TESIS DOCTORAL

Ecodiseño de alimentos mediante el análisis de ciclo de vida

Saioa Ramos Fernández

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Director:

Dr. Gumersindo Feijoo Costa

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RESUMEN

Según datos de la agencia europea de medio ambiente el sector alimentario es una de las actividades humanas que más contribuye al impacto ambiental. Por ello, tanto las políticas europeas como consumidores están demandando productos más sostenibles con el entorno.

Los productos alimentarios, desde la agricultura hasta el consumidor final, pasando por los procesos de ganadería, pesca, acuicultura, transporte, transformación y venta, acumulan a lo largo de su cadena una serie de impactos ambientales como el potencial de cambio climático, eutrofización o el agotamiento de recursos. El uso de fertilizantes y pesticidas, la quema de combustibles fósiles o el consumo de agua son unos problemas de alcance mundial que debemos reducir con el fin de disminuir estos impactos.

Sin embargo, antes de invertir esfuerzos en minimizar el impacto ambiental asociado a los productos alimentarios, es indispensable evaluar este impacto teniendo en cuenta todos los aspectos ambientales de una cadena alimentaria. Para ello el Análisis de Ciclo de Vida se presenta como una herramienta aceptada a nivel internacional. Sin embargo, los procesos alimentarios tienen una serie de características intrínsecas que dificultan la aplicación de esta metodología.

En esta tesis se analizan los retos del uso del Análisis de Ciclo de Vida para calcular y reducir los potenciales impactos ambientales asociados a los productos alimentarios. Unidas a esta hipótesis se plantean y responden tres preguntas:

1. ¿Cómo afecta la variabilidad temporal a los impactos ocasionados por la producción primaria? A lo largo de la tesis se demuestra que en los procesos de producción primaria existen unas variables no controladas (clima, plagas, etc.) que varían significativamente los resultados de impacto ambiental de un año para otro. Con el fin de reducir la incertidumbre asociada a esta variabilidad se propone el uso de la metodología combinada de ACV+DEA, por la cual se puede establecer el número mínimo de años necesarios de recolección de datos de inventario para obtener una evaluación ambiental representativa.

2. ¿Qué método de evaluación de impactos es el más idóneo para calcular el impacto ambiental de los productos alimentarios? En esta tesis se definen una serie de impactos para evaluar la sostenibilidad de los productos alimentarios. Asociados a estos impactos también se establecen son los aspectos más relevantes para cada etapa de la cadena alimentaria. Además, debido a que el consumo de agua es uno de los aspectos clave de

la industria alimentaria se han valorado 3 metodologías para evaluar el impacto ambiental regionalizado asociado al consumo de agua.

3. ¿Cómo se puede facilitar a las empresas la evaluación ambiental? Con el fin de transferir la metodología al sector alimentario se ha desarrollado una herramienta de fácil manejo para evaluar de una forma sencilla y armonizada el impacto ambiental de los alimentos. Además, con el fin de facilitar no solo la evaluación, sino también la reducción del impacto ambiental asociado a los alimentos, se ha planteado y probado una nueva metodología de ecodiseño de alimentos en un caso real.

LABURPENA

Europear ingurugiro-agentziak jakinarazi duenez, elikagaien sektorea, beste sektoreekin alderatuz, inpaktu gehien sortzen duen giza ekintzetariko bat da. Horregatik, gaur egungo kontsumitzaileak, baita egungo lege edo ekimen publikoak ere, inpaktu gutxiko elikagaiak eskatzen ari dira.

Elikagaien transformazioaren bizi-zikloan zehar, ingurugiroarekiko elkarrekintza ugari egon ohi dira. Nekazaritzatik abiatuta azken kontsumora arte; ura, energia eta materiale asko erabiltzen dira, baita zabor, aire-emisio eta ur-emariak sortu ere. Honek guztiak inpatu desberdinak eragiten ditu, besteak beste, klima aldaketa edo eutrofizazioa.

Dena dela, inpaktu hauek murriztu aurretik, era fidagarri eta adierazgarri batean neurtzea beharrezkoa da. Horretarako, Bizi-Zikloaren Analisia (BZA), maila internazionalean adostua dagoen metodologia gisa aurkezten da. Hala eta guztiz ere, elikagaiek berezkoa duten ezaugarri askok, metodologia honen aplikazioa oztopa dezakete.

Tesi honek, bizi-zikloaren analisiaren erabilera balioztatzen du, bai elikagaien inpaktua baita inpaktu hori murrizteko ekimenen balioa ere ebaluatzeko. Hipotesi honen inguruan, tesi honek hiru galdera nagusiri erantzuten dio:

1. Zein da ekoizpen primarioaren urteen arteko aldakortasunak, neurtutako ingurumen-inpaktuaren adierazgarritasunean duen esangura? Ekoizpen primarioak (nekazaritza, abeltzaintza, arrantza eta akuikultura) berezko duen aldakortasunak, ingurumen-inpaktua neurtzeko orduan eragin handia du. Urte batetik besterako eurien aldaketak, izurrite batek edo arrain stock-aren leku aldatzeak, ingurumen-inpaktuan eragin adierazgarria izan dezake. Tesi honen bidez, BZA+DEA (*Data envelopment analysis*) metodologia konbinatuaren bitartez, ingurumen-inpaktuaren balio esanguratsua lortzeko beharrezkoak diren urte kopuru minimoa ezarri daitekeela frogatua izan da.

2. Zein da metodologiarik egokiena elikagaiek eragiten duten ingurumen-inpaktua neurtzeko? Tesi honetan zehar, elikagaiek ingurumenean duten eragina ebaluatzeko, ezinbestekoak diren adierazleak aukeratu dira: klima aldaketa, estres hidrikoa edo azidifikazioa, esaterako. Halaber, adierazle hauekin erlazionaturik dauden elikagai-kateen etapa bakoitzeko alderdiak identifikatu egin dira (ongarrien erabilera, aire-emisioak edo pestiziden erabilera). Horrez gain, giza-jarduera baten ondorioz, alor ezberdinetan suerta daitezkeen estres hidrikoa neurtzeko plazaratu berri diren hiru metodo ezberdin ere frogatu egin dira.

3. Nola erraztu daiteke elikagaien ingurugiro ebaluazioa eskala erreal batean ezartzen? Aukeratu diren adierazleak eta metodologia enpresei iritsi arazteko asmoz, tesi honetan, tresna sinple eta sendo bat ere garatu izan da, SENSE-tool. Honetaz gain, elikagaien ekodiseinua errazteko, metodologia bat ere garatu da.

ABSTRACT

According to the European Environment Agency, food industry is one of the human activities which contribute most to the environmental impact. Therefore, both European policies as consumers are demanding more sustainable products with less environmental impact. Food products, from farming to the final consumer, through the processes of livestock, fishing, aquaculture, transportation, processing and sales, accumulate along their chain a series of environmental impacts. The use of fertilizers or pesticides, fossil fuel combustion or water consumption are a few global challenges that need to be faced.

However, before investing efforts to minimize the environmental impact of food products, first it is necessary to evaluate them; for that purpose the life-cycle assessment is presented as an internationally accepted methodology. However, food products have a number of intrinsic characteristics that hinder the implementation of this methodology.

This thesis evaluates the challenges to apply life cycle analysis to assess and reduce the environmental impact of food product and processes. Within this hypothesis three questions are answered:

1. How could be overcome the impact variation caused by the interanual variability in the primary production? The thesis shows that, especially in primary production processes there is an uncontrolled aspects which could affect considerably to the final environmental impact. In order to reduce the uncertainty associated with the variability, the combined use of LCA + DEA methodology is proposed as a suitable tool to determine the minimum number of years necessary for representative data collection.

2. What impact characterization methodologies are the most suitable for measuring the environmental impact associated to food products? In this thesis a series of impacts are defined to assess the sustainability of food products. Associated with these impacts is also defined a set key environmental performance indicators to each stage of the food chain. In addition, three methods have been tested to assess the environmental impact associated with the consumption of water.

3. How could this methodology be applied to evaluate the environmental impact in a company? In order to transfer the methodology to the food industry, a user-friendly tool has been developed to evaluate the environmental impact of food in a simple and harmonized way. A new methodology of ecodesign for food has also been proposed and tested in a real case.

ÍNDICE

ABREVIATURAS / ABREVIATIONS

ACV	Análisis de Ciclo de Vida
CAPV	Comunidad Autonoma del País Vasco
CE	Comunidad Europea
DEA	Data envelopment Analysis
DMU	Decision Making Unit
EC	European Comunity
EEA	European Environment Agency
FAO	Food and Agriculture Organization
GEI	Gases de efecto Invernadero
GHG	GreenHouse Gases
INE	Instituto Nacional de Estadística
IPCC	Intergovernmental Panel on Climate Change
JRC	Joint Research Center
LCA	Life Cycle Assessment
PYME	Pequeña Y Mediana Empresa
SENSE	Harmonised Environmental Sustainability in the European food and drink chain
SME	Small and Medium entrerprise
WSI	Water Stress Index
WTA	Withdrawal To Availability

Nire familiari

Capítulo 1 | INTRODUCCIÓN

La cadena alimentaria es uno de los sectores con mayor crecimiento a nivel mundial y es clave para el desarrollo del planeta. Según las proyecciones de la Organización mundial para la Agricultura y Alimentación (FAO) para el 2050 la producción de alimentos deberá aumentar hasta un 70 % con el fin de alimentar a la creciente población.

Cabe destacar que la producción de alimentos también es uno de los sectores con mayor impacto ambiental, según un estudio llevado a cabo por el Centro Común de Investigación más conocido por JRC (*Joint Research Centre*), la producción de alimentos supone entre el 20 y el 30 % del impacto ambiental de las actividades humanas de la Unión Europea (Tukker et al., 2006).

Sin embargo, existen muchas iniciativas a nivel internacional tanto públicas como privadas para disminuir los impactos asociados a la producción de alimentos. Dentro de estas iniciativas el Análisis de Ciclo de Vida resulta una herramienta indispensable para evaluar el impacto ambiental asociado a todo el ciclo de vida del producto, desde la producción de primaria hasta el consumo o disposición final.

1.1. La industria alimentaria en Europa, España y País Vasco

El sistema alimentario se define como una red compleja entre los productores primarios y las industrias que enlazan con ellos. Los vínculos entre estos agentes incluyen empresas auxiliares de envases, proveedores de materias primas o ingredientes, así como los transportistas. El sistema también incluye las industrias de comercialización de alimentos que enlazan la producción primaria con los consumidores finales, y que incluyen los procesadores de alimentos, mayoristas, minoristas y establecimientos de servicios de alimentos.

En la Figura 1 se muestra una imagen de los vínculos existentes entre las diferentes etapas de la cadena alimentaria. En términos generales la cadena alimentaria comienza con la producción de materias primas animales o vegetales, obtenidas en explotaciones ganaderas, agrícolas o pesqueras. Estas materias primas se transportan hasta las empresas donde son procesadas, envasadas y de nuevo transportadas hasta llegar al alcance del consumidor final. Es necesario tener en cuenta que a lo largo de la cadena existen una serie de demandas y necesidades de materiales auxiliares (envases, ingredientes, aditivos, etc.), energía y agua, así como una serie de salidas como residuos, subproductos y vertidos, que también forman parte de la cadena alimentaria.

La industria agroalimentaria es la principal actividad de la industria manufacturera europea, representando el 14,6 % de su facturación total y un valor superior a los 1.048.000 millones de euros (Food Drink Europa, 2013). Cuenta con unas 286.000 empresas, siendo la mayoría de ellas Pequeña y Mediana Empresas (PYME) con menos de 250 trabajadores (un 99,1 % del total), que dan empleo a 4,24 millones de personas y representan el 51,6 % del total de las ventas del sector agroalimentario de la UE y el 64,3 % del conjunto de empleos que se genera. La industria alimentaria española ocupa el quinto puesto en valor de ventas tras Alemania, Francia, Italia y Reino Unido.

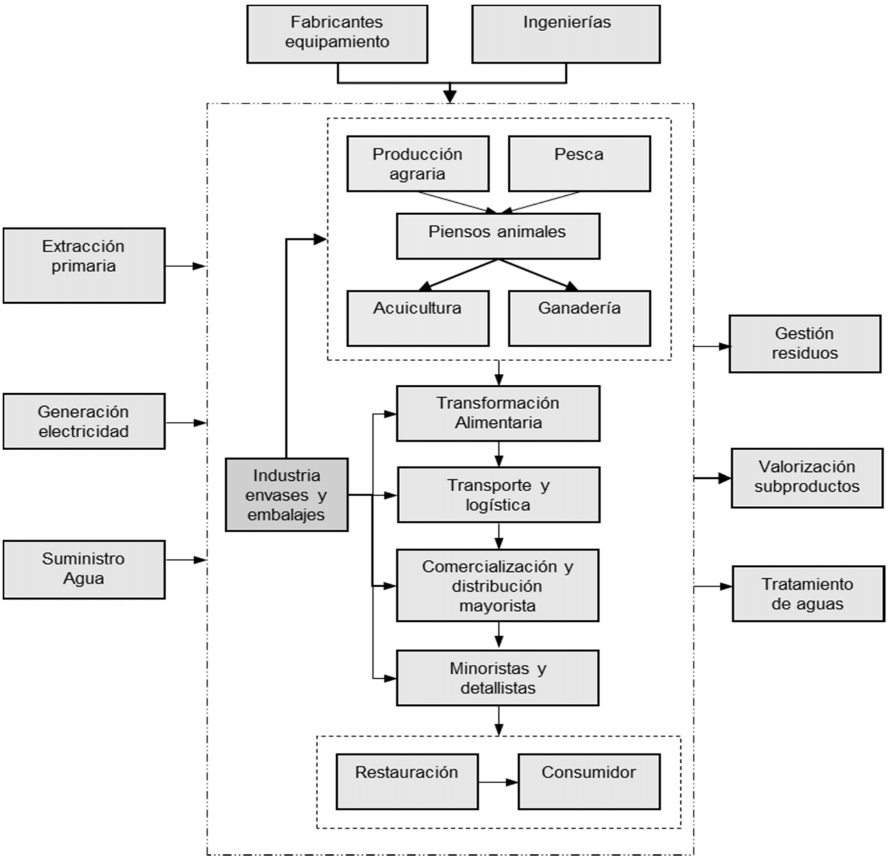

Figura 1. Imagen generalista de cadena alimentaria global.

En España, la industria de alimentación y bebidas es la primera rama industrial, según la encuesta Industrial de Empresas del Instituto Nacional de Estadística (INE), a 31 de diciembre de 2013, representando el 20,6 % de las ventas netas de producto, el 18,2 % de personas ocupadas, el 16,8 % de las inversiones en activos materiales y el 15,3 % del valor añadido. Las agrupaciones de actividad con mayor contribución al total de ventas netas del sector industrial en 2013 fueron Alimentación y bebidas (20,6 %), Vehículos a motor y Energía y agua (12,5 % cada uno) y Metalurgia y fabricación de productos metálicos (11,4 %). En total, el consumo de alimentos y bebidas realizado por las familias españolas en 2014 alcanzó un valor de 69.225 millones de euros, con un valor medio por hogar de 127,0 € al mes (Mercasa, 2014).

En la demanda de alimentación de los consumidores españoles priman los productos frescos. Los principales productos demandados son, la carne que supone un 22,1 % sobre el gasto total, las patatas, frutas y hortalizas frescas (17,2 %) los pescados un 13,1 % y la leche y derivados lácteos (12,2 % sobre el gasto total). Al mismo tiempo, también se configuran como productos relevantes el pan (5,7 %), los productos de bollería y pastelería (4,0 %), los platos preparados (3,0 %) y el aceite de oliva (2,0 %) (MAGRAMA, 2015).

Por otro lado, en base al anuario de estadística en 2014 en la Comunidad Autónoma del País Vasco (CAPV) los sectores de pan y pastelería, vino y transformación de pescado fueron los sectores con mayor importancia en lo que respecta al número de empleados (Tabla 1).

Tabla 1. Información sobre los sectores alimentarios de la CAPV (fuente: MAGRAMA, 2015).

	Personas ocupadas		**Ventas**		**Compra mp***	
	Nº empl.	**%**	**10^6 €**	**%**	**10^6 €**	**%**
Industria cárnica	1.051	7,5	168	5,0	109	5,4
Transformación de pescado	1.442	10,3	214	6,3	108	5,4
Conservas vegetales	180	1,3	23	0,7	9	0,5
Grasas y aceites	377	2,7	546	16,2	510	25,5
Industria láctea	1.158	83,0	337	10,0	196	9,8
Pan, pastelería y pastas	4.327	30,9	437	12,9	149	7,5
Azúcar, chocolate y confitería	569	4,1	73	2,2	34	1,7
Otros	1.210	8,6	224	6,6	93	4,7
Alimentación animal	403	2,9	247	7,3	173	8,7
Vinos	2.348	16,8	453	13,4	133	6,6
Otras bebidas alcohólicas	278	2,0	34	1,0	12	0,6
Aguas y bebidas no alcohólicas	642	4,6	624	18,5	473	23,6
Total Ind. Alimentaria CAPV	13.985	100	3.380	100	1.999	100
Total Industria País vasco	178.817		41.390		22.269	

*mp = materias primas

Según la Organización de Agricultura y Alimentación de las Naciones Unidas (FAO, por sus siglas en inglés Food and Agriculture Organization) la cadena alimentaria es uno de los sectores con mayor crecimiento a nivel mundial y es clave para el desarrollo de la población (FAO, 2009). Así, se espera que la población mundial aumente hasta 9.200 millones en 2050, y según las proyecciones, para alimentar a la población mundial será necesario aumentar la producción de alimentos en un 70 %. Asociado a esta intensificación de la producción de alimentos, también aumentara de manera significativa el impacto que ejercen las diferentes etapas de la cadena alimentaria sobre el cambio climático, uso del suelo, explotación de recursos o toxicidad humana entre otras. Todo ello hace que asegurar la sostenibilidad en la cadena alimentaria sea un tema crítico para el futuro del planeta.

1.2. Principales impactos y retos ambientales del sector alimentario

Es muy importante tener en cuenta el enfoque holístico y global de la cadena alimentaria. En las últimas décadas, la cadena alimentaria ha pasado de considerarse como un sector primario a ser un sector casi completamente industrializado (Booth y Coveney, 2015). Este fenómeno se debe a las necesidades cada vez más exigentes en políticas de seguridad alimentaria, que exigen que los alimentos una vez producidos deban ser procesados con el fin de garantizar inocuidad (tratamientos térmicos, envasados, etc.) y adaptarlos a las necesidades de los consumidores (eliminar partes no deseadas, precocinados, etc.).

La cadena de alimentos afronta desafíos inéditos que surgen de la acelerada urbanización, la globalización del comercio de alimentos, el cambio en los hábitos de consumo y las técnicas más intensivas de producción de alimentos. Si bien ya a principios del siglo XXI se comenzó a tratar la cadena alimentaria de una forma integrada de cara a respetar la inocuidad y seguridad alimentaria, no ha sido hasta los años 90 cuando se ha adoptado el mismo enfoque para asegurar el mínimo impacto ambiental de la producción de los alimentos.

Según la Agencia Europea del Medio Ambiente (EEA), las emisiones directas de CO_2 de la industria alimentaria en 2005 representaron el 0,9 % del total de emisiones de gases efecto invernadero (GEI) en la UE15. Las emisiones de GEI de la cadena alimentaria no dependen exclusivamente del proceso de fabricación y producción propio de la industria. Es más, según datos de DEFRA, la mayor fuente de GEI en la cadena alimentaria es la agricultura y ganadería con el 49 % de las emisiones, debido a las emisiones de metano y N_2O, seguida por los consumidores con el 18 % sobre el total de las emisiones. La producción de alimentos figura en tercer lugar, con un 11 % de las emisiones de GEI de la cadena alimentaria (Sánchez et al., 2011).

La Ley Española 2/2011, de 4 de marzo de Economía Sostenible (LES) es en este sentido, un ejemplo de cómo los diferentes Gobiernos mundiales están demandando a las empresas mayor compromiso que asegure su desarrollo sostenible, y la capacidad del ciudadano para contar con la información necesaria para tomar sus propias decisiones.

Por otro lado, en los últimos años han surgido a nivel de la Unión Europea (UE), diferentes iniciativas programáticas y legales de cara a fomentar el consumo y la producción sostenible. En este sentido, cabe destacar el Plan de Acción sobre Consumo y Producción Sostenible y una Política Industrial Sostenible, de julio de 2008, así como diferentes iniciativas legales tales como la ampliación del ámbito de aplicación de la Directiva sobre diseño ecológico de productos que consumen energía, la modificación de la Directiva Europea 2010/30/CE sobre etiquetado energético o el Reglamento (CE) nº 66/2010 sobre la etiqueta ecológica.

Desde principios de los años 90 la sociedad y las diferentes administraciones comenzaron a tomar conciencia de la importancia de salvaguardar el medio ambiente y minimizar los impactos que la fabricación de productos pudiera ocasionar. Sin embargo, pese a que es necesario tratar la cadena alimentaria de una forma integrada, cada etapa de la cadena posee unas características únicas que es necesario tratar por separado.

1.2.1. Agricultura

La agricultura se define como “una actividad llevada a cabo por el hombre que a través de cultivar la tierra produce alimentos para la población humana.”

En términos generales la agricultura representa el mayor usuario de agua dulce, suponiendo un 70 % del consumo mundial y el 37 % del consumo de agua en la UE-27 (Lopez-Gunn et al., 2012; FAO, 2003). Asimismo, según el Panel Intergubernamental para el Cambio Climático (IPCC, de sus siglas en inglés *Intergovernmental Panel on Climate Change*) también es responsable de más del 14 % de las emisiones totales de GEI a nivel mundial (IPCC, 2014).

Los aspectos más importantes son entre otros, la aplicación de fertilizantes y pesticidas, el consumo de agua para regadío, el consumo de diésel o el uso de suelo (Brentup et al., 2004). A continuación se describen cada uno de ellos:

1.2.1.1. Fertilizantes

Con el fin de aumentar la eficiencia de los cultivos, los fertilizantes son un insumo esencial en la agricultura. Los cultivos necesitan micronutrientes como nitrógeno (Figura 2), fósforo y potasio, aparte de otros micronutrientes como azufre, manganeso, boro, molibdeno y zinc. Estos elementos se encuentran en los suelos en forma de sales inorgánicas (urea, fosfatos, etc.), sin embargo en muchas ocasiones los suelos no contienen la cantidad adecuada de elementos y es necesario un aporte externo. El uso de fertilizantes tanto orgánicos como inorgánicos conlleva altas emisiones de amonio y nitratos que producen impactos de eutrofización y acidificación de las aguas superficiales y subterráneas y en los suelos (EC, 2009; EEA, 2003). Debido a los microorganismos presentes en el suelo que realizan procesos de nitrificación y desnitrificación el aporte de nitrógeno también crea gases de efecto invernadero como el N_2O. El uso de fertilizantes químicos como el nitrógeno mineral y la urea son también importantes fuentes que conducen a emisiones de NH_3. En 2010 en la UE, la agricultura fue responsable del 94 % de las emisiones de NH_3 (EEA, 2012).

Otra emisión, pero esta vez al agua, relativa a los procesos agrícolas es debida a los lixiviados de nitratos. La magnitud depende de muchos factores tales como el tipo de suelo, el clima, el tipo de cultivo, el sistema de laboreos, las tasas de aplicación de fertilizantes nitrogenados y la aplicación de estiércol. Estas pérdidas de nitratos afectan directamente al potencial impacto de eutrofización y acidificación (Smolders et al., 2010).

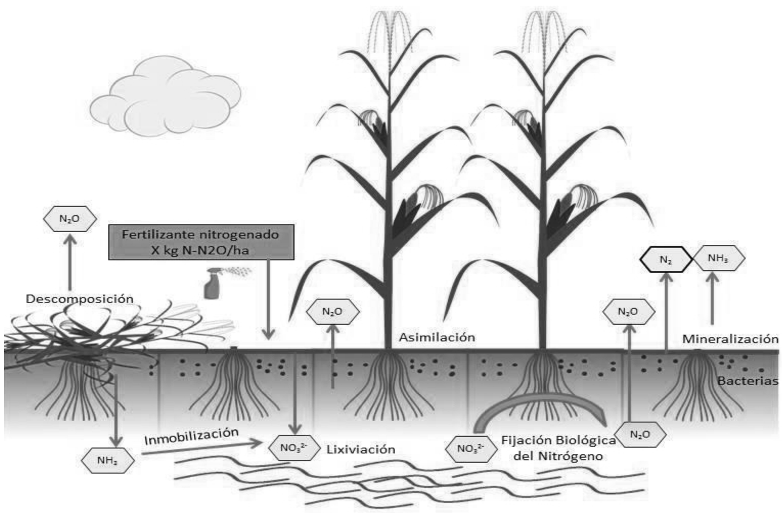

Figura 2. Principales flujos del nitrógeno en los campos de cultivo para pasto y pienso de ganadería.

Pero no solo el uso de fertilizantes crea un potencial impacto ambiental directo, la producción industrial de los fertilizantes conlleva también un potencial de impacto "indirecto" al cambio climático debido a las emisiones de CO_2, CH_4 y N_2O que se generan durante su síntesis (IFA y UNEP, 2002).

1.2.1.2. Uso de Pesticidas

Con el fin de evitar la pérdida de cultivos debido a posibles plagas, la aplicación de pesticidas es una práctica habitual. Al ser sustancias biológicamente activas con una toxicidad inherente al compuesto específico, los pesticidas son considerados como fuentes significativas de contaminantes, con consecuencias sobre la integridad de salud humana y el medio ambiente (Juraske et al., 2011; Juraske et al., 2009). Otras actividades agrícolas relacionadas como la deforestación, el drenaje de los humedales, de uniformidad genética en las tierras de cultivo, tienen también un efecto notable en la reducción de la biodiversidad.

Aunque los pesticidas se aplican en cantidades más pequeñas en comparación con los fertilizantes y correctores del suelo, su grado de persistencia ambiental es mucho mayor. El cobre, que generalmente se usa como fungicida, es una de las emisiones que más contribuyen a la toxicidad (Sanjuán et al., 2005). Sin embargo, su efecto tóxico varía considerablemente en función de las características del terreno, disminuyendo con la cantidad de materia orgánica por ejemplo (Kellogg et al., 2000).

1.2.1.3. Uso de agua: Riego

La agricultura requiere grandes cantidades de agua para regadío, además de agua de calidad para los distintos procesos productivos. El sector agrícola se posiciona como el mayor consumidor de agua del planeta dada su función productiva, no solo de alimentos, sino también de otros cultivos no comestibles como el algodón, el caucho o los aceites industriales cuya producción no deja de crecer. El regadío demanda hoy en día cerca del 70 % del agua dulce extraída para uso humano (FAO, 2003).

1.2.1.4. Consumo de energía

El consumo de energía en la etapa de la agricultura incluye principalmente el diésel necesario para las labores de campo, así como la energía eléctrica requerida para las instalaciones de las granjas. Estos usos de energía suponen principalmente impactos de cambio climático, acidificación y agotamiento de recursos fósiles (Beckman et al., 2013).

1.2.1.5. Uso de suelo

El uso y la transformación de la tierra para el cultivo agrícola es un factor esencial en el análisis de los sistemas agrícolas. El uso del suelo se refiere al área utilizada para el cultivo dividido por el período de vida medio de la labranza (Mourad, 2007).

La expansión de las tierras de cultivo se produce por lo general, a expensas de los bosques y praderas, que también tienen usos esenciales. Este proceso conduce a problemas ambientales asociados con la deforestación, de las cuales la pérdida de biodiversidad y emisiones de CO_2 son los más importantes. Por otra parte la calidad del suelo puede cambiar debido a la producción agrícola con los cambios físicos que alteran los ecosistemas y afectan a la biodiversidad (Müller-Wenk y Brandao, 2010; Koellner et al., 2013).

1.2.2. Pesca

La actividad pesquera produce alrededor del 57 % del total de pescado consumido el cual representa el 16 % del consumo mundial de proteínas, siendo esta proporción considerablemente más elevada en algunas naciones en desarrollo y en regiones muy dependientes de los recursos marinos. No obstante, esta actividad está cercana a su límite de crecimiento ya que la mayoría de las poblaciones de especies comerciales se encuentran en un elevado grado de explotación (FAO, 2014).

Como la demanda está acercándose a los límites de la producción, muchos recursos pesqueros están sufriendo deterioro (FAO, 2014). La contaminación procedente de las áreas industriales, urbanas y agrícolas, el uso de la tierra en las cuencas hidrográficas y el uso y gestión del agua, son algunos factores que están ejerciendo impactos negativos en la pesca.

1.2.2.1. Impacto al suelo marino

Principalmente las redes de arrastre distorsionan significativamente los sistemas de los fondos marinos. En un estudio realizado recientemente por el Consejo Superior de Investigaciones Científicas (Puig et al., 2012), se demuestra que el arado reiterativo de los sedimentos blandos del lecho marino por las artes de arrastre ha alterado la dinámica sedimentaria natural y ha suavizado y simplificado la morfología submarina original. Este estudio compara los efectos de las redes y puertas de arrastre con los efectos de las labores agrícolas.

1.2.2.2. Sobreexplotación de stocks

El principal efecto ecológico negativo directo de la pesca de captura es la explotación excesiva de los recursos piscícolas. La pesca desmesurada no sólo degrada la población de las especies objetivo, cambiando su tamaño y estructura, sino que también influye en las otras especies, relacionadas con la cadena de alimentos.

La Organización Mundial para la Agricultura y la Alimentación estima que el 77 % de las especies con valor comercial están afectadas en mayor o menor grado de sobrepesca: 8 % ligeramente, 17 % en sobreexplotación y 52 % en sobreexplotación máxima (FAO, 2014).

1.2.2.3. Descartes

En muchas ocasiones el uso de ciertos equipos y artes de pesca, que no atrapan exclusivamente la especie deseada, o que destruyen los hábitats, perjudican o mata, involuntariamente, las especies no objetivo (Vazquez-Rowe et al., 2011). Estas capturas que son devueltas al mar ya muertas son los descartes.

Un estudio llevado a cabo por la FAO en 2014, estimó en 7,3 millones de toneladas la tasa mundial de descartes, siendo las pesquerías de arrastre de camarón las que mayores porcentajes de descarte obtienen, con alrededor del 60 % de las capturas descartadas en barco. En el mismo informe declaran que por ejemplo, la pesquería de arrastre de merluza en el golfo de Bizkaia puede alcanzar un 56 % de tasa de descartes, lo cual es uno de los mayores valores a nivel mundial. Si bien los descartes son rápidamente reincorporados en la cadena trófica mediante los carroñeros o aves marinas, suponen un grave problema para la estabilidad de los ecosistemas naturales.

El problema de los descartes pesqueros está siendo duramente legislado por la comunidad europea, quien en 2013 publicó la nueva Política Pesquera Común mediante el Reglamento (UE) n° 1380/2013, de 11 de diciembre de 2013, donde se prevé una total prohibición de descartes en barco para el año 2019.

1.2.2.4. Consumo de fuel

Recientemente se ha empezado a tener conciencia ambiental del alto consumo de combustibles fósiles de las diferentes pesquerías. El aumento del esfuerzo pesquero y la disminución de las capturas ha llevado a muchos agentes a tomar decisiones con el fin de reducir la demanda de los combustibles fósiles (Watson et al., 2013). Debido a la sobreexplotación de los stocks comerciales, los barcos tienen que faenar más lejos obteniendo capturas de menor calidad, por lo que el consumo de combustible relativo al desembarco ha aumentado con el tiempo en algunas pesquerías (Hospido y Tyedmers 2005, Schau et al., 2009; Basurko et al., 2013). A nivel económico este hecho se ve mitigado gracias a los subsidios con los que cuenta la pesca (Arnason et al., 2008). A nivel mundial la actividad pesquera supone un requerimiento de 1,2 % del consumo mundial de petróleo, sin embargo, existen muchas diferencias entre las diferentes pesquerías, dependiendo del arte de pesca, región, especie objetivo, etc. (Tyedmers et al., 2005).

Se calcula que a nivel mundial, la pesca consume casi 50 mil millones de litros de combustible con poco más de 80 millones de toneladas de peces e invertebrados marinos, lo cual supone en términos generales para una tasa promedio de 620 L por tonelada desembarcada (Tyedmers et al., 2005). Basurko et al. (2013) estudiaron el consumo de fuel por especies para la flota vasca obteniendo valores de entre 1646 L por tonelada para peces demersales capturados por pesca arrastre o 98 L por tonelada para el Jurel o Chicharro con pesca de cerco.

1.2.3. Ganadería

La ganadería es una de las etapas de la cadena alimentaria con mayor impacto ambiental debido, principalmente a las emisiones de metano que contribuyen con un promedio de 52 % de las emisiones totales de la cadena de productos lácteos (FAO, 2010).

1.2.3.1. Fermentación entérica

Las principales emisiones de metano de los sistemas ganaderos provienen de la fermentación entérica que los rumiantes producen como subproducto de su metabolismo lo cual contribuye al calentamiento global y la oxidación fotoquímica (Crosson et al., 2011). Aproximadamente el 6 % de la ingesta en forma de energía se convierte en metano pero puede variar entre 2-12 % dependiendo de la dieta. Por ejemplo, según un estudio de Jhonson y Johnson de 1995, el forraje de baja calidad aumenta la producción de metano biogénico. Actualmente, se están realizando investigaciones sobre la forma de reducir estas emisiones aumentando la productividad de cada unidad de ganado u optimizando los ingredientes de los piensos (Storlien et al., 2015; Kamra et al., 2015), pero por mucho que se reduzca al mínimo estás emisiones, son parte del metabolismo del ganado y seguirán existiendo.

1.2.3.2. Piensos

Otro aspecto muy importante a tener en cuenta es la producción de los piensos necesarios para la producción ganadera a nivel industrial. Debido a la necesidad de generar piensos estables a lo largo del año las explotaciones ganaderas han pasado del pastoreo a la alimentación mediante piensos comerciales (Cederberg y Mattsson, 2000; Eide, 2002). La mayor parte de los ingredientes de estos piensos proceden de cultivos agrícolas, con lo que la ganadería suma así los impactos descritos anteriormente sobre este tipo de productos (Eide, 2002).

1.2.3.3. Gestión del estiércol

Las emisiones de metano y óxido de nitrógeno generadas durante la gestión y manipulación del estiércol son otro de los aspectos más importantes a tener en cuenta dentro de la ganadería (Henriksson et al., 2011). El alcance de las emisiones de GEI del almacenamiento y tratamiento de los purines, depende directamente de la cantidad de estiércol, la relación C/N, la proporción que se descompone anaeróbicamente y la temperatura, duración y tipo de almacenamiento (Montes et al., 2013).

Como norma general, los purines más líquidos generan proporcionalmente más CH_4 procedente de una descomposición anaerobia, mientras que los sólidos producen más N_2O (Amon et al., 2006). También las emisiones de amoníaco del estiércol durante almacenamiento y la aplicación como abono, representan una importante pérdida de nitrógeno y contribuyen considerablemente a los potenciales impactos de eutrofización y acidificación (Montes et al., 2013). Cabe destacar que en un estudio llevado a cabo por la FAO en 2010, donde estudiaron la sensibilidad del potencial de cambio climático respecto a los diferentes sistema de gestión del estiércol, se concluyó que las emisiones del sistema de gestión del estiércol son relativamente robustas y no muestran una variación significativa dependiendo de las prácticas de gestión del estiércol.

1.2.4. Acuicultura

Si bien la pesca es un aporte esencial de proteínas a nivel mundial, los sistemas acuícolas se presentan como una alternativa viable para evitar los problemas derivados de la pesca (Tidwell y Allan, 2001).

1.2.4.1. Vertidos

Los vertidos de agua contaminada procedentes de las piscinas pueden contaminar los ambientes acuáticos cercanos. Esta contaminación orgánica está muy influenciada por el tipo y procedimiento de gestión de dichas corrientes, los mecanismos de las piscinas para la renovación y purificación de las aguas y la tasa de ingesta de los peces (Heldbo et al., 2013). Asimismo, las substancias químicas que se utilizan en las piscinas para la esterilización y control de plagas, pueden suponer un impacto en la toxicidad de las aguas locales. La calidad de las aguas de recepción en el momento de vertido, así como su capacidad de dilución y dispersión, determinará el efecto del afluente en el ambiente acuático receptor (Henriksson et al., 2011).

1.2.4.2. Piensos

Al igual que ocurre en la etapa de ganadería, para obtener un mayor rendimiento en los sistemas de acuicultura, también son necesarios los piensos comerciales. Los ingredientes de estos piensos proceden de materias primas de origen agrícola o de pesca, por lo que los productos de acuicultura, suman también los impactos originados en esas etapas (Pelletier et al., 2009, Papatryphon et al., 2004)

1.2.4.3. Escapes de especies exóticas

Otros impactos negativos potenciales de la piscicultura se originan en el uso de las especies exóticas. En este sentido, se producen efectos negativos en las especies nativas silvestres a raíz de la difusión, mediante las especies exóticas, de las enfermedades y los parásitos, o la fuga de los peces del estanque (Silva et al., 2009; Gausen y Moen, 1991).

1.2.5. Transformación y procesado

Casi todos los alimentos pasan por algún tipo de proceso antes de poder ser consumidos. La transformación de los alimentos puede tener muchas causas, como la conservación, la mejora de la disponibilidad de nutrientes, garantizar la higiene y seguridad alimentaria, aumentar la vida útil del producto, adaptar el producto a los consumidores o facilitar el transporte y la disponibilidad.

Sin embargo, el procesado de alimento lleva consigo una serie de aspectos ambientales (consumo de agua y energía, entre otros) que impactan de forma considerable en el medio ambiente (Sanjuan et al., 2014).

1.2.5.1. Consumo energético

El consumo energético en los procesos productivos suele estar concentrado en los procesos de cocinado o de almacenamiento en frío. Las técnicas tradicionales de cocinado por convección o cocción, requieren grandes cantidades de energía, normalmente en forma de vapor. Este vapor normalmente es generado a partir agua en calderas de gas o electricidad.

En lo que respecta al consumo energético global en 2010 en la UE, la industria de la alimentación y bebidas consumió 29 millones de toneladas equivalentes de petróleo, lo cual representa una participación del 10 % de la energía total consumida por la industria

(EUROSTAT, 2012). Esto coloca a la industria alimentaria en la cuarta posición, detrás de la industria del hierro y el acero, los productos químicos y petroquímicos y los minerales no metálicos.

A pesar de que muchas de las actividades *upstream* y *downstream* en la cadena de valor de los alimentos, como la agricultura, la distribución y las ventas, representan una parte significativa de los GEI de la industria, alrededor de un tercio de la energía bruta requerida se asocia a la fase de producción (O'Shaughnessy, 2013; DEFRA, 2006). En la tabla 2 se muestran algunos valores medios de consumos energéticos para distintos tipos de producto.

Tabla 2. Valores medios de consumo energético por tipo de producto (MAGRAMA, 2005abd, 2006).

	Consumo eléctrico (kWh/tn)
Transformados de pescado	828,2
Producto lácteo	39,0 - 448,0
Cerveza	8,4 - 14,4
Conservas vegetales	50,0 – 275,0
Congelados vegetales	200,0 – 6000,0

Al analizar en detalle los datos mostrados, observamos que los valores varían significativamente entre los diferentes productos. Sin embargo, los tratamientos de calor y frío son parte fundamental de los procesos del sector lácteo, siendo el proceso de pasteurización el procedimiento más habitual. Para la mayoría de los productos lácteos los requerimientos de energía térmica son mayores que los eléctricos, con procesos que incluyen la concentración (por ejemplo, de leche en polvo) como el proceso de mayor consumo y la producción de leche en bruto como el de menor consumo.

Cabe decir que el impacto de la energía eléctrica varía significativamente de un país a otro, siendo dependiente del mix eléctrico regional. Los países como España con un alto porcentaje de energía eléctrica procedente de combustibles fósiles como el carbón, tienen un mayor potencial de impacto al cambio climático, debido a los GEI generados durante la combustión. La intensidad de emisiones del sector eléctrico español en 2008 (326 g CO_2/kWh) era ligeramente inferior a la media europea (395 g CO_2/kWh) por emplearse una mayor proporción de energías renovables, aunque muy superior a la de países donde se emplea energía nuclear en gran porcentaje, como por ejemplo Francia (98 g CO_2/kWh). Este hecho hace que la huella de carbono del sector español de la alimentación y bebidas sea superior a la de otros países europeos como Austria, Bélgica, Eslovaquia, Francia, Finlandia, Letonia, Lituania o Suecia, en lo que a emisiones indirectas de consumo eléctrico se refiere (Sims et al., 2007). El hecho de que el impacto al cambio climático se ve fuertemente afectado por este hecho, implica que las industrias no tengan por tanto capacidad de control sobre dicho factor. Además del impacto al cambio climático, el consumo energético supone también un impacto en el agotamiento de recursos fósiles, oxidación fotoquímica y agotamiento de la capa de ozono.

En la actual situación de crisis financiera las empresas están invirtiendo esfuerzos y recursos económicos en minimizar los requerimientos energéticos de los procesos productivos, lo cual es beneficioso también para el medio ambiente. Hoy en día las industrias están apostando por metodologías más limpias y con menores requerimientos energéticos.

1.2.5.2. Consumo de agua

El agua es básica tanto como ingrediente para la producción de alimentos y bebidas, como elemento indispensable para muchas etapas de procesado (lavado, cocción, refrigeración, etc.), así como para garantizar el cumplimento de las normas de higiene. Por lo tanto, es otra prioridad del sector. En Europa el consumo de agua de las empresas alimentarias se ha reducido un 33 % entre 2006 y 2010, y los vertidos se han reducido considerablemente lo cual se refleja en el hecho de que el número de las instalaciones de depuración ha aumentado desde el 6,1% en 2004 hasta el 8,38% en 2008 (Sánchez et al, 2011).

No todos los productos requieren la misma magnitud de agua para su producción. Al igual que para el consumo energético, a continuación se muestran los valores genéricos de consumo de agua para una serie de productos tipo (Tabla 3).

