the determination of the environmental impacts of this productive activity is evident.

Regarding water footprint results, DMUs 104 and 40 stand out with 96.74 and 91.67 L per kg of FPCM, respectively. These results can be linked to concentrate consumption, which is a key factor in the environmental impact of dairy farms in terms of their water footprint. On the opposite, DMUs 70, 85 and 98 can be highlighted for their low water footprint. In fact, these three farms have crop/concentrate feed ratio over 86%, reaching 100% in the DMU 85. Moreover, if a ratio of concentrate/m^3 milk produced is calculated, these DMUs present the lowest values, always below 260 kg of forage per m^3 of raw milk, while the average for the entire sample is 435 kg of forage per m^3 of raw milk. As can be observed in Figure 2.3, there is no clear relationship between the two indicators used. CF mainly depends on direct emissions, which are related to the livestock and manure produced, while WF depends on 90% of the consumption of feed.

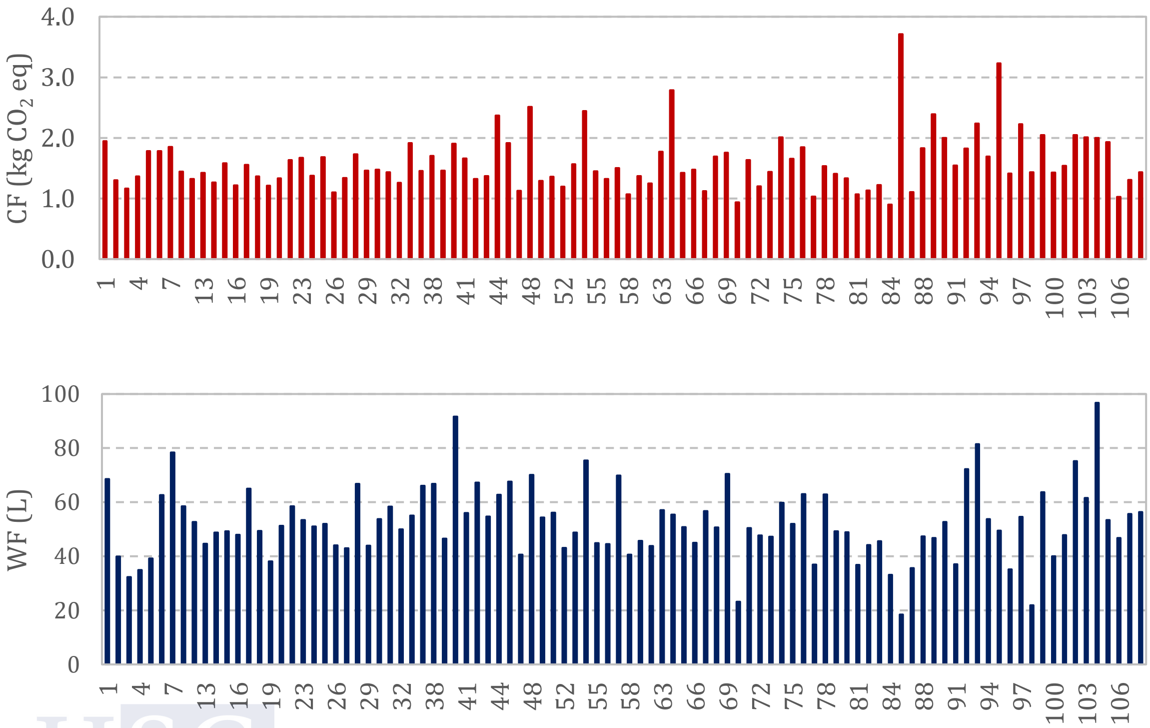


Figure 8.3. Carbon footprint (top) and water footprint (below) per kg of FPCM produced across the sample dairy farms.

8.3.3. DEA computation and efficiency scores

In order to compute the efficiency scores and the operational benchmarks, the DEA matrix (Table D.3 of Appendix I) was implemented in the optimisation model. Table 8.2 presents the efficiency scores computed for the dairy farms. Table D.4 of Appendix I presents the target reduction percentages relative to original values for all the inputs considered in the analysis.

The results show that this methodology is suitable for identifying the link between the operational and environmental performance of multiple similar units. Of all the farms evaluated, just 21 of 96 dairy farms proved to be fully efficient (Φ=1). In fact, the efficiency ratio can be considered acceptable, an average efficiency of 0.58 is achieved in the analysed sample, while only 27 farms present efficiency values below 0.4. For those inefficient farms (Φ<1), important reduction targets are proposed. Thus, average reductions that range from a minimum of 13.6% in maize silage consumption to 53.7% in silage plastic are achieved. If the operational reductions proposed by the model for each of the inputs are considered as the maximum potential for input reduction that can be achieved in milk production, the sample of farms evaluated has a greater margin for improvement than other agricultural and livestock systems previously evaluated (Lozano et al., 2010; Vázquez-Rowe et al., 2012).

Table 8.2. Efficiency scores (Φ) for the sample of dairy farms

DMU	Φ	DMU	Φ	DMU	Φ	DMU	Φ	DMU	Φ	DMU	Φ
1	0.33	20	0.58	40	0.54	58	1	76	0.33	93	0.29
2	0.68	22	0.70	41	1	59	0.53	77	0.67	94	0.22
3	0.43	23	0.78	42	0.66	62	0.51	78	0.49	95	0.12
4	0.35	24	1	43	1	63	1	79	0.61	96	0.41
5	0.46	25	0.62	44	0.31	64	1	80	0.50	97	0.24
6	0.42	26	0.94	45	0.33	65	0.44	81	1	98	1
7	0.31	27	0.64	46	1	66	0.40	82	0.57	99	0.25
8	0.44	28	0.47	48	0.14	67	1	83	1	100	0.65
12	0.61	29	0.67	50	0.64	68	1	84	1	101	0.57
13	0.51	30	0.65	51	0.35	69	0.27	85	1	102	0.28
14	0.60	31	1	52	1	70	1	86	0.61	103	0.27
15	0.46	32	0.51	53	0.38	71	0.49	88	0.40	104	0.25
16	0.60	34	0.31	54	0.23	72	0.72	89	0.39	105	0.37
17	0.56	35	0.49	55	0.25	73	0.59	90	0.22	106	1
18	1	38	0.39	56	0.45	74	0.24	91	0.43	107	0.58
19	1	39	0.28	57	0.45	75	0.49	92	1	108	0.49

8.3.4. Environmental impact of virtual DMUs

The last stage of the methodology is to analyse the reduction targets set by the SBM-I model, which involves modifying the life cycle inventories of inefficient farms. In this way, a relationship can be established between inefficient operations and environmental impacts by comparing the environmental profile before and after considering the recommendations for reducing impacts (current and virtual dairy farms), depicted in Figure 8.4.

All environmental profiles of farms with an efficiency value below 1 have improved by applying the DEA recommendations. The average percentage of carbon footprint reduction is around 49% in the set under study. However, it can reach maximum reduction values of 77% in the case of DMU 95. This farm is characterised by a very low efficiency value (0.11), so reductions in material consumption are expected to be significant and, consequently, also a reduction in environmental impacts. This DMU is characterised by a very traditional farm, with a low degree of modernisation, few heads of cattle and, therefore, low milk production. In fact, it is the farm with the lowest productivity, barely reaching 2.8 m^3 of milk production per cow, while the average for the rest of the sample analysed is above 9.3 m^3 per cow.

The average production value is within the expected range 8,000-11,000 L/cow per year according to the last National dairy report carried out by the Ministry of Agriculture and Fisheries, Food and Environment in 2017 (MAPAMA, 2017). In detail, the DMU 95 does not consume concentrate, since the cattle are fed exclusively on the grass of the surrounding land, which means that direct emissions are the greatest "hot spots".

The reduction of environmental impacts is more evident in terms of WF with an average reduction around 55% as this element is 90% dependent on the environmental impacts of feed production. Reductions in this element have a direct positive impact on the environmental performance of the farm. Thus, observing the recommended percentages of reduction in, the farms with the highest reduction in concentrate are DMUs 104, 48 and 8, with 83.8%, 74.2% and 67.6% respectively, which imply the greatest reduction in their water footprint: 81.7, 73.8 and 73.8% respectively.

8.3.5. Eco-efficiency evaluation over time

Given that the sample analysed comprises a wide range of livestock farms of different sizes, it is interesting to establish the relationship between farm size and the value of operational efficiency, as reported in Iribarren et al. (2011). Figure 8.5 shows the efficiency scores against farm size in terms of total raw milk production for 2011 (grey square) and 2019 (orange circle). There is an apparent correlation between farm size and its efficiency score.

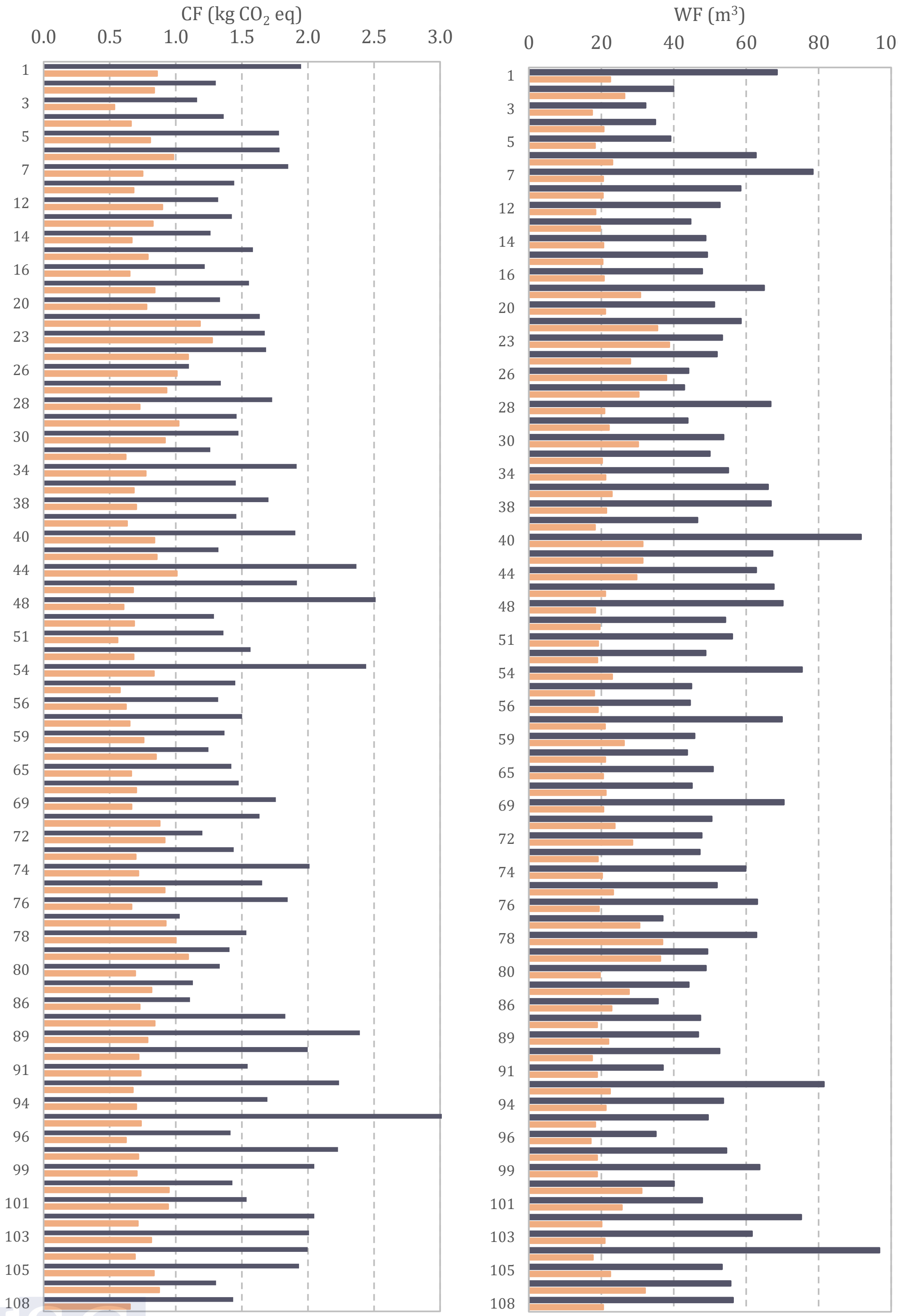


Figure 8.4. Environmental impacts in terms of carbon footprint (left) and water footprint (right) per kg FPCM for real (black) and virtual (orange) farms.

As shown in Table 8.2 and Figure 8.5, almost 22% of dairy farms (21 or 96) were considered efficient (Φ=1). This value is lower than that obtained by Iribarren et al. (2011), where 31 out of 72 farms were considered efficient. This difference can be attributed to the fact that Iribarren et al. (2011) considered fewer elements in the DEA analysis when handling data from a smaller sample. In both studies, two main groups were distinguished in terms of feeding system. On the one hand, import-based feeding refers to feed products that are produced abroad and then imported into the farm (mainly concentrate) and on the other hand, farm-based feeding, where the main feed is composed by maize and grass cultivated in the farm. No relationship was found in any case, only that a high percentage of efficient farms (22 out of 31) used maize and concentrate as the two main feed products in 2011. It is remarkable the progression of Galician dairy farms towards local and sustainable food, consisting mainly in feeding on the farm itself and following the principles of the circular economy. Thus, the sample of farms evaluated in this chapter presents an average percentage of on-farm feeding above 80% and only 6 farms present a percentage below 70%.

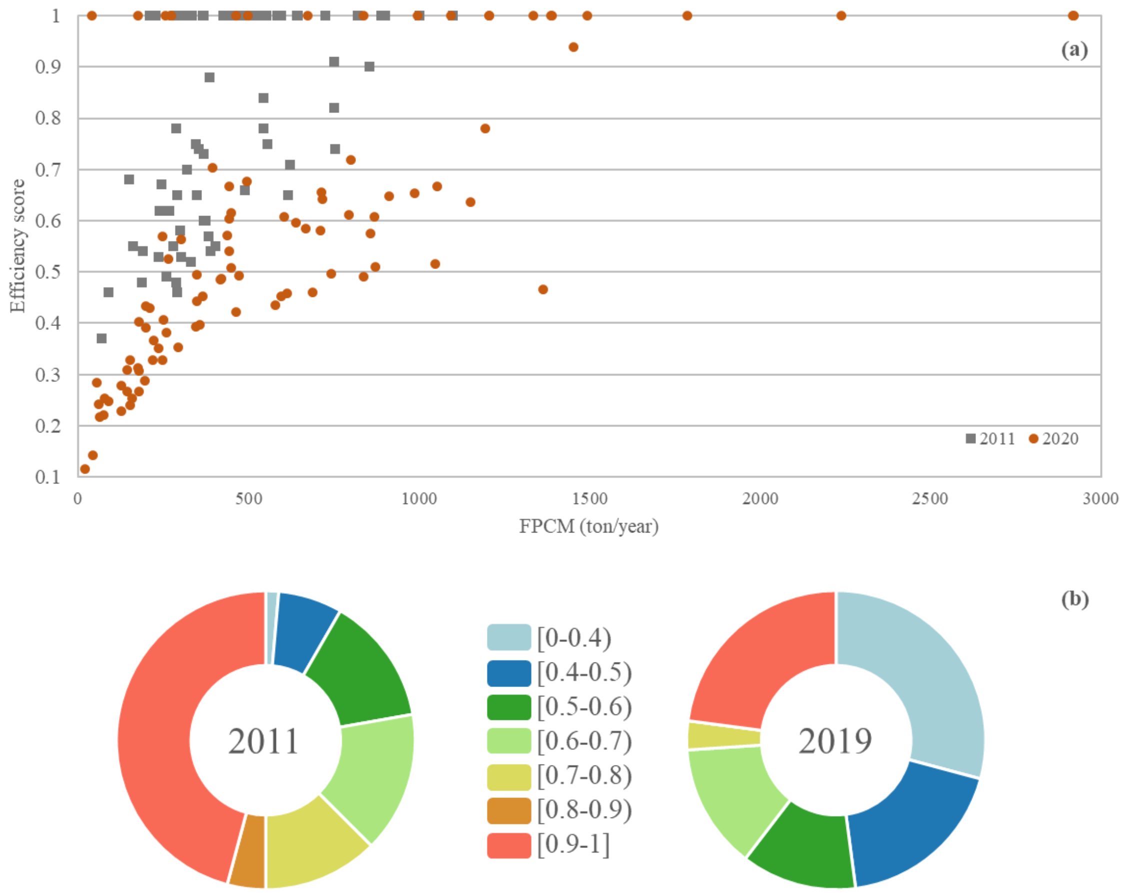


Figure 8.5. (a): Efficiency scores against raw milk production for 2011 (grey square) and 2019 (orange circle). (b): Relative distribution of dairy farms according to their efficiency score.

In addition, the overall decrease in the average eco-efficiency of inefficient farms in 2019 should be noted. In addition, the positive correlation between farm size and operational efficiency observed in a previous study in 2011 is even more evident in this chapter. This fact has been made possible by an expansion in the total number of farms assessed and their size, pointing out that the Galician dairy sector needs to continue carrying out improvement actions that lead to better operational and environmental performance.

8.4. CONCLUSIONS

The life cycle impact has been evaluated based on the carbon footprint and water footprint of milk production in 96 livestock farms distributed throughout Galicia. Feed production (mainly concentrate and on-farm maize and grass), as well as direct CH_4 and N_2O emissions have been identified as the critical processes of the system. The carbon footprint for an average medium-sized farm has been estimated at 1.33 kg CO_2 per kg of FPCM, a value that is within the range found in similar studies. The water footprint according to ISO 14046 is 52.5 L per kg FPCM.

However, the range of environmental results found is very wide, which demonstrates the high variability of the operational characteristics of this type of production system. The eco-efficiency analysis carried out has shown that of the 96 farms evaluated, 21 are currently fully efficient. This analysis has also made it possible to identify actions that inefficient farms should carry out. Thus, reductions in the consumption of silage plastic (-53.7%) and the production of wastewater (-49.9%) stand out as the principal elements to improve the overall efficiency of the analysed farms. It can be stated that the eco-efficiency of milk production has decreased over the last decade, going from an average of 0.64 in 2011 to an average of 0.58 in 2019. This fact marks the path that the Galician dairy sector must follow, in which it must reduce its environmental impacts so that the production of a basic foodstuff such as milk pursue the compliance of international standards, especially in terms of environmental certification.

8.5. REFERENCES

Aguirre-Villegas, H.A., Passos-Fonseca, T.H., Reinemann, D.J., Armentano, L.E., Wattiaux, M.A., Cabrera, V.E., Norman, J.M., Larson, R., 2015. Green cheese: Partial life cycle assessment of greenhouse gas emissions and energy intensity of integrated dairy production and bioenergy systems. J. Dairy Sci. 98, 1571–1592. https://doi.org/10.3168/jds.2014-8850

Aguirre-Villegas, H.A., Passos-Fonseca, T.H., Reinemann, D.J., Larson, R., 2017. Grazing intensity affects the environmental impact of dairy systems. J. Dairy Sci. 100, 6804–6821. https://doi.org/10.3168/jds.2016-12325

Althaus, H.J., Chudacoff, M., Hischier, R., Junbluth, N., Osses, M., Primas, A., 2007. Life Cycle Inventories of Chemicals. Ecoinvent report No. 8, v2.0 EMPA.

Baldini, C., Bava, L., Zucali, M., Guarino, M., 2018. Milk production Life Cycle Assessment: A comparison between estimated and measured emission inventory for manure handling. Sci. Total Environ. 625, 209–219. https://doi.org/10.1016/j.scitotenv.2017.12.261

Baldini, M., Da Borso, F., Rossi, A., Taverna, M., Bovolenta, S., Piasentier, E., Corazzin, M., 2020. Environmental sustainability assessment of dairy farms rearing the Italian simmental dual-purpose breed. Animals 10. https://doi.org/10.3390/ani10020296

Berton, M., Bittante, G., Zendri, F., Ramanzin, M., Schiavon, S., Sturaro, E., 2020. Environmental impact and efficiency of use of resources of different mountain dairy farming systems. Agric. Syst. 181. https://doi.org/10.1016/j.agsy.2020.102806

Castanheira, É.G., Dias, A.C., Arroja, L., Amaro, R., 2010. The environmental performance of milk production on a typical Portuguese dairy farm. Agric. Syst. 103, 498–507. https://doi.org/10.1016/j.agsy.2010.05.004

Cooper, W.W., Seiford, L.M., Tone, K., 2007. Data Envelopment Analysis: A compresensive text with models, applications, references and DEA-Solver software. Springer, New York.

de Léis, C.M., Cherubini, E., Ruviaro, C.F., da Silva, V.P., Lampert, V. do N., Spies, A., Soares, S.R., 2015. Carbon footprint of milk production in Brazil: a comparative case study. Int. J. Life Cycle Assess. 20, 46–60. https://doi.org/10.1007/s11367-014-0813-3

Denier van der Gon, H., Bleeker, A., 2005. Indirect N_2O emission due to atmospheric N deposition for the Netherlands. Atmos. Environ. 39, 5827–5838. https://doi.org/10.1016/j.atmosenv.2005.06.019

Djekic, I., Petrovic, J., Božičković, A., Djordjevic, V., Tomasevic, I., 2019. Main environmental impacts associated with production and consumption of milk and yogurt in Serbia – Monte Carlo approach. Sci. Total Environ. 695. https://doi.org/10.1016/j.scitotenv.2019.133917

Dones, R., Bauer, C., Bolliger, R., Burger, B., Emmenegger, M., Frischknecht, R., Heck, T., Jungbluth, N., Röder, A., Tuchschmid, M., 2007. Life Cycle Inventories of Energy Systems: Results for Current Systems in Switzerland and other UCTE Countries. Final report Ecoinvent No. 5.

Egas, D., Ponsá, S., Colon, J., 2020. CalcPEFDairy: A Product Environmental Footprint compliant tool for a tailored assessment of raw milk and dairy products. J. Environ. Manage. 260. https://doi.org/10.1016/j.jenvman.2019.110049

Escribano, M., Elghannam, A., Mesias, F.J., 2020. Dairy sheep farms in semi-arid rangelands: A carbon footprint dilemma between intensification and land-based grazing. Land use policy 95. https://doi.org/10.1016/j.landusepol.2020.104600

Eurostat, 2020. Greenhouse gas emissions from agriculture. https://ec.europa.eu/eurostat/tgm/table.do?tab=table&init=1&language=en&pcod

e=tai08&plugin=1 (accessed 2.7.20).

Eurostat, 2019. Agriculture, forestry and fishery statistics - 2019 edition. https://ec.europa.eu/eurostat/documents/3217494/10317767/KS-FK-19-001-EN-N.pdf/742d3fd2-961e-68c1-47d0-11cf30b11489 (accessed 2.7.20).

Famiglietti, J., Guerci, M., Proserpio, C., Ravaglia, P., Motta, M., 2019. Development and testing of the Product Environmental Footprint Milk Tool: A comprehensive LCA tool for dairy products. Sci. Total Environ. 648, 1614–1626. https://doi.org/10.1016/j.scitotenv.2018.08.142

FEGA, 2019. Declaraciones obligatorias del sector vacuno de leche. https://www.fega.es/datos-campanas-clasificadas-por-actividad/actividad/declaracionesleche (accessed 8.6.20).

Finnegan, W., Goggins, J., Clifford, E., Zhan, X., 2017. Global warming potential associated with dairy products in the Republic of Ireland. J. Clean. Prod. 163, 262–273. https://doi.org/10.1016/j.jclepro.2015.08.025

González-García, S., Castanheira, É.G., Dias, A.C., Arroja, L., 2013. Using Life Cycle Assessment methodology to assess UHT milk production in Portugal. Sci. Total Environ. 442, 225–234. https://doi.org/10.1016/j.scitotenv.2012.10.035

Grossi, G., Goglio, P., Vitali, A., Williams, A.G., 2019. Livestock and climate change: Impact of livestock on climate and mitigation strategies. Anim. Front. 9, 69–76. https://doi.org/10.1093/af/vfy034

Hischier, R., 2007. Life Cycle Inventories of Packagings and Graphical Papers. Ecoinvent report No. 11.

Horrillo, A., Gaspar, P., Escribano, M., 2020. Organic farming as a strategy to reduce carbon footprint in dehesa agroecosystems: A case study comparing different livestock products. Animals 10. https://doi.org/10.3390/ani10010162

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M.D.M., Hollander, A., Zijp, M., van Zelm, R., 2016. ReCiPe 2016 v1.1. A harmonized life cycle impact assessment method at midpoint and endpoint level. The Netherlands.

IDF, 2015. A common carbon footprint approach for the dairy sector. The IDF guide to standard life cycle assessment methodology.

IPCC, 2006. 2006 IPCC Guidelines for national greenhouse gas inventories. Intergovernmental panel on climate change.

Iribarren, D., Hospido, A., Moreira, M.T., Feijoo, G., 2011. Benchmarking environmental and operational parameters through eco-efficiency criteria for dairy farms. Sci. Total Environ. 409, 1786–1798. https://doi.org/10.1016/j.scitotenv.2011.02.013

ISO, 2014. ISO 14046 - Environmental Management - Water footprint - Principles, Requirements and Guidelines.

ISO, 2006a. ISO 14040 - Environmental Management - Life Cycle Assessment - Principles and Framework.

ISO, 2006b. ISO 14044 - Environmental Management - Life Cycle Assessment - Requirements and Guidelines.

