Modelling location-dependent environmental impacts in life cycle assessment: water use, desertification and soil erosion

Doctoral thesis

Montse Núñez Pineda

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Application to energy crops grown in Spain

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A thesis submitted in fulfilment of the requirements for the European Mention of the PhD degree in Environmental Sciences and Technology

Sostenipra research group
Institut de Ciència i Tecnologia Ambientals (ICTA)
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IRTA RECERCA I TECNOLOGIA AGROALIMENTÀRIES

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Deja que sean tus pies los que te marquen el camino

Para mi padre,

Para mi hermano,

Y para mis madres, la de allá y las de aquí.

The present thesis entitled “Modelling location-dependent environmental impacts in life cycle assessment: water use, desertification and soil erosion. Application to energy crops grown in Spain” was carried out at the Institute of Agriculture and Food Research and Technology (IRTA), under the supervision of Dr. Assumpció Antón, from IRTA and the Department of Chemical Engineering at the Universitat Rovira i Virgili (URV), Dr. Pere Muñoz, from IRTA, and Dr. Joan Rieradevall, from the Institute of Environmental Sciences and Technology (ICTA) and the Department of Chemical Engineering at the Universitat Autònoma de Barcelona (UAB).

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Cabrils, April 2011

Dra. Assumpció Antón　　　　Dr. Pere Muñoz　　　　Dr. Joan Rieradevall

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Acronyms and abbreviations

A	Average perspective in EI99
BW	Blue water
CF	Characterisation factor
dGW	Delta green water
DNAP-Spain	Spanish Desertification National Action Programme
E	Egalitarian perspective in EI99
ECEC	Ecological Cumulative Exergy Consumption
EI99	Eco-indicator 99
ET_c	Crop evapotranspiration
ET_0	Potential evapotranspiration
GDP	Gross domestic product
GIS	Geographic information systems
GW	Green water
GWSI	Green water scarcity index
H	Hierarchist perspective in EI99
HWSD	Harmonized World Soil Database
I	Individualist perspective in EI99
K_c	Crop coefficients
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
NPP_{act}	Net primary production of the actual vegetation
NPP_0	Net primary production of the potential vegetation
NPPD	Net primary production depletion
Pr’	Effective precipitation or soil-water infiltration
PAF	Potentially affected fraction of species
PDF	Potential disappeared fraction of species
Pr	Precipitation
RUSLE	Revised universal soil loss equation
SSP On Cultivos	Singular and Strategic Project On Cultivos
SOC	Soil organic carbon
SOM	Soil organic matter
USLE	Universal soil loss equation

VC	Vegetative cover
WSI	Water stress index
YLD	Years lived disabled
YLL	Years of life lost

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A tall personal, l'Àngel Garcia m'ha assessorat en el disseny de la tesi i m'ha ajudat en moltes coses més...

Mi agradecimiento final para toda mi familia: la de casa y la de la oficina. Los del presente y los del pasado.

Resumen

Suelo y agua dulce son dos recursos imprescindibles para los ecosistemas y la humanidad. La agricultura depende de la disponibilidad de estos recursos, que por tanto, debe gestionar correctamente. En caso contrario, las prácticas agrícolas pueden provocar impactos adversos en el medio ambiente y poner en peligro la disponibilidad de suelo y agua para futuras actividades agrícolas. Los suelos agrícolas representan sólo el 12% de la superficie terrestre mundial. Sin embargo, aproximadamente el 70% de las extracciones de agua de la naturaleza se utilizan en la agricultura de irrigación y el 30-40% de los suelos destinados a la agricultura están degradados. La desertificación, entendida como la degradación irreversible del suelo, es uno de los mayores problemas para la sostenibilidad de las tierras áridas, áreas que cubren el 40% de la superficie terrestre. Por estos motivos, deben evaluarse los impactos ambientales debidos al uso del suelo y del agua en la agricultura.

El análisis de ciclo de vida (ACV) es un método para evaluar el perfil ambiental de sistemas productivos. El ACV se desarrolló inicialmente para estudiar la producción industrial, pero en los últimos años la investigación se ha dirigido a la adaptación del método para poder aplicarlo también en los sistemas agrícolas. La metodología convencional de ACV no determina los impactos ambientales debidos al uso del suelo y del agua, siendo ésta una importante deficiencia para evaluar el perfil ambiental de los sistemas agrícolas. Además, al contrario de otras categorías de impacto ambiental global, como el calentamiento global, los impactos ambientales derivados del uso del suelo y del agua son distintos en cada lugar del planeta, en función de las condiciones espacio-temporales del sitio. Por tanto, es necesario extender la metodología actual de ACV.

Esta tesis se centra en el desarrollo de la metodología de ACV para incorporar los impactos ambientales resultantes del uso del suelo y del agua. La variabilidad espacio-temporal de estos recursos se tiene en cuenta en los métodos propuestos utilizando la herramienta complementaria de los sistemas de información geográfica (SIG). Para el uso del agua, se presentan dos métodos de aproximación para medir los impactos debidos al consumo de agua de las reservas del suelo, cuando, hasta la fecha, los estudios han intentado evaluar los impactos ambientales debidos al consumo de agua para la irrigación. Para el uso del suelo, se propone una aproximación multi-indicador para modelar el impacto de la desertificación, una categoría nunca antes incluida en ACV, así como una metodología para incluir los impactos de la erosión del suelo, donde la pérdida de suelo se relaciona con la pérdida de carbono orgánico, como medida de la calidad del suelo, y finalmente, con la disminución de producción de biomasa de los ecosistemas. Los

métodos desarrollados comprenden las fases de inventario de ciclo de vida (ICV) y de evaluación de impacto de ciclo de vida (EICV).

Además, para comprobar la aplicabilidad de los métodos regionalizados de ACV y de los factores de caracterización desarrollados, estos se aplican en rotaciones de cultivos con cultivos energéticos en España, con el objetivo de cuantificar los efectos colaterales de producir bioenergía sobre los recursos suelo y agua, muy disputados en el país. Los resultados revelan que no hay una solución idónea, con una rotación de cultivos sembrados en una zona específica del país, que sea capaz de reducir, simultáneamente, los impactos ambientales debidos al uso de suelo y agua. Esto se debe, en primer lugar, a que los cultivos de secano muestran mayores impactos relacionados con el uso del suelo, pero, al contrario, no utilizan agua de irrigación. Y en segundo lugar, a que las zonas con más reservas de agua en superficie, acuíferos y suelos están también sometidas a lluvias más intensas y erosivas, y en consecuencia, a un mayor deterioro del suelo.

Entre otras importantes líneas de investigación a seguir, próximos trabajos deben centrarse en el estudio de unidades funcionales adecuadas para el ACV de sistemas agrícolas, el cálculo de las incertidumbres de los métodos desarrollados en la tesis, así como en la identificación de una escala geográfica significativa y de aplicación factible que aborde la diferenciación espacial de los factores de caracterización para los impactos del uso del suelo y del agua, y, en general, para cualquier categoría de impacto ambiental regional.

Resum

Sòl i aigua dolça són dos recursos imprescindibles pels ecosistemes i la humanitat. L'agricultura depèn de la disponibilitat d'aquests recursos que, per tant, ha de gestionar correctament. En cas contrari, les pràctiques agrícoles poden provocar impactes adversos en el medi ambient i posar en perill la disponibilitat de sòl i aigua per a futures activitats agrícoles. Els sòls agrícoles representen només el 12% de la superfície terrestre mundial. Malgrat això, aproximadament el 70% de les extraccions d'aigua de la natura s'utilitzen en l'agricultura d'irrigació i el 30-40% dels sòls destinats a l'agricultura estan degradats. La desertificació, entesa com la degradació irreversible del sòl, és un dels problemes més greus per a la sostenibilitat de les terres àrides, àrees que ocupen el 40% de la superfície terrestre. Por aquests motius, s'han d'avaluar els impactes ambientals derivats de l'ús del sòl i de l'aigua en l'agricultura.

L'anàlisi de cicle de vida (ACV) és un mètode per avaluar el perfil ambiental de sistemes productius. L'ACV es va desenvolupar inicialment per estudiar la producció industrial, però en els últims anys la investigació s'ha dirigit a l'adaptació del mètode per a poder aplicar-lo també en els sistemes agrícoles. La metodologia convencional d'ACV no determina els impactes ambientals deguts a l'ús del sòl i de l'aigua, essent aquesta una important deficiència per a avaluar el perfil ambiental dels sistemes agrícoles. A més a més, al contrari que altres categories d'impacte ambiental global, com l'escalfament global, els impactes ambientals derivats de l'ús del sòl i de l'aigua són diferents a cada lloc del planeta, en funció de les condicions espaials i temporals de la localització. Per tant, es fa necessària l'extensió de la metodologia actual d'ACV.

Aquesta tesi es centra en el desenvolupament de la metodologia d'ACV per a incorporar els impactes ambientals resultants de l'ús del sòl i de l'aigua. La variabilitat espaial i temporal d'aquests recursos es considera en els mètodes proposats utilitzant l'eina complementària dels sistemes d'informació geogràfica (SIG). Per a l'ús de l'aigua, es presenten dos mètodes d'aproximació per a mesurar els impactes deguts al consum d'aigua de les reserves del sòl, quan, fins a la data, els estudis han intentat avaluar els impactes ambientals deguts al consum d'aigua d'irrigació. Per a l'ús del sòl, es proposa una aproximació multi-indicador per a modelar l'impacte de la desertificació, una categoria mai inclosa abans en ACV, així com una metodologia per a incloure els impactes de l'erosió del sòl, on la pèrdua del sòl es relaciona amb la pèrdua de carboni orgànic, com a mesura de la qualitat del sòl, i finalment, amb la disminució de producció de biomassa dels ecosistemes. Els mètodes desenvolupats comprenen les fases d'inventari de cicle de vida (ICV) i d'avaluació d'impacte de cicle de vida (AICV).

A més, per a comprovar l'aplicabilitat dels mètodes regionalitzats d'ACV i dels factors de caracterització desenvolupats, aquests s'apliquen en rotacions de conreus amb conreus energètics a Espanya, amb l'objectiu de quantificar els efectes col·laterals de produir bioenergia sobre els recursos aigua i sòl, molt disputats al país. Els resultats evidencien que no hi ha una solució idònia, amb una rotació de conreus sembrats en una zona específica del país, que sigui capaç de reduir, simultàniament, els impactes ambientals derivats de l'ús de sòl i d'aigua. Això es deu, en primer lloc, a què els conreus de secà tenen impactes més grans relacionats amb l'ús del sòl, però, al contrari, no utilitzen aigua d'irrigació. I en segon lloc, a què les zones amb més reserves d'aigua en superfície, aqüífers i sòls estan també sotmeses a les pluges més intenses i erosives, i en conseqüència, a un deteriorament més gran del sòl.

Entre altres importants línees d'investigació a seguir, pròxims treballs s'han de centrar en l'estudi d'unitats funcionals adequades per a l'ACV de sistemes agrícoles, el càlcul de les incerteses dels mètodes desenvolupats a la tesi, així com en la identificació d'una escala geogràfica significativa i d'aplicació factible que abordi la diferenciació espaial dels factors de caracterització pels impactes de l'ús del sòl i de l'aigua, i, en general, per a qualsevol categoria d'impacte ambiental regional.

Abstract

Soil and freshwater are two absolutely essential resources for ecosystems and humanity. Agriculture depends very much on these resources, and so, without their correct management, farmland practices can trigger many adverse impacts on the environment and jeopardise the availability of soil and water for future agricultural activities. Agricultural lands represent only 12% of the world's land area. However, roughly 70% of water withdrawals from nature are for irrigated agriculture and 30-40% of the agricultural land is affected by soil degradation. Desertification, irreversible soil degradation, is one of the main problems for sustainability in drylands, areas that cover 40% of the earth's surface. For these reasons, the environmental impacts of the use of water and land by agricultural activities should be measured.

Life cycle assessment (LCA) is a method to construct the environmental profile of production systems. It was initially developed for industrial production, but a considerable amount of research has been undertaken in recent years to adapt LCA to agricultural systems as well. Conventional LCA methodology does not determine the environmental impacts of water and land use, which is a very significant shortcoming when evaluating the environmental performance of agricultural systems. Furthermore, contrary to other global environmental impact categories such as global warming, the environmental impacts of water and land use vary in every location of the globe, depending on the spatio-temporal conditions of the location, requiring therefore an extension of current LCA methodology.

This thesis focuses on the development of the LCA methodology to incorporate the environmental impacts arising from the use of water and land. The spatio-temporal variability of these resources is taken into account in the proposed methods using the complementary tool of geographic information systems (GIS). For water use, two screening frameworks are built to capture the impacts of soil-water consumption by plants, when, until now, efforts have been directed towards evaluating the environmental impacts of irrigation water consumption. For land use, a multi-indicator approach for a new impact category, desertification, until now never modelled in the LCA context, is provided, as well as a methodology for including soil erosion impacts, in which the soil loss has been related to the loss of organic carbon, as a measure of the soil quality, and finally, to the loss of biomass productivity of ecosystems. The methods developed deal with the life cycle inventory (LCI) and the life cycle impact assessment (LCIA) phases.

In addition, to verify the applicability of the developed location-dependent methods and characterisation factors, these are applied to agricultural crop rotations with energy crops

growing in Spain, with the aim of quantifying the side effects of producing bioenergy on the disputed water and land resources in the country. The outcomes indicate that there is no best solution of a single crop rotation grown in a specific location capable of minimising water and land use environmental impacts simultaneously. This is because, firstly, rainfed crop rotations exhibit higher land use related impacts but, in contrast, they are not irrigated. And secondly, locations with more surface, ground and soil water reserves are subjected to more intensive and erosive rainfalls, thus, to higher land use damages.

Among other important follow-up lines of research, future work should focus on the study of suitable functional units for agricultural LCA, calculate the uncertainties of the developed methods as well as try to identify a feasible and relevant geographical scale at which to address the spatial differentiation of the characterisation factors for water and land use impacts, and in general, for any location-dependent impact category.

Preface

This thesis belongs to the Environmental Sciences and Technology PhD programme of the Universitat Autònoma de Barcelona. The work was carried out within the research group on Sustainability and Environmental Prevention (Sostenipra) at the Institute of Agriculture and Food Research and Technology (IRTA) of the regional Government of Catalonia, from May 2007 to April 2011.

The thesis aims to contribute to the methodological development of LCA, namely providing new methods to incorporate non-global environmental impacts, the ecological consequences of which depend on regional conditions, such as climate, soils or the ecosystem patters of the location. These are biogeograhical factors traditionally overlooked in existing LCA methods.

The thesis is essentially based on the following papers, which have either been published or accepted or are under review in international peer-reviewed journals:

- Núñez M, Civit B, Muñoz P, Arena AP, Rieradevall J, Antón A (2010) Assessing potential desertification environmental impact in life cycle assessment. Part 1: methodological aspects. International Journal of Life Cycle Assessment 15 (1): 67-78.