Tabla 3. Valores medios de consumo de agua por tipo de producto. (MAGRAMA, 2005abd, 2006).

	m^3 / tn de producto
Conservas de frutas	2,5 – 4,0
Conservas vegetales	3,5 – 6,0
Congelados vegetales	5,0 – 8,5
Zumo de frutas	6,5
Mermelada	6,0
Transformados de pescado	19,5
Productos lácteos	1,0 – 6,0
Cervezas	4,4 – 8,6

Sin embargo, en muchas zonas de España, la calidad del agua de abastecimiento no permite que pueda ser utilizada directamente por la industria. En estos casos, las industrias aplican diferentes tratamientos al agua para adecuar sus características a las requeridas en los procesos (reducción de sales, desinfección, filtración, etc.). Por lo general, estos tratamientos requieren un consumo adicional de agua para compensar los rechazos o lavados de las columnas de intercambio, haciendo que el consumo total de agua sea superior al realmente necesario para desarrollar la actividad. Este consumo "extra" puede aumentar significativamente el consumo específico de agua de la industria (calculado como consumo de agua por unidad de producción), entre industrias del mismo sector situadas en diferentes zonas geográficas (FIAB, 2008).

La reutilización de agua dentro de la propia instalación es una de las posibles estrategias para reducir el consumo de agua, así como los costes directamente relacionados con este consumo, como son los costes de depuración o el canon de vertido. Sin embargo, la reutilización de agua en la industria alimentaria está fuertemente limitada por los riesgos que puede suponer para el mantenimiento de la seguridad higiénica de los productos alimenticios elaborados (Real Decreto 1620/2007, de 7 de diciembre, por el que se establece el régimen jurídico de la reutilización de las aguas depuradas). Por otro lado, la reutilización de agua dentro de la instalación también está condicionada desde el punto de vista técnico, por la disponibilidad de sistemas de regeneración o acondicionamiento adaptados a las características del agua a reutilizar, y por el coste económico de dichos sistemas. Debido a estas limitaciones, el porcentaje de reutilización de agua en las industrias alimentarias es sólo del 2,4 %, cifra bastante inferior al del conjunto de la industria manufacturera, la cual se sitúa en el 8,9 % (Datos elaborados a partir de la Encuesta sobre el uso del agua en el sector industrial del INE, 1999).

1.2.5.3. Generación de vertidos

Como se describía anteriormente, en muchas ocasiones, la demanda de agua no se atribuye únicamente a la formulación del producto, también es esencial para su uso en procesos productivos y de limpieza, por lo que estas aguas se desechan en modo de vertido. Además estos vertidos suponen no solo un impacto debido a los sólidos y otros contaminantes que puedan ser arrastrados, en muchas ocasiones suponen también una pérdida de materia prima.

El tratamiento de aguas residuales en la industria de alimentación y bebidas representa un importante reto para el sector, es por ello que la industria está aplicando importantes esfuerzos en este sentido. La inversión en sistemas para la depuración de aguas residuales en la industria alimentaria es muy elevada (en el año 2006 fue de 50.989.279 €), lo que representa el 36 % de la inversión en depuración de agua de toda la industria manufacturera) y supone la mayor de las inversiones realizadas por la industria alimentaria en corrección ambiental (O'Shaughnessy, 2013).

Los elementos con mayor preocupación respecto a los vertidos de las industrias agroalimentarias son: la demanda bioquímica de oxígeno (DBO), los sólidos en suspensión (SST), la carga excesiva de nutrientes (compuestos de nitrógeno y fósforo), organismos patógenos y los niveles de cloro y pesticidas residuales (UNIDO, ND).

En la Tabla 4 se muestran algunos de los valores de estos elementos en los vertidos de las industrias alimentarias.

Tabla 4. Valores medios de caudal y carga contaminante (DBO y sólidos en suspensión SST) de unos productos alimentarios tipo por tonelada de producto (MAGRAMA, 2005abd, 2006).

	Caudal (m^3/tn)	Carga
Conservas de frutas	3,5 - 8,5	DBO5: 140 a 7.000 mg/l SST: 60 a 3.000 mg/l
Conservas y congelados vegetales	3,5 - 8,5	DBO5: 140 a 7.000 mg/l SST: 60 a 3.000 mg/l
Zumo de frutas	0,5 - 6,5	DBO5: 500 a 3.200 mg/l SST: 300 a 1.300 mg/l
Transformados de pescado	10,9	DBO5: 104-3.800 mg/l SST: 2-1.400 mg/l
Litro de leche	3,5	DBO5: 728–5.973 mg/l SST: ND
Cerveza	2,5-7,2	ND

ND: No data

1.2.5.4. Generación de subproductos y residuos

De acuerdo con los datos de la Comisión Europea (2012), en España se pierden ocho millones de toneladas de alimentos anualmente. La mayoría de este desperdicio se produce en los hogares (42 %) aunque todavía sigue habiendo un 39 % que corresponde al procesado de los alimentos, un 14% a la restauración y un 5 % a la distribución (Bagherzadeh et al, 2014; MAGRAMA, 2013).

En muchas ocasiones el desperdicio alimentario viene dado por las características intrínsecas del propio procesado del alimento., como por ejemplo 10 % de residuos orgánicos en cadena de transformación de pescado, 50 % de pérdida de materia prima en zumos de frutas o 25 % de residuos orgánicos en la elaboración de cervezas (MAGRAMA, 2005abd, 2006),. Sin embargo, existen muchos casos en los que el desperdicio es evitable y se produce debido a ineficiencias de proceso, maquinaria de procesado en mal estado, al planteamiento no adaptado de la ingeniería de la planta, falta de regulación de la producción, almacenamiento inadecuado o la falta de previsión en el transporte, entre otras.

Frente a esta problemática la Comisión Europea mediante la Directiva marco de residuos 2008/98/EC se ha fijado como objetivo reducir a la mitad este desperdicio en 2020.

La industria de la alimentación y bebidas en lo que a la gestión de residuos se refiere, pasa por el máximo aprovechamiento posible de las materias primas empleadas y tiene como objetivo final comportarse como biorefinerías, en las que los recursos primarios se separan en diferentes componentes.

La generación de residuos tiene un problema ambiental añadido que son las emisiones de CH_4 que también pueden ocurrir cuando la materia orgánica se descompone. Estos residuos por tanto suponen no solo una pérdida de materia prima, sino también un potencial de impacto de cambio climático.

1.2.5.5. Envasado

Los actuales hábitos de consumo, como son la reducción progresiva del número de ocupantes en los hogares o el aumento de las compras por internet, han derivado en una mayor generación de envases (Ecoembes, 2012). La industria de la alimentación y bebidas es un importante usuario de envases debido a que el envase es esencial para garantizar la calidad, seguridad e higiene del alimento.

El envase maximiza el tiempo de caducidad de los alimentos, evitando de este modo una mayor generación de desperdicios alimenticios en las etapas de venta y consumo. Según un estudio del WRAP (2010), del total de la comida que se compra se llega a tirar cerca de un tercio, y la mitad de ésta podría haber sido utilizada si los sistemas de envasado y manejo hubieran sido los adecuados (Sharp et al., 2010).

Sin embargo, la producción y gestión final de los materiales utilizados en el envasado suponen un impacto ambiental. Así, los residuos de envase representan en la actualidad un 17 % de los residuos sólidos urbanos en peso, y entre un 20-30 % en volumen (Ecoembes, 2012). Si estos residuos no son aprovechados y se gestionan en incineración o vertedero puede llegar a suponer un potencial impacto al cambio climático y a la acidificación (Beccali, 2009).

En este sentido, la reducción del peso de los envases es uno de los aspectos donde la industria ha realizado mayores esfuerzos en la última década, observándose que la evolución de la relación entre el peso del envase y el peso del producto puesto en el mercado ha disminuido un 14 % desde el año 2000 (Ecoembes, 2001). No obstante, existen una serie de limitaciones legales, técnicas y económicas al diseño de los envases, como son los aspectos relacionados con la sanidad e higiene (De Pablo y Moragas, 2015), las características de algunos productos que exigen envases con ciertas propiedades (como la rigidez, resistencia a la presión interna) o la inversión necesaria para llevar a cabo el cambio de un tipo de envase por otro.

1.2.6. Transporte y distribución

Se estima que entre el 15 y el 30 % de la huella de carbono de la industria de alimentación y bebidas se debe a las actividades de transporte y, teniendo en cuenta que en la mayoría de los casos este es un servicio prestado por proveedores externos, el transporte es un aspecto de la cadena alimentaria a tener en especial consideración por la industria (Virtanen et al., 2011; Wakeland et al., 2012). Además, es necesario mencionar que debido a la internacionalización de las cadenas alimentarias las operaciones de transporte dentro de la industria han incremento notablemente en las últimas décadas, tanto en volumen como en distancia.

El transporte de mercancías en España está basado principalmente en el transporte por carretera (95,9 %), siendo la tasa de utilización de ferrocarril del 4,1 %, muy inferior a la media europea, donde el porcentaje de uso se sitúa entre el 11,7 % y el 61,3 %, dependiendo del país (Perez-Arriaga et al., 2008; EC, 2012). En lo que respecta al transporte de mercancías por carreteras, cabe destacar que la vida media de los vehículos influye en la cantidad de emisiones de GEI emitidas a la atmósfera (Lipman y Delucchi, 2002). En este sentido, Según el RD 89/2013 España se sitúa entre los países de la UE con un parque de vehículos de transporte de mercancías más envejecido (más del 70 % de sus vehículos con más de 7 años de antigüedad) y por lo tanto más contaminante.

1.2.7. Resumen de los principales impactos ambientales

A continuación, en la Tabla 5 se muestra a modo de resumen los principales aspectos ambientales de las etapas de la cadena alimentaria y los impactos ambientales asociados a cada uno de ellos.

Tabla 5. Resumen de los principales aspectos asociados a las diferentes etapas alimentarias, y su impacto asociado.

Etapa	Aspecto	Impacto
Agricultura	Fertilizantes	Eutrofización
	Pesticidas	Toxicidad humana y ecotoxicidad
	Consumo de Fuel	Cambio Climático
	Riego	Estrés hídrico
	Uso de Suelo	Uso de suelo
Pesca	Pesca excesiva	Sobreexplotación
	Descartes	Pérdida de Biodiversidad
	Consumo de fuel	Cambio Climático
		Agotamiento de recursos fósiles
	Arrastre en suelo marino	Cambios en los ecosistemas
Ganadería	Emisión entéricas	Cambio Climático
	Gestión de purines	Cambio Climático
		Eutrofización
	Consumo de agua	Estrés hídrico
	Pesticidas	Toxicidad
	Uso de suelo	Uso de suelo
Acuicultura	Vertidos	Eutrofización
	Escapes de especies exóticas	Pérdida de biodiversidad
	Piensos	Eutrofización
		Toxicidad humana y ecotoxicidad
		Cambio climático
		Estrés hídrico
Procesado	Consumo de agua	Estrés hídrico
	Consumo de energía	Cambio climático
		Acidificación
		Agotamiento de recursos
	Consumo de agua	Estrés hídrico
	Generación de residuos	Cambio climático
		Acidificación
		Agotamiento de recursos
	Generación de vertidos	Eutrofización
		Acidificación
		Ecotoxicidad
Transporte	Consumo de fuel	Cambio climático
		Acidificación
		Agotamiento de recursos

Ante esta situación de sostenibilidad, la industria de alimentación y bebidas debe enfocar sus esfuerzos en reducir de forma significativa los impactos ambientales asociados a su propia actividad, enfocando los esfuerzos a implantar acciones de mejora en cada una de las etapas de la cadena alimentaria. Sin embargo, para poder evaluar la mejora obtenida, y saber si los esfuerzos se están enfocando a los aspectos ambientales más significativos, es necesario identificar, medir y calcular el estado actual de impacto ambiental de las diferentes actividades. Para ello, el Análisis de Ciclo de Vida (ACV) es una herramienta que está obteniendo muy buenos resultados.

1.3. Análisis de Ciclo de Vida como herramienta de evaluación ambiental

La preocupación del consumidor por el medio ambiente, está marcando las tendencias de mercado. La industria debe adaptarse, por tanto, a los requisitos ambientales y sociales a la vez que logra que sus procesos sean más eficientes. No solo la demanda por parte de los consumidores es quién ejerce esta presión de reducir los impactos ambientales de los productos, las actuales políticas europeas y estatales también están enfocadas a este objetivo.

La iniciativa promovida por la Unión Europea desde el año 2013 *Single Market for green product* merece una especial mención en este sentido. A finales de 2012, los resultados de la encuesta llevada a cabo por el eurobarómetro (flash 367) mostraban que la mitad de los consumidores, el 47 % de los consumidores europeos EU-27 y el 43 % de los españoles, no confían o tienden a desconfiar de las declaraciones ambientales de los productos. Con este marco se presentó en Abril de 2013 (COM/2013/0196 final) la iniciativa *Single market for green products* con 4 ejes principales:

- Desarrollo de dos metodologías para la Evaluación de la Huella ambiental de Producto y de Organización
- Desarrolla una recomendación a nivel Europeo para armonizar las metodologías de evaluación de Ciclo de Vida (2013/179/EU)
- Proporciona los principios para comunicar el comportamiento ambiental, fomentando la transparencia, fiabilidad, integridad, comparabilidad y claridad del análisis
- Apoyo los esfuerzos internacionales hacia una mayor coordinación en el desarrollo metodológico y la disponibilidad de datos

La iniciativa está basada en el análisis de ciclo de vida (ACV), por lo que hoy por hoy esta metodología se presenta como una herramienta indispensable para calcular los impactos ambientales y la toma de decisiones enfocadas la mejora ambiental. Esta metodología es una herramienta de diseño que investiga y evalúa los impactos ambientales de un producto o servicio durante todas las etapas de su existencia: extracción, producción, distribución, uso y fin de vida (reutilización, reciclaje, valorización y disposición final de los residuos).

Este enfoque de ciclo de vida completo del producto se basa en ser capaz de identificar claramente todos los aspectos ambientales (uso de recursos del medio ambiente, la generación de residuos y emisiones) de la cadena alimentaria que tiene un impacto ambiental. De esta forma se tienen en cuenta todos los impactos producidos a lo largo de la cadena productiva, no sólo los producidos por la propia fábrica o en una etapa específica del ciclo.

El ACV permite entender las relaciones causa técnica-efecto ambiental de las distintas operaciones y proceso que se llevan a cabo durante la producción y comercialización de un producto o servicio. Asimismo, cuenta también con otra serie de ventajas como la identificación de los *hot-spot* o aspectos con mayor impacto ambiental, simulación de escenarios para comparar alternativas de producción o la proporción de información para orientar sobre los objetivos a la hora de plantearse modificaciones en procesos o productos o la comparación entre productos similares.

Los resultados de los ACV también se han utilizado como una herramienta de información para la toma de decisión por parte de los consumidores (Jungbluth *et al.*, 2000), así como en el desarrollo de criterios de eco-etiquetado, para informar a los consumidores de las características medioambientales de los productos que se demandarán en el futuro (Mungkung, et al, 2006). Los resultados indican también la limitación inherente de los estudios de LCA en términos de la elección de la metodología y de las categorías de impacto estudiadas ya que algunas, como uso del suelo y biodiversidad, no están todavía bien definidas y estudiadas.

1.3.1. Metodología de Análisis de Ciclo de Vida

La metodología de ACV está estandarizada según la norma ISO 14040:2006. En este estándar se definen las principales etapas a tener en cuenta a la hora de realizar un estudio de ACV (Figura 3).

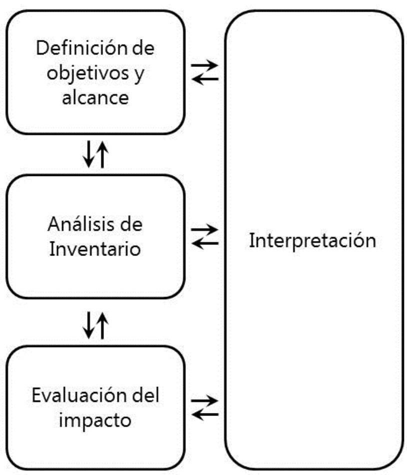

Figura 3. Etapas del ACV marcadas por la ISO 14040:2006.

1.3.1.1. Definición del objetivo y alcance del estudio

Un ACV comienza con una definición del objetivo y alcance del estudio. La norma ISO establece que el objetivo y el alcance de un ACV deben estar claramente definidos y ser consistentes con la aplicación prevista. Así el objetivo y ámbito de aplicación marcarán los detalles técnicos que guían el trabajo posterior. En esta fase es muy importante definir las categorías de impacto que van a ser estudiadas con el fin de obtener unos resultados que reflejen la realidad del comportamiento ambiental del objeto de estudio.

1.3.1.2. Inventario del ciclo de vida

El Inventario del Ciclo de Vida (ICV) implica la creación de un inventario de los flujos de entradas y salidas a la naturaleza de un sistema de producto. Los flujos de inventarios incluyen aportes de agua, energía y materias primas, y las emisiones a la atmósfera, al suelo y al agua. El modelo de flujo se ilustra con un diagrama de flujo que incluye las etapas que van a ser evaluadas en la cadena de suministro y ofrece una imagen clara de los límites del sistema. Los datos deben estar relacionados a la unidad funcional definida en la fase anterior.

1.3.1.3. Evaluación del impacto del ciclo de vida

El análisis de inventario es seguido por la evaluación de impacto. Esta fase está dirigida a evaluar de una forma estandarizada y comparable los potenciales impactos ambientales que ocurren a lo largo de la cadena.

En esta etapa se transforman los flujos de inventario en potenciales impactos ambientales de acuerdo con las metodologías seleccionadas en la primera fase. Dicho de otra forma, todas las entradas y salidas al sistema de estudio se traducen a potenciales impactos (Figura 4). Así, los diferentes flujos de inventario se les asignan unas unidades de equivalencia comunes que se suman para ofrecer una categoría de impacto. Estas unidades de equivalencia son los factores de caracterización de impacto.

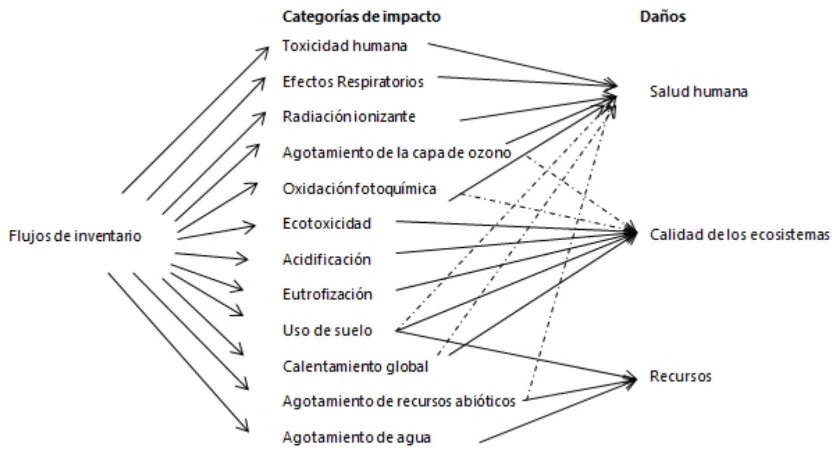

Figura 4. Esquema de caracterización de impactos, desde los flujos de inventario hasta los daños finales en los ecosistemas, salud humana y disponibilidad de recursos.

Habitualmente el ACV concluye en la caracterización de impactos ya que esta es la última etapa obligatoria según la norma ISO 14044:2006. Sin embargo, además de los pasos anteriores, existen otros elementos opcionales como la normalización, agrupación y ponderación de impactos que se pueden llevar a cabo en función del objetivo y el alcance del estudio. En la normalización, los resultados de las categorías de impacto del estudio se comparan con los efectos totales en la región de interés, como pueden ser los valores medios europeos. La agrupación consiste en la clasificación de las categorías de impacto en 3 grandes grupos de impacto: impacto a los recursos, ecosistemas y salud humana. El tercer paso es la ponderación, donde los diferentes impactos ambientales se ponderan con respecto al otro de modo que a continuación pueden ser sumados para obtener un solo número para el impacto medioambiental total.

1.3.1.4. Interpretación de resultados

La Interpretación es una técnica sistemática para identificar, cuantificar, verificar y evaluar la información de los resultados del inventario del ciclo de vida y / o la evaluación del impacto del ciclo de vida. El resultado de la fase de interpretación es un conjunto de conclusiones y recomendaciones para el estudio. De acuerdo con la norma ISO 14040:2006, la interpretación debe incluir:

- identificación de las etapas críticas basadas en los resultados;
- evaluación de la integridad, sensibilidad y consistencia de estudio teniendo en cuenta los controles; y

- conclusiones, limitaciones y recomendaciones.

1.3.2. Estudios e iniciativas sobre el ACV en los productos alimentarios

El ACV ha sido una herramienta ampliamente utilizada en sectores industriales clásicos desde 1960. Pero no fue hasta mediados de los noventa cuando salió a la luz uno de los primeros trabajos publicados sobre la utilización del Análisis de Ciclo de Vida en productos alimentarios. El estudio de Andersson et al. (1994) sobre el uso de ACV en los productos y procesos alimentarios estableció las reglas de uso de esta metodología como base para posteriores análisis de impacto.

Sin embargo no fue hasta cerca del año 2010 cuando realmente comenzó a utilizarse esta metodología de manera más amplia y consolidada (Hou et al., 2015) (Figura 5).

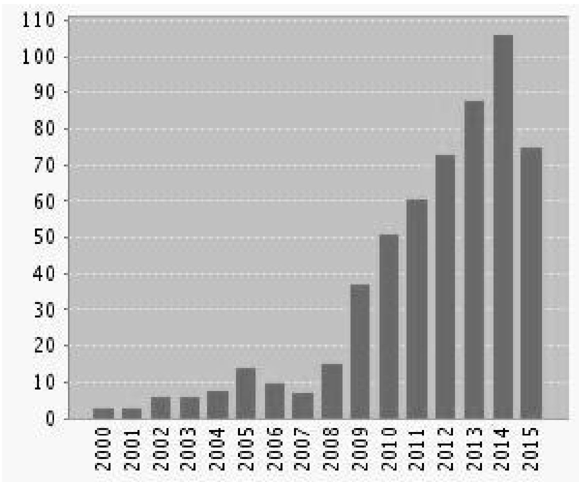

Figura 5. Evolución del número de artículos en material de ACV (Scientific Report Index, consultado en Noviembre 2015).

Como se irá viendo a lo largo de la presente tesis, uno de los principales aspectos clave a la hora de realizar un ACV representativo es seleccionar los indicadores adecuados que reflejen la realidad del sistema. El cambio climático o huella de carbono por ejemplo, evaluar mediante las emisiones GEI el impacto al calentamiento global o efecto invernadero. Sin embargo existen otros indicadores que describen el comportamiento ambiental de un producto, entre ellos el potencial de eutrofización, la acidificación o el aumento de la toxicidad en el medio.

1.3.3. Impactos ambientales

1.3.3.1. Cambio climático

El llamado efecto invernadero es un proceso normal en la atmósfera sin el cual la vida en la Tierra sería imposible. La radiación del sol pasa a través de la atmósfera calentando la superficie terrestre, parte de esa radiación es reflejada de vuelta a la atmósfera contribuyendo al aumento de la temperatura de la superficie. Sin embargo, las actividades humanas como la quema de combustibles fósiles o la descomposición de residuos orgánicos que emiten GEI, amplifican el efecto invernadero natural y provocan el calentamiento anormal de la superficie terrestre. Las principales sustancias que contribuyen al calentamiento global son los llamados Gases de Efecto Invernadero (GEI) como el dióxido de carbono (CO_2), metano (CH_4), óxido nitroso (N_2O) o hidrocarburos (Solomon et al., 2007).

1.3.3.2. Potencial de eutrofización

El impacto de eutrofización se produce debido a un aporte excesivo de nutrientes a los ecosistemas, que puede llegar a colapsar todos los procesos naturales del sistema donde sea vertido (EEA, 2012).

El desarrollo de la biomasa en un ecosistema viene limitado, la mayoría de las veces, por la escasez de algunos elementos químicos, como el nitrógeno en los ambientes continentales y el fósforo en los sistemas de agua dulce, que los productores primarios necesitan para desarrollarse y a los que llamamos por ello factores limitantes. La contaminación puntual de las aguas, por efluentes urbanos, o difusa, por la contaminación agraria o atmosférica, puede aportar cantidades importantes de esos elementos limitantes. El resultado es un aumento de la producción primaria (fotosíntesis) con importantes consecuencias sobre la composición, estructura y dinámica del ecosistema (Schindler et al., 2006; Schindler et al., 2009). La eutrofización produce de manera general un aumento de la biomasa y un empobrecimiento de la diversidad.

1.3.3.3. Acidificación

La acidificación se produce cuando la acidez del suelo se modifica debido a la deposición atmosférica de sustancias tales como sulfatos, nitratos y amoníaco. La acidificación se define como la reducción del pH del suelo o del agua hasta un punto en el que se producen cambios indeseables en los ecosistemas, por tanto, el potencial de acidificación se basa en la saturación de ácidos en el suelo (Posch et al., 2008; Krewitt et al., 2001; Seppälä et al., 2006).

Los compuestos sulfurados y de nitrógeno reactivo como óxidos de nitrógeno generados durante la quema de combustibles fósiles o las emisiones de amoniaco (NH_3) contribuyen significativamente a los procesos de acidificación (EEA, 2012).

1.3.3.4. Toxicidad Humana y ecotoxicidad

Para evaluar el potencial impacto en la toxicidad humana y en los ecosistemas de una substancia emitida, es necesario calcular la cadena de causa de los efectos toxicológicos. El impacto se evalúa en función de la cantidad en destino final de la substancia (atmósfera, suelo agrícola, suelo industrial, agua dulce o agua marina), la exposición a la substancia y de los efectos en la salud humana (Rosenbaum et al., 2008; Goedkoop et al., 2009).

En cuanto a la producción de alimentos, toxicidad humana se ve afectada principalmente por el uso de pesticidas o por metales pesados incluidos en los fertilizantes de fósforo (Foster et al., 2006; Tukker et al., 2006).

1.3.3.5. Uso de suelo

La inclusión de los aspectos ambientales relacionados con el uso del suelo en la metodología del ACV ha estado en desarrollo activo en los últimos años. Los diferentes enfoques se han desarrollado y propuesto para evaluar el uso del suelo impactos sobre la biodiversidad, los servicios ambientales o el cambio climático, por efectos tales como, la captura de carbono, la producción biótica, o la regulación de la erosión (Cherubini y Strømman, 2011; Köllner y Scholz, 2007; Milà i Canals et al., 2007).

1.3.3.6. Agotamiento de recursos abióticos

En relación con el agotamiento de recursos abióticos, el sector de alimentos y bebidas no difiere significativamente de otros productos de la industria. El agotamiento de los recursos abióticos de los alimentos depende en gran medida de los combustibles fósiles en la cadena de producción, como el petróleo crudo, carbón, gas natural y carbón marrón, mientras que los metales y minerales no presentan una contribución de gran relevancia El cobre, el zinc, el plomo y el fósforo son los más importantes recursos abióticos no fósiles en la cadena de productos típicos.

1.3.3.7. Agotamiento de agua

El potencial impacto ambiental asociado a la extracción de recursos hídricos está ganando cada vez más atención, siendo la Huella Hídrica uno de los indicadores de impacto más utilizados junto con el cambio climático. Tanto en la producción primaria como en la industria alimentaria en general el agua es un recurso básico y ampliamente utilizado tanto para usos de riego el mantenimiento de la higiene en las industrias. Sin embargo, dependiendo de la región y la disponibilidad de agua existente en cada cuenca el impacto asociado a este consumo puede variar significativamente. El análisis del agotamiento del agua se enfoca principalmente en la cantidad de agua extraída de una cuenca. La degradación de la calidad del agua a menudo se evalua dentro de las categorías de impacto independientes como la ecotoxicidad o eutrofización (Hoekstra et al., 2011; Pfister et al., 2009; Ripoll et al., 2010; Ridoutt y Pfister, 2010; Ridoutt y Pfister 2014).

1.4. Limitaciones del ACV a superar en el sector alimentario

Desde el punto del ACV, los sistemas alimentarios tienen diferencias o complejidades respecto a los procesos industriales convencionales. Una de las características principales de los sistemas alimentarios es que se basa en la utilización directa de recursos vivos, lo que hace que muchos procesos biológicos como el ciclo del nitrógeno o de carbono, que son muy difíciles de controlar y afecten significativamente a los posibles resultados de análisis de impactos. En la mayoría de las ocasiones estos procesos o ciclos de elementos varían considerablemente dependiendo de las condiciones locales (tipo de suelo, caudal, precipitación, evapotranspiración, etc.) por lo que requieren el uso de modelos de simulación adaptados (Mungkung y Gheewala, 2007).

Además, en los sistemas de producción primaria, los modelos de intercambio de sustancias entre el emisor y el medio ambiente receptor son complejos, y al contrario que en otros procesos industriales, estos modelos de intercambio afectan significativamente a los resultados del impacto. Por otra parte, muchos de los sistemas de producción primarios están relacionados entre sí. Así, pequeños cambios en un sistema podrían afectar significativamente a otros sistemas (Harris y Narayanaswamy, 2009). Por otra parte, los impactos ambientales de la producción primaria, así como el procesamiento de productos alimenticios pueden influir en los ecosistemas y los seres humanos dependiendo de donde están localizados (Rodriguez et al., 2014). La regionalización de los factores de caracterización para determinadas categorías de impacto puede ayudar a obtener resultados más precisos y contribuir a una mejor comprensión de los impactos ambientales en regiones específicas (Pfister, 2013; Manneh et al., 2010; Sedlbauer et al., 2007).

Debido a la complejidad descrita de las cadenas alimentarias, existe hoy por hoy un desacuerdo a la hora de aplicar la metodología de ACV en productos alimentarios.

1.4.1. Selección de la unidad funcional

La unidad funcional se define como la unidad objetivo del estudio, y cuantifica el servicio ofrecido por el sistema de producto. La unidad funcional proporciona, a su vez una referencia estándar para las entradas y salidas del inventario del sistema. La principal preocupación en la industria alimentaria es la selección de unidades funcionales que reflejen no solo el objetivo de un estudio en concreto, también el papel del producto en particular dentro de una dieta (Cederberg y Mattsson, 2000).

De esta forma, la unidad funcional de un producto puede seleccionarse en base a la masa del producto (kg de alimento, kg de producto comestible, etc.), al volumen o a los aspectos nutricionales (aporte energético, aporte vitamínico y proteico, etc.) (Tyszler et al., 2004).

1.4.2. Límites del sistema

También existe controversia a la hora de fijar los límites del sistema. El fin de vida de la mayoría de los alimentos es la propia ingesta, por lo que para tener en cuenta la disposición final del producto deberían modelizarse los procesos digestivos. No obstante, debido a que existen muchos factores que influyen en la funcionalidad de estos procesos, son difíciles de caracterizar (Jungbluth, 1999; Muñoz et al., 2008). La mayoría de los estudios están enfocados en los productos una vez fabricados, sin tener en cuenta el consumo de los mismos, ya que los procesos de compra y preparación de alimentos son muy diversos y difíciles de predecir (Pardo et al., 2012; Jungbluth, 1999).

1.4.3. Escala temporal

La selección de los límites temporales del estudio también es un tema delicado en los ACV alimentarios, sobre todo en aquellos basados en el aprovisionamiento de materias primas salvajes o dependientes de factores no controlados por el ser humano. Así los stocks pesqueros varían considerablemente de un año a otro, por lo que se recomienda obtener medias de impacto de al menos 3 años para poder ofrecer un resultado robusto (Ramos et al., 2011; Vazquez-Rowe et al 2012b). Para los procesos agrícolas también dependientes de las condiciones climáticas es también necesario obtener datos de más de un año ya que la intensidad de las labores, del riego puede variar considerablemente de un año para otro haciendo que el valor del impacto de un año en concreto no se representativo para el cultivo en cuestión.

1.4.4. Asignación de cargas

Otro de los aspectos con mayor controversia a la hora de realizar un ACV de un producto alimentario es debido a que en la mayoría de los sistemas no hay un solo producto como salida del sistema, por lo que resulta necesario asignar las cargas ambientales acumuladas a los diferentes productos de salida. Así, en el caso de los productos cárnicos por ejemplo, parte del ganado (los terneros) se cría sólo para producir carne, mientras que otra parte es destinada a producir leche cruda (vacas lecheras). Por lo tanto, puede haber dificultades a la hora de asignar los impactos ambientales entre la carne, los productos lácteos y otros subproductos como el cuero. La norma ISO 14044:2006 proporciona diferentes opciones para esta tarea, tales como la expansión del sistema, la asignación de masas o la asignación económica.

1.4.5. Selección de metodologías de impacto

A día de hoy la estandarización y homogenización de los modelos de categorías de impactos es un tema de alta prioridad tanto para las instituciones como para el mundo científico. Así, desde la mesa redonda para el consumo y producción sostenible de alimentos de la Comisión Europea se ha publicado la metodología ENVIFOOD (Food SCP RT (2013), basada en el set de indicadores publicados por el JRC (EU, 2013). Sin embargo existen todavía a día de hoy una serie de impactos que no están contemplados en ninguna metodología de ACV. Estos son, entre otros pérdida de biodiversidad (Curran, 2010), descartes (Vazquez-Rowe et al., 2012a), daños en el suelo marino (Ziegler y Valentinsson, 2008).

Respecto a las metodologías de impacto, la falta de regionalización de los factores de caracterización de algunas de las categorías de impacto resulta un inconveniente añadido (Manneh et al. 2010; Gotway y Young 2002; Perveen y James 2009). Categorías de impacto como agotamiento de agua, eutrofización, uso de suelo o toxicidad están muy influencias por el lugar donde se produce el impacto. De esta forma, no es lo mismo consumir un litro de agua en un una cuenca con abundancia de agua, que consumirla en una zona árida. Lo mismo ocurre por ejemplo con las emisiones de nitrógeno y la eutrofización. Así es necesario también adaptar las metodologías a las condiciones de cada zona o región.

1.4.6. Aplicabilidad a una escala real

Otro de los retos más importantes del ACV es la interpretación de los resultados y su aplicabilidad en una escala real. Actualmente, muchas empresas están llevando a cabo acciones de mejora de proceso o producto que podrían traducirse en un menor impacto ambiental por unidad de producto, sin embargo no realizan evaluaciones ambientales ni comunican la mejora al consumidor final. Así, el ACV es aún hoy una metodología complicada alejada de las prioridades de las empresas, debido sobre todo a la dificultad de realizar evaluaciones sencillas, y a la falta de oportunidades para comunicar de forma estandarizada los potenciales impactos ambientales asociados a un producto.

1.5. Ecodiseño de alimentos

La mejora ambiental enfocada a producto aporta una visión de mercado y ambiental al mismo tiempo ya que la función de la cadena alimentaria es proveer de productos alimentarios que sean comercializados, vendidos y consumidos. Con base en el ACV, el ecodiseño es hoy por hoy, una herramienta consistente para disminuir los impactos ambientales asociados a los productos manufacturados. El ecodiseño se refiere a la introducción de argumentos ambientales en el momento en que se toman las decisiones, además de los aspectos que históricamente han sido cosideradas como la estética o el coste y su principal objetivo es obtener productos sostenibles (Figura 5).

Figura 6. Principales aspectos a tener en cuenta en el ecodiseño de alimentos.

Frente a los problemas ambientales asociados con la producción y el consumo de alimentos, debido al aumento de la conciencia y de la presión de la opinión pública en relación con el deterioro del medio ambiente natural, las instituciones públicas, como la

Unión Europea, por medio de la Recomendación de la Comision de 9 de abril de 2013 sobre el uso de métodos comunes para medir y comunicar el comportamiento ambiental de los productos y las organizaciones a lo largo de su ciclo de vida, han incorporado estrategias de protección del medio ambiente de productos.

De esta manera, el trabajo comienza con la visión de mejorar el producto, a partir de los materiales utilizados en el proceso, a través de la gestión de residuos con el resultado final de reducir el consumo de energía y reducción de las emisiones (Zufia et al., 2009, IHOBE, 2000).

En el camino hacia el desarrollo sostenible hay diferentes etapas que ayudan a reducir el impacto medioambiental de los productos, desde los actos individuales, tales como la reducción del consumo energético durante un proceso determinado, a las acciones globales de prevención que persiguen la integración de los aspectos ambientales, sociales y empresariales (Rieradevall y Vinyets, 1999).

Como tal, ecodiseño se convierte en el eslabón clave hacia la sostenibilidad y el consumo responsable incorporando nuevos conceptos como la visión del sistema de producto, el concepto de ciclo de vida y la integración de todos los elementos que intervienen en la mejora de los aspectos ambientales del producto (Fiksel, 1997; Rieradevall y Vinyets, 2000).

El uso del ecodiseño como un sistema de gestión de producto ofrece interesantes beneficios a las empresas, tales como la reducción de los impactos ambientales, reducción de costes, la innovación y el cumplimiento de la legislación ambiental, el cumplimiento de las demandas de los clientes, el aumento de la calidad del producto, y la mejora de la imagen del producto y de la empresa (Gurauskienė y Varžinskas, 2006).

Todos estos beneficios están asociados con la realización de determinadas estrategias y acciones durante el proceso de diseño del producto. Las estrategias, entre otros, pueden estar centradas en los siguientes aspectos:

- Desarrollar productos alimenticios más eficientes que generen menos residuos durante todo el ciclo de vida.
- Reducir los contaminantes del producto y la generación de residuos a lo largo del ciclo de vida, ampliando su vida útil con la posibilidad de reutilizar o reciclar los envases de productos alimenticios.
- Optimizar todas las etapas de la cadena alimentaria para reducir el material, consumo de agua y energía, así como que los residuos asociados a ineficiencias.

- Determinar y superar los posibles problemas entre la reducción del impacto ambiental del diseño de un producto y la preservación de la seguridad y calidad de los alimentos.

Mediante la aplicación de las estrategias de ecodiseño, las empresas pueden generar nuevos productos, denominados ecoproductos, que junto con una reducción en el impacto ambiental global, crean más beneficios para la empresa.

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Capítulo 2 | HIPÓTESIS Y OBJETIVOS

El interés de esta tesis radica en superar parte de los retos asociados a la utilización del Análisis de Ciclo de Vida (ACV) para realizar una evaluación ambiental de los en productos alimentarios. Así, la hipótesis con la que se presenta esta tesis es la siguiente:

"Superar los retos de aplicación del Análisis de Ciclo de Vida como metodología para evaluar y reducir el potencial impacto ambiental asociado a la producción y comercialización de los productos alimentarios de una forma rigurosa y representativa"

Los retos de la aplicación del ACV en productos alimentarios pueden ser varios como ya se ha comentado en el apartado 1.4. Sin embargo esta tesis pretende dar respuesta a los siguientes:

¿Cómo afecta la variabilidad temporal a los impactos ocasionados por la producción primaria?

¿Qué métodos de caracterización de impactos son los más idóneos para medir el impacto ambiental de los productos alimentarios?ñ

¿Cómo se puede facilitar la aplicación del ACV por parte de las empresas en un entorno real?

2.1. Objetivos

Alrededor de estas preguntas se enmarcan los siguientes objetivos parciales:

<u>¿Cómo afecta la variabilidad temporal a los impactos ocasionados por la producción primaria?</u>

1. Demostrar la variabilidad temporal no controlada en la producción primaria: En las contribuciones I y II se presenta la variabilidad asociada a las variaciones de los stocks de caballa y merluza, mientras que en el artículo III se ha demostrado la variabilidad en el cultivo de guisantes.
2. Probar la viabilidad de utilizar la metodología de ACV + DEA con el fin de reducir variabilidad temporal: Se ha probado la validez de esta tecnología en la pesca de cerco de verdel (contribución I) y la pesca de arrastre de merluza (contribución II)

<u>¿Cómo medir el impacto ambiental de los productos alimentarios de una forma robusta y representativa?</u>

3. Determinar las categorías de impacto más relevantes para evaluar el impacto ambiental de los productos alimentarios: En la contribución IV se han seleccionado

las metodologías de caracterización de impactos asociadas a los aspectos más significativos del sector alimentario.

4. Establecer la regionalización de impactos: En la contribución V se evalua la importancia de incluir factores de caracterización regionalizados para evaluar el impacto en el estrés hídrico de forma precisa.

¿Cómo se puede facilitar a las empresas la evaluación ambiental?

5. Establecer los flujos de inventario representativos para cada etapa de la cadena alimentaria: En la contribución IV se han seleccionado una serie de flujos de inventario con el fin de seleccionar los aspectos más relevantes de las cadenas alimentarias y poder facilitar la recogida de datos representativos.
6. Desarrollar una herramienta sencilla de evaluación ambiental: En la contribución V se describe la herramienta SENSE-tool desarrollada con el fin de facilitar a las PYMEs la evaluación ambiental de sus productos alimentarios
7. Por último en la contribución VI se busca validar y establecer una metodología de Ecodiseño específica para productos alimentarios para reducir de forma significativa los impactos asociados al producto a lo largo de su ciclo de vida.

2.2. Plan de trabajo

Las contribuciones presentadas dan respuestas a los retos que se presentan a la hora de llevar a cabo el ACV en los productos alimentarios (Figura 7).

Fases según ISO 14040	Retos	Contribución Tesis
Inventario de Ciclo de vida	Variabilidad temporal no controlada	I, II, III
	Representatividad de datos de inventario	V
Evaluación de impacto	Metodologías de impacto representativas del sector alimentario	V
	Regionalización de factores de impacto ambientales	IV y V
Interpretación de resultados	Facilitar la aplicación real en empresas	V
	Proceso de ecodiseño adaptado a alimentos	IV

Figura 7. Plan de trabajo de la tesis con el encaje de las contribuciones.

2.3. Contribuciones

I **Ramos S**, Vázquez-Rowe I, Artetxe I, Moreira MT, Feijoo G, Zufia, J. 2011. Environmental assessment of Atlantic mackerel (Scomber scombrus) in the Basque Country. Increasing the time line delimitation in fishery LCA studies. *International Journal of Life Cycle Assessment* 16 (7), 599-610

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VI **Ramos S,** Pardo G, Zufia J. *Under revision.* Environmental improvement of a chicken product through life cycle assessment methodology. *Journal of Cleaner Production - Under revision*

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Capítulo 3 | METODOLOGÍA

Tal y como se ha descrito anteriormente en la sección 1.3, el Análisis de Ciclo de Vida es una metodología ampliamente aceptada para medir el impacto ambiental de los productos alimentarios. En este apartado y siguiendo la metodología definida por la ISO 14040:2006, se describe la sistemática seguida para la consecución de las contribuciones anteriormente citadas: representatividad de los productos objeto de estudio, el inventario de ciclo de vida, la evaluación de impactos y el uso o aplicación del ACV.