Knudsen, M.T., Dorca-Preda, T., Djomo, S.N., Peña, N., Padel, S., Smith, L.G., Zollitsch, W., Hörtenhuber, S., Hermansen, J.E., 2019. The importance of including soil carbon changes, ecotoxicity and biodiversity impacts in environmental life cycle assessments of organic and conventional milk in Western Europe. J. Clean. Prod. 215, 433–443. https://doi.org/10.1016/j.jclepro.2018.12.273

Laca, Amanda, Gómez, N., Laca, Adriana, Díaz, M., 2020. Overview on GHG emissions of raw milk production and a comparison of milk and cheese carbon footprints of two different systems from northern Spain. Environ. Sci. Pollut. Res. 27, 1650–1666. https://doi.org/10.1007/s11356-019-06857-6

Lijó, L., Lorenzo-Toja, Y., González-García, S., Bacenetti, J., Negri, M., Moreira, M.T., 2017. Eco-efficiency assessment of farm-scaled biogas plants. Bioresour. Technol. 237, 146–155. https://doi.org/10.1016/j.biortech.2017.01.055

Lozano, S., Gutiérrez, E., 2011. Slacks-based measure of efficiency of airports with airplanes delays as undesirable outputs. Comput. Oper. Res. 38, 131–139. https://doi.org/10.1016/j.cor.2010.04.007

Lozano, S., Iribarren, D., Moreira, M.T., Feijoo, G., 2010. Environmental impact efficiency in mussel cultivation. Resour. Conserv. Recycl. 54, 1269–1277. https://doi.org/10.1016/j.resconrec.2010.04.004

Lozano, S., Iribarren, D., Moreira, M.T., Feijoo, G., 2009. The link between operational efficiency and environmental impacts. A joint application of Life Cycle Assessment and Data Envelopment Analysis. Sci. Total Environ. 407, 1744–1754. https://doi.org/10.1016/j.scitotenv.2008.10.062

Lu, Y., Payen, S., Ledgard, S., Luo, J., Ma, L., Zhang, X., 2018. Components of feed affecting water footprint of feedlot dairy farm systems in Northern China. J. Clean. Prod. 183, 208–219. https://doi.org/10.1016/j.jclepro.2018.02.165

MAPA, 2019. Informe de coyuntura del sector vacuno de leche. Subdirección general de productos ganaderos, Dirección genera de producciones y mercados agrarios.

MAPAMA, 2017. Informe Nacional de Vacuno de Leche-2016. Ministerio de Agricultura y Pesca, Alimentación y Medio Ambiente.

Mc Geough, E.J., Little, S.M., Janzen, H.H., McAllister, T.A., McGinn, S.M., Beauchemin, K.A., 2012. Life-cycle assessment of greenhouse gas emissions from dairy production in Eastern Canada: A case study. J. Dairy Sci. 95, 5164–5175. https://doi.org/10.3168/jds.2011-5229

McEldowney, J.F., 2021. Climate change and the law, The Impacts of Climate Change. Elsevier Inc. https://doi.org/10.1016/b978-0-12-822373-4.00018-5

Mekonnen, M.M., Neale, C.M.U., Ray, C., Erickson, G.E., Hoekstra, A.Y., 2019. Water productivity in meat and milk production in the US from 1960 to 2016. Environ. Int. 132. https://doi.org/10.1016/j.envint.2019.105084

Morais, T.G., Teixeira, R.F.M., Rodrigues, N.R., Domingos, T., 2018. Carbon footprint of milk from pasture-based dairy farms in Azores, Portugal. Sustainability 10, 1–22. https://doi.org/10.3390/su10103658

Nemecek, T., Käggi, T., 2007. Life Cycle Inventories of Agricultural Production Systems. Final Report Ecoinvent v2.0 No. 15a.

Noya, I., González-García, S., Bacenetti, J., Arroja, L., Moreira, M.T., 2015. Comparative life cycle assessment of three representative feed cereals production in the Po Valley (Italy). J. Clean. Prod. 99, 250–265. https://doi.org/10.1016/j.jclepro.2015.03.001

Noya, I., González-García, S., Berzosa, J., Baucells, F., Feijoo, G., Moreira, M.T., 2018. Environmental and water sustainability of milk production in Northeast Spain. Sci. Total Environ. 616–617, 1317–1329. https://doi.org/10.1016/j.scitotenv.2017.10.186

Payen, S., Falconer, S., Ledgard, S.F., 2018. Water scarcity footprint of dairy milk production in New Zealand – A comparison of methods and spatio-temporal resolution. Sci. Total Environ. 639, 504–515. https://doi.org/10.1016/j.scitotenv.2018.05.125

Pirlo, G., Lolli, S., 2019. Environmental impact of milk production from samples of organic and conventional farms in Lombardy (Italy). J. Clean. Prod. 211, 962–971. https://doi.org/10.1016/j.jclepro.2018.11.070

PRé Consultants, 2017. SimaPro Database Manual (No. Methods Library). The Netherlands.

Schmidheiny, S., Stingson, B., Lehni, M., 2000. Eco-efficiency: creating more value with less impact.

Spielmann, M., Dones, R., Bauer, C., 2007. Life Cycle Inventories of Transport Services. Final report Ecoinvent No. 14.

Thomassen, M.A., van Calker, K.J., Smits, M.C.J., Iepema, G.L., de Boer, I.J.M., 2008. Life cycle assessment of conventional and organic milk production in the Netherlands. Agric. Syst. 96, 95–107. https://doi.org/10.1016/j.agsy.2007.06.001

Tone, K., 2011. Slacks-Based measure of efficiency. Int. Ser. Oper. Res. Manag. Sci. 164, 195–209. https://doi.org/10.1007/978-1-4419-6151-8_8

Üçtuğ, F.G., 2019. The Environmental Life Cycle Assessment of Dairy Products. Food Eng. Rev. 11, 104–121. https://doi.org/10.1007/s12393-019-9187-4

Usva, K., Virtanen, E., Hyvärinen, H., Nousiainen, J., Sinkko, T., Kurppa, S., 2019. Applying water scarcity footprint methodologies to milk production in Finland. Int. J. Life Cycle Assess. 24, 351–361. https://doi.org/10.1007/s11367-018-1512-2

Vázquez-Rowe, I., Villanueva-Rey, P., Iribarren, D., Moreira, M.T., 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. https://doi.org/10.1016/j.jclepro.2011.12.039

Vida, E., Tedesco, D.E.A., 2017. The carbon footprint of integrated milk production and renewable energy systems – A case study. Sci. Total Environ. 609, 1286–1294. https://doi.org/10.1016/j.scitotenv.2017.07.271

Wang, X., Ledgard, S., Luo, J., Guo, Y., Zhao, Z., Guo, L., Liu, S., Zhang, N., Duan, X., Ma, L., 2018. Environmental impacts and resource use of milk production on the North China Plain, based on life cycle assessment. Sci. Total Environ. 625, 486–495. https://doi.org/10.1016/j.scitotenv.2017.12.259

Woldegebriel, D., Udo, H., Viets, T., van der Harst, E., Potting, J., 2017. Environmental impact of milk production across an intensification gradient in Ethiopia. Livest. Sci. 206, 28–36. https://doi.org/10.1016/j.livsci.2017.10.005

Chapter 9

Eco-efficiency assessment of shrimp aquaculture production in Mexico

Summary

Globally, human society faces the challenge of providing food to a growing population, at the same time that the effects of climate change and resource depletion must be addressed. Aquaculture allows to ensure a safe supply of different marine species and is a major technological and biological undertaking. Taking into account that in Sonora (Mexico), there are more than 200 aquaculture plants, the analysis of this sector implies a joint and harmonised assessment, considering not only life cycle assessment (LCA), but also data envelopment analysis (DEA). This chapter focuses on the application of LCA + DEA methodology to assess the ecoefficiency of 38 semi-intensive shrimp farms located in the state of Sonora. LCA results showed that feed management and electricity consumption are the main critical points in almost all the impact categories. Further improvement actions were evaluated, the replacement of wheat meal for Dried Distiller Grains with Solubles (DDGS) resulted in environmental impact reductions ranged from 2% to 57%, depending on the impact category. On the other hand, the installation of photovoltaic panels in the area was evaluated, looking for a shift towards a less carbon-intensive energy production. Overall, the implementation of these improvement measures will contribute to increased environmental protection and resource efficiency.

This chapter, under the consent of the publisher, was redrafted after: Cortés, A., Casillas-Hernández, R., Cambeses-Franco, C., Bórquez-López, R., Magallón-Barajas, F., Quadros-Seiffert, W., Feijoo, G., Moreira, M. T. (2021). Eco-efficiency assessment of shrimp aquaculture production in Mexico. Aquaculture, 544, 737145. DOI: https://doi.org/10.1016/j.aquaculture.2021.737145. More information related to this article can be found in Appendix III.

OUTLINE

9.1. INTRODUCTION

Fish consumption per capita is increasing by an average of about 1.5% per year (FAO, 2020). Considering this situation of increasing demand, aquaculture shows strong potential for food security and can be used as a promising alternative to current intensive fishing (Little et al., 2016). In fact, in the latest statistics recorded by FAO, global fish consumption peaked at approximately 171 million tonnes, and aquaculture represented 47% of total production (FAO, 2018). Moreover, current aquaculture is very diverse, with the most produced species ranging from finfish such as carp, rainbow trout or salmon, to all kinds of molluscs and bivalves such as white shrimp, clams or oysters (OECD, 2017). Focusing on shellfish farming, the white shrimp (*Penaeus vannamei*) is the most common, constituting 53% of the total crustaceans produced worldwide. Although Asian countries dominate the production, there is also a contribution from some American countries, with the outstanding share of Mexico (OECD, 2017).

Due to its physical and social characteristics, Mexico has a real potential to be a leader in aquaculture. In 2017, Mexico had its highest historical record of 404,551 tonnes of aquaculture production, with shrimp being one of the main species cultivated with a production of 270,000 tonnes. Most of the aquaculture production for this species is located in two states, Sonora and Sinaloa (Porchas-Cornejo et al., 2018). Shrimps are produced in three models of farming systems: extensive, semi-intensive and intensive. The differences lie in the level of technology applied, the control of physical-chemical and biometric variables, water consumption and the frequency of meal dosage. In recent years, the expansion of aquaculture systems has been accompanied by an intensification of the system and has generated social concerns on the associated sustainability issues. In this context, LCA can be considered an appropriate methodology to evaluate the environmental impacts associated with shrimp farming. Among the different LCA studies of aquaculture production systems for marine products, some focus on the study of the environmental profile of shrimp farming (Henriksson et al., 2015; Järviö, 2018; Medeiros et al., 2017). While DEA can identify those shrimp farms that perform better in terms of benefits and services by reducing resource use and waste generation. In this regard, several reports analysed the technical efficiency performance of aquaculture farms using DEA in Taiwan, Europe and Brazil (Chang et al., 2010; Gutiérrez et al., 2020; Santos et al., 2019). However, there are no previous studies in the literature that combine both methodologies: LCA and DEA, with the aim of evaluating this important sector of the Mexican economy and improving the environmental and operational performance of shrimp farms.

Within this framework, the goal of this chapter is to apply the large amount of data available to carry out an environmental and eco-efficiency assessment of 38 semi-intensive farms located in Sonora using a combined LCA and DEA approach. As in Chapter 8, the environmental and eco-efficiency analyses were conducted to detect critical activities in the environmental profile of the process, identifying operational inefficiencies, setting input reduction objectives and computing the environmental impacts of inefficient practices in shrimp farming in this case. The results of the eco-efficiency analysis will allow a realistic proposal of alternatives to improve environmental performance by identifying those facilities that under similar conditions may act as reference for their peers. This chapter also proposes the definition of a roadmap for more sustainable aquaculture production with a view to future environmental certification. In this regard, taking into account the data used in this chapter, which represent a broad sample of the available data on shrimp production in Mexico, the analysis would not only focus on the assessment of similar facilities in a given geographical area, but also on a clear overview of the sector.

9.2. MATERIALS AND METHODS

9.2.1. System overview

In Mexico, most of the national shrimp production is concentrated in the northwest region, specifically in the states of Sonora and Sinaloa, where semi-intensive farms are the most abundant. Shrimp aquaculture in Mexico started in Sinaloa in the late 1960s in an artisanal way and carried out by fisherman and farmers. It gained momentum as a consequence of the economic stimulus caused by exports to the United States. From 1970 to 1988, shrimp aquaculture underwent technical, operational, administrative and organisational reforms. It was in 1980 when the first shrimp farm operating under the semi-intensive system was built in Sinaloa. From 1988 to 2000, shrimp farming in Mexico grew significantly, driven by government policy. Above all, pronounced growth was observed in the states of Sinaloa and Sonora (Arreola-Lizárraga et al., 2019). The environmental impacts by aquaculture activities in Sonora are well documented, mainly pollution problems produced by the addition of feed and fertilisers to the ponds, triggering a phytoplankton bloom along the coastal areas of wastewater discharge (González et al., 2003). As detailed in Ponce-Palafox et al. (2011), the vegetation associated with shrimp farming in the state of Sonora is mostly semi-desertic, xeromorphic and succulents shrub-dominated, arranged in a matrix, so environmental impacts associated with land use change are not expected to be relevant. For this reason, in this chapter, these impacts were excluded, attention was focused on impacts related to nutrient emissions to water.

These semi-intensive farms are characterised by their diversity in terms of land use and size, while maintaining traditional working conditions (Van et al., 2017). Generally, pond filling operations start in March and the shrimp fattening period lasts until October or November. Feed is applied in daily doses, in amounts that are adjusted according to pond biomass (Casillas-Hernández et al., 2006). The average stocking density is 20-30 individuals/m^2 and adult shrimps reach sizes around 30 g. The criteria for determining the ideal sales weight of shrimp are determined by the production margin, the market price and the production costs (Lucien-Brun, 2016).

In general, the operation of these typical Mexican semi-intensive farms consists of the following phases:

(i) Pond preparation: Operations of ploughing and bottom levelling are carried out. In parallel, applications of quick lime and monitoring of the pond are performed, as well as the application of fertilisers to the soil. After these operations, seawater is pumped in for filling. Energy consumption is mainly attributed to pumping operations.
(ii) Pond operation: The first stage carried out in the prepared ponds is larvae sowing. During this phase priority is given to feeding the post larvae and maintaining water quality. Therefore, maintenance operations such as water exchanges and operations to control environmental variables such as salinity, temperature and dissolved oxygen are carried out.
(iii) Harvesting: This operation is performed by reducing the water level of the pond and using nets to collect the products. Generally, the time of collection is determined by several criteria, based on the shrimp size, market prices or when the water quality decreases, and a complete renewal is necessary.
(iv) Ancillary operations: These actions complement the shrimp farms facilities, such as surveillance booths, warehouses and cellars, offices, and laboratories.

9.2.2. The LCA + DEA framework

The five-step framework (Vázquez-Rowe et al., 2010) has been applied to analyse 38 Mexican shrimp farms. The methodology, as stated in Chapter 2, is structured in 5 steps: i) data collection and construction of life cycle inventory for each DMU; ii) determination of the life cycle environmental impacts of each DMU through the LCA methodology; iii) implementation of the DEA model to obtain the efficiency scores and operational objectives for each DMU. These operational objectives represent reductions in input consumption while maintaining output production; iv) impact assessment of LCI for new virtual DMUs based on the operational reductions established in step 3; v) interpretation of the results obtained, comparison among DMUs and verification of inefficient practices.

9.2.3. LCA methodology

The ISO 14040 and 14044 standards have been used as the basic methodology to carry out environmental assessment. These standards define the LCA phases as: goal and scope definition, inventory analysis, impact assessment and interpretation.

9.2.3.1. Goal and scope definition

The main objective of this chapter is to analyse the significant environmental burdens of shrimp aquaculture and link them to operational inefficiencies. A secondary objective is to identify operational improvement actions to reach, totally or partially, the proposed theoretical goals. The Functional Unit selected for the study was the production of one tonne of commercial size shrimp. This FU was the reference unit to which all inputs and outputs are referred. The aquaculture system evaluated was divided into three main subsystems. Figure 9.1 represents the system boundaries considered for shrimp production by a semi-intensive aquaculture system.

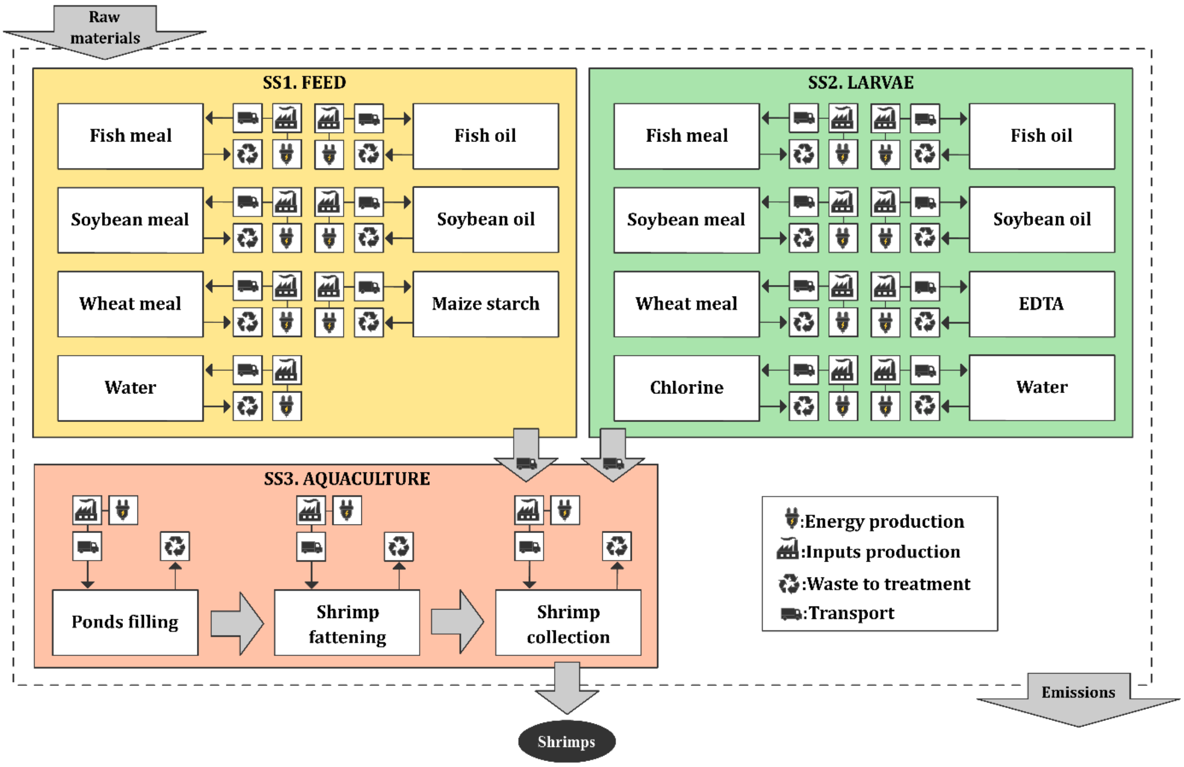


Figure 9.1. Description of the shrimp aquaculture production system in Mexico

SS1. Feed: This subsystem covers the processing and transport of raw materials to produce shrimp feed. This feed is formulated with 35% protein content and is mainly composed of fish meal, fish oil, soybean meal, wheat meal, maize-starch, and soybean oil.

SS2. Larvae includes the production of post-larvae in laboratories distributed throughout the state. This system includes chemicals for maintenance and cleaning operations of the ponds, energy consumption to achieve adequate aeration conditions and the feed used for larvae farming. Larvae feed is similar to that used for adult shrimp, and consists of fish meal, fish oil, soybean meal, wheat meal, maize-starch and soybean oil.

SS3. Aquaculture: This subsystem includes both the transport of feed and the shrimp production operations themselves (filling and preparation of ponds, fattening and harvesting of shrimp). Maintenance operations, such water exchange and the monitoring of environmental parameters are carried out during this phase to maintain water quality. The transport of shrimps for processing, packaging and market operations associated were excluded from the study due to the lack of reliable data.

9.2.3.2. Data collection and life cycle inventory

A total of 38 Mexican shrimp farms were inventoried in this chapter. The approximate location of the farms evaluated along the southern coast of Sonora is shown in Figure 9.2.

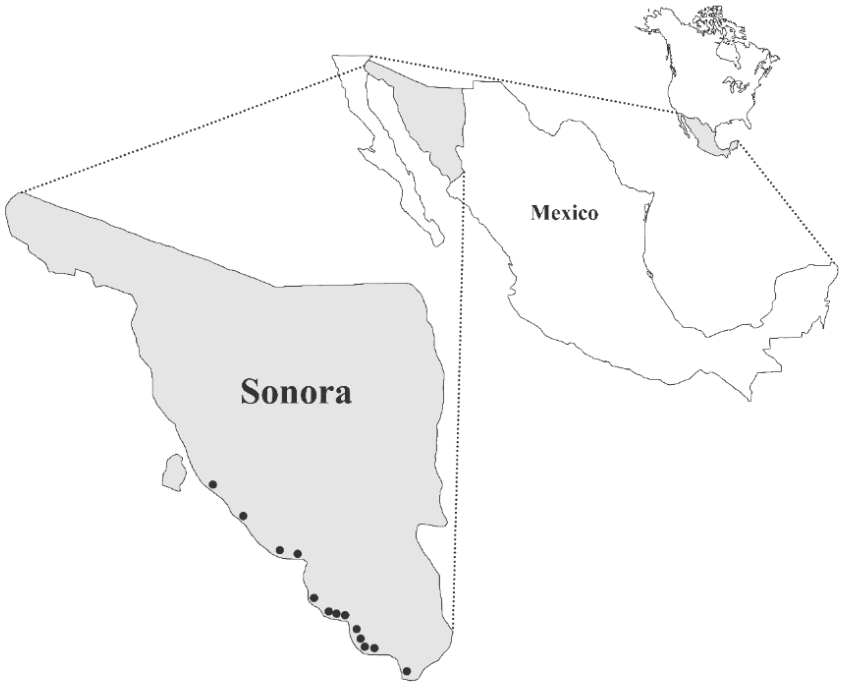


Figure 9.2. Geographical coverage of the shrimp farms under assessment

The information to analyse the environmental burdens in shrimp production comes from primary sources. In particular, the data was provided by the State Committee on Aquatic Health of the State of Sonora (COSAES-Spanish acronym). In any LCA study, the data collection stage is a fundamental step to ensure the reliability and reproducibility of the environmental profile. Therefore, the use of primary sources ensures compliance with the requirements of ISO 14040 and 14044 standards for data quality.

The information provided compiles relevant data to understand the operation of the different farms and comprises the following variables: Farming area (ha), stock density (organisms/m^2), total shrimp production (t), survival rate (%) and Feed Conversion Ratio-FCR. FCR is a ratio that measures the efficiency with which the bodies of seafood convert feed into the desired output. Primary data were verified in order to calculate other determinant elements, these elements include production efficiency, amount of feed provided and electricity consumption, among others. Appendix II shows in more detail how these variables were calculated.

Water exchange in ponds was based on agricultural records of the region: evaporation was estimated from historical 10-year average monthly rates from 5.7 to 10.7 mm/day (Garatuza-Payan et al., 1998). Daily exchange rate was calculated at 11% according to a previous crop cycle analysis (Casillas-Hernández et al., 2006).

Direct emissions of suspended solids, nitrogen and phosphorus were obtained following the guidelines provided by farm managers. The nitrogen emitted to the environment has been determined as the difference between the nitrogen supplied to the system in the feed and the nitrogen that is part of the composition of the adult shrimp. It has been taken into account that the protein content in the feed is 32% (Lee and Lee, 2018), while 16% of this protein is nitrogen (Pupim et al., 2013). Nitrogen in the adult shrimp body has been calculated considering a total protein content of 17.3%, a very similar value to that proposed by Dayal et al. (2013). Phosphorus emissions were obtained in the same way as nitrogen, considering that 1.7% of the feed supplied is phosphorus (Chatvijitkul et al., 2018). Similarly, a total phosphorus content of 0.3% was assumed for shrimp (Dayal et al., 2013). Finally, the total amount of suspended solids was estimated to be 5.3 times the total weight of produced shrimp. This emission factor was provided directly by the facility managers and it was calculated as the average of a series of estimates of six ponds over a year (Casillas-Hernández et al., 2006).

Table 9.1. Life cycle inventory data for an average medium-size farm per FU

Inputs					
SS1. Feed		**SS2. Larvae**		**SS3. Aquaculture**	
Materials	**kg**	**Materials**	**kg**	**Materials**	**t**
Fishmeal	495.1	Chloride	0.3	Feed from SS1	2
Fish oil	128.5	EDTA	0.1	Larvae from SS2	0.07
Soybean meal	495.1	Fishmeal	279.3	**Transport**	**t·km**
Wheat meal	368.9	Fish oil	55.1	Feed	140
Maize starch	337.9	Soybean meal	279.3	**Energy**	**MJ**
Soybean oil	66.8	Wheat meal	208.1	Electricity	13,333
Raw materials	**L**	Soybean meal	28.7	**Raw materials**	**m^3**
Water	240	Energy	kWh	Water	3,380.1
		Electricity	1,948.8		**ha**
		Raw materials	**m^3**	Land	0.3
		Water	19.3		
Outputs					
SS1. Feed		**SS2. Larvae**		**SS3. Aquaculture**	
Products	**t**	**Products**	**t**	**Products**	**t**
Feed to SS3	2	Larvae to SS3	0.07	Shrimp	1
				Emissions to water	**t**
				Suspended solids	5.3
				Nitrogen	0.076
				Phosphorus	0.03

It is important to highlight the high volume of data handled in this chapter, since life cycle inventory data has been collected from 38 shrimp farms. These inventories were classified according to farm size, so small farms with a total production of less than 150 t, medium farms between 150 and 500 t and large farms for shrimp production of more than 500 t were considered. Table 9.1 represents the life cycle inventory of an average medium-size farm, although the life cycle impact analysis was carried out for each of the 38 evaluated farms. It is also important to note that inventories of Subsystems 1 and 2 are similar for all farms while Subsystem 3 is specific for each facility.

9.2.3.3. Impact assessment

To convert the extensive list of life cycle inventory results into a useful list of environmental indicators, the ReCiPe 2016 v1.1 in a hierarchist perspective was used (Huijbregts et al., 2016). According to Henriksson et al. (2012), the following impact categories were selected: Global warming (GW), Stratospheric ozone depletion (SOD), Terrestrial acidification (TA), Freshwater eutrophication (FE), Marine eutrophication (ME), Marine ecotoxicity (MET), Fossil resources scarcity (FRS) and Water consumption (WC). SimaPro 9.0 (PRé Consultants, 2017) was the software used for the computational implementation of the inventories.