- Núñez M, Pfister S, Antón A, Muñoz P, Hellweg S, Koehler A, Rieradevall J. Assessing the environmental impacts of water consumption on energy crops grown in Spain. Accepted to appear in Journal of Industrial Ecology.

- Núñez M, Muñoz P, Rieradevall J, Antón A. Assessment methodology to establish new USLE cover and management C-factors. Case study on food and energy crops. Submitted in January 2011 to Agronomy for Sustainable Development.

- Núñez M, Antón A, Muñoz P, Rieradevall J. Inclusion of soil erosion impacts in life cycle assessment: application to energy crops in Spain. Submitted in February 2011 to Environmental Science & Technology.

In addition, the work included in the thesis was presented in several oral communications and posters in national and international conferences:

- Núñez M, Antón A, Muñoz P, Rieradevall J (2007) Incorporació d'eines d'anàlisi ambiental en la producció hortícola. 5th conference of the Institució Catalana d'Estudis Agraris (ICEA), 4-6 July 2007, Castelldefels, Barcelona, Spain.

- Núñez M, Antón A, Rieradevall J, Muñoz P (2008) Assessment methodology of the USLE C-factor to include land erosion impact in LCA. 6th international conference on life cycle assessment in the agri-food sector (LCA food), 12-14 November 2008, Zurich, Switzerland.

- Núñez M, Muñoz P, Rieradevall J, Antón A, Carrasco J (2009) Identifying the most suitable places for bioenergy crop growth in Spain in relation to soil erosion. 3rd international conference on life cycle assessment in Latin America (CILCA), 27-29 April 2009, Pucón, Chile.

- Núñez M, Civit B, Muñoz P, Arena AP, Rieradevall J, Antón A (2009) A framework for assessing potential desertification environmental impact in LCA. 3rd international conference on life cycle assessment in Latin America (CILCA), 27-29 April 2009, Pucón, Chile.

- Antón A, Núñez M, Milà i Canals Ll, Pfister S, Muñoz P (2009) Desarrollo de una categoría de impacto para evaluar la utilización del agua mediante el análisis de ciclo de vida. 7th meeting of the Spanish Network of LCA, 7-8 June 2009, Santiago de Compostela, Spain.

- Núñez M, Pfister S, Antón A, Muñoz P, Hellweg S, Koehler A, Rieradevall J (2010) Using geographic information systems and spatial data to assess water consumption of bioenergy crops in life cycle assessment. 20th SETAC Europe annual meeting, 23-27 May 2010, Seville, Spain.

- Núñez M, Antón A, Muñoz P, Rieradevall J, Carrasco J (2010) Contribution to modelling soil erosion and water consumption in life cycle assessment of agricultural systems. 7th international conference on life cycle assessment in the agri-food sector (LCA food), 22-24 September 2010, Bari, Italy.

Part of the research was conducted during a three-month stay (October 2008 – January 2009) at the Institute of Environmental Engineering of the Swiss Federal Institute of Technology (ETH), in Zurich, Switzerland.

Furthermore, during the period of the thesis, research focusing on environmental fields other than the topic of this thesis was also carried out. This parallel research was published in peer-reviewed journals or book chapters:

- Oliver-Solà J, Núñez M, Gabarrell X, Boada M, Rieradevall J (2007) Service sector metabolism. Accounting for energy impacts of the Montjuïc urban park in Barcelona. Journal of Industrial Ecology 11 (2): 83-98.

- Muñoz P, Antón A, López M, Huerta O, Núñez M, Rieradevall J, Ariño J (2008) Aplicación de compost de fracción orgánica de residuos sólidos municipales en la fertilización de cultivos hortícolas en la comarca del Maresme. In Subvenciones de I+D+i en el ámbito de la prevención de la contaminación. Balance 2004-2007, Ministerio de Medio Ambiente (ed.), Madrid, Spain, pp 45-51, ISBN 978-84-8320-420-7.

- Núñez M, García-Lozano R, Boquera P, Gabarrell X, Rieradevall J (2009) Temporary structures as a generator of waste in covered trade fairs. Waste Management 29 (7): 2011-2017.

- Núñez M, Martínez J, Muñoz P, Antón A, Rieradevall J (2009) Estudios preliminares de evaluación de impacto ambiental global en la aplicación de compost como fertilizante en cultivos de tomate al aire libre y en invernadero. In Proceso y destino del compost. Formación, información e interrelaciones entre los agentes del sector, Huerta O, López M, Martínez FX (eds.), Barcelona, Spain, pp 184-189, ISBN 978-84-692-1993-5.

- Núñez M, Oliver-Solà J, Rieradevall J, Gabarrell X (2010) Water management in integrated service systems: accounting for water flows in urban areas. Water Resources Management 24 (8): 1583-1604.

Structure of the thesis

The thesis is organised into 7 chapters, with the main contents summarised below. For clarity, the structure of the thesis is outlined in figure A. As shown in Figure A, chapter 1 sets out an introduction to the most important piece of the thesis, which consists of four chapters, from chapter 2 to 5. The final chapters 6 and 7 state the general conclusions of the thesis and the lines of future work. Figure A can be used during the reading of the manuscript as a *thesis map*.

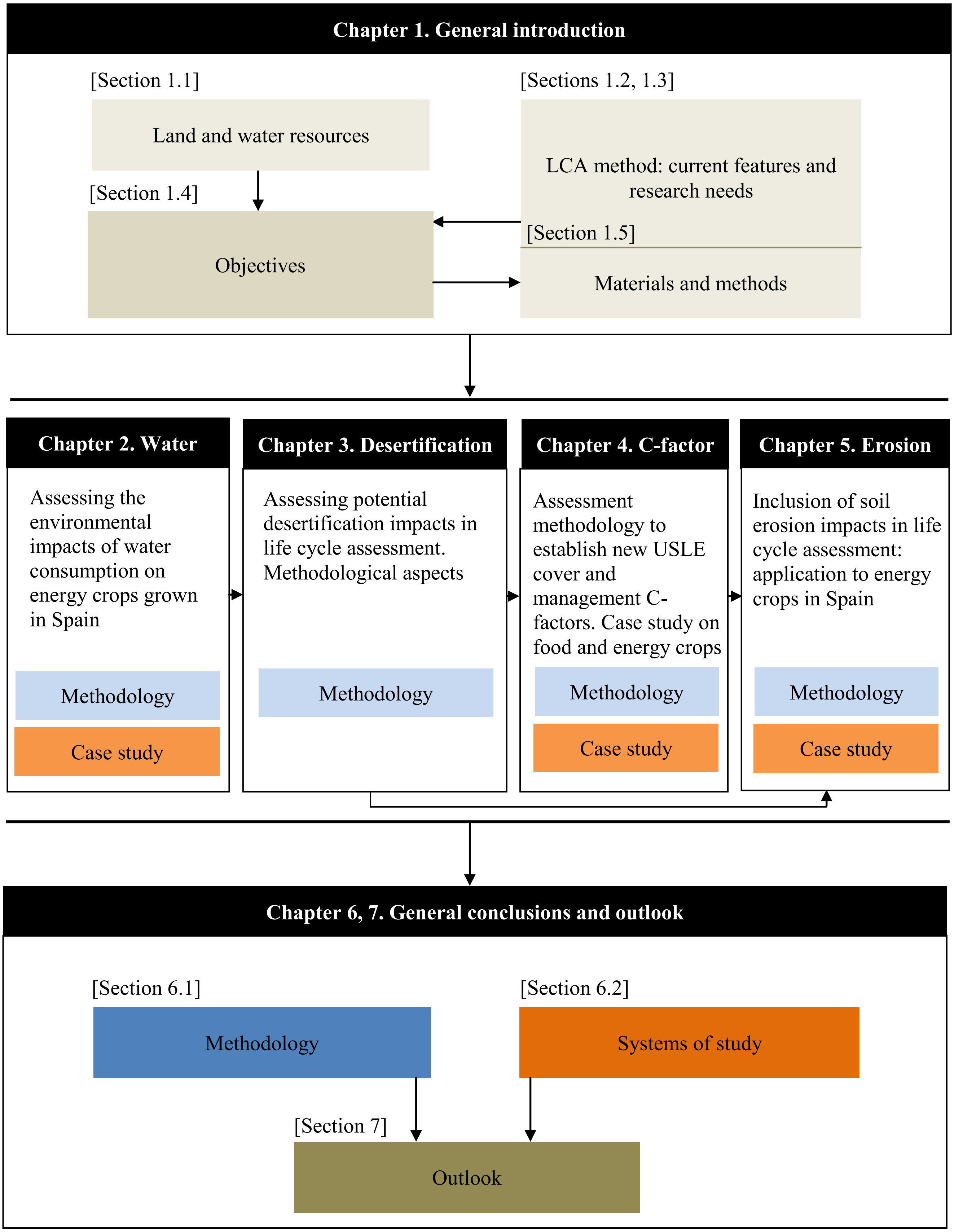


Figure A Flow chart of the thesis structure.

Chapter 1 [*Introduction*] presents an overview of the topic of land and water use, focusing on the available resources and the different human functionalities they currently cover. Particular attention is paid to the agricultural sector, as it is a key activity worldwide managing a large part of the land and water resources, and therefore greatly influencing their environmental condition. Next, it outlines the framework of the life cycle assessment (LCA) environmental tool, which is the base methodology used in this thesis. LCA is a comprehensive quantitative environmental method that is widely applied, but that still has some shortcomings which must be overcome. These shortcomings, which have motivated the development of this thesis, are then highlighted, paying particular attention to the study of agricultural systems. Chapter 1 follows with the objectives of the thesis and subsequently presents a description of several methodological aspects which are very important for the understanding of the subsequent sections, namely specific aspects of the LCA method and the usefulness of geographic information system (GIS) tools applied together with LCA. This chapter ends with a description of the systems of study considered throughout the research.

Chapter 2 [*Assessing the environmental impacts of water consumption on energy crops grown in Spain*] addresses the issue of freshwater consumption in LCA. To deal with this topic, water consumption and the environmental impacts of growing four agricultural crop rotations with food and energy crops are evaluated. These rotations are also compared to two non-energetic rotations of reference. Two origins of water are distinguished in the assessment and therefore separately evaluated: blue water, used to irrigate and coming from surface and ground water bodies, and green water, stored as soil moisture from rainfall that does not become runoff. This differentiation between water types results in different impacts for irrigated and rainfed crops. As the selection of the functional unit in LCA is a controversial issue, the analysis is carried out taking two perspectives: one that considers the use of a specified land area (m^2) and the other the production of crops (kg). Finally, the assessment of water consumption is complemented by other environmental indicator, the use of land, that provides a quantification of the transfer of water/land use impacts between irrigated and rainfed agriculture. The obtained results show the interrelationship between land and water as well as the importance of taking into account both indicators at the same time to avoid the transfer of impacts from one impact category to the other.

The following three chapters, from chapter 3 to chapter 5, deal with land use aspects. **Chapter 3** [*Assessing potential desertification impacts in life cycle assessment. Methodological aspects*] focuses on the use of land in arid climates. These areas are especially vulnerable to poor land management practices because, after degradation, ecosystems cannot recover due to their low soil resilience capacity. This chapter illustrates the development of a method to consider the desertification impact in LCA. The proposed method, built following a multi-indicator approach, requires the estimation of four environmental variables linked to the activity evaluated and the location where it takes place: aridity index, soil erosion, aquifer overexploitation and fire risk. The chapter also explains the procedure to estimate these variables and the model to calculate the environmental impact within the LCA framework.

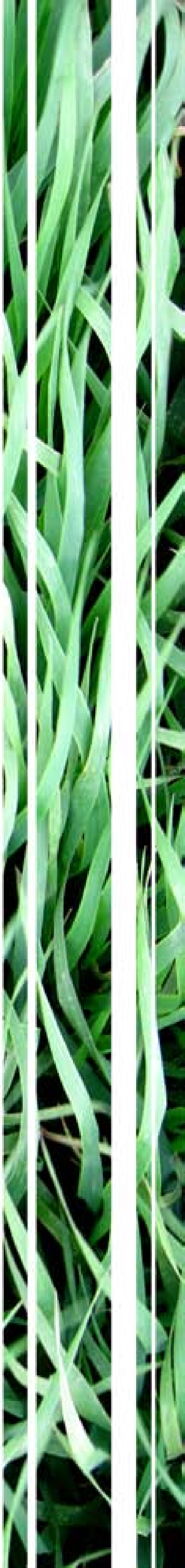
Chapter 3 provides evidence of the importance of soil erosion at the global level. One of the most widely methods applied for estimating soil losses by water is the universal soil loss equation (USLE). This method, used in LCA to gather information of the analysed system, cannot always be applied. This is because one of the variables of the equation, namely the cover and management factor (C-factor), is only available for a limited number of cropping systems and management practices. To overcome this situation, **chapter 4** [*Assessment methodology to establish new USLE cover and management C-factors. Case study on food and energy crops*] develops a method to derive new C-factors for crops and management practices that have not yet been calculated. The developed method is illustrated step by step with a case study of a two year rotation of a food and an energy crop.

Chapter 5 [*Inclusion of soil erosion impacts in life cycle assessment: application to energy crops in Spain*] presents a follow-up of chapter 4. While the previous chapter focuses on calculating soil losses under a certain activity, this one puts forward, under the umbrella of the LCA, a model to quantify the environmental consequences of these soil losses on both ecosystem quality and the availability of soil resources. As was done throughout the thesis, the model is tested with five agricultural rotations, two of them with energy crops, which are compared to the non-energetic crop rotation systems studied.

Finally, **chapter 6** [*Conclusions*] and **chapter 7** [*Outlook*] conclude the thesis. Chapter 6 provides the most important and general conclusions of the research. Conclusions are organised into two sections: those related to the methodology, which describe the improvements of the LCA method for the impact categories land and water use achieved with this thesis; and those related to the application of the new LCA methods to the systems of study. Finally, chapter 7 presents some recommendations for further work, to continue enhancing the integration of land and water use impacts in the LCA environmental tool.

Chapter 1.