3.1. Productos de estudio

Esta tesis tiene como objetivo principal evaluar la utilidad del ACV en los productos alimentarios. Por ello, dada la propia variabilidad de la industria alimentaria, se han seleccionado alimentos que cubran parte de la diversidad de la actual industria agroalimentaria.

Las **contribuciones I y II** referentes a productos pesqueros se han enfocado a especies como el verdel y la merluza por su significancia en el mercado de la Comunidad Autónoma del País Vasco (CAPV) (Tabla 6). Desde un punto de vista de cantidad de desembarcos, el verdel supone alrededor de la mitad de las capturas (49 %), por lo que, aunque no tiene un gran valor económico unitario, es una especie con mucha importancia para la industria pesquera.

La merluza es, tras el bonito, la especie que más valor aporta a las pesquerías vascas, con un 22 % del valor total obtenido. Además, la merluza es una especie con una gran relevancia a nivel cultural, y con mucha proyección de futuro en la industria alimentaria. Es por ello que se ha seleccionado esta especie.

Tabla 6. Principales especies pescadas en la CAPV clasificadas por valor económico y cantidad (EUSTAT, 2014).

	Desembarcos		Valor	
	tn	**%**	**€**	**%**
Atún	1.254	2,3	5.035.287	7,77
Bonito	6.005	11,31	19.419.177	29,96
Chicharro	4.570	8,60	2.471.411	3,81
Merluza	3.780	7,12	13.747.144	21,21
Sardina	6.096	11,48	1.900.575	2,93
Verdel	26.003	48,98	8.686.531	13,40
Otros	5.371	10,11	13.567.037	20,93
TOTAL	53.081	100	64.827.164	100

Con la **contribución III** se ha realizado una evaluación del impacto de los guisantes congelados. Este estudio llevado a cabo junto con el centro sueco SP (anterior SIK). A nivel mundial se producen anualmente alrededor de 17 Mt (FAOSTAT, 2013), siendo Europa el segundo productor (6 %) detrás de Asia (86 %). Un alto porcentaje de estos cultivos son destinados a alimentación animal ya que presentan alto valor nutritivo debido sobre todo a la cantidad de proteínas, entre el 22 y 24 %. Sin embargo es un alimento muy demandado para consumo humano. Pese a no ser producido en la CAPV es ampliamente utilizado tanto por los consumidores vascos, como por la industria de platos preparados.

La **contribución IV** se centra en la cadena láctea. Este estudio cubre tanto la etapa ganadera como la transformación de la leche cruda en diferentes productos (nata, queso, mantequilla, yogures y leche en polvo). En España, se consumieron alrededor de 94 kg de leche per cápita en 2011, ligeramente superior a la media europea de 88 kg de leche per cápita (FAOSTAT, 2011). Los productos lácteos son alimentos básicos para la población, y como hemos visto en el capítulo anterior, con un gran potencial de impacto ambiental, por lo que el estudio del impacto ambiental se presenta necesario.

Por otro lado, para validar el sistema de evaluación ambiental armonizado que se presenta en la **contribución V,** se han seleccionado las cadenas alimentarias de los sectores zumos de frutas, acuicultura y cárnico y lácteo. Estas cadenas se han seleccionado en función de la importancia de las mismas en el mercado europeo, la tendencia en aumento de la producción de los productos, los problemas ambientales asociados a las etapas y debido a que cubren las diferentes etapas productivas asociadas a las cadenas alimentarias.

Por último, la **contribución VI** se ha centrado en un producto ampliamente utilizado por los consumidores españoles como es la bandeja de pechuga de pollo.

3.2. Inventario de ciclo de vida

En esta tesis se ha tratado de solventar los problemas asociados a la obtención de datos de inventario precisos. Para la mayoría de los estudios de análisis de ciclo de vida de productos alimentarios se utilizan bases de datos comerciales, como Ecoinvent, o públicas, como la desarrollada por la unión europea ELCD ampliamente aceptadas a nivel científico. Sin embargo, en muchas ocasiones carecen de datos adaptados a las diferentes casuísticas, regiones o niveles tecnológicos, por se puede incrementar la incertidumbre de los resultados (Sanjuan et al, 2014). Con el fin de evitar o reducir al mínimo la incertidumbre originada por la falta de datos directos en las **contribuciones I, II, III, V y VI** se ha invertido un esfuerzo extra en obtener la mayor parte de datos directos. Estos datos en la mayoría de las ocasiones se han obtenido a partir de cuestionarios directos a las empresas, datos propios de la experiencia de AZTI en estudios previos de investigación, directamente de otros de los centros en colaboración o de conversaciones directas con los responsables de las empresas implicadas

Además, muchos de los impactos están directamente relacionados con datos procedentes de las emisiones de substancias al medio como las emisiones compuestos nitrogenados asociadas a la aplicación de fertilizantes, sustancias tóxicas debido a la aplicación de pesticidas o las emisiones de GEI debidas a la fermentación entérica del ganado. Con el fin de adaptar las cantidades de fertilizantes y emisiones de GEI del ganado, en las **contribuciones III, V y VI** se han utilizados los modelos expuestos por el EcoInvent y IPCC respectivamente (Nemecek y Kägi, 2007; Dong et al., 2006). Para estimar el nivel de emisiones de los pesticidas, en la **contribución III** se ha utilizado la metodología PestLCI desarrollada por Stenemo et al. (2005).

3.2.1. Set de indicadores para la industria alimentaria

A lo largo de los anteriores estudios se ha detectado que para cada tipo de alimento existen una serie de flujos de inventario que tienen una mayor influencia en los resultados de impacto ambiental. Con el fin de asegurar una representatividad del inventario, pero a su vez evitando interferir con la actividad empresarial con una recolección de datos tediosa, en la **contribución IV** se ha desarrollado una lista de los flujos de inventario más relevantes para cada etapa de una cadena alimentaria tipo.

Para realizar esta selección primero se han seleccionado 3 cadenas de alimentos tipo representativas de la diversidad alimentaria existente en la Unión Europea, como son el sector de zumos de frutas, acuicultura y el cárnico y lácteo. Para cada cadena seleccionada se ha realizado un ACV exhaustivo desde la producción de las materias primas (frutas, piensos de acuicultura y piensos ganaderos) hasta la puerta del supermercado. Los datos de inventario de estas cadenas se han obtenido directamente de dos empresas de cada sector.

Una vez realizado el ACV se han seleccionado para cada etapa de la cadena alimentaria aquellos datos de inventario más representativos para las metodologías de impacto ambiental. Las etapas definidas han sido agricultura, pesca, ganadería y acuicultura como sector primario, transformación y transporte y para cada una de ellas se han definido un máximo de 10 flujos o datos de inventario.

3.2.2. Metodología combinada ACV+DEA para reducir variabilidad temporal

En la **contribución I** se ha identificado como un reto por parte de la cadena alimentaria el hacer frente a la variabilidad en los resultados debido a la variabilidad temporal no controlada por la tecnología.

Con el fin de reducir la incertidumbre en los resultados procedente de la variabilidad temporal, en la **contribución II** se ha utilizado la metodología combinada de ACV+DEA desarrollada por Vazque-Rowe et al. (2010).

El Análisis Envolvente de Datos (DEA, por sus siglas en inglés *Data Envelopment Analysis*,) es una metodología de programación lineal que proporciona una comparativa de la eficiencia empírica de múltiples unidades similares (Cooper et al., 2007). Esta metodología se ha aplicado al sector primario como una metodología independiente en varios estudios (Kao et al., 1993; Idda et al., 2009; Griffin y Woodward, 2011; Picazo-Tadeo et al., 2011). De hecho, gracias a su capacidad para discriminar entre diferentes unidades dentro de múltiples conjuntos de datos, se ha utilizado en el sector pesquero para analizar la eficiencia técnica de una amplia gama de flotas pesqueras (Tingley et al., 2003; 2005; Färe et al, 2006; Maravelias y Tsitsika, 2008), como una herramienta para evaluar la capacidad de las flotas pesqueras europeas, con el fin de proponer estrategias para mejorar y homogeneizar la eficiencia (Herrero y Pascoe, 2003; Griffin y Woodward, 2011).

Sin embargo, el uso de DEA junto con el ACV tiene como objetivo relacionar los impactos ambientales con la evaluación operativa, como una manera de verificar de la ecoeficiencia a través de la optimización de las entradas y salidas (Lozano et al., 2009). Dicho esto, hay que señalar que los resultados de la optimización son solamente teóricos y no sugieren acciones de mejora concretas para alcanzar los estándares de eficiencia ecológica. Además, esta metodología proporciona también una alternativa al uso de datos promedio en sistemas de unidades múltiples, reduciendo las fuentes comunes de la incertidumbre en los estudios de ACV (Vázquez-Rowe et al., 2010).

A pesar de que este método ha sido desarrollado muy recientemente, se ha aplicado con éxito a una amplia gama de sistemas de producción en el sector primario, incluyendo la pesca o la viticultura (Iribarren et al, 2011; Vázquez-Rowe et al. 2011; 2012). Sin embargo, una serie de potencialidades no utilizadas de esta metodología permanecen aún sin explorar (Iribarren et al., 2010). Uno de ellos es el uso de una opción para DEA, el *window analysis*, para determinar la eficiencia de impacto ambiental de sistemas de producción sobre una base temporal (Charnes et al., 1985).

En la **contribución II** se ha utilizado esta metodología combinada para evaluar la ecoeficiencia de la pesquería de arrastre de Bizkaia. Existen dos variantes en la metodología combina de ACV+DEA: el método de 5 pasos y el método de 3 pasos. El método de 5 pasos se ha utilizado principalmente para la verificación de la ecoeficiencia y determinar las consecuencias sobre los impactos ambientales de las ineficiencias operativas (Vázquez-Rowe et al., 2010). El método de 3 pasos busca principalmente la estimación de la eficiencia de impacto ambiental mientras se realiza una evaluación comparativa simultánea de un conjunto de parámetros operacionales y ambientales (Iribarren et al., 2011).

En la **contribución II** referente a la variación en la pesca de merluza por parte de la flota vasca de arrastre, se ha seleccionado la metodología de tres pasos, utilizando un método de análisis temporal *window analysis* para el DEA (Lozano et al., 2010):

Paso 1: Datos de inventarios: Los datos de inventario para la flota de arrastre vasca se obtuvieron un total de 7 buques, pertenecientes a los puertos de Pasaia y Ondarroa para un periodo de 6 años comprendido 2001 y 2006.

Paso 2: Caracterización ambiental de cada buque: Para cada buque o Unidad de Decisión se ha realizado una evaluación ambiental mediante el Análisis de Ciclo de Vida para las categorías de impacto más representativas del sector pesquero.

Paso 3: Análisis DEA: La matriz de la DEA en el método de 3 pasos se realiza con las entradas y salidas de inventario más relevantes (consumo de diésel o cantidad de desembarcos) conjuntamente con los resultados de las 4 categorías de impacto ambiental evaluados en el Paso 2. La matriz se realizó utilizando dos longitudes de ventana diferentes: 1 y 6. La matriz de longitud 1 se ha utilizado para comparar los resultados de la eficiencia anual entre buques individuales y sus fluctuaciones interanuales. Por otro lado, la longitud de ventana de 6 ha hecho posible la comparación de todo el período de seis años bajo el mismo conjunto de referencia.

3.3. Evaluación de impactos

Como se ha mencionado en la sección 1.4.5 otro aspecto importante del ACV es la selección de las categorías de impacto ambiental más representativas de un producto o servicio. Desde la Comisión Europea recientemente se ha publicado el manual para estandarizar y regular los análisis de ciclo de vida (JRC, 2013), dentro del cual se recomienda utilizar para cualquier tipo de producto un total de 14 diferentes categorías de impacto. Sin embargo, es necesario seleccionar qué categorías y métodos se adaptan mejor a la realidad ambiental de las cadenas alimentarias. Para ello, esta tesis se ha centrado en seleccionar aquellas categorías y modelos de impacto ambiental más representativos de los productos alimentarios, así como en la regionalización de los factores de impacto para aquellas categorías sensibles a las condiciones del medio o lugar donde se realiza la actividad.

3.3.1. Selección de categorías y modelos de impacto ambiental

Con el fin de obtener una interpretación viable del impacto ambiental asociado a un producto, es necesario reducir a un número manejable el número de categorías de impacto ambiental, seleccionando así aquellas que sean representativas para el sector alimentario. Por ello en la **contribución IV**, y dentro del proyecto europeo SENSE (FP7 288974), se ha realizado una selección de las categorías de impacto más relevantes para el sector alimentario europeo.

Las categorías se han seleccionado con el fin de evaluar los aspectos más significativos de la incidencia de las diferentes cadenas alimentarias. Para ello, primero se ha realizado una análisis exhaustivo de los aspectos ambientales más significativos de cada una de las etapas de las tres cadenas o productos estudiados: zumo de frutas, acuicultura y cárnico/lácteo. Con estas cadenas se ha cubierto la problemática asociadas a la agricultura, pesca, ganadería y acuicultura, a la transformación y envasado y a la distribución. Una vez seleccionados los aspectos ambientales más significativos se ha procedido a seleccionar aquellas categorías de impacto que reflejen mejor la realidad ambiental de los alimentos seleccionados. Por ejemplo: siendo la emisión de substancias nitrogenadas uno de los aspectos más relevantes para los procesos agrícolas, se ha seleccionado la eutrofización terrestre y acuática como una de las categorías más relevantes.

Por último, en base a los últimos estudios realizados en materia de ACV y en base a la propuesta del JRC (2013) se han seleccionado los modelos de análisis de impactos más apropiados para evaluar cada impacto ambiental asociado a los productos alimentarios (**contribución IV**).

3.3.2. Regionalización de factores de impacto

Para transformar un valor de inventario en las unidades de impacto ambiental se emplean los factores de caracterización (FC) desarrollados por las diferentes metodologías. Sin embargo, en la mayoría de las ocasiones estos factores son de carácter global y no se adaptan a las condiciones o realidades de cada medio. El desarrollo de FC ha sido determinado históricamente por la falta de información espacial y temporal en la recogida de datos de inventario. Estos FC genéricos están bien adaptados para evaluar los impactos de carácter global, como el cambio climático o el agotamiento de la capa de ozono, pero tienen algunas limitaciones inherentes a la hora de evaluar las categorías de impacto que no son de naturaleza global, como la acidificación o la eutrofización o que son categorías de impacto típicamente regionales a nivel territorial. Los impactos de uso de suelo, agotamiento de agua, o de recursos tienen un alcance aún más local, mientras que la toxicidad y la ecotoxicidad pueden variar desde muy local a los impactos globales, dependiendo de las sustancias (Sedlbauer et al, 2007).

Por todo, con el fin de reducir la incertidumbre en los resultados debido a las diferencias de los potenciales de impacto en las diferentes regiones, en la **contribución IV** se han adaptados los FC de las categorías de impacto de acidificación, eutrofización y agotamiento de agua a nivel de país. Estos factores se han seleccionado de los estudios de Posch et al., 2008, para el caso de la acidificación y eutrofización, y del estudio de Frischknecht et al. 2009 para el potencial impacto de agotamiento de agua.

Puesto que la industria alimentaria, y sobre todo la agricultura suponen una parte muy importante del consumo de agua mundial, se ha detectado la necesidad de aplicar métodos de impacto que reflejen el potencial de impacto debido al uso del agua. En la **contribución V** se ha seleccionado el nuevo método desarrollado por el WULCA y en el que la Unión Europea está trabajando activamente y se ha comparado con 2 de los métodos más utilizados para la evaluación del impacto del uso del agua para una cadena de producción láctea. El trabajo no ha pretendido realizar un análisis exhaustivo del ajuste del nuevo método a la realidad hidrológica del país, sino que ha tratado de realizar un análisis sobre la utilidad de los resultados para alcanzar conclusiones y potenciar mejoras en el sector de la alimentación. A continuación se detalla cada uno de los métodos seleccionados:

<u>1. Método AWaRe</u>: Desarrollado por el WULCA, 2015, este grupo ha creado un índice (escala del 1 a 1000) basado en la demanda y la disponibilidad de agua de las diferentes regiones del planeta (Boulay et al, 2015). La metodología tiene una serie de asunciones:

- considera el consumo directo de agua, sin contabilizar la evapotranspiración.
- la escala temporal seleccionada debe ser mensual.
- la escala geográfica seleccionada es por cuenca. Sin embargo, de no haber datos disponibles de los lugares específicos involucrados en todas las etapas de la cadena del producto objeto de estudio, también establece factores por países.

- Existen diferentes índices para el consumo de agua de la agricultura y el industrial o doméstico. Esto es debido al supuesto de que en la agricultura el consumo se localiza en los meses de menos precipitación, mientras que en la industria o a nivel doméstico el consumo se mantiene constante a lo largo del año.

2. Nuevo método WSI revisado: Desarrollado por Ridoutt y Pfister (2010), es el método recomendado por el protocolo ENVIFOOD (Food SCP Rt, 2013) de la Unión Europea. El método utiliza los índices de disponibilidad de agua desarrollados por Pfister et al (2009), pero los normaliza en función de un factor global de estrés hídrico (1,6 m^3 H2O eq.). Los resultados se expresan en unidades de H2O equivalentes respecto a la situación media de estrés hídrico (WSI) global.

3. Método nuevo $WSI_{HH, EQ}$: En 2014, Ridoutt y Pfister presentaron en el congreso LCA agri-food de Asia (Bangkok) un nuevo método para medir el impacto del consumo de agua en la salud humana y los ecosistemas. Teniendo en cuenta que una de las preocupaciones a la hora de interpretar los resultados de impacto por el consumo de agua es la falta de coherencia entre los modelos de endpoint o de punto final y los de midpoint o de estrés hídrico de las cuencas, en ese estudio se presentó un nuevo método de cálculo regionalizado que integra los impactos del propio estrés hídrico de la cuenca con el daño causado en los humanos y los ecosistemas.

3.4. Aplicación y uso del Análisis de Ciclo de Vida

Las claves para una aplicación e interpretación efectiva de los resultados de evaluación de impactos ambientales pasan desde la selección efectiva del objeto o producto de estudio hasta la facilitación de la interpretación de resultados.

3.4.1. Sistema integral para la evaluación de impactos ambientales

Con el fin de facilitar a las empresas el uso e implantación de evaluaciones ambientales de producto, en el marco de la tesis y dentro del marco del proyecto SENSE (FP7 288974) *Harmonised Environmental Sustainability in the European food and drink chain,* se ha desarrollado la herramienta online SENSE-Tool que ofrece la posibilidad de evaluar de forma sencilla pero robusta el impacto ambiental de los productos alimentarios (**contribución IV**).

El concepto de la herramienta se ha estructurado en cuatro secciones

- PERFIL: La primera sección se ha organizado con el fin de captar la información necesaria para generar un perfil de las empresas que van a realizar el análisis ambiental. Los apartados que se han seleccionado son los siguientes: información general sobre la empresa (nombre, CIF, etc.), el lugar de producción y el portfolio de los productos a los que se les vaya a realizar la evaluación ambiental.

- DIAGRAMA DE FLUJO: El segundo apartado se ha constituido con el fin de obtener el diagrama de flujo general del producto. Se han seleccionado las 6 etapas más relevantes de los sistemas alimentarios: agricultura, pesca, ganadería, acuicultura, transformación y transporte y se ha desarrollado la posibilidad de realizar el diagrama de flujo en base a estos módulos y a diferentes niveles.
- INVENTARIO DE CICLO DE VIDA: Una vez definido el diagrama el siguiente módulo se ha basado en la captación de datos de inventario para cada una de las etapas. Se ha desarrollado un módulo online basado en los datos de inventario seleccionados (apartado 3.2.1). Con el fin de facilitar la recogida de datos de proveedores e incluir todo el ciclo de vida del producto, se ha desarrollado un módulo donde las empresas pueden enviar invitaciones *online* a los proveedores.
- RESULTADOS o CARACTERIZACIÓN DE IMPACTOS: En base a las metodologías de impacto definidas en la **contribución IV** (sección 3.3.1) se han desarrollado unos algoritmos para calcular los impactos finales asociados a la unidad de producto. Estos algoritmos tienen en cuenta los flujos entre productos intermedios, así como los factores de caracterización de impactos finales. El programa realiza una suma del impacto ambiental proporcional de cada proceso involucrado en la cadena alimentaria.

Para facilitar la explotación de resultados se ha diseñado un apartado de comunicación de resultados donde la empresa puede obtener un documento que resuma los resultados para la caracterización de impactos.

3.4.2. Ecodiseño de alimentos

Con el fin de acercar la metodología de análisis de Ciclo de Vida a las industrias y consumidores, se ha desarrollado una nueva metodología de ecodiseño adaptada al sector alimentario. Para el desarrollo de la misma se han tenido en cuenta tanto aspectos de ACV e impacto ambiental, así como aspectos más puros referentes al diseño de alimentos como son la calidad nutricional, sensorial, económica o de conveniencia para el usuario. El objetivo final ha sido establecer una metodología para diseñar o rediseñar un producto alimentario con menor impacto ambiental.

En la metodología desarrollada (Figura 8) se presenta el esquema típico de diseño de alimentos donde se han incluido las acciones adicionales desarrolladas para incluir la evaluación ambiental en cada etapa del diseño de alimentos.

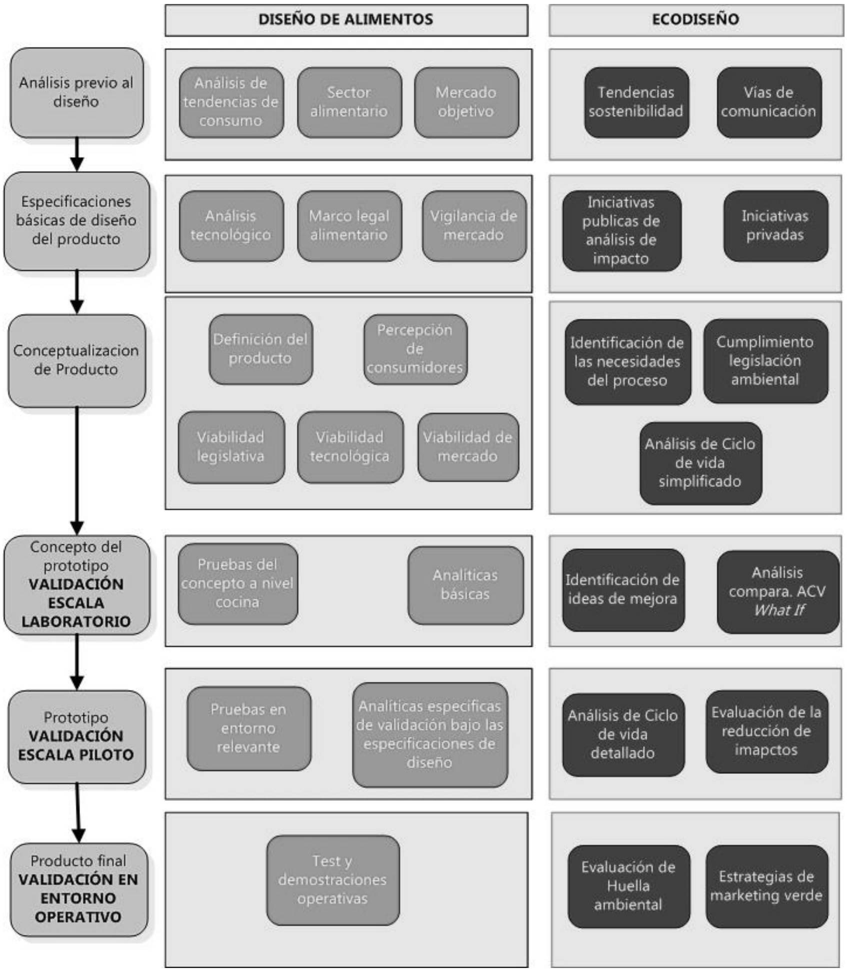

Figura 8. Metodología de ecodiseño de alimentos.

Así, en el análisis previo al diseño, donde se establece el primer concepto de producto y su función en el mercado, a parte de la vigilancia de tendencias y la definición del potencial mercado objetivo, también se ha incorporado la búsqueda de tendencias de sostenibilidad y vías de comunicación ambiental. En el siguiente paso, donde se definen las especificaciones básicas del producto, junto con el análisis tecnológico, la vigilancia de mercado y el marco legal, se ha planteado la realización de una identificación de requisitos en materia de normativa ambiental o iniciativas público-privadas. También incluye una vigilancia de mejores técnicas disponibles (MTD).

En la siguiente etapa de conceptualización del producto (donde se definen detalladamente cuales son las características del producto) se ha propuesto realizar un Análisis de Ciclo de Vida de producto. Para ello primero será necesario identificar las necesidades de agua, energía y materiales del proceso, así como los residuos y emisiones al aire y al agua que potencialmente se van a generar. Como resultado de este análisis se obtendrá una primera evaluación ambiental del producto definido.

Una vez definido el concepto, se realiza la primera validación a escala laboratorio. En esta etapa se realizan los primeros prototipos a nivel cocina y las analíticas básicas del producto. Para mejorar el desempeño ambiental del producto, en esta fase se propone realizar un brainstorming con el fin de identificar acciones específicas basadas en los resultados del análisis de ciclo de vida. Una vez identificadas las acciones más viables se ha propuesto realizar un análisis de escenarios, o análisis *what if,* para seleccionar aquellas acciones de mejora con mayor potencial de implantación. Si se observasen grandes desviaciones respecto a los valores objetivo, se propone revisar las formulaciones y procesos o rediseñar el producto si fuera necesario.

Tras la validación a escala laboratorio o cocina, se realizan las pruebas a escala piloto donde se comprueban las características del procesado. En esta etapa, se establece la realización un ACV completo y detallado del producto final. Los impactos ambientales a tener en cuenta se deciden en base a la bibliografía, reglamentación ambiental, iniciativas ambientales identificadas, etc. Se plantea estudiar el interés de incorporar al producto alguna de las etiquetas o certificaciones sobre impacto ambiental, como pueden ser la huella de carbono o huella hídrica. Se propone también la identificación de los requerimientos y metodologías de evaluación de impactos requeridas por estas certificaciones con el fin de facilitar los procesos de marketing del producto final.

Tras los primeros prototipos se realizará una prueba industrial en una fábrica, elaborando el alimento según la fórmula y el proceso aprobado a escala piloto. Se plantea que las empresas que realizan el esfuerzo de realizar un ecodiseño, tengan la posibilidad de utilizar dichas mejoras como argumentos de venta adicionales a los ya utilizados tradicionalmente (beneficios para la salud, sabor o comodidad).

En la **contribución VI** se valida y demuestra cómo mediante el ecodiseño de alimentos es posible reducir significativamente el impacto ambiental asociado a los productos alimentarios. El protocolo se ha validado en un producto de bandeja de pollo, y las mejoras ambientales se han enfocado a todo el ciclo de vida del producto, empezando por los piensos avícolas hasta el envase final del producto.

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Capítulo 4 | RESULTADOS

4.1. CONTRIBUCIÓN I

Environmental assessment of the Atlantic mackerel (*Scomber scombrus*) season in the Basque Country. Increasing the time line delimitation in fishery LCA studies.

Saioa Ramos[a], Ian Vázquez-Rowe[*b], Iñaki Artetxe[c], Mª Teresa Moreira[b], Gumersindo Feijoo[b] and Jaime Zufía[a].

[a] Food Research Division, AZTI-Tecnalia, 48160, Derio, Spain.

[b] Department of Chemical Engineering, School of Engineering, University of Santiago de Compostela. 15782 – Santiago de Compostela, Spain.

[c] Marine Research Division, AZTI-Tecnalia, 20110 Pasaia, Spain

*Corresponding author: Ian Vázquez-Rowe. Email: ian.vazquez@rai.usc.es

Telephone number: 0034-881816020. Fax number: 0034-881816702

Abstract

Goal, scope and background. The purpose of this study was to evaluate the environmental impacts linked to fish extraction on a temporal basis, in order to analyze the effect that stock abundance variations may have on reporting environmental burdens. Inventory data for the North-East Atlantic Mackerel (NEAM) fishing season were collected over an eight-year period and used to carry out a Life Cycle Assessment (LCA). The selected fishery corresponds to the Basque coastal purse seining fleet.

Methods. The functional unit (FU) was set as 1 ton of landed round fish in a Basque port during the NEAM fishing season for each of the selected years. The selected datafor the life cycle inventory were gathered from personal communication from ship-owners and from a fish first sale register in the Basque country. A series of fishery-specific impact categories and indicators were included in the evaluation together with conventional impact categories.

Results. Conventional LCA impact categories showed that the environmental impact is dominated by the energy use in the fishery, despite of the low fuel effort identified with respect to other purse seining fisheries. Nevertheless, strong differences were identified between annual environmental impacts, attributed mainly to remarkable variations in

NEAM stock abundance from one year to another, whereas the fishing effort remained quite stable throughout the assessed years. Fishery-specific categories, such as the discard rate or seafloor impact showed reduced impacts of this fishery respect to other small pelagic fish fisheries. Finally, the Fishery in Balance (FiB) Index identified the evolution of NEAM stock abundance for this particular fishery.

Conclusions. To our knowledge, this is the first fishery LCA study in which there is sufficient inventory data in order to conduct the methodology throughout a wide period of time. The outstanding variance in environmental impacts from one season to another evidences the need to expand fishery LCAs in time, in order to attain a more integrated perspective of the environmental performance of a certain fishery or species.

Recommendations. The extension of LCA inventories in the timeline may be an important improvement for activities that rely entirely on the extraction of organisms from wild ecosystems. For instance, future research will have to determine the importance of increasing the timeline in fishery LCAs for species that do not show large stock abundance variations through time, unlike NEAM.

Keywords: Atlantic mackerel; Life Cycle Assessment; purse seining; *Scomber scombrus*, timeline analysis.

4.1.1. Introduction

The Basque country has traditionally been an important fishing region at a European level since late medieval times, when whale fishing was an important industry and the first transatlantic vessels started commercializing cod fished in Newfoundland (Macías-Pereda and Muruaga 1992). Currently, the importance of the Basque fishing fleet has not diminished, but fleet characteristics and target species have shifted considerably due to the depletion or overexploitation of many traditional fisheries (Pauly et al. 2002; Worm and Myers 2004; SOFIA 2008).

The Basque fishing fleet is made up of a coastal fleet that targets small pelagic fish, a cod trawling fleet, an offshore trawling fleet that targets hake in the Northern (ICES Divisions VIIIabd and VII) and Southern stocks (ICES Divisions VIIIc) and finally a strong tuna industry made up of 24 vessels (Table 1). Despite of the strong reduction in the number of vessels, tonnage and overall landings in the past few years, the importance of this fleet is obvious since its current total gross tonnage (GT) is comparable to that of Denmark, Ireland or Germany (Murua et al. 2003; EUROSTAT, 2009).

Table 1. Number of vessels in the Basque fishing fleet (1992-2007). Source: EUSTAT 2010.

Fishing fleet	1992	1999	2007
Coastal fleet*	399	340	226
Offshore trawling	107	63	36
Freezer-trawlers	25	5	0
Deep-sea purse seiners	29	29	24
Cod freezer-trawlers	24	8	5
Total Basque fishing fleet	584	445	291

*The analyzed purse seiners in this article are included in *Coastal fleet* vessels.

The decrease in landings, however, has not only been affected by the reduction of the fishing fleet forced by the Common Fisheries Policy (CFP), but also by the increasing limits to the total allowable catch (TAC) for certain species such as European hake or by the closure of the anchovy fishery (2005-2009). Within this frame, North-East Atlantic mackerel –NEAM- (*Scomber scombrus* Linnaeus 1758), a pelagic shoaling species belonging to the Scombridae family that is widely distributed in European waters (Punzón et al. 2004; Uriarte et al. 2001), presents itself as the only major target fishing species that has not only maintained, but also increased its landings in recent years. The Basque coastal fleet mainly extracts this species in ICES Division VIIIc in the late winter and early spring months (Table 2), coinciding with the peak of spawning activity in the East and

Central Cantabrian Sea (Uriarte and Lucio 2001). Most of the vessels involved in the NEAM season are purse seiners, but other gears that occasionally target this species are handlines, trawls and, to a lesser extent, gillnets.

Table 2. Calendar of catches for the Basque coastal purse seining fleet.

Species	Jan.	Feb.	Mar.	Apr.	May	Jun.	Jul.	Aug.	Sep.	Oct.	Nov.	Dec.
Atl. Mack	■	■	■									
Anchovy				■	■							
Albacore						■	■	■	■	■		

Atlantic mackerel (*Scomber Scombrus*); Anchovy (*Engraulis encrasicholus*); Albacore (*Thunnus alalunga*).

NEAM is one of the main pelagic fish species commercialized in Spain. In 2009, 19,400 tons of this species were sold fresh across the nation, and 3,000 tons were sold canned (Martín-Cerdeño 2009; MARM 2009). Additionally, it is also used as bait in many long lining and trolling fisheries. NEAM landings in the Basque country represent approximately 15% of fresh NEAM commercialized in Spain (Mercados Municipales 2010). However, it is important to highlight that over 90% of NEAM landed in the Basque country corresponds to the February-April period, when together with the neighboring region of Cantabria, they represent 90% of national landings and sales (MERCASA 2010).

The apparent robustness of the NEAM fishery in ICES Division VIIIc is reflected by the increasing spawning stock biomass (SSB) pattern in recent years (ICES 2010) and the strong increase in landings, despite the notable reduction in the number of vessels (Figure 1). Nevertheless, the fishing fleet analyzed in this article is also threatened by overexploitation, mainly due to the failure to comply with the fixed TACs, and the high fishing mortality (F) of NEAM in the Northeastern Atlantic stock (Table 3), which is considered by ICES as a unique stock (ICES 2007a; 2007b). In fact, recent findings suggest that current stock abundance in the southern section (ICES Divisions VIIIc and IXa) may be linked to a variety of hydrographical factors, such as plankton abundance or temperature shifts (Reid 2001; Hannesson 2007). Therefore, the stock increase in this area would be linked to a changing spatial distribution of the species, rather than on a net improvement of the stock.

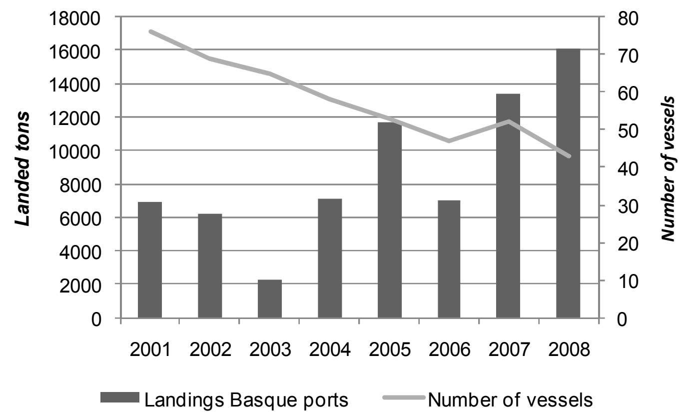


Figure 1. NEAM landings in the Basque Country in the 2001-2008 period respect to total ICES areas VIIIc and IXa NEAM landings. *Source*: ICES; AZTI.

Table 3. Stock assessment summary for the NEAM stock. NEAM catches and TACs for ICES Divs. VIIIc and IXa in the assessed period. *Source:* ICES.

Year	SSB*	F	Recrut*	NEAM landings Divs. VIIIc, IXa*	Total allowable catch (TAC)	% over catch respect to TAC
2001	2,138,374	0.40	4,853	43,198	40,180	7.5
2002	1,749,298	0.45	7,854	49,576	41,100	20.6
2003	1,748,701	0.44	3,475	25,823	35,000	-26.2
2004	1,848,672	0.40	4,437	34,840	32,310	7.8
2005	2,290,881	0.28	6,794	49,618	24,870	99.5
2006	2,409,602	0.23	6,915	52,751	26,180	101.5
2007	2,540,759	0.24	3,818	62,834	29,610	112.2
2008	2,709,395	0.23	4,507	59,859	27,010	121.6

*Landings reported in tons; SSB= Spawning Stock Biomass; F= Fish mortality.

In an attempt to identify and quantify environmental impacts linked to the industrial aspects of fish extraction, Life cycle assessment (LCA) appears as an internationally recognized methodology (Pelletier et al., 2007). Nevertheless, the importance of including innovative methodological improvements in LCA to broaden its scope and shift to a more comprehensive environmental analysis of fisheries is a major concern for LCA seafood practitioners. Consequently, in recent years there has been a series of publications that have proposed the inclusion of new impact categories in fishery LCA, such as seabed disturbance (Ziegler et al. 2003; Nilsson and Ziegler, 2007); biotic resources use (Papatryphon et al. 2004) or discard rate (Ziegler et al. 2009). However, to date, fishery LCA studies have been based on relatively short periods of time - in most cases one season or year – (Ziegler et al. 2003; Ziegler et al. 2009; Ramos et al. 2010; Vázquez-Rowe et al. 2010b; Vázquez-Rowe et al. 2011b), mainly due to the difficulty of obtaining thorough inventory data for a prolonged period of time (Weidema and Wesnaes 1996; Reap et al 2008). This situation has lead to LCA publications that have not taken into account the irregular cycles that fisheries may be subject to (Pet et al. 1997), especially for those fish species, mainly small pelagic fish (e.g. NEAM), that suffer natural interdecadal abundance fluctuations (Pauly et al. 2002; Fréon et al. 2008).

Therefore, in this study, the main objective is to analyze the NEAM season capture by Basque purse seining vessels during an extended period of time (2001-2008) with the aim of identifying possible environmental performance variations during the assessed period and including a series of fishery-specific categories or indicators that aid in the understanding of this fishery from an integrated perspective.

4.1.2. Materials and Methods

4.1.2.1. Goal and scope definition

The main goal of this LCA study is to assess the environmental impacts related to NEAM extraction by the Basque coastal purse seining fleet in the Gulf of Biscay (ICES Division VIIIc) on a temporal basis, in order to analyze how a long period of time may affect the environmental performance of fishing fleets that target species with strong annual stock abundance variation. Therefore, an 8 year period (2001-2008) was set as the time line in this particular study.

The functional unit (FU) that was selected for this particular research corresponded to 1 ton of landed round fish in a Basque port during the NEAM fishing season for each of the selected years. This FU is based on the assumption that the main objective of the study was to compare the environmental profile of one same seasonal fishery that was assessed for an 8 year period. The rationale behind using this FU rather than adopting a product perspective (i.e., exclusively NEAM landings) is linked to the fact that it is more realistic to assess a fishery in terms of the total catch and landings, rather than on the independent landing rates of the targeted species, especially when analyzing and discussing fishery-specific indicators or categories.

The selected fishing fleet was chosen based on the fact that it represented roughly 75% of annual landings of NEAM in Basque ports. The system under study was made up of the different operational stages performed by the assessed coastal purse seining vessels, including diesel consumption, anti-fouling, marine lubricant oil and trawl net use and ice consumption. The construction and maintenance of the vessels, as well as cooling agent emissions were also included. However, it is important to note that this study only focuses on the Atlantic mackerel fishing season performed by the selected vessels each year. Therefore, the inventory and the environmental impacts that will be associated to the life cycle inventory (LCI) will only correspond to the assigned resource use and related emissions for the seasonal period that corresponds to NEAM extraction, as will be discussed further on.

NEAM is the main target species, although a series of by-catch species, mainly European pilchard (*Sardina pilchardus* Walbaum 1792), are also landed (Table 4). These species were analyzed ranging from the production of the supply materials until landing operations for sale at Basque ports. Therefore, this assessment constituted a "cradle to gate" analysis (Guinée et al. 2001). The backup for this decision is the fact that on land seafood operations are not subject to the strong yearly fluctuations that are expected for fishery activities. However, it is important to note that landing operations included only take into account landing operations done on board, while on land operations at port were excluded (Vázquez-Rowe et al. 2010b).

Table 4. Selected vessel samples for the 2001-2008 period

	2001	2002	2003	2004	2005	2006	2007	2008
Sample size	35	28	30	31	41	27	45	35
% over total fleet	46.1	40.6	46.2	53.4	77.4	57.4	86.5	81.4
Average beam (m)	29.2	28.0	27.2	30.8	31.6	33.0	32.3	32.1
Average daily capture by vessel (tons)	5.77	5.94	2.04	6.76	11.16	10.02	17.82	22.13
Total NEAM landings (tons)	3,222	2,286	164	2,990	7,255	2,897	10,171	11,394
% over total NEAM landings	63.3	68.9	36.9	77.2	96.9	87.8	99.6	97.3
% of NEAM landings over total	79.8	85.8	33.4	83.9	93.3	89.3	97.6	98.1

4.1.2.2. Data acquisition

The samples used for this study corresponded to a set of purse seining vessels belonging to the Basque coastal fishing fleet obtained according to availability for the different years, as observed in Table 3. The primary data for fishing vessel operations were obtained mainly from a specific Basque register of fish at first sale provided by the Marine Research Division at AZTI. Landings, vessel characteristics (beam, GT, etc), fishing operations and fishing areas were the most relevant data obtained from the register. It is important to highlight that data from this register were obtained through a series of questionnaires filled out by AZTI observers, in direct collaboration with skippers from the most important purse seining ports in the Basque country. The response rate to these questionnaires can be observed in Table 4. A series of additional information, such as the number of seine nets used per vessel or the consumed ice were obtained through personal communication from Basque fishermen and skippers (J.A. Luzarraga, shipowner, personal communication, November, 2010). Cooling agent emissions were provided by AZTI's Marine Research Division (Aboitiz and Pereira 2009; Xabier Aboitiz 2011, personal communication). Finally, background data associated with the production of diesel, nets or anti-fouling and boat paint were obtained from the ecoinvent database (Frischknecht et al. 2007).