9.2.4. DEA model selection

Finally, as well as in Chapter 8, SBM model was selected as it follows a non-radial approach, which allows greater discrimination power to assess the efficiency of DMU than radial methods (Samuel-Fitwi et al., 2013). Convexity, scalability and free arrangement of inputs and outputs are also assumed for the determination of the efficient production frontier. The applicability of this model is so high that many authors use it in very different fields, i.e. in the production of grapes for winemaking (Vázquez-Rowe et al., 2012), bioelectricity production from biomass gasification (Rajabi Hamedani et al., 2019), farm-scaled biogas plants (Lijó et al., 2017), grocery stores (Álvarez-Rodríguez et al., 2019) or wind farms (Iribarren et al., 2013). On the other hand, an input-oriented approach was chosen because the main objective of this chapter is to minimize the use of resources (inputs) and possible environmental impacts without affecting the production of shrimp (outputs) but also considering that shrimp production is limited to the degree of technological development of each farm.

9.2.5. Input/output selection

The DEA matrix calculated in this chapter was composed of 7 inputs and 1 output (Table 9.2). These units were chosen for their operational importance and associated environmental impacts, according to the previous life cycle analysis. It is important to emphasise that elements I-6 and I-7 are undesirable outputs, although they were considered as inputs for calculation purposes. The complete DEA matrix is shown in table

C.1 of Appendix I**.** The computational implementation of the DEA matrix was carried out through the DEA-solver Pro software (Cooper et al., 2007).

Table 9.2. Elements considered in the DEA matrix, codification, and measurement units.

	Label	Element	Unit
	I – 1	Seawater	m^3
	I – 2	Feed	t
	I – 3	Larvae	t
Inputs	I – 4	Electricity	MJ
	I – 5	Transport	t·km
	I – 6	Nitrogen	t
	I – 7	Phosphorus	t
Outputs	O – 1	Shrimps	t

9.2.6. Improvement actions

Once the critical stages in the environmental profile were determined, some improvement actions were determined to reduce the environmental impact of the system. Specifically, the variation of the life cycle impact was estimated with respect to two fundamental elements: the formulation of the feed and the energy requirements of the larvae tanks.

Feed management is a key factor that significantly affects water quality, final product quality and economic management of aquaculture facilities (Kong et al., 2020). In addition, it should be noted that environmental burdens from water discharge are derived from the portion of feed that is not consumed by the animals and remains in the pond water (Smárason et al., 2017). All this leads to the proposal to replace some components of the feed with others of lesser environmental impact that result in similar levels of growth and survival. Oatmeal, barley meal, rye meal, rapeseed meal and Distillers Dried Grain with Solubles (DDGS) were chosen as possible options for the substitution of wheat meal. Wheat meal was proposed for replacement because it is the element with the greatest overall impact. In order to make a reliable substitution, the feed conversion factor and the nutritional composition of each alternative were analysed.

The shift from electricity production to photovoltaic panel generation was evaluated as another improvement action. This action follows the path set by the United Nations in the fight against climate change (United Nations, 2015). Even so, plant managers must adopt good practices to minimize energy use (Cao et al., 2011). Considering that Mexico is geographically located between 14º and 33º latitude and that the average daily irradiation is around 3.1 MJ m-2 day-1 (Lobit et al., 2018), photovoltaic energy seems to be a good option to reduce the environmental footprint of energy consumption. In order to consider the electricity generation through photovoltaic generators within the system boundaries, the Ecoinvent® process "Electricity, low voltage {MX}|electricity production, photovoltaic, 3kWp slanted-roof installation, single-Si, panel, mounted" was used.

9.3. RESULTS

9.3.1. Environmental burdens of current DMUs

Tables C.2, C.3 and C.4 of Appendix I shows the contribution of each of the elements of the inventory to the impact categories for average small, medium, and large farms. Figure 9.3 breaks down the relative contribution of the subsystems involved for the different farm sizes. There are no major differences between the distribution of impacts by subsystems for the different sizes assessed. SS1. Feed and SS2. Larvae are primarily responsible for environmental burdens in most impact categories, except for freshwater eutrophication and water consumption. In these two categories, subsystem SS3. Aquaculture is the most relevant with 95-96% in FE and 92-97% in WC.

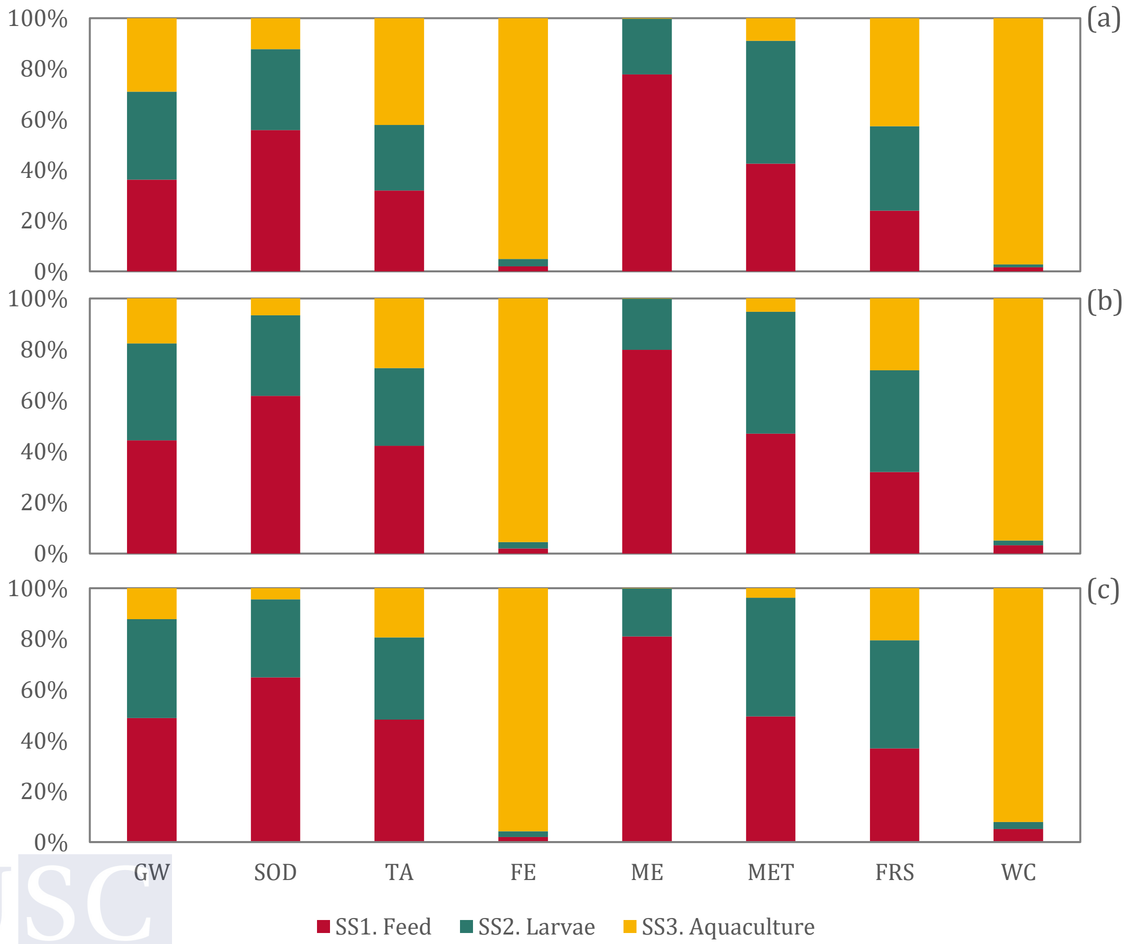


Figure 9.3. Relative contribution of the different subsystems in shrimp production for average small (a), medium (b) and large (c) farms

The environmental burdens in the GW category come mainly from the electricity requirements of SS1 and SS2. These electrical consumptions are related to the milling of wheat and soybean grains to obtain meals and the need for aeration in the larvae tanks to maintain optimal growth conditions (Tien et al., 2019), respectively. The contribution of these two sub-systems is equally relevant in the SOD category, coming from the emissions of dinitrogen monoxide (N_2O) from agricultural production of maize, wheat, and soybean.

Regarding the eutrophication categories, some differences can be found in the behaviour of the FE and ME categories. With respect to FE, SS3. Aquaculture is the main contributor due to direct phosphorus emissions (95%). This phosphorus, although an essential nutrient in aquaculture ecosystems, can play a central role in environmental pollution (Luo et al., 2018). Direct nitrogen emissions are also emitted from this subsystem, although they are not very relevant. In ME category, this subsystem has little influence, highlighting the contributions related to agricultural activities related to feed production. Finally, the impact on the WC category is also important, in which SS3. Aquaculture stands out with percentages from 92% to 97%, due to the large amount of water pumped to fill the ponds.

9.3.2. DEA calculation and efficiency scores

The DEA matrix was implemented into the DEA-solver Pro software (Cooper et al., 2007). The results that can be extracted from the model are the efficiency scores for each of the DMUs under assessment. The obtained DEA efficiency scores can be found in Table 9.3. Of all the shrimp farms evaluated, just over 13% (5 of 38 farms) were found fully efficient (Φ=1). However, although only 5 farms were considered fully efficient, the efficiency index can be considered high in general, as only four farms have efficiency values below 0.6 and an average efficiency of 0.79 is achieved. This average efficiency of the sample is relatively high compared to a previous study applied to similar systems. Chang et al. (2010) performed a DEA analysis to 70 seafood aquaculture facilities with an average efficiency of 0.55. It is also important to note that the farms with high efficiency index correspond to those of larger size, while small and medium farms obtained worse results.

For the inefficient farms (Φ<1), the software also suggests important operational reduction targets to make them efficient, these operational reductions are shown in table C.5 of Appendix I. An average operational reduction of 24.4% is proposed for the complete set of DMUs and inputs analysed. The reductions in I-1 (seawater) and I-2 (electricity) stand out, with 36.7% in both. It is important to note that these reductions are based on the theoretical efficient frontier and their achievement may be limited for technical or operational reasons (Lijó et al., 2017), such as providing sufficient nutrients to shrimps or the impossibility of purchasing new and more efficient equipment. Therefore, these operational reductions should be considered as the maximum potential for input reduction that can be achieved in shrimp aquaculture production and hence the sample of farms evaluated has a greater margin for improvement in the future. If the proposed reductions are analysed in detail, the DMUs with low efficiency values are identified. For example, the high reduction rates present in all inputs of DMUs 7, 23, 31 and 34 should be highlighted.

Table 9.3. Efficiency scores (Φ) of the complete sample of shrimp farms

DMU	Φ	DMU	Φ	DMU	Φ	DMU	Φ
1	0.78	11	0.71	21	0.58	31	0.44
2	0.77	12	0.83	22	0.87	32	0.76
3	0.81	13	1	23	0.38	33	1
4	0.79	14	0.81	24	0.68	34	0.57
5	0.77	15	0.88	25	0.76	35	0.89
6	0.94	16	0.75	26	0.83	36	0.84
7	0.55	17	0.84	27	0.74	37	0.80
8	0.76	18	0.85	28	0.87	38	1
9	1	12	0.77	29	0.64		
10	0.76	20	1	30	0.89		

9.3.3. Environmental burdens of virtual DMUs

This section contains the results obtained after the last stage of the methodology. The last stage of the LCA/DEA methodology consists of the estimation of the life cycle impacts for "virtual" operations resulting from the application of the theoretical operational reductions proposed in Section 7.3.2. In this way, the environmental savings due to efficient operation can be estimated by comparing the environmental profile before and after the operational reductions proposed by the DEA methodology. It is important to note that, due to the high variability of results between impact categories, the ReCiPe normalisation factors were applied to achieve an overview of the environmental performance of each DMU. As shown in Figure 9.4, the reduction targets applied to DMUs significantly affected the environmental performance.

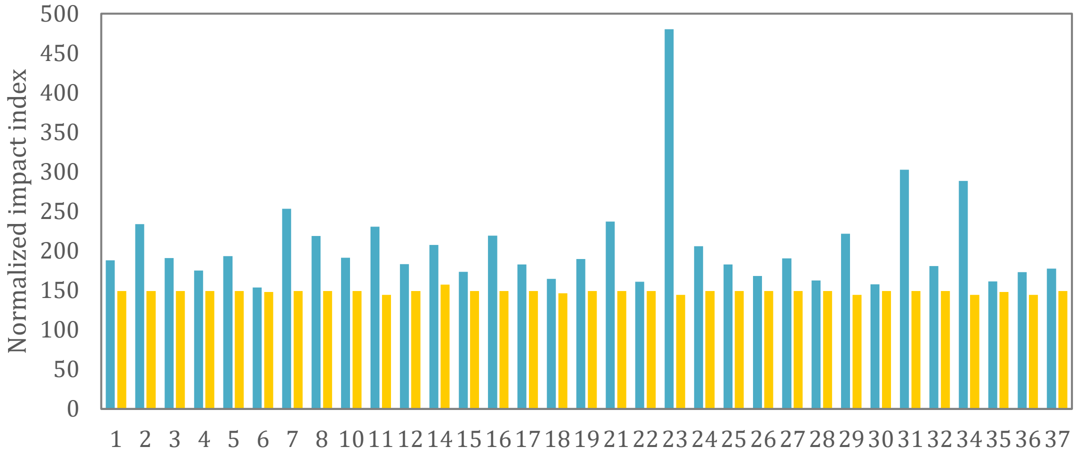


Figure 9.4. Normalisation results for original (blue) and virtual (yellow) DMUs.

Some percentages of reduction in environmental impact from 3.6% to 69.9% are achieved. As expected, the greatest reductions occurred on the farms with the lowest efficiencies, such as DMU 23 (69.9%) and DMU 31 (50.7%). While the smallest reductions were found on farms that were already close to full efficiency (DMUs 6 and 30). The results show that this methodology can be considered adequate to identify the link between operational and environmental performance of multiple, as all virtual farms have a similar environmental profile, corresponding to the optimal level of operation.

9.3.4. Improvement actions

The proposed improvement measures were evaluated independently, comparing the environmental profile with that of the base scenario. Figure 9.5 plots the environmental results related to feed production when wheat meal is substituted by any of the proposed alternatives.

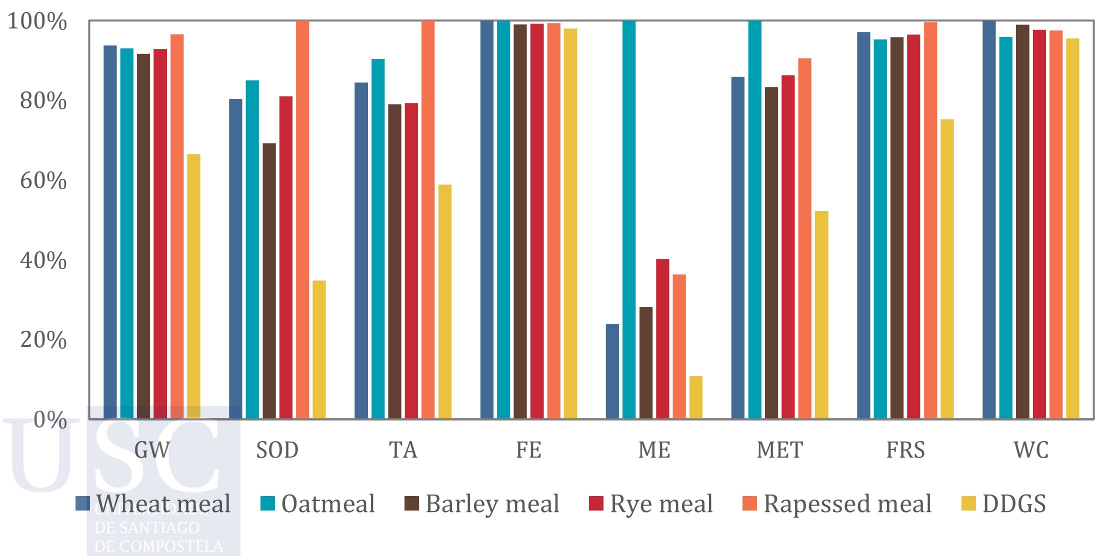


Figure 9.5. Environmental impact related to feed formulation with the different alternatives proposed.

In the view of the results, only the replacement of wheat meal by barley meal or by DDGS seems to be environmentally friendly. Analysing barley meal in detail, the reductions in environmental impacts are limited, although a 14% decrease in the SOD category stands out. Regarding DDGS, environmental improvements are more notable, such as 56.7% in SOD, 39.2% in MET or 30.2% in TA categories. DDGS therefore would be a viable substitute of wheat meal for feed production due to the following factors: (i) acceptable nutritional value, with a high protein content (Rhodes et al., 2015); (ii) relatively low market cost; (ii) no competition with the food industry and (iv) budding production linked to the growing importance of bioethanol industry.

Regarding electricity consumption, Figure 9.6 shows the variation in environmental impact with regards to the operation of a farm if, instead of considering the average Mexican profile for electricity supply, all energy is considered as solar photovoltaic energy.

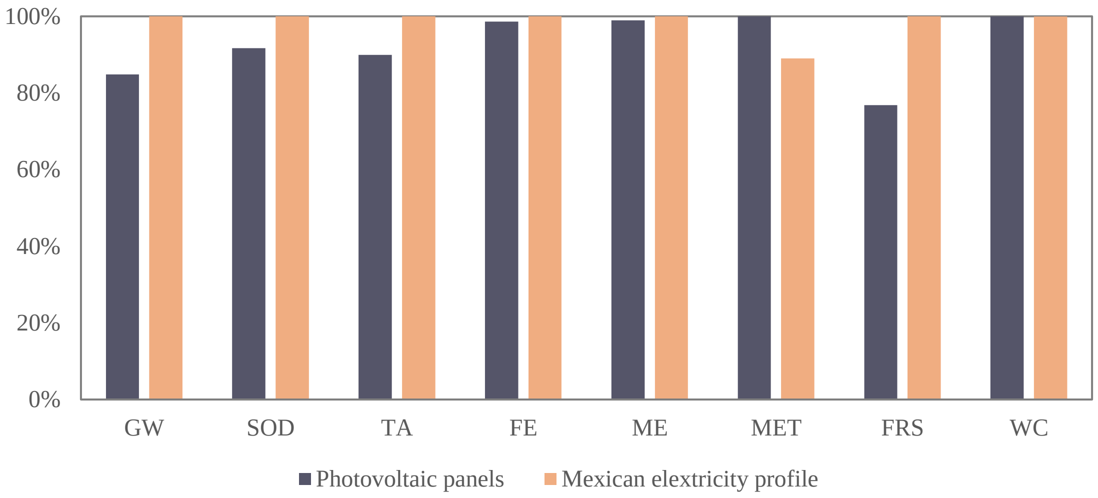


Figure 9.6. Environmental results regarding farm operations considering the use of photovoltaic energy or the Mexican electricity profile

The installation and use of photovoltaic panels would result in a 15% reduction in carbon footprint, in addition to a 10% reduction in TA and 23.2% in FRS. Only in MET category the impact would increase slightly by 12.3%. This category would increase due firstly to the mounting structure and secondly to the manufacture of the photovoltaic panel (Ling-Chin et al., 2016). Bearing in mind that the high impact in this category is derived from a structure whose useful lifetime is quite long, it can be concluded that the implementation of these photovoltaic panels in the facilities will have a positive effect on the environmental impact (Corcelli et al., 2019).

9.4. DISCUSSION

In this chapter, a wide range of impact categories were used to obtain an overview of the environmental performance of the process. The impact categories analysed have made it possible to cover a wide spectrum of environmental impacts related to different environmental aspects. Thus, this chapter includes, among others, the GW, TA, FE and ME categories which, according to Henriksson et al. (2012), cover the most commonly used in LCA applied to aquaculture systems.

The results obtained in this chapter are comparable with previous results reported in LCA studies on shrimp aquaculture performance. Jonell and Henriksson (2015) applied the life cycle assessment to mangrove-shrimp farms. Their study, similarly to what has been done for Mexican farms, considered a “cradle to farm-gate” approach, although with some difference. Both studies included energy supply, raw material extraction, agriculture, shrimp larval production, and shrimp cultivation and harvesting. In addition, infrastructure was not included in either study because of its negligible influence. It is important to note the operational differences between them, as Mexican farms continue to use traditional techniques and the use of chemicals is not reported. Chemicals are responsible for improving productivity in aquaculture systems by improving larval survival rates, feeding efficiency, and pathogen control, but they also have a negative impact on the environment due to their ecotoxicity. Jonell and Henriksson (2015), who considered the same FU (1 tonne of shrimp), only evaluated the most common impact categories (eutrophication, acidification and global warming). The carbon footprint obtained in this chapter is 7.6 kg CO_2 eq per kg live shrimp at the farm gate, while Jonell and Henriksson (2015) reported 27.4 kg CO_2 per kg for conventional aquaculture shrimp production. They also carried out the same evaluation to organic production, obtaining better results, around 13.3 kg CO_2 eq per kg. This same trend can be found in acidification category, worse results in traditional than organic production (10.1 and 8.1 kg SO_2 eq, respectively), but much higher than the value of 37 g SO_2 per kg reported in this chapter. The use of diesel for removing pond sediments, zeolites used in the shrimp grow-out phase and the applications of NPK and P_2O_5 to increase productivity could explain the differences between both studies.

Medeiros et al. (2017) analysed the production of *Macrobrachium amazonicum* in Brazil. Although it is not the same species as that produced in Mexico, the operating conditions are similar. In addition, it should be noted that the components of the feed used are also similar, with vegetable components such as soybean meal and wheat flour and other animal elements such as fish oil. However, the life cycle of the two species are different, which is reflected in the results obtained, since in in this chapter, 2 kg of feed per kg live shrimp at farm gate are needed, while in Medeiros et al. (2017) 2.7 kg per kg are required. In conclusion, almost identical results of 38 g SO_2 eq per kg are reported for the acidification category, while the carbon footprint results are 11.1 kg CO_2 eq per kg shrimp.

In contrast, the carbon footprint values reported in this study are slightly higher than those obtained in a previous study that evaluated organic shrimp production in Taiwan (Chang et al., 2017). Although the authors considered the distribution and use stage within the system boundaries, these life cycle stages were omitted for the comparison of results. Thus, the cradle-to-gate carbon footprint of organic shrimp production in Taiwan was 5.7 kg CO_2 eq per kg shrimp, a significantly lower value than the obtained in this study (7.6 kg CO_2 eq per kg shrimp). Furthermore, the author concludes that both electricity consumption and feed formulation are determining elements in the final environmental burdens. Therefore, it can be considered that the farms evaluated in this study have, in general, a good environmental performance, at least in terms of carbon footprint and terrestrial acidification, at similar levels to organic and certified production. It is important to note that results in terms of eutrophication have not been compared due to methodological differences between the CML (Guinée et al., 2002) and ReCiPe methodologies. Moreover, the high reduction of the environmental impacts in terms of eutrophication (freshwater and marine) should be highlighted. Both categories were reduced by almost 18% and 19% respectively.

With regard to efficiency scores, the results obtained from the DEA study showed that only 5 of the 38 farms evaluated were considered efficient, which represents a low value compared to previous LCA/DEA studies applied to the agri-food sector (Iribarren et al., 2011; Laso et al., 2018; Lozano et al., 2010; Vázquez-Rowe et al., 2010). It should be noted that these previous studies did not analyse aquaculture production but focused on other agro-industrial production systems such as fishing, mussel rafts or wine production. Although it was found that few DMUs were fully efficient, it is important to note that most DMUs achieved efficiency values above 0.5. In fact, only 5 were found to be below 0.6. Therefore, the average efficiency value of the sample evaluated is a reasonably high value of 0.79 (see Section 7.3.2). The robustness of this instrument in handling these heterogeneous and different datasets should be highlighted, allowing to go one step further than the computation of an average inventory (Lorenzo-Toja et al., 2015). The results obtained seems consistent when checking the behaviour of some key factors for the farm operation (Section 7.3.1), which showed that electricity and feed consumption are the main hot spots of the process. In addition, the study area was also considered as a key and limiting point in different types of aquaculture (Theodoridis et al., 2017). Thus, Figure 9.7 shows the ratios of production/area, production/feed and production/energy for efficient and inefficient farms.

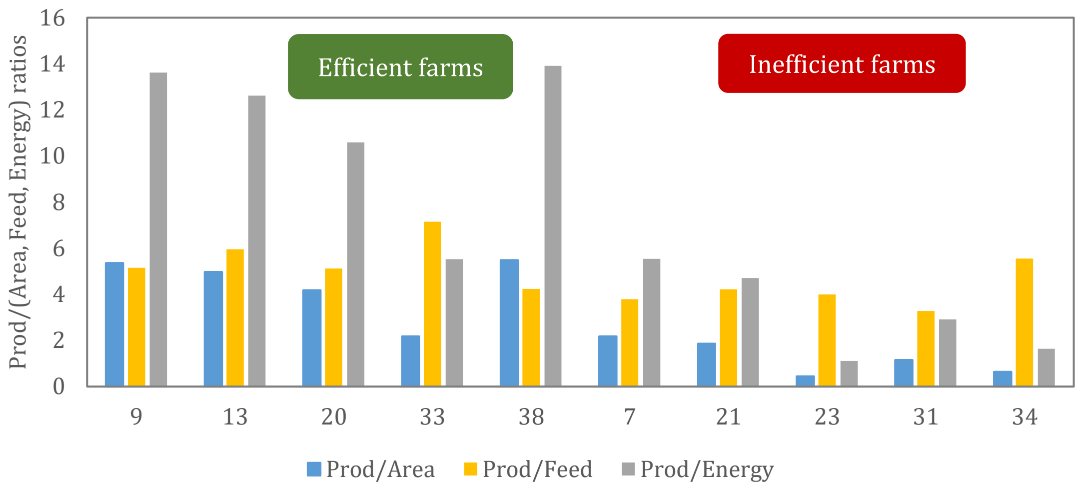


Figure 9.7. Production/area, production/feed and production/energy ratios for efficient farms (left of the dottle line) and inefficient farms (right of the dotted line).