Introduction



Soil and freshwater are two absolutely essential resources for ecosystems and humanity. Both land and water use have been traditionally considered local environmental issues, but they are becoming forces of global importance. Land provides critical natural resources and ecosystem services, such as food, fibre and freshwater, but current land use practices are degrading the natural environment throughout the globe. Also, the combination of increased water use and the expected effect of climate change on the hydrologic cycle may have consequences worldwide. The sustainable management of land and water resources is therefore a priority, especially when they are intensively used, such as in the case of agricultural activities. In this regard, although modern agriculture has been successful in increasing food production, it has also caused extensive environmental damage to water and soil reserves. Taking into account these (and other) environmental impacts in desision-making processes should be necessary in order to carry out a comprehensive evaluation of alternatives. One of the strategies most applied globally to evaluate the sustainability of products and processes throughout their life cycle, "*from cradle to grave*", is the life cycle thinking. This environmental strategy aims to minimise the negative impacts while avoiding transferring the problem from one life cycle stage of a product or process to another. Among the tools offered by life cycle thinking, life cycle assessment (LCA) represents one of the most thorough and sound methods. In fact, LCA has been broadly developed and applied during the last decades. However, the method still has some limitations that need to be addressed with further research. Two of these are related to the use of soil and freshwater, which have been overlooked in LCA studies and which, as stated before, are fundamental for biosphere and human needs. This thesis is a contribution to the development of methods to assess land and freshwater use in LCA, with particular attention to the use of these resources in agricultural activities, in particular to the cultivation of energy crops in a Mediterranean country, Spain. To fulfil the methodological objective of improving the LCA method for land and water use, the potentialities of spatial analysis brought by geographic information system tools (GIS) are used in the LCA environment.

Chapter 1 presents an introduction to the thesis topic and is structured as follows:

Land and water resources: use and environmental status

Outline of the LCA environmental tool. LCA applied to agricultural systems

Current challenges of the LCA method. Motivation of the thesis

Objectives of the thesis

Materials and methods: LCA aspects and GIS

1.1 Land and water resources

Land and freshwater are two limited resources which are crucial to all forms of life and provide the basis for many human activities. Herein the importance of an environmentally sustainable use and management of land and water that allows for preservation for future users. In this section, current land uses and environmental threats to the conservation of soil reserves are first described. Subsequently, this same frame of analysis is applied to the study of water resources.

1.1.1 Land use and environmental status of soil resources

According to the FAO/UNEP organisation, land use is characterised by the arrangements, activities and inputs people undertake in a certain land cover to produce, change or maintain it (Di Gregorio and Jansen 2000). Land cover is described as the physical coverage of land and it is the expression of human activities. Statistics for the year 2005 showed that, globally, cultivated land covered about 12% of the land area, permanent pastures 25%, forests and woodlands 33%, urban areas 1.5%, and the remainder was devoted to other uses (World Factbook 2011). This land area represents 29% of the total world area, while water occupies the other 71%.

In Europe, for the year 2006, a larger proportion of soil was covered with crops (25%), forests (35%) and urban areas (4%) at the expense of permanent pastures (17%) (figure 1).

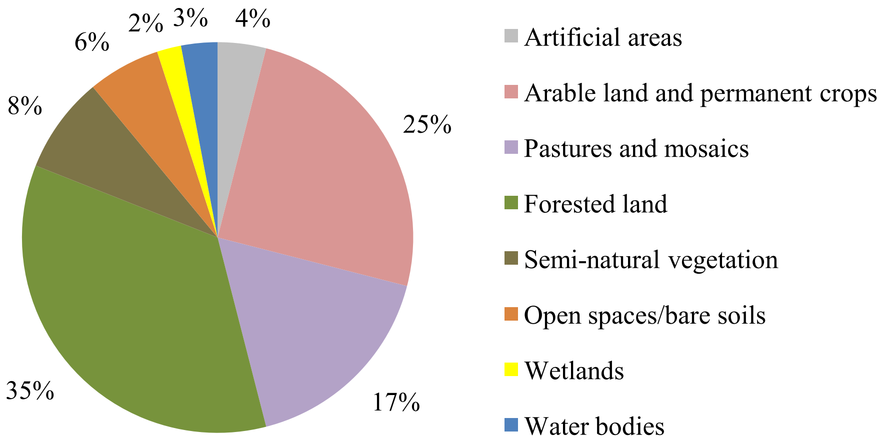


Figure 1 Share of land cover types in Europe (% of total). Year 2006 (EEA 2010a).

As can be seen in figure 1, farmers manage more than 40% of the European land area (25% of arable land and permanent crops plus 17% of pastures and mosaics).

In Spain, according to the Corine land cover classification (EEA 2005), the proportion of land cover types are in line with those for Europe, with more than 40% of the country being cultivated land. However, here, a much larger area is covered with semi-natural vegetation (26% in Spain versus 8% in the whole of Europe) at the expense of forested land (18% in Spain versus 35% in Europe).

From the abovementioned data, it can be observed that the agricultural sector is a key activity in Europe and Spain and, if not correctly managed, it can trigger adverse effects on the environment. The loss of traditional farming practices to spread intensive agriculture has led to many environmental problems, of which the European Environment Agency (EEA 2011a) highlights soil erosion, water pollution, over-exploitation of water resources, loss of biodiversity, pesticide-born damage and risk for human health. The adverse impacts of inappropriate agricultural land management affect the capacity of the land to function effectively within an ecosystem. This phenomenon is denoted as **land degradation**, and according to FAO/UNEP/UNESCO (1979), it concers

> *changes in the physical, chemical and biological properties of the soil, which affect the capacity of the land to provide ecosystem goods and services.*

Soil degradation affects around 30% of the world's irrigated lands and 40% of rainfed agricultural lands (UNEP/US NASA/World Bank 1998).

When degradation affects drylands, the phenomenon is called **desertification**. According to the United Nations (Holtz 2003), desertification means

> *degradation of land and vegetation, soil erosion and the loss of topsoil and fertile land in arid, semi-arid and dry sub-humid areas*[1]*, caused primarily by human activities and climatic variations*.

[1]Arid, semi-arid and dry sub-humid areas means areas, other than polar and sub-polar regions, in which the ratio of annual precipitation to potential evapotranspiration falls within the range of 0.05 to 0.65 (United Nations 1994).

Land over-exploitation, overgrazing, mechanised farming, bad irrigation practices, illegal and excessive logging, bush and forest fires and deforestation are human activities that can cause desertification. Along with these human activities, a range of climatic factors are believed to influence the process of land degradation, such as high variability in rainfall or recurrent drought (Holtz 2003). The main consequences of desertification are the inability of already degraded lands to be recovered and the expansion of the desert area. Drylands cover 40% of the earth's surface and between 10% and 20% of these areas are degraded or unproductive. As stated in section 3.4.1, the highest desertification risk is found in Africa, some Arab countries, Australia, the western edge of South America, southwest China and several Mediterranean countries.

Before using any land area for a specific purpose, it is usually necessary to change its initial properties to obtain a land suitable for the activity. As soil is a limited resource, tensions rise almost everywhere between resource users for occupying the same territorial resource. This conflict of interests leads to **changes in land use and land cover**, altering landscape, environment and the quality of land, and leaving large and often irreversible land-use impacts (EEA 2011b).

One associated environmental aspect of land use is the freshwater demand. The type of land use directly affects the freshwater supplies through withdrawals and diversions. On the other hand, land use changes disrupt the regional water balance by modifying the rainwater available for other users through changes in runoff, infiltration and evapotranspiration flows (Foley et al. 2005). Therefore, **dealing with land use impacts implies also dealing with freshwater use impacts**.

1.1.2 Freshwater use and environmental status of freshwater resources

Freshwater resources include surface and ground water systems. **Surface freshwater** systems – rivers, lakes and wetlands– occupy only 1% of the earth's surface area and contain 0.3% of the overall water resources in the world (including oceans and seas, permanent snow and ice caps). Two types of **groundwater** bodies are differentiated: **shallow aquifers**, which are replenished by surface runoff, and **fossil** (or deep) **aquifers**, which are not connected to runoff, thus they are non-renewable. Groundwater extraction contributes approximately 20% to global freshwater use (Penning de Vries et al. 2003).

The International Water Management Institute (Penning de Vries et al. 2003) pinpoints three major threats for freshwater-dependent ecosystem conservation: fragmentation of rivers, water withdrawal and pollution of surface and ground water by agricultural and industrial chemicals and animal wastes. Withdrawal can lead to river desiccation or reduced flow during the dry season.

Roughly **70% of water withdrawals from nature are for irrigated agriculture**, with the remainder being for domestic, industrial and hydropower uses (FAO 2008). In Europe, 45% of water abstraction is for cooling in energy production, followed by agriculture, 22%; public water supply, 21%, and industry, 12% (EEA 2010b). This sectoral average water use differs significantly within Europe (figure 2). For example, in southern European countries agriculture accounts for more than half of the total water abstraction, even reaching as much as 80% in Spain, where the remaining 15% is for the domestic sector and only 5% for the industrial sector (MARM 2008).

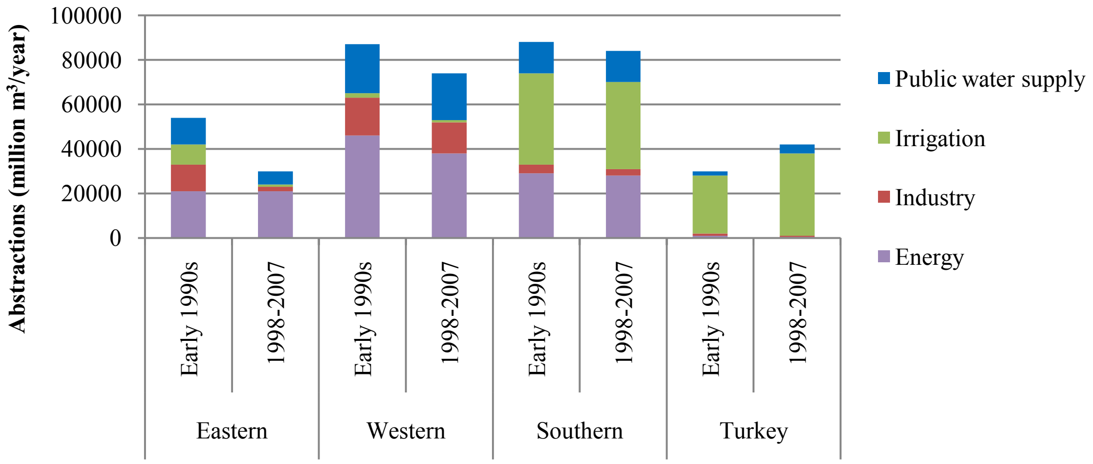


Figure 2 Sectoral water use across Europe in the early 1990s and 1998-2007. Million m^3 y^{-1} (EEA 2010b).

Freshwater is an abundant resource. However, as water and populations are unequally distributed, even nationally, water scarcity is a problem in many regions of the world. The United Nations Water Programme (United Nations 2006) defines **water scarcity** as

> *the point at which the aggregate impact of all water users affects the quantity or quality of water under prevailing institutional arrangements to the extent that the demand by all sectors, including the environment, cannot be satisfied.*

Some studies predict that regional water shortage problems will worsen in the future due to the increasing human population and the projected climate change (Alcamo et al. 2000; Falkenmark and Rokström 2004).

The **water scarcity situation will also be determined by the forthcoming uses of land**, since, as stated in 1.1.1, the amount of water required in a specific area depends on the activity developed there. In this regard, optimistic scenarios suggest that, by 2050, 30-40% more freshwater will be used to irrigate crops, due to an increase of the agricultural land area. If, however, agricultural water productivity does not improve, agricultural water demands will grow by 70-90% (De Frailure et al. 2007).

Water scarcity has many economic impacts on agriculture, particularly reducing crop yields. As the largest worldwide water consumer presently and in the future, agriculture is the core sector for improving water management and availability.

One of the disciplines that can be used to study the environmental impacts of human activities, such as those from agricultural systems is LCA. This environmental tool is further explained in the following section.

1.2 Outline of the life cycle assessment environmental tool

In this section, an overview of the LCA methodology is outlined. Firstly, a general picture of the framework of this environmental assessment tool is provided, and then the main difficulties encountered when applying the LCA method to agricultural activities are presented.

1.2.1 The life cycle assessment methodology

LCA is a tool to evaluate the environmental performance of products (goods and services), which provide one or more functions. LCA takes into account a **product's full life cycle**, from the extraction of resources and processing of raw material, through production, use and recycling, to the disposal of remaining waste. LCA studies thereby help to avoid resolving one environmental problem while creating others (JRC 2010a). Ideally, all relevant resources consumed (including energy) and emissions released into the air, water and soil to fulfil the function(s) are identified and quantified. Afterwards, the potential contribution of these resource consumption and emissions to several types of environmental impacts are evaluated. Environmental impacts are related to the function(s) of the product, being the basis for comparisons.

The ISO 14040 (2006) and 14044 (2006) standards provide the indispensable framework for LCA. The international life cycle data system (ILCD) handbook (JRC 2010a,b) provides the technical basis for consistent and quality assured LCA data and studies, conforming to the ISO standards. The standardised LCA framework consists of four phases (ISO 14040 2006), as outlined in figure 3: i) goal and scope definition, ii) inventory analysis, iii) impact assessment, and iv) interpretation.

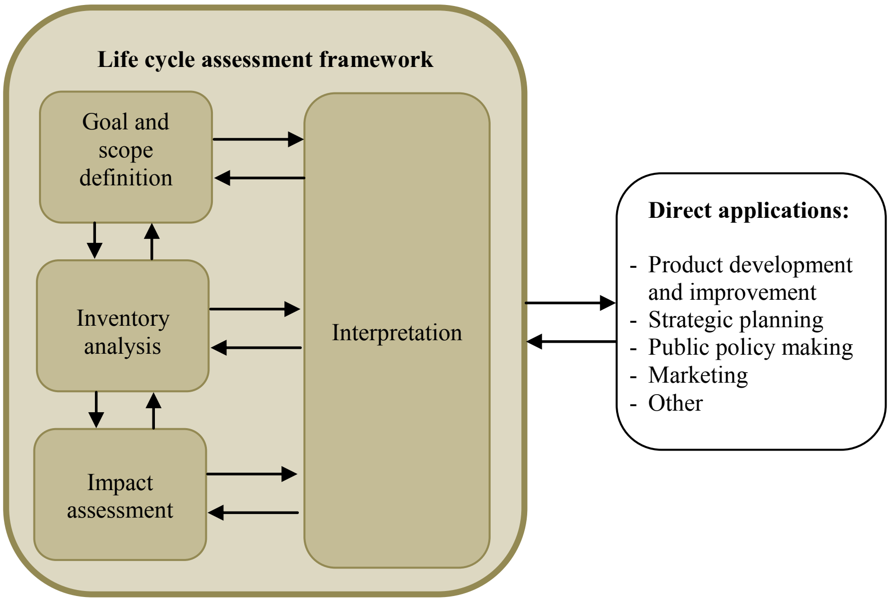


Figure 3 The phases of LCA according to ISO 14040 (2006).

From figure 3 it can be seen that LCA is not a single sequence, but it follows an iterative process, in which the level of detail can be increased with subsequent iterations.

i. **Goal and scope definition**: designed to obtain the required specifications for the LCA study. It includes the following steps:

   Definition of the purpose of the study, explaining the goal of the study and specifying the intended application, the method assumption and impact limitations, the reasons for carrying out the study, the decision context, the commissioner, the practitioner and the target audience.