4.1.2.3. Life Cycle Inventory

The development of the life cycle inventory (LCI) involves the collection and computation of the data in order to quantify the relevant inputs and outputs in the production system, including resource usage and emissions related to the analyzed system (ISO 14040:2006). This phase is usually the most time consuming compared to other LCA phases, mainly due to the difficulty in collecting comprehensive data. Hence, this last issue is probably the main responsible for the scarce appearance of long period analysis in fisheries LCA, together with the fact that this methodology has only been implemented recently in this field. Therefore, data in this study were collected for an increased period of time, in order to achieve a reliable and representative picture of the environmental performance of the analyzed system.

A simplified inventory summary regarding the main inputs and outputs of the studied system is shown in Table 5, while additional data were given in Online Resource 1. Inventory data relating to NEAM landings were obtained from a range of 27 to 45 purse seiners depending on the assessed year, representing at least 40% of the total purse seining fleet (Table 3). Unfortunately, only 6 vessels were assessed for the entire period, due to vessel scrapping in order to meet CFP regulations and lack of data availability for certain years.

Discard amounts were not available for this fleet. Nevertheless, discussion on average ranges relating to discards in these types of fisheries was included. The seafloor impact potential (SIP) proposed by Nilsson and Ziegler (2007) was assumed to be minimal for the gear used by this particular fleet. Nevertheless, this issue will be analyzed in the discussion section.

Table 5. Inventory for fish landed in the NEAM season inBasque ports by coastal purse seiners in selected years of the 2001-2008 period (Data per FU: 1 ton of landed round fish during the NEAM season).

INPUTS				
From the technosphere				
Materials and fuels	**Units**	**2001**	**2004**	**2008**
Diesel	kg	31.53	34.63	14.62
Steel	kg	7.01	9.80	7.15
Anti-fouling	g	884	1,249	930.5
Boat paint	g	310	440	332.1
Marine lubricant oil	g	80.0	87.8	37.1
Ice	kg	125	125.2	122.6
Seine net (nylon+lead+cork)	kg	3.68	3.69	2.65
OUTPUTS				
To the technosphere				
Products	Units	2001	2004	2008
Total round fish	t	1	1	1
NEAM	t	0.798	0.839	0.981
Other pelagic fish	t	0.202	0.161	0.019
To the environment				
Emissions to the atmosphere				
1. CO_2	kg	100.0	109.8	46.33
2. SO_2	g	315	346	146
3. VOC	g	75.7	83.1	35.1
4. NO_x	kg	2.27	2.49	1.05
5. CO	g	233	256	108
6. R22	g	4.08	4.52	3.40
Emissions to the ocean				
1. Xylene	g	80.9	114.4	85.2
2. Dicopper oxides	g	183	259	193
3. Zinc oxides	g	82.8	117.1	87.2
4. Nylon	g	421	423	304
5. Lead	g	93.2	93.5	67.3

4.1.2.4. Allocation strategies and other assumptions

The purse seining fishing fleet under study presents three distinct fishing seasons, as mentioned in Table 2. The first two seasons of the year, the NEAM and the anchovy season, take place in the first half of the year, while the albacore fishing season takes place throughout the second half of the year. Therefore, temporal allocation for construction and maintenance materials in the LCI was performed by assigning half of the annual inputs/outputs to the albacore season, while the other half was assigned proportionally to the other two fishing seasons depending on their annual length. It is important to note that this procedure is influenced by the fact that the anchovy fishery has suffered strong restrictions or closure in recent years (ICES 2009). Additionally, the same procedure was implemented to allocate cooling agent emissions to the studied fishing season.

No further allocations were needed in the selected case study due to the characteristics of the chosen FU. In other words, the fact that NEAM and by-catch are analyzed globally in terms of total landings makes it possible to disregard other allocation procedures, such as mass or economic allocation (Ayer et al. 2007). The rationale for disregarding mass allocation, which would be appropriate due to the similar economic value of the by-catch (mainly sardine landings) when compared to NEAM, is linked to the fact that it is more realistic to assess a fishery in terms of the total catch and landings, rather than on the independent landing rates of the targeted species, especially when analyzing and discussing fishery-specific indicators or categories. Additionally, the highly specialized NEAM fishing season involves low by-catch rates. For instance, as seen in Figure 2, NEAM landings represent at least 80% of the total catches during the NEAM fishing season in all the assessed years, except for the year 2003, coinciding with the *Prestige* oil spill, in which NEAM landings represented only 33% of the total.

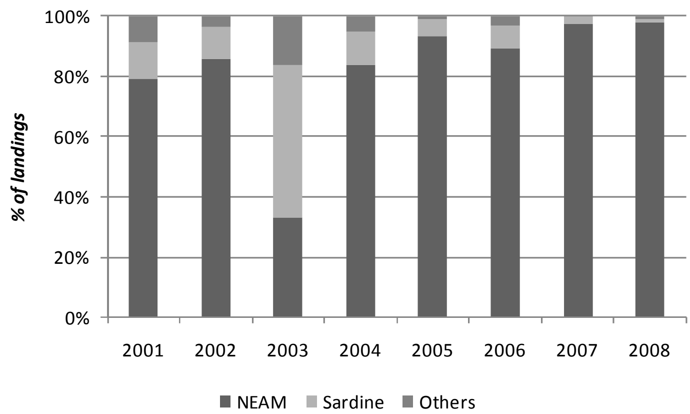


Figure 2. Relative landings of selected fishing species by the Basque purse seining fleet in the assessed fishing seasons.

4.1.2.5. Impact category selection

CML baseline 2000 method was selected as the computational framework for the attributional (retrospective) LCA analysis (Heijungs et al. 1992; Guinée et al. 2001). The impact categories that were included in the assessment were: Abiotic Depletion Potential (ADP), Acidification Potential (AP), Eutrophication Potential (EP), Global Warming Potential (GWP), Ozone Layer Depletion Potential (ODP) and Marine aquatic Eco-Toxicity Potential (METP). The software that was used for the computational implementation of the inventories was Simapro 7 (PRè Consultants 2011). Additionally, a series of fishery-specific categories were discussed in this research, including discard reporting, SIP as proposed by Nilsson and Ziegler (2007) and the Fisheries in Balance (FiB) Index as proposed by Pauly et al. 1998. The FiB Index aims at identifying the *fishing down marine food webs* phenomenon, which suggests that when fish species at the top of the trophic chain are overexploited, the captures of species lower down in the trophic level increase (Pauly et al. 1998; Villasante 2009).

4.1.3. Results

4.1.3.1. Characterization results identified on a temporal basis for landings during the NEAM season

Vessel operations were the main activities linked to fish extraction in the NEAM season that contributed to the environmental impact in all the conventional impact categories assessed, except for ODP, in which no environmental emissions were generated by this subsystem, and ADP. Nevertheless, a series of differences were found between the evaluated years. For instance, for GWP contributions ranged from 48% in the year 2002, to 62% in 2004, while contributions to METP were in all years above 83% (year 2001).Diesel consumption was identified as the main contributor to environmental impact within vessel operations for all the impact categories, except for METP in which the main burden was linked to anti-fouling emissions to the ocean.

For the ADP impact category the main environmental burdens were linked to the diesel production subsystem. For this particular activity, impacts ranged from 54% in 2002 to 71% in 2004. The relative contribution of diesel production to the other impact categories was in all cases below 10%.

ODP relative contributions were overwhelmingly related to cooling agent emissions by the refrigeration systems on board. Their contribution to this impact category was at least 90% (years 2001 and 2004). Additionally, cooling agents (mainly R22), also presented relevant environmental impacts for GWP, ranging from 4% in 2001 to 9% in 2003.

Finally, the net production and transportation subsystem also appeared as an important contributor in the ADP and GWP categories, with values ranging from 17 in 2004 to 34% in 2002 for ADP and from 14 in 2004 to 29% in 2002 for GWP. Other relevant activities or processes regarding environmental impact were the ice production system and to a lesser extent operations relating to the construction and maintenance of the vessels (anti-fouling and steel production). More detailed data on individual contributions per activity may be consulted in Online Resource 2.

When the total environmental burdens for the different seasons are compared, as can be observed in Figure 3, 2008 appears as the year with lowest associated burdens per FU for all impact categories, except for METP. The lowest impacts for the average vessel for METP were achieved in the year 2001. On the contrary, the season in 2003 had the highest associated impacts for all impact categories.

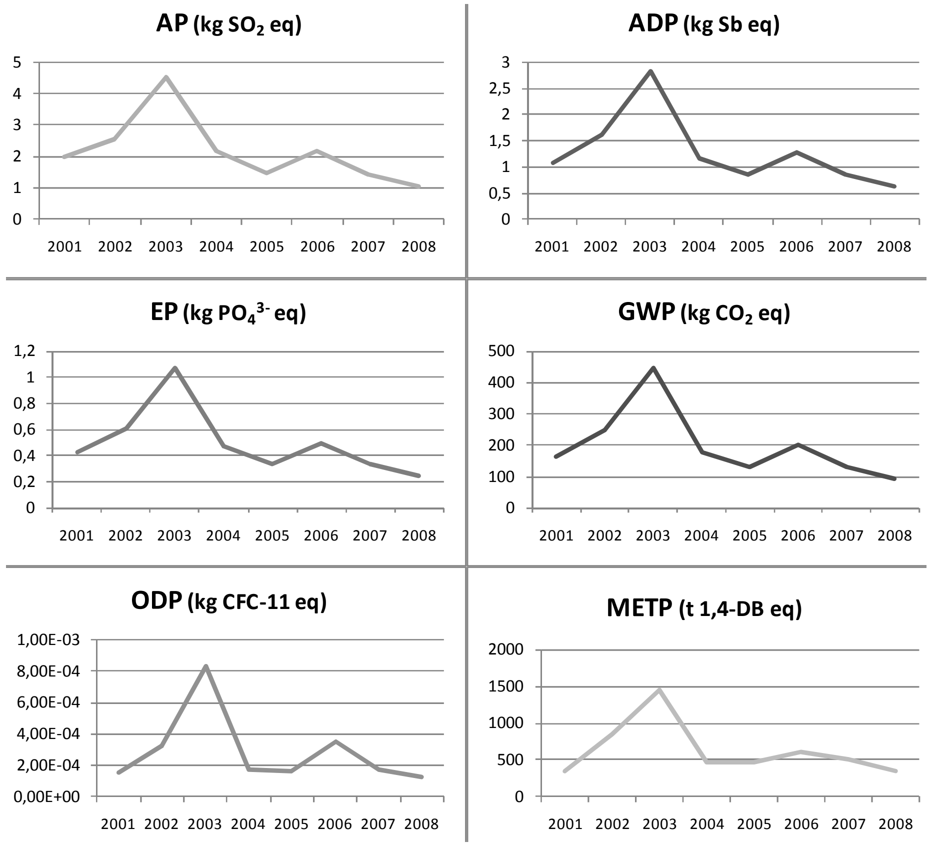


Figure 3. Environmental impact potentials for the average vessel per FU in the assessed period.

If the season in 2001 is taken as the reference, since it is the first assessed period in the selected time scale, a high oscillation in the environmental impacts can be observed from one season to another. On the one hand, the NEAM season in year 2003 shows environmental impacts at least 130% higher respect to the reference year (AP), while in some impact categories it is 324% higher (METP). Additionally, other NEAM seasons in which the associated burdens are above those registered for 2001 are 2002, 2004 and 2006.

On the other hand, the NEAM seasons in years 2005, 2007 and 2008 showed reduced environmental impacts when compared to the reference year. The lowest impacts corresponded to year 2008, in which the associated burdens were, for example, 43% lower than in the reference year for ADP and GWP. The results for years 2005 and 2007 were very similar, with environmental impacts ranging from 19 to 27% less than in the year 2001 for ADP, AP, EP and GWP.

4.1.3.2. Fishery-specific environmental impacts

Discard data were not available in this fleet for any of the assessed years. Nevertheless, according to a series of personal communications in Basque ports, skippers and fishermen confirmed that the discards generated through captures in the NEAM season by the Basque purse seining fleet are close to the average 1.6% reported by Kelleher (2005) for pelagic purse seining fisheries (J. Ruiz, marine researcher, personal communication, November 3, 2010).

As observed in Figure 4, the FiB Index shows a strong decline in the 2001-2003 period, a relatively stable period from 2004 to 2006 and a moderate increase in the final two years of the study. The year with the strongest fall in the index was 2003, in which the decrease was above 1, while 2008 appeared as the year with highest increase in the FiB Index (0.26). These results are in accordance with the mean trophic level (MTL) observed in the different years (Figure 4), showing, in the first place, a strong decline in the 2001-2003 period, and secondly a quick recovery and stabilization at a trophic level of around 3.6 until 2008. This tendency would translate in a 0.225 increase in the trophic level per decade.

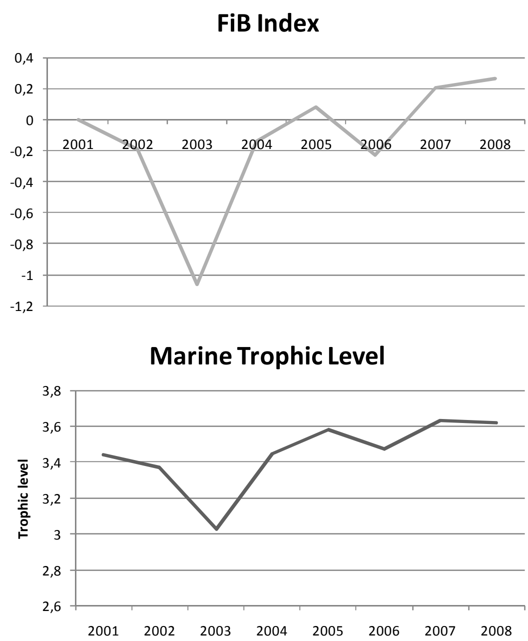


Figure 4. a) FiB Index for the Basque NEAM season (years 2001-2008). b) Calculated MTL for the Basque captures during the NEAM season (years 2001-2008).

4.1.4. Discussion

4.1.4.1. The importance of applying fisheries LCA for a significant extent of time

Environmental burdens related to the landing of 1 ton of pelagic fish in Basque ports show similar trends to other landings fished by purse seiners in other fisheries (Thrane 2004; Hospido and Tyedmers 2005; Vázquez-Rowe et al. 2010b), despite the increased variance that was observed between the selected years. Furthermore, due to the reduced fuel consumption of the analyzed fleet during the NEAM season, fuel related vessel operations only represented from 48% (2002) to 62% (2004) of the total environmental impact for GWP. Therefore, the importance of other vessel subsystems, such as net or ice production is greater than in other fleets that are more fuel intensive (Thrane 2004; Vázquez-Rowe et al. 2010a, Vázquez-Rowe et al. 2011b).

Nevertheless, the fact that this study comprises a relatively long period of time shows that there can be a great difference in the environmental burdens for a given impact category from one year to another. For instance, regarding GWP, the associated environmental impact per FU in the year 2003 was of 445 kg CO_2 eq, 4.68 times more than in 2008 (95 kg CO_2 eq). This tendency was observed for all the conventional impact categories that were included in this study, highlighting the importance of extending LCA inventories to wider periods of time, in order to obtain a broader perspective of the impacts associated to a particular fishery.

Additionally, this improvement may be extremely useful for those species that show erratic biomass and fecundity patterns (Fréon et al. 2008) or for those species that are under recovering schemes in depleted fisheries, since stock abundance variations and fishing overcapacity may generate a context that triggers fluctuations in environmental impact per FU.

The particular circumstances that surround fisheries as an industrial system make them unpredictable, since they are majorly dependent on fish abundance in a given period of time and a given spatial distribution. Other factors that may influence a fishery, such as management policies, are just a consequence of guaranteeing the sustainability of a limited resource (Clover 2006). Therefore, the extension of LCA inventories in the time line may be an important improvement for activities that rely entirely on the extraction of organisms from wild ecosystems (Hospido and Tyedmers 2005).

Furthermore, an extended timeline in fishery LCAs not only allows identifying tendencies in a particular fishery, but may also help detect specific circumstances that create a brusque variation in LCA characterization values. For instance, the outstandingly high environmental impact results attained by in the 2003 NEAM fishing season coincide with the wreck of the oil tanker *Prestige* off the Galician coast (November 19th 2002), which affected great part of the surface in the Cantabrian sea shelf. In fact, Sánchez et al. (2006) identified significant reductions in the abundance of megrim (*Lepidorhombus boscii*), Norway lobster (*Nephrops norvegicus*) and Pandalid shrimp (*Plesionika heterocarpus*) during the year 2003, with noteworthy recoveries during the 2004 season. Despite the fact that none of these species are pelagic, it is highly probable that NEAM suffered similar consequences linked to the oil spill, especially taking into account that during NEAM spawning, which starts usually in late February, there was an elevated presence of oil masses in the Cantabrian sea continental shelf (Sánchez et al. 2006).

Therefore, obtained results suggest the need to increase the timeframe of fisheries LCA on a regular basis. However, the handling of the results attained when timeline analysis is applied may give rise to biased or incorrect conclusions, due to the increased difficulty linked to multiple result interpretation (Vázquez-Rowe et al. 2010a). Hence, given that yearly results can be somewhat misleading, revealing the need to smoothen out short-term fluctuations at the same time as highlighting longer-term cycles, a five year moving average was proposed (Hamilton 1994), as can be observed in Figure 5 for the GWP impact category.

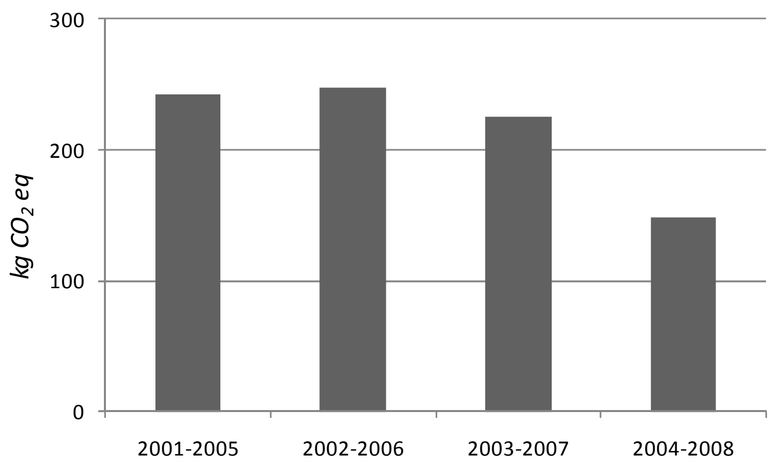


Figure 5. 5-year moving average for the GWP impact category.

4.1.4.2. Energy use

In terms of direct fuel consumption in the analyzed fishery, the average consumption ranges from 14.6 kg fuel/t fish in 2008 to 41.1 kg fuel/t fish in 2002, except for the year 2003, in which the energy use rocketed to 75.9 kg fuel/t fish. Therefore, the tendency observed for the assessed years shows that fuel consumption per ton of landed fish has decreased considerably in this period (see Online Resource 1). Recent literature (Schau et al. 2009), suggested that strong declines in this ratio are usually linked with important increases in the fuel price. However, the low fuel consumption linked to this fishery makes the fleet less sensitive to the fluctuations in fuel price. In fact, the increase in the amount of landings per day and vessel (Table 4), as well as the overall increase in landings for the years that presented lower energy use and environmental impacts (Figure 1), suggest that a leading factor influencing the environmental impact in this fishery is fish availability.

Comparison of these results with other studies shows that they are on the lower range of fuel intensity for purse seiners (Tyedmers 2001; Schau et al. 2009; Winther et al. 2009; Driscoll and Tyedmers 2010). More specifically, when the fuel effort of NEAM season landings is compared to that of other NEAM landing fleets, the fuel intensity in the Basque fishery is considerably lower than in other important NEAM fishing regions, such as Galicia, 176 kg of fuel/t NEAM or Norway, 90 kg of fuel/t NEAM (Tyedmers 2001; Schau et al. 2009; Vázquez-Rowe et al. 2010b)[1].

The increased fuel intensity reported not only for Galician NEAM landings by purse seiners, but also for NEAM landings regarding the Galician coastal bottom trawling fleet (Vázquez-Rowe et al. 2010b), when compared to those in the Basque Country, evidences the high environmental impact variability between regions showing the risks of reporting LCA results at a national scale for a particular coastal species (Table 6). Additionally, the relevance of studying these spatial variations increases when the fisheries of the analyzed country show independent patterns regarding fishing fleet characteristics and fishery management. It is important to note that results reported in Table 6 for the three fishing fleets are reported for 1 ton of landed NEAM following mass allocation.

[1] The case studies included from the bibliography take into account mass allocation,.

Table 6. Comparative characterization values for selected impact categories for 1 ton of round NEAM in three different Northern Spain fisheries (year=2008).

	Unit	F1	F2	F3
Sample size		35	30	24
Average beam	m	32.1	17	28
Energy use	kg fuel/t fish	14.6	176	496
Impact categories				
ADP	kg Sb eq	0.62	4.99	12.27
AP	kg SO_2	1.04	10.2	27.21
EP	kg PO_4^{3-}	0.24	1.95	4.97
GWP	kg CO_2	94.6	797	2,279
ODP	kg CFC-11 eq	1.24E-4	8.66E-4	7.86E-3
METP	t 1,4DCB	351	226	440
Discards	kg/FU	16.3	33.1	727
SIP	km^2	0	0	0.68

ADP= Abiotic Depletion Potential; AP= Acidification Potential; EP= Eutrofication Potential; GWP= Global Warming Potential; SIP= Seafloor Impact Potential; F1= Basque purse seining fleet; F2 = coastal purse seining; F3 = coastal bottom trawling.

The main reasons related to this low fuel intensity are mainly linked to the specialized season of NEAM catches in the gulf of Biscay, together with other key factors such as the reduced width of the continental platform in this area compared to the Galician coast, as well as the prohibition of purse seiners to fish within the Galician *rias*, forcing the fishing fleet in that area to target NEAM stocks at an increased distance from the coastline (MARM 2004).

4.1.4.3. Environmental impacts identified through fishery-specific impact assessment

The inclusion of fishery-specific results in LCA studies, as mentioned above, is a growing concern. However, in this particular research study the lack of specific discard data for the assessed fishery may have skewed the fishery-specific impact categories to a certain extent. Nevertheless, other publications relating to discards in pelagic fisheries in NW Spain, together with Basque skippers and fishermen comments, suggest that the discard rate for the NEAM season is very low. In fact, Vázquez-Rowe et al. (2011a) reported a discard rate of 3.2% for the Galician purse seining fleet targeting NEAM, horse mackerel and sardines, while Kelleher (2005) reports that seining linked discards in this area are close to the estimated 1.6% for this fishing gear worldwide.

SIP was not applied to this fishery, since it was assumed that purse seining is a fishing gear that causes negligible direct damage on the seafloor according to this index, despite the fact that lost nets can potentially create ghost fishing (Brown and Macfayden 2007; ICES 2000). Additionally, as can be observed in Table 5, NEAM landings performed by trawlers imply a considerable impact on the seafloor, showing that trawling fleets can create an increased impact on benthic ecosystems (Vázquez-Rowe et al. 2010b; Ziegler et al. 2003).

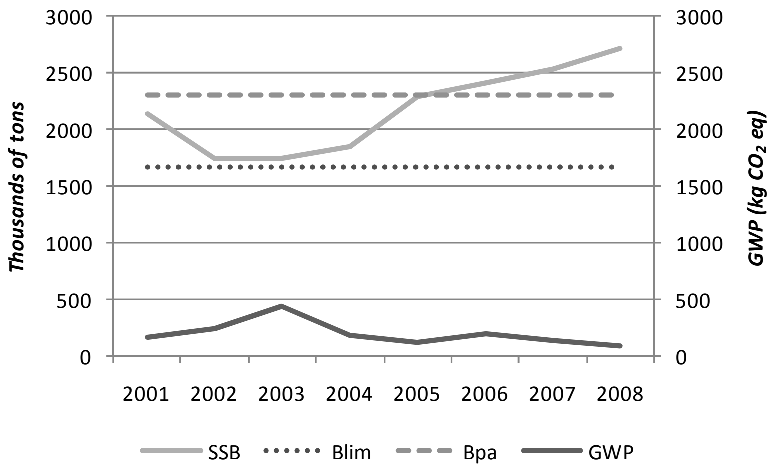
Regarding fishery exploitation, the increasing pattern for MTL (0.225 per decade) is quite remarkable when taking into account that fisheries assessed worldwide present a decreasing MTL (Pauly et al. 1998; Villasante 2009), especially those in which the targeted species are those with a higher trophic level (Branch et al. 2010). This increase is reflected in the FiB Index, which shows increasing positive values in the 2005-2008 period, contrasting with a sharp negative value for 2003. This tendency suggests that the increase in landings in the last few years of the assessed timeline is stronger than the net primary productivity may sustain through time or an expansion in the spatial distribution of the fishery (Pauly et al. 1998). The latter does not seem likely, since skippers from the Basque seining fleet reported not having changed their fishing zones in the assessed years.

A high number of ecosystems analyzed in previous studies worldwide regarding MTL show that an increase in fish landings is usually linked to higher landings of species with a low trophic level. However, the theory of *fishing down marine food webs* is not completely valid for this particular case study, mainly because of the fact that at least 79% of the landings of the fishery correspond to NEAM in each season. Furthermore, taking into account that this study also focuses exclusively on the NEAM fishing season does not make it possible to see the effects that the variable landings of the species may have on other coastal fishing seasons, not only of the coastal purse seining fleet, but also of other fleets that work in the area. Nevertheless, recent studies have identified a clear tendency in the last couple of decades in which the peak of catches for NEAM in the Cantabrian sea has shifted forward (Punzón and Villamor 2009), a situation that could also have important consequences on the ecosystem and the management of the fishery.

A certain correlation between yearly SSB variations and fluctuations in annual environmental impact for the selected categories was observed for this fishery, as can be observed in Figure 6. More specifically, the lowest levels of SSB, close to the biomass limit reference point (B_{lim}) are observed in the years with highest environmental impacts (2002-2003), while the lowest impacts for all the selected categories were found in years in which SSB levels were above the biomass precautionary approach reference point (B_{pa}).

The fact that energy use, as mentioned in the previous section, is lowest coinciding with the years with highest captures and SSB, suggests that environmental impacts in pelagic fisheries may be considerably influenced by the availability of fish in a given time period, provided that the vessels' fishing patterns do not experiment significant changes.

Nevertheless, a number of factors can influence the obtained results, such as the spatial distribution of the species and fishing management policies (e. g. the fulfillment of the NEAM TAC for Spain may cause increased environmental impacts per functional unit if strict daily quotas were to be enforced). Therefore, further research in this field should be taken in order to determine to what extent stock abundance affects the assessed environmental impacts.



SSB= spawning stock biomass

GWP= global warming potential

B_{lim}= biomass limit reference point

B_{pa}= biomass precautionary approach reference point

Figure 6. Annual spawning stock biomass (SSB) for the NEAM stock compared to annual global warming potential (GWP) environmental impacts for the assessed fishing fleet. *Source:* ICES 2010.

Finally, an additional factor that must be taken into account is the fact that the strong increase in stock abundance and landing in the Basque NEAM fishery may cause an increasing building capacity due to the expansion of the resources, which could develop into fleet and industry overcapacity whenever there is a new decline in the resources (Fréon et al. 2008; Villasante 2010; Villasante and Sumaila 2010).

4.1.5. Conclusions

To our knowledge, this is the first fishery LCA study in which there is sufficient data in order to conduct the methodology throughout a wide period of time. To date, LCA studies, despite having a broad and praiseworthy work behind when developing the LCI, failed to display the variations in environmental impact that a particular fishery or species could have from one year to another. The results obtained in this study suggest the need to increase the timeframe of fisheries LCA on a regular basis when assessing small pelagic species, such as NEAM, since they show strong annual environmental impact variations, in order to increase their feasibility and accurateness. Nevertheless, with the aim to avoid misleading multiple result interpretations, a five year moving average is proposed for result reporting. Further research is recommended in order to assess the importance of increasing the time line in fisheries LCA for those species that show small annual variations in environmental impacts.

More specifically, the life cycle environmental impact of NEAM extraction in the Basque country displayed low environmental impacts per FU, with a similar range to herring landing found in literature. When compared with other fisheries targeting NEAM, such as the Galician coastal seining fleet, the Basque fleet presented environmental impacts up to 88% lower, demonstrating the high regional variability that can be identified within the same country and the risks of reporting fishery LCA results at a national scale.

Finally, the Basque purse seining fleet has shown minimal fishery-specific impacts when regarding discards or seafloor impact. Furthermore, the increasing abundance of NEAM stocks in this area of the Bay of Biscay demonstrates an acceptable state of the stock in its southern area. Nevertheless, the strong increase in landings in the studied period, which has been brought about due to extensive overfishing, may create a future overcapacity of this particular industry (fleet and processing plants) if institutions were to allow an uncontrolled expansion of the sector.

4.1.6. Acknowledgments

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4.2. CONTRIBUCIÓN II

Operational efficiency and environmental impact fluctuations of the Basque trawling fleet using LCA+DEA methodology

Saioa Ramos[1]*; Ian Vázquez-Rowe[2]; Iñaki Artetxe[3]; María Teresa Moreira[2]; Gumersindo Feijoo[2]; Jaime Zufia[1]

[1] Food Research Division, AZTI-Tecnalia, 48160, Derio, Spain.

[2] Department of Chemical Engineering, School of Engineering, University of Santiago de Compostela. 15782 – Santiago de Compostela, Spain.

[3] Marine Research Division, AZTI-Tecnalia, 48395 Sukarrieta, Spain

*Corresponding author: Saioa Ramos. Email: sramos@azti.es

Telephone number: +34-667174357. Fax number: +34-946572555

Abstract

A recent study, using Life Cycle Assessment (LCA), suggests that natural fluctuations in stock abundance in fisheries may cause high variability in environmental impacts related to the Atlantic mackerel fishery in the Basque Country. The aim of this study is to analyze environmental fluctuations through time of a demersal species, European hake (*Merluccius merluccius*), caught by Basque bottom trawlers in European waters. The three-step LCA+DEA method, which combines LCA with data envelopment analysis (DEA), a linear programming tool, was implemented to assess annual variability of the environmental impacts in the period 2001-2006. The identification of the varying operational efficiency levels between vessels and the potential environmental gains of input minimization were explored. Results showed variations of up to 25% in the environmental impacts between years, although minimal environmental gains were identified through operational benchmarking, given the similar efficiency values between vessels. Hence, it was observed that despite substantial interannual changes in the impacts, there is limited potential for environmental impact reduction for the assessed environmental dimensions. Environmental and operational differences between years impeded setting a particular best-performing target for this production system, attributable to the high variance observed in input/output distribution through time. Finally, results seem to confirm the lower fluctuations in environmental impacts for demersal species fishing in comparison with those of small pelagic fish.

Keywords: data envelopment analysis; fishing vessels; hake; life cycle assessment; trawling.

4.2.1. Introduction

Protein supply from fishing and aquaculture activities constitutes an outstanding source of economic revenue for coastal communities (Cooley *et al.*, 2009). However, commercial fisheries are currently facing a serious crisis on a worldwide level, due to the overexploitation of fishing stocks (Pauly *et al.*, 2002; Worm *et al.*, 2009). This situation has led to important economic and social effects on local fishing communities (Hamilton, 2007; Hannesson, 2006). In virtue of this complex international context, European nations agreed in the 1970s to set common rules in European waters to regulate fish landings, protect fishing communities from abrupt changes in seafood trade and manage European fisheries with a continental perspective (Song, 1995). This strategy derived in what is now known as the Common Fisheries Policy (CFP).

Spain, the main fishing nation in Europe in terms of gross tonnage, landings and employment, concentrates most of its fishing infrastructure along the Cantabrian and Atlantic coasts, mainly in the Basque Country and Galicia (European Commission, 2012). While Galicia is responsible for nearly 50% of the Spanish fishing vessels and is characterized by mainly artisanal fishing vessels, (MARM, 2011), the Basque Country is noted for its small, specialized, and in many cases, industrial fishing fleets (Freire and García-Allut, 2000; Iriondo *et al.*, 2010; Murua, 2010). Even so, the size of the Basque fleet is comparable to that of important fishing nations, such as Denmark (EUROSTAT, 2009). Given the characteristics of these two regions, shifts in decision making in the CFP can have important consequences on the local communities and on the economy. Consequently, due to the increasing predominance of environmental issues in fisheries management, a spate of scientific research regarding environmental sustainability of fisheries has been observed in research centers throughout NW Spain (Borja *et al.*, 2000, 2011; Carballa-Penela and Domenech, 2010; Vázquez-Rowe *et al.*, 2012a).

While environmental sustainability in fisheries has usually been limited to the effects that fishing causes in marine ecosystems, a broader interpretation is starting to be applied, in which energy demand and materials used in industrial fishing are evaluated in order to assess their environmental and operational impacts (Hospido and Tyedmers, 2005; Vázquez-Rowe *et al.*, 2012a; Avadí and Fréon, 2013). One of the methodologies commonly used to analyze these environmental impacts is Life Cycle Assessment –LCA, the only internationally standardized environmental assessment tool (ISO, 2006a; Kloepffer, 2008).

LCA allows compiling and analyzing the inputs and the outputs, as well as the potential environmental impacts of a production system throughout its entire life cycle (ISO, 2006a;

2006b). Its use for the environmental assessment of food systems has shown a strong development in the past two decades (Roy *et al.*, 2009; De Vries and De Boer, 2010). More specifically, its application to fishery and seafood systems first appeared in Scandinavian countries (Eyjolfsdottir *et al.*, 2003; Ziegler *et al.*, 2003), as a tool to measure environmental impacts, such as global warming, toxicity or eutrophication, generated by operational activities in fisheries and fish processing systems (Hospido and Tyedmers, 2005). However, triggered by the holistic approach of LCA, a series of methodological innovations have been developed which take into consideration a series of fishery-specific impacts, such as the computation of discards, seafloor impacts or biotic resource use – BRU (Ziegler *et al.*, 2003; Pelletier *et al.*, 2007; Vázquez-Rowe *et al.*, 2012a).

The use of LCA in the Spanish seafood sector has undergone substantial development, with studies analyzing the environmental profile of a wide range of coastal and offshore fisheries (Vázquez-Rowe *et al.*, 2012a), as well as aquaculture (Iribarren *et al.*, 2012). These studies have centered mainly on hake fisheries worldwide (Vázquez-Rowe *et al.*, 2011b; 2013), since hake is the most consumed seafood product in Spanish households (Martín-Cerdeño, 2010; Asche and Guillen, 2012).

The integrated perspective of evaluating a wide range of environmental studies is definitely one of the main advantages of the LCA methodology. However, certain constraints can be linked to the applicability of regular LCA studies, such as temporal details of the case studies, or the way in which multiple datasets are handled (Weidema and Wesnaes, 1996; Reap *et al.*, 2008; Udo de Haes *et al.*, 2004). Together with methodological innovation within the tool, LCA practitioners have taken advantage of other existing methodologies in order to obtain suitable combined methods to solve the specific methodological barriers in a particular case study. One of these tools is the Data Envelopment Analysis (DEA), which has been combined with LCA in a wide range of publications under the name of the LCA+DEA method (Vázquez-Rowe *et al.*, 2010; 2011a; 2012b; Iribarren *et al.*, 2010; 2011; Jan *et al.*, 2012).

DEA is a linear programming methodology that provides a comparative empirical efficiency of multiple similar units (Cooper *et al.*, 2007). DEA has been applied to fisheries and other primary sector activities as an independent methodology in several studies (Kao *et al.*, 1993; Idda *et al.*, 2009; Griffin and Woodward, 2011; Picazo-Tadeo *et al.*, 2011). In fact, DEA, thanks to its ability to discriminate between different units within multiple datasets, has been used in the fishing sector to analyze the technical efficiency (TE) or the capacity utilization (CU) of a wide range of different fishing fleets (Tingley *et al.*, 2003; 2005; Färe *et al.*, 2006; Maravelias and Tsitsika, 2008), as a tool to evaluate the degree of overcapacity of fishing fleets worldwide (namely European fleets) while proposing strategies to improve and homogenize their efficiency (Herrero and Pascoe, 2003; Griffin and Woodward, 2011).

The use of DEA with LCA aims at linking the environmental impacts with operational benchmarking, as a way of attaining eco-efficiency verification through the theoretical optimization of inputs and outputs (Lozano *et al.*, 2009). Having said this, it should be noted that these theoretical optimization standards do not suggest specific improvement actions to attain eco-efficiency standards, even though some studies (Vázquez-Rowe and Tyedmers, 2013) do explore the specific sources of environmental inefficiencies. Furthermore, it also makes it possible to avoid the use of average data in multiple unit systems, reducing common sources of uncertainty in LCA studies (Vázquez-Rowe *et al.*, 2010), which enhances the delivery of best-performing targets for individual units (e.g. fishing vessels). Despite the fact that this method has only been developed quite recently, it has been successfully applied to a range of production systems in the primary sector, including fisheries, viticulture or dairy farms (Iribarren *et al.*, 2011; Vázquez-Rowe *et al.*, 2011a; 2012b). However, a series of unexploited potentials of this joint methodology remain unexplored (Iribarren *et al.*, 2010). One of these is the use of a timeframe methodological option for DEA, window analysis, to determine the environmental impact efficiency of production systems on a time frame basis (Charnes *et al.*, 1985).

Previous studies have highlighted the strong variations in stock abundance in small pelagic fisheries which have occurred long before human exploitation of marine resources and, therefore, cannot be associated with fishing activities (Holmgren-Urba *et al.*, 1993; Schwartzlose *et al.*, 1999; Fréon *et al.*, 2008). Furthermore, a study conducted by Ramos *et al.* (2011a) suggested strong changes in environmental impacts on a seasonal basis in the Basque Atlantic mackerel (*Scomber scombrus*) fishery. In fact, they used the *Fisheries in Balance* (FiB) method as an auxiliary tool to LCA, finding a strong correlation between years with low stock abundance and higher environmental impacts. In this case study, the LCA+DEA method is applied to the Basque trawling fleet in ICES Division VIIIabd, with the aim of determining whether the abovementioned environmental profile variability of pelagic species can also be observed in demersal species (i.e., mainly European hake – *Merluccius merluccius* – in the present study), which tend to show lower natural fluctuations in stock abundance (ICES, 2013; see Figure 1).

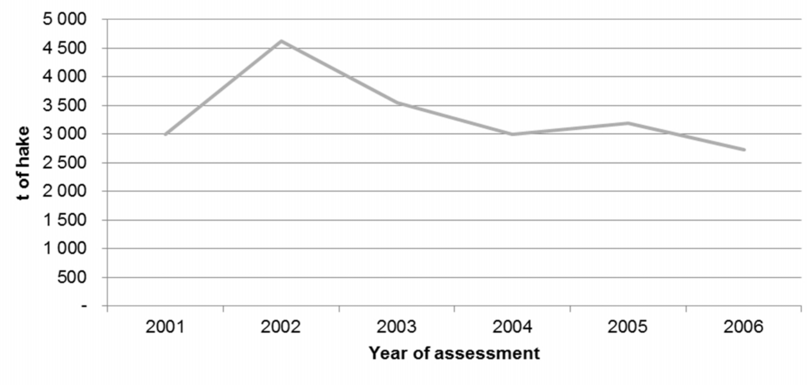

Figure 1. European hake landings in the Basque Country in the 2001-2006 period. Source: AZTI (2010).

Furthermore, the complementary use of DEA aims to detect not only timeline variance, but also differences between fishing vessels, as well as estimating the environmental consequences of inefficiencies in vessel operations. Consequently, the main objective of the study is focused on assessing annual variability of the environmental impacts of fishing activity in the Basque bottom trawling fishery.

4.2.2. Methodological framework

4.2.2.1. Definition of the case study

Characteristics of the production system analyzed

The offshore trawling fleet in the Basque Country had a total of 20 vessels in 2006, which target a set of high and medium economic value demersal fishing species in the Celtic Sea (Figure 2), sharing the fishery stocks with vessels from France, United Kingdom, Ireland, Denmark and other Spanish regions, mainly Galicia (Murillas *et al.*, 2008). European hake constitutes the main target species for the Basque trawling fleet, due to its culinary importance and its attractive sale price in Basque fish markets. Nevertheless, other species, such as blue whiting (*Micromestimius poutassou*), megrim (*Lepidorhombus* spp.) and common sole (*Solea solea*), are also landed by these vessels.

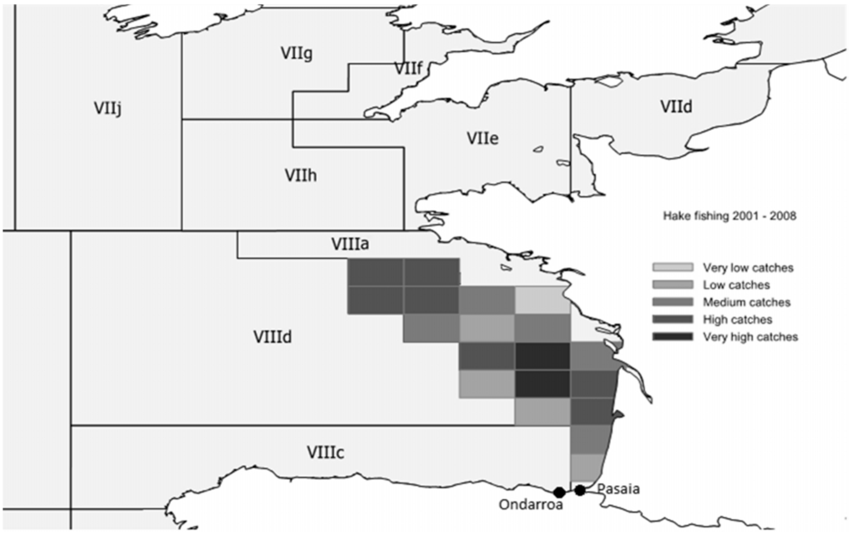

Figure 2. Map illustrating the main fishing areas of the Basque trawling fleet. Roman numbers on graph refer to the ICES areas. The variable tones of grey in zone VIII refer to the amount of hake caught by Basque vessels. Dark grey tones reflect increasing levels of catch.