As shown in Figure 9.7, the production/area and production/energy ratios are very high for the most efficient farms. However, the case of DMU 33 should be noted, as neither of these ratios is high, but the farm presents an efficient operation. This can be explained by the fact that it has a very high production/feed ratio, which highlights the importance of feed in the environmental and operational performance of the process. While it is true that inefficient farms (to the right of the dotted line) do not have significantly lower production/feed values than efficient ones, the combination of the three ratios clearly gives the worst results. This makes it clear that, in order to seek operational and environmental efficiency, action must be taken on all possible lines of action, prioritising a balanced improvement of all variables.

These measures can be expected to have a positive long-term impact on the receiving water body (Gulf of California). Since, as determined above (Ahrens et al., 2008) the reduction of N losses to surface waters will buffer denitrification events and N_2O emissions to Valley drains, and N export events to estuaries and the Gulf of California.

9.5. CONCLUSIONS

The path towards a real Circular Economy in the seafood sector requires the proposal of sustainable alternatives that address wild fisheries and improve food security. Life Cycle Assessment was used to evaluate the environmental aspects associated with shrimp production farms in the state of Sonora. The results showed that feed formulation and electricity consumption in larval tanks are the main "hot-spots" of the process.

The joint application of LCA + DEA provided a comprehensive approach that made possible to distinguish operationally inefficient farms and, although only 5 out of 38 were considered fully efficient, the average efficiency of the sample was 0.79. The expected reductions in input consumption were significant, resulting in estimated reductions from 3.6% to 69.9% in the normalised impact index depending on the DMU evaluated. The farms with the largest reductions were generally small and medium-sized, while the larger farms were the most efficient.

As a result of the eco-efficiency analysis, several improvement actions were proposed that resulted in the convenience of installing photovoltaic panels and decreasing the food conversion ratio by substituting wheat meal in the feed. Substitution by DDGS proved to be the most promising option, ensuring reductions of between 2% and 57% depending on the impact categories.

In conclusion, the potential of aquaculture to meet the demand for seafood is shown as an excellent opportunity to contribute to the healthy nutrition of the population, while paying attention to the conservation of marine resources. Since the main priority is the use of environmentally sustainable alternatives, this chapter can be considered as a useful guide for shrimp farm managers, particularly in Mexico.

9.6. REFERENCES

Ahrens, T.D., Beman, J.M., Harrison, J.A., Jewett, P.K., Matson, P.A., 2008. A synthesis of nitrogen transformations and transfers from land to the sea in the yaqui valley agricultural region of northwest mexico. Water Resour. Res. 45, 1–13. https://doi.org/10.1029/2007WR006661

Álvarez-Rodríguez, C., Martín-Gamboa, M., Iribarren, D., 2019. Combined use of Data Envelopment Analysis and Life Cycle Assessment for operational and environmental benchmarking in the service sector: A case study of grocery stores. Sci. Total Environ. 667, 799–808. https://doi.org/10.1016/j.scitotenv.2019.02.433

Arreola-Lizárraga, J.A., Garatuza-Payán, J., Yépez-González, E.A., Robles-Morúa, A., 2019. Capital natural y bienestar social de la comunidad Yaqui, Oficina de Publicaciones, Instituto Tecnológico de Sonora. Ciudad Obregón, Sonora, Mexico.

Cao, L., Diana, J.S., Keoleian, G.A., Lai, Q., 2011. Life cycle assessment of chinese shrimp farming systems targeted for export and domestic sales. Environ. Sci. Technol. 45, 6531–6538. https://doi.org/10.1021/es104058z

Casillas-Hernández, R., Magallón-Barajas, F., Portillo-Clarck, G., Páez-Osuna, F., 2006. Nutrient mass balances in semi-intensive shrimp ponds from Sonora, Mexico using two feeding strategies: Trays and mechanical dispersal. Aquaculture 258, 289–298. https://doi.org/10.1016/j.aquaculture.2006.03.027

Chang, C.C., Chang, K.C., Lin, W.C., Wu, M.H., 2017. Carbon footprint analysis in the aquaculture industry: Assessment of an ecological shrimp farm. J. Clean. Prod. 168, 1101–1107. https://doi.org/10.1016/j.jclepro.2017.09.109

Chang, H.H., Boisvert, R.N., Hung, L.Y., 2010. Land subsidence, production efficiency, and the decision of aquacultural firms in Taiwan to discontinue production. Ecol. Econ. 69, 2448–2456. https://doi.org/10.1016/j.ecolecon.2010.07.020

Chatvijitkul, S., Boyd, C.E., Davis, D.A., 2018. Nitrogen, phosphorus, and carbon concentrations in some common aquaculture feeds. J. World Aquac. Soc. 49, 477–483. https://doi.org/10.1111/jwas.12443

Cooper, W.W., Seiford, L.M., Tone, K., 2007. Data Envelopment Analysis: A compresensive text with models, applications, references and DEA-Solver software. Springer, New York.

Corcelli, F., Fiorentino, G., Petit-boix, A., Rieradevall, J., Gabarrell, X., 2019. Transforming rooftops into productive urban spaces in the Mediterranean . An LCA comparison of agri-urban production and photovoltaic energy generation. Resour. Conserv. Recycl. 144, 321–336. https://doi.org/10.1016/j.resconrec.2019.01.040

Dayal, J.S., Ponniah, A.G., Khan, H.I., Babu, E.P.M., Ambasankar, K., Vasagam, K.P.K., 2013. Shrimps - a nutritional perspective. Curr. Sci. 104, 1487–1491. https://doi.org/jstor.org/stable/24092471

FAO, 2020. The State of World Fisheries and Aquaculture - Sustainability in action. Food and Agriculture Organization of the United Nations, Rome.

FAO, 2018. The State of World Fisheries and Aquaculture 2018 - Meeting the sustainable development goals. Food and Agriculture Organization of the United Nations, Rome.

Garatuza-Payan, J., Shuttleworth, W.J., Encinas, D., McNeil, D.D., Stewart, J.B., DeBruin, H., Watts, C., 1998. Measurement and modelling evaporation for irrigated crops in north-west Mexico. Hydrol. Process. 12, 1397–1418. https://doi.org/10.1002/(SICI)1099-1085(199807)12:9<1397::AID-HYP644>3.0.CO;2-E

González, O.H.A., Beltrán, L.F., Cáceres-Martínez, C., Ramírez, H., Hernández-Vazquez, S., Troyo-Dieguez, E., Ortega-Rubie, A., 2003. Sustainability development analysis of semi-intensive shrimp farms in Sonora, México. Sustain. Dev. 11, 213–222.

Guinée, J.B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Sleeswijk, A.W., de Haes, H., de Bruijn, H., van Duin, R., Huijbregts, M.A.J., 2002. Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards. Kluwer Academic Publishers, Dordrecht.

Gutiérrez, E., Lozano, S., Guillén, J., 2020. Efficiency data analysis in EU aquaculture production. Aquaculture 520, 734962. https://doi.org/10.1016/j.aquaculture.2020.734962

Henriksson, P.J.G., Guinée, J.B., Kleijn, R., De Snoo, G.R., 2012. Life cycle assessment of aquaculture systems-A review of methodologies. Int. J. Life Cycle Assess. 17, 304–313. https://doi.org/10.1007/s11367-011-0369-4

Henriksson, P.J.G., Rico, A., Zhang, W., Ahmad-al-nahid, S., Newton, R., Phan, L.T., Zhang, Z., Jaithiang, J., Dao, H.M., Phu, T.M., Little, D.C., Murray, F.J., Satapornvanit, K., Liu, L., Liu, Q., Haque, M.M., Kruijssen, F., Snoo, G.R. De, Heijungs, R., Bodegom, P.M. Van, Guine, J.B., 2015. Comparison of Asian Aquaculture Products by Use of Statistically Supported Life Cycle Assessment. https://doi.org/10.1021/acs.est.5b04634

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M.D.M., Hollander, A., Zijp, M., van Zelm, R., 2016. ReCiPe 2016 v1.1. A harmonized life cycle impact assessment method at midpoint and endpoint level. The Netherlands.

Iribarren, D., Hospido, A., Moreira, M.T., Feijoo, G., 2011. Benchmarking environmental and operational parameters through eco-efficiency criteria for dairy farms. Sci. Total Environ. 409, 1786–1798. https://doi.org/10.1016/j.scitotenv.2011.02.013

Iribarren, D., Martín-Gamboa, M., Dufour, J., 2013. Environmental benchmarking of wind farms according to their operational performance. Energy 61, 589–597. https://doi.org/10.1016/j.energy.2013.09.005

Järviö, N., 2018. Including GHG emissions from mangrove forests LULUC in LCA : a case study on shrimp farming in the Mekong Delta , Vietnam 1078–1090. https://doi.org/10.1007/s11367-017-1332-9

Jonell, M., Henriksson, P.J.G., 2015. Mangrove-shrimp farms in Vietnam-Comparing organic and conventional systems using life cycle assessment. Aquaculture 447, 66–75. https://doi.org/10.1016/j.aquaculture.2014.11.001

Kong, W., Huang, S., Yang, Z., Shi, F., Feng, Y., Khatoon, Z., 2020. Fish Feed Quality Is a Key Factor in Impacting Aquaculture Water Environment: Evidence from Incubator Experiments. Sci. Rep. 10, 1–15. https://doi.org/10.1038/s41598-019-57063-w

Laso, J., Vázquez-Rowe, I., Margallo, M., Irabien, Á., Aldaco, R., 2018. Revisiting the LCA+DEA method in fishing fleets. How should we be measuring efficiency? Mar. Policy 91, 34–40. https://doi.org/10.1016/j.marpol.2018.01.030

Lee, C., Lee, K.J., 2018. Dietary protein requirement of Pacific white shrimp *Litopenaeus vannamei* in three different growth stages. Fish. Aquat. Sci. 21, 1–6. https://doi.org/10.1186/s41240-018-0105-0

Lijó, L., Lorenzo-Toja, Y., González-García, S., Bacenetti, J., Negri, M., Moreira, M.T., 2017. Eco-efficiency assessment of farm-scaled biogas plants. Bioresour. Technol. 237, 146–155. https://doi.org/10.1016/j.biortech.2017.01.055

Ling-Chin, J., Heidrich, O., Roskilly, A.P., 2016. Life cycle assessment (LCA) – from analysing methodology development to introducing an LCA framework for marine photovoltaic (PV) systems. Renew. Sustain. Energy Rev. 59, 352–378. https://doi.org/10.1016/j.rser.2015.12.058

Little, D.C., Newton, R.W., Beveridge, M.C.M., 2016. Aquaculture: A rapidly growing and significant source of sustainable food? Status, transitions and potential. Proc. Nutr. Soc. 75, 274–286. https://doi.org/10.1017/S0029665116000665

Lobit, P., López-Pérez, L., Lhomme, J.P., 2018. Retrieving air humidity , global solar radiation , and reference evapotranspiration from daily temperatures : development and validation of new methods for Mexico . Part II : radiation. Theor. Appl. Climatol.

133, 799–810. https://doi.org/10.1007/s00704-017-2212-8

Lorenzo-Toja, Y., Vázquez-Rowe, I., Chenel, S., Marín-Navarro, D., Moreira, M.T., Feijoo, G., 2015. Eco-efficiency analysis of Spanish WWTPs using the LCA+DEA method. Water Res. 68, 651–666. https://doi.org/10.1016/j.watres.2014.10.040

Lozano, S., Iribarren, D., Moreira, M.T., Feijoo, G., 2010. Environmental impact efficiency in mussel cultivation. Resour. Conserv. Recycl. 54, 1269–1277. https://doi.org/10.1016/j.resconrec.2010.04.004

Lucien-Brun, H., 2016. Critical decisions for shrimp harvesting and packing, Part 1. Glob. Aquac. Advocate.

Luo, Z., Hu, S., Chen, D., 2018. The trends of aquacultural nitrogen budget and its environmental implications in China. Sci. Rep. 8, 1–9. https://doi.org/10.1038/s41598-018-29214-y

Medeiros, M. V., Aubin, J., Camargo, A.F.M., 2017. Life cycle assessment of fish and prawn production: Comparison of monoculture and polyculture freshwater systems in Brazil. J. Clean. Prod. 156, 528–537. https://doi.org/10.1016/j.jclepro.2017.04.059

OECD, 2017. OECD Review of Fisheries 2017 - General Survey of Fisheries Policies.

Ponce-Palafox, J.T., Ruiz-Luna, A., Castillo-Vargasmachuca, S., García-Ulloa, M., Arredondo-Figueroa, J.L., 2011. Technical, economics and environmental analysis of semi-intensive shrimp (*Litopenaeus vannamei*) farming in Sonora, Sinaloa and Nayarit states, at the east coast of the Gulf of California, México. Ocean Coast. Manag. 54, 507–513. https://doi.org/10.1016/j.ocecoaman.2011.03.008

Porchas-Cornejo, M.A., Álvarez-Ruiz, P., Alvarez-Tello, F.J., Martínez-Porchas, M., Martínez-Córdova, L.R., López-Martínez, J., García-Morales, R., 2018. Detection of the white spot syndrome virus in zooplankton samples collected off the coast of Sonora , Mexico. Aquac. Res. 49, 48–56. https://doi.org/10.1111/are.13431

PRé Consultants, 2017. SimaPro Database Manual (No. Methods Library). The Netherlands.

Pupim, L.B., Martin, C.J., Ikizler, T.A., 2013. Assessment of protein and energy nutritional status, in: Nutritional Management of Renal Disease. Elsevier Inc., pp. 137–158. https://doi.org/10.1016/B978-0-12-391934-2.00010-2

Rajabi Hamedani, S., Del Zotto, L., Bocci, E., Colantoni, A., Villarini, M., 2019. Eco-efficiency assessment of bioelectricity production from Iranian vineyard biomass gasification. Biomass and Bioenergy 127, 105271. https://doi.org/10.1016/j.biombioe.2019.105271

Rhodes, M.A., Yu, D., Zhou, Y., Davis, D.A., 2015. Use of lipid-extracted distillers dried grain with solubles (DDGS) in diets for pacific white shrimp. N. Am. J. Aquac. 77, 539–546. https://doi.org/10.1080/15222055.2015.1067263

Samuel-Fitwi, B., Nagel, F., Meyer, S., Schroeder, J.P., Schulz, C., 2013. Comparative life cycle assessment (LCA) of raising rainbow trout (*Oncorhynchus mykiss*) in different production systems. Aquac. Eng. 54, 85–92.

https://doi.org/10.1016/j.aquaeng.2012.12.002

Santos, W., Rosa, J., Mauad, C., Vogel, E., Jorge, O., Favarini, C., 2019. Sustainability and technical efficiency of fish hatcheries in the STATE of MATO GROSSO do SUL , Brazi l. Aquaculture 500, 228–236. https://doi.org/10.1016/j.aquaculture.2018.10.024

Smárason, B.Ö., Ögmundarson, Ó., Árnason, J., Björnsdóttir, R., Davíösdóttir, B., 2017. Life cycle assessment of icelandic arctic char fed three different feed types. Turkish J. Fish. Aquat. Sci. 17, 79–90. https://doi.org/10.4194/1303-2712-v17_1_10

Theodoridis, A., Batzios, C., Ragkos, A., Angelidis, P., 2017. Technical efficiency measurement of mussel aquaculture in Greece. Aquac. Int. 25, 1025–1037. https://doi.org/10.1007/s10499-016-0092-z

Tien, N.N., Matsuhashi, R., Chau, V.T.T.B., 2019. A sustainable energy model for shrimp farms in the Mekong delta. Energy Procedia 157, 926–938. https://doi.org/10.1016/j.egypro.2018.11.259

United Nations, 2015. Transforming our world: the 2030 Agenda for Sustainable Development. A/RES/70/1. UN General Assembly. https://doi.org/10.1007/s13398-014-0173-7.2

Van, T.P.T.H., Rhodes, M.A., Zhou, Y., Davis, D.A., 2017. Feed management for Pacific white shrimp Litopenaeus vannamei under semi-intensive conditions in tanks and ponds. Aquac. Res. 48, 5346–5355. https://doi.org/10.1111/are.13348

Vázquez-Rowe, I., Iribarren, D., Moreira, M.T., Feijoo, G., 2010. Combined application of life cycle assessment and data envelopment analysis as a methodological approach for the assessment of fisheries. Int. J. Life Cycle Assess. 15, 272–283. https://doi.org/10.1007/s11367-010-0154-9

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. https://doi.org/10.1016/j.jclepro.2011.12.039

SECTION V

GENERAL CONCLUSIONS

Chapter 10

General findings and conclusions of the thesis

Summary

The main goal of this doctoral thesis was to assess the environmental sustainability of the application of circular economy strategies in key primary sectors in Galicia through the application of LCA and other complementary tools. Given the great importance of the primary sector for agricultural and fishing activities in this region, it seems coherent to think that the conclusions drawn from this thesis can be representative of the Galician primary sector. The topic addressed in this thesis is in line with the growing concern for the application of circular economy in different aspects of our daily life and in the search for a sustainable lifestyle. In this context, the environmental assessment and eco-efficiency methodologies applied in this thesis have been shown to be useful for this purpose. It has been demonstrated that the application of a biorefinery approach in the primary sector to valorise solid waste by producing different value-added products is a good strategy that reduces the environmental impact by avoiding the manufacture of new products from virgin materials. Combining this multi-product approach with the application of traditional techniques in fish and seafood extraction proved to be a vital option for reducing the environmental burdens of the global fisheries sector. At the same time, eco-efficiency analyses demonstrated how operationally efficient practices translate into lower environmental impact, opening the door to improvement actions that enhance both operational and environmental efficiency. The potential of aquaculture and dairy production to meet the demand for seafood and dairy products was shown to be an excellent opportunity to contribute to the healthy nutrition of the population, while paying attention to the conservation of natural resources and environmental pollution. The main findings and conclusions extracted from Sections II, Section III, and Section IV are detailed below.

Section II: Circular economy in the agricultural sector. The main objective of the section is to estimate the environmental impacts of different ways of valorisation of agro-industrial waste, mainly from the wine industry. In addition, this section aims at comparing these alternatives with current practices carried out on the different waste analysed in each of the chapters. In this sense, this section is composed of 3 separate case studies, with different valorisation pathway which, however, can be complementary. Wine lees valorisation for the production of value-added products in a biorefinery scheme is evaluated in chapter 3, while the organic valorisation of grape marc trough vermicomposting for the production of bioactive compounds and organic amendment is evaluated in chapter 4. Finally, chapter 5 attempts to determine the environmental implications of using bioethanol from distillery wastes as a source of biohydrogen production and subsequent electricity generation in a solid fuel cell. The main outcomes from these chapters are detailed below.

Chapter 3 is aimed to delve into the different strategies for valuing wine lees, which are one of the two main winemaking-derived waste, apart from grape marc in a biorefinery scheme. The main outputs with marketable added value are some bio-based products as yeast cells, bioethanol, calcium tartrate and a polyphenol-rich antioxidant extract. For this purpose, data were collected from a biorefinery where the wine lees were processed, including all relevant energy and material consumption, and secondary data were collected from literature sources to determine the environmental impacts associated with wine production. In this regard, the results show that the valorisation of wine lees can be considered a very attractive process to obtain value-added products as, in comparison with other systems to obtain antioxidant-rich extracts, it presents the best environmental profile, improving on average 75% in all impact categories. However, there is always room for improvement, as the environmental analysis showed that steam consumption was the most important hotspot in the process, reaching very important contributions in almost all the impact categories. In this sense, it would be necessary to keep reducing this consumption in the future, for example, by using other residues from the winery, such as grape stalks, which could be used as raw material to obtain high temperature steam in a boiler.

Chapter 4 analyses the environmental impacts related to the valorisation of grape marc through vermicomposting to produce four main outputs: a nutrients-rich biofertiliser, marketable brandy spirit, an extract rich in polyphenols and oil rich in fatty acids. To this regard, primary date was collected from pilot-scale vermireactors, and data related to the operational aspects of the consumption of resources, waste management and the use of machinery were considered. Secondary data was also collected for modelling all the extraction process of polyphenols from the seeds. When economic allocation is taken into account to the different co-products, most environmental burdens are allocated to seed oil, as its price is very high. It is important to highlight that vermicomposting process is proven as an innovative and environmentally sustainable valorisation treatment, with the exception of the energy needed for distillation processes, so that in future scenarios, it could be interesting to replace fossil energy sources with renewable ones. The comparative analysis between this end-of-life treatment and other biowaste treatment practices shows that, although the vermicomposting presented poor results in terms of both carbon footprint and normalised impact index. However, when the economical revenue is considered, the environmental profile of vermicomposting is much better than the other alternatives, mainly because of the high quantity of co-products, which is a very relevant aspect, not only for an environmental but also from an economic point of view.

As the last part of Section II, Chapter 5 studies the environmental impacts of valorising alcoholic residues from distilleries by steam reforming, so that the main output product is hydrogen and small amounts of other gases. This output hydrogen is used to produce electricity in a 3 kW SOFC, which has been demonstrated as a clean technology, allowing the direct conversion of hydrogen into electricity in a fast, clean and safe way. When the environmental performance of this valorisation strategy is compared with current management methods, stem reforming appeared to be the most environmentally friendly process, reducing by 30 and 33% the GHG emissions with respect to incineration and landfill, respectively. In addition, as specific literature on biohydrogen production can be found, this valorisation alternative was compared with a total of 7 peer-reviewed studies to obtain hydrogen from renewable or non-renewable sources, resulting in steam reforming as the option with better environmental results in general terms, despite presenting poor results in GWP and POF categories. Both steam reforming and SOFC are robust technology that allow the valorisation of waste with impurities, so these technologies emerge as a good option at the end of their valorisation routes in which some output by-products are alcoholic stream with low value-added. In this sense, it may be interesting to connect this alternative to either of the two valorisation choices proposed in the previous chapters, since in all of them there was always an alcoholic stream coming out the distillation process.

Section III: Circular economy in the fisheries sector. This section is focused on the fisheries sector with two main objectives. On the one hand, Chapter 6 seeks to continue advancing in the construction of a comprehensive life cycle inventory on the capture of different Galician commercial species as well as demonstrating that the use of traditional techniques in capture and processing generates a lower environmental impact than other fisheries. In this regard, the detailed inventory of Atlantic scallop capture and processing by traditional techniques was compiled and environmentally assessed. On the other hand, the second objective is to know in depth the environmental impacts related to circular practices in the fishing sector, more specifically in the caning industry. With this purpose, in Chapter 7, a Galician canning industry whose main purpose is the commercialisation of canned tuna and other related by-products was analysed from an environmental point of view.

Chapter 6 analyses the environmental sustainability of the inshore great scallop fishery in the "Ría de Arousa". This fleet landed more than 90,000 kg of scallops during the study period, representing more than 75% of total Galician landings. Due to the lack of a specific inventory dedicated to the capture and processing of great scallops, data from 14 vessels of the artisanal fleet of Cambados were collected. Primary data was also collected on the processing, packaging, and freezing of scallops in the processing plant in the port of Cambados. The environmental profile of great scallops catch and processing show that electricity consumption in the processing facilities and the fishing stage were the critical points. More specifically, three main activities produced most of the environmental impacts, diesel production and combustion, anti-fouling and, in a lesser extent, manufacture and use of nets. These processes accounted for a total of 90% of the impact of scallop fishing. The main conclusion that can be extracted from this chapter is that the environmental impact of scallop fishing and processing is relatively high in comparison to other fisheries due to the combination of two factors, a high fuel use intensity and a low edible yield. However, when comparing the protein content with carbon footprint, it was shown that the great scallop presented of the best profiles within the evaluated fish and seafood as the protein content was one of the highest, at the level of some meats such beef or chicken, while the carbon footprint remained at the level of other molluscs and crustaceans.

The premise underlying Chapter 7 is that, although the Galician canning sector has traditionally employed circular economy-based measures in its activities, it still has enormous room for improvement to continue applying further actions. It was demonstrated that the inclusion of by-products valorisation processes improves the environmental profile of the process from a product approach. Thus, when the focus is placed on the assessment of a value chain, system expansion avoids allocating environmental burdens among the main products. The breakdown of the environmental impacts shows fishing and primary processing as the most relevant subsystems. In fishing stage, the consumption of diesel and antifouling, as well as the sardine bait used for fishing are the most relevant processes, while in primary processing, the manufacture of primary aluminium packaging stand out. From a general point of view, this chapter demonstrates how the Galician canning sector has the potential to reach the EU target of achieving the goal of zero biowaste in industry by valorising the biowaste into by-products. Similar studies need to be further applied to specific primary sub-sectors in the future to continue the path towards a more sustainable and circular food system.

Section IV: Eco-efficiency assessment. The combined five-step LCA+DEA methodology has been proven to be a convenient methodology for calculating the eco-efficiency and the environmental impacts derived by inefficient practices. In this section, two different case studies were carried out for dairy (Chapter 8) and aquaculture (Chapter 9) systems, which are detailed below.

Chapter 8 analyses the life cycle impact in terms of water and carbon footprint as well as the eco-efficiency of 96 livestock farms distributed throughout Galicia, considering each farm as an independent production unit. Feed production is found as the main responsible of water footprint, while direct CH_4 and N_2O emissions are found as the critical processes regarding carbon footprints. Considering the proposed LCA + DEA methodology, twenty-one farms are found as fully efficient. Reductions of silage plastic consumption and the production of wastewater are identified as the principal actions that the inefficient farms should carry out to improve their overall efficiency. Taking into account these results, the eco-efficiency of the Galician dairy sector has decreased over the last decade, going for an average of 0.64 to 0.58 in the last ten years. However, this fact must be the turning point that marks the path that the Galician dairy sector must follow, seeking to reduce its environmental impacts so that the production of milk pursues the compliance of international standards in terms of environmental protection.