Definition of the scope of the study, answering questions regarding what to analyse and how. It includes the drawing of a flow diagram of the system under study, which leads to a first estimation of the system boundaries between the system analysed and the ecosphere[2] and of the inputs entering and outputs leaving the system (elementary flows). It also includes the definition of the life cycle inventory (LCI) modelling framework, the levels of cut-off criteria, and the identification of the technical, geographical and time-related representativeness, as well as method consistency. A crucial element of this phase is the definition of the functional unit, which is a measure of the function of the studied system and provides a quantitative reference for the study.

Specification of data requirements, which entails the identification of the main types, quality and sources of data and other information both for the inventory and the impact assessment steps.

For an LCA study in the agricultural sector this could be the comparison of alternative site-specific soil and weed management methods, such as conventional tillage or zero-tillage practices. The functional unit could be the sustainable long term production of 1 kg. of a dried, specific harvested crop (Guinée et al. 2006).

ii. **Inventory analysis**: often referred to as LCI, it involves collecting all data to quantify relevant inputs and outputs of the unit processes of the product system (figure 4) and relating them to the functional unit of the study.

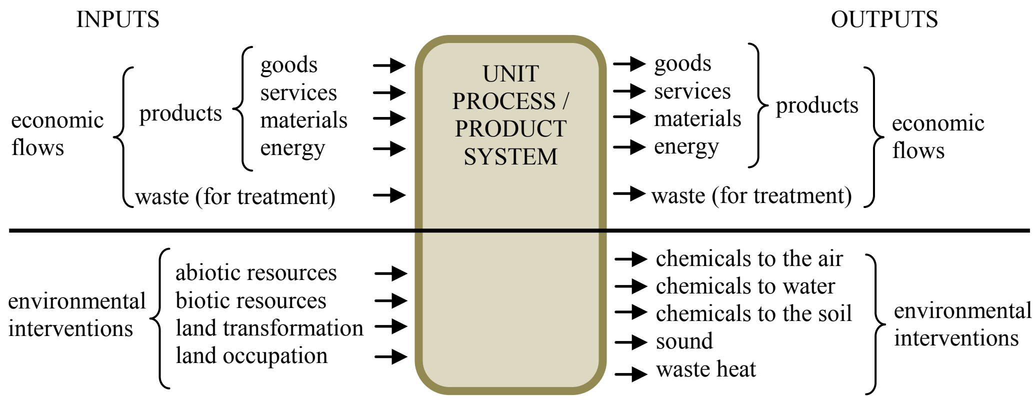


Figure 4 Basic structure of a unit process (or product system) in terms of its inputs and outputs (Guinée et al. 2002).

[2] The term “ecosphere” in JRC (2010a) is referred to as “environment” in ISO 14044 (2006).

The following steps should be carried out:

Collection of data for the unit processes of the foreground and the background systems[3].

Normalisation of data to the functional unit, which means scaling all data collected to the quantitative output of the study.

Resolution of multifunctionality problems by distributing the resource inflows and outflows of a given process over the different products that the process may yield. For example, the cultivation of wheat providing two co-products: wheat and straw.

Calculation of LCI results, summing up all the inputs and outputs of all processes within the system boundaries. LCI results are usually summarised in an inventory table.

The LCI results are the input to the subsequent impact assessment phase.

iii. **Impact assessment**: often referred to as life cycle impact assessment (LCIA), this phase involves sorting together all substances of the LCI that contribute to a particular type of environmental impact (figure 5).

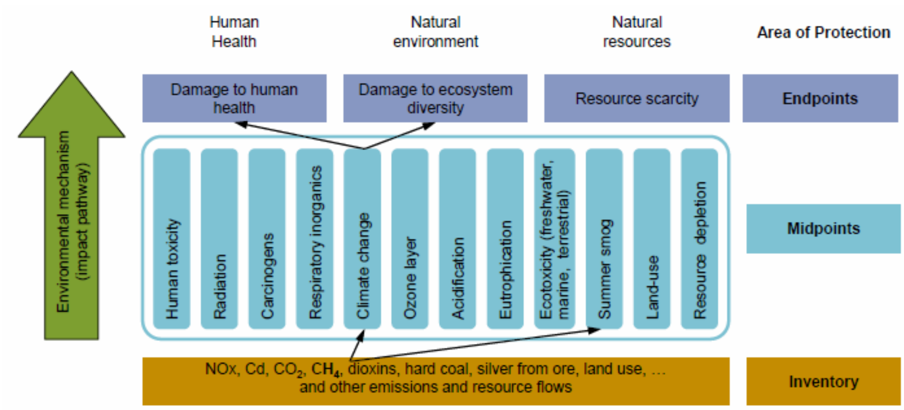


Figure 5 Schematic steps of the LCIA framework, linking life cycle inventory results to endpoint damages via midpoint impact categories (JRC 2010a).

Several steps may be distinguished, of which the first four are mandatory according to ISO 14040 (2006):

[3] According to JRC (2010a), the foreground system includes the processes that are directly affected by decisions analysed in the study in relation to their selection or mode of operation (i.e., direct control). In contrast, the background system comprises those processes that are operated as part of the system but that are not under direct control.

Selection of midpoint impact categories and category endpoints (mandatory). The midpoint and the endpoint approaches are two possible levels at which to quantify the environmental impacts. In the midpoint approach, the impact category indicator is defined close to the intervention (i.e., problem-oriented, such as global warming), while in the endpoint approach (i.e., damage-oriented) indicators are close to recognisable values for society, also called areas of protection (AoP). The ILCD handbook (JRC 2010a) distinguishes three AoP: human health, natural environment and natural resources. Section 1.5.1 provides a more detailed description of midpoint and endpoint methods.

Selection of characterisation methods (mandatory), either at the midpoint or the endpoint level. Of the currently available impact assessment methods, the most used are the CML 2002 (Guinée et al. 2002) at the midpoint level and the Eco-indicator 99[4] (EI99, Goedkoop and Spriensma 2001) at the endpoint level. Other novel methods combine both approaches (e.g., IMPACT 2002+, Jolliet et al. 2003).

Classification (mandatory), assigning the LCI results to the corresponding impact categories.

Characterisation (mandatory), which means that the impact of each resource consumption and emission is modelled quantitatively. The result is expressed in a unit common to all contributions within a given impact category by applying the so-called characterisation factors. For example, the common unit for the impact category climate change is kg. of CO_2-equivalents. Here, the characterisation factor of carbon dioxide is 1, while methane has a characterisation factor of more than 20, reflecting a higher climate change potential.

Normalisation (optional), where the characterised impact scores are expressed in relation to a common reference, by dividing the indicator results by the respective reference. The ILCD handbook (JRC 2010a) recommends the total annual environmental inventory globally per capita as normalisation basis (i. e., the global total divided by the number of citizens).

[4] As the Eco-indicator 99 is the impact assessment method used in chapter 2 for the assessment of water consumption impacts and since it is widely referenced throughout the thesis, a brief description of its foundations can be found in section 1.5.1.

Weighting (optional), where the environmental impact categories and/or AoP (typically normalised) are each multiplied by a specific weighting factor, that is intended to reflect the relative importance of the different impact categories/endpoint damages among each other. For example, the impact category acidification potential may get a weight of e. g., 2 and the impact category of eutrophication potential a weight of e. g., 3. Weighting sets can be developed by different mechanisms, such as setting them by public policy makers, industry panels or expert panels.

iv. **Interpretation**: this phase aims to evaluate the results from the LCI and the LCIA. Three activities can be distinguished, illustrated in figure 6:

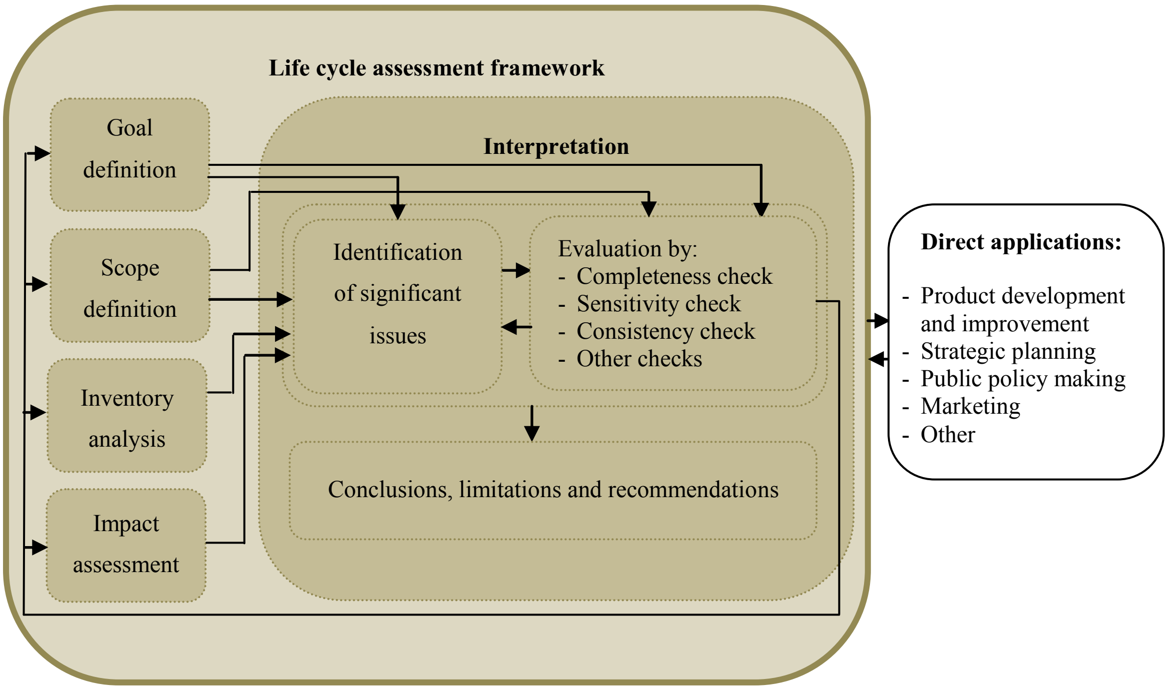


Figure 6 The steps of the interpretation phase and their relation to other phases of the LCA and within the interpretation phase (JRC 2010a).

Evaluation of the most important results obtained in the LCI and the LCIA steps.

Analysis of the quality and the robustness of these results.

Drawing conclusions and recommendations.

The results of the interpretation phase may lead to a new iteration round of the study, including a possible adjustment of the original goal and scope.

1.2.2 Life cycle assessment of agricultural systems

LCA was initially developed for the assessment of industrial systems. Several groups began to apply LCA to agricultural systems in the 1990s (e.g., Audsley 1997; Wegener Sleeswijk et al. 1996). Results of these studies showed that there were considerable **differences between industrial and agricultural systems** (table 1), which hindered the applicability of the LCA method to agricultural products.

Table 1 Differences between industrial and agricultural systems.

Characteristic	Industrial systems	Agricultural systems
Dependency on location	Independent	Dependent
Functionality	One or few functions	Multifunction
Obtained products	Typically one	Typically more than one (co-production)
System boundaries	Clearly defined	Unclear, both physically and temporally
Main sources of impacts	Energy and materials consumption	Land and water use, pesticide emissions, energy and materials consumption
Degree of knowledge	High (simple and pre-designed processes)	Relatively low (complex, natural processes)

Source: Adapted from Milà 2003.

The differences, recapitulated in table 1, are the following:

- The evaluation of environmental effects in LCA is not specific for a certain location, whereas **agricultural production is dependent on temporal and spatial conditions**. Soil quality, climate, water availability and farming practice influence yields very much. For these reasons, the production of a given crop varies between locations and years.

- Agricultural systems are **multi-purpose**, **having primary and secondary functions**. For example, a food crop plot, apart from providing foodstuff, helps to preserve biodiversity and landscape aesthetic and cultural values. A biomass energy crop plot provides useful energy but also allows for the preservation of land under agricultural practice, which is important for social and food security reasons (Patyk and Reinhardt 2000). The most appropriate functional unit in each case will depend on the service to be studied, bearing in mind that the alternatives should be function equivalent.

- Agricultural systems typically **produce more than one output (co-products)**. Therefore, the resolution of multifunctionality problems plays an important role in results.

- Arable crops grow in **rotation systems of several years**, with the aim of reducing the use of external inputs though internal nutrient recycling, maintaining the long-term productivity of the land and avoiding pests, thus consequently increasing crop yields (Zegada-Lizarazu and Monti 2011). As planting one specific crop in a rotation can have effects on subsequent crops, the temporal boundaries of the analysed system should cover the whole rotation.
- The **boundary between the technosphere** (i.e., the studied system) **and the ecosphere** (i.e., the environment) **is much more difficult to define**, especially because the soil can be seen as a part of both. Changes in land quality that influence the present production (soil fertility) should not be assessed in the LCA, because the economic output, that is, the crop yield, is already defined in the functional unit. However, changes in the non-economic values of functions (intrinsic value) should be part of the LCA (Guinée et al. 2006).

- **Importance of diffuse emissions** and atmospheric deposition of nutrients and heavy metals in agricultural activities.

- Though materials and energy consumption are a source of global impacts both in industrial and agricultural systems, many **key impacts of agriculture** occur at the same agricultural field and **are related to land and water use and pesticide emissions**.

- Agricultural systems are modified natural systems, of which **knowledge is very limited** due to their complex biological processes. On the contrary, industrial systems have been human designed, so they are well-known.

Applying LCA to agricultural systems without due consideration of the abovementioned specific characteristics of agriculture may lead to erroneous results. These and other issues concerning system boundaries, allocation and specific environmental impact categories relevant for agricultural systems have been addressed by many authors (Cowell 1998; Guinée et al. 2006; Milà 2003). Considerable methodological advances have been currently achieved. However, according to the EULCIA project on the recommendation of methods and characterisation factors for a standardised LCIA framework (JRC 2007), existing LCIA methods do not sufficiently address or even ignore several environmental impacts of agricultural systems. In current LCA methods, the representation of some life cycle stages is out of proportion, while others do not get the attention they deserve. This trend is especially visible in the land use intensive sectors of agriculture and silviculture.

All this means that the application of available LCA methods to agricultural products gives only a partial vision of the real impacts of agriculture, as current methods do not consider the environmental impacts of, among others, water and land use: two resources that, as seen in section 1.1, are essential to carry out agricultural activities. The lack of due regard of these types of environmental impact categories in LCA needs to be addressed. The current challenges of the life cycle assessment methodology are further described in the following section.