This fleet is constituted by two different types of bottom trawlers, known as *baka* and bottom pair trawlers. *Baka* trawlers operate as single boat trawlers and are, therefore, using otter doors to spread the trawl. Their trips last on average 6 days, with haul durations that range from 4 to 5 h. Catch is generally landed in two specific Basque ports: Ondarroa and Pasaia. Bottom pair trawlers are composed of two vessels trawling a single net. The average trip for these vessels is usually 5 or 6 days, with longer hauls - 7-8 h (Murillas *et al.*, 2008).

Unit of assessment determination and data acquisition

Decision making units (DMUs) are each of the independent entities that make up the multiple unit system (Cooper *et al.*, 2007). When assessing fishing systems with the LCA+DEA method, the chosen unit of assessment in previous studies has been the fishing vessel, since this approach guarantees a realistic perception of the vessels' performance (Vázquez-Rowe *et al.*, 2010). Nevertheless, for this specific fleet it may be argued that pair trawlers do not represent two separate entities, since they operate under the same operational and environmental conditions. However, in this article all vessels were considered as independent DMUs for two main reasons: on the one hand, the fact that all pair trawlers also performed trips as single operating vessels at given times of the year; on the other, pair trawlers did not always operate with the same vessels, with several changes observed on an annual and interannual scale.

Figure 3 shows a schematic representation of the main material and energy flows that were considered in the production system.

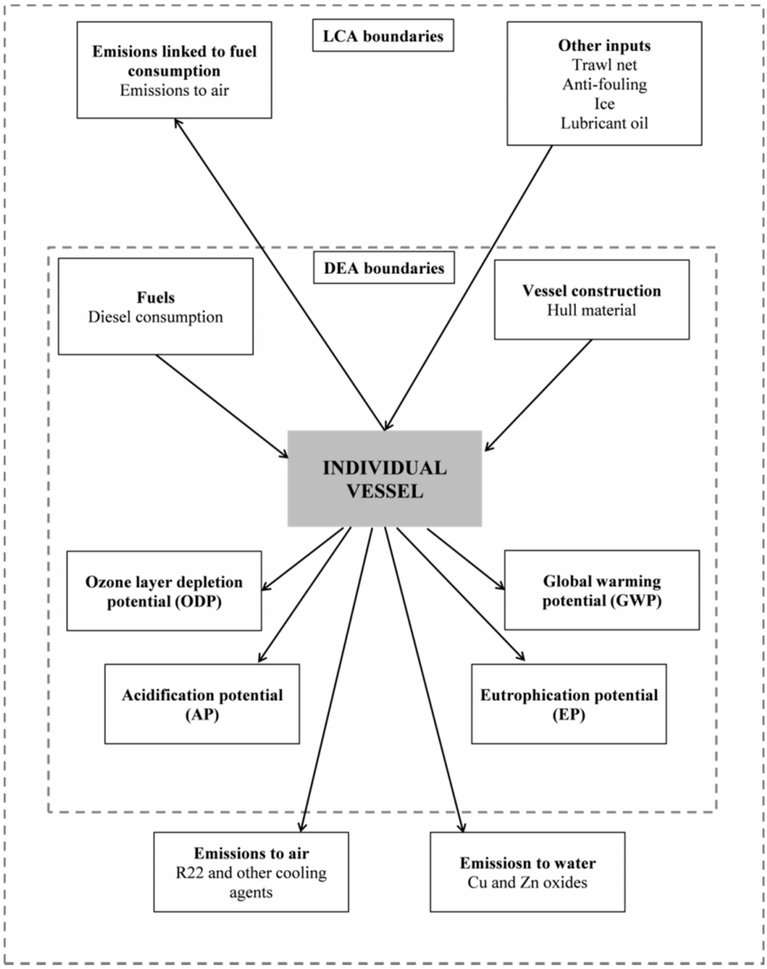

Figure 3. Inputs and outputs included in the production system for LCA+DEA implementation.

The system boundaries were limited to the fishing activities and their background processes, excluding the on-land phases of fish supply chains. This perspective was considered due to the fact that industrial processes of transformation and transport are not expected to vary much from one year to another in terms of environmental impact, at least when the existing processing industry is working close to its full capacity (Benedetto et al., 2014). In fact, any variations in these stages would not be primarily affected by changing landing rates by the vessels. A final issue that was taken into account to limit the boundaries to the fishing stage was the complexity of the supply chain. Fish products, especially those consumed fresh, such as hake in Spain, are part of highly complex market flows in which the existence of clearly comparable units of assessment, as needed for DEA implementation, are very diffuse and not homogenized with the fishing stage (Kaplan, 2000; Ilbery and Maye, 2005; Martín-Cerdeño, 2010).

Regarding the selection of inputs and outputs to be included in the system, there are certain discrepancies that can be identified between LCA and DEA. For the LCA methodology an integrated life cycle inventory, as defined by ISO 14044 (ISO, 2006b) was followed, considering all the sources of potential environmental impacts, as described in published LCA review and case studies of fishing fleets (Pelletier *et al.*, 2007; Parker, 2012; Vázquez-Rowe *et al.*, 2012a; Avadí and Fréon, 2013), especially those analyzing trawling fleets (Vázquez-Rowe *et al.*, 2011b; 2012b). However, it should be noted that the inclusion of certain inputs are only attributed to LCA due to the fact that DEA analysis only takes into account a selection of the most significant inputs and outputs (Vázquez-Rowe *et al.*, 2010).

The inputs that have been considered for LCA computation include a series of operational items in the vessels, such as diesel consumption, anti-fouling paints, ice consumption or vessel characteristics. Inputs such as vessel characteristics, fuel consumption or fish landings were obtained mainly from a specific register of fish at first sale available at AZTI-Tecnalia. Furthermore, a series of additional information, such as the number of nets used per vessel or the amount of refrigerant loss to the atmosphere, were retrieved through anonymous surveys carried out on Basque skippers. Background data associated with the production of fuel, nets or anti-fouling paints were taken from the ecoinvent® database (Frischknecht *et al.*, 2007). The LCA stage of the study also included the computation of the derived emissions from the different processes, such as diesel combustion or anti-fouling loss to sea (Hospido and Tyedmers, 2005). Finally, fishery-specific inventory items, such as the amount of area swept by trawlers or discards were not available for this dataset and were excluded from the assessment. Nevertheless, it is important to remark that these excluded items are currently not required items in standardized LCA impact categories (Pelletier *et al.*, 2007).

In contrast, the DEA matrix employed only included operational inputs that have proved to be either of key importance in previous environmental assessment studies or which imply an important economic expenditure. Consequently, based on a preliminary LCA assessment of the analyzed fishing fleet (Ramos *et al.*, 2011b) and other LCA trawling fleet studies, diesel production and consumption, and hull material (provision and use) were the two inputs selected while total landed catch was the output (Vázquez-Rowe *et al.*, 2010; 2011a; Ziegler *et al.*, 2011). Other operational activities, such as the use of trawl net or the emissions of refrigerants involved lower environmental burdens, except for specific impact categories (e.g. cooling agents in terms of ozone depletion). Therefore, following the three step LCA+DEA, which is explained in section 2.2, a set of commonly used impact categories in fisheries LCA were also integrated as inputs in the DEA matrix so as to account for an integrated environmental assessment of the fishing vessels.

4.2.2.2. LCA+DEA framework

Two different methods have been developed within the LCA+DEA methodology. In the first place, the five-step method has been used mainly for eco-efficiency verification and to determine the consequences on environmental impacts of operational inefficiencies (Vázquez-Rowe *et al.*, 2010). Secondly, the three-step method seeks mainly the estimation of environmental impact efficiency, while performing a simultaneous benchmarking of a set of operational and environmental parameters (Iribarren *et al.*, 2011). To analyze the Basque trawling fleet over the selected years a modified three-step LCA+DEA methodology has been chosen (Lozano *et al.*, 2010), with the objective of estimating environmental impact efficiencies from a timeline approach (Figure 4):

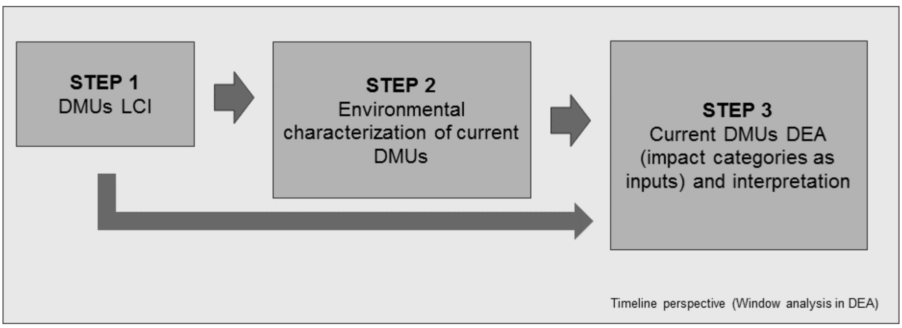

Figure 4. Three-step LCA+DEA method adapted to window analysis. NOTE: DMU= decision making unit; LCI= life cycle inventory; DEA= data envelopment analysis.

i. *LCI for each of the DMUs:* The first step was to obtain a representative LCI of each selected DMU. Each DMU, as mentioned above, is a trawl vessel. For the inventory the most relevant aspects which influence the impact analysis have been taken into account, as discussed in section 2.1.

ii. *Life Cycle Impact Assessment (LCIA) for each of the DMUs:* The second step consists of an environmental impact characterization based on the LCI developed in the first step. For this characterization the CML baseline 2000 method was selected as the computational framework for the LCA analysis (Guinée *et al.*, 2001).

iii. *DEA analysis from the characterization values obtained in the second step.* The final step of the three step approach consists of the DEA computation. Hence, the DEA matrix is generated by compiling a set of operational items and environmental impact categories as inputs, as well as the desired output. In this modified version of the method, and in order to capture the variations of efficiency over time, the technique called 'window analysis' was proposed (Charnes *et al.*, 1985). Window analysis assesses the performance of a particular DMU over time by treating it as a different entity in each time period (Charnes *et al.*, 1985). Therefore, the performance of a DMU during a particular period is compared not only to the performance of other units, but also to its own performance in other periods.

When computing window analysis in DEA, it is important to note that a window length must be selected, which determines the extent of the relative comparability between DMUs (Charnes *et al.*, 1985). For instance, if a window length of 1 is assumed, this implies that the DMU efficiencies are calculated independently from a temporal perspective. In contrast, if the window length is expanded to 2 or more, efficiency calculation is based on the total entities in this period. If this second approach is extended to the complete panel dataset the reference set will refer to the entire matrix, but will not account for changes in technology, natural resources or other assumptions over time (Wu, 2005).

4.2.3. Application of the proposed method and results

4.2.3.1. Step 1: inventory data

Inventory data for Basque trawlers were obtained for a total of 7 vessels, belonging to the ports of Pasaia (43° 19´N, 1° 55´ O) and Ondarroa (43° 19´ N, 2° 25´O). Despite the fact that data of up to 27 vessels were available in some of the specific years, only the included vessels reported data for the entire period of study. The selected period included annual data for a total of 6 years (2001-2006). The most important target species of this fleet is European hake, as can be seen in Table 1.

Table 1. Annual relative amount of landed individual marine species in the assessed sample (Values in % over total catch) and brief description of the assessed fleet.

Fishery data						
Species	2001	2002	2003	2004	2005	2006
European hake	54.33	83.29	88.03	84.16	85.91	83.29
Other species	45.67	16.71	11.97	15.84	14.09	16.71
Total	100.00	100.00	100.00	100.00	100.00	100.00
Northern Stock trawling fleet data						
Number of vessels	52	50	47	45	40	34
Inventoried sample	7	7	7	7	7	7
% over total	13.5	14	14.9	15.6	17.5	20.6
Total gross tonnage (GT)	15,396	14,984	14,195	14,229	13,058	11,287
Average beam (m)	30.8	30.8	30.6	30.6	30.3	29.6

Nevertheless, given the vessel perspective used in this study, inventory data, which are shown in Table 2, were assigned to the total catch, rather than allocating the inputs and outputs to hake. Therefore, the functional unit (FU) was set as one tonne of gutted fresh landed fish caught by bottom trawlers from 2001 to 2006 in ICES division VIIIabd.

Table 2. Brief average life cycle inventory of the selected sample (data per tonne of landed catch).

Inventory	2001	2002	2003	2004	2005	2006
INPUTS						
From the technosphere						
Fuel (kg)	2776	1896	1793	2625	2008	2034
Steel (kg)	45.7	16.7	12.0	15.8	14.1	16.7
Trawl net (kg)	6.6	4.4	5.1	6.0	5.2	4.6
Anti-fouling (kg)	152.1	152.1	150.0	149.2	151.0	148.3
OUTPUTS						
To the technosphere						
Landed fish (t)	1.0	1.0	1.0	1.0	1.0	1.0
Emissions to air						
CO_2 (kg)	8801	6010	5683	8322	6365	6447
CO (kg)	20.5	14.0	13.3	19.4	14.9	15.1
NO_x (kg)	199.9	136.5	129.1	189.0	144.6	146.4

Direct air emissions from fuel combustion were calculated based on the EMEP-Corinair Emission Inventory Handbook of 2006 (EMEP-Corinair, 2006). Cooling agent emissions were included in the inventory based on the data provided by a specialized retailer (José Manuel Juncal, Kinarca S.A., June 2010, personal communication).

4.2.3.2. 3.2. Step 2: Environmental characterization of current DMUs

Impact category selection in the assessment was based on commonly used categories in fishery systems (Pelletier *et al.*, 2007; Vázquez-Rowe *et al.*, 2012a), from the CML Baseline 2000 method (Guinée *et al.*, 2001): Acidification Potential (AP), Eutrophication Potential (EP), Global Warming Potential (GWP) and Ozone Layer Depletion Potential (ODP). Moreover, these impact categories have been identified as having a high level of convergence with ILCD recommendations in order to compute the LCIA (ILCD, 2011). The software used for the impact assessment was Simapro 7.3 (Goedkoop *et al.*, 2010).

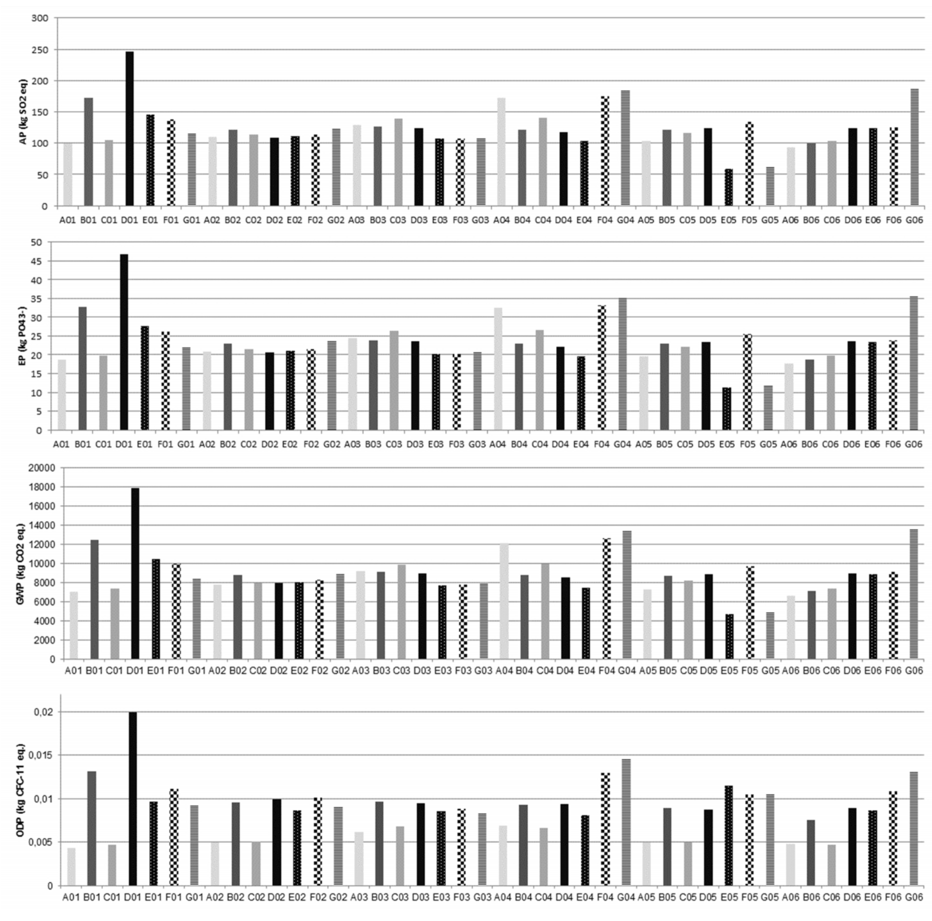

Figure 5. Current environmental characterization values per FU for individual DMUs (letters A-F represent the different vessels; the two final digits represent the year of assessment: 01= 2001; 02= 2002, etc.).

Figure 5 presents the characterization results for each of the assessed vessels on an annual basis referred to the FU for the different impact categories. The results show that in all categories there is a considerable variability in environmental impacts between the assessed vessels. For instance, in the GWP category the average impact for the entire period per FU was 9,300 kg CO_2, although the annual averages ranged from 7,150 kg CO_2 to 10,900 kg CO_2. In addition, variability between vessels within one year of operation varied from a standard deviation of 3,700 kg CO_2 in 2001 to 640 kg CO_2 in 2002. Finally, it should be noted that the remaining impact categories showed similar trends.

4.2.3.3. Step 3: Current DMUs DEA and result interpretation

Once step 2 was accomplished, a DEA matrix was established based on the LCI data gathered in step 1. As mentioned above, the DEA matrix in the 3-step method jointly computes inventory inputs and outputs together with environmental input results. Consequently, the four impact categories assessed in step 2 were included as inputs in the matrix, together with the two operational inputs (diesel and hull material) and the output (landed catch), as can be seen in Figure 3. Table 3 presents the matrix referring to the first year of assessment (2001). The DEA matrices for the other years evaluated are available in Online Resource 1.

Table 3. DEA matrix with individual vessel inputs and outputs for the selected trawling fleet sample for one selected year (2001).

DMU	O Catch (kg/year)	I-1 Diesel (l/year)	I-2 Hull material (kg/year)	I-3 AP (kg SO_2 eq./year)	I-4 EP (kg PO_4^{3-} eq./year)	I-5 GWP (t CO_2 eq./year)	I-6 ODP (g CFC-11 eq./year)
1	438,567	801,223	3754	43,415	8321	3426	8973
2	251,376	798,997	6810	27,659	5244	2155	5164
3	400,480	769,605	37,455	41,771	7948	3285	8211
4	159,668	724,860	6671	39,290	7454	2869	3463
5	327,119	881,456	5406	47,762	9057	3615	6800
6	284,848	724,860	6671	39,290	7454	2991	5911
7	327,119	707,861	6052	38,369	7279	2970	6742

DMU= decision making unit; O= output; I-1= input 1; I-2= input 2; I-3= input 3; I-4= input 4; I-5= input 5; I-6= input 6; AP= acidification potential; EP= eutrophication potential; GWP= global warming potential; ODP= ozone layer depletion potential.

Window analysis in the slacks-based measure (SBM) framework was selected as the model to compute the matrix. More specifically, an input oriented model was selected for two main reasons. On the one hand, European hake and the other landed species constitute a limited natural resource (Vázquez-Rowe and Tyedmers, 2013). On the other hand, the existence of a rigid quota system in this fishing area for the different national fleets (Council Regulation, 2009) also involves an important constraint that makes it more feasible to target a minimization of inputs while maintaining outputs. A constant return to scale (CRS) approach was assumed for the model given the fact that the fleet operates in a competitive market (Cooper *et al.*, 2007; Lozano *et al.*, 2009). Model formulation can be consulted in Appendix A.

DEA-solver Pro was the specific software used to compute the DEA matrix (Saitech, 2012). The matrix was then assessed using two different lengths of window: 1 and 6. This choice was based on two factors. On the one hand, the vessels in the sample operate in a given area every year under a series of specific quota limitations and biological moratoria. Therefore, year after year, these vessels compete for the same natural resource, but under different environmental, social and political conditions, due to changes in fisheries management, stock abundance, etc. Consequently, a length of window of 1 in the DEA results is justified as a way of comparing the annual efficiency between vessels and their individual interannual fluctuations. This perspective is commonly referred to as a *contemporaneous* approach (Tulkens and Vanden Eeckaut, 1995). On the other hand, a window length of 6 makes it possible to compare the entire six year period under the same reference set, permitting a broader comparison throughout the entire window. This perspective was chosen to provide efficiency trends in the fishing fleet over time in order to evaluate if any underlying factor may influence the overall annual results. This approach is named *intertemporal* since it provides an assessment based on the observation of the entire study period. However, it should be noted that in the selected case study the selected period only illustrates the timeline for which data were available, while the entire lifespan of the vessels or the existence of the fishery is not accounted for; therefore, in literature it is named *local intertemporality* (Cullimane and Wang, 2006).

Average efficiency scores, based on the first approach of window analysis (length of window = 1), are shown in Table 4. As can be observed, 57.1% of DMUs were efficient ($\Phi = 1$). In fact, vessels 1 and 3 presented an efficiency of 100% for the entire period. Additionally, only 21.4% of the DMUs showed average efficiency scores below 95%. Consequently, the average efficiency scores for each individual vessel for the entire period were all above 90%, with vessel 7 showing the lowest value ($\Phi = 0.93$). Finally, on an annual basis, average efficiencies in the 2001-2003 period were higher ($\Phi > 0.98$) than in the second period of assessment (<95%), in which standard deviations ranged from ±8.0 (2006) to ±11.1 (2005).

Table 4. Average efficiency scores (Φ_0) in percentage (%) per fishing vessel for the assessed period (length of window = 1).

Year	V 1	V 2	V 3	V 4	V 5	V 6	V 7	Aver.	SD
2001	100	97.0	100	94.3	98.4	98.6	99.3	98.2	±2.0
2002	100	99.2	100	100	100	99.8	99.0	99.7	±0.4
2003	100	99.1	100	100	100	100	100	99.9	±0.3
2004	100	94.8	100	100	100	82.8	80.7	94.0	±8.6
2005	100	75.0	100	79.1	100	100	98.4	93.2	±11.1
2006	100	100	100	89.6	100	84.3	82.9	93.8	±8.0
Average	100	94.2	100	93.8	99.7	94.3	93.4		
SD	±0.0	±9.6	±0.0	±8.4	±0.6	±8.3	±9.0		

SD= standard deviation, length of window= indicates the number of temporal units that are assessed together for DEA computation.

Figure 6 presents a comparison between the average original DMU and the average virtual target DMU in terms of the environmental impact category values (per FU) that were included in the DEA matrix.

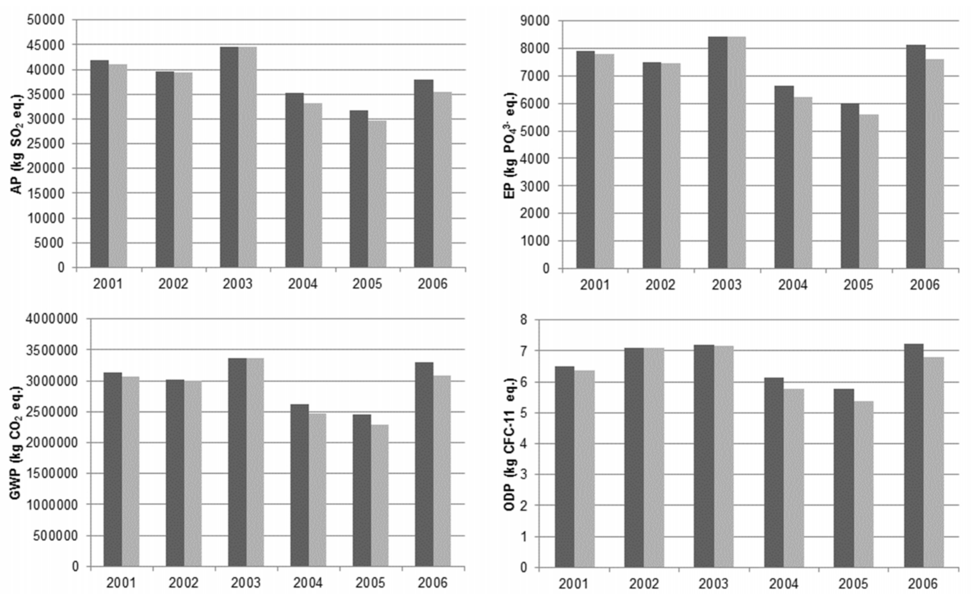

Figure 6. Environmental impact potentials for the current average vessel (dark bar) and the target average vessel (light bar).

The alternative approach that was followed for window analysis took into account the entire panel (length of window = 6). In this case, the best performing DMUs were limited to 14.3% of the assessed sample (Table 5). In fact, only 3 vessels attained full efficiency at least in one year of assessment. Average fleet efficiencies ranged from 91.8% in 2005 to 71.6% in 2001. Concerning individual vessels, their average efficiency scores throughout the window ranged between 91.2% (vessel 1) and 70.9% (vessel 7). All vessels presented high standard deviations from year to year while higher variability in standard deviation was observed on an annual basis.

Table 5. Average efficiency scores (Φ_0) in percentage (%) per fishing vessel for the assessed period (length of window = 6).

Year	V 1	V 2	V 3	V 4	V 5	V 6	V 7	Average	SD
2001	100	53.5	100	40.8	67.8	63.8	75.3	71.6	±22.3
2002	91.7	72.4	93.5	75.6	82.3	73.0	73.3	80.1	±8.9
2003	77.8	70.6	73.3	75.8	78.5	86.7	80.5	77.6	±5.2
2004	84.3	73.1	73.7	100	81.7	53.1	50.3	73.7	±17.5
2005	98.2	74.0	100	77.7	97.7	100	95.0	91.8	±11.1
2006	96.4	87.1	100	77.1	80.0	64.3	51.2	79.4	±17.3
Average	91.2	71.8	90.1	74.5	81.3	73.5	70.9		
SD	±8.7	±10.7	±13.1	±19.0	±9.6	±17.2	±17.4		

SD= standard deviation, length of window= indicates the number of temporal units that are assessed together for DEA computation.

4.2.4. Discussion

4.2.4.1. Environmental and operational performance of the Basque trawling fleet

The *contemporaneous* approach results obtained for this fleet shows low relative inefficiency levels between vessels within each time period (Table 4), suggesting similar operational patterns in all the vessels assessed. In fact, in Figure 5 it can be seen that despite substantial changes in environmental impacts between the assessed years, the potential for obtaining environmental benefits if vessels were to perform in an efficient manner is limited. For instance, the vessels assessed would have saved approximately 595 tonnes of GHG emissions in the period under analysis if they had performed in an efficient way. Nevertheless, results for years 2001-2003 showed an insignificant potential for reducing environmental impacts, while the following three years showed a higher potential for reduction.

Consequently, the results suggest that, taking into account the political and stock abundance constraints that exist in the fishery, the vessels assessed are operating at a high capacity level. Nevertheless, this conclusion should be taken with caution due to the fact that DEA only measures relative efficiencies (Charnes *et al.*, 1994). Hence, another alternative may be that vessels simply showed similar levels of inefficiency, and that due to the absence of a best performing vessel throughout the period evaluated, these inefficiencies are not visible (Vázquez-Rowe and Tyedmers, 2013).

Furthermore, the similar operational and environmental results for the different vessels suggest that the effects of the "skipper-effect", which is defined as the potential that the skill of fishermen has on the correct operation of fishing vessels, is minimal in this particular fleet. This observation is in line with previous studies that defend that the "skipper-effect" is more visible in fleets, such as purse seining, where individual strategies by skippers may have a higher influence on creating a higher yield, and therefore, minimizing environmental impacts (Gaertner *et al.*, 1999; Ruttan and Tyedmers, 2007; Vázquez-Rowe and Tyedmers, 2013). Moreover, it should be noted that the variations in efficiency between the vessels assessed may also be attributable to other factors, such as technical efficiency or data misreporting (Tingley *et al.*, 2005; Parker and Tyedmers, 2011).

From an LCA perspective, it is important to note that the environmental impact results obtained per FU for the different vessels throughout the period assessed are in accordance with impacts reported by other Spanish fishing fleets in the Northern Stock (Vázquez-Rowe *et al.*, 2011b). This finding does not only suggest similar operational patterns for the two fleets, but also advocates an extended validity of the environmental impact trends for other similar fleets operating in the same area.

Time-dependent variations in environmental impacts identified in pelagic fisheries could also be occurring in other types of fisheries (Ramos *et al.*, 2011a). Results for the demersal trawling fishery assessed, despite showing changes on an annual basis, do not show substantial variations. Hence, provided that there are no significant changes in the way the fishery is being run, and biomass levels maintain their recovery (ICES, 2011), it is feasible to presume that environmental impacts in the fishery should not suffer abrupt changes in future years, unless specific technological, climatic or fishery management (including policy) changes occur.

Finally, an interesting future assessment would be to analyze the differences in operational efficiency at an inter-assessment level (Iribarren *et al.*, 2011), comparing the evaluated fleet with other national fishing fleets targeting demersal species in the area. This issue is of great importance since the European Union considers different quota limitations depending on the vessel flag. It would also have a relevant role when linking inter-fleet CU with the environmental profile of the vessels (Vázquez-Rowe and Tyedmers, 2013). Moreover, recent studies highlight that fisheries management can have a determining effect on changes in environmental impacts (Misund *et al.*, 2002; Driscoll and Tyedmers, 2010).

4.2.4.2. The importance of timeline analysis in environmental impact determination and operational inefficiency mitigation

When the results obtained with a window length of the entire period –*intertemporal* approach– (length of window = 6) are analyzed, the average efficiencies for the average vessel varies considerably between years, suggesting that vessels have difficulties in maintaining their operational patterns, as well as the catch rates (Table 5). This inability, which can also be seen through the differing environmental impacts from year to year (Figure 5[2]), can be due to a varied combination of factors, including stock abundance and distribution, changes in total allowable catches (TACs), meteorological conditions or even the price market (Asche and Guillen, 2012). However, it was not possible to establish any type of consistent pattern when crossing operational efficiency with a series of potential influencing factors (i.e., total biomass in the stock, total landings and TAC limitations) or environmental consequences evaluated in this research (ICES 2011). Nevertheless, expected technological improvements in this fleet, in accordance with average trends in European fleets that estimate a ~5% increase per year (Gelchu and Pauly, 2007; Villasante and Sumaila, 2010), would suggest an increase in efficiency through the 6 year panel using the *intertemporal* approach. However, this was not the case for the analyzed sample, indicating that despite an expected technological improvement of the vessels, management and fishery-linked factors are the main underlying issues behind variability between years.

[2] Note that data in Figure 5 were elaborated using the efficiency values in Table 4, corresponding to a length of window in DEA of 1. The inappropiateness of using a length of window of 6 for the calculation of the target environmental impact results has previosuly been discussed in the manuscript.

Consequently, the results illustrate the difficulty in setting a particular best-performing target for this type of production system, as developed in previous LCA+DEA analysis in other primary sector activities, such as farming (Vázquez-Rowe *et al.*, 2012b), given the high variance in input/output distribution in a timeline perspective detected in the system.

4.2.5. Conclusions and perspectives

The combined LCA+DEA approach has been applied in this study in order to assess the variation of potential environmental impacts over time. As suggested in previous studies, environmental impacts in many primary sector activities, including fishing, are strongly influenced by temporal fluctuations in natural resources. In this context, this study was carried out focusing on the timeline variations in environmental impacts linked to the fishing of a demersal species, European Hake, and hence, detecting efficiency differences on vessels over the selected period.

While results certified the variable environmental impacts on an interannual basis, these fluctuations were substantially lower than those obtained for the Basque small pelagic fish fleet Ramos et *al.*, 2011a). Additionally, the use of DEA highlighted the reduced space for input minimization under current operational patterns, given the similar performance of vessels within each year. Nevertheless, the high variability in efficiency levels when the entire period is examined underlines the existence of external factors that influence vessel performance. However, no correlations were found with any specific underlying factor that would explain this variation.

In any case, this study stresses the appropriateness of implementing a timeline perspective in the environmental assessment of fishing systems, as well as proving that LCA+DEA is an adequate method for identifying operational inefficiencies under these terms. Therefore, future development in this field may explore the specific sources that lead to operational inefficiency in fishing fleets, in order to provide specific strategies to implement the reduction of environmental impacts on a stakeholder or political level.

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Appendix A. Window analysis model formulation

The DEA model used for the Basque bottom trawling fishery was the "window analysis" model. Moreover, a specific DEA framework, the "input-oriented slacks based measure of efficiency" (SBM-I), was used to implement window analysis (Table A.1).

Table A.1. Legend of symbols used in the formulation of the model.

Symbol	Description	Symbol	Description
n	Number of DMUs	x_{kj}	Amount of input k consumed
t	Number of periods	y_j	Amount of output generated
p	Window length (p ≤ k)	0	Index of the DMU being
j	1,2,..., N (index on the DMU)	$(\lambda_{10},\lambda_{20},\ldots,\lambda_{N0})$	Linear coefficient vectors for
M	Number of different inputs	σ_{k0}	Slack in the consumption of
k	1,2,...,M (index of inputs	Φ_0	Efficiency score for DMU 0

The formula in order to calculate the total number of DMUs is as follows (Charnes and Cooper, 1991):

$$n(t - p + 1)p \qquad \text{[Eq. A.1]}$$

Furthermore, the SBM-I formulation used for window analysis is presented below:

$$\Phi_0 = \mathrm{Min}\left(1 - \frac{1}{M}\sum_{k=1}^{M}\frac{\sigma_{k0}}{x_{k0}}\right)$$

subject to

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

$$\sum_{j=1}^{N}\lambda_{j0}y_{j0} = y_0 \quad \forall\, k$$

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

The objective function of this model is non-linear, even though it is easily linearized. It represents the average reduction in the inputs consumed by DMU 0. This model seeks feasible operating points that consume fewer inputs as compared to the current units (DMU 0), without reducing output levels. If this objective is accomplished, then: $\Phi_0<1$. On the contrary, if it is not viable to reduce the consumption of any input without output loss, then $\Phi_0 = 1$ (because $\sigma_{k0} = 0\ \forall\, k$). In the latter case DMU 0 is said to be efficient.

4.3. CONTRIBUCIÓN III

Green Peas becoming greener: timeline environmental impact assessment of Swedish frozen green peas

Saioa Ramos[a], Anna Aronsson[b], Ulf Sonesson[b] and Jaime Zufia[a]

[a] Food Research Division, AZTI-Tecnalia, Bizkaia Technology Park, Derio, Spain.

sramos@azti.es & jzufia@azti.es

[b] Sustainable Food Production, SIK, Frans Perssons väg 6, Gothenburg, Sweden,

anna.aronsson@sik.se; ulf.sonesson@sik.se

*Corresponding author: Saioa Ramos. Email: sramos@azti.es

Telephone number: 0034-667174357. Fax number: 0034-946572555

Abstract

The goal of this study was (i) to assess the environmental sustainability of green peas by analysing the environmental impact related to the production and processing of frozen green peas over a time period of 25 years by analysing every fifth year between 1980 and 2005 and (ii) to identify opportunities for further reduction of emissions. LCA methodology with CML, USE-Tox and CED impact assessment were used to assess the environmental impact of green peas from cradle to retail. Over six studied years, the global warming potential impact of frozen green peas decreased almost 40%. The agricultural stage was the main contributor to the overall environmental impact but transport made a substantial contribution to ozone depletion and fossil depletion. Hot spots identified were fuel and pesticide use at the agricultural stage and efficiency in processing technology at the processing stage. Annual variation in weather and occurrence of pests played an important role for the differences in environmental impact between years, but this study demonstrates that management decisions also have a large impact on the life cycle environmental impact for green peas.

Keywords: legumes**,** agriculture, LCA, timeline, toxicity, freezing

4.3.1. Introduction

Grain legumes have long been an important protein source for mankind and animals (Van Kessel and Hartley, 2000). The green pea (Pisum sativum L.) is cultivated in many parts of the world and among most important legumes (FAOSTAT, 2012). This legume has significant nutritional and health advantages for consumers as it is a good source of vitamins, antioxidants (Ismail et al., 2009) and slowly digestible carbohydrate, fibre and vegetable protein (Sievenpiper et al., 2009).

Worldwide the production of green pea increased by an average of 12% per year between 1990 and 2010 with the main production countries being China, India and France (FAOSTAT, 2012). In Europe green peas are mainly produced in southern Scandinavia and along the west coast of northern Europe down to France because of a suitable temperate climate.

These legumes are usually consumed frozen, as frozen vegetables allow conserving taste, texture, and nutritional value in foods better than any other preservation method (FAO, 2005). In Sweden approximately 42800 t of green peas were harvested in 2011 (SJV, 2012) with at least 56% going to export as frozen peas to Europe; mainly Italy (Findus, 2012, personal communications).Peas, beans, carrots and a variety of cabbages (broccoli and cauliflower) have the largest share in consumption of frozen vegetables (CBI, 2009).Italy is the largest market for frozen vegetables accounting for 20% of the EU market value in 2008, followed closely by Germany with 19%.

One of the main environmental or economic benefits of cultivating legumes including the green pea is their N-fixating properties. Legumes access atmospheric N_2 through symbiosis with soil bacteria, collectively called rhizobia, and therefore require minimal nitrogen fertilizer inputs. When part of the fixated atmospheric nitrogen is made available to a subsequent crop, the use of legumes in a rotation can lead to a reduction in fertilizer-N use (Van Kessel and Hartley, 2000) which in turn can improve the sustainability of the crop production.

Life Cycle Assessment (LCA) methodology is increasingly being applied in the agri-food sector, being an objective and transparent methodology to quantify and assess environmental burdens of products and services (Audsley et al., 1997; Sonesson et al., 2010). The ISO standards (ISO-14040, 2006a; ISO-14044, 2006b) identify guidelines to be followed in an LCA study in order to guarantee this objectivity. However, biological systems are variable and LCA is usually a static tool that studies a steady state (de Haes, 2006). Recently, new LCA approaches with a timeline perspective have been carried out and can be useful to assess temporal changes in environmental impacts (Ramos et al., 2011; Vazquez-Rowe et al., 2012).

Previous LCA studies of pea production have shown environmental benefits compared to wheat at crop level (Charles and Nemecek, 2002) but some impacts have been higher for pea due to lower yields than wheat (Nemecek et al., 2005). However, the environmental impact of green pea from cradle-to-grave has not been well studied. In addition, fewer studies take into account the processing stage of the whole food product (Mila I Canals et al, 2008; Fuentes et al, 2006). UK-grown frozen broccoli and green beans were estimated to have a GWP of 2.64 and 1.74 kg CO2-eq/kg vegetable served on a plate (Mila-I-Canals et al., 2008).In those studies pesticide use and temporal variations were not included.

The main objective of this study is to perform a life cycle analysis of frozen green pea to evaluate the main environmental hot spots and improvement opportunities within the whole chain of production and distribution. A second objective was to examine how supply chain management changes have affected the environmental impacts over time by analysing six individual years of pea production.

4.3.2. Materials and Methods

4.3.2.1. Goal and scope definition

The goal of this study was to assess the environmental impact related to the production and processing of frozen green peas over a time period of 25 years by analysing every fifth year between 1980 and 2005 (in total 6 years). Attributional LCA methodology was used. An LCA quantifies the environmental impacts of a given product or process by accounting for all resources used in the process. An assessment over the full life cycle of a product is referred to as a cradle-to grave analysis for materials that are not recycled in the system (Owens, 1997). This study can be seen as a cradle-to-retail analysis as the assessment ends when the frozen pea product arrives to the retailer. The inclusion of several years of production means that efforts made to improve environmental sustainability of the product and variations in time influencing the environmental impact can be quantified.

For each year the selected functional unit (FU) was 1 bag of frozen green peas weighing 1 kg, which corresponds to 1.13-1.17 kg of harvested pea.

4.3.2.2. System description

The system boundaries included all activities from cradle (production of fertilizer and pesticides) to retail (Figure 1) excluding the consumer stage and waste handling. .

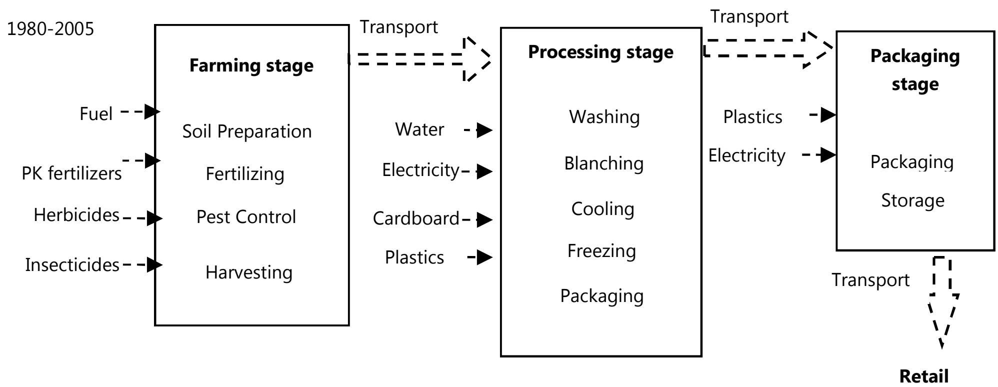

Figure 1: System definition including all the stages and inputs took into account. Thick arrow represents road or rail transport.

The studied system has been divided into three different phases: the agriculture stage where the farm operations and transport to the processing facility is included and the processing stage, where blanching, freezing and packaging are included and finally the distribution stage where transportation of the product to Europe is taken into account.

The Agriculture stage

Green peas require a relatively cool and humid climate, preferably in the temperature range of 7-30°C (table 1). In this case study peas for preservation are cultivated in Skåne and southern Halland in Sweden (Figure 2). Pea seed is drilled in spring after seed-bed preparation similar to cereals. The drilling date is adjusted for different cultivation sites so that the crop matures gradually from site to site which simplifies harvesting which takes place in July-August.

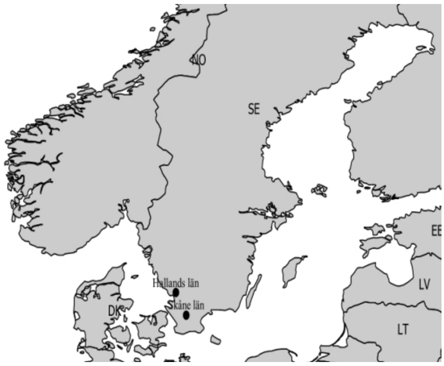

Figure 2: The location where green peas were collected from production in Halland and Skåne in southern Sweden.