Finally, Chapter 9 studies the environmental characterisation and eco-efficiency of semi-intensive shrimp production farms in the state of Sonora (Mexico). These semi-intensive farms are characterised by their diversity in terms of land use and size, while maintaining traditional working conditions. The results show that feed formulation and electricity consumption during larval growing in tanks were quantified as the main "hot-spots" on many of the evaluated farms. Only 5 of 38 farms are found as fully efficient, while the average efficiency of the sample is 0.79 over 1. The reductions in input consumption allow to estimate environmental impact reductions until a maximum of 69.9% in the normalised impact index. Different improvement actions were proposed, resulting in the convenience of installing photovoltaic panels as the most suitable option in terms of energy efficiency, while the substitution of wheat meal by DDGS proved to be the most promising options in term of food substituting. As general conclusion, the potential of aquaculture to meet the demand for seafood is shown as an excellent opportunity to contribute to the healthy nutrition of the population, while paying attention to the conservation of marine resources.

ADDITIONAL CONTENTS

USC

UNIVERSIDADE
DE SANTIAGO
DE COMPOSTELA

Appendix I. Complementary tables

Table A.1 and Table A.2 include the inventories related to the valorisation of wine waste in a biorefinery scheme to obtain value added products **(Chapter 3)**.

Table A.1. Inventory for Subsystem 1.1 per functional unit: 1 tonne of wine lees

Inputs from Technosphere			
Materials	**kg**	**Transport**	**t·km**
Thiocarbamates	5.00	Organic fertiliser	1190
Dithiocarbamates	33.57	**Energy**	**MWh**
Acetamide-aniline	0.80	Electricity	1.25
Nitriles	5.24	**Field operations**	**ha**
Cyclic-N compounds	1.33	Field sprayer	11.15
Phtalamide	13.00	Fertilising	1.57
Triazine	26.61	Tillage	2.44
Glyphosate	27.18	Rotary mower	1.57
Fosetyl-Al	56.47	Hoeing	1.57
Unspecified pesticides	219.51		
	t		
Steel	10.14		
Water	105.40	**Inputs from Environment**	
Organic fertiliser	273.52	**Land operations**	**ha**
Diesel	5.72	Land use	11.15
Outputs to Environment		**Outputs to Technosphere**	
Emissions to water	**kg**	**Products**	**t**
NO_3^-	1730.75	Grapes	95.82
PO_4^{3-}	11.18		

Table A.2. Inventory for Subsystem 1.2 per functional unit: 1 tonne of wine lees

Inputs from Technosphere		Inputs from Environment	
Energy	**MWh**	**Land operations**	**ha**
Electricity	61.85	Land use	0.13
Materials	**kg**		
Propane	166.38		
Water	666.38		
Lubricant oil	5.93		
Bentonite	93.21		
NaOH	16.67		
Ethylene glycol	143.73		
Polypropylene	48.00		
Liquified SO_2	3.27		
Outputs to Technosphere		**Outputs to Environment**	
Products	**m³**	**Emissions to atmosphere**	**t**
Wine	65.33	CO_2 (biogenic)	6.69
Wastes	**m³**		**kg**
Grape stalk	30.75	Ethanol	14.81
Grape pomace	30.49		**g**
Wine lees	1.00	Methanol	5.03
Wastewater	666.38	Acetaldehyde	0.56
		H_2S	10.45

Table B.1 shows the questionnaire for data collection. Tables B.2, B.3 and B.4 show the contribution of each element to the final carbon footprint, the Monte Carlo analysis and the data sources for the comparative analysis **(Chapter 6)**.

Table B.1. Example of the questionnaire used for the collection of fishery-related data.

Category		Unit (if applicable)	Data
Base port			Cambados
Discharge port			Cambados
Year of vessel construction			1990
Dredge material			Iron
Dredge renewal			Every 2-3 years
Weight of the nets		kg	2
Scallop fishing days per year		day	60
Net breaking			Once a month
Target species			Great scallop (90%)
Bycatch			Variegated scallop (10%)
Vessel material			Wood
Vessel´s dimensions	Length	metres	9
	Beam	metres	3.2
Capacity		tonne	6.8
Crew		persons	3
Lifetime		years	35
Operating time		months per year	11
Maintaining time		months per year	1
Distance to fishing zone		miles	6
Total fishing days per year		days	Working days for 11 months
Average speed with dredge		knots	2
Hours per day with dredge		hours	5
Number of sets per day		number	---
Material consumptions should refer to the scallop fishery			
Power of main engine		HP	85
Type of fuel used			Diesel
Fuel consumption		L/year	2,500 per scallop campaign
Lube oil consumption		L/year	50
Water consumption		m^3/year	
Paint consumption		L/year	25
Antifouling consumption		kg/year	10
Bilge water	Frequency of discharge		Once a week
	Discharge	L/Discharge	2

Table B.2. Results per FU applying mass and economic allocation factors. Relative contribution of each of the elements considered within the LCI.

	Impact category	GW	SOD	FE	ME	FET	MET	FRS
	Units	**kg CO_2 eq**	**kg CFC11 eq**	**kg P eq**	**kg N eq**	**kg 1,4-DCB**	**kg 1,4-DCB**	**kg oil eq**
	Mass allocation	$4.77\cdot10^{-1}$	$1.41\cdot10^{-7}$	$3.93\cdot10^{-5}$	$6.02\cdot10^{-6}$	$3.41\cdot10^{-3}$	$1.62\cdot10^{-1}$	$1.62\cdot10^{-1}$
	Economic allocation	$4.75\cdot10^{-1}$	$1.40\cdot10^{-7}$	$3.92\cdot10^{-5}$	$6.01\cdot10^{-6}$	$3.41\cdot10^{-3}$	$1.62\cdot10^{-1}$	$1.62\cdot10^{-1}$
Relative contribution	**Diesel**	81.4%	67.0%	17.0%	14.8%	15.4%	0.7%	79.5%
	Gillnets	1.4%	4.3%	1.8%	28.3%	2.5%	0.2%	1.3%
	Anti-fouling	0.1%	0.5%	8.7%	3.5%	18.0%	97.3%	0.1%
	Paint	0.0%	0.1%	0.1%	0.1%	0.1%	0.01%	0.1%
	Lubricant	0.2%	0.1%	0.5%	0.2%	0.7%	0.02%	0.7%
	Vessel construction	0.4%	0.9%	3.1%	5.1%	8.6%	0.3%	0.3%
	Bilge waters	0%	0%	0%	0%	0%	0.01%	0%
	Polypropylene	0.01%	0%	0.01%	0.1%	0.01%	0%	0.01%
	Chromium steel	0.01%	0.01%	0.01%	0.01%	0.02%	0.01%	0.01%
	Packaging film	5.3%	1.6%	8.4%	4.9%	6.1%	0.2%	9.4%
	Corrugated board	0.4%	1.9%	3.1%	10.2%	2.8%	0.1%	0.4%
	Electricity	10.6%	18.8%	55.8%	31.2%	43.9%	1.2%	8.2%
	Waste treatment	0.2%	4.9%	1.4%	1.7%	1.9%	0.1%	0.1%

Table B.3. Final results of the Monte Carlo analysis referred to the FU

Impact Category	Units	Mean	Median	SD	Coeff. of Variation	2,5%	97,5%	Std. Err. Of the mean
GW	kg CO_2 eq	$4.79 \cdot 10^{-1}$	$4.79 \cdot 10^{-1}$	$1.21 \cdot 10^{-1}$	25.20%	$2.44 \cdot 10^{-1}$	$7.27 \cdot 10^{-1}$	$3.82 \cdot 10^{-3}$
SOD	kg CFC11 eq	$1.39 \cdot 10^{-7}$	$1.31 \cdot 10^{-7}$	$4.84 \cdot 10^{-8}$	34.80%	$6.96 \cdot 10^{-8}$	$2.53 \cdot 10^{-7}$	$1.53 \cdot 10^{-9}$
FE	kg P eq	$3.87 \cdot 10^{-5}$	$3.38 \cdot 10^{-5}$	$1.92 \cdot 10^{-5}$	49.50%	$1.61 \cdot 10^{-5}$	$9.27 \cdot 10^{-5}$	$6.06 \cdot 10^{-7}$
ME	kg N eq	$5.78 \cdot 10^{-6}$	$5.71 \cdot 10^{-6}$	$7.02 \cdot 10^{-7}$	12.20%	$4.64 \cdot 10^{-6}$	$7.39 \cdot 10^{-6}$	$2.22 \cdot 10^{-8}$
FET	kg 1,4-DCB	$3.13 \cdot 10^{-3}$	$2.80 \cdot 10^{-3}$	$1.31 \cdot 10^{-3}$	42.00%	$1.65 \cdot 10^{-3}$	$6.29 \cdot 10^{-3}$	$4.15 \cdot 10^{-5}$
MET	kg 1,4-DCB	$1.59 \cdot 10^{-1}$	$1.59 \cdot 10^{-1}$	$6.45 \cdot 10^{-2}$	40.40%	$3.72 \cdot 10^{-2}$	$2.85 \cdot 10^{-1}$	$2.04 \cdot 10^{-3}$
FRS	kg oil eq	$1.64 \cdot 10^{-1}$	$1.61 \cdot 10^{-1}$	$4.62 \cdot 10^{-2}$	28.20%	$8.16 \cdot 10^{-2}$	$2.64 \cdot 10^{-1}$	$1.46 \cdot 10^{-3}$

Table B.4. Data collected for the representation of the comparative analysis

Category	Product	Protein[1]	Consumption[2]	CF[3]	Reference
Seafood	Scallop	19.0	116.2	3.44	This chapter
	G. barnacle	13.6	316.3	5.34	(Vázquez-Rowe et al., 2013)
	Lobster	17.7	2,103.1	8.80	(Vázquez-Rowe et al., 2014)
	Pilchard	18.1	20,257.0	0.51	(González-García et al., 2015)
	H. mackerel	15.4	15,254.8	2.28	(Vázquez-Rowe et al., 2010)
	Hake	11.9	88,091.3	5.44	(Vázquez-Rowe et al., 2011)
	Anchovy	17.6	44,447.6	1.45	(Laso et al., 2018)
	Mussel	10.8	50,784.5	0.47	(Iribarren et al., 2010)
	Prawn	18.0	14,036.5	14.85	(Clune et al., 2017)
	Turbot	16.1	4,371.3	14.51	(Clune et al., 2017)
	Trout	15.7	11,723.1	2.75	(Medeiros et al., 2017)
	Squid	14.0	34,080.5	3.86	(Iribarren et al., 2010)
	Sea bass	18.0	24,375.9	3.55	(Clune et al., 2017)
Meat	Beef	20.2	163,648.0	26.60	(Clune et al., 2017)
	Chicken	20.8	575,313.6	2.25	(González-García et al., 2014)
	Lamb	15.6	54,864.4	10.85	(Jones et al., 2014)
	Pork	21.0	457,181.7	4.96	(Noya et al., 2017)
Dairy	Milk	3.1	3,196,814.7	1.32	(Noya et al., 2018)
	Yogurt	4.0	450,923.9	1.77	(González-García et al., 2013)
	Butter	0.7	15,562.0	7.30	(Vergé et al., 2013)
	Cheese	21.6	354,415.0	5.30	(Vergé et al., 2013)
	Ice-cream	3.5	133,134.3	2.80	(Werner et al., 2014)
Fruits and vegetables	Orange	0.8	779,954.7	0.15	(Aguilera et al., 2015a)
	Banana	1.2	538,026.5	0.30	(Aguilera et al., 2015a)
	Apple	0.3	431,935.5	0.12	(Aguilera et al., 2015a)
	Strawberry	0.7	121,164.5	0.65	(Clune et al., 2017)
	Melon	0.6	348,900.6	0.24	(Aguilera et al., 2015a)
	Grape	0.6	86,759.6	0.12	(Aguilera et al., 2015a)
	Avocado	1.5	55,395.5	0.30	(Aguilera et al., 2015a)
	Potato	2.2	937,388.3	0.24	(Aguilera et al., 2015b)
	Tomato	0.9	605,132.6	0.26	(Aguilera et al., 2015b)
	Onion	1.1	325,443.3	0.22	(Aguilera et al., 2015b)
	Garlic	3.9	36,210.0	0.24	(Aguilera et al., 2015b)
	Spinach	2.4	56,552.8	0.54	(Clune et al., 2017)
	Carrot	0.8	148,844.4	0.22	(Clune et al., 2017)
	Zucchini	1.8	176,659.6	0.42	(Clune et al., 2017)
	Pineapple	0.5	87,086.7	0.72	(Aguilera et al., 2015b)
	Asparagus	2.9	31,922.6	1.07	(Vazquez-Rowe et al., 2016)

[1] Protein content. Grams of protein per 100 of edible portion. Data obtained from the Spanish Agency of Food Security and Nutrition (AESAN, 2018).

[2] Spanish annual consumption in 2018. Tonnes. Data extracted from household consumption survey carried out by the Spanish Ministry of Agriculture, Fisheries and Food (MAPA, 2020).

[3] Carbon Footprint. kg CO_2 eq per kg food

AESAN, 2018. Spanish Food Composition Database (BEDCA). URL: https://www.bedca.net/ (accessed 9.3.20).

Aguilera, E., Guzmán, G., Alonso, A., 2015a. Greenhouse gas emissions from conventional and organic cropping systems in Spain. II. Fruit tree orchards. Agron. Sustain. Dev. 35, 725–737. https://doi.org/10.1007/s13593-014-0265-y

Aguilera, E., Guzmán, G., Alonso, A., 2015b. Greenhouse gas emissions from conventional and organic cropping systems in Spain. I. Herbaceous crops. Agron. Sustain. Dev. 35, 713–724. https://doi.org/10.1007/s13593-014-0267-9

Clune, S., Crossin, E., Verghese, K., 2017. Systematic review of greenhouse gas emissions for different fresh food categories. J. Clean. Prod. 140, 766–783. https://doi.org/10.1016/j.jclepro.2016.04.082

González-García, S., Castanheira, É.G., Dias, A.C., Arroja, L., 2013. Environmental life cycle assessment of a dairy product: The yoghurt. Int. J. Life Cycle Assess. 18, 796–811. https://doi.org/10.1007/s11367-012-0522-8

González-García, S., Gomez-Fernández, Z., Dias, A.C., Feijoo, G., Moreira, M.T., Arroja, L., 2014. Life Cycle Assessment of broiler chicken production: A Portuguese case study. J. Clean. Prod. 74, 125–134. https://doi.org/10.1016/j.jclepro.2014.03.067

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. Int. J. Life Cycle Assess. 20, 1019–1032. https://doi.org/10.1007/s11367-015-0887-6

Iribarren, D., Moreira, M.T., Feijoo, G., 2010. Revisiting the Life Cycle Assessment of mussels from a sectorial perspective. J. Clean. Prod. 18, 101–111. https://doi.org/10.1016/j.jclepro.2009.10.009

Jones, A.K., Jones, D.L., Cross, P., 2014. The carbon footprint of lamb: Sources of variation and opportunities for mitigation. Agric. Syst. 123, 97–107. https://doi.org/10.1016/j.agsy.2013.09.006

Laso, J., Vázquez-Rowe, I., Margallo, M., Crujeiras, R.M., Irabien, Á., Aldaco, R., 2018. Life cycle assessment of European anchovy (*Engraulis encrasicolus*) landed by purse seine vessels in northern Spain. Int. J. Life Cycle Assess. 23, 1107–1125. https://doi.org/10.1007/s11367-017-1318-7

MAPA, 2020. Household consumption database. URL: https://www.mapa.gob.es/app/consumo-en-hogares/ (accessed 9.2.20).

Medeiros, M. V., Aubin, J., Camargo, A.F.M., 2017. Life cycle assessment of fish and prawn production: Comparison of monoculture and polyculture freshwater systems in Brazil. J. Clean. Prod. 156, 528–537. https://doi.org/10.1016/j.jclepro.2017.04.059

Noya, I., Aldea, X., González-García, S., M. Gasol, C., Moreira, M.T., Amores, M.J., Marín, D., Boschmonart-Rives, J., 2017. Environmental assessment of the entire pork value chain in Catalonia – A strategy to work towards Circular Economy. Sci. Total Environ. 589, 122–129. https://doi.org/10.1016/j.scitotenv.2017.02.186

Noya, I., González-García, S., Berzosa, J., Baucells, F., Feijoo, G., Moreira, M.T., 2018. Environmental and water sustainability of milk production in Northeast Spain. Sci. Total Environ. 616–617, 1317–1329. https://doi.org/10.1016/j.scitotenv.2017.10.186

Vazquez-Rowe, I., Kahhat, R., Quispe, I., Bentín, M., 2016. Environmental profile of green asparagus production in a hyper-arid zone in coastal Peru. J. Clean. Prod. 112, 2505–2517. https://doi.org/10.1016/j.jclepro.2015.09.076

Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2013. Carbon footprint analysis of goose barnacle (*Pollicipes pollicipes*) collection on the Galician coast (NW Spain). Fish. Res. 143, 191–200. https://doi.org/10.1016/j.fishres.2013.02.009

Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2011. Life Cycle Assessment of fresh hake fillets captured by the Galician fleet in the Northern Stock. Fish. Res. 110, 128–135. https://doi.org/10.1016/j.fishres.2011.03.022

Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2010. Life cycle assessment of horse mackerel fisheries in Galicia (NW Spain): Comparative analysis of two major fishing methods. Fish. Res. 106, 517–527. https://doi.org/10.1016/j.fishres.2010.09.027

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. https://doi.org/10.1007/s13280-013-0426-2

Vergé, X.P.C., Maxime, D., Dyer, J.A., Desjardins, R.L., Arcand, Y., Vanderzaag, A., 2013. Carbon footprint of Canadian dairy products: Calculations and issues. J. Dairy Sci. 96, 6091–6104. https://doi.org/10.3168/jds.2013-6563

Werner, L.B., Flysjö, A., Tholstrup, T., 2014. Greenhouse gas emissions of realistic dietary choices in Denmark: The carbon footprint and nutritional value of dairy products. Food Nutr. Res. 58, 1–16. https://doi.org/10.3402/fnr.v58.20687

Tables C.1, C.2 and C.3 include additional life cycle inventories compiled for subsystems not included in the main body of **Chapter 7**.

Table C.1. Life cycle inventory of the Subsystem 1. Tuna fishery per FU (1 tonne of raw tuna at processing plant).

SUBSYSTEM 1. TUNA FISHERY					
Inputs from the Technosphere					
Materials	**Unit**	**Value**	**Materials**	**Unit**	**Value**
Diesel	L	548.9	Lubricant oil	kg	5.3
Antifouling	g	671.3	R410-A	g	251.1
Ice	kg	230.2	Bait	kg	410.8
Boat paint	g	234.2			
Outputs to the Technosphere					
Products	**Unit**	**Value**	**Waste to treatment**	**Unit**	**Value**
Katsuwonus pelamis	kg	1,000	Plastic to recycling	g	624.6
			Cardboard to recycling	kg	1.0
			Plastic to landfill	kg	2.3
			Cardboard to landfill	g	471.4
Outputs to the Environment					
Emissions to Ocean	**Unit**	**Value**	**Emissions to Atmosphere**	**Unit**	**Value**
Xylene	g	61.4	CO_2	tonne	1.5
Cobalt	mg	35.6	SO_2	kg	14.0
Zinc	g	62.9	NO_X	kg	36.6
Copper	g	139.1	CO	kg	3.4

Table C.2. Life cycle inventory of the Subsystem 3. By-products valorisation per FU (1 tonne of raw tuna at processing plant).

SUBSYSTEM 3. BY-PRODUCTS VALORISATION					
Inputs from the Technosphere					
Materials	**Units**	**Value**	**Energy**	**Units**	**Value**
Inedible by-products from SS2	kg	627	Electricity	kWh	59.8
Bactericide	g	290.3	**Transport**	**Unit**	**Value**
Antioxidant	g	243.8	Inedible by-products	t·km	840.2
Water	L	754.7			
Polipropylene	g	94.2			
Outputs to the Technosphere					
Products	**Units**	**Value**	**Waste to treatment**	**Units**	**Value**
Fishmeal	kg	172.2	Mineral oil	g	5.8
Fish oil	kg	36.1	Oil filters	mg	580.6
			Hazardous waste	g	13.1
			Wastewater	m^3	1.4

Table C.3. Life cycle inventory of the Subsystem 4. Tuna pâté per FU (1 tonne of raw tuna at processing plant).

SUBSYSTEM 4. TUNA PÂTÉ PRODUCTION					
Inputs from the Technosphere					
Materials	**Unit**	**Value**	**Transport**	**Unit**	**Value**
Edible by-products from SS2	kg	8	Olive	t·km	3.96
Olive	kg	3.4	EVOO	t·km	1.57
EVOO	kg	1.8	Mashed potatoes	t·km	1.62
Mashed potatoes	kg	1.4	Onion	kg·km	180
Onion	kg	0.15	Garlic	kg·km	180
Potato starch	kg	0.15	Potato starch	kg·km	180
Garlic	kg	0.15	Black pepper	kg·km	180
Black pepper	kg	0.15	Salt	kg·km	180
Salt	kg	0.15	Glass jar	kg·km	298.3
Glass jar	kg	18.0	Aluminium lid	kg·km	16.3
Aluminium lid	g	984.6	Plastic label (PP)	kg·km	1.0
Plastic label (PP)	g	61.5	Plastic film (LDPE)	kg·km	14.7
Plastic film (LDPE)	g	183.3	Corrugated board	kg·km	59.4
Corrugated board	kg	3.3	Cardboard waste	kg·km	21.4
Water	L	679.8	Plastic waste	kg·km	165.1
Energy	**Unit**	**Value**			
Electricity	kWh	11.1			
Natural gas	kWh	158.3			
Outputs to the Technosphere					
Products	**Unit**	**Value**	**Waste to treatment**	**Unit**	**Value**
Tuna pâté jar	Amount	123	Biowaste	kg	1.06
			Plastic to recycling	g	171.0
			Cardboard to recycling	kg	1.32
			Wastewater	L	679.8

Table D.1 summarises the characteristics of the inventoried farms. Table D.2 shows the allocation factors calculated for each farm. Table D.3 and Table D.4 show the DEA matrix and the DEA results, respectively **(Chapter 8)**.

Table D.1. Main products and cattle heads of the 96 evaluated farms

Farms	Products		Animals			
	FPCM (kg/year)	Beef (kg/year)	Dairy cows	Dry cows	Culled cows	Heifers
1	226,466	900	23	3	3	26
2	510,631	3,000	44	5	10	14
3	215,226	1,800	27	4	6	4
4	243,552	2,100	30	4	7	10
5	630,727	6,000	54	10	20	55
6	477,765	1,200	50	8	4	30
7	181,812	1,200	23	7	4	9
8	595,360	0	60	7	0	35
12	623,216	1,200	59	6	4	45
13	899,870	6,000	80	1	20	45
14	658,893	4,200	60	10	14	30
15	709,046	6,000	72	10	20	48
16	456,667	0	38	6	0	40
17	310,672	0	35	5	0	20
18	3,012,728	0	225	30	0	180
19	3,007,328	19,500	230	30	65	195
20	733,730	2,400	55	14	8	50
22	407,305	900	38	5	3	30
23	1,229,972	9,000	100	16	30	70
24	1,244,570	4,500	105	7	15	70
25	462,741	2,400	40	4	8	29
26	1,497,688	9,000	110	20	30	70
27	1,188,309	15,000	99	14	50	94
28	1,407,251	7,500	155	25	25	130
29	458,336	3,000	41	4	10	25
30	1,017,543	3,000	104	10	10	60
31	1,430,496	15,000	130	15	50	115
32	1,078,899	7,500	95	15	25	60
34	184,999	1,200	25	4	4	8
35	485,833	3,000	40	15	10	35
38	355,177	1,800	40	4	6	32

Table D.1. (Cont.). Main products and cattle heads of the 96 evaluated farms

Farms	Products		Animals			
	FPCM (kg/year)	Beef (kg/year)	Dairy cows	Dry cows	Culled cows	Heifers
39	130,881	600	16	2	2	8
40	457,182	3,900	58	14	13	50
41	263,671	2,100	30	6	7	25
42	734,554	4,500	65	6	15	49
43	1,127,645	4,500	94	9	15	70
44	146,990	900	16	4	3	19
45	253,872	1,800	35	5	6	23
46	479,043	3,000	35	7	10	32
48	45,386	600	10	3	2	4
50	737,574	3,000	65	5	10	40
51	303,778	1,500	35	6	5	20
52	1,377,536	7,500	112	9	25	110
53	267,609	1,800	36	7	6	8
54	132,013	1,500	20	10	5	13
55	91,752	600	12	3	2	4
56	377,333	2,700	40	6	9	20
57	614,156	6,600	47	8	22	40
58	694,135	6,000	60	12	20	40
59	273,986	1,800	31	6	6	16
62	463,789	3,000	40	6	10	25
63	180,545	600	18	2	2	8
64	863,286	3,000	78	5	10	43
65	358,147	900	36	7	3	37
66	368,212	2,400	46	5	8	11
67	1,027,494	5,100	93	14	17	0
68	278,750	900	40	5	3	11
69	182,853	2,400	26	5	8	9
70	1,539,545	6,000	120	15	20	30
71	359,308	1,800	32	7	6	16
72	824,275	3,000	60	10	10	37
73	687,514	5,400	66	7	18	53
74	157,670	900	26	3	3	11
75	431,785	4,200	45	5	14	29
76	158,565	600	22	3	2	11
77	1,085,516	6,000	89	9	20	5

Table D.1. (Cont.). Main products and cattle heads of the 96 evaluated farms

Farms	Products		Animals			
	FPCM (kg/year)	Beef (kg/year)	Dairy cows	Dry cows	Culled cows	Heifers
78	430,366	1,200	48	3	4	31
79	897,436	2,100	76	7	7	40
80	766,836	3,000	65	10	10	62
81	1,842,248	12,000	154	11	40	9
82	449,963	1,800	35	1	6	26
83	1,244,519	10,200	102	12	34	67
84	2,309,842	12,000	170	25	40	0
85	41,139	1,500	14	2	5	18
86	819,717	6,000	65	10	20	30
88	184,385	1,500	26	2	5	15
89	205,679	3,600	27	10	12	42
90	65,376	600	10	2	2	2
91	204,702	900	21	1	3	10
92	513,979	600	53	15	2	60
93	203,130	2,700	28	8	9	48
94	75,984	900	8	1	3	7
95	20,200	600	7	1	2	3
96	257,472	2,400	27	2	8	19
97	60,952	300	13	1	1	4
98	283,056	2,100	34	8	7	14
99	81,208	600	13	4	2	4
100	941,822	5,100	84	10	17	110
101	253,980	900	22	4	3	16
102	57,187	300	10	1	1	3
103	149,318	600	17	6	2	15
104	164,006	900	19	3	3	12
105	227,138	1,200	28	1	4	22
106	1,431,906	6,000	119	21	20	45
107	883,500	3,600	78	9	12	60
108	864,076	7,500	78	12	25	59