1.3 Current challenges of the life cycle assessment methodology in relation to impact assessment categories. Motivation of the thesis

Impact assessment categories commonly included in LCA studies reflect the focus of LCA methodological development on industrial systems and their characteristic impacts, such as global warming, resource depletion or summer smog formation. When the environmental performance of an agricultural system is analysed with the LCA method, many important impacts of agriculture are not covered by these common environmental indicators. The main important shortcomings affecting the life cycle assessment of agricultural systems concern the following impacts:

- Land use, including impacts on biodiversity and services provided by terrestrial ecosystems, such as the capacity to produce biomass or to stabilise soils to reduce erosion.
- Water use, including impacts due to withdrawals from surface and ground water bodies to irrigate crops as well as the effects of water uptake from soil moisture by plant roots.
- Human and ecological toxicity impacts due to the use of pesticides and fertilisers

For **land use-related impacts**, in recent years studies have been performed to extend LCIA to issues such as erosion, desiccation and biodiversity in order to assess the sustainability of agricultural systems, though these topics are still undergoing debate. It is wort mentioning the ongoing work of the taskforce on natural resources and land use within the UNEP/SETAC[5] life cycle initiative partnership, who worked out an agreed set of principles for the LCIA of land use

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[5] United Nations Environment Programme/Society for Environmental Toxicology and Chemistry.

(Milà i Canals et al. 2007). The group is presently working on the provision of practicable methods compatible with the set of principles to assess land use anywhere in the globe. The impact assessment method and characterisation factors for the carbon sequestration supporting service (Müller-Wenk and Brandão 2010) are already available. Methods and impact factors for biodiversity and for the ecosystem services of biotic production, erosion regulation, freshwater regulation and water purification are expected to be available very soon.

For **water use-related impacts**, Heuvelmans (2005) detected that water quantity was absent in LCAs or at most, the amount used was simply recorded in the LCI. In her thesis, she proposed an indicator framework for the impact assessment evaluation of water use. The lack of globally applicable methods to assess water use was also recognised by the UNEP/SETAC life cycle initiative, which in 2008 created a working group on the assessment of use and depletion of water resources. So far, the methodological framework to support the quantitative modelling of water use has been proposed (Bayart et al. 2010), including recommendations for LCI and possible indicators on the midpoint and endpoint levels.

The ongoing work of the LC-IMPACT[6] European project, in which the Sostenipra research group is participating, is addressing key aspects for developing and improving LCIA methods, characterisation factors and normalisation factors for land and water use, among other impact categories for which no widely agreed methods yet exist. This project complements the work already carried out or in progress of the UNEP/SETAC life cycle initiative working groups.

While the UNEP/SETAC life cycle initiative working groups and the LC-IMPACT project focus on improving environmental methodologies for LCA, the PROSUITE[7] European project aims to develop and standardise life cycle methods for a sustainable assessment of technologies, taking into account the environmental performance but also the economic and social dimensions of technologies. The Sostenipra research group is also engaged in the development of this project.

Apart from these international initiatives, the thesis by Antón (2004) is the other important background to this thesis. Antón (2004), in her PhD dissertation, carried out an environmental assessment of tomatoes grown in a Mediterranean climate (Maresme, Barcelona, Spain),

[6] Acronym for "Development and application of environmental Life Cycle Impact assessment Methods for imProved sustAinability Characterisation of Technologies" (http://www.lc-impact.eu/).

[7] Acronym for "PROspective SUstaInability Assessment of TEchnologies" (http://www.prosuite.org/).

identified the key parameters of agricultural systems from an environmental point of view and pointed out methodological aspects of the LCA method to be enhanced, these being the use of pesticides, land and water. Juraske (2007) contributed to the development of best available practice in fate and exposure assessment of pesticides for evaluating their impacts on human health and ecosystems. This thesis endeavours to cover the aspects of water and land use, with the aim of providing advances towards their integration in the LCA methodology. For land use, efforts were addressed to model soil erosion impacts and, due to the irreversible effects of land degradation in arid lands, special attention was given to going forward in the inclusion of desertification impacts in LCA.

As stated in the section 1.2.2, impacts from land and water use are highly dependent on the location of the system under study. Water availability and many biogeographical factors regarding landscape, climate, vegetation patterns and a range of soil properties determine the severity of the environmental effects from water and land use. There is therefore a growing recognition of the necessity to develop LCIA methods for these non-global impact categories that account for the spatial variability of water and land availability. This was taken into account in the methodological improvements proposed in this research.

1.4 Objectives of the thesis

The main objective of this thesis is **to contribute to the methodological development of LCA, in order to allow for a better evaluation of the environmental impacts of agricultural systems, focusing on the issues of water and land use**.

To achieve this main aim, the following goals are addressed:

- To suggest a spatially-dependent characterisation approach to account for the environmental impacts of green water consumption.
- To provide a methodology for including potential desertification from land use as a new regional impact category.
- To propose a geographically differentiated framework to include impacts of land use on the soil erosion regulation ecosystem service.
- To provide viable methods to build the inventories for the impact categories studied.

- To evaluate the usefulness of the geographic information system tools to define location-dependent characterisation factors and to calculate environmental indicators for the regional impact categories of water and land use.

- To verify the applicability of the developed methods, applying the methods of water consumption and soil erosion to a system of study, namely agricultural crop rotations with energy crops grown in Spain.

- To assess the effects of producing energy from crops on the limited water and land resources in Spain.

- To evaluate agricultural and natural reference systems as reference situations to compare the environmental impacts of water consumption and soil losses of the energy crop rotations.

- To determine suitable areas of Spain to grow energy crops in agricultural rotations regarding water and soil erosion environmental impacts.

1.5 Materials and methods

This section presents the main methodological aspects that have been involved in the development of the thesis. These concern two issues: LCIA aspects and GIS tools. In addition, the systems of study that have been used to exemplify the methodological developments of the thesis are also outlined here.

1.5.1 Life cycle impact assessment aspects: approaches and methods

Midpoint and endpoint approaches

LCIA aims to connect emissions and extractions of the LCI on the basis of impact pathways to their potential environmental damages. An impact pathway (or environmental mechanism or cause-effect chain), is the basis for modelling in LCA, and represents a series of effects that together can create a certain level of damage. The first two steps of the impact assessment phase (see section 1.2.1) are the selection of the category indicators and the characterisation methods.

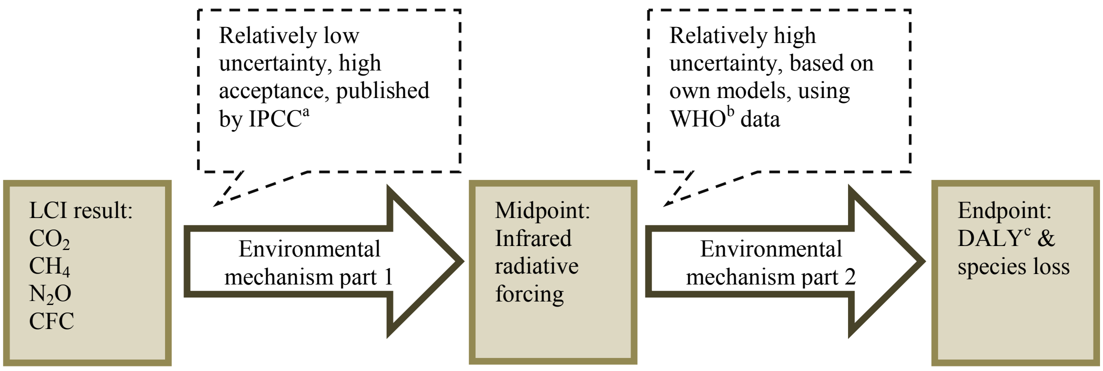
A category indicator may be located at any place between the LCI intervention and the final environmental damage. Based on that, there are **two main schools or approaches of LCIA methods**:

- Classical impact assessment methods, which link LCI results to so-defined **midpoint categories**, e.g., ozone depletion or acidification.

- **Damage oriented methods**, which model the cause-effect chain up to the environmental damages to human health, natural environment and natural resources, e.g., human health impairment or species endangerment.

The term midpoint expresses that the indicator is located on the impact pathway at an intermediate position between LCI results and the ultimate environmental damage, often referred to as endpoints. As a consequence, an additional step may allocate these midpoint categories to one or more damage categories, the latter representing the ultimate object of society's concern (Jolliet et al. 2004).

Figure 7 shows an example of representation of the midpoint and endpoint approaches for the climate change impact pathway. According to this figure, we know that a number of substances, registered in the LCI, increase the radiative forcing, which is the impact category indicator at the midpoint level. As a consequence of radiative forcing, heat is prevented from being emitted from the Earth to space. This means that more energy is trapped on Earth, and temperature increases. The expected effects of this on the endpoint level are twofold: changes in habitats for living organisms, which may lead to species extinction, and human life impairments.



[a] Intergovernmental Panel of Climate Change; [b] World Health Organization; [c] Disability adjusted life years.

Figure 7 Example of a midpoint-endpoint model for climate change, linking the LCI to human health and ecosystem damages (Goedkoop et al. 2009).

From the example above, it can be observed that the longer the modelled impact pathway, the higher the uncertainties obtained. While the radiative forcing is a physical parameter of relatively easy measure in a laboratory, the resulting temperature increase is less straightforward to determine, and it is even more difficult to predict the final consequences of climate change. Midpoint indicators are therefore more certain than endpoint indicators. However, LCIA methods contain many midpoint indicators (about 15, but the quantity varies among methods) and a small number of endpoint indicators (usually 3), the latter having generally a more understandable meaning for society.

One of the multiple endpoint methods available in the market is the Eco-indicator 99 method, applied in chapter 2 for the assessment of water consumption impacts. This method is further described in the following section.

Impact assessment methods: the Eco-indicator 99 method

A characterisation method, or impact assessment method, refers to the collection of individual impact pathways that together address the different impact categories. Most existing characterisation methods are either at the midpoint or the endpoint level. Since the early nineties many attempts have been made to harmonise both approaches, yet without resulting in a consistent accepted method. In this regard, the ReCiPe 2008 method (Goedkoop et al. 2009) is the latest attempt, though acceptance and degree of applicability is still to be proven due to the young age of the method.

Of the endpoint methods, the most accepted and applied is the so-called EI99 method (Goedkoop and Spriensma 2001), developed by PRé Consultants. This LCIA method aims to simplify the interpretation and weighting of results. One of the intended applications was the calculation of the environmental load of a product in a single score. This single-score indicator can be used by designers for product development applications, but it is also used as a general purpose impact assessment method in LCA (Goedkoop and Spriensma 2001; JRC 2010b).

The method covers **three endpoint impacts**:

- **Damages to human health**: expressed as DALY (disability adjusted life years). Four sub-steps are used to establish the link between the LCI table and the potential damages:

  Fate analysis, linking an emission to a temporary change in concentration.

Exposure analysis, linking this temporary concentration to a dose.

Effect analysis, linking the dose to a number of health effects.

Damage analysis, linking health effects to the number of years lived disabled (YLD) and years of life lost (YLL).

- **Damages to ecosystem quality**: expressed as PDF (potential disappeared fraction of species). Two different approaches are used:

  - Ecotoxicity, acidification and eutrophication. Ecotoxicity is firstly expressed as PAF (potentially affected fraction of species) and later transformed to PDF with a rather crude conversion. Three sub-steps are used:

    Fate analysis, linking emissions to concentration.

    Effect analysis, linking concentrations to toxic stress or increased nutrients or acidity levels.

    Damage analysis, linking the effects to the increased potential disappeared fraction of plants.

  - Land use. Based on empirical data of the quality of ecosystems (occurrence of vascular plants) as a function of land-use type and the area size.

- **Damages to resources**: expressed as MJ of surplus energy required to extract an additional unit of the resource. Only fossil fuels and minerals are modelled. Two sub-steps are followed:

  Resource analysis, linking the resource extraction to the decrease of the resource concentration.

  Damage analysis, linking lower concentration to the increased efforts to extract the resource in the future.

EI99 also performs the optional phases of normalisation and weighting. Normalisation is carried out at the European level, based on environmental interventions from production in 1990-1994. In the weighting step, the relative importance of the three damage indicators is compared among each other in order to get a single-score indicator. This is done by direct questioning of a panel of LCA experts and LCA users. Results of the questionnaires were grouped according to three different cultural perspectives, which result in four sets of weighting factors: individualist (I), egalitarian (E), hierarchist (H) and an average for all the panellists (A). The authors of the method recommend the use of the hierarchist perspective and the average weighting set.

Figure 8 shows the general representation of the EI99 methodology.

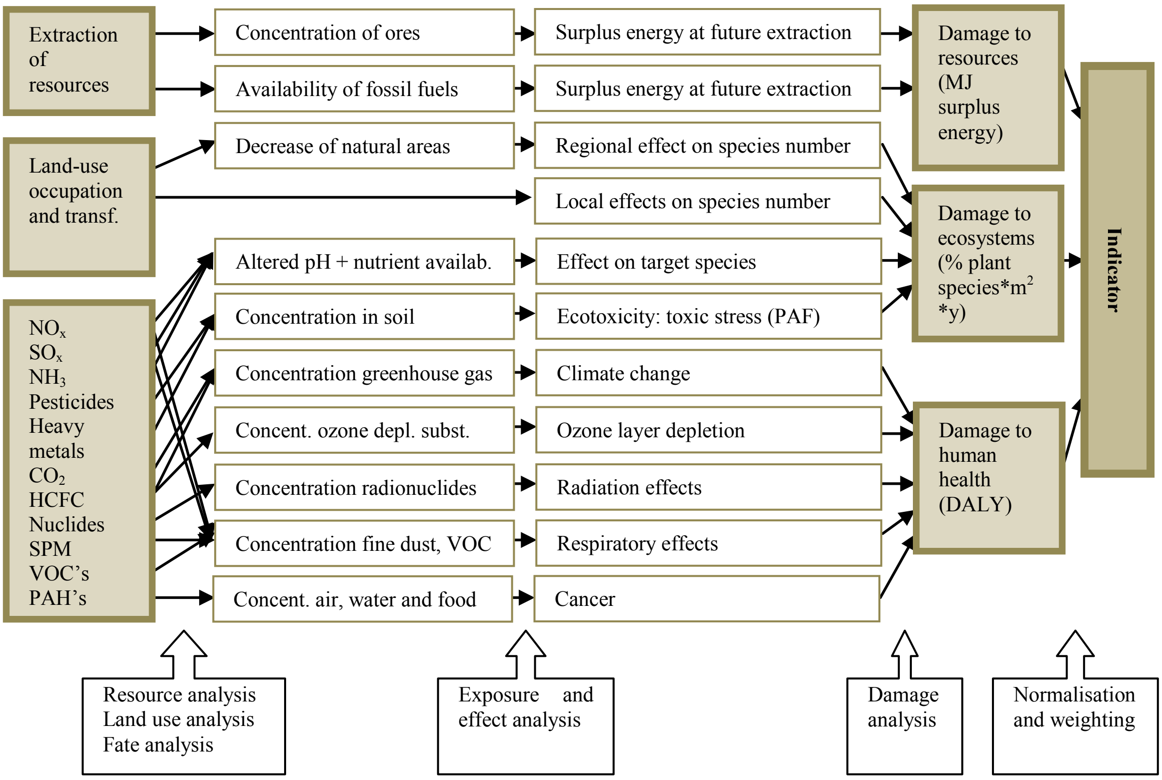


Figure 8 Impact categories and pathways covered by the EI99 method (Goedkoop and Spriensma 2001).