Table 1: Summarized average monthly temperatures and rainfall data for the cultivation areas of Green pea.

	Skåne and Halland area					
	T (°C)			P (mm)		
	June	July	August	June	July	August
1980	N/A	N/A	N/A	N/A	N/A	N/A
1985	13.9	16.4	15.7	65.7	74.4	104.6
1990	15.9	16.3	17.3	58.1	38.2	135.7
1995	14.7	18	18.6	66.4	36.4	24.1
2000	12.8	14.5	15.6	47.3	126.8	61.7
2005	14.6	18.3	16.0	67.4	110.7	68.6

During the period of study two important changes in farm management occurred that possibly affected the environmental impact on a product level. Weeds are a significant problem, especially creeping thistle (*Cirsium arvense*), wild chamomile (*Tripleurospermum perforatum)* and black nightshade (*Solanum nigrum*). Pest and weed control is importantto sustain yields and the farmers made a transition from conventional pesticide management to integrated pest management using e.g. pheromone traps to reduce the amount of pesticide and herbicide used. At the same time a method to identify pea root rot (combinations of different common fungi such as (*Pythium ultimum, Fusarium solani* f. sp. *pisi,* and *Rhizoctonia solani)* was developed and implemented resulting in reduced disease incidence which increased yields.

The average soil was estimated to have a P-class of III and K-class of III. Phosphorus and potassium fertilizer was applied according to recommendations from agronomists at the Agricultural society in Malmö, Southern Sweden, at the following levels: 15 kg P/ha and 75 kg K/ha in 1980 and to 10 kg P/ha and 40 kg K/ha in 1985- 2005.

Growing peas as a preceding crop for both grain and rape seed in the crop rotation helps creating diversity in the agricultural system and to use N-resources in an efficient way. It has been shown that winter wheat yields can increase. In this study we assume that the extra wheat generated replaces the need for N fertilization in the following crops. Apart from increasing the yield, peas can also reduce the need for pesticides in the subsequent crop. This is however not taken into account in this study.

The Processing stage

After harvest peas are transported in trucks, to the processing plant in Bjuv, Southern Sweden. At the processing plant peas are washed and debris from the field is removed. The peas are then transported in water to the grading section where large, medium and small peas are separated, followed by blanching of the peas for 1 min at 90°C. After blanching, peas are cooled with cold water to 6-7°C. The water is drained from the peas and frozen at -30°C for 7 min in a flo-freezer. A flo-freezer is a container where the bottom is drilled with small holes in a specific pattern, cooling elements and fans. Cold air is circulated through the chamber and when the peas meet the cold air they move around, hence each pea is frozen individually. The water used in the system is reused several times.

After the freezing peas pass metal detectors, and then are weighed and packed in big containers of thick cardboard with an optional inner polyethylene bag and normally measures 120 × 120cm and in height 200cm, with a capacity of approximately 1000kg.

The frozen peas are transported to a distribution centre where they are packaged in 1 kg bags made of 30g plastic mix (90% PE and 5% Bisphenol A) before further distribution to retail.

Transport to retail

The frozen peas are transported from the distribution centre's freeze storage to retailers. Two-thirds of the product is exported to Europe; Italy being the most important market (acquiring 50% of the products), followed by Spain and Germany. One third of the peas are sold in Sweden. The transport mode changed over time during the study. At first the whole distance was covered by refrigerated trucks but was from 1995 gradually replaced by refrigerated rail transport.

Data acquisition and methodology

Data regarding farm operations such as fuel, pesticide and fertilizer use were taken directly from a series of surveys to farmers. Emissions related to those inputs are mainly taken from Ecoinvent (Nemecek and Kagi, 2007; Rouchette and Janzen, 2005; IPCC, 2006) or from the PestLCI model in case of pesticides (Stenemo et al, 2005).

Direct N_2O emissions were estimated according to IPCC (2006) and Rouchette and Janzen (2005). Indirect N2O emissions to air were estimated according to IPCC (2006). Nitrate e leaching to water was calculated according to Aronsson and Torstensson (2004) and the computer software "STANK IN MIND" (Stank in Mind, 2004) adapted to Swedish conditions and was dependent on the subsequent crop, see Table 2.

Table 2: Estimated N-leaching from field per year. WOSR= winter oilseed rape, WW= winter wheat.

Year	N leaching	Subsequent crop	
	kg N/ha*year	WOSR (%)	WW (%)
1980	52	80	20
1985	52	80	20
1990	52	80	20
1995	57	30	70
2000	58	20	80
2005	59	10	90

For direct emissions, N_2O from decomposition of crop residues and from the root exudates were included. Direct N_2O emissions from ammonia denitrification were not taken into account due to the fact that they mainly derive from application of mineral N fertilizers. Also N_2O emissions from biological N fixation (BNF) itself were is not accounted for as several studies have reported that the denitrification ratio for the symbiotic bacterium growing with legumes is not well understood and do not differ significantly from natural soil denitrification (Rochette and Hanzen, 2005; Garcia-Plazaola et al., 1993). Carbon emissions from the decomposition of crop wastes are not taken into account according to Nemecek & Kägi, (2007).

For the processing, storage and packaging direct data from an anonymous producer were recorded through direct interview and questionnaires except for water use which was taken from Mila I Canals et al., 2008.

For transport inventory data, an independent and intermodal refrigeration unit with capacity of 22 tons of product was modelled. The cooling container had a diesel consumption of 3l/h (Emanuelsson et al, 2010). and a capacity of 6,5kg of refrigerants (mainly R143a) with a ratio of leakage of 5-10%. For the allocation of refrigerants, 2500h working hours each year was recorded.

Background data associated with the production of fuel, pesticides or fertilizer were taken from the Ecoinvent 2.2 ® database. In the Table 3 data sources for the study are shown.

Table 3: Data sources for the inventory of green peas farming, processing and distribution

Inputs from the environment		Data Source
Agriculture		
	Land Occupation	Average data
	Fuel consumption	Average data
	Pesticides	Average data
	Fertilizer	Recommendation
	Avoided Fertilizer	Average data
Processing		
	Water (Blanching)	Mila I Canals et al., 2008.
	Electricity (freezing)	Average data
	Plastics (packaging)	Average data
	Transport	Emanuelsson et al., 2010
Output to the technosphere		
	Green Peas	Direct data
Outputs to the environment		
To the air		
	Air emissions from the fuel combustion	Nemecek and Kägi, 2007
	Pesticides emissions	PestLCI; Stenemo et al, 2005
	N_2O direct and indirect emissions	Rouchette & Janzen, 2005; IPCC, 2006
To the water		
	P emissions from fertilizers	Rapport 5823 Naturvardsverket
	N leaching	Aronsson and Torstensson, 2004
	Pesticides emissions	PestLCI; Stenemo et al, 2005
To the soil		
	Heavy metals emissions from rubber erosion	Nemecek and Kägi, 2007
	Pesticides emissions	PestLCI; Stenemo et al, 2005

4.3.2.3. Life cycle inventory

For the agriculture stage fuel use for farming operations, P and K fertilizers, and the use of insecticides and herbicides have been taken into account for the six years form 1980-2005, namely: 1980, 1985, 1990, 1995, 2000 and 2005. As input at this stage two different origins of pea seeds were used; one transported from U.S.A and one Swedish seed type. Emissions to the atmosphere from fuel use, nitrogen emissions from green pea waste and phosphorus emissions from fertilizers have been taken into account as main outputs (Table 4a). For the processing stage, electricity and water consumption was accounted for. Plastic material needed for the packaging have been also included as well as the different transport modes, rail and road transport (Table 4b). Used technology for this stage has not changed significantly since 1980, thus no changes over time for processing data was accounted for in the LCI. Possible changes that may have been made should have been focused on improving efficiency. Consequently, it can be said that this stage is not as dependent on external factors as the agriculture is.

Table 4a: Summarized average inventory for agriculture referred to the functional unit 1kg of green peas

	1980	1985	1990	1995	2000	2005
Input						
Fuel Use (L)	42.21	40.40	49.04	40.07	38.48	37.31
Seeds SE (%)	5	5	10	15	20	34
Seeds USA (%)	95	95	90	85	80	66
P (Kg)	4.79	3.18	3.40	3.13	3.05	3.08
K (Kg)	23.95	12.71	13.58	12.52	12.21	12.30
Output to the water						
N Leach (g)	16.61	11.30	10.76	13.75	12.37	12.57
Output to the atmosphere						
Carbon dioxide (g)	142.67	83.49	91.71	96.13	76.44	72.91
Carbon Monoxide (g)	4.68	2.74	3.01	3.15	2.51	2.39
NO2-N emission (g)	22.34	13.07	14.36	15.05	11.97	11.42

Table 4b: Summarized average inventory for processing referred to the functional unit

	1980	1985	1990	1995	2000	2005
Input						
Electricity (kWh)	0.26	0.26	0.26	0.26	0.26	0.26
Water (L)	10.90	10.90	10.90	10.90	10.90	10.90
Plastic bag (LDPE + BA) (g)	30.00	30.00	30.00	30.00	30.00	30.00
Rail transport (kgkm)	0	227	445	682	1140	2270
Road Transport (kgkm)	2270	2050	1820	1590	1140	0
Output to the atmosphere						
Carbon dioxide (g)	312.54	286.91	261.27	235.63	184.36	56.18
Carbon monoxide (g)	0.34	0.34	0.33	0.32	0.31	0.27
HFC-134a	1.03E-02	1.03E-02	1.03E-02	1.03E-02	1.03E-02	1.03E-02
HFC-143a	9.21E-02	8.31E-02	7.40E-02	6.49E-02	4.67E-02	1.18E-03
HFC-125	8.71E-03	8.71E-03	8.71E-03	8.71E-03	8.71E-03	8.71E-03

4.3.2.4. Impact category selection

Impact category selection in the assessment was based on known key impacts from agricultural production (Garnett, 2011), food processing (Kramer, 2003) and transport (Sim et al., 2007) and were as follows: abiotic depletion potential (ADP), acidification potential (AP), eutrophication potential (EP), global warming potential (GWP) and Ozone layer depletion potential (ODP). The impact assessment used was CML baseline 2000.

As there is an on-going debate on the suitability of the existing impact assessment methodologies for ecotoxicity we made a screening between different available toxicity methodologies in order to select the one which suited our product best (Table 5): the USE-Tox toxicity model (Rosenbaun et al., 2008), ReCiPe, CML baseline 2000 (Heijungs et al., 1992) and EcoIndicator 99 were evaluated. After the screening, it was decided to use the USE-Tox methodology, due to the fact that it had the most pesticide compounds included. The impact categories human toxicity and ecotoxicity were chosen for the USE-tox evaluation. Simapro 7.3 (PRé-Consultants, 2011) was used for implementing the impact assessment.

Additionally, Cumulative Energy Demand, CED, (Frischknecht et al., 2004) was chosen as an energy flow indicator, since energy consumption is one of the most relevant aspect food industries have to face.

Table 5: USE-tox, ReCiPe and CML **t**oxicity methodologies comparison based on the appearance of applied active substances.

	USEtox			ReCiPe				CML			
	HTc	HTnc	ET	HT	TE	FwE	ME	HT	TE	FwE	ME
Cyanazine	swa		swa	s	s	s	s	swa	swa	swa	swa
Benthazone		swa	swa	s	s	s	s	swa	swa	swa	swa
MCPA			swa	s	s	s	s	swa	swa	swa	swa
Pendimetalin		swa	swa	s	s	s	s				
Metribuzine		swa	swa	s	s	s	s				
Pirimicarb		swa	swa	s	s	s	s	swa	swa	swa	swa
Deltametrin		swa	swa	s	s	s	s	swa	swa	swa	swa
Tau-Fluvalin.											
Aclonifen			swa	s	s	s	s				

HTc = Human Toxicity, cancer s = output to the soil
HTnc = Human toxicity, non-cancer w = output to water
ET= ecotoxicity a = air emissions
HT = human Toxicity
TE = Terrestial Ecotoxicity
FwE = Fresh Water Ecotoxocity
ME = Marine Ecotoxicity
Car. = Carcinogenics

4.3.3. Results and discussion

Overall, one kg of frozen peas from cradle to retail shows a substantial reduction for all impact categories over the studied years. In general, the year with the greatest impact was 1980 and the year with lowest impact was 2005. The eutrophication potential is the only impact category where there has been no reduction of emissions over time if the year of 1980 is disregarded.

LCA studies of processed food products generally shows that the agricultural stage is the main contributor to the whole environmental impact assessment. In this study, the agricultural stage contributed an average of 30% to the Global Warming Potential, 90% to the Eutrophication Potential and up to 50% in the other characterization impacts (Figure 3).

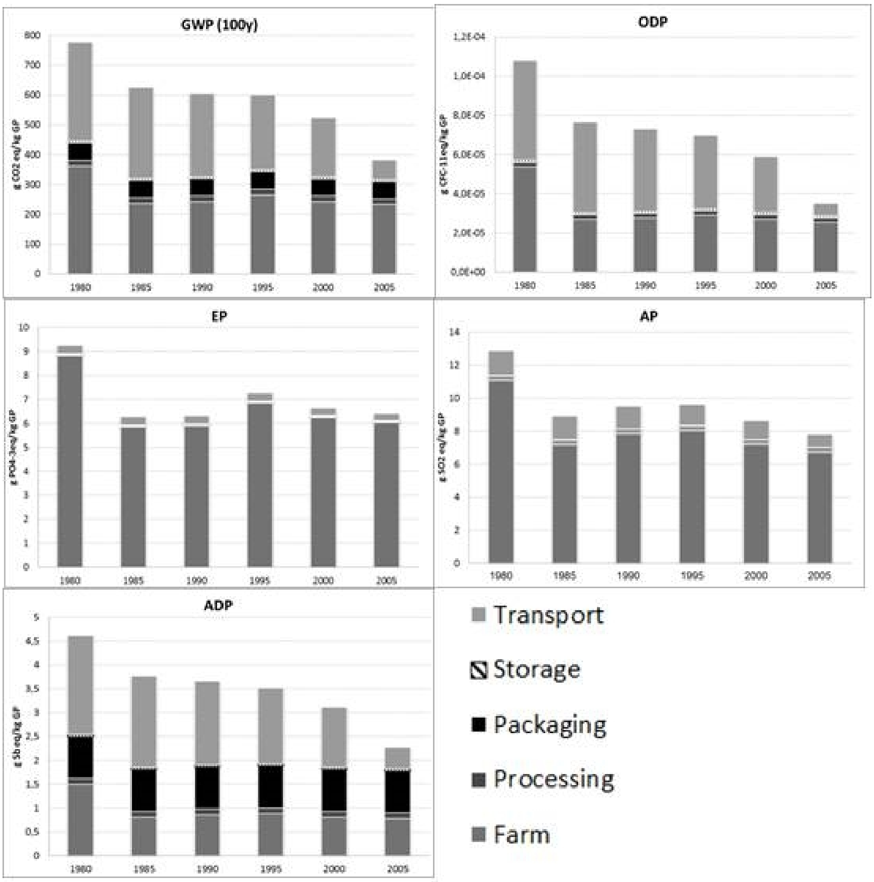

Figure 3: Full LCIA characterisation profile for Swedish-grown green peas, from cradle to retail per kg frozen green pea

In the following sections the e main environmental impacts, the main life cycle stages, energy consumption and transport are described and discussed.

4.3.3.1. Impact assessment of frozen green peas

Global Warming Potential

From 1980 to 2005 the global warming potential impact decreased with almost 40%, from 396g CO2eq/kg frozen green pea (GP) to 297 g CO_2 eq/kg. The main reason may be the decrease in fuel use for farm operations, from 132L/ha in 1980 to 120L/ha in 2005.

However the fuel use for farming only accounts for 40% (average between all the years) of the total global warming potential. Another 40% of the total GWP is mostly due to the N_2O emitted to the atmosphere from the denitrification processes made by soil bacteria during the decomposition of the crop wastes after harvesting. The N_2O emissions have slowly increased over time, due to the higher yields. The N_2O emissions are dependent on the crop yields and on the nitrogen absorption capacity of the subsequent crop. In 1990 and 1995 even without much productivity (t/ ha), the subsequent crop was mostly winter rape, which adsorbs more nitrogen than winter wheat and, hence, the leakage is not so high. In 2000 and 2005 winter wheat was the dominating following crop, so there were more emissions of N_2O per hectare, however in those years the yields was higher and thus the N_2O emissions per product weight were lower than in 1995.

The year 1980 shows the greatest impact compared to all the selected years, mainly due to low productivity values. The peak in almost all the impacts observed in 1995 relates to the increase in the N_2O emissions to the atmosphere. The remaining 20% of the total GWP is due to the transport of seeds.

Eutrophication and Nutrient Management

Following the same tendency as the GWP, the eutrophication potential has decreased more than 35% since 1980 to 2005. However, the reduction has not been gradual: in 1985 and 1990 impact values are reduced to 65 and 60% respectively, and then in 1995 there is a peak rising again to 70% of the impact. In the years 2000 and 2005, the impact falls back to 65% compared to 1980.

The main reason is the differences between N_2O emissions during the selected time, as nitrogen together with phosphor emissions are the most relevant factors when regards to the eutrophication potential (Gallego, 2008).

4.3.3.2. Impact Assessment of Green Peas cultivation

Regarding the timeline evolution of the pea cultivation the results suggest an improvement in all of the selected impact categories from 1980 to 2005 (Figure 4).

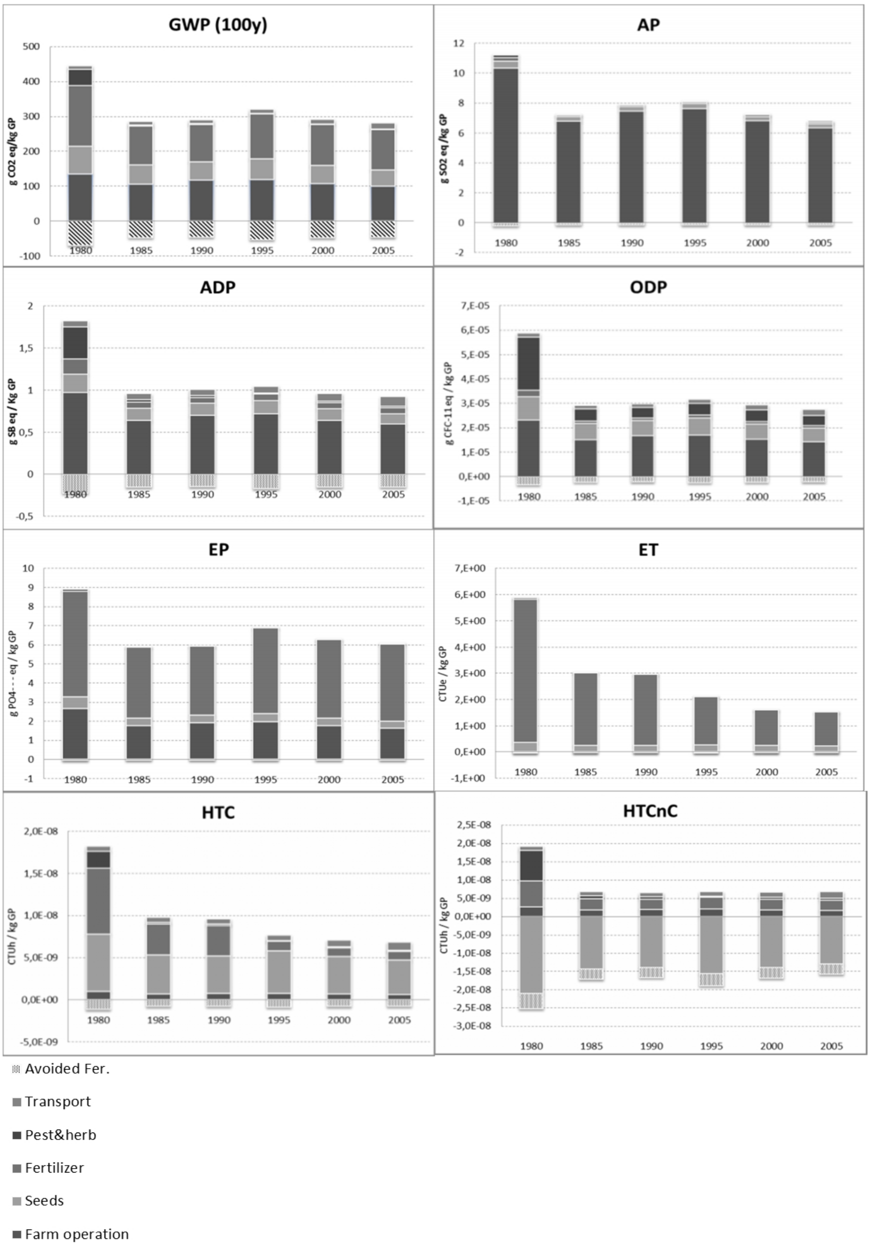

Figure 4: LCIA characterisation profile for the agricultural stage for Swedish-grown green peas per kg of green pea, from 1980 to 2005

On the whole, 1980 was, the year with highest environmental impact of all impact categories; the other years have a 60% less impact compared to 1980. The summer of 1980 was in Sweden cold and rainy resulting in low yields which may be part of the reason of a higher environmental impact per kg of product. In addition, differences between other years (1985, 1990, 1995, 2000 and 2005) are less significant, with not more than a 10% difference in environmental impact for all of the categories. Hence, it could be assumed that there are no relevant differences. The general trend of reduced environmental impact over the selected period is broken by the years 2000 and 2005 where the impacts increase (Table 6). Only in 2005 there is a noticeable drop in all impact categories of around 20% comparing to 1995.

Table 6: Averages, standard deviation and percentage of maximum and minimum variation (vari.) between the years on each of selected categories from CML methodology per kg of green peas

	Abiotic Depletion Potential	Acidification Potential	Eutrophication Potential	Global Warming Potential (100y)	Ozone Layer Deletion Potential
Unit	g SB eq	g SO2 eq	g PO4--- eq	g CO2 eq	g CFC-11 eq
Average	0.96	7.98	6.63	266.30	3.19E-05
σ	0.32	1.60	1.15	54.53	1.16E-05
% min vari.	45.16	27.84	22.53	29.02	47.56
% max vari.	51.54	39.70	31.81	37.81	54.27

Toxicity and pesticide uses

It can be observed how the reductions of pesticides greatly reduced the human- and ecotoxicity (table 7). As in the other impact categories described above, the year 1980 show the highest values in both categories. However, values for 1985 and 1990 are close to 50% of the impact of 1980. Over the following years (1995, 2000 and 2005) the values of the impact reduces again, reaching 40 % of the impact related to 1980.

The effort made by the farmers and the processing company in order to reduce the amount of insecticide and pesticide use has reduced the ecotoxicity impacts. However, these results have to be regarded with care since these impact categories are associated with a high uncertainty (van Zelm *et al.*, 2009).

In this study, heavy metal uptake by the crop was not included, but the LCI inventory for the pea seed from the Ecoinvent database includes plant uptake of copper and zinc. For the impact category of Non-Cancer Human Toxicity this results in negative impacts which can be misleading (table 7) as crop residues on the field will release heavy metals back into the soil after harvest.

Table 7: Results USE-Tox methodology impact categories, averages and standard deviation per kg of green peas

	Human toxicity, cancer (CTUh)	Human toxicity, non-cancer (CTUh)	Ecotoxicity (CTUe)
1980	1.25E-08	-1.6E-08	5.790203
1985	6.52E-09	-1.5E-08	2.982777
1990	6.35E-09	-1.4E-08	2.907425
1995	4.11E-09	-1.7E-08	2.064901
2000	3.69E-09	-1.5E-08	1.56222
2005	3.42E-09	-1.3E-08	1.489439
Average	6.1E-09	-1.5E-08	2.799494
σ	3.41E-09	1.12E-09	1.598518

4.3.3.3. Impact Assessment of processing

In general the energy consumption in the processing stage for food products is high (Pardo and Zufia, 2012), while other aspects such as impact associated to the water consumption or other material uses can play a minor part depending on process and product.

With respect to the energy consumption, CED was chosen in order to obtain a benchmarked list of efficient years (Figure 5). Following the same tendency as in other impact categories, year 1980 had the highest CED, and the impact has gradually reduced since 1980.

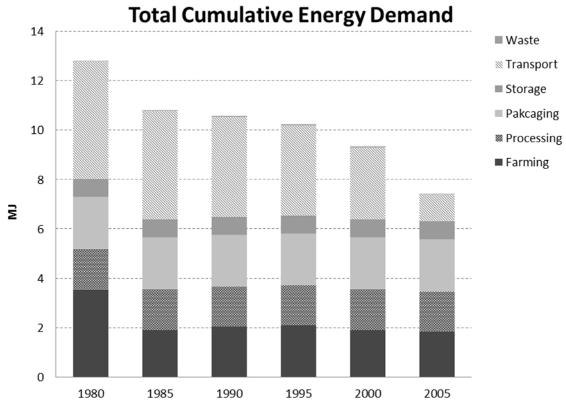

Figure 5: Cumulative energy demand in MJ per kg for frozen green peas from cradle to retail

In all the selected years the main problem of this aspect is the electricity consumed in the processing and fuel and electricity used in the transport stage. The other most influential aspect for energy consumption would be cold storage for 30 days. Note that all of the stages with high energy consumption have a great potential for reduction by reducing the storage time or by reducing transport distances.

4.3.3.4. Impact Assessment of transportation

Transportation of goods is a very important stage for the total environmental impact of food products. Comparing with 1980 all the selected impacts caused by transports have decreased significantly (Figure 6).

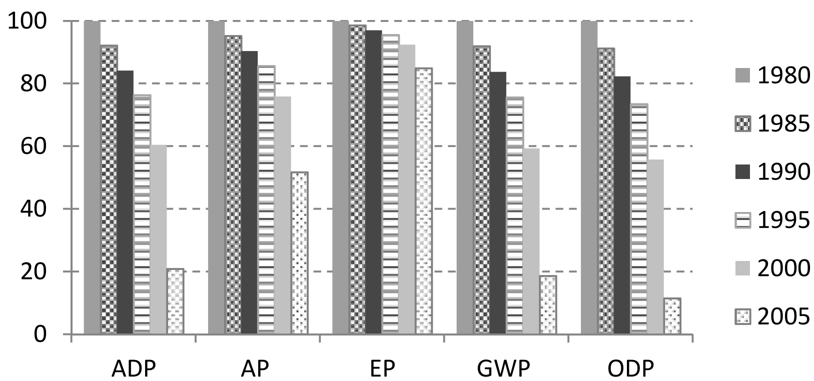

Figure 6: LCIA characterisation profile for transport to retail of frozen green pea from 1985 to 2005

The production company has made an important effort to replace road transport by train transport, which has led to a reduction of 60% of the GWP for all selected years. The reduction is all due to the substitution of fuel used by trucks to the electricity used by train. Also if the transport distances could be decreased an additional reduction of the carbon footprint could be achieved. This is however a market effect and reducing transport distances would mean that the peas need to be sold in markets closer to Sweden where prices and demands may be lower.

Ozone depletion potential (ODP) is used as a simple measure for quantifying the effect of various ozone-depleting compounds (Such as chlorofluorocarbons) on the ozone layer (Fisher et al., 1990). As refrigerants use decreased over time due to the change of transport mode the ODP also decreased significantly.

4.3.4. Conclusions and perspectives

Overall it can be concluded that decisions made along the supply chain from cradle to gate for frozen green peas have reduced the environmental impacts over time.

Although significant reductions were identified in all impact categories selected, specific hot-spot improvement options have been detected:

- Minimize fuel use in the agricultural stage
- Reduce pesticide use
- Reduce distances between production and consumption; reducing the gap between food production and their final destination is one of the current challenges facing the global food system.
- Substitute road transport by train transport; this study has confirmed that there are environmental benefits of replacing road transport with rail transport.
- Optimization of freezing technologies; in recent years new freezing technologies have emerged to make processes more efficient and/or to provide more value to the final product.
- Minimize storage time in freezers.

Annual variation in weather and occurrence of pests plays an important role in the differences of environmental impact between years for the cultivation of green peas, but it has been demonstrated that management decisions also have a large impact on the life cycle environmental impact. Hence management is an important aspect when making our food systems more sustainable.

Comparing with other LCA studies of frozen vegetable products suggest that Swedish frozen green peas has lower CED and GWP when retail and consumer stage is not included (Carlsson-Kanyama et al., 2003; Mila I Canals et al., 2008). Frozen green beans from the UK were found to have a global warming potential of 1.72 kg CO2eq/kg beans on a plate including waste handling and humane excretion (Mila I Canals et al., 2008). Dried and canned yellow- and chick peas and pinto beans cultivated in or imported to Sweden had a carbon footprint ranging from 0.2 to 1.4 kg CO2eq/kg product depending on differences in cultivation, processing and transport stages (Fuentes et al., 2006). It is however difficult and not appropriate to directly compare these products as the quantification methodology may be different.

4.3.5. Acknowledgements

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4.3.6. References

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4.4. CONTRIBUCIÓN IV

Environmental improvement of a chicken product through life cycle assessment methodology

Saioa Ramos[a*], Guillermo Pardo[a,b], Jaime Zufía[a]

[a] Food Research Division, AZTI-Tecnalia, Parque Tecnológico de Bizkaia, Edif. 609 - 48160, Derio (Spain).

[b] Current address: BC3. Basque Centre for Climate Change, Alameda Urquijo 4 , 4ª 48008 Bilbao (Spain)

*Corresponding author: Saioa Ramos. Email: sramos@azti.es

Telephone number: 0034-667174357. Fax number: 0034-946572555

Abstract

Life Cycle Assessment (LCA) methodology was applied to a chicken product in order to identify the more relevant sources contributing to environmental impact. A number of potential improvements were proposed and developed at different stages of the food chain, based on a technical and environmental point of view, with special focus on the application of innovative solutions. Improved food chain was also evaluated through LCA in order to compare results. Considerable impact reductions were achieved in the feed production stage by replacing conventional ingredients with tomato by-products in the poultry diet, (freshwater eutrophication and global warming potential decreased 7% and 6% respectively). Water consumption and wastewater generation was minimized about 16% in the slaughterhouse through reutilization alternatives and pulsed light decontamination. Conventional plastic tray and film were replaced with biodegradable materials avoiding up to 20% of greenhouse gases emissions. Finally product distribution was also optimized by improving refrigerated storage and logistics achieving relevant reductions in electrical and diesel consumption estimated around 10-15%.

Keywords: Poultry, Broiler, Life Cycle Assessment, Ecodesign

4.4.1. Introduction

Broiler production industry is an important component of food sector in Spain. Chicken, together with pork, are traditionally the most demanded types of meat within the Spanish market accounting up to 80% of the total share of meat consumed. According to FAO (2012) domestic supply rose to 1.26 million tons in 2009 accounting one of the highest consumption ratios of chicken meat in the UE (around 27 kg/person/year) (Magdelaine et al., 2008). As can be seen in figure 1 during the last decade consumption trends reveal a diminution in pig meat demand together with an slight increase for poultry meat. Due to its good quality for nutritional and dietary purposes and its affordable price, future prospects predict that this trend could continue for the next decade so a further growth in poultry meat production can be expected (Horne, 2008).

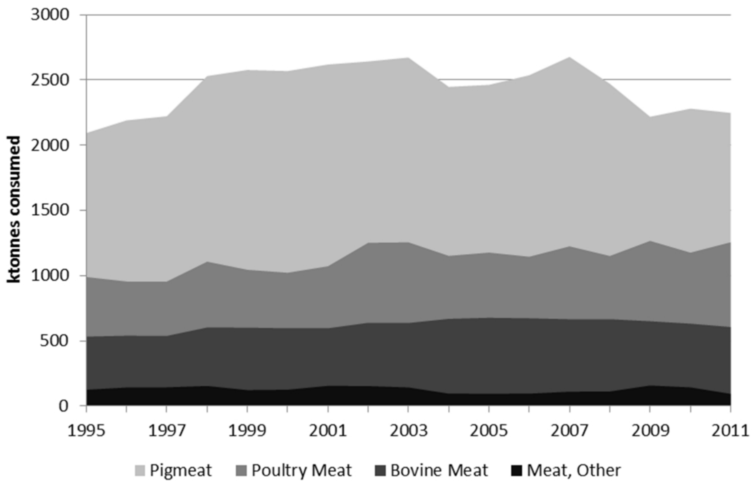

Figure 1. Evolution in consumption of different types of meat in Spain (2000–2009).

On the other hand, nowadays increasing attention is being paid to the environmental burdens associated to the food supply chain, and specifically to livestock production since its global importance has been pointed out by certain studies. Among others, "Environmental Impact of Products" (EIPRO) study (Tukker et al., 2006) showed that food and drink sector was responsible for 20 to 30 % of the environmental impact of private consumption in the EU, identifying meat and dairy products as the items with the greatest contribution in this area. Also Food and Agriculture Organisation (FAO) report "Livestock long shadow: environmental issues and options" (Steinfeld et al., 2006) remarked livestock production as a major contributor to the world's environmental problems, accounting about 18% of global anthropogenic greenhouse gas (GHG) emissions.

Other relevant authors have also pointed out the environmental issues related to meat production with special regard to acceptable levels of nitrogen and phosphorus in the effluents, or the contribution of the sector to the global emission of greenhouse gases (Galloway et al., 2010; Garnett, 2007; Steinfeld and Wassenaar, 2007; Sutton et al., 2011). In short term, increasing concern about these burdens may lead to more restrictive regulations which could involve a rise in costs within the poultry sector in order to adapt the production systems to these circumstances.

Hence recent studies have proposed the implementation of different measures as a way to produce more efficient and sustainable chicken products. IMPRO project (Weidema et al., 2008) identified options at three main areas: i) involving agricultural actions to reduce water and air emissions, ii) power savings at different stages and iii) meal planning tools to reduce wastage at household level. Furthermore, some authors have also pointed out strategies to reduce the impact at farm level, by focusing on crop cultivation, poultry diets or improvement options at housing facilities (Leinonen et al., 2012; Nguyen et al., 2012; Seguin et al., 2011) whereas Davis and Sonesson (2007) investigated the influence of various measures to increase efficiency at post-farm stages.

With special attention to the application of innovative approaches, the aim of the case-study was to select and develop a number of improvement measures to be implemented at different stages of the chicken supply chain. Their feasibility at pilot and industrial scale was evaluated, and their potential effect on the overall environmental impact of the product was also analysed through life cycle assessment (LCA) methodology in order to quantify the impact reduction achieved in every scenario.

4.4.2. Materials and methods

4.4.2.1. Life Cycle Assessment

LCA has become an important tool for environmental evaluation of production chains, allowing the identification of those parts of the life cycle where the greatest improvements can be made. According to international standards (ISO 14040:2006; ISO 14044:2006) this methodology involves a four-stage procedure: definition of the goal and scope, life cycle inventory (LCI), life cycle impact assessment (LCIA) and interpretation. In the first step, a functional unit (FU), to which all inputs and outputs are related, system boundaries and allocation strategies (partitioning of inputs and outputs) are defined, depending on the subject and intended use of the study. Through LCI phase all the inputs and outputs of the system under study are identified and quantified. Finally, results from LCI are related to environmental impact categories and indicators through the LCIA, and interpreted in relation to the objectives of the study.

4.4.2.2. Goal and scope definition

The final goal of the present study was to apply LCA methodology to minimise the environmental impact associated to a meat-based product, allowing the identification of critical stages along the life cycle and comparison between potential improvement scenarios. From a multidisciplinary approach, four different research centers were involved in the project to investigate innovative technologies and processes in their respective areas of expertise under the common objective of reducing the environmental impact of a chicken based product along various stages of the supply chain (Figure 2).

The product chosen for the study was a tray of sliced chicken breast packaged in modified atmosphere, which is a commonly commercialized item in the Spanish market. The most significant impact sources were identified using life cycle assessment (LCA) methodology and several improvement options were selected at different life cycles stages, according to their potential to reduce the environmental load of the product and their suitability from a technical point of view. The following improvements were proposed to be explored:

- Processing of food industry by-products as valuable ingredients for animal feed.

- Pulsed light technology for water decontamination and re-use in food sector

- Development of a biodegradable packaging for meat products

- Energy and resource savings in the distribution system by means of storage efficiency, routes optimisation and reverse logistics options.

Proposed improvements were studied and developed at each research institution involved in the project, according to their field of knowledge, and feasibility was evaluated for their implementation to the industry. Finally LCA methodology was applied in order to estimate the environmental impact of the new food chain and to compare results with the original scenario.

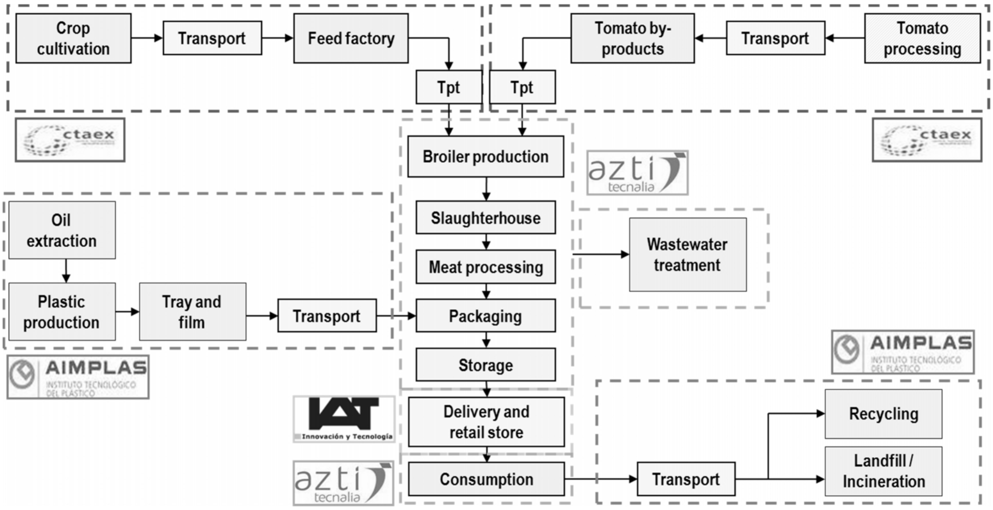

Figure 2. Overview of the life cycle of the chicken product and expertise areas of every partner. (Tpt.=Transport)

Functional unit and system boundaries

Selected functional unit was a 0.6 kg tray of sliced chicken breast packaged in modified atmosphere. The study ranged all the stages of the life cycle of the product from cradle to grave. Therefore, activities involved in the agricultural phase, including crop cultivation for feed production, poultry housing and production and consumption of supply materials (diesel, fertilizers) were considered but also the post-farm stages of product's life cycle, such as slaughtering, food packaging, distribution, retailing, consumption at household level and packaging disposal.

The production and maintenance of infrastructure and capital goods (buildings and machinery) were excluded of the analysis, although the occupied land of these facilities was taken into account.

Allocation strategies

In the multi-output subsystems involved along the LCA it was necessary to distribute environmental burdens among the multiple co-products obtained through the analysed process. The principal objetive of the analysed system was to produce chicken meat for human consumption. Therefore, environmental impacts embeded from agricultural and slaughtering stages were assigned entirely to the meat products and not to other low value by-products obtained from poultry processing (e.g. blood, livers, lungs, feathers, etc...).

In order to assign the proportional impacts to the selected item (chicken breast) among the others meat co-products, an allocation procedure based on a physical criteria was considered adequate. Given that the different meat products derive from the same system and slaughtering process, a mass allocation procedure was applied by accounting the share in quantity of different poultry meat co-products from overall carcass weight. Mass allocation is also recomended when system expansion is not possible by the ISO 14040 and the ENVIFOOD protocol (2012) developed by the European Food Sustainable Consumption and Production (SCP) Round Table. Table 1 shows the allocation factors used. As can be observed, breast had a high allocation factor which pointed out that a relevant amount of the environmental burden related to the broiler production is ascribed to the analysed product.

Mass allocation procedure was also performed at distribution, retailing and household stages to account for the energy consumption and refrigerant leakages attributed to the target product, according to the average capacity of cold storage devices involved in the chain.

Table 1. Chicken co-products and derived mass allocation factors.

Co-Product	Weight (g)	Allocation factor
Breast	480	0.32
Drumsticks	240	0.16
Thighs	500	0.34
Wings	267	0.18
Total (carcass)	1487	1.00

4.4.2.3. Impact category selection

Environmental impacts associated with chicken life cycle were estimated using the ILCD 2011 method (JRC, 2011). Based on the previously mentioned ENVIFOOD protocol (Food SCP Rt, 2013), a number of impact categories were selected taking into account their relevance in the food processing: acidification potential (AP), Fresh water eutrophication potential (FE), global warming potential (GWP), ozone depletion potential (ODP), land use (LU) and Water Depletion Potential (WD)..

4.4.2.4. Life Cycle Inventory

In order to describe the system under study (Fig. 2) a detailed data gathering step was required. Life cycle was divided in main stages and the research institutions involved in the project were assigned to collect data from different phases according to their area of expertise.

Most of the data for inventory were acquired from companies directly involved in the project. Four broilers production farms and two poultry slaughterhouses were analysed during the study. Primary data with regards to water and energy consumption, packaging, fodder use and related outputs was collected through questionnaires, meetings and telephone conversations.

When complementary information was required, additional contacts in food industry were consulted (e.g. technology suppliers) as well as bibliographical sources such as peer-reviewed literature, sectorial reports and government statistical publications. Finally main background data to complete the life cycle inventory such as transportation, or production of fuels, plastics and fertilizers, among others, were obtained from existing databases, mainly ecoinvent v2.0 (Ecoinvent Centre, 2009). Moreover, electricity was modelled updating ecoinvent electricity production mix to average Spanish grid in 2009 based on data from the Spanish Government (REE, 2009).

Feed ingredients

Feed formulation for standard broiler production was defined based on expert information. Mean value of starter and finisher diets was estimated. Maize, soybean, wheat, sunflower meal and palm oil were identified as the main components in the formulation of the poultry fodder.