Table D.2. Mass, economic, and biological allocation factors for the different products

Farms	Mass allocation		Economic allocation		Biological allocation	
	Milk	Beef	Milk	Beef	Milk	Beef
1	1.00	0.00	0.96	0.04	0.98	0.02
2	0.99	0.01	0.94	0.06	0.96	0.04
3	0.99	0.01	0.92	0.08	0.95	0.05
4	0.99	0.01	0.92	0.08	0.95	0.05
5	0.99	0.01	0.91	0.09	0.94	0.06
6	1.00	0.00	0.98	0.02	0.98	0.02
7	0.99	0.01	0.94	0.06	0.96	0.04
8	1.00	0.00	1.00	0.00	1.00	0.00
12	1.00	0.00	0.98	0.02	0.99	0.01
13	0.99	0.01	0.94	0.06	0.96	0.04
14	0.99	0.01	0.94	0.06	0.96	0.04
15	0.99	0.01	0.92	0.08	0.95	0.05
16	1.00	0.00	1.00	0.00	1.00	0.00
17	1.00	0.00	1.00	0.00	1.00	0.00
18	1.00	0.00	1.00	0.00	1.00	0.00
19	0.99	0.01	0.94	0.06	0.96	0.04
20	1.00	0.00	0.97	0.03	0.98	0.02
22	1.00	0.00	0.98	0.02	0.99	0.01
23	0.99	0.01	0.93	0.07	0.95	0.05
24	1.00	0.00	0.96	0.04	0.98	0.02
25	0.99	0.01	0.95	0.05	0.97	0.03
26	0.99	0.01	0.94	0.06	0.96	0.04
27	0.99	0.01	0.89	0.11	0.92	0.08
28	0.99	0.01	0.95	0.05	0.97	0.03
29	0.99	0.01	0.94	0.06	0.96	0.04
30	1.00	0.00	0.97	0.03	0.98	0.02
31	0.99	0.01	0.90	0.10	0.93	0.07
32	0.99	0.01	0.93	0.07	0.96	0.04
34	0.99	0.01	0.94	0.06	0.96	0.04
35	0.99	0.01	0.94	0.06	0.96	0.04
38	0.99	0.01	0.95	0.05	0.97	0.03
39	1.00	0.00	0.96	0.04	0.97	0.03
40	0.99	0.01	0.92	0.08	0.95	0.05
41	0.99	0.01	0.92	0.08	0.95	0.05
42	0.99	0.01	0.94	0.06	0.96	0.04

Table D.2. (Cont.). Mass, economic, and biological allocation factors for the different products

Farms	Mass allocation		Economic allocation		Biological allocation	
	Milk	Beef	Milk	Beef	Milk	Beef
43	1.00	0.00	0.96	0.04	0.98	0.02
44	0.99	0.01	0.94	0.06	0.96	0.04
45	0.99	0.01	0.93	0.07	0.96	0.04
46	0.99	0.01	0.94	0.06	0.96	0.04
48	0.99	0.01	0.88	0.12	0.92	0.08
50	1.00	0.00	0.96	0.04	0.97	0.03
51	0.99	0.01	0.95	0.05	0.97	0.03
52	0.99	0.01	0.95	0.05	0.97	0.03
53	0.99	0.01	0.94	0.06	0.96	0.04
54	0.99	0.01	0.90	0.10	0.93	0.07
55	0.99	0.01	0.94	0.06	0.96	0.04
56	0.99	0.01	0.93	0.07	0.96	0.04
57	0.99	0.01	0.90	0.10	0.93	0.07
58	0.99	0.01	0.92	0.08	0.95	0.05
59	0.99	0.01	0.94	0.06	0.96	0.04
62	0.99	0.01	0.94	0.06	0.96	0.04
63	1.00	0.00	0.97	0.03	0.98	0.02
64	1.00	0.00	0.97	0.03	0.98	0.02
65	1.00	0.00	0.98	0.02	0.98	0.02
66	0.99	0.01	0.94	0.06	0.96	0.04
67	0.99	0.01	0.95	0.05	0.97	0.03
68	1.00	0.00	0.97	0.03	0.98	0.02
69	0.99	0.01	0.88	0.12	0.92	0.08
70	1.00	0.00	0.96	0.04	0.98	0.02
71	0.99	0.01	0.95	0.05	0.97	0.03
72	1.00	0.00	0.96	0.04	0.98	0.02
73	0.99	0.01	0.93	0.07	0.95	0.05
74	0.99	0.01	0.94	0.06	0.96	0.04
75	0.99	0.01	0.91	0.09	0.94	0.06
76	1.00	0.00	0.96	0.04	0.98	0.02
77	0.99	0.01	0.95	0.05	0.97	0.03
78	1.00	0.00	0.97	0.03	0.98	0.02
79	1.00	0.00	0.98	0.02	0.99	0.01
80	1.00	0.00	0.96	0.04	0.98	0.02
81	0.99	0.01	0.94	0.06	0.96	0.04

Table D.2. (Cont.). Mass, economic, and biological allocation factors for the different products

Farms	Mass allocation		Economic allocation		Biological allocation	
	Milk	Beef	Milk	Beef	Milk	Beef
82	1.00	0.00	0.96	0.04	0.98	0.02
83	0.99	0.01	0.92	0.08	0.95	0.05
84	0.99	0.01	0.95	0.05	0.97	0.03
85	0.96	0.04	0.73	0.27	0.77	0.23
86	0.99	0.01	0.93	0.07	0.95	0.05
88	0.99	0.01	0.92	0.08	0.95	0.05
89	0.98	0.02	0.85	0.15	0.89	0.11
90	0.99	0.01	0.91	0.09	0.94	0.06
91	1.00	0.00	0.96	0.04	0.97	0.03
92	1.00	0.00	0.99	0.01	0.99	0.01
93	0.99	0.01	0.88	0.12	0.92	0.08
94	0.99	0.01	0.89	0.11	0.93	0.07
95	0.97	0.03	0.77	0.23	0.81	0.19
96	0.99	0.01	0.91	0.09	0.94	0.06
97	0.99	0.01	0.95	0.05	0.97	0.03
98	0.99	0.01	0.93	0.07	0.95	0.05
99	0.99	0.01	0.93	0.07	0.95	0.05
100	0.99	0.01	0.95	0.05	0.97	0.03
101	1.00	0.00	0.97	0.03	0.98	0.02
102	0.99	0.01	0.95	0.05	0.97	0.03
103	1.00	0.00	0.96	0.04	0.97	0.03
104	0.99	0.01	0.95	0.05	0.97	0.03
105	0.99	0.01	0.95	0.05	0.97	0.03
106	1.00	0.00	0.96	0.04	0.97	0.03
107	1.00	0.00	0.96	0.04	0.97	0.03
108	0.99	0.01	0.92	0.08	0.95	0.05

Table D.3. DEA matrix for the complete set of farms under assessment

Farms	Inputs							Undesirable outputs				Products
	Concentrate (kg)	Grass silage (kg)	Maize silage (kg)	Electricity (kWh)	Diesel (kg)	Silage plastic (kg)	Water (m^3)	CH_4 (kg)	N_2O (kg)	NH_3 (kg)	Wastewater (m^3)	FPCM (kg)
1	113,515	494,575	0	9,852	2,036	135	2,013	5,208	365	1,150	220	219,444
2	174,470	526,330	321,200	20,295	2,394	225	1,558	7,869	741	3,591	584	494,797
3	53,655	184,325	246,375	9,863	1,234	295	1,370	4,582	144	992	290	208,552
4	57,670	375,950	0	14,051	1,151	3,750	1,455	5,445	220	1,455	290	236,000
5	217,540	592,030	492,750	94,436	6,483	320	2,191	12,869	1,145	6,365	720	611,170
6	246,740	392,740	456,250	20,944	2,948	630	2,590	9,530	351	2,075	630	462,951
7	112,420	179,580	209,875	16,943	1,772	315	1,874	4,428	188	977	292	176,175
8	271,560	368,285	777,450	29,472	4,856	480	3,875	10,890	560	3,006	560	576,899
12	281,963	246,558	538,375	43,185	3,411	250	2,885	11,599	450	2,220	400	603,891
13	350,400	770,150	992,800	34,976	5,160	480	3,196	15,170	815	3,454	720	871,967
14	259,150	481,800	653,350	25,030	1,835	350	2,873	11,670	382	2,547	375	638,463
15	283,970	509,540	1,264,360	35,423	4,409	640	3,557	14,862	633	3,223	365	687,060
16	170,090	265,720	554,800	17,436	2,085	270	1,661	8,158	243	1,285	182	442,507
17	149,650	120,450	428,875	14,364	3,189	250	1,097	6,385	263	1,206	230	301,038
18	1,215,450	1,817,700	3,312,375	74,360	25,438	480	10,561	44,250	1,685	10,079	1,200	2,919,310
19	975,463	3,027,675	2,283,075	55,889	10,165	814	10,345	50,130	2,057	13,841	730	2,914,077
20	292,730	379,600	775,625	45,471	3,644	370	3,002	12,160	589	3,130	400	710,979

Table D.3. (Cont.). DEA matrix for the complete set of farms under assessment

Farms	Inputs							Undesirable outputs				Products
	Concentrate (kg)	Grass silage (kg)	Maize silage (kg)	Electricity (kWh)	Diesel (kg)	Silage plastic (kg)	Water (m^3)	CH_4 (kg)	N_2O (kg)	NH_3 (kg)	Wastewater (m^3)	FPCM (kg)
22	177,025	150,380	605,900	20,944	667	250	1,861	7,638	405	2,080	185	394,675
23	512,460	1,401,600	1,020,540	41,889	1,136	885	4,895	21,140	1,658	7,441	547	1,191,833
24	531,440	1,022,000	1,366,925	35,479	278	560	4,074	20,260	1,304	6,607	365	1,205,979
25	183,595	272,290	568,670	15,641	941	423	1,774	8,105	609	2,354	365	448,392
26	540,200	1,259,250	1,324,950	37,500	1,959	722	4,299	22,760	943	4,933	511	1,451,248
27	483,260	367,920	1,839,600	36,627	2,525	1,200	5,258	23,734	1,065	5,839	1,095	1,151,463
28	726,350	1,627,900	1,810,400	48,474	3,997	1,050	8,210	32,780	1,513	6,912	1,280	1,363,615
29	155,855	191,260	524,870	12,283	1,423	240	2,103	8,111	373	1,845	365	444,124
30	449,680	337,260	1,190,630	38,247	3,048	934	4,540	19,344	1,113	4,978	548	985,991
31	706,275	574,875	1,761,125	24,428	5,487	800	5,968	29,380	1,486	7,134	550	1,386,140
32	428,875	830,375	1,177,125	40,854	6,851	840	4,519	19,420	878	4,767	547	1,045,444
34	77,015	231,775	0	8,879	960	230	1,213	4,440	387	1,422	1,000	179,262
35	233,600	613,200	365,000	14,453	2,479	270	2,242	9,340	470	2,402	548	470,768
38	185,420	470,120	292,000	11,264	2,469	525	2,085	8,170	423	2,004	584	344,164
39	49,640	195,640	0	8,043	1,061	180	919	2,956	127	546	401	126,823
40	334,340	423,400	275,210	22,732	1,870	500	2,839	12,893	518	2,733	584	443,006
41	108,405	408,800	0	10,082	1,954	210	1,398	6,550	297	1,307	292	711,778

Table D.3. (Cont.). DEA matrix for the complete set of farms under assessment

Farms	Inputs							Undesirable outputs				Products
	Concentrate (kg)	Grass silage (kg)	Maize silage (kg)	Electricity (kWh)	Diesel (kg)	Silage plastic (kg)	Water (m³)	CH_4 (kg)	N_2O (kg)	NH_3 (kg)	Wastewater (m³)	FPCM (kg)
42	412,450	593,125	427,050	20,862	4,466	745	3,151	13,390	463	2,862	730	711,778
43	477,420	596,775	830,010	27,588	1,641	495	3,880	18,894	1,318	6,781	730	1,092,679
44	54,020	202,210	0	7,328	1,283	105	859	3,866	382	1,670	310	142,432
45	131,035	366,825	0	12,928	1,192	180	1,990	6,970	403	1,922	657	246,000
46	163,520	462,820	204,400	14,087	1,627	252	1,632	7,945	363	1,690	328	464,189
48	25,915	122,275	0	5,869	974	372	476	1,945	94	285	182	43,978
50	341,275	487,275	711,750	14,997	1,742	325	3,282	12,415	656	4,186	912	714,703
51	135,780	428,875	0	14,617	2,254	324	1,509	6,775	221	1,020	365	294,358
52	499,685	1,340,645	704,815	30,682	4,327	866	5,604	24,592	1,105	6,864	1,095	1,334,822
53	110,960	355,875	0	7,623	1,376	230	1,521	6,261	291	1,555	401	259,311
54	75,190	346,750	0	6,911	1,639	160	982	4,540	295	870	292	127,919
55	30,660	112,420	0	6,222	1,488	130	888	2,217	87	410	547	88,907
56	135,780	643,860	0	9,493	1,204	370	1,909	7,715	356	1,754	438	365,632
57	342,370	730,000	0	16,144	5,099	360	2,758	10,942	760	4,347	750	595,112
58	255,500	562,100	284,700	19,309	1,957	843	2,771	12,840	349	2,593	547	672,611
59	102,200	310,250	169,725	12,713	1,794	200	1,696	6,036	252	1,406	700	265,490
62	151,840	675,000	0	20,061	879	406	3,054	8,105	368	2,193	1,168	449,408

Table D.3. (Cont.). DEA matrix for the complete set of farms under assessment

Farms	Inputs							Undesirable outputs				Products
	Concentrate (kg)	Grass silage (kg)	Maize silage (kg)	Electricity (kWh)	Diesel (kg)	Silage plastic (kg)	Water (m^3)	CH_4 (kg)	N_2O (kg)	NH_3 (kg)	Wastewater (m^3)	FPCM (kg)
63	65,700	206,590	0	5,232	74	260	1,386	3,228	210	1,087	547	174,946
64	406,975	669,410	683,280	32,333	532	986	4,364	14,378	657	4,324	1,270	836,517
65	134,685	292,730	316,820	9,862	1,710	977	2,232	7,951	274	1,168	730	347,042
66	120,450	417,560	409,530	10,936	2,060	1,140	2,364	7,816	450	1,916	620	356,794
67	563,560	916,515	407,340	52,131	8,556	1,137	4,273	14,663	580	4,095	829	995,634
68	128,480	292,000	0	15,789	10	450	2,054	6,675	387	1,517	529	270,106
69	96,360	189,800	0	8,000	3,934	288	1,169	4,966	193	913	803	177,183
70	224,475	438,000	1,253,775	62,213	2,161	0	6,047	20,545	1,116	6,464	985	1,491,807
71	133,590	186,880	490,560	10,574	2,758	475	1,436	6,237	306	1,288	427	348,166
72	319,010	501,875	542,025	33,978	2,187	1,650	2,913	11,865	548	2,851	492	798,716
73	289,080	517,570	602,250	26,817	2,669	260	2,672	14,046	553	3,233	438	666,196
74	68,985	280,320	0	7,181	1,797	430	1,514	4,641	199	1,071	401	152,781
75	205,495	187,975	580,350	11,108	1,712	1,415	2,442	9,240	464	2,238	985	418,397
76	74,460	105,850	231,775	6,157	1,348	880	1,660	4,032	238	1,088	949	153,648
77	372,665	420,480	1,230,780	31,586	1,490	2,740	4,306	14,314	853	4,441	1,040	1,051,856
78	205,130	433,985	438,000	17,791	783	680	2,954	8,998	358	1,936	1,405	417,021
79	339,450	388,360	1,113,250	24,700	1,959	2,560	3,833	13,846	799	3,074	876	869,609

Table D.3. (Cont.). DEA matrix for the complete set of farms under assessment

Farms	Inputs							Undesirable outputs				Products
	Concentrate (kg)	Grass silage (kg)	Maize silage (kg)	Electricity (kWh)	Diesel (kg)	Silage plastic (kg)	Water (m^3)	CH_4 (kg)	N_2O (kg)	NH_3 (kg)	Wastewater (m^3)	FPCM (kg)
80	333,610	441,650	835,850	31,214	1,897	895	6,028	14,170	570	3,278	3,285	743,058
81	580,715	988,420	1,967,350	51,653	2,493	0	6,309	24,844	1,479	9,095	1,168	1,785,124
82	160,965	166,805	443,475	17,583	2,705	650	2,066	6,905	321	1,376	657	436,011
83	458,805	855,195	1,178,220	44,464	105	3,696	4,906	21,217	1,054	5,152	785	1,205,929
84	647,875	496,400	1,675,350	62,690	1,964	1,927	7,193	27,345	1,629	9,216	913	2,238,219
85	0	58,400	0	3,992	1,423	275	919	3,529	184	622	492	39,863
86	266,450	472,675	1,047,550	23,942	1,571	2,137	3,194	12,740	698	4,162	657	794,299
88	74,825	129,575	151,110	13,777	1,219	1,120	1,102	4,966	311	1,030	255	178,668
89	73,730	295,650	197,100	6,651	784	1,818	1,293	7,832	537	2,206	292	199,301
90	32,120	56,940	0	7,680	637	415	892	1,750	131	430	584	63,348
91	61,320	155,125	207,685	6,520	1,050	796	1,108	3,766	364	1,616	438	198,355
92	308,790	621,960	270,830	21,949	872	1,012	3,433	12,213	730	3,707	1,168	498,042
93	119,720	305,140	0	10,399	743	956	1,498	8,033	251	1,734	547	196,831
94	28,835	114,975	0	3,591	1,258	448	706	1,803	120	492	438	73,628
95	9,125	40,150	0	2,715	719	466	494	1,342	65	423	328	19,573
96	82,855	269,005	217,905	8,717	2,669	430	1,434	5,557	311	1,578	438	249,489
97	28,835	89,425	0	4,647	250	372	670	2,158	96	535	219	59,062

Table D.3. (Cont.). DEA matrix for the complete set of farms under assessment

Farms	Inputs							Undesirable outputs				Products
	Concentrate (kg)	Grass silage (kg)	Maize silage (kg)	Electricity (kWh)	Diesel (kg)	Silage plastic (kg)	Water (m^3)	CH_4 (kg)	N_2O (kg)	NH_3 (kg)	Wastewater (m^3)	FPCM (kg)
98	72,270	328,500	124,100	10,821	1,772	1,420	1,614	6,509	497	1,882	522	274,280
99	45,260	178,850	0	4,625	278	545	727	2,418	131	609	321	78,690
100	337,260	398,580	766,500	34,178	1,406	1,720	4,311	20,329	1,090	4,582	621	912,618
101	109,500	193,450	160,600	8,174	1,024	1,240	1,309	4,487	403	1,428	401	246,104
102	32,120	68,985	42,705	6,871	737	380	837	1,685	79	341	529	55,414
103	73,365	253,675	0	5,564	1,282	1,032	860	3,807	247	1,196	292	144,688
104	149,650	324,120	0	7,117	1,082	930	1,030	3,754	152	861	401	158,920
105	98,550	332,150	255,500	6,929	1,022	1,600	1,357	5,563	362	1,996	438	220,095
106	550,055	762,120	1,520,225	21,133	2,602	2,448	5,984	21,774	752	4,046	912	1,387,505
107	388,725	440,190	1,011,780	21,148	1,829	2,290	4,674	15,873	756	3,556	1,186	856,105
108	393,105	745,330	788,400	43,213	2,289	1,530	4,165	16,032	1,106	5,362	912	837,283

Table D.4. Efficiency scores (Φ) and operational reduction percentages for the inefficient farms

	Φ	Concentrate	Grass silage	Maize silage	Electricity	Diesel	Silage plastic	Water	CH_4	N_2O	NH_3	Wastewater
Farm 1	0.33	70.6	74.5	0	68.5	70.4	52	78.6	62.6	62.6	62.6	59.1
Farm 2	0.68	37.1	32.2	0	51.8	37.8	16.2	20.5	39.4	39.4	39.4	56.5
Farm 3	0.43	40.8	35	35.7	70.1	53.6	79.1	70.1	58.2	58.2	58.2	70.5
Farm 4	0.35	37.7	64	0	76.2	43.8	98.1	68.1	62	62	62	66.6
Farm 5	0.46	57.2	40.7	5.7	90.8	74.1	43.7	45.2	65.7	65.7	65.7	65.2
Farm 6	0.42	71.4	32.3	22.9	68.7	56.9	78.3	64.9	55.6	55.6	55.6	69.9
Farm 7	0.31	76.1	43.7	36.2	85.3	72.7	83.5	81.5	63.9	63.9	63.9	75.3
Farm 8	0.44	67.7	10	43.6	72.3	67.4	64.5	70.8	54.3	54.3	54.3	57.7
Farm 12	0.61	67.8	28.1	5.7	41.7	74.4	100	15.2	20.2	20.2	20.2	0.3
Farm 13	0.51	62.1	35	33.2	64.7	53.6	46.4	46.4	48.6	48.6	48.6	50.3
Farm 14	0.6	62.5	23.9	25.7	63.9	4.5	46.2	56.3	49.9	49.9	49.9	30.2
Farm 15	0.46	63.2	22.6	58.7	72.5	57.2	68.3	62.1	58	58	58	22.8
Farm 16	0.6	60.4	4.4	39.4	64.1	41.7	51.7	47.7	47.7	47.7	47.7	0.3
Farm 17	0.56	55.7	0	47.1	56	82.8	30.8	29.2	45.5	45.5	45.5	46.5
Farm 20	0.58	63.1	0	29.3	71.7	50.1	51.5	46.4	43.9	43.9	43.9	20.7
Farm 22	0.7	44	33.9	49.8	41.9	36	0	28.5	32.6	32.6	32.6	0
Farm 23	0.78	30.1	69.7	5.2	23.5	0	0	24	30.9	30.9	30.9	14.8

Table D.4. (Cont.). Efficiency scores (Φ) and operational reduction percentages for the inefficient farms

	Φ	Concentrate	Grass silage	Maize silage	Electricity	Diesel	Silage plastic	Water	CH_4	N_2O	NH_3	Wastewater
Farm 25	0.62	51.2	25.5	40.4	45.7	0	48	39.5	45.7	45.7	45.7	49.7
Farm 26	0.94	14.8	25.6	0	7.1	0	1.2	0	6	6	6	0
Farm 27	0.64	31	30.6	53.2	12	60	17.4	29.6	35.9	35.9	35.9	57.1
Farm 27	0.47	71.4	51.9	42.8	60.2	6.4	61.7	67.4	62.1	62.1	62.1	56.3
Farm 28	0.67	55.7	0	32.5	0	35.3	68.3	36.8	34.9	34.9	34.9	35.6
Farm 29	0.65	46.8	0	37.7	40	51	29.5	39.5	41.3	41.3	41.3	26.5
Farm 30	0.51	62.9	27.7	32.5	63.8	58.1	63.3	54.6	52.2	52.2	52.2	21.6
Farm 32	0.31	64.6	55.6	0	71.4	48.8	77	71	67.1	67.1	67.1	92.7
Farm 34	0.49	69.3	55.9	2	53.9	47.9	48.6	58.7	55.8	55.8	55.8	64.7
Farm 35	0.39	71.7	58	10.4	56.7	61.7	80.7	67.6	62.8	62.8	62.8	75.8
Farm 38	0.28	61.1	62.8	0	77.7	67.2	79.2	72.9	60	60	60	87
Farm 39	0.54	68.3	20.7	0	58.7	27.2	64.3	58.8	56.6	56.6	56.6	59.1
Farm 40	0.66	56.4	5.7	0	24	50.2	60	38.1	29.7	29.7	29.7	45.5
Farm 42	0.31	59.8	59.6	0	72.5	69.5	60	67.4	72.4	72.4	72.4	81.2
Farm 44	0.33	71.4	61.5	0	73.1	43.4	59.7	75.7	69.7	69.7	69.7	84.6
Farm 45	0.14	74.2	79.3	0	89.4	87.6	96.5	81.8	78.3	78.3	78.3	90.1
Farm 48	0.62	51.2	25.5	40.4	45.7	0	48	39.5	45.7	45.7	45.7	49.7

Table D.4. (Cont.). Efficiency scores (Φ) and operational reduction percentages for the inefficient farms

	Φ	Concentrate	Grass silage	Maize silage	Electricity	Diesel	Silage plastic	Water	CH_4	N_2O	NH_3	Wastewater
Farm 50	0.64	68.2	25.5	21.8	1.4	0	50.5	46.4	45.3	45.3	45.3	63.2
Farm 51	0.35	67	60.6	0	71.5	64.2	73.2	61.7	57.9	57.9	57.9	66.9
Farm 53	0.38	64.4	58.2	0	51.8	48.3	66.7	66.5	63.4	63.4	63.4	73.5
Farm 54	0.23	74.1	78.8	0	73.8	78.6	76.4	74.4	74.3	74.3	74.3	82
Farm 55	0.25	55.8	54.6	0	79.8	83.6	79.8	80.3	62.5	62.5	62.5	93.3
Farm 56	0.45	59	67.4	0	45.4	16.7	70.8	62.4	57.4	57.4	57.4	65.8
Farm 57	0.45	73.5	53.2	0	47.8	68	51.2	57.6	57.5	57.5	57.5	67.5
Farm 59	0.53	40.8	37.2	0	57.7	55.1	48.4	60.1	47.3	47.3	47.3	80.2
Farm 62	0.51	55.3	73.2	0	30.7	0	87.2	52.4	32.7	32.7	32.7	78.4
Farm 65	0.44	60.8	31.9	16.7	50.2	44.3	89.5	69.5	57.7	57.7	57.7	80.5
Farm 66	0.4	54.9	50.9	33.8	53.8	52.5	90.8	70.4	59.9	59.9	59.9	76.4
Farm 69	0.27	72	46.4	0	68.6	87.6	81.9	70.2	66.6	66.6	66.6	91
Farm 71	0.49	56.5	0	46.1	48.5	67.9	74	49.1	46.4	46.4	46.4	66.6
Farm 72	0.72	44.4	0	0	52.5	2.8	78.7	30.5	22.6	22.6	22.6	21.5
Farm 73	0.59	64.9	26.1	15.9	64.8	31.5	24.4	51	57.2	57.2	57.2	37.6
Farm 74	0.24	66.3	68.7	0	69.9	76.7	89.5	80.2	70.4	70.4	70.4	84.4
Farm 75	0.49	59.1	0	45.5	28.2	49.1	85.3	58.8	52.9	52.9	52.9	82.6