None of the available LCIA methods implemented in LCA software are able to capture the inherent spatial variability of land and freshwater throughout the globe. In this thesis, to develop new LCA methods of land and water use that are site-specific, geographic information tools, which are more and more used coupled with other environmental tools, were used.

1.5.2 Combining geographic information systems and life cycle assessment tools

Geographic information systems, abbreviated as GIS, are *special-purpose digital databases in which a common spatial coordinate system is the primary means of reference*. The term GIS refers both to the specific software and to the data sets to be used with the software. GIS software contains subsystems for: 1) data input; 2) data storage, retrieval, and representation; 3) data management, transformation, and analysis; and 4) data reporting and product generation. A GIS supports spatial data collection, analysis, and decision making.

The release of free and accurate GIS software and data sets provides a relatively novel spatial and temporal resolved data source which can be applied together with other environmental assessment tools such as LCA. GIS and LCA are different and complementary. The former is specifically designed to organise and analyse spatial data, while the latter inventories and assesses product system data without providing geospatial information. Although the lack of spatial differentiation of LCA has been frequently criticised, it was not until recently that LCA began to take advantage of GIS properties (Boulay et al. 2010; Geyer et al. 2010; Saad et al. 2011).

In this thesis, the strengths of combining GIS and LCA were applied to develop spatially-explicit life cycle assessment methods for water and land use, two impact categories that have been traditionally overlooked in the existing LCIA methodologies. Both land and water use environmental impacts rely on spatial and temporal conditions where the evaluated activity takes place, and current (site-generic) LCA methods do not consider this geographic information. With the introduction of GIS tools in this process, the spatial detail of the LCA information can be captured.

Concretely, in this thesis **GIS-LCA merging** was implemented in two stages, according to the LCA framework:

- **In the LCI**, to create a spatial-dependent inventory, where the elementary flows of land and water use vary according to the location. For example, the amount of rainfall of a specific area during a certain time will determine the quantity of soil eroded, which is required data for desertification (chapter 3) and for erosion (chapter 5) LCIA models.Without using GIS, rainfall information would have been referred to a less detailed spatial scale and thus, the resulting inventory flow would have been less accurate.

- **In the LCIA**, to derive spatially-dependent characterisation factors to be applied in the regionalised impact models. For example, in the erosion impact assessment model (chapter 5), the environmental damage of losing a specific soil quantity will depend on the on-site soil resources. This locally available soil reserve, which is used as a characterisation factor, was obtained from publically available GIS maps with information at global scale.

To show how GIS-LCA coupling has been applied throughout the thesis, the procedure is exemplified here with the developed soil erosion model (chapter 5). Table 2 reports the necessary input data and sources used to construct the LCI as well as to derive the regionalised characterisation factors of soil erosion. To create a regionalised inventory of soil erosion for each studied agricultural plot, a number of country, regional and local geospatial databases and web map service interfaces containing edaphoclimatic and landscape information were consulted. The GIS software MiraMon® 6.1 (Pons 2008) supported data collection and management by assigning to the created layer (or map) of georeferenced plots the spatially-resolved information gathered in the LCI. Lastly, the layer(s) of the input parameters that are used as characterisation factors were overlaid and queries were created to calculate, by means of the impact assessment model, the spatially-differentiated environmental impact scores for soil erosion. Table 2 also shows data range values for each input parameter according to two different resolution schemes: whole country and specific plot. The large variability of data values from plot to plot, not captured by the average country data, could be accounted for in this thesis to assess land and water use impacts thanks to the use of GIS tools.

Table 2 Spatially resolved input parameters and characterisation factors for the soil erosion LCA model.

Input parameter		Data range value[a]		Source
		Average Spain	Plot	
Life cycle inventory				
Georeference location		-09º - +03º lon +36º - +44º lat	All, within Spanish boundaries	- Spanish Soils Edaphic Properties database (Trueba et al. 2000) - Agricultural plots GIS (SIGPAC) (MARM 2011a)
Soil properties	Soil texture	Loam	All	- Spanish Soils Edaphic Properties database (Trueba et al. 2000) - Sample analysis
	Sand (%)	43.8	0.7-84.5	
	Silt (%)	30.2	6.0-72.3	
	Clay (%)	25.7	0.5-68.7	
	Soil structure	Not granular	All	- Spanish Soils Edaphic Properties database (Trueba et al. 2000) - Sample analysis
	Organic matter content (%)	2.1	0.1-14.8	- Spanish Soils Edaphic Properties database (Trueba et al. 2000) - Sample analysis
	Soil unit	Cambisol	11 types	- Harmonized World Soil Database (FAO/IIASA/ISRIC/ISSCAS/JRC 2009)
Landscape properties	Slope gradient (%)	3.1	0-8	- Spanish Soils Edaphic Properties database (Trueba et al. 2000) - Agricultural plots GIS (SIGPAC) (MARM 2011a)
Crop properties	USLE C-factor[b]	0.9	0.5-1.2	- Field work - Wischmeier and Smith (1978)
Climatic properties	Annual rainfall (mm y^{-1})	456	201-1040	- Agrarian data GIS (SIGA) (MARM 2011b)
Characterisation factors				
Soil properties	Soil depth (cm)	104	48-125	- Effective soil depth raster map (FAO/UNESCO 2007)
Ecosystem properties	Net primary productivity of the potential vegetation (NPP_0) (g C m^2 y^{-1})	506	302-628	- NPP_0 raster map (Haberl et al. 2007)

[a] Data range values of the 120 agricultural plots analysed in the case study of chapter 5.
[b] C-factor of the universal soil loss equation (chapter 4). Example of data range values for the winter barley-winter wheat-oilseed rape rotation.

1.5.3 Systems of study: energy crops in Spain

The research presented in chapters 2, 4 and 5 uses the case of energy crops as the systems of study, but the LCA methods used in the thesis might be applicable to any kind of agricultural system. An energy crop is defined (CIEMAT 1996) as

> *a plant grown and used to generate biomass that can be transformed into fuels or directly exploited for its energy content*.

In Europe, the production of crops with energy purposes is seeking to increase the share of renewable energy consumption at the expense of fossil fuels, in order to reduce greenhouse gas emissions that contribute to climate change. Furthermore, bioenergy crops can help Europe to reduce dependency on energy imports and to be a source of income for farmers.

With the aim of determining environmentally compatible paths for bioenergy cultivation, several studies about the environmental performance of energy crops have been recently carried out, with the main subject of discussion being the energy balance (e.g., Iriarte 2010; Martínez Gasol 2009). However, apart from the input-output energy yield, other pressures of current agricultural activities on the environment should be evaluated, in order to obtain a complete picture of the real impacts of energy crops. In the case of Spain and other arid climate zones of southern Europe, it is of great importance to measure the environmental impacts due to the use of land and water, since both resources are already subjected to many pressures. To this end, the Spanish Ministry of Science and Innovation is supporting an in-depth study on bioenergy production in the country. The project, Singular and Strategic Project for the development, demonstration and evaluation of the viability of the commercial production of energy from dedicated crops in Spain (SSP On Cultivos), includes the assessment of soil erosion as well as of water consumption due to the growth of energy crops.

In the thesis, the systems of study were of crops cultivated in rotations of several years (chapters 2, 4 and 5). Chapter 4 also includes the assessment per individual annual crop. In all cases, a comparative analysis of the energy crops with other agricultural systems –including non-energy crops and land in fallow– and/or with natural vegetation systems was performed. Table 3 summarises the crops of the systems of study of the following chapters.

Table 3 Crops of the systems of study in the thesis.

Crop		Chapter	Application considered in the thesis		Type of analysis	
			Energy	Non-energy	Individual crop	In rotation
Winter barley (*Hordeum vulgare L.*)		2, 5		×		×
Winter wheat (*Triticum aestivum L.*)		2, 4, 5		×	×	×
Oilseed rape (*Brassica napus*)		2, 4, 5	×		×	×
Sunflower (*Helianthus anuus L.*)		2	×			×
Maize (*Zea mays L.*)		2		×		×
Sorghum (*Sorghum bicolor L.*)		2	×			×
Poplar (*Populus spp.*)		2, 5	×			×
Rye (*Secale cereale L.*)		5		×		×
Pea (*Pisum sativum*)		5		×		×
Unseeded fallow		2, 5		×		×
Mediterranean forest		2		×		×

The following chapters, from chapter 2 to chapter 5, describe the contributions of the thesis towards the inclusion of the location-dependent impact categories of water and land use in LCA.

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Chapter 2.

Assessing the environmental impacts of water consumption on energy crops grown in Spain.

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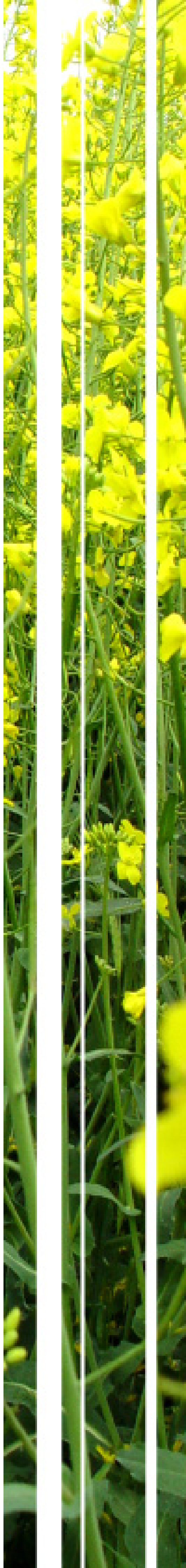
Stefanie Hellweg, Annette Koehler, Joan Rieradevall.



Abstract

The environmental impacts of water consumption of four typical crop rotations grown in Spain, including energy crops, were analyzed and compared against Spanish agricultural and natural reference situations. The LCA methodology was used for the assessment of the potential environmental impacts of blue water (withdrawal from water bodies) and green water (uptake of soil moisture) consumption. The latter has so far been disregarded in LCA. To account for green water, two approaches have been applied: the first accounts for the difference in green water demand of the crops and a reference situation. The second is a green water scarcity index, which measures the fraction of the soil-water plant consumption and available green water. Our results show that, in the Spanish water basins and rotations, there are major differences in blue and green water consumption and in their related environmental impacts, as well a trade-off between water consumption and land use impacts. Further research on integration of quantitative green water assessment in LCA is crucial in studies of systems with a high dependence on green water resources.

Keywords Bioenergy crops • Blue water • Green water • Green water scarcity index (GWSI) • LCIA • Water footprint

2.1 Introduction

2.1.1 Water consumption and energy crops

Freshwater is an essential natural resource for human and ecosystem health. While water is abundant globally, the resource is under increasing pressure in many parts of the world. Agriculture is by far the largest water-use sector, accounting for around 70% of the water withdrawn worldwide from rivers and aquifers for agricultural, domestic and industrial purposes (FAO 2008). In Europe, agricultural water use is a serious concern especially in southern and southeastern regions such as Spain, where water is scarce and highly variable during the year and from year to year.

An increasing agricultural area throughout Europe is devoted to growing crops for energy purposes. The use of bioenergy in Europe offers significant opportunities to replace conventional fossil fuel energy sources, decrease greenhouse gas emissions and improve energy supply security. However, it is clear that the increasing demand for energy crops will lead to further pressure on water resources, if their cultivation leads to an increased share of irrigated crop areas. Virtual water content (Allan 1998), defined as the volume of water from irrigation (blue water, BW) and precipitation (green water, GW) consumed through evapotranspiration during crop growth (Allan 1998), was recently quantified in Spain as 1.03-1.20 m^3 kg^{-1} y^{-1} for irrigated wheat, 1.03-1.08 m^3 kg^{-1} y^{-1} for irrigated barley and 0.59 m^3 kg^{-1} y^{-1} for irrigated maize (Aldaya and Llamas 2009).

Of the total water consumption, the proportions of green water and blue water differ as a function of spatial conditions (e.g., climate, soil and vegetation properties). In the case of energy crops, for instance, the BW requirements of a poplar bioenergy system located in the Duero basin (central Spain) amounts to about 1,870 m^3 ha^{-1} y^{-1} in a short rotation of 15 years, adding up to 28,000 m^3 ha^{-1} over the complete cycle (Gasol et al. 2009). These authors underlined the high water demands of poplar as the determining factor restricting its spread in Spain. This example highlights the importance of selecting a balanced combination of crops and locations that minimise water stress.

2.1.2 Water use and life cycle assessment

A number of water consumption analyses using different hydrological models (Hoff 2010) for many agricultural products (Gerbens-Leenes et al. 2009; Hoekstra and Hung 2005) have been published recently. Most of these studies focus on the quantification of water use while ignoring the environmental consequences of water consumption. Water use and water resource depletion have gradually gained importance in LCA, a methodology for quantifying the environmental impacts of products and activities. However, there is still only preliminary scientific consensus on the parameters to consider and the methodology to follow to account for water use-related impacts.

Recently, two regionalized methodologies dealing with how to assess consumptive water use in LCA studies have been published: Milà i Canals et al. (2009) suggest direct environmental impact indicators, that is, midpoint indicators, and Pfister et al. (2009) developed environmental damage indicators, that is, endpoint indicators, for the human health, ecosystem quality and resources attributes of concern. Both methodological approaches propose the virtual water (Allan 1998) and the water footprint (Chapagain and Hoekstra 2004) concepts to generate the LCI data, and they both consider only blue water consumption. However, Pfister et al. (2009) recognized that neglecting potential changes in green water flows, for example, due to different vegetation types, is a simplification, and that the related effects should be addressed in future research. Milà i Canals et al. (2009) determined changes in green water flows as a consequence of land use changes, as these aspects are strongly related. The methods of Milà i Canals et al. (2009) and Pfister et al. (2009) take into account the globally unequal distribution of freshwater resources by the spatial differentiation of impacts at the watershed level.

Despite the fact that many environmental impacts of agricultural production depend on land use (e.g., water consumption, carbon emission or sequestration, and erosion) only a limited number of LCA studies on bioenergy resources have measured environmental impacts of currently occupied land against a land-use reference system.