Most likely source countries for the production of fodder ingredients were identified and transport burdens were accounted according to aproximate distances (see Table 2). Wheat, sunflower and maize were assumed to be produced in Spain, soybean meal in Brazil and palm oil in Malaysia. Inventory data for the production of the feed ingredients were obtained from the ecoinvent database (Jungbluth et al., 2007; Nemecek and Kägi, 2007) involving field work activities, fertilizers, pesticides, and direct field emissions related to agricultural step, as well as the main inputs and outputs from the transformation process.

The amount of land ocuppied in the agricultural stages was quantified from the specific crop yield, based on the area required to produce 1 kg of product (FAOSTAT, 2012). It was considered that all crops require the use of land for 1 whole year, as proposed by (Milà i Canals et al., 2012). For the case of soybean and palm fruit production in Brazil and Malaysia respectively, the equivalent carbon dioxide (CO_2) emissions related to the land transformation from tropical rainforest were accounted according to Jungbluth et al. (2007).

Table 2. Proportion in poultry feed, yields and main country of origin of crops considered in this study as ingredients of a standard broiler diet.

Components	Proportion (%)	Crop yield (kg/ha·year)	Transport (km)	Source country
Maize	40.0	9886 [a]	150	Spain
Soybean	31.0	2719 [a]	8000	Brazil
Wheat	16.4	2541 [a]	150	Spain
Sunflower seed	6.0	1018 [a]	150	Spain
Palm oil	3.0	21120 [a]	12000	Malaysia
Supplements [b]	3.6	-	-	Various

[a] FAOSTAT (2012) [b] include minerals, vitamins and amino acids

Broiler production

Table 3 presents the main technical indicators of the broiler production system considered in this study.

Table 3. Average production and food intake figures of broiler production system considered in this case study

Parameters	Units	Value
Production cycle	days	44
Average Live Weight	kg	2.2
Average mortality	%	3.6
Feed intake	kg/bird	4.4
Feed conversion ratio		1.9
Stocking density	birds/m^2	20.8

Data for the inventory of this stage were provided by three consulted farms. The average values obtained are shown in table 4 and they were considered to be representative of overall Spanish broiler industry since are in accordance with consulted bibliography on governmental statistics and sectorial reports (MAPA, 2006).

Direct emissions of ammonia (NH_3), nitrous oxide (N_2O) and methane (CH_4) from housing and poultry manure application were estimated based on country specific guidelines and measurements reported in previous studies (Calvet et al., 2011; EMEP-CORINAIR, 2007; EPER, 2007). Nitrate and phosphate leaching from land application of manure were computed according to Williams et al., (2006). Additionally the benefit from nutrient recycling due to the agronomical use of poultry manure was accounted as the equivalent production of mineral fertiliser avoided in terms of its nitrogen content.

Table 4. Inventory for production of 1 broiler (2.2kg live weight).

	Units	Value
Inputs		
Feed	kg	4.40
Electricity	MJ	0.15
Natural gas	MJ	2.67
Water	l	11.5
Bedding material[1]	kg	0.40
Outputs		
Emissions to the atmosphere		
NH_3 (housing)	g	20.0
N_2O (housing)	g	2.1
CH_4 (housing)	g	1.2
NH_3 (manure application)	g	21.7
N_2O (manure application)	g	0.5
CH_4 (manure application)	g	15.8
Emissions to water		
NO_3^- (manure application)	g	57.3
PO_4^{3-} (manure application)	g	3.1

Slaughtering, meat processing and packaging

Annual inventory data were obtained from two slaughterhouses located in Spain. Average allocated values per FU of chicken meat are shown in table 5, which includes the main inputs and outputs of the analyzed processing systems. Low value by-products, others than meat, obtained from poultry processing (e.g. blood, livers, lungs, feathers, etc...) are incorporated by different associated industries. Hence, environmental burdens related to these activities were not considered in the study.

Plastic tray and film were accounted following packaging details provided from a meat industry, as well as the gas mixture applied for shelf life extension purposes.

Table 5. Inventory of meat processing stage. Data per FU (0.6 kg of chicken breast)

Inputs		
	Units	Value
Processing		
Electricity	kWh	0.09
Natural gas	MJ	0.36
Water	l	4.3
Packaging		
Polystyrene *(tray)*	g	9.3
Polyethylene high density *(tray)*	g	1.1
Polyethylene high density *(film)*	g	3.4
Ethylene vinyl acetate *(film)*	g	0.9
Gases		
Carbon dioxide (liquid)	g	0.5
Oxygen (liquid)	g	0.3
Nitrogen (liquid)	g	0.3
Outputs		
Emissions to water		
N total	mg	4.3
P total	mg	0.8

Product delivery

Logistics of the selected product were modelled according to a realistic distribution system of a Spanish chicken company, involving 14 main delivery routes. Refrigerant leakages and additional energy required for refrigeration during transport were accounted following reviewed literature (Tassou et al, 2009).

The selected transport chain takes into account first transport between the production of broilers to the logistic platform (average distance of 744 km in 60 tn trailer) and a second delivery from this platform to different retailers (average distance of 279,36 Km in 7 Tn truck)

Retail store

Energy for cold storage of the product during retailing stage was also included in the study. Electricity consumption for cooling at stores was estimated using refrigeration equipment details and product throughputs of ten supermarkets within the distribution area.

Consumption stage

Consumer habits related to purchasing, storage period and conditions, cooking, washing and waste disposal were modelled according to the results revealed from a specific market research survey on chicken meat consumption involving 900 consumers interviewed (Zufía and Pardo, 2011).

The methodology employed in this consultation has been a quantitative. A semi-structured questionnaire *ad hoc* has been applied by personal telephone interviews. The specific characteristics of the market survey were:

- Universe: Buyer-consumer whole chicken breasts in plastic tray in Spain.
- Sample: Men and women over 18 years living in different Autonomous Communities in Spain which are responsible for buying and cooked whole chicken breasts in plastic tray regularly (at least once a month).
- Segmentation of the sample: Two variables have been differentiated:
    - Territory (unit: Autonomous Community)
    - Proportional to the number of residents over 18 years.
    - Type of home.
- Sample size and statistical justification:
    - Sample size n = 900
    - Confidence level of 95%
    - Sample e = + / Error - 3.3% 2s

Waste disposal

Based on data revealed from the market research survey, the amount of chicken meat spoiled or wasted after consumption period of the selected item (chicken breast fillets) was considered negligible. With regards to the packaging, 75% of the interviewed consumers admitted to dispose of the tray and film of the product in the recycling bin. However, according to experts consulted, current Spanish recycling system does not segregate this kind of multilayer materials so it was assumed that they are finally processed as a refuse of plastic recycling plant, conducted mainly to waste-to-energy facilities.

Proposed improvements

Three main strategies for environmental improvement were defined. First, tomato by-products (seeds and skins) were considered as a partial substitute of conventional ingredients included in poultry diet. As shown in Figure 3, in the alternative feed formulation the proportion of soybean was reduced whereas the other components were adjusted to keep adequate nutrient levels. However, tomato by-products inclusion was limited to a maximum of 3% due to its high fiber content. Energy for estabilisation of drying process of the tomato was measured on-site at CTAEX facilities.

Other improvement taken into account was to introduce Polylactide (PLA) based tray and film as alternative packaging materials for chicken meat products. An additional co-extruded poly-vinyl alcohol (PVOH) layer was included to provide the gas barrier proprieties of modified atmosphere packaging. Data of the production process for PLA were based on Althaus et al. (2007) whereas production of vinyl acetate copolymer was used as a proxy for PVOH according to literature reviewed (Humbert et al., 2009).

Furthermore, additional energy, water and material requirements related to other proposed improvements (e.g. pulsed light technology) were estimated mainly from prototypes and pilot scale trials carried out along the project and from contacts in industry and technology suppliers.

4.4.3. Results

4.4.3.1. Life cycle impact assessment results

As can be observed in table 6, among the relative contributions of the different stages involved in the chicken meat supply chain, the major impacts to the environment are caused during the feed production stage. Those impact are mainly related to the crop cultivation phase, followed by broiler production stage due to on-farm operations such as heating of broiler house or storage and spreading of poultry manure.

Moreover according to figure 3, Fodder production showed crucial impacts in most categories but especially for GWP (60 %), ODP (60 %), LU (90 %) and WDP (30 %). In terms of global warming, N_2O emissions appeared most important (34%), followed by CO_2 emissions from burning fossil fuels and transport (33%). Specifically soy used in the poultry feed has a particular influence in the analysed system since it is mainly imported from Brazil, which is associated to impacts linked to long distance transport and deforestation for grain production.

Table 6. Life Cycle Impact Assessment of the selected functional unit 0,6 kg of consumed chicken breast. GWP = global warming potential; ODP = Ozone Layer Depletion; AP = Acidification potential; FE = Freshwater eutrophication potential; WDP: Water resource Depletion; LU = Land use.

	Unit	Fodder	Farming	Proce.	Packaging	Distribution	Consum.	TOTAL
GWP	kg CO2 eq	1,30E+00	1,45E-01	9,55E-02	5,29E-02	2,41E-01	2,65E-01	2,10E+00
ODP	kg CFC-11 eq	9,81E-08	6,64E-09	6,72E-09	6,73E-10	3,71E-08	1,22E-08	1,61E-07
AP	molc H+ eq	1,23E-02	1,13E-01	5,54E-04	2,23E-04	1,19E-03	2,23E-03	1,30E-01
FEP	Kg P eq.	3,40E-04	1,12E-06	1,70E-05	1,56E-06	4,48E-06	1,75E-05	3,82E-04
LU	kg C deficit	7,55E+01	5,88E-02	7,34E-02	2,07E-02	5,87E-01	1,77E-01	7,64E+01
WDP	m3 water eq	1,66E-01	1,11E-02	6,22E-02	1,07E-02	6,93E-02	2,55E-01	5,74E-01

Environmental burdens related to broiler production phase were found to be the most relevant for AP (85 %) but also significant for other categories such as GWP (7%). Ammonia (NH_3) emissions from poultry manure handling and storage are predominantly responsible to the impacts on AP associated to this stage, but also nitrate leaching ocurred after land application which has particular influence on groundwater eutrophication. Additionally, other substances emitted from poultry manure decomposition process, mainly N_2O and CH_4, contribute notably to GWP together with indirect CO_2 emissions related with natural gas combusted for heating and electricity consumption for ventilation and lighting purposes.

Slaughtering and meat processing stage contributes moderately (around 5 to 10 %) to most of the impact categories analysed. It is important to high light potential impact on water depletion, around 15 % of the total WDP, due to the important amount of water required for cleaning operations in this kind of facilities.

With regards to the packaging, its main influence is related to GWP (< 5 %), mainly associated with the processes of oil extraction, plastic manufacturing and, to a lesser extent, waste disposal.

Product delivery and storage at retailer's store accounted together for 20 % of the total impact for ODP, and 10 % for GWP, which is attributed to emisions related to transport and energy consumption for refrigeration but also to refrigerant leakages during transport under cold temperature.

Impact associated to the consuptiom of the selected product at household stage, are mainly related to WDP (40 %) due to water consumption during the wahsing activities.

Moreover it's also accounts for GWP (5 %) mainly related to energy demanding processes, especifically natural gas and electricity for cooking and cold storage at fridge.

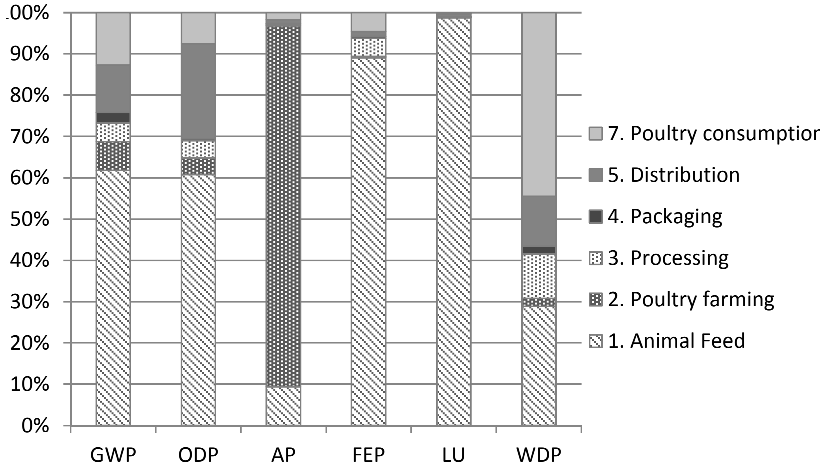

Figure 3. Main contributions from different stages to the environmental impact of the chicken product

4.4.4. Discussion

4.4.4.1. Hot-spots identification

According to the results of the present study, the feed production stage, and specifically the crop cultivation phase, is responsible of a most of the environmental impact associated with whole product chain, as previously highlighted by other authors (Alvarenga et al., 2012; Bengtsson and Seddon, 2013; González-García et al., 2014; Leinonen et al., 2012; Pelletier, 2008), whereas emissions from manure management have a strong contribution on certain categories such as AP or FE.

When considering in the analysis the whole chicken meat supply chain, the importance of other phases of the life cycle is revealed, since they can have a remarkable influence in terms of energy demand or gaseous emissions, contributing particularly to global warming and ozone layer depletion (Katajajuuri, 2009). In this study those stages has been taken into account, however, no remarkable impacts have been associated to those downstream stages. Only for water depletion potential impact household consumption has been relevant stage.

4.4.4.2. Improvement options identified

First of all, since feed production was identified as one of the major contributors to the environmental burdens of the chicken meat, the option of processing food industry by-products as valuable ingredients for poultry feed was explored. Thus tomato skins and seeds were dried and conditioned for its use as feed ingredients (up to 3%) allowing to reduce the amount of imported soy in the formulation.

In relation to water consumption at slaughterhouse, two main optimisation actions were identified. Firstly, direct reutilisation of water from washing boxes process to clean installations and transport lorries was pointed out. Secondly, internal water line from meat processing was selected for its potentially re-use in scalding after decontamination through pulsed light and additional clorination. Combining both options it was estimated that 18% of water consumption at this stage could be reduced.

Conventional tray and film were also ecodesigned. Increased use of plastic packaging has led to serious ecological problems due to their total non-biodegradability (Siracusa et al., 2008). Specifically food packaging materials are often contaminated by foodstuff and organic substances, making recycling impracticable and economically not convenient most of the times. As a consequence thousands of plastic materials from food items are either landfilled or incinerated every year, increasing the problem of municipal waste collection and disposal (Kirwan and Strawbridge, 2003). As a proposal to face this issue, a biodegradable packaging for the chicken item was studied, based on polylactic acid and ethylene vinyl alcohol materials.

Finally, another major stage investigated through the project was product delivery phase, leading to a series of improvement actions identified. Significant reductions related to electrical and diesel consumption were estimated around 10-15% at this phase by means of improved refrigerated storage, distribution routes optimisation and reverse logistics solutions.

4.4.4.3. Compared LCA results based on potential improvements

Significant impact reductions were achieved through the improvements identified and developed at different stages of the life cycle, as can be observed in Table 7. In the feed production stage reductions of around 5% were estimated for FE, GWP and LU by replacing conventional ingredients with tomato by-products in the poultry diet, avoiding the import of grain from long distance.

Water consumption and wastewater generation were minimised about 16% in the slaughterhouse and meat processing stage through recycling and re-use alternatives, by applying innovatives techniques such as pulsed light decontamination. These improvement had also positive influence on other categories, which is reflected in reductions on FE and GWP.

Conventional plastic tray and film were replaced with biodegradable materials avoiding up to 5% of greenhouse gases emissions associated to life cycle packaging. However biodegradable packaging showed an increased impact on categories such as ODP, EP, WDP and LU, linked to the crop cultivation stage required to obtain raw materials for bioplastic production.

Environmental impacts during product delivery stage were also reduced by improving different aspects along the supply chain, among others, implementing modularity at refrigerated storage spaces, optimising delivery routes and promoting eco-driving lessons. Estimated savings in electrical and diesel consumption at this phase lead to a decrease between 13-15% in all the assessed impact categories.

In terms of the whole life cycle of the analysed chicken product combined improvements have resulted in considerable reductions mainly in three categories: global warming potential (-4.3%), freshwater eutrophication (-5.85%) and land use (-5.61%).

Table 7. Environmental results per FU for the considered improvement options. GWP = global warming potential; ODP = Ozone Layer Depletion; AP = Acidification potential; FE = Freshwater eutrophication potential; WDP: Water resource Depletion; LU = Land use.

	Poultry Feed		Slaughtering		Packaging		Product delivery		Combined scenario	
	Value	Dif (%)	Value	Dif (%)	Value	Dif (%)	Value	Dif (%)	Value	Dif (%)
GWP	2.03	-3.20	2.10	- 0.02	2.10	-0.08	2.07	-1.1	2.01	-4.3
ODP	1.6E-7	-0.47	1.6E-07	- 0.03	1.6E-07	1.93	1.5E-07	-2.3	1.5E-07	-2.8
AP	0.13	-0.24	0.13	0.00	0.13	0.02	0.13	-0.1	0.13	-0.3
FE	3.6E-04	-5.53	3.8E-04	- 0.21	3.8E04	1.50	3.8E-04	-0.1	3.6E-04	-5.8
WDP	0.57	-0.40	0.58	- 0.14	0.59	2.75	0.57	-0.1	0.57	-1.7
LU	72.16	-5.53	76.39	0.00	76.64	0.33	76.33	-1.2	72.10	-5.6

4.4.4.4. Comparison with previous published literature

Comparability of the results with previous LCA studies of poultry meat may be difficult due to differences in the methodology, especially with regards to the functional unit selected, system boundaries of the study and the methods chosen for impact assessment stage.

Table 8 shows results of the present study expressed per 1000kg of edible carcass weight, in order to facilitate the comparison with previous LCA studies on poultry meat production. Moreover, comparable impact assessment methodologies have been selected from ReCiPe 1.04 Midpoint. Results were found to be of the same magnitude of those reported by previous authors, although in the high range for most of the analyzed impact categories.

The high value obtained in the current study for AP (116 kg SO_2eq/ton) can be attributed to the relationship between NH_3 emissions and temperature, since climatic conditions in Spain tend to be warmer than in Northern European areas.

With regards to GWP (2.8 ton CO_2eq/ton) results were very similar to obtained by several authors (Katajajuuri, 2007; Leinonen et al., 2012) but in general higher than the rest studies. This can be explained attending to differences in the scope and functional unit selected for the analysis which often refers to poultry meat at farm or slaughterhouse gate. Moreover it could be related to the current Spanish electricity mix, which depends more on fossil fuel combustion than other energy with less GHG emissions such as nuclear or renewable energies.

When compared to LCA studies from Northern European countries, present work obtained values in the same range than previously reported for EP (36 kgPO_3^{-4}/ton). Eutrophication potential impacts of poultry meat has been usually associated to emissions linked to fertilizing in the crop cultivation stage, followed by manure management processes (Leinonen et al., 2012; Seguin et al., 2011). Nevertheless, in this study case, although the role of nitrate and phosphate leaching from fertilization or manure application is still dominant, its contribution is limited to some extent due to dry climatic conditions; whereas the role of deposition of atmospheric ammonia emissions becomes more relevant, favored by warmer temperatures.

Table 8. Environmental comparison between chicken LCA studies: impact results per 1000kg of edible carcass weight. AP = acidification potential; Units: kgSO2 eq., EP = eutrophication potential; Units: kg PO3-4 ,GWP = global warming potential; Units: ton CO2 eq. In some studies, results were converted from live weight into carcass weight assuming a carcass yield of 70%.

	AP kg SO2 eq	EP kg PO4--- eq	GWP kg CO2 eq	Country
This study	116,67	36,76	2.889,90	Spain
This study ECO	116,34	36,22	2.777,92	Spain
Spies 2003	60,00	16,40	1.410,00	Brazil
Katajajuuri 2008	35,00	2,10	2.079,00	Finland
Pelletier 2008	15,80	3,90	1.395,00	USA
Williams 2009	25,90	14,00	1.800,00	UK
Williams 2009	30,80	23,50	2.000,00	UK
Leinonen 2012	32,70	14,20	3.087,00	UK
Leinonen 2012	40,00	16,20	3.437,00	UK
Prudencio da Silva 2014	28,70	13,80	2.216,00	France
Prudencio da Silva 2014	47,20	19,30	2.696,00	Brazil
Prudencio da Silva 2014	31,40	14,00	2.058,00	France
Prudencio da Silva 2014	34,50	14,40	1.449,00	Brazil

4.4.5. Conclusion

Through the ECOALIM project, LCA methodology has been successfully applied in order to identify critical stages and operations along the life cycle of a food product from an environmental point of view, but also for comparative analysis between different technologies and potential improvement options. LCA proved to be a useful tool directly involved in the decision making process when minimising the environmental impact associated to food chains, and additionally a considerable option to promote competitiveness, innovation and sustainability through the eco-design of food products.

According to the results of the present study the strong contribution of the poultry feed production stage on the environmental impact of a chicken product has been pointed out, as previously highlighted by other authors (Alvarenga et al., 2012; Leinonen et al., 2012). Nevertheless, the relevance of other phases of the life cycle should not be underestimated, since environmental improvements at every stage can lead to significant global reductions due to the high volume of consumption of the analysed product.

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4.5. CONTRIBUCIÓN V

SENSE tool: Easy-to-use web-based tool to calculate food product environmental impact

Saioa Ramos[a,*], Lohitzune Larrinaga[b], Unai Albinarrate[b], Niels Jungbluth[c], Gyda Mjöll Ingolfsdottir[d], Eva Yngvadottir[d], Birgit Landquist[e], Anna Woodhouse[e], Gudrun Olafsdottir[f], Aintzane Esturo[a,g], Jaime Zufía[a], Begoña Perez-Villareal[a]

[a] AZTI-Tecnalia Food Research Division, Parque Tecnológico de Bizkaia Edif. 609, 48160 Derio, Spain

[b] INGENET Computer and Control Engineering, Gorbeia Kalea, 14, 48410, Orozko, Spain

[c] ESU-services Ltd., Margrit-Rainer-Strasse 11C, 8050 Zurich, Switzerland

[d] EFLA Consulting Engineers, Höfðabakki 9, 110 Reykjavík, Iceland

[e] SP Technical Research Institute of Sweden, Frans Perssons v. 6, Sweden

[f] University of Iceland, Dunhagi 5, 107 Reykjavík, Iceland

[g] SGF International s.V. Am Hahnenbusch 14a, 55268 Nieder Olm, Germany

*Corresponding author: Saioa Ramos. Email: sramos@azti.es

Telephone number: 0034-667174357. Fax number: 0034-9465725 55

Abstract

Purpose: The purpose of the European SENSE project was to define an integral system to assess and communicate the environmental impacts of food products and to develop a web based tool for Small and Medium size Enterprises (SMEs). The tool has been tested in salmon, beef-and-dairy and fruit juice production sectors.

Methods. The SENSE project has evaluated several existing methodologies for environmental impact assessment over the life cycle including also social aspects, in order to deliver a new integral system for the environmental and social assessment of agricultural and aquaculture food products.

Results and discussion. The system includes a standardization of a data gathering system; a selection of relevant key environmental performance indicators for food supply chains and a common methodology to perform simplified life cycle impact assessment. The

results are based on collected information on the use of resources and emissions generated along the supply chain of food or drink products. The main result is a web-based software tool that is based on a summation of the partial impacts of the different steps in food supply chains. In this software different actors in the supply chain can enter their own data and link them to the data of other companies. The results obtained in the tool could be used for at least 6 different approaches: i) environmental impact assessment of the product, ii) food chain hot spot identification iii) comparison of hypothetical or real improvement scenarios iv) assessment of the environmental impact development over the years, v) benchmarking opportunity for the companies and vi) a business to business communication strategy. The scientific robustness of the tool has been tested comparing the obtained results with the same analysis with commercial software.

Conclusions. The SENSE tool is a simplified tool designed for food and drink SMEs to assess their sustainability on their own. This cannot be fully compared to a complete LCA study. The testing with SMEs showed that they need additional support for filling in the questionnaires correctly and interpret the results.

Recommendations. The simplified evaluation of environmental impacts based on a life cycle approach could lead to benefits to SMEs within the food industry. The future application and development of the tool will be focused on adapting the tool to the Product Environmental Footprint initiative requirements and self-assessment opportunities.

Keywords: Food SME, Online LCA tool, impact assessment, Benchmarking, SENSE Project, orange juice, salmon, beef, dairy.

4.5.1. Introduction

Food production and consumption cause significant strain on the environment. It is estimated that 29% of global emissions of greenhouse gases (GHG) are from agriculture and food production (Vermeule et al, 2012). In the EU, food consumption accounts for 20–30% of various environmental impacts and, in the case of eutrophication, more than 50% (Tukker et al., 2005). In the UK, the food and drink sector is responsible for 14% of industrial energy consumption and 7 Mt of carbon emissions per year; it also uses 10% of all industrial water supplies and produces 10% of the industrial and commercial waste stream (DEFRA, 2006). Moreover, in Switzerland i.e., nutrition causes about 12% of total energy demand and 18% of greenhouse gas emissions due to Swiss consumption patterns (Jungbluth et al. 2011; Jungbluth et al. 2013; Frischknecht et al. 2009).

The food and drink industry in Europe, of which according to Eurostat (2001) 99% are small and medium enterprises, is highly fragmented, and food chains are very complex. Hence, to assess the environmental sustainability of a product there is a need for applying integrated, harmonized and scientifically robust methodologies, together with appropriate communication strategies for making environmental sustainability understandable to the different actors in the food chain and to the market.

Nowadays the proactive communication of environmental impact of products can lead to great benefits to the industries which, in many cases, can lead to brand differentiation. However, most of the industries in the food sector, especially SMEs, neither have a strong background nor the capability to assess the environmental impact of their products.

Environmental impacts linked to food production can be identified and quantified with the Life Cycle Assessment (LCA) methodology, an internationally recognized methodology (Pelletier et al., 2007). Nevertheless, the importance of including innovative methodological improvements in LCA to broaden its scope and shift to a more comprehensive and harmonized environmental analysis is a major concern for LCA practitioners. Consequently, in recent years there are initiatives to harmonize and simplify the LCA methodology for the so called "Product Environmental Footprints" (PEF).

In order to implement a representative LCA in the food and drink sector the following aspects have to be defined:

i) Functional unit (FU) selection: The FU is defined as the unit which is being studied and quantifies the service delivered by the product system, providing a reference to which the inputs and outputs can be related. Units selected should reflect both the goal of the study and the role of a particular food product in a diet. Nevertheless this generates multiple questions in the food systems due to the fact that different functional units can be selected, for example, based on the mass (total product kg, kg of edible product), the volume (L, m3, etc.) or nutritional aspects (the caloric contribution, vitamin supply, protein supply, etc.) (Tyszler et al, 2014).

ii) System boundary selection: There is also a challenge when setting the system boundaries of the investigated life cycle according to the goal and scope of the study. Some studies include the full life cycle including transportation home, storage, preparation and disposal of waste and faces (Jungbluth , 2000; Muñoz et al, 2008). Other studies are focused only in the gate-to-gate systems because the buying and preparation processes are out of the scope of the studies (Pardo et al 2012).

iii) Timescale: The selection of the time limits of the study is a difficult issue in the food LCAs especially in the studies based in the sourcing of "wild" raw materials or dependent on factors not controlled by humans. For example the fish stocks vary considerably from year to year and therefore it is recommended to collect data of at least 3 years in order to obtain a robust result (Ramos et al, 2011; Vazquez-Rowe et al 2012b).

iv) Allocation procedure. Food product LCA is often complex because there is not just one product as output from the system. In the case of meat, some cattle are bred just to produce meat, but others are also part of the dairy chain. Hence, there can be difficulties assigning or allocating environmental impacts between the beef, dairy and by-product components. ISO 14044:2006 leaves different options for this task, such as system expansion, mass allocation or economic allocation.

v) Impact methodology selection: There are still many impacts that are not being considered in the life cycle thinking such as loss of biodiversity (Curran, 2010), discards (Vazquez-Rowe et al, 2012a) and damages to the marine seafloor (Ziegler and Valentinsson, 2008). The globalization of the production systems forces the creation of some impact characterization factors according to the conditions in each region.

Within this framework, the European research project SENSE aims to deliver a harmonized system for the environmental impact assessment of food and drink products for SME producers and to propose a common approach to the above mentioned aspects. Using the defined system and tool, all companies using the tool will be using the same methodology. The research evaluates existing relevant environmental impact assessment methodologies, to deliver a new integral system that can be linked to monitoring and traceability data. The system integrates:

(a) methodology for environmental impact assessment (described in section 2.1) which defines the impact methodology selection;

(b) a set of key environmental performance indicators plus a harmonized system to collect data (described in section 2.2);

(c) a common LCA framework for the food industry (described in section 2.3) answering the functional unit, allocation procedure, definition of system boundaries and the time scale.

(d) a web-based tool which allows the environmental impact calculation for a food product (described in section 3) in a simplified way.

The sustainability information is collected along the supply chain of the food product and reflected into an Environmental Identification Document (EID). This should contribute to making information on the environmental sustainability of a product as a part of the usual decision making and purchasing behavior in B2B. It also could provide a competitive advantage to those products and companies which choose to use the EID, through a comprehensive environmental communication between the industries.

4.5.2. Methods

In order to develop the tool the main aspects as outlined before have been defined as follows.

4.5.2.1. LCIA methodology

In parallel to the SENSE project, the Roundtable for the Food and Drink Sustainable Consumption developed the ENVIFOOD protocol in 2012. A set of environmental impact indicators for the food supply chains was selected based on literature reviews (Landquist et al., 2013a). After the definition of the main impact categories, a specific characterization method for each impact was agreed on after the review of public and private initiatives. The life cycle assessment methodologies chosen for each impact category are listed in Table 1 along with the corresponding indicators and references. This task was done at the same time the ENVIFOOD protocol was being developed.

Table 1. Life cycle impact assessment methodologies used in the SENSE-tool.

Impact category	Unit	Selected LCIA method	Reference
Climate change	kg CO_2 eq	Bern Model – IPCC	Solomon, 2007
Eutrophication, Terrestrial	molc N-eq	Accumulated Exceedanc	Posch et al., 2008
Eutrophication, Freshwater	kg P-eq	ReCiPe v1.05	Goedkoop et al., 2009
Eutrophication, Marine	kg N-eq	ReCiPe v1.05	Goedkoop et al., 2009
Acidification	molc H+-eq	Accumulated Exceedanc	Posch et al., 2008
Human toxicity	CTUh	USEtox Model	Rosenbaum et al., 2008
Ecotoxicity	CTUe	USEtox Model	Rosenbaum et al., 2008
Land use	kg C/m^2/a	Soil organic matter mod.	Milà i Canals 2007
Abiotic resource depletion	kg Sb eq	CML 2002	Guinée et al., 2002
Water depletion	m^3 H_2O eq	Ecological scarcity model	Frischknecht et al., 2009

2.2. Key Environmental Performance indicators for easy data collection

A key issue in LCA is the collection of data for all stages in the life cycle. Often the list of required data is quite long. Therefore a simplified approach has been developed in this project by focusing the data collection on a list of key environmental performance indicators (KEPI)

KEPIs are used in the SENSE tool to calculate the environmental impacts. The KEPIs are essentially parameters that will be used as life cycle inventory inputs or outputs. These inputs are related to one year period and have been selected in order to have a set of parameters which together account for at least the 90 % of the impact related to a product (figure 1). KEPIs are yardsticks which can be easily measured by an actor in the food chain. For the selection of those parameters three detailed LCAs have been performed in the beef and dairy, orange juice and salmon aquaculture sectors based on the impact methodologies described in section 2.1 (Doublet et al., 2014). Those LCA studies contribute to verifying the relevance of the KEPIs considering their contribution to the environmental impact, the data availability and the easiness of measurement. As a main result for the environmental impact assessment, the most relevant KEPIs for each chain have been selected to create a list of inputs and outputs. The following main group of inputs and outputs for each chain were described by Doublet et al., (2014):

i) Beef and dairy chain: The production of feed, the emissions from the use of fertilizers, and the manure and diesel used for the agricultural machinery are important contributors to environmental impact, representing up to 70% of the total potential impact for all the potential impacts selected (Doublet et al., 2013a).

ii) Orange Juice production chain: The four main contributors to the orange cultivation impact assessment are the electricity use for the irrigation (50 % of the climate change potential impact), the N-fertilizer and P2O5-fertiliser use (95 % of the most impact studied) and the production and application of pesticides (50 % of the total ecotoxicity potential impact). Regarding post harvesting stages, the most relevant aspects for the juice pressing are the electricity use and thermal energy use (representing up to 50% of the total Global Warming potential). Moreover, the main contributor to the bottling process is the manufacture of the PET bottle (Doublet et al., 2013b).

iii) Salmon aquaculture chain: Feed is identified as a KEPI for the hatchery and the aquaculture farm representing up to 60 % of the most environmental impact studied as well as water use which is also identified for the secondary processing. Additionally, organic matter to sea from the aquaculture farm is identified as a KEPI, representing 97 % of the total impact of marine eutrophication (Ingolfsdottir et al., 2013)

Taking into account main results obtained in these studies, in Table 2 the selected set of KEPIs for the production of all the food and drink supply chains is shown (Landquist et al, 2013b). The selection was based on their relevance for the environmental impact in the studies. Also the previous bibliography and the expertise of the project partners were taken into account. For example, use of fertilizers has been defined as KEPI, where production of fertilizers is also taken into account into the tool. For the moment the tool does not include all types of fertilizers used for the agriculture, although it is prepared for it.

Table 2. Selected key environmental performance indicators for the European food and drink sector.

INPUT	UNIT		DS
Land use	Ha/year	Land occupation for agricultural uses: permanent crops, arable land or grazing.	ecoinvent
Fertilizers	Kg N, P or K/year	Inorganic fertilizer consisting of nitrous compounds such as ammonium nitrate or ammonium sulphate and phosphorous or potassium compounds.	ecoinvent ESU
Organic fertilizer	Kg/year	Fertilizers derived from animal or vegetable matter (e.g. compost, manure)	ecoinvent

Pesticides	Kg AI/year	Pesticides are plant protection products. The term "pesticides" covers insecticides, acaricides, herbicides, fungicides, plant growth regulators, rodenticides or biocides. The user has to provide the commercial name for the pesticide (i.e. RoundUp ®) in a free-text box and introduce the amount per hectare used. Once it is defined, an addition table will appear where they have to specify the percentage of active ingredient (AI) (i.e. glyphosate). If the AI is not in the list, generic pesticides could be used, such as, "fungicides" or "herbicides" or "pesticides". When those AI are used, 100% of the content should be entered..	ecoinvent
Energy	energy unit / year	Energy consumption in agriculture systems are mainly related to fuel used during land labors (tractor), energy required for buildings maintenance and greenhouses maintenance, in the fisheries systems to the use of fossil fuel for the fishing vessels and in aquaculture, livestock and food processing systems the energy use is mainly related to the machinery requirements and building general consumption.	ESU
Freshwater use	L or m^3/year	For water requirements the user has to introduce the total water requirements for 1 year. Rain water is not taken into account, only tap-water	ecoinvent
Feeds	Kg/year	Data on feed can be obtained directly from the feed supplier as guest user and should then be added as an incoming product or Data on feed can be selected from a drop down menu, offering different kind of feed ingredients (crop and marine). In the questionnaire, the user should specify the different feed ingredients and add the relative amount by weight.	ecoinvent
Packaging	Kg/year	For the packaging the user should specify the type of final packaging (glass, plastic bottle or so) and the amount used per year. In some cases, intermediate packaging will be relevant too.	ecoinvent
Livestock	nº animals /year	For the livestock, the specific animal has to be selected. Specify the amount produced in one year and the share of the product in turnover (%).	IPCC
OUTPUT			
Wastewater	L or m^3/year	For inland aquaculture systems the user need to specify the amount (l or m3) of wastewater discharges per year. For marine aquaculture systems an average N direct emissions to the marine environmental due to faeces and uneaten feed per kg of fish has been taken into account (Heldbo, et al. 2008).	ecoinvent

Wastes	Kg/year	The user chooses the waste material (organic waste, plastics, cardboard, glass or other type) and the disposal way (incineration, recycling landfill)	ecoinvent

Some assumptions were taken in order to adapt the datasets to the studies. Following main assumptions are described:

- Inorganic and organic fertilizer emissions: Calculations for direct emissions due to the application of fertilizers are based on scientific emission models and not on real measurements. The nitrogen and phosphorous emissions have been calculated depending on the N and P kg applied. Nitrogen emissions calculated according to ecoinvent report 15 (Nemecek & Schnetzer, 2011) and IPCC 2006 guidelines (de Klein et al, 2006) as follows: emissions to the air 1% of N in NO_x and NH_3; and 0.75% of nitrate; and 30 % of applied N to groundwater as nitrate.

- Livestock emissions: biogenic GHG emissions from manure management and enteric fermentation were calculated depending on the type of animal and country according to Tier 1 formula from IPCC 2006 guidelines (Dong et al, 2006).

- Feed ingredients: Vegetable ingredients of the feed were taken directly from the ecoinvent dataset; however, for the marine feed ingredients, datasets were created for the catching and processing of 15 different fish species, including the reduction processes of fish to meal and oil. Datasets on feed were created for average feed in Norway in 2010 (Hognes et al 2011) and average Icelandic feed from 2013 (Olafsdottir et al., 2014) Background data for those datasets regarding fuel use for fisheries was obtained from SINTEF in Norway and information on the resource use for the salmon feed reduction process was based on published results (DEFRA, 2007; Winther et al. 2009).

- Pesticide emissions: Due to the lack of specific inventory models for pesticide emission quantification, most of the food LCA studies assume that 100% is emitted to the soil in accordance with ecoinvent approach (Nemecek & Schnetzer, 2011).

- Fuel emission from combustion: The emission of tractor operation was taken from ecoinvent report 15 (Nemecek & Kägi, 2007) while data for the diesel combustion in vessel was taken from ecoinvent report 14 (Spielmann et al, 2007).

- Refrigerated transports: refrigerant agent needed for the refrigerated transport was taken from Emanuelsson approach (2010).

- Packaging: For plastics containers (such as PET bottles or GPS foam) a combined dataset between the type of plastic and a plastic transformation was taken from ecoinvent, assuming also a percentage of material losing according to ecoinvent dataset. For glass, can and film, direct dataset from ecoinvent were taken (Hischier, 2007).

- Waste water: Two different types of waste water have been taken from ecoinvent: one representing waste water with high organic load (assimilated to potato starch production effluent) and another representing low organic load (assimilated to sewage, unpolluted). The discharge of organic matter from marine based system was estimated using an average N discharge (41 kg N eq/ 1 ton fish) to the marine environment due to feces and uneaten feed per kg of live fish (Heldbo et al., 2013).

For KEPI collection, yearly data gathering system has been described in detail (Alvarez et al., 2013). In order to facilitate the data gathering, the tool offers the possibility to send the questionnaires to the main suppliers of the chain.

4.5.2.2. Common LCA framework

In order to create a common framework a reference flow of kg edible product is recommended for all the food products. It is not foreseen to introduce any functionality to the calculations nor including distribution or food consumption.

For the system boundary a cradle-to-gate limit has been selected, excluding retailing, consumption and end of life processes. This reflects the fact that the tool is designed for internal use and for business-to-business communication.

The main idea of the tool is to enter data for a specific year of production. This makes it also possible to monitor the environmental performance between different years.

For the allocation some procedures proposed in ISO are out of the scope of this easy to use tool for SMEs. However, allocation cannot be avoided and allocation rules should be made as simple as possible. Thus, economic allocation has been selected; however, the tool allows introducing manually the percentage of the allocation of different incoming materials, such as packaging or main ingredients, if these can be clearly assigned to one output.

Furthermore social aspects are tackled in the tool. Therefore a questionnaire with relevant aspects has been developed. The questions were focused on adherence to labour standards and national laws, and communication of the companies´ policy regarding labour standards. The questions also addressed workers´ rights to join trade unions, their employment conditions, wages and working hours. Additionally, questions were included on the status of occupational health and safety training, training related to employees wellbeing and the actions of the companies to address issues regarding the influence of the company on the local communities both concerning remedies and additional costs as well as offering opportunities to local people. A grading scheme was also developed but so far the results of this questionnaire have not been evaluated quantitatively (Olafsdottir et al., 2015).

4.5.2.3. Validation of the tool

Once the tool was developed and prior to implementing the SENSE tool in companies, the tool was tested by checking the functionality of the user interphase and the performed calculations. The aim was to verify that the outcome of the tool calculations were comparable with the results obtained when using the same input data in a commercial software (SimaPro and GaBi).

The validation of the integrated SENSE tool was based on performing simplified environmental impact assessment representing three food chains (fruit juice, meat and aquaculture fish) in different European regions. An acceptance criteria was defined of less than 10% of variation in the impact characterization in comparison with a conventional LCA (Olafsdóttir et al, 2014). The first validation was based on the comparison between the results of the complete LCA and the results from the simplified SENSE-tool (Olafsdóttir et al, 2014). A complete LCA takes into account more inputs to the system such as secondary packaging materials or auxiliary materials and also models for each case study the emissions from the substances emissions such as pesticides, fertilizers or manure emissions.

The second validation of the tool was to assess the functionality with 23 food SMEs. The results of SENSE tool calculations for the products assessed from the SMEs were checked by exploring if the range of values obtained were within the range of earlier SENSE tool case studies that had been validated in the SENSE project. Additionally, literature values for similar products (e.g. raw and pasteurized milk, orange juice, salmon and arctic char) were used for comparison. The methodology or impact characterization factors applied in some of the studies reported in the literature vary and therefore only the climate change impact was assessed in this second validation (Olafsdóttir et al., 2015).

4.5.3. Results

4.5.3.1. Development of the tool

Taking into account the impact assessment methodology and the set of KEPIs selected, a web based tool, the SENSE tool, has been designed and developed with a common server and database allowing an active interaction between users. The developed tool aims to be used by food SMEs without a strong LCA background and to provide easy to be interpreted environmental information.

The tool compiles the information available at different levels in the food chain. The collected data are characterized and evaluated in order to obtain the key indicators associated with the evaluated product. This tool provides a common framework to users from different stages of the supply chain and by introducing a simplified set of production data they can compare respective environmental impacts. The tool has been designed in a user friendly way and very intuitive to facilitate its use by SMEs.