Table D.4. (Cont.). Efficiency scores (Φ) and operational reduction percentages for the inefficient farms

	Φ	Concentrate	Grass silage	Maize silage	Electricity	Diesel	Silage plastic	Water	CH_4	N_2O	NH_3	Wastewater
Farm 76	0.33	68.6	16.6	49.6	64.7	68.7	94.9	81.8	67.2	67.2	67.2	93.4
Farm 77	0.67	18.3	44.5	36	6.7	38.1	67	21.5	8.5	8.5	8.5	58.7
Farm 78	0.49	41.2	78.7	28.7	34.4	53.3	47.2	54.6	37	37	37	87.9
Farm 78	0.61	25.9	50.3	41.5	1.4	61.1	70.8	27.1	16.3	16.3	16.3	59.5
Farm 80	0.5	62.9	9.5	32.5	62.9	0	70.7	74.2	50.4	50.4	50.4	90.7
Farm 82	0.57	38.5	0	25.9	46.2	72.1	59.6	44.2	26.5	26.5	26.5	72.9
Farm 86	0.61	37.8	27.8	42.7	34.2	0	80.4	38.4	37.9	37.9	37.9	50.5
Farm 88	0.4	63.6	20.8	10.1	81.6	59.8	95.3	68.1	67.6	67.6	67.6	71.3
Farm 89	0.39	58.8	61.3	23.1	57.6	30.2	96.8	69.7	78.4	78.4	78.4	72
Farm 90	0.22	70	36.1	0	88.3	72.7	95.5	86	68.6	68.6	68.6	95.6
Farm 91	0.43	50.7	26.6	27.4	56.9	48.1	92.7	64.8	60.5	60.5	60.5	81.4
Farm 93	0.29	75	63	0	73.2	27.3	93.9	74.2	77.5	77.5	77.5	85.2
Farm 94	0.22	61.1	63.2	0	71	83.9	95.2	79.5	65.1	65.1	65.1	93.1
Farm 95	0.12	67.3	72	0	89.8	92.5	98.8	92.2	87.8	87.8	87.8	97.6
Farm 96	0.41	54.1	46.7	13	59.5	74.4	82.9	65.8	61.6	61.6	61.6	76.6
Farm 97	0.24	68.8	62.1	0	82	35.1	95.3	82.7	75.7	75.7	75.7	88.9
Farm 99	0.25	73.5	74.7	0	75.9	22.3	95.7	78.7	71.5	71.5	71.5	89.9

Table D.4. (Cont.). Efficiency scores (Φ) and operational reduction percentages for the inefficient farms

	Φ	Concentrate	Grass silage	Maize silage	Electricity	Diesel	Silage plastic	Water	CH_4	N_2O	NH_3	Wastewater
Farm 100	0.65	21.7	49.2	10.9	25.2	43	54.3	32	40.1	40.1	40.1	40.1
Farm 101	0.57	49.4	16.3	0	43.4	32.8	88.1	50.1	40.5	40.5	40.5	66.9
Farm 102	0.28	73.7	53.9	1.4	88.6	79.4	95.7	87	69.9	69.9	69.9	95.7
Farm 103	0.27	70	67.2	0	63.2	69	95.9	67	68.4	68.4	68.4	79.7
Farm 104	0.25	83.8	71.8	0	68.4	59.7	95	69.7	61.8	61.8	61.8	83.7
Farm 105	0.37	66	61.9	34.5	55	40.9	95.9	68.1	68.2	68.2	68.2	79.4
Farm 107	0.58	43.5	40.5	36.4	0	38.1	72.9	46.6	33.3	33.3	33.3	70.5
Farm 108	0.49	67.4	35.7	19.3	72.4	0	83.7	60.4	57.3	57.3	57.3	62.3

Table E.1 shows the DEA matrix. Tables E.2, E.3 and E.4 summarise the environmental results of the small, medium and large-size farms, respectively. Table E.5 include the objective reduction percentage in operating income for shrimp farms considered inefficient **(Chapter 9)**.

Table E.1. DEA matrix for the complete set of shrimp farms under assessment

DMU	O	I-1	I-2	I-3	I-4	I-5	I-6	I-7
1	401.9	1,150,848	795.7	29.5	4,539,624.4	55,702.0	30.1	11.9
2	1,223.3	2,697,300	2,544.5	155.9	10,639,744.8	178,112.5	98.0	38.2
3	187.9	479,520	362.6	15.6	1,891,510.2	25,382.6	13.6	5.4
4	81.0	479,520	140.1	4.3	1,891,510.2	9,809.1	5.0	2.1
5	183.9	549,450	353.1	15.3	2,167,355.4	24,717.5	13.2	5.2
6	230.0	751,648	370.3	12.6	2,964,942.2	25,924.4	12.8	5.4
7	391.1	1,789,209	1,032.6	36.3	7,057,697.4	72,279.0	42.7	15.9
8	431.5	899,100	988.0	40.6	3,546,581.6	69,163.0	39.3	15.0
9	2,170.0	4,040,955	4,209.8	104.7	15,939,913.9	294,686.0	158.2	62.6
10	244.9	707,292	499.5	17.8	2,789,977.5	34,967.4	19.1	7.5
11	72.4	499,500	122.4	8.6	1,970,323.1	8,564.9	4.3	1.8
12	206.2	531,468	381.5	16.3	2,096,423.8	26,702.9	14.1	5.6
13	2,273.5	4,569,426	3,819.5	121.3	18,024,515.8	267,363.6	135.1	55.7
14	951.9	1,808,190	2,303.5	68.0	7,132,569.6	161,243.4	93.0	35.1
15	1,088.9	2,364,633	2,068.8	72.0	9,327,509.6	144,818.4	77.1	30.7
16	792.0	1,858,140	1,702.8	81.6	7,329,601.9	119,196.0	66.3	25.7
17	2,007.5	4,195,800	4,235.8	125.6	16,550,714.1	296,507.8	164.0	63.7
18	187.9	1,080,918	300.6	9.6	4,263,779.2	21,039.2	10.4	4.3
19	1,025.6	2,766,231	2,153.8	69.3	10,911,649.4	150,766.1	83.3	32.4
20	832.9	1,993,005	1,624.2	35.2	7,861,589.2	113,694.9	61.1	24.2
21	176.1	949,050	417.3	15.3	3,743,613.9	29,210.0	16.8	6.4
22	220.9	654,345	388.7	12.2	2,581,123.3	27,212.4	14.0	5.7
23	14.0	319,680	35.0	3.6	1,261,006.8	2,450.0	1.4	0.5
24	154.6	659,340	327.7	11.6	2,600,826.5	22,936.7	12.7	4.9
25	216.8	847,152	411.9	14.1	3,341,668.0	28,831.7	15.4	6.1
26	204.3	759,240	357.5	12.2	2,994,891.1	25,024.3	12.9	5.2
27	296.6	989,010	602.2	20.5	3,901,239.7	42,152.5	23.0	9.0
28	273.1	809,190	480.7	15.6	3,191,923.4	33,650.8	17.4	7.1
29	39.0	339,660	74.1	3.1	1,339,819.7	5,187.0	2.8	1.1
30	191.2	585,414	325.1	10.5	2,309,218.7	22,757.6	11.6	4.7
31	111.5	969,030	340.1	10.5	3,822,426.8	23,805.3	14.5	5.3

Table E.1. (Cont.). DEA matrix for the complete set of shrimp farms under assessment

DMU	O	I-1	I-2	I-3	I-4	I-5	I-6	I-7
32	109.0	598,401	196.2	6.4	2,360,447.1	13,734.0	7.2	2.9
33	76.0	348,651	106.4	3.8	1,375,285.5	7445.1	3.4	1.5
34	37.0	574,425	66.6	4.5	226,5871.6	4,662.0	2.4	1.0
35	143.3	579,420	229.3	8.6	2,285,574.8	16,048.5	7.9	3.3
36	30.1	190,809	45.1	2.0	752,663.4	3,155.3	1.5	0.6
37	125.7	543,456	213.6	8.8	2,143,711.5	14,954.7	7.6	3.1
38	657.4	1,198,800	1,551.5	50.0	4,728,775.5	108,607.4	62.2	23.6

Table E.2. Environmental characterisation of an average small-size farm

		GW	SOD	TA	FE	ME	MET	FRS	WD
		kg CO_2	kg	kg SO_2	kg P eq	kg N eq	kg 1,4-DCB	kg oil eq	m^3
Total		$7.8 \cdot 10^{3}$	$1.8 \cdot 10^{-2}$	$4.1 \cdot 10^{1}$	$2.9 \cdot 10^{1}$	$2.4 \cdot 10^{0}$	$1.0 \cdot 10^{2}$	$1.8 \cdot 10^{3}$	$6.7 \cdot 10^{3}$
SS1. Feed	**Soybean**	$1.2 \cdot 10^{3}$	$3.5 \cdot 10^{-3}$	$1.3 \cdot 10^{0}$	$1.0 \cdot 10^{-1}$	$7.6 \cdot 10^{-1}$	$8.3 \cdot 10^{0}$	$4.6 \cdot 10^{1}$	$3.4 \cdot 10^{0}$
	Maize starch	$3.1 \cdot 10^{2}$	$3.4 \cdot 10^{-3}$	$2.4 \cdot 10^{0}$	$1.2 \cdot 10^{-1}$	$1.1 \cdot 10^{0}$	$1.4 \cdot 10^{1}$	$6.6 \cdot 10^{1}$	$8.4 \cdot 10^{0}$
	Soybean oil	$4.1 \cdot 10^{2}$	$1.1 \cdot 10^{-3}$	$4.2 \cdot 10^{-1}$	$3.8 \cdot 10^{-2}$	$2.6 \cdot 10^{-1}$	$3.0 \cdot 10^{0}$	$1.5 \cdot 10^{1}$	$4.4 \cdot 10^{0}$
	Water	$1.3 \cdot 10^{-1}$	$8.3 \cdot 10^{-8}$	$5.0 \cdot 10^{-4}$	$6.9 \cdot 10^{-5}$	$5.1 \cdot 10^{-6}$	$5.4 \cdot 10^{-3}$	$3.2 \cdot 10^{-2}$	$2.3 \cdot 10^{-1}$
	Wheat meal	$5.3 \cdot 10^{2}$	$5.9 \cdot 10^{-3}$	$3.8 \cdot 10^{0}$	$3.5 \cdot 10^{-1}$	$8.9 \cdot 10^{-1}$	$2.1 \cdot 10^{1}$	$1.1 \cdot 10^{2}$	$1.0 \cdot 10^{2}$
	Fish meal	$3.1 \cdot 10^{2}$	$-2.9 \cdot 10^{-3}$	$4.1 \cdot 10^{0}$	$-3.5 \cdot 10^{-2}$	$-8.9 \cdot 10^{-1}$	$-2.2 \cdot 10^{0}$	$1.4 \cdot 10^{2}$	$-6.2 \cdot 10^{0}$
	Fish oil	$7.9 \cdot 10^{1}$	$-7.3 \cdot 10^{-4}$	$1.1 \cdot 10^{0}$	$-8.9 \cdot 10^{-3}$	$-2.3 \cdot 10^{-1}$	$-5.6 \cdot 10^{-1}$	$3.7 \cdot 10^{1}$	$-1.6 \cdot 10^{0}$
SS2. Larvae	**Emissions**	0	0	0	0	0	0	0	$1.4 \cdot 10^{0}$
	Water	$3.2 \cdot 10^{0}$	$2.1 \cdot 10^{-6}$	$1.3 \cdot 10^{-2}$	$1.8 \cdot 10^{-3}$	$1.3 \cdot 10^{-4}$	$1.4 \cdot 10^{-1}$	$8.2 \cdot 10^{-1}$	$5.8 \cdot 10^{0}$
	Chlorine	$1.9 \cdot 10^{-1}$	$2.9 \cdot 10^{-7}$	$9.8 \cdot 10^{-4}$	$1.8 \cdot 10^{-4}$	$1.7 \cdot 10^{-5}$	$1.4 \cdot 10^{-2}$	$4.8 \cdot 10^{-2}$	$7.7 \cdot 10^{-3}$
	EDTA	$5.8 \cdot 10^{-1}$	$2.3 \cdot 10^{-7}$	$1.9 \cdot 10^{-3}$	$1.8 \cdot 10^{-4}$	$4.4 \cdot 10^{-4}$	$2.0 \cdot 10^{-2}$	$2.3 \cdot 10^{-1}$	$8.2 \cdot 10^{-3}$
	Electricity	$1.2 \cdot 10^{3}$	$1.6 \cdot 10^{-3}$	$4.0 \cdot 10^{0}$	$5.5 \cdot 10^{-1}$	$3.5 \cdot 10^{-2}$	$3.2 \cdot 10^{1}$	$3.7 \cdot 10^{2}$	$2.4 \cdot 10^{0}$
	Larvae feed	$1.5 \cdot 10^{3}$	$4.3 \cdot 10^{-3}$	$6.5 \cdot 10^{0}$	$2.8 \cdot 10^{-1}$	$4.9 \cdot 10^{-1}$	$1.8 \cdot 10^{1}$	$2.2 \cdot 10^{2}$	$6.3 \cdot 10^{1}$
SS3. Aquaculture	**Emissions**	0	0	0	$2.8 \cdot 10^{1}$	0	0	0	$6.5 \cdot 10^{3}$
	Diesel	$2.2 \cdot 10^{3}$	$2.2 \cdot 10^{-3}$	$1.7 \cdot 10^{1}$	$4.8 \cdot 10^{-2}$	$5.6 \cdot 10^{-3}$	$8.3 \cdot 10^{0}$	$7.4 \cdot 10^{2}$	$3.7 \cdot 10^{0}$
	Transport	$2.8 \cdot 10^{1}$	$1.2 \cdot 10^{-5}$	$8.1 \cdot 10^{-2}$	$2.6 \cdot 10^{-3}$	$2.0 \cdot 10^{-4}$	$8.6 \cdot 10^{-1}$	$9.6 \cdot 10^{0}$	$7.8 \cdot 10^{-2}$

Table E.3. Environmental characterisation of an average medium-size farm

		GW	SOD	TA	FE	ME	MET	FRS	WD
		kg CO_2	kg	kg SO_2	kg P eq	kg N eq	kg 1,4-DCB	kg oil eq	m^3
Total		$6.8\cdot10^{3}$	$1.8\cdot10^{-2}$	$3.3\cdot10^{1}$	$3.1\cdot10^{1}$	$2.5\cdot10^{0}$	$1.0\cdot10^{2}$	$1.4\cdot10^{3}$	$3.6\cdot10^{3}$
SS1. Feed	**Soybean**	$1.3\cdot10^{3}$	$3.7\cdot10^{-3}$	$1.3\cdot10^{0}$	$1.1\cdot10^{-1}$	$8.1\cdot10^{-1}$	$8.9\cdot10^{0}$	$5.0\cdot10^{1}$	$3.6\cdot10^{0}$
	Maize starch	$3.3\cdot10^{2}$	$3.6\cdot10^{-3}$	$2.5\cdot10^{0}$	$1.3\cdot10^{-1}$	$1.2\cdot10^{0}$	$1.5\cdot10^{1}$	$7.0\cdot10^{1}$	$9.0\cdot10^{0}$
	Soybean oil	$4.3\cdot10^{2}$	$1.2\cdot10^{-3}$	$4.5\cdot10^{-1}$	$4.1\cdot10^{-2}$	$2.8\cdot10^{-1}$	$3.2\cdot10^{0}$	$1.6\cdot10^{1}$	$4.7\cdot10^{0}$
	Water	$1.3\cdot10^{-1}$	$8.9\cdot10^{-8}$	$5.3\cdot10^{-4}$	$7.3\cdot10^{-5}$	$5.5\cdot10^{-6}$	$5.7\cdot10^{-3}$	$3.4\cdot10^{-2}$	$2.4\cdot10^{-1}$
	Wheat meal	$5.7\cdot10^{2}$	$6.3\cdot10^{-3}$	$4.0\cdot10^{0}$	$3.8\cdot10^{-1}$	$9.5\cdot10^{-1}$	$2.3\cdot10^{1}$	$1.2\cdot10^{2}$	$1.1\cdot10^{2}$
	Fish meal	$3.3\cdot10^{2}$	$-3.1\cdot10^{-3}$	$4.4\cdot10^{0}$	$-3.7\cdot10^{-2}$	$-9.5\cdot10^{-1}$	$-2.3\cdot10^{0}$	$1.5\cdot10^{2}$	$-6.7\cdot10^{0}$
	Fish oil	$8.5\cdot10^{1}$	$-7.8\cdot10^{-4}$	$1.1\cdot10^{0}$	$-9.6\cdot10^{-3}$	$-2.4\cdot10^{-1}$	$-5.9\cdot10^{-1}$	$3.9\cdot10^{1}$	$-1.7\cdot10^{0}$
SS2. Larvae	**Emissions**	0	0	0	0	0	0	0	$1.4\cdot10^{0}$
	Water	$3.1\cdot10^{0}$	$2.0\cdot10^{-6}$	$1.2\cdot10^{-2}$	$1.7\cdot10^{-3}$	$1.2\cdot10^{-4}$	$1.3\cdot10^{-1}$	$7.9\cdot10^{-1}$	$5.5\cdot10^{0}$
	Chlorine	$1.8\cdot10^{-1}$	$2.7\cdot10^{-7}$	$9.4\cdot10^{-4}$	$1.7\cdot10^{-4}$	$1.6\cdot10^{-5}$	$1.4\cdot10^{-2}$	$4.6\cdot10^{-2}$	$7.4\cdot10^{-3}$
	EDTA	$5.6\cdot10^{-1}$	$2.2\cdot10^{-7}$	$1.8\cdot10^{-3}$	$1.7\cdot10^{-4}$	$4.2\cdot10^{-4}$	$1.9\cdot10^{-2}$	$2.2\cdot10^{-1}$	$7.8\cdot10^{-3}$
	Electricity	$1.1\cdot10^{3}$	$1.5\cdot10^{-3}$	$3.8\cdot10^{0}$	$5.2\cdot10^{-1}$	$3.4\cdot10^{-2}$	$3.0\cdot10^{1}$	$3.5\cdot10^{2}$	$2.3\cdot10^{0}$
	Larvae feed	$1.5\cdot10^{3}$	$4.1\cdot10^{-3}$	$6.2\cdot10^{0}$	$2.7\cdot10^{-1}$	$4.7\cdot10^{-1}$	$1.8\cdot10^{1}$	$2.1\cdot10^{2}$	$6.0\cdot10^{1}$
SS3. Aquaculture	**Emissions**	0	0	0	$3.0\cdot10^{1}$	0	0	0	$3.4\cdot10^{3}$
	Diesel	$1.2\cdot10^{3}$	$1.2\cdot10^{-3}$	$8.9\cdot10^{0}$	$2.5\cdot10^{-2}$	$2.9\cdot10^{-3}$	$4.4\cdot10^{0}$	$3.9\cdot10^{2}$	$1.9\cdot10^{0}$
	Transport	$3.0\cdot10^{1}$	$1.3\cdot10^{-5}$	$8.7\cdot10^{-2}$	$2.7\cdot10^{-3}$	$2.1\cdot10^{-4}$	$9.2\cdot10^{-1}$	$1.0\cdot10^{1}$	$8.4\cdot10^{-2}$

Table E.4. Environmental characterisation of an average large-size farm

		GW	SOD	TA	FE	ME	MET	FRS	WD
		kg CO_2	kg	kg SO_2	kg P eq	kg N eq	kg 1,4-DCB	kg oil eq	m^3
Total		$6.2\cdot10^{3}$	$1.7\cdot10^{-2}$	$2.9\cdot10^{1}$	$3.1\cdot10^{1}$	$2.5\cdot10^{0}$	$9.5\cdot10^{1}$	$1.2\cdot10^{3}$	$2.3\cdot10^{3}$
SS1. Feed	**Soybean**	$1.3\cdot10^{3}$	$3.7\cdot10^{-3}$	$1.4\cdot10^{0}$	$1.1\cdot10^{-1}$	$8.1\cdot10^{-1}$	$9.0\cdot10^{0}$	$5.0\cdot10^{1}$	$3.6\cdot10^{0}$
	Maize starch	$3.3\cdot10^{2}$	$3.7\cdot10^{-3}$	$2.6\cdot10^{0}$	$1.3\cdot10^{-1}$	$1.2\cdot10^{0}$	$1.5\cdot10^{1}$	$7.1\cdot10^{1}$	$9.0\cdot10^{0}$
	Soybean oil	$4.4\cdot10^{2}$	$1.2\cdot10^{-3}$	$4.6\cdot10^{-1}$	$4.1\cdot10^{-2}$	$2.8\cdot10^{-1}$	$3.2\cdot10^{0}$	$1.6\cdot10^{1}$	$4.7\cdot10^{0}$
	Water	$1.4\cdot10^{-1}$	$8.9\cdot10^{-8}$	$5.3\cdot10^{-4}$	$7.4\cdot10^{-5}$	$5.5\cdot10^{-6}$	$5.8\cdot10^{-3}$	$3.5\cdot10^{-2}$	$2.4\cdot10^{-1}$
	Wheat meal	$5.7\cdot10^{2}$	$6.3\cdot10^{-3}$	$4.0\cdot10^{0}$	$3.8\cdot10^{-1}$	$9.5\cdot10^{-1}$	$2.3\cdot10^{1}$	$1.2\cdot10^{2}$	$1.1\cdot10^{2}$
	Fish meal	$3.3\cdot10^{2}$	$-3.1\cdot10^{-3}$	$4.4\cdot10^{0}$	$-3.8\cdot10^{-2}$	$-9.6\cdot10^{-1}$	$-2.3\cdot10^{0}$	$1.6\cdot10^{2}$	$-6.7\cdot10^{0}$
	Fish oil	$8.5\cdot10^{1}$	$-7.9\cdot10^{-4}$	$1.1\cdot10^{0}$	$-9.6\cdot10^{-3}$	$-2.4\cdot10^{-1}$	$-6.0\cdot10^{-1}$	$4.0\cdot10^{1}$	$-1.7\cdot10^{0}$
SS2. Larvae	**Emissions**	0	0	0	0	0	0	0	$1.3\cdot10^{0}$
	Water	$2.9\cdot10^{0}$	$1.9\cdot10^{-6}$	$1.1\cdot10^{-2}$	$1.6\cdot10^{-3}$	$1.2\cdot10^{-4}$	$1.2\cdot10^{-1}$	$7.3\cdot10^{-1}$	$5.1\cdot10^{0}$
	Chlorine	$1.7\cdot10^{-1}$	$2.6\cdot10^{-7}$	$8.7\cdot10^{-4}$	$1.6\cdot10^{-4}$	$1.5\cdot10^{-5}$	$1.3\cdot10^{-2}$	$4.3\cdot10^{-2}$	$6.9\cdot10^{-3}$
	EDTA	$5.2\cdot10^{-1}$	$2.0\cdot10^{-7}$	$1.7\cdot10^{-3}$	$1.6\cdot10^{-4}$	$3.9\cdot10^{-4}$	$1.8\cdot10^{-2}$	$2.0\cdot10^{-1}$	$7.3\cdot10^{-3}$
	Electricity	$1.1\cdot10^{3}$	$1.4\cdot10^{-3}$	$3.6\cdot10^{0}$	$4.9\cdot10^{-1}$	$3.1\cdot10^{-2}$	$2.8\cdot10^{1}$	$3.3\cdot10^{2}$	$2.1\cdot10^{0}$
	Larvae feed	$1.4\cdot10^{3}$	$3.8\cdot10^{-3}$	$5.8\cdot10^{0}$	$2.5\cdot10^{-1}$	$4.4\cdot10^{-1}$	$1.6\cdot10^{1}$	$1.9\cdot10^{2}$	$5.6\cdot10^{1}$
SS3. Aquaculture	**Emissions**	0	0	0	$3.0\cdot10^{1}$	0	0	0	$2.1\cdot10^{3}$
	Diesel	$7.3\cdot10^{2}$	$7.3\cdot10^{-4}$	$5.5\cdot10^{0}$	$1.6\cdot10^{-2}$	$1.8\cdot10^{-3}$	$2.7\cdot10^{0}$	$2.4\cdot10^{2}$	$1.2\cdot10^{0}$
	Transport	$3.0\cdot10^{1}$	$1.3\cdot10^{-5}$	$8.7\cdot10^{-2}$	$2.8\cdot10^{-3}$	$2.1\cdot10^{-4}$	$9.2\cdot10^{-1}$	$1.0\cdot10^{1}$	$8.4\cdot10^{-2}$

Table E.5. Percent reduction (%) in operating income for shrimp farms considered inefficient

DMU	I-1	I-2	I-3	I-4	I-5	I-6	I-7	Φ
1	-29.8%	-15.2%	-27.2%	-29.8%	-15.2%	-20.7%	-17.1%	0.78
2	-8.8%	-19.2%	-58.1%	-8.8%	-19.2%	-25.9%	-21.5%	0.77
3	-21.3%	-13.0%	-35.6%	-21.3%	-13.0%	-17.9%	-14.6%	0.81
4	-66.1%	-2.9%	-0.6%	-66.1%	-2.9%	-4.2%	-3.3%	0.79
5	-32.7%	-12.5%	-35.7%	-32.7%	-12.5%	-17.3%	-14.1%	0.77
6	-18.8%	0.0%	-4.3%	-18.8%	0.0%	0.0%	0.0%	0.94
7	-56.1%	-36.4%	-42.4%	-56.1%	-36.4%	-45.6%	-39.7%	0.55
8	-3.6%	-26.6%	-43.3%	-3.6%	-26.6%	-34.7%	-29.5%	0.76
10	-30.4%	-17.6%	-26.5%	-30.4%	-17.6%	-23.9%	-19.8%	0.76
11	-33.5%	-17.2%	-58.1%	-33.5%	-17.2%	-25.1%	-19.8%	0.71
12	-22.0%	-9.2%	-32.4%	-22.0%	-9.2%	-12.9%	-10.5%	0.83
14	0.0%	-22.6%	-30.7%	0.0%	-22.6%	-28.9%	-24.9%	0.81
15	-7.5%	-11.6%	-19.3%	-7.5%	-11.6%	-16.1%	-13.1%	0.88
16	-14.3%	-21.9%	-48.2%	-14.3%	-21.9%	-29.1%	-24.4%	0.75
17	-3.8%	-20.4%	-14.7%	-3.8%	-20.4%	-27.3%	-22.8%	0.84
18	-39.6%	-5.0%	0.0%	-39.6%	-5.0%	-7.4%	-5.8%	0.85
19	-25.5%	-20.0%	-21.1%	-25.5%	-20.0%	-26.8%	-22.4%	0.77
21	-62.7%	-29.1%	-38.6%	-62.7%	-29.1%	-37.6%	-32.1%	0.58
22	-32.2%	-4.5%	-3.0%	-32.2%	-4.5%	-6.5%	-5.2%	0.87
23	-79.9%	-44.0%	-80.5%	-79.9%	-44.0%	-55.9%	-48.3%	0.38
24	-52.9%	-20.8%	-28.9%	-52.9%	-20.8%	-27.7%	-23.2%	0.68
25	-48.6%	-11.6%	-17.8%	-48.6%	-11.6%	-16.1%	-13.1%	0.76
26	-45.9%	-4.0%	-10.3%	-45.9%	-4.0%	-5.8%	-4.6%	0.83
27	-39.7%	-17.2%	-22.8%	-39.7%	-17.2%	-23.4%	-19.4%	0.74
28	-32.2%	-4.5%	-6.7%	-32.2%	-4.5%	-6.5%	-5.2%	0.87
29	-47.3%	-26.3%	-36.9%	-47.3%	-26.3%	-36.6%	-29.8%	0.64
30	-34.3%	-1.2%	-2.7%	-34.3%	-1.2%	-1.7%	-1.4%	0.89
31	-76.9%	-44.9%	-43.3%	-76.9%	-44.9%	-54.4%	-48.5%	0.44
32	-63.4%	-6.7%	-8.5%	-63.4%	-6.7%	-9.5%	-7.6%	0.76
34	-70.4%	-22.2%	-58.8%	-70.4%	-22.2%	-31.6%	-25.4%	0.57
35	-32.1%	0.0%	-12.5%	-32.1%	0.0%	0.0%	0.0%	0.89
36	-27.7%	-6.7%	-24.2%	-27.7%	-6.7%	-10.3%	-7.8%	0.84
37	-53.5%	-1.2%	-24.1%	-53.5%	-1.2%	-1.7%	-1.4%	0.80

Appendix II. Calculation of relevant parameters in eco-efficiency assessment

On-farm emissions estimation in dairy farms

On-farm emissions were estimated following the IPCC Guidelines for National Greenhouse Gas Inventories. Direct emissions related to agriculture, forestry and other land use are divided in methane (CH_4), direct dinitrogen monoxide (N_2O) and indirect dinitrogen monoxide, which is divided mainly in ammonia (NH_3) and nitrate (NO_3^-).