In this paper, we assessed the environmental impacts associated with water consumption of several energy crop rotation systems grown in Spain. The aims of the study were to (i) assess the environmental impacts of blue water consumption in real agricultural settings with a recently developed LCIA method (Pfister et al. 2009), examining a large number of case-study sites distributed all over Spain, (ii) propose a characterisation approach to account for the environmental impacts of green water consumption, (iii) compare environmental impacts caused by water consumption of the tested energy crops against reference agricultural and natural situations in the country, (iv) identify appropriate production areas and energy crop rotations to minimize the environmental effects of water consumption for growing energy crops in Spain, and (v) quantify the land and water trade-offs between rainfed and irrigated crops.

2.2 Materials and methods

2.2.1 Agricultural systems and water consumption

Crop rotation systems

Water consumption in crop rotation systems was estimated in 117 plots located throughout Spain, covering 23 out of the 47 Spanish watersheds of the WaterGAP 2 global model (Alcamo et al. 2003) (figure 9).

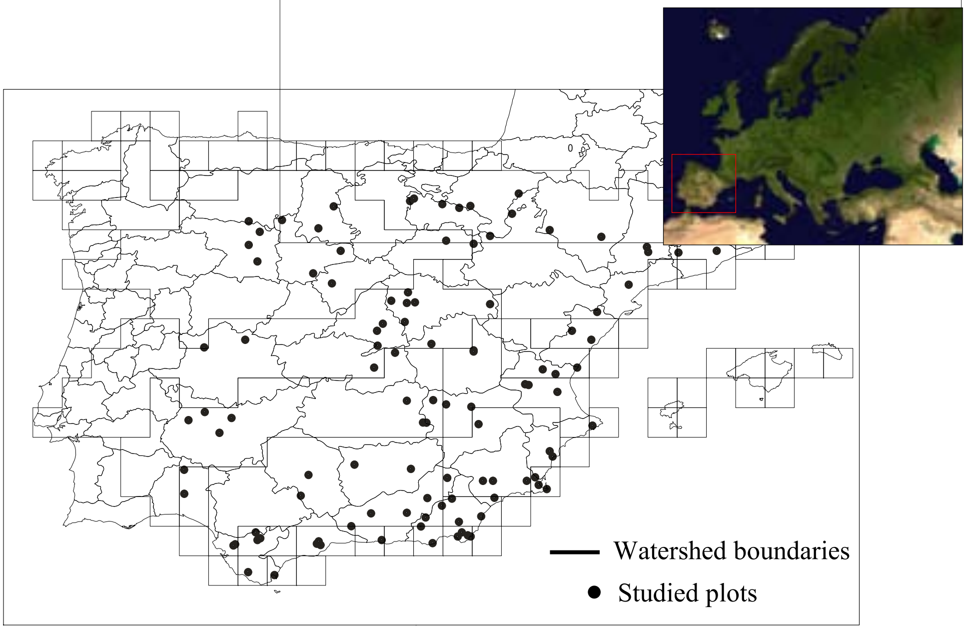


Figure 9 Location of the studied plots in the Spanish watersheds. Location of watersheds (black straight lines), political boundaries of provinces and agricultural production plots (black dots), modified and adapted from Alcamo et al. (2003) and Trueba et al. (2000).

Some data on the geographical and physical properties of the plots included in the study were collected from the Spanish Soils Edaphic Properties database (Trueba et al. 2000), which contains information on more than 2,000 plots of agricultural, forestry and grazing land. Other data were specifically measured for a current project studying the viability of energy crops in Spain (SSP On Cultivos, www.oncultivos.es). Of the selected plots from Trueba et al. (2000), all are devoted to agriculture or are bare soils. The 117 studied plots are considered a representative subsample of the geographical and physical properties of the plots from the Spanish Soils Edaphic Properties database and the SSP On Cultivos project.

Arable crops are usually grown in a crop rotation system. This means that the comparison of agricultural systems at the level of single crops may be misleading, as management practices implemented during the cultivation of a crop (e.g., application of fertilisers) may also benefit subsequent crops. Each crop in a crop rotation has its own function and the cultivation of one could have an effect on the yield of another. This means it is important to extend the system boundaries to the whole crop rotation when an LCA study is carried out (Nemecek et al. 2008; van Zeijts et al. 1999).

This was taken into consideration in our assessment of water consumption of four possible energy crop production systems in Spain (table 4): three systems of annual energy crops in a 3-year crop rotation, and a short rotation coppice of the perennial crop, poplar, with a life-span of 15 years (in contrast to the long rotation times of forestry plantations, with cycles of at least 50 years), cut every three years. The rotations were of crops with food and energy purposes.

Agricultural and natural reference situations for modelling water use impacts

A reference is needed to compare the damaging effects of water consumption caused by the use of land for the crop rotations. As there is no scientific agreement on a reference situation when land use impacts are modelled in LCA (Brentrup et al. 2002; Lindeijer et al. 2002), two different approaches, proposed by Milà i Canals et al. (2007b), were applied.

The first situation was the potential natural land situation. In the system classifications proposed by many authors (Bailey 1998; Folch et al. 1984; Olson and Dinerstein 2002), the potential natural vegetation in Spain (except a narrow coastal line in the north) is Mediterranean forest (311 classes of broad-leaved forest and 313 classes of mixed broad-leaved and coniferous forest, according to the CORINE land cover classification; EEA 2000). Considering that a majority of the energy crop rotations are located in the Mediterranean area, the Mediterranean forest (MF) was chosen as the natural reference for all the plots, with an MF-MF-MF reference accounting for the water consumption over a 3 year period.

The second reference system chosen was the, until recently, common agricultural practice of a two-year rotation of cereals plus a third year of bare soil (winter barley-winter wheat-unseeded fallow, B-W-F) (Boellstorff and Benito 2005). Unseeded fallow was traditionally used in arid and semi-arid agricultural zones in the centre of Spain to enhance soil moisture and fertility for subsequent rotations. In 1992, the Common Agricultural Policy set-aside program provided European Union subsidies (EEC 1992) as incentives for farmers to decrease production, managing land as unseeded fallow. This measure was rescinded in 2007, due to the decrease in cereal production and the escalation of prices in the European Union.

Irrigation schemes of the rotation systems

Crops of the studied rotations (table 4) were grown under three different irrigation schemes, depending on the water supplied in relation to their requirements for optimal production and yield under the given climatic conditions. Barley (B), wheat (W), oilseed rape (R) and sunflower (SF) are non-irrigated crops, so their water supply was from precipitation accumulated as soil moisture (i.e., available green water or effective precipitation Pr'). Both sorghum (SG) and poplar (P) were grown under support-irrigated schemes (deficit irrigation). In this situation, the water supplied is below the requirements of the plant to achieve optimal yield, to obtain an increase in water-use efficiency, defined as yield per water amount consumed, in comparison to rainfed conditions. This type of crop management is typical in low precipitation and water-limited areas such as Mediterranean countries. Maize (M) is the only crop where irrigation completely satisfied the crop water requirements, i.e., standard conditions and maximum yield. Finally, neither reference situation (B-W-F and MF-MF-MF) were irrigated.

Table 4 Characteristics of the studied rotations and the reference systems. Cells in grey indicate crops produced for potential energy use.

Scenario	**Code**	**1st year**	**2nd year**	**3rd year**	**Cropping system**	**Irrigation scheme**
1	B-W-R	Winter **b**arley (*Hordeum vulgare L.*)	Winter **w**heat (*Triticum aestivum L.*)	Oilseed **r**ape (*Brassica napus*)	Three annual crops	Three rainfed crops
2	B-SF-W	Winter **b**arley (*Hordeum vulgare L.*)	**S**unflower (*Helianthus anuus L.*)	Winter **w**heat (*Triticum aestivum L.*)	Three annual crops	Three rainfed crops
3	M-SG-M	**M**aize (*Zea mays L.*)	**S**orghum (*Sorghum bicolor L.*)	**M**aize (*Zea mays L.*)	Three annual crops	Maize: irrigated Sorghum: support-irrigated[a] If $Pr_{April\text{-}September} \leq 0.30\ m^3m^{-2} \rightarrow$ $BW = 0.15\ m^3m^{-2}y^{-1}$ If $Pr_{April\text{-}September} > 0.30\ m^3m^{-2} \rightarrow$ $BW = 0\ m^3m^{-2}y^{-1}$
4	P-P-P	**P**oplar (*Populus spp*)	**P**oplar (*Populus spp*)	**P**oplar (*Populus spp*)	Perennial crop (life-span 15 years, 5 cuts 3 years each)	Support-irrigated[a] If $Pr_{April\text{-}September} \leq 0.30\ m^3m^{-2} \rightarrow$ $BW = 0.30\ m^3m^{-2}y^{-1}$ If $Pr_{April\text{-}September} > 0.30\ m^3m^{-2} \rightarrow$ $BW = 0.10\ m^3m^{-2}y^{-1}$
Ref.sys. 1	B-W-F	Winter **b**arley (*Hordeum vulgare L.*)	Winter **w**heat (*Triticum aestivum L.*)	**F**allow	Two annual crops + bare soil	Two rainfed crops + bare soil
Ref.sys. 2	MF-MF-MF	**M**editerranean **F**orest	**M**editerranean **F**orest	**M**editerranean **F**orest	Natural vegetation	Non-irrigated

Pr = precipitation; BW = blue water.
[a] Irrigation restricted to a predetermined threshold. The April-September period defines the amount of support-irrigation, as crop water requirements are highest in the spring and summer (these are the hottest and driest months in Spain). Data is from records of water consumption of poplar and sorghum cultivated for energy purposes (SSP On Cultivos, www.oncultivos.es).

Calculation of water consumption in each rotation system

The total water consumption of each rotation was the sum of blue water and green water consumed yearly by each of the three crops grown in the rotation. To calculate the water consumed by each crop, the evapotranspiration losses from seeding to harvest was added to the evaporation losses from harvest to the seeding of the next crop. Crop evapotranspiration (ET_c) can be calculated from the FAO approach (equation 1, Allen et al. 1998):

$$ET_c = K_c \times ET_0 \qquad (1)$$

Crop coefficients (K_c) are used to relate ET_c to potential evapotranspiration (ET_0). ET_0 describes the evapotranspiration from a reference surface, calculated based on climate data using the Penman-Monteith approach. ET_0 is independent of crop type, crop development and management practices. Primary ET_0 data at local level for the period 1998-2007 were gathered as monthly data and averaged for each of the 117 studied plots. K_c values for several crops grown under standard conditions in sub-humid climates are available from Allen et al. (1998). These K_c values have been adjusted for rainfed and support-irrigated crops in our study (table 5, following Allen et al. 1998). The dates for seedbed, harvest and the length of the intermediate crop-stages were collected on a province level (see figure 9 for boundaries of Spanish provinces), from regional agri-food institutes, and allocated to each plot using geographic information systems.

Table 5 Adjusted rainfed and support-irrigated monthly crop coefficients (K_c) from seeding until the month before the seeding of the next crop.

Month after sowing	winter **B**arley[a]	winter **W**heat[a]	oilseed **R**ape[b]	**S**un**F**lower[c]	**M**aize[c]	**S**or**G**hum[c]	**P**oplar 1[st]y	**P**oplar 2[nd]y	**P**oplar 3[rd]y	**F**allow
1	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30[d]
2	0.30	0.30	0.45	0.30	0.67	0.67	0.30	0.30	0.30	0.30[d]
3	0.40	0.40	0.55	0.40	1.10	1.10	0.30	0.30	0.30	0.30[d]
4	0.75	0.75	0.65	1.00	1.20	1.20	0.30	0.30	0.30	0.30[d]
5	1.00	1.00	0.75	0.60	1.20	1.20	0.70	0.70	0.70	0.30[d]
6	0.75	0.75	0.85	0.40	0.60	1.00	1.05	1.05	1.33[f]	0.30[d]
7	0.50	0.50	0.95	0.28[e]	0.28[e]	0.28[e]	1.05	1.05	1.33[f]	0.30[d]
8	0.20	0.20	1.00	0.28[e]	0.28[e]	0.28[e]	1.05	1.05	1.33[f]	0.30[d]
9	0.28[e]	0.28[e]	0.50	0.28[e]	0.28[e]	0.28[e]	0.50	0.50	0.80	0.30[d]
10	0.28[e]	0.28[e]	0.23	0.28[e]	0.28[e]	0.28[e]	0.30	0.30	0.30	0.30[d]
11	0.28[e]	0.28[e]	0.28[e]	0.28[e]	0.28[e]	0.28[e]	0.30	0.30	0.30	0.30[d]
12	0.30[d]	0.30[d]	0.30[d]	0.30[d]	0.30[d]	0.30[d]	0.30	0.30	0.30	0.30[d]

[a] Winter crops sowed from first November to first December and harvested at the beginnings of July.
[b] Winter crop sowed from the end of September to the earliest November and harvested the two latest weeks of June.
[c] Crops with sowing dates in spring (SF: March, 15 – May, 15; M: March, 1 – April, 15; SG: April, 15 – April, 30) and harvesting dates in summer-first autumn (SF: August, 15 – October, 15; M: September, 1 – October, 15; SG: September, 15 – September 30).
[d] Bare soil. Same value than K_c of 1[st] month after sowing, 0.30. (Allen et al. 1998).
[e] Surface covered with 12% plant residues. K_c value is K_c of 1[st] month after sowing (0.30) reduced by 5% for each 10% of soil surface that is effectively covered (Allen et al. 1998).
[f] Guidi et al. (2008).

As soil water infiltration (Pr') depends on specific soil characteristics (e.g., soil texture and structure, vegetation types and cover), Pr' of each plot and for each crop was calculated on a monthly basis, using the runoff curve number method adapted to Spain (Ferrer 1993; MOPU 1990). To apply the curve number method, primary precipitation data at the local level, for the period 1998-2007, were gathered monthly and averaged for each of the 117 plots studied. The fraction of the rainfall below a predetermined threshold (P_0) enters the soil and is available for plants and/or later percolates if the water holding capacity of the soil is exceeded. As this later rarely occurs in semi-arid regions, as in most parts of Spain (Bellot and Ortiz de Urbina 2008), we assumed all the effective rainfall as available for the crops. Rainwater above the threshold (P_0) will generate runoff. Values for P_0 for specific land uses, landscape slopes and soil textures are provided in table 6.

Table 6 Thresholds P_0 (mm.) for average soil moisture content.