The tool is accessible via internet; therefore it is not necessary to install any software, making its use even simpler. This computer application has been developed using Visual Basic .Net, on Visual Studio 2010. The database engine used is SQL Server 2008 R2, where all the application's information is stored. As far as the application imaging, both design and pictures implemented, were done using Photoshop CS 6 and Gimp 2.8.

The SENSE tool application calculates the environmental impacts related to the selected impact categories. The impact characterization can be shown for each functional unit (kg of product) or the production of one year. The results of the tool are presented in the following ways:

A) Environmental impact per process and year. Those results are shown in a bar chart and show the impact generated for the selected environmental indicator and process. A table with the impact value is also shown under the graphic (Figure 1).

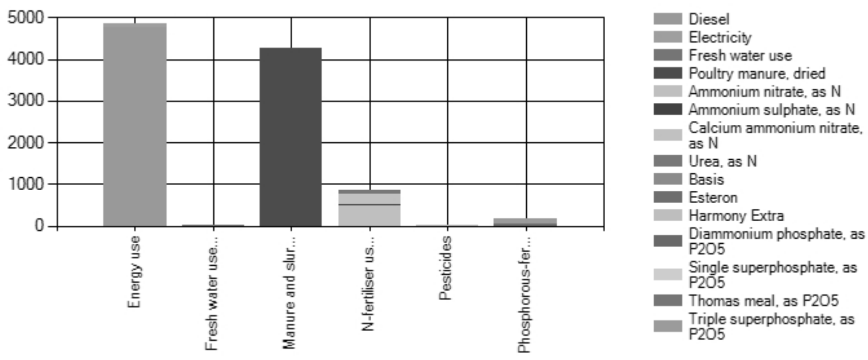

Figure 1. Captured figure of the SENSE-tool results for the climate change potential characterization results for the farm stage of the dairy production chain.

B) Complete impact analysis: For each impact category a pie graph is shown with the contribution of each process to the total impact. Additionally, a histogram is also shown with the summary of all the impact and processes (Figure 2).

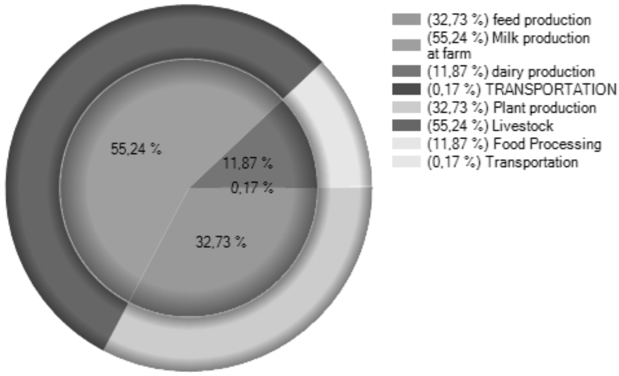

Figure 2. Captured figure of the SENSE-tool results for climate change potential characterization for a dairy production chain expressed in pie chart.

C) Comparison between product's environmental impacts. It is also possible to compare the impact of different product by process or by impact. When comparing the environmental impact by process, the weight of the different processes on the final impact of each product will be shown for each impact category. When comparing by impact, a complete graph will be shown comparing the final impact of each product.

D) Evolution of the product impact. A line chart is shown with the evolution of the environmental impacts of the product along the years. With this data the tendency of the environmental behavior of a given product could be assessed.

E) Product benchmarking: In the future this option would allow the user to benchmark its products internally (comparison between the companies own product in different years) and externally (with other similar products and between different companies). When selecting benchmarking option, a spider graph can show the deviation of the actual product impact assessment towards the average value for that product. For the moment, there is not enough data for the external benchmarking; however the tool is ready for the future improvements.

F) Coupled with those graphics, there is a possibility to extract the Environmental Identification Document (EID) where a summary of main environmental results is described. The main objective of this document is to transmit the information along the B2B stakeholders involved in the food supply chains.

4.5.3.2. Validation of the tool

In order to ensure the functionality of the tool, main results of the complete LCA versus the SENSE tool results were compared. Although this comparison was made for the three selected chains, the results are just shown for juice and beef and dairy chain. For the aquaculture chain the results of the complete LCA study performed were not comparable with the output from the SENSE tool since new datasets for feed had been implemented in the tool. Furthermore, organic emissions to sea were accounted for as BOD, nitrogen and phosphorus per whole fish in the LCA study (Ingólfsdóttir et al., 2013). However, in the SENSE tool, this was simplified and only the release of nitrogen is accounted for based on published information (Heldbo et al., 2013).

The relative percentage difference between the environmental impacts calculated in the SENSE tool and the LCA on orange juice is below 10 % for some impact categories such as climate change, human toxicity, acidification, eutrophication terrestrial, eutrophication marine, abiotic resource depletion and water depletion (table 3). Moreover, the relative percentage difference between the environmental impacts of the SENSE tool and the LCA on Romanian beef is highly dependent on the impact category. Results for climate change, human toxicity cancer and non-cancer effects, ecotoxicity, freshwater and land use had a difference smaller than 10 % for meat products (table 3) (Olafsdóttir et al, 2014). The impacts with higher differences (eutrophication of freshwater, ecotoxicity of freshwater and land use in orange use and acidification, terrestrial, freshwater and marine eutrophication and water depletion in beef supply chain) are also due to differences in the data bases and the system boundaries as in the case of the aquaculture.

Table 3. Main results obtained for the comparison of the impact characterization between the SENSE-tool and SimaPro LCA approach for orange juice (1l at pressing company) and beef (1kg at slaughterhouse)

Impact category	Differences orange	Differences beef
Climate change	-2%	2%
Human toxicity, cancer effects	10%	7%
Human toxicity, non-cancer	-4%	-1%
Acidification	2%	-69%
Eutrophication, terrestrial	-3%	-73%
Eutrophication, freshwater	-14%	-76%
Eutrophication, marine	-7%	-60%
Ecotoxicity, freshwater	-95%	2%
Land use	-70%	-7%
Water depletion	-1%	33%

Moreover, a validation with 23 food SME's was carried out. Overall the results of the SENSE tool calculations of climate change for the products of the external companies fulfilled the testing criteria. The literature values for raw milk vary between 0.74 and 2.8 kg CO_2 eq/kg raw milk (Doublet et al., 2013a) and the results obtained in the pilot studies are in line with the expected results from literature. The values for climate change were 1.40, 2.47 and 2.88 kg CO_2 eq per liter raw milk. For aquaculture companies participating in the pilot studies, the results of the SENSE tool calculations for climate change impact for conventional net pen system were similar, around 2.4 kg CO_2 eq/kg fish. The results are similar to earlier reported values by Ytrestöyl et al. for the 2010 feed (2.6 kg CO_2 eq/kg). The land based systems had higher impacts ranging from 3.20 - 5.1 kg CO_2 eq /kg (Olafsdóttir et al., 2015). Comparison with other impact categories was not possible due to lack of references with the selected methodology. Most of the selected case studies for this comparison have been carried out with the CML or ReCiPe methodologies, both of them measure toxicity as 1,4 DB eq, while ILCD and SENSE uses CTU. The same happens for example with the acidification, while CML and ReCiPe use kg SO2 eq, ILCD uses molc H+. Therefore, is not possible to compare.

4.5.4. Discussion

4.5.4.1. Allocation procedures

The method used when distributing the environmental burden between the main product and its by-product can have a significant impact on the final results of a LCA (Svanes et al., 2011). Since the aim of the project is to obtain a simplified environmental analysis of the food and drink products, some limitations have to be accepted. Although it may be controversial, economic allocation is chosen in the SENSE tool as the default allocation approach which can be easily implemented for all production systems.

For the beef and dairy chain, the allocation procedures recommended by the international dairy federation (IDF, 2010) to allocate the environmental impacts of beef and milk production at farm as well as the allocation matrix to distribute the environmental impacts of the individual dairy products are complex and time-consuming for somebody not familiar with life cycle assessment. However, the results for single dairy products are quite sensitive to the allocation approach chosen (Feitz et al. 2007). Feitz et al. (2007) suggested using economic allocation for inter-industry sectorial flows while Kim et al. (2013) allocated the incoming raw milk to the individual dairy products on a milk solids basis. Moreover, the allocation of environmental impacts to by-products is also an issue for the slaughtering process in the beef chain (Cederberg et al., 2009). These differences in the allocation methodology explain deviations of results in different tools and literature.

In the aquaculture chain, the use of economic allocation has been criticized as it does not reflect the biophysical properties of the production system and it is sensitive to changes in market prices (Pelletier & Tyedmers, 2011; Svanes et al., 2011; Ytrestøyl et al., 2011). Mass allocation methods have been applied in studies on feed and aquaculture as well as fisheries (Boissy et al., 2011) while others have used gross nutritional energy (Pelletier et al., 2009) or economic allocation (Ellingsen et al., 2009). However, the use of by-products from environmentally costly productions such as livestock production or demersal fish trimmings in salmon feed production contribute substantially to the outcome of an LCA analysis in terms of energy use and CO_2 emissions (Ytrestøyl et al., 2011). The economic allocation used in this case study gives a higher burden on the main product than if mass allocation would have been used. At the aquaculture farm 10 % of the biomass at the farm is guts which are given away for free and therefore has zero environmental loads (Ingolfsdottir et al., 2013).

The decision to use economic allocation rules for the SENSE tool may be the simplest approach for SMEs. However, since the SENSE tool offer the possibility to implement different allocation factors for each incoming product; this is a good approach that could be used if SMEs are willing to invest more time to obtain a more scientific environmental assessment.

4.5.4.2. Validation of the Life Cycle approach of the SENSE tool

As mentioned in section 3.2, relevant differences were identified for some of the selected impacts when comparing the results from the SENSE-tool and the SimaPro.

Differences identified in acidification and eutrophication during the fruit juice validation experience could be explained because the modelling of the emissions due to the land use and the application of manure as well as the additional data taken into account in the complete LCA. Moreover, differences in the type of herbicides applied and the emissions from the land use explain the large deviation in the freshwater ecotoxicity and the freshwater eutrophication impact categories. In the SENSE tool, the modelling of the land use does not include the transformation from and to permanent crop. This explains the deviation in those impact categories (Doublet et al., 2013b).

For dairy and aquaculture products the results of the earlier LCA case studies (Doublet et al., 2013a; Ingolfsdóttir et al., 2013) were not comparable with the SENSE tool calculations due to e.g. difference in methodologies, allocation rules or in background datasets applied (Olafsdóttir et al, 2014).

Overall, the methodology established for the SENSE-tool does not replace complete LCA because the collection of data is simplified to be adapted to SMEs and, on the other hand, there is a limited amount of freedom degrees for the modeling of the studied system. However this limitation is one of the objectives of the project itself, since all the studies are intended to be performed according to the same method in order to obtain comparable results.

4.5.4.3. Usefulness in SMEs

The testing of the deployment of the SENSE tool was performed by users in 23 companies in meat and dairy, fruit juice and salmonid aquaculture sectors. The common impressions from the SMEs in all sectors indicates that the companies were at first reluctant to implement the SENSE-tool into their company mainly due to lack of resources (time or people). However, when given support training and help to introduce data into the SENSE tool the users agreed on several benefits of using the SENSE tool for sustainability assessment. Main benefits of the SENSE tool identified for the companies were; i) a user friendly tool that facilitates harmonized data gathering for life cycle inventory; ii) benchmarking was considered a very interesting option (not implemented yet); iii) the results can be used in sustainability reporting (environmental and social); iv) aquaculture farms see benefit of carbon footprint calculations, to fulfill requirements of e.g the Aquaculture Stewardship Council standard; v) the possibility to identify the hot-spot of the processes; vi) the possibility to use as a tool to improve suppliers performance.

While the milk and meat sector was least interested to participate in the testing, the aquaculture companies, mainly from Iceland, were most willing to participate. The willingness to test the tool in the aquaculture industry may be related to the fact that they are already under pressure to demonstrate their performance due to pressures from regulations, green accounting requirements and upcoming standards. Data on the KEPIs that is needed to perform environmental impact assessment using the SENSE tool is therefore already available. In the meat and dairy sector as well as the fruit sector, the perception was that SMEs were reluctant to share data due to the fear of data misuse. In the juice sector, the lack of willingness is explained mainly by the lack of time and personal resources available for testing the tool. This is a common obstacle for the entire food SMEs, but the fruit sector is particularly seasonal and the time period where the testing was scheduled coincided with the peak activity. So, even though the companies are aware of the environmental issues, they could not invest time in testing a tool.

4.5.4.4. Using the SENSE tool for food sector

One of the main objectives of this project was to obtain a harmonized methodology to measure the environmental impact of the foods produced and consumed in Europe. For that purpose, case studies have been performed, but only in the previously mentioned three sectors and thus the results do not represent the whole food and drink sector. Therefore, it would be necessary to test this tool in other food sectors. Nevertheless, it is likely that the tool could be adapted to other food sectors, since the impact indicators and inventory flows were selected by taking also into account experience of experts and bibliography.

4.5.5. Conclusions

The SENSE tool has been designed to be suitable for SMEs in food and drink sector and it has been shown by the testing companies that the objective has been achieved. However, it is important to highlight that the main aim is to obtain a simplified tool, and thus it won't be an alternative for the complete LCA studies in any case.

This tool is aligned with the new emerging initiative developed by the European Union Single Market for Green Products. This initiative is nowadays developing a new framework for measuring the "product environmental footprint" of all kind of good and services commercialized in the EU. Although the SENSE-tool is not completely adjusted to this methodology, it is open for modifications in order to adapt it to the PEF initiative.

The integration of social aspects in product assessment is still not fully developed. Further research work is necessary in order to integrate this in a harmonized way according to CSR and UNEP/SETAC initiatives.

Finally, as a recommendation, it is important to highlight that there is a need to encourage food companies to include the environmental issues in the decision-making processes making the stakeholders of the food chains aware about the sustainability of their products.

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4.6. CONTRIBUCIÓN VI

Evaluating the suitability of three water scarcity footprint methods: case study for the Spanish dairy industry

Ramos S[1], Ridoutt BG[2,3], Sanguansri P[2], Zufía J[1]

[1] AZTI-Tecnalia, Bizkaiko Parque Tecnologikoa, Astondoa Bidea Edif 609, 48160 Derio, Spain

[2] Commonwealth Scientific and Industrial Research Organisation (CSIRO), Private Bag 10, Clayton South, Victoria 3169, Australia

[3] University of the Free State, Department of Agricultural Economics, Bloemfontein 9300, South Africa

Abstract

Water footprints are becoming an important metric used by the business sector to identify environmental performance improvement opportunities and to report performance to stakeholders. This has been supported by the publication of a new International Standard on the subject (ISO 14046). Presently, the greatest interest is in water scarcity footprints which specifically address consumptive water use. However, right now the comparability of different water scarcity footprint results is inhibited by the use of different characterization factors for local water stress. Using a case study based on dairy production in northern Spain, water scarcity footprints were calculated using three alternative methods (revised WSI, WSI_{HH_EQ} and AWaRe). The results varied significantly, from 2.8 to more than 6,000 L H_2Oe/kg product. In the near term, operators of product water footprint programmes should specify the particular water stress factors to be applied. This will enable comparability of product water footprints within a programme. However, over time, effort should also be made to achieve consistency between programmes, and this study discusses some important considerations such as the reference unit and range.

Keywords: Water Stress, LCA, Aware, WSI, ISO14046, environmental labeling, sustainable consumption and production

4.6.1. Introduction

According to the European Environment Agency (EEA, 2009) water scarcity occurs where there are insufficient water resources to satisfy long-term average requirements. It refers to long-term water imbalances, combining low water availability with a level of water demand exceeding the supply capacity of the natural system. Although water scarcity often happens in areas with low rainfall, human activities contribute to the problem, especially in areas with high population density, tourist inflow, intensive agriculture and water demanding industries.

Water use efficiency (WUE; water use per unit process) has been widely used as a performance indicator in regards to water use. It is a measure of the technical efficiency of a factory or of an appliance (e.g. a washing machine). However WUE does not make sense from a life cycle perspective as there is no logic in aggregating water of different types from locations of different water scarcity. As such, WUE cannot be meaningfully applied to the life cycles of products and services. In 2014, the International Organization for Standardization addressed this problem with the publication of an International Standard for water footprinting (ISO 14046). This was a major development, providing a consistent framework for businesses to evaluate and report impacts related to water use from a life cycle perspective. ISO14046 also provides a basis for businesses to make water footprint claims with respect to products and services, akin to the product carbon footprint claims (i.e. labels and declarations) which have begun to appear in many jurisdictions.

ISO14046 includes within its scope both water consumption and pollution. When only water consumption is evaluated the term *water scarcity footprint* is recommended. A limitation at present is that ISO14046 does not specify a particular characterization model for water scarcity. This is not such a problem for LCA studies designed to provide strategic insights for environmental improvement, as the focus is generally upon identification of hotspots and it is the relative size of the contribution from different life cycle stages that matters. However, for water footprint claims where there is communication to stakeholders in society (including consumers), the absolute numbers do matter.

Within this framework a water scarcity footprint study was carried out for a large dairy producer located in the north of Spain. Since there is not one commonly agreed method for the evaluation of the local water stress, three different methods were tested and compared.

4.6.2. Alternative water stress characterization factors

Within the Life Cycle Assessment approach several methods have been published to quantify the environmental impacts of water use (Kounina et al., 2013). In relation to consumptive water use, among the most relevant and recent methods are the following:

- The revised Water Stress Index (WSI) developed by Ridoutt and Pfister (2010) as recommended by the European Food Sustainable Consumption and Production Roundtable in the ENVIFOOD Protocol (Food SCP Rt, 2013). This index utilizes the regionalized withdrawal-to-availability based factors reported by Pfister et al. (2009), but with moderation by the global average WSI. In this way, water scarcity footprint results are reported in units of H_2Oe (equivalent) where the reference situation is water consumption at the global average WSI.

- The regionalized WSI of Ridoutt and Pfister (2014; www.ifu.ethz.ch/ESD/downloads/WSI_HH_EQ.kmz) that integrates separate models for water stress on humans and ecosystems. In some regions, the human health impacts of water stress are moderated by the importation of water intensive goods, such as food, as well as investments in alternative water supply technologies and water use efficiency measures. As such, the water withdrawal-to-availability ratio can be a poor proxy for environmental harm. This alternative index (WSIHH_EQ) overcomes this problem and delivers results which are consistent with LCA endpoint models for water use (Pfister et al., 2009).

- Aware (Available Water Remaining) method (http://www.wulca-waterlca.org/): This new index is the preliminary recommendation by the UNEP-SETAC Life Cycle Initiative's project group on water use (http://www.lifecycleinitiative.org/).

Case Study: Water footprint of dairy production in northern Spain

The selected functional unit or target of the study was the processing of 1 kg of raw milk at a dairy company located in Asturias (North of Spain). The system boundary was from cradle to gate, taking into account the production of feed from raw materials to the processing of milk into dairy products.

Direct water resources used for feed crop irrigation, farm operations and dairy plant operations were taken into account. The farming subsystem, which was a conventional system supplemented by pasture and purchased feed, was modeled using data for 54 farms providing raw milk to the dairy plant. The feed used was a conventional feed for milking cows, prepared by 45 % of maize produced in Spain, 12 % of Barley produced in Spain, and 10 % of soy imported from Argentina and Brazil among other supplements. These data covered the 2013 financial year. The dairy factory subsystem was modeled using data provided by the plant engineer. This system included inputs of reticulated town water, the recovery of water from the evaporator, on-site wastewater treatment, selected water reuse, and the discharge to a local stream of freshwater, which is regarded by the local water management agency as an environmentally beneficial flow.

Additionally, water evaporation from hydropower reservoirs was taken into account, using evaporation rates given by the Spanish Ministry for Agriculture, Food and Environment for each region in Spain. In this dairy product case study, so-called green water (soil moisture from natural rainfall) consumed through pasture evapotranspiration was not included in the inventory of water use.

Background inventory data for the crop cultivation, electricity generation and water supply datasets were taken from Ecoinvent and Agri-Footprint (Blonk Agri-footprint BV, 2014) databases. In addition, emissions to freshwater from fertilizers, pesticides, and industrial processes were not included because the environmental impacts are generally considered under other LCA impact categories such as eutrophication and freshwater ecotoxicity and do not form part of a water scarcity footprint.

To calculate the water footprint, each instance of consumptive water use was multiplied by the relevant WSI and then summed across the product life cycle.

4.6.3. Result and discussion

The production of 1 kg of raw milk in Asturias, northern Spain, was found to depend on 37.7 L of water consumption. Almost 95 % of this water consumption was associated with the supplementary irrigation of the maize grown in Spain and used for animal feed. On the other hand, 4.8 L of water were used in the dairy plant to process each 1 kg of raw milk into dairy products. Almost all of this water was used for the pasteurization of the raw milk (Table 1).

Table 1: Inventory of direct and indirect water consumption to process 1 kg of raw milk.

	water consumption	
	L	**%**
Farm subsystem		
Livestock direct	1.46	3.44
Hydropower	0.46	1.09
Feed	35.83	84.27
Barley (SP)	*4.63*	*10.90*
Maize (SP)	*31.17*	*73.32*
Soy (BR)	*0.00*	*0.01*
Soy (AR)	*0.02*	*0.05*
Dairy industry		
Direct use	3.93	9.24
Hydropower	0.83	1.96
TOTAL	42.52	100

The water scarcity footprint results are presented in Table 2. Regardless of which method was used, feed production was identified as the most important life cycle stage (87 to 92 % of the overall water scarcity footprint). In all cases, the production of maize was most critical, although differences in the contribution of the different feed components was evident when the different methods were compared. In all cases, dairy processing contributed less than 10% of the overall water scarcity footprint (Table 2). These results demonstrate that for strategic LCA studies, where results are used internally within the business and not shared with stakeholders, the choice of water stress index is not so critical as the relative importance of different life cycle stages do not differ so much.

Table 2: Water scarcity footprint results for the processing of 1 kg of raw milk calculated using three methods (WSI, WSIHHEQ and AWARE). See text for details about the methods.

	WSI		WSI_{HHEQ}		AWARE	
	L H_2Oe	**%**	**L H_2Oe**	**%**	**L world eq.**	**%**
Farm subsystem						
Livestock direct	0.03	0.30	0.04	1.47	2.69	0.04
Hydropower	0.46	4.39	0.04	1.53	250.13	4.00
Feed	9.00	86.64	2.51	90.30	5,767.78	92.23
Barley (SP)	*3.38*	*32.57*	*0.33*	*11.82*	*2,681.51*	*42.88*
Maize (SP)	*5.61*	*54.01*	*2.18*	*78.37*	*3,086.26*	*49.35*
Soy (BR)	*2.4E-04*	*0.00*	*1.2E-04*	*0.00*	*2.6E-03*	*0.00*
Soy (AR)	*7.1E-03*	*0.07*	*2.8E-03*	*0.10*	*5.7E-03*	*0.00*
Dairy industry						
Direct use	0.08	0.79	0.11	3.95	7.23	0.12
Hydropower	0.82	7.88	0.08	2.75	225.84	3.61
TOTAL	**10.39**	100	**2.78**	100	**6,253.67**	100

However, the absolute results varied enormously, from less than 3 to more than 6,000 L H_2Oe/kg (Table 2). In large part, this is explained by the differences in the ranges of the different water stress indexes applied: in this study from 0.17 to 0.99 for the revised WSI, from 0.01 to 14 for the WSI_{HHEQ} and from 1.11 to 600 for the AWaRe methodology. Therefore, a major concern arises when selecting the final value to report or communicate. It is therefore considered critical that in the near term, operators of product water footprint programmes specify the particular water stress factors to be applied. This will, at least, enable comparability of product water footprints within a programme. If companies making water scarcity footprint claims have liberty to choose any WSI method, then the results going into the public domain will vary so much that there will be no meaningful basis for non-technical people to make any sensible comparisons.

However, over time, effort should also be made to achieve consistency between programmes. In this regard, consideration must be given to both reference unit and range. Using this dairy product case study as an example, consumers are likely to have a very different attitude to the product if its water scarcity footprint is reported as 2.8 L H_2Oe per kg compared to more than 6,000 L H_2Oe per kg. Water scarcity footprint results may need to be scaled so that they are comparable to other forms of direct water use that consumers are familiar with, e.g. flushing a toilet (3 to 6 litre) or taking a shower (7 to 15 L/min), operating a dishwasher (10-12 L per cycle). As such, the reference unit is most important. Expressing water footprints in relation to water consumption at the global average water stress index is helpful in this regard.

Another consideration is the variation in characterization factors between water rich and water scarce locations. If the contrast is too great, it could discourage water efficiency behaviors in water scarce regions because even small amounts of water use lead to very large water scarcity footprints. As a practical suggestion, the range in water stress index values should perhaps not exceed 100.

4.6.4. References

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Capítulo 5 | DISCUSIÓN GENERAL

En el presente apartado de discusión general se realiza una valoración integral en torno a la hipótesis planteada: Superar los retos de aplicación del Análisis de Ciclo de Vida en los productos alimentarios con el fin de evaluar y reducir el impacto ambiental asociado de una forma rigurosa y robusta. Los tres retos a los que la tesis ha dado respuesta son: reducir la variabilidad temporal, identificar unos métodos de caracterización de impactos representativos del sector alimentario y facilitar la aplicación de la metodología de ACV a las empresas.

5.1. ¿Cómo afecta la variabilidad temporal a los impactos ocasionados por la producción primaria?

A lo largo de las **contribuciones I, II y III** se analiza en la existencia de una variabilidad temporal en la productividad del sector primario agroalimentario no dependiente directamente de la tecnología utilizada. Esta variación en la productividad implica una variación en el impacto ambiental asociado a la unidad de producto. La razón y magnitud de estas variabilidades son diferentes en función del sector e incluso entre las diferentes especies productivas dentro de un mismo sector:

Así, en las producciones agrícolas las variaciones en las condiciones meteorológicas afectan directamente a la productividad de los cultivos. Debido a las variaciones climatológicas puede cambiar la productividad, incrementarse las plagas o variar las necesidades de riego de los cultivos (Hoogenboom, 2000) modificando así el esfuerzo relativo a la productividad y variando significativamente el potencial impacto ambiental por unidad de producto. Tal y como se muestra en el **contribución III** las variaciones en los sistemas productivos de un año a otro pueden llegar a significar más de un 20 % de variación en la productividad. Sin embargo, se ha demostrado que también las decisiones puntuales tomadas sobre el manejo del cultivo clave a la hora de determinar el potencial impacto ambiental de un producto.

En el sector pesquero, las propias variaciones intrínsecas de los stocks hacen que el esfuerzo invertido varíe significativamente de un año a otro (**contribución I y II**). Este esfuerzo se traduce sobre todo en horas de pesca invertidas lo que implica un mayor o menor consumo de fuel por unidad de producto desembarcado, respecto a años anteriores. Debido a que el factor más influyente a la hora de evaluar el impacto ambiental de las pesquerías es el consumo y combustión de fuel, esta variación implica un cambio significativo en el potencial de impacto ambiental de la actividad pesquera por especie desembarcada.

Por otro lado sistemas, más controlados como el sector ganadero y acuícola muestran menores variaciones entre los diferentes años, debido sobre todo a la industrialización de estos sistemas productivos. La cría de animales en cautividad ha reducido la incertidumbre de la productividad asociada a factores no contralados como el clima o las enfermedades. El uso de antibióticos, el ajuste de los piensos o la estabulación controlada en el caso de la ganadería son algunos de los factores de la industrialización que hacen más estable la relación entre el esfuerzo invertido y la producción en un periodo de tiempo determinado.

Con el fin de establecer un valor medio de impacto ambiental que sea representativo y robusto es necesario identificar si el grado de variabilidad existente en los ciclos de vida de los diferentes productos es significativo. En este apartado se concluye que para cada especie productiva a analizar sería necesario realizar una análisis conjunto DEA (data envelopment analysis) y ACV, tanto para probar la variabilidad interanual como para definir el número medio de años necesarios.

Dentro de los diferentes análisis que ofrece el DEA, con la realización de un análisis de ventana o *Windows analysis,* se consigue determinar la cantidad mínima de años necesaria para recopilar los datos de inventario indispensables para obtener mediante un ACV una media significativa de análisis de impacto ambiental para la especie o sector objetivo (Iribarren et al., 2015). La validez de esta esta metodología en la presente tesis ha sido probada en dos especies del sector pesquero: caballa y merluza. Para la caballa, se ha establecido una media móvil de 5 años con el fin de obtener un resultado significativo. Sin embargo, para la merluza se establece una media inferior, 3 años. Estas diferencias en la cantidad de años son debidas a las fluctuaciones intrínsecas de cada stock o especie, ya que la merluza atlántica es una especie de por sí más estable que la caballa. Asimismo, se concluye que sería necesario aplicar la metodología ACV+DEA a las diferentes especies productivas con el fin de establecer el número de años necesarios para obtener una media representativa del impacto ambiental asociado a la unidad de producto final.

5.2. ¿Qué método de caracterización de impactos es el más idóneo para medir el impacto ambiental de los productos alimentarios?

Debido a las características intrínsecas de la producción de alimentos detalladas en el capítulo 3 de metodología, las cadenas alimentarias son uno de los sistemas productivos más complicados a la hora de realizar evaluaciones ambientales con enfoque de ciclo de vida.

En el marco del proyecto SENSE (FP7 288974) donde se ha desarrollado una herramienta armonizada y robusta para facilitar a las empresas la evaluación ambiental de los productos alimentarios (SENSE-tool), se han seleccionado una serie de indicadores de impacto ambiental específicos para el sector alimentario. Como se indica en la **contribución IV** estos indicadores están basados a su vez en la metodología ILCD (JRC, 2013) desarrollada por la comisión europea y en el protocolo ENVIFOOD (Food SCP RT, 2013) donde se establecen las pautas a seguir para realizar una ACV para el sector alimentario.

Sin embargo, tal y como se indica en la **contribución IV**, algunos indicadores ambientales están muy influenciados por las condiciones específicas del lugar donde se esté realizando la actividad. Muchas de las substancias emitidas a lo largo de la producción de alimentos tienen una interacción *in situ* con el medio en el que son emitidas. Así, por ejemplo la emisión de un compuesto nitrogenado en un medio ya colmatado en nitrógeno, no tiene el mismo potencial de eutrofización que la emisión de la misma substancia en un medio con carencia de nitrógeno. Lo mismo ocurre con el impacto de estrés hídrico. Así, no es lo mismo extraer un litro de agua en una zona árida que de una cuenca con abundante disponibilidad de agua (Pfister et al., 2009; Ridoutt y Pfister, 2014). Por ello, muchos de los factores de caracterización globales que se ofrecen en las actuales metodologías de evaluación de impactos, no son representativos del potencial impacto que puede estar teniendo debido a esta interacción con el medio (Pfister et al., 2009; Ridoutt y Pfister, 2014) Indicadores como el potencial de eutrofización, acidificación, el estrés hídrico o la toxicidad son algunos de los impactos con mayor necesidad de regionalización.

A lo largo de la presente tesis se ha destacado la importancia de desarrollar una metodología específica sobre todo, para el potencial impacto de estrés hídrico **(Contribución V)**. Se ha seleccionado este indicador debido a su nivel de importancia en el estado Español y en otras regiones del mundo. El estudio se ha centrado en evaluar tres nuevas metodologías de impacto en una cadena láctea, desde la fabricación de los piensos hasta la comercialización de los productos lácteos. Como hemos podido comprobar, las tres metodologías seleccionadas ofrecen resultados relativos muy similares en cuanto a impacto causado por las diferentes etapas, ya que las tres coinciden en que el mayor impacto se concentra en el agua de riego utilizada en los cultivos de los piensos. Sin embargo, al evaluar los resultados absolutos, la metodología AWaRe aumento en un rango de 1000 los valores obtenidos con las otras dos metodologías. Se ha comprobado también que la metodología AWaRe ofrece unos factores de caracterización de impacto demasiado elevados, por lo que resulta muy difícil la interpretación de los resultados a nivel absoluto. Es necesario matizar también que la metodología WSI_{hhEQ} es un indicador del daño a los ecosistemas y salud humana ejercida por una actividad, mientras que las metodologías AWaRe y WSI ofrecen información sobre el potencial impacto de agotamiento de agua debido al consumo de agua. En este sentido, tal y como se concluye en la contribución, es necesario definir bien los objetivos del estudio y seleccionar la metodología que se ajuste más a los mismos.

5.3. ¿Cómo se puede facilitar a las empresas la evaluación ambiental?

La sostenibilidad de los productos alimentarios está tornándose como tema preferente de las actuales políticas europeas y estatales. La comisión europea con la iniciativa *Single Market for Green products* tiene como objetivo impulsar la evaluación ambiental de todos los bienes y servicios que se comercializan en la unión europea de una forma armonizada y robusta con el fin de promocionar la sostenibilidad de los productos. A nivel nacional, la Dirección General de la Industria Alimentaria del Ministerio de Agricultura, Alimentación y Medio Ambiente, está desarrollando el "Programa para la sostenibilidad integral de la industria agroalimentaria", cuyo objetivo general es mejorar las condiciones ambientales, sociales y económicas del sector, fomentando la eficiencia y la creación de valor de los productos agroalimentarios.

Por otro lado, los consumidores exigen cada vez más productos responsables con el medio ambiente (Vittersø y Tangeland, 2014). Factores como la contribución al cambio climático, el uso de pesticidas o los productos "eco" están siendo valorados positivamente por los consumidores. Sin embargo muchos de ellos se muestran confusos antes las múltiples etiquetas logos o auto-alegaciones, que en ocasiones no muestran un rigor científico (Vanhonacker et al., 2013; DeBoer et al., 2007).

Por todo ello, las empresas perciben la necesidad de incorporar criterios de sostenibilidad ambiental dentro de las políticas internas de Responsabilidad Social Corporativa (RSC). No obstante, para las empresas alimentarias europeas donde el 99% son PYMEs es muy complicado introducir el enfoque de ciclo de vida de sus productos. La falta de tiempo, recursos humanos y técnicos son las principales razones de la imposibilidad de manejar herramientas de evaluación ambiental. También el tedioso trabajo de recopilar tanto datos propios como datos de proveedores es otro de los impedimentos para promover la evaluación ambiental. Por otro lado muchas de ellas no ven un claro beneficio en comunicar tanto a los consumidores como a los diferentes agentes de la cadena de valor los potenciales impactos ambientales de sus productos.

Con el fin de facilitar a las empresas la evaluación ambiental de sus productos, a lo largo de la tesis y financiada también por el proyecto europeo SENSE (FP7 288974) se ha desarrollado una sistemática específica para evaluar el impacto ambiental de los productos alimentarios. Dentro de esta sistemática se han seleccionado tanto los aspectos clave de comportamiento de las empresas, una metodología de toma de datos estandarizada y un modelo de evaluación de impactos armonizada. Todo ello se plasmado en una herramienta informática, la herramienta SENSE-tool, que permite a las empresas evaluar de forma fácil y sencilla, pero a su vez obteniendo unos resultados robustos y significativos, el potencial impacto ambiental de sus productos a lo largo de la cadena.

Una vez evaluada la herramienta se ha procedido a su validación mediante el uso de la misma en 27 empresas de los sectores de acuicultura, lácteo y de zumos de frutas. Con todo lo recogido de las pruebas en las empresas llevadas a cabo dentro de la **contribución V**, se identificaron algunas ideas clave con el fin de mejor la herramienta. Por un lado, las empresas demandaban mayor conjunto de "*background*" data. La herramienta SENSE se nutre de datos de inventario proporcionados por la base de datos de ecoinvent, sin embargo muchos flujos de inventario no están disponibles. Por otro lado, y con el fin de validar los datos introducidos, se ha propuesto incorporar un sistema de certificación por terceros.

Por otro lado, la herramienta también se ha validado comparando los resultados obtenidos con la misma y los resultados de un ACV completo realizado con el software SIMAPRO. Los datos obtenidos de la comparativa verifican que, sobre todo para los impactos de cambio climático, acidificación y toxicidad, el SENSE-tool ofrece unos resultados muy similares al de un ACV completo (< 10 % de variación), ofreciendo unos resultados robustos de evaluación de impacto ambiental.

Una parte muy importante para incorporar de forma real estos nuevos sistemas de evaluación ambiental es la certificación. Los últimos años han proliferado mucho los sistemas de gobernanza privada, debido, sobre todo a una falta de regulación estricta por parte de la administración pública. De esta forma las empresas que promueven la mejora ambiental de sus procesos o productos pueden certificarse en iniciativas privadas (Casey, 2015).

Otra forma de reducir el impacto ambiental de los productos es incluir la variable ambiental desde la fase del diseño conceptual de los productos alimentarios, el llamado ECODISEÑO. Las razones para reducir el impacto pueden ser varias, desde la propia ética ambiental, hasta la reducción de costes asociada. Tanto en la guía de ecodiseño que se ha desarrollado dentro de esta tesis y en la **Contribución VI**, se plantean una serie de alternativas de minimización de impacto. Algunas de ellas podrían suponer un aumento de costes para las empresas, sin embargo Schiesser et al (2011) remarcan que la reducción de impacto ambiental de los productos alimentarios puede ser un verdadera oportunidad para las empresas. Mediante la validación del protocolo de ecodiseño realizada en la **contribución VI** los principales focos donde las empresas deberían invertir esfuerzos son:

- Mejora de la producción primaria: evitar el uso intensivo de fertilizantes y pesticidas, minimizar las labores de labranza, minimizar el esfuerzo pesquero, seleccionar piensos con locales o con menor impacto, etc.
- Reducción en el uso de materiales: ajustes de raciones, disminuir peso del envase, uso de materiales de fácil reciclaje, etc.
- Introducción de técnicas de producción más eficientes: optimizar los tratamientos térmicos, reducir los tiempos de refrigeración o congelación, uso de energías renovables, minimización de consumo de agua o reducir el desperdicio alimentario
- Promoción del consumo de alimentos sostenibles: consumo local, promover dietas con menor cantidad de carnes, reciclado del envase, ajustes de formato de venta o aumentar la vida útil

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Capítulo 6 | CONCLUSIONES Y PERSPECTIVAS FUTURAS

6.1. Conclusiones

A continuación se presentan las conclusiones generalas derivadas de la presente tesis:

1. La combinación de metodologías de ACV y el DEA-window analysis aplicadas en esta tesis se presenta como un método viable y útil para establecer el mínimo de años necesarios de captura de datos de inventario de cada especie, con el fin de obtener unos valores de impacto ambiental representativo y robusto.
2. Con el fin de reportar los impactos ambientales asociados a un producto concreto de una forma precisa y representativa, es necesaria la regionalización de los factores de caracterización de las metodologías de impacto con componente local como son la acidificación, eutrofización y estrés hídrico. Dicha regionalización supone una mejora substancial en la exactitud de los posteriores resultados de evaluación ambiental.
3. Disponer de un sistema armonizado, sencillo y robusto de evaluación ambiental de productos alimentarios, permite a las empresas la posibilidad de mejorar los procesos productivos, facilitar la toma de decisiones, compararse con otros productos o compararse con productos de la competencia y aportar al consumidor una información que facilite una decisión de compra basada en información veraz.
4. El establecimiento de un set de indicadores de comportamiento ambiental el cual abarcar el 90 % de los impactos ambientales asociados a un producto, permite obtener una evaluación simplificada y armonizada de una cadena de alimento y facilita a las empresas la obtención de datos de inventario.
5. Los impactos ambientales más relevantes para las cadenas y productos alimentarios identificados son: Cambio climático, agotamiento de la capa de ozono, eutrofización, acidificación, toxicidad humana, ecotoxicidad, agotamiento de recursos abióticos, agotamiento de agua y uso de suelo.
6. Utilizando la metodología de ecodiseño basada en el enfoque de ACV las empresas pueden reducir significativamente sus impactos ambientales asociados, ya que el 70 % del impacto asociado a un producto se define ya en las primeras etapas del diseño. Al mismo tiempo esta metodología permite también aumentar la eficiencia productiva, evitando a futuro una costosa inversión para la mejora global de la cadena.

6.2. Perspectiva Futura

El futuro de la evaluación ambiental de la industria alimentaria se presenta como un reto muy importante a para la sostenibilidad del planeta. Los esfuerzos llevados a cabo por los investigadores y en general por el mundo científico han avanzado considerablemente en los últimos 15 o 20 años y así, hoy día, es posible realizar una evaluación ambiental de producto fiable y robusta. Además, debido a la situación ambiental mundial, muchas administraciones y organismos públicos están apostando cada vez más por impulsar la sostenibilidad en todos los sectores y a todos los niveles.

Sin embargo, el uso de las metodologías planteadas en esta tesis y el establecimiento de las emergentes iniciativas está aún lejos de la implantación real. Las empresas alimentarias no confían en los métodos establecidos y, pese a que los consumidores exigen cada vez más productos sostenibles, no establecen la evaluación y comunicación ambiental en sus productos. Por ello, en el futuro próximo será necesario impulsar la evaluación ambiental de productos alimentarios y adaptar las herramientas actuales de evaluación para que los usuarios finales las puedan manejar de forma sencilla. De esta forma se podrán incorporar argumentos ambientales en la toma de decisiones de las empresas alimentarias.

Además, con el fin de ofrecer una información contrastada tanto al consumidor final, como a las administraciones y agentes interesados, es necesario avanzar en la creación de esquemas de certificación ambiental internacionales que cubran todos los aspectos del ciclo de vida de los productos, desde la producción primaria hasta el consumidor final, y que ofrezcan garantías a las empresas y consumidores. Estos esquemas de certificación deberán tener en cuenta la escala temporal y la regionalización de las bases de datos, así como la utilización de modelos de impacto específicos para el tipo de actividad.

Con el fin de garantizar también la estandarización de los resultados es imprescindible avanzar en la creación de bases de datos actualizadas y regionalizadas. Las bases de datos existentes hoy por hoy, no cubren toda la gama de alimentos, y menos aún las características de producción de los mismos. Por ello, es importante crear nuevas bases de datos asequibles y con mayor variedad de datos regionalizados.

Por último, y a nivel más metodológico, es fundamental avanzar en regionalizar los modelos de caracterización que dependen de las características específicas de un lugar como son el estrés hídrico, la eutrofización o la acidificación. También será necesaria la creación de nuevas categorías de impacto que cubran aspectos no incluidos en los actuales modelos de caracterización, como por ejemplo el potencial impacto en el suelo marino o la pérdida de biodiversidad.