CH_4 emissions from enteric fermentation: Calculated through IPCC Tier 1 emission factors, considering as primary information the number of head of livestock present on each farm. These emissions were calculated following the Equation (1):

$$CH_4(kg\ yr^{-1}) = (EF_{DAIRY} * N_{DAIRY}) + (EF_{REST} * N_{REST}) \tag{1}$$

Where:

- EF_{DAIRY} and EF_{REST} are the enteric fermentation emission factors for dairy and non-dairy cattle, respectively. Values taken for cattle from Western Europe (117 and 57 kg CH_4 head^{-1} yr^{-1} for dairy and non-dairy cattle, respectively.
- N_{DAIRY} and N_{REST} are the number of dairy and non-dairy cattle, respectively.

CH_4 emissions from manure management: Based on the livestock population data by animal category and climate temperature. These emissions were calculated following the Equation (2):

$$CH_4(kg\ yr^{-1}) = (EF_{DAIRY} * N_{DAIRY}) + (EF_{REST} * N_{REST}) \tag{2}$$

Where:

- EF_{DAIRY} and EF_{REST} are the emission factor dairy and non-dairy cattle, respectively. Values taken for cattle from Western Europe as 27 and 8 kg CH_4 head^{-1} yr^{-1} for dairy and non-dairy cattle, respectively. These values were chosen taking into account that the average annual temperature in 2019 was around 13ºC
- N_{DAIRY} and N_{REST} are the number of dairy and non-dairy cattle, respectively.

N_2O emissions from manure management: It was calculated by multiplying the total amount of N excretion from all livestock categories in each type of manure management system by an emission factor for that type of manure management system. Thus, these emissions were estimated following the Equation (3):

$$N_2O(kg\ yr^{-1}) = \left[\sum_S \left[\sum_T \left(N_T * Nex_T * MS_{T,S}\right)\right] * EF_{3,S}\right] * \frac{44}{28} \tag{3}$$

Where:

- N_T is the number of head of livestock per category T.
- Nex_T is the annual average N excretion per head of livestock per category T, in kg N head^{-1} yr^{-1}. Values taken for cattle from Western Europe (0.48 and 0.33 kg N (1000 kg animal mass^{-1}) day^{-1} for dairy and non-dairy cattle, respectively.
- $MS_{T,S}$ is the fraction of total annual nitrogen excretion for each livestock category T that is managed in manure management system S. In this case, only 1 type of manure management system is considered, so it takes the value of 1.
- $EF_{3,S}$ is the emission factor for direct N2O emissions from manure management system S. In this case, the manure management system is a pit storage below animal confinements, so it takes the value of 0.002 kg N_2O-N/kg N
- 44/28 is the conversion of N_2O-N emissions to N_2O emissions

N_2O emissions from managed soils: An increase in available N enhances nitrification and denitrification rates which then increase the production of N_2O. This category quantifies the human-induced N additions that mineralise soil organic N. These emissions were estimated following the Equation (4) and the respective Equations (4.1) and (4.2):

$$N_2O_{DIRECT} - N = N_2O{-}N_{N\ INPUTS} + N_2O - N_{OS} \tag{4}$$

$$N_2O{-}N_{N\ INPUTS} = (F_{SN} + F_{ON} + F_{CR} + F_{SOM}) * EF_1 \tag{4.1}$$

$$N_2O - N_{OS} = F_{OS} * EF_2 \tag{4.2}$$

Where:

- N_2O_{DIRECT}-N is the annual direct N_2O emissions produced from managed soils, in kg N_2O-N yr^{-1}
- N_2O-$N_{N\ INPUTS}$ is the annual direct N_2O-N emissions from N inputs to managed soils, in kg N_2O-N yr^{-1}
    - F_{SN} is the annual amount of synthetic fertiliser N applied to soils, in kg N yr^{-1}. It was calculated through primary information regarding NPK fertilisers consumption.
    - F_{ON} is the annual amount of animal manure N additions applied to souls, in kg N yr^{-1}. It was calculated through primary information regarding re-use of cow manure in land crops.

  - FCR is the annual amount of N in crop residues, in kg N yr^{-1}. In this case, it was considered irrelevant.
  - FSOM is the annual amount of N in mineral soils that is mineralised, in kg N $yr^{-1.}$ Also considered negligible.
  - EF1 is the emission factor for N_2O emissions from N inputs, in kg N_2O-N (kg N input)$^{-1.}$ It was taken as 0.01 according to the IPCC guidelines.
- N_2O-N_{OS} is the annual direct N_2O-N emissions from managed organic soils, in kg N_2O-N yr^{-1}
  - F_{OS} is the annual area of managed soils, in ha. It was calculated based on the primary information proportionate by the farmers.
  - EF2 is the emission factor for temperate organic crop and grassland soils. According to the IPCC guidelines, it was taken as 8 kg N_2O-N yr^{-1}

Computation of key parameters from primary data in shrimp aquaculture

Production efficiency (kg/ha):

$$\gamma = \frac{P \cdot 1000}{A} \tag{5}$$

Where:

- γ: Production efficiency (kg/ha)
- P: Total shrimp production (t)
- A: Farming area (ha)

Amount of feed provided (t):

$$Feed = \frac{(\gamma \cdot FCA) \cdot A}{1000} \tag{6}$$

Where:

- FCA: Feed Conversion Ratio

Volume of seawater needed to fill ponds (m^3)

$$V_{seawater} = A \cdot h \cdot 10000 \tag{7}$$

Where:

- $V_{seawater}$: volume of seawater in the pond (m^3)
- H: Height of the water volume. It was considered 1 m

Number of larvae required (t):

$$M_{larvae} = \frac{\rho_{larvae} \cdot A \cdot m_{larvae}}{100} \tag{8}$$

Where:

- M_{larvae}: Total weight of larvae used in one year of operation (t)
- P_{larvae}: Stock density (org/m^2)
- m_{larve}: Average weight of a larvae

Energetic consumption (MJ):

$$E = V_{Seawater} \cdot 0.0823 \tag{9}$$

Where:

- E: Energy consumed on each farm per year of operation (MJ)

Feed transport (t·km):

$$T = Feed \cdot D \tag{10}$$

Where:

- T: Feed transport (t·km)
- D: Distance. 70 km was considered as an average

Appendix III. Published articles used in this thesis

Integrated evaluation of wine less valorization to produce value-added products

Cortés, A.; Moreira, M. T.; Feijoo, G.

CRETUS, Department of Chemical Engineering, Universidade de Santiago de Compostela, 15782, Santiago de Compostela, Spain.

Journal: Waste Management (2019), ed. Elsevier (ISSN: 0956-053X), 95, 70-77. DOI: https://doi.org/10.1016/j.wasman.2019.05.056. Impact Factor in 2019: 5.448, ranking it 35 of 265 in Environmental Sciences (Q1). This Journal allows for the use of this full article in the present thesis, both in print and electronic format without requiring further permission, provided that it is not published commercially and that it is properly referenced.

Integrated evaluation of wine lees valorization to produce value-added products

Author: Antonio Cortés,Maria Teresa Moreira,Gumersindo Feijoo
Publication: Waste Management
Publisher: Elsevier
Date: 15 July 2019

Journal Author Rights

Please note that, as the author of this Elsevier article, you retain the right to include it in a thesis or dissertation, provided it is not published commercially. Permission is not required, but please ensure that you reference the journal as the original source. For more information on this and on your other retained rights, please visit: https://www.elsevier.com/about/our-business/policies/copyright#Author-rights

BACK CLOSE WINDOW

Author contribution (CRediT taxonomy): Investigation, Writing-original draft, Formal analysis, Conceptualization.

Chapters reproducing the article content: Chapter 3 is based on this publication.

Unraveling the environmental impacts of bioactive compounds and organic amendment from grape marc

Cortés, A.[a]; Moreira, M. T.[a]; Domínguez, J.[b]; Lores, M.[c]; Feijoo, G.[a]

[a] CRETUS, Department of Chemical Engineering, Universidade de Santiago de Compostela, 15782, Santiago de Compostela, Spain.

[b] Grupo de Ecoloxía Animal (GEA), Universidade de Vigo, 36310, Vigo, Spain.

[c] CRETUS, Department of Analytical Chemistry, Nutrition and Food Sciences, Universidade de Santiago de Compostela, 15782, Santiago de Compostela, Spain.

Journal: Journal of Environmental Management (2019), ed. Elsevier (ISSN: 0301-4797), 95, 70-77. DOI: https://doi.org/10.1016/j.jenvman.2020.111066. Impact Factor in 2019: 5.647, ranking it 33 of 265 in Environmental Sciences (Q1). This Journal allows for the use of this full article in the present thesis, both in print and electronic format without requiring further permission, provided that it is not published commercially and that it is properly referenced.

Unraveling the environmental impacts of bioactive compounds and organic amendment from grape marc

Author: Antonio Cortés,Maria Teresa Moreira,Jorge Domínguez,Marta Lores,Gumersindo Feijoo

Publication: Journal of Environmental Management

Publisher: Elsevier

Date: 15 October 2020

Journal Author Rights

Please note that, as the author of this Elsevier article, you retain the right to include it in a thesis or dissertation, provided it is not published commercially. Permission is not required, but please ensure that you reference the journal as the original source. For more information on this and on your other retained rights, please visit: https://www.elsevier.com/about/our-business/policies/copyright#Author-rights

BACK CLOSE WINDOW

Author contribution (CRediT taxonomy): Investigation, Acquisition of data, Writing-original draft, Formal analysis.

Chapters reproducing the article content: Chapter 4 is based on this publication.

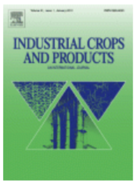
Unraveling the environmental impacts of bioactive compounds and organic amendment from grape marc

Cortés, A.[a]; Feijoo, G.[a]; Chica, A.[b]; Da Costa-Serra, J. F.[b]; Moreira, M. T.[a]

[a] CRETUS, Department of Chemical Engineering, Universidade de Santiago de Compostela, 15782, Santiago de Compostela, Spain.

[b] Institute of Chemical Technology, Universitat Politècnica de València-Consejo Superior de Investigaciones Científicas, 46022 València, Spain

Journal: Industrial Crops & Products (2019), ed. Elsevier (ISSN: 0962-6690), 138, 111465. DOI: https://doi.org/10.1016/j.indcrop.2019.111465. Impact Factor in 2019: 4.244, ranking it 8 of 91 in Agronomy (Q1). This Journal allows for the use of this full article in the present thesis, both in print and electronic format without requiring further permission, provided that it is not published commercially and that it is properly referenced.



Environmental implications of biohydrogen based energy production from steam reforming of alcoholic waste

Author: Antonio Cortés,Gumersindo Feijoo,Antonio Chica,Javier Francisco Da Costa-Serra,María Teresa Moreira

Publication: Industrial Crops and Products

Publisher: Elsevier

Date: 5 October 2019

Journal Author Rights

Please note that, as the author of this Elsevier article, you retain the right to include it in a thesis or dissertation, provided it is not published commercially. Permission is not required, but please ensure that you reference the journal as the original source. For more information on this and on your other retained rights, please visit: https://www.elsevier.com/about/our-business/policies/copyright#Author-rights

BACK CLOSE WINDOW

Author contribution (CRediT taxonomy): Investigation, Acquisition of data, Writing-original draft, Formal analysis.

Chapters reproducing the article content: Chapter 5 is based on this publication.

Evaluation of the environmental sustainability of the inshore great scallop (*Pecten maximus*) fishery in Galicia

Cortés, A.; González-García, S.; Franco-Uría, A.; Moreira, M. T.; Feijoo, G.

CRETUS, Department of Chemical Engineering, Universidade de Santiago de Compostela, 15782, Santiago de Compostela, Spain.

Journal: Journal of Industrial Ecology (2021), ed. Wiley (ISSN: 1530-9290), 1-13. DOI: https://doi.org/10.1111/jiec.13153. Impact Factor in 2021: 7.202, ranking it 49 of 279 in Environmental Sciences (Q1). This Journal allows for the use of this full article in the present thesis, both in print and electronic format without requiring further permission, provided that it is not published commercially and that it is properly referenced.

Evaluation of the environmental sustainability of the inshore great scallop (Pecten maximus) fishery in Galicia

Author: Gumersindo Feijoo, Maria Teresa Moreira, Amaya Franco-Uría, et al
Publication: Journal of Industrial Ecology
Publisher: John Wiley and Sons
Date: May 16, 2021

Licensed Content

Licensed Content Publisher	John Wiley and Sons
Licensed Content Publication	Journal of Industrial Ecology
Licensed Content Title	Evaluation of the environmental sustainability of the inshore great scallop (Pecten maximus) fishery in Galicia
Licensed Content Author	Gumersindo Feijoo, Maria Teresa Moreira, Amaya Franco-Uría, et al
Licensed Content Date	May 16, 2021
Licensed Content Volume	0
Licensed Content Issue	0
Licensed Content Pages	14

Order Details

Type of use	Dissertation/Thesis
Requestor type	Author of this Wiley article
Format	Print and electronic
Portion	Full article
Will you be translating?	No

Author contribution (CRediT taxonomy): Investigation, Acquisition of data, Writing-original draft, Formal analysis.

Chapters reproducing the article content: Chapter 6 is based on this publication.

Multi-product strategy to enhance the environmental profile of the canning industry towards circular economy

Cortés, A.; Esteve-Llorens, X.; González-García, S.; Moreira, M. T.; Feijoo, G.

CRETUS, Department of Chemical Engineering, Universidade de Santiago de Compostela, 15782, Santiago de Compostela, Spain.

Journal: Science of the Total Environment (2021), ed. Elsevier (ISSN: 0048-9697), 791, 148249. DOI: https://doi.org/10.1016/j.scitotenv.2021.148249. Impact Factor in 2021: 10.753, ranking it 26 of 279 in Environmental Sciences (Q1). This Journal allows for the use of this full article in the present thesis, both in print and electronic format without requiring further permission, always under the terms of the Creative Commons CC-BY license provided that it is properly referenced.

Author contribution (CRediT taxonomy): Investigation, Acquisition of data, Writing-original draft, Formal analysis.

Chapters reproducing the article content: Chapter 7 is based on this publication.

Pursuing the route to eco-efficiency in dairy production: The case of Galician area

Cortés, A.[a]; Feijoo, G.[a]; Fernández, M.[b]; Moreira, M. T.[a]

[a] CRETUS, Department of Chemical Engineering, Universidade de Santiago de Compostela, 15782, Santiago de Compostela, Spain.

[b] Galician Association of Agri-food Cooperatives, 15703, Santiago de Compostela, Spain.

Journal: Journal of Cleaner Production (2020), ed. Elsevier (ISSN: 0959-6526), 285, 124861. DOI: https://doi.org/10.1016/j.jclepro.2020.124861. Impact Factor in 2020: 9.297, ranking it 3 of 44 in Green & Sustainable Science & Technology (Q1). This Journal allows for the use of this full article in the present thesis, both in print and electronic format without requiring further permission, provided that it is not published commercially and that it is properly referenced.

Pursuing the route to eco-efficiency in dairy production: The case of Galician area

Author: Antonio Cortés,Gumersindo Feijoo,Mario Fernández,Maria Teresa Moreira

Publication: Journal of Cleaner Production

Publisher: Elsevier

Date: 20 February 2021

Journal Author Rights

Please note that, as the author of this Elsevier article, you retain the right to include it in a thesis or dissertation, provided it is not published commercially. Permission is not required, but please ensure that you reference the journal as the original source. For more information on this and on your other retained rights, please visit: https://www.elsevier.com/about/our-business/policies/copyright#Author-rights

BACK CLOSE WINDOW

Author contribution (CRediT taxonomy): Investigation, Acquisition of data, Writing-original draft, Formal analysis.

Chapters reproducing the article content: Chapter 8 is based on this publication.

Eco-efficiency assessment of shrimp aquaculture production in Mexico

Cortés, A.[a]; Casillas-Hernández, R.[b]; Cambeses-Franco, C.[a]; Bórquez-López, R.[b]; Magallón_Barajas, F.[c]; Quadros, Seiffert, W.[d]; Feijoo, G.[a]; Moreira, M. T.[a]

[a] CRETUS, Department of Chemical Engineering, Universidade de Santiago de Compostela, 15782, Santiago de Compostela, Spain.

[b] Department of Agricultural and Veterinary Sciences, Sonora Institute of Technology, Cd. Obregón, Mexico.

[c] Northwestern Center for Biological Research (CIBNOR), La Paz B.C.S., Mexico

[d] Laboratory of Shrimp Marine, Department of Aquaculture, Federal University of Santa Catarina, Florianopolis, SC, Brazil

Journal: Aquaculture (2021), ed. Elsevier (ISSN: 0044-8486), 737145. DOI: https://doi.org/10.1016/j.aquaculture.2021.737145. Impact Factor in 2021: 5.135, ranking it 8 of 113 in Marine & Freshwater Biology (Q1).

Author contribution (CRediT taxonomy): Investigation, Acquisition of data, Writing-original draft, Formal analysis.

Chapters reproducing the article content: Chapter 9 is based on this publication.

Appendix IV. List of publications

Peer-reviewed journals

Antonio Cortés, Maria Teresa Moreira, Gumersindo Feijoo. “Integrated evaluation of wine lees valorization to produce value-added products”. *Waste Management*, 2019, 95, 70-77. https://doi.org/10.1016/j.wasman.2019.05.056.

Antonio Cortés, Gumersindo Feijoo, Antonio Chica, Javier Francisco Da Costa-Serra, Maria Teresa Moreira. “Environmental implications of biohydrogen based energy production from steam reforming of alcoholic waste”. *Industrial Crops & Products*, 2019, 138, 111465. https://doi.org/10.1016/j.indcrop.2019.111465.

Sara González-García, Manuel Rama, **Antonio Cortés**, Fernando García-Guaita, Andrés Núñez, Lucía González-Louro, Maria Teresa Moreira, Gumersindo Feijoo. “Embedding environmental, economic and social indicators in the evaluation of the sustainability of the municipalities of Galicia (northwest of Spain)”. Journal of Cleaner Production, 2019, 234, 27-42. https://doi.org/10.1016/j.jclepro.2019.06.158.

Antonio Cortés, Maria Teresa Moreira, Jorge Domínguez, Marta Lores, Gumersindo Feijoo. “Unravelling the environmental impacts of bioactive compounds and organic amendment from grape marc”. *Journal of Environmental Management*, 2020, 272, 111066. https://doi.org/10.1016/j.jenvman.2020.111066.

Antonio Cortés, Luis F.S. Oliveira, Valdecir Ferrari, Silvio R. Taffarel, Gumersindo Feijoo, Maria Teresa Moreira. “Environmental assessment of viticulture waste valorisation through composting as a biofertilisation strategy for cereal and fruit crops”. *Environmental Pollution*, 2020, 264, 114794. https://doi.org/10.1016/j.envpol.2020.114794.

Antonio Cortés, Gumersindo Feijoo, Mario Fernández, Maria Teresa Moreira. “Pursuing the route to eco-efficiency in dairy production: The case of Galician area”. *Journal of Cleaner Production*, 2021, 285, 124861. https://doi.org/10.1016/j.jclepro.2020.124861.

Antonio Cortés, Sara González-García, Amaya Franco-Uría, Maria Teresa Moreira, Gumersindo Feijoo. “Evaluation of the environmental sustainability of the inshore great scallop (*Pecten maximus*) fishery in Galicia”. *Journal of Industrial Ecology*, 2021, 1-13. https://doi.org/10.1111/jiec.13153.

Antonio Cortés, Ramón Casillas-Hernández, Cristina Cambeses-Franco, Rafael Bórquez-López, Francisco Magallón-Barajas, Walter Quadros-Seiffert, Gumersindo Feijoo, Maria Teresa Moreira. "Eco-efficiency assessment of shrimp aquaculture production in Mexico". *Aquaculture*, 2021, 544, 737145. https://doi.org/10.1016/j.aquaculture.2021.737145.

Israel Ruiz-Salmón, Jara Laso, María Margallo, Pedro Villanueva-Rey, Eduardo Rodríguez, Paula Quinteiro, Ana Cláudia Días, Cheila Almeida, Maria Leonor Nunes, António Marques, **Antonio Cortés**, Maria Teresa Moreira, Gumersindo Feijoo, Philippe Loubet, Guido Sonnemann, Andrew P. Morse, Ronan Cooney, Eoghan Clifford, Leticia Regueiro, Diego Méndez, Clémentine Anglada, Christelle Noirot, Neil Rowan, Ian Vázquez-Rowe, Rubén Aldaco. "Life cycle assessment of fish and seafood processed products – A review of methodologies and new challenges". *Science of the Total Environment*, 2021, 761, 144094. https://doi.org/10.1016/j.scitotenv.2020.144094.

Antonio Cortés, Xavier Esteve-Llorens, Sara González-García, Maria Teresa Moreira, Gumersindo Feijoo, "Multi-product strategy to enhance the environmental profile of the canning industry towards circular economy". *Science of the Total Environment*, 2021, 791, 148249. https://doi.org/10.1016/j.scitotenv.2021.148249.

Eduardo Entrena-Barbero, **Antonio Cortés**, Xavier Esteve-Llorens, Maria Teresa Moreira, Pedro Villanueva-Rey, Diego Quiñoy, Cheila Almeida, António Marques, Paula Quinteiro, Ana Cláudia Días, Jara Laso, Israel Ruiz-Salmón, María Margallo, Rubén Aldaco, Gumersindo Feijoo. "Methodological guidelines for the calculation of a Water-Energy_Food nexus index for seafood products: moving towards a blue future". Submitted to *Environmental Science and Policy*.

UNIVERSIDADE DE SANTIAGO DE COMPOSTELA

CAMPUS VIDA
CAMPUS DE EXCELENCIA INTERNACIONAL

This doctoral thesis focuses on the environmental analysis through the methodology of Life Cycle Assessment of innovative alternatives for the application of the principles of circular economy in strategic primary sectors in Galicia. Specifically, the valorisation of wine residues to obtain value-added products, the application of traditional techniques in the in-shore fishery, the application of multi-product strategies in the canning sector, and the level of eco-efficiency in dairy and aquaculture farms were analysed. The results of this thesis have made it possible to determine the "hot spots" of the different practices, as well as benchmarking with other management alternatives.