Land use	Slope (%)	Hydrological properties	Soil group[a] A	B	C	D
Fallow	≥3	R[b]	15	8	6	4
		N[c]	17	11	8	6
	<3	R/N	20	14	11	8
Row crops	≥3	R	23	13	8	6
		N	25	16	11	8
	<3	R/N	28	19	14	11
Winter cereals	≥3	R	29	17	10	8
		N	32	19	12	10
	<3	R/N	34	21	14	12
Rotation of low density crops	≥3	R	26	15	9	6
		N	28	17	11	8
	<3	R/N	30	19	13	10
Rotation of high density crops	≥3	R	37	20	12	9
		N	42	23	14	11
	<3	R/N	47	25	16	13
Grasslands	≥3	Poor	24	14	8	6
		Medium	53	23	14	9
		Good	*	33	18	13
		Very good	*	41	22	15
	<3	Poor	58	25	12	7
		Medium	*	35	17	10
		Good	*	*	22	14
		Very good	*	*	25	16
Forest plantations	≥3	Poor	62	26	15	10
		Medium	*	34	19	14
		Good	*	42	22	15
	<3	Poor	*	34	19	14
		Medium	*	42	22	15
		Good	*	50	25	16
Forest areas (woodlands, scrublands)		Very low density	40	17	8	5
		Low density	60	24	14	10
		Medium	*	34	22	16
		High density	*	47	31	23
		Very high density	*	65	43	33

Source: MOPU (1990).
* Rainfall totally entering the soil.
[a] A = sandy, sandy silt texture; B = sandy loam, loam, sandy clay loam, silty loam texture; C = clay loam, silty clay loam, sandy loam texture; D = clay texture.
[b] crops in the direction of the maximum slope.
[c] contour cropping.

The LCI data to derive water consumption of each crop were the potential evapotranspiration and adjusted crop coefficients to obtain crop evapotranspiration values, and the precipitation, land use (classification of MOPU 1990), soil texture and land slope to obtain the effective precipitation or green water available for evapotranspiration.

The amount of water that a rainfed crop can evapotranspire in a month is limited by effective precipitation (monthly plant water deficiency = ET_c – Pr'). For irrigated crops, the monthly irrigation water requirement is equal to the plant water deficiency. For support-irrigated crops, irrigation is restricted to a predetermined threshold, which depends on climate and crop type (table 4).

The FAO approach could not be used to calculate evapotranspiration of the natural reference vegetation (i.e., MF), since K_c values have been developed only for crops. Instead, the evapotranspiration of MF was derived on an annual basis from a method proposed by Piñol et al. (1999). These authors pointed out the quotient Pr/ET_0 (precipitation to potential evapotranspiration) as the key variable for determining water balance of catchments. The authors highlighted that, when Pr is much greater than ET_0 (humid climate), the evaporative demand can be totally satisfied, annual evapotranspiration (ET_c) can be considered constant and water surplus becomes runoff. In contrast, Pr is lower than ET_0 in arid climates, and even in wetter years the water supply is not enough to satisfy the evaporative demand. Thus, annual ET_c depends on the water surplus of rainy years, which is evaporated rather than lost by runoff. In arid climates, the correlation of Pr and ET_c is highly significant (Piñol et al. 1999). This relationship is described in equation 2. This model is only applicable at low values of Pr/ET_0 (≤1, arid catchments), as in all the studied plots, with Pr/ET_0 values between 0.15 and 0.80.

$$ET_c = \left(\frac{\left(\frac{Pr}{ET_0}\right)^k}{1+\left(\frac{Pr}{ET_0}\right)^k} \right)^{\left(\frac{1}{k}\right)} \times ET_0 \qquad (2)$$

where k=2.0 is a group parameter of the non-climatic catchment characteristics relevant for the water balance.

2.2.2 Environmental assessment

Basis for comparison

As agricultural systems are multifunctional, two approaches were used to carry out comparisons between systems:

Spatial agricultural management scope. The cultivation should be performed by minimising the environmental impacts per area and time unit (Nemecek et al. 2008). In terms of water consumption, this means not using more water than sustainably available at a particular site. This is an indicator for an absolute ecological impact rather than a comparative functional unit for an LCA study, since impacts of the system under study refer to the farming area, not to the functionality of the crop cultivated. Such an m^2-based assessment is useful to quantify the absolute impact of water use related to an area, indicating an impact intensity which is useful in the spatial management of agricultural land. This impact intensity is expressed as volume of water consumed (m^3) per area cultivated (m^2) during the 3-year crop rotation. This unit can be used as a basis for calculating the environmental impacts of other functional units (e.g., tonnes of product).

Productive scope. Crops are grown for food, feed, fibre or biomass for bioenergy. The objective is to minimise the environmental impacts per unit of product (Nemecek et al. 2008). In terms of water, it means reducing the quantity of water applied per unit output of, for example, harvested dry matter, raw protein yield or edible energy yield, which denote different productive functions for comparison. Tonne of harvested dry matter was the functional unit used to identify rotations with the highest water-use efficiency. To simplify the analysis, we assumed that one tonne of product was equivalent in function, independent of the crop type. The physical unit of the productive function was the overall m^3 of water consumed per tonne of harvested yield, as dry matter per area, during the 3-year rotation. Water consumed per crop was allocated to the dry matter of harvested grains for barley, wheat, oilseed rape, sunflower and maize, and to the whole biomass for sorghum and poplar, considering these as the products from each crop. The Mediterranean forest output was related to its annual aboveground biomass production.

Current yield data of barley, wheat, oilseed rape, sunflower and maize, regionalised at a province level, were from the Spanish Ministry of the Environment and Rural and Marine Affairs, for the period 2003-2006. Province yield data were assigned to the watershed resolution according to the province area share within a specific watershed. Information on the harvested dry matter for crop and rotation in each watershed is provided in table 5SI of the supporting information, at the end of this chapter. Average whole biomass production was taken as 7.50 t $ha^{-1}y^{-1}$ for sorghum and 13.50 $tha^{-1}y^{-1}$ for poplar in all watersheds, as regional yield data of these energy crops are not yet available. These yield values come from deficit-irrigated experimental plots cultivated in Spain to produce energy from biomass (SSP On Cultivos). The annual net aboveground primary production of the Mediterranean forest was considered as 5.65 t $ha^{-1}y^{-1}$ (Ibàñez et al. 1999).

Assessment of blue water consumption

The LCIA method developed by Pfister et al. (2009) was used for evaluating the ecological impacts of the area-based and the yield-based systems. This method includes the use of a water stress index (WSI) as a characterisation factor for a midpoint water deprivation category as well as an assessment of damages to resources, human health and ecosystem quality that is compatible with the EI99 framework (Goedkoop and Spriensma 2001). The WSI, ranging from 0.01 to 1, is based on the ratio of freshwater withdrawals to blue water availability in each watershed, and indicates the portion of consumptive water use that deprives downstream users of freshwater. Damages to resources are measured using surplus energy units (MJ), human health damages using DALYs, and damages to ecosystems using PDFs. In developed countries such as Spain, water scarcity does not affect human health, for instance causing malnutrition or diarrhoea, as these countries are able to compensate for reduced freshwater availability by for example, water desalination. The midpoint water deprivation category, the endpoint factors for resources and ecosystem damages, as well as the aggregated EI99 single-scores (Pfister et al. 2009) were applied.

Assessment of green water consumption

A two-tiered approach was used, depending on the unit of the assessment.

Spatial agricultural management scope. For the area-based assessment, the green water scarcity index (GWSI) defined by the Water Footprint Network (http://www.waterfootprint.org) was applied (equation 3). Data collected for the LCI were used to obtain results for this index, which provides information at the plot level.

$$\mathrm{GWSI} = \frac{\mathrm{GW}}{\mathrm{P'}} \qquad 0 \le \mathrm{GWSI} \le 1 \qquad (3)$$

Where GWSI is dimensionless, Pr’ is the effective precipitation per area during the 3 years of the rotation (m^3m^{-2} rotation^{-1}) and GW is the amount of Pr’ consumed by the plant in this same area and time unit, that is, green water consumption (m^3m^{-2} rotation^{-1}). GWSI indicates the aridity stress where crops grow. Low values are more favourable for the environment, as they indicate less stress upon available soil water. As effective precipitation and green water depend on the vegetation, the green water scarcity index is especially useful to decide on the spatial location of rainfed rotations and to compare green water demands of these rainfed rotations with those of natural reference systems at a particular site. GWSI could be used as an additional indicator in environmental assessments of agricultural and forestry systems, so supplementing the results of LCA studies by ranking different cultivation areas/regions in regard to the absolute aridity stress.

Productive scope. For the yield-based assessment, the WSI used for assessing impacts of blue water consumption was also applied to the delta green water consumption (dGW, that is, GW consumed by the system studied minus GW consumed by the reference system), providing a weighted dGW value (WSI×dGW). A change in green water use compared to the reference systems modifies river discharge, and thus long-term downstream water availability, which may contribute to intensify or reduce water scarcity. Following this approach, the evaluation of green water flows is compatible with the evaluation for blue water flows. Nevertheless, impacts from both types of water consumption should not be added together for a single-impact score as their economic and ecological values differ (Ridoutt and Pfister 2010). Opportunity costs of blue water are generally higher than those of green water, because blue water consumption by an activity can deprive many downstream users, whilst green water is mostly only available naturally on land for plants, except when it is converted into blue water or causes more runoff (blue water).

Based on the blue water and the green water consumption assessments, the most appropriate places and rotations to minimise environmental impacts of water consumption for growing energy crops in Spain can be selected. As we did not define a single-score indicator with simultaneous consideration of the blue and green water scores, suitable locations and rotations will depend on the type of water (i.e., blue or green) analysed.

Assessment of land use

As (blue and green) water use and land use are closely linked, a screening assessment of impacts due to land occupation of the studied rotations was conducted to show the trade-offs existing between water and land use. This analysis was done using the damage-oriented EI99 LCIA method (Goedkoop and Sprinsma 2001) on the level of biodiversity damage as well as for the aggregated EI99 results. The default normalisation and weighting factors (hierarchist perspective, EI99HA) were used to calculate single-scores.

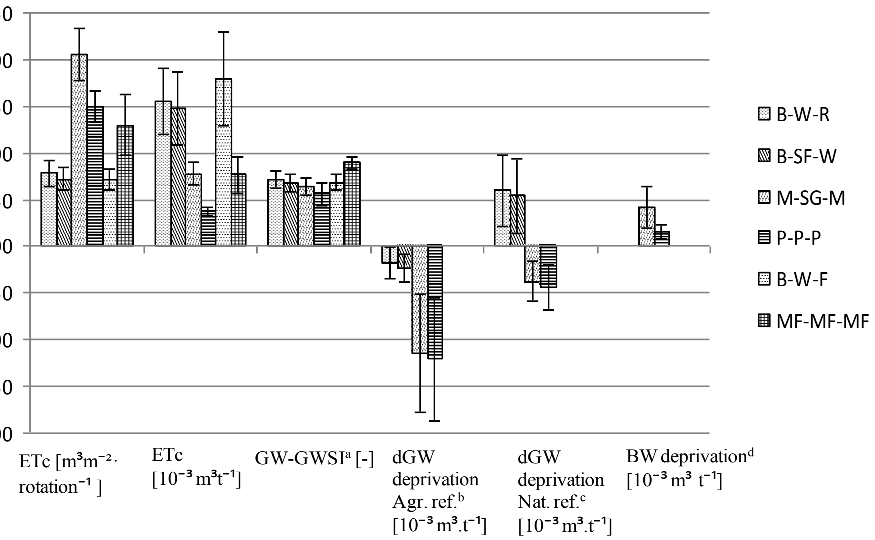
2.3 Results

2.3.1 Water use assessment

Life cycle inventory

The blue and green water consumption of the crop rotations regionalised at river basin level is shown in table 1SI of the supporting information, at the end of this chapter. Irrigated and deficit-irrigated rotations were more water-intensive per area but more water-use efficient per product yield than rainfed rotations (figure 10, ET_c m^3 m^{-2} $rotation^{-1}$ and ET_c m^3 t^{-1}), due to higher yields of irrigated and deficit-irrigated crops. Considering the two crop rotations with irrigation, there was higher blue water consumption by M-SG-M than P-P-P in all watersheds for both the m^2 (average of 1.78 times more) and the tonne based assessments (average of 2.65 times more). This is because a fully irrigated crop (maize) was grown in two of the three years in the M-SG-M rotation, whereas poplar is a support-irrigated crop. Basins with the lowest blue water consumption were located in northeast Spain, both for M-SG-M and P-P-P rotations. In this region, rainfall in the warm months of spring and summer (average of these months > 300 mm) is sufficient for reducing or refraining from watering of (support-) irrigated crops. In addition, ET_0 (and consequently ET_c), does not reach the high levels of other regions (average ET_0 in the northeast Spain is 860 mm y^{-1} *versus* 1,000-1,500 mm y^{-1} in the other studied watersheds). In contrast, several basins located in the southeast, with low precipitation and high evapotranspiration rates, recorded the highest blue water consumptions (table 1SI of the supporting information).

As far as green water is concerned, consumption by the natural reference system (MF) was, on average, 80% higher than the crop production systems in absolute terms ($m^3 m^{-2}$ $rotation^{-1}$). On the other hand, when water productivity was taken into account ($m^3 t^{-1}$), the agricultural reference system (B-W-F) recorded the greatest green water consumption per product yield, which is, on average, 52% more than for the energy crop production systems.



B: winter barley; W: winter wheat; R: oilseed rape; SF: sunflower; M: maize; SG: sorghum; P: poplar; F: fallow; MF: Mediterranean forest.

[a] GWSI: green water consumption to soil-water availability ratio (GW/Pr')

[b] dGW deprivation under agricultural reference:
WSI×(GW consumption system studied - GW consumption B-W-F rotation)

[c] dGW deprivation under natural reference:
WSI×(GW consumption system studied - GW consumption MF-MF-MF rotation)

[d] BW deprivation: WSI×BW consumption

Figure 10 Comparison of water consumption and its environmental effects among energy crop rotations and the reference agricultural and natural systems. Values are the average for all the Spanish watersheds considered in this study. Error bars denote variation among watersheds.

Blue water life cycle impact assessment

The potential environmental damages of blue water consumption for 5 relevant Spanish basins covering different regions of the country are shown in table 7. For a complete list of all basins, see table 2SI of the supporting information.

Using the WSI as assessment indicator for water deprivation, most watersheds in the south and southeast of Spain have the greatest environmental implications, both under the spatial agricultural management scope and the product life cycle assessment. For the yield-based assessment, basins in these areas scored between 11-times higher for the M-SG-M rotation and 5-times higher for the P-P-P rotation than basins with the lowest deprivation impacts, located in the north and northeast (table 2SI). The fact that in 18 out of the 23 basins the WSI was >0.5 indicates the severe water stress affecting a large part of the basins in the country, mainly those in the south, southeast and east.

Damage to resources and ecosystems according to endpoint factor analyses as well as the aggregating EI99 method were generally higher for the M-SG-M rotation system than for the P-P-P rotation (table 2SI). For the resources category, there was no water depletion in 13 out of the 23 assessed basins. These watersheds are scattered all over the country. Major regional water depletion was recorded in southeastern basins.

The general trend of the EI99 single-score factor revealed that northern and northeastern basins in Spain appeared to be the best choice for reducing blue water consumption and ecological impacts from cultivating irrigated rotations, whereas these rotations should not be cultivated in several basins of southeast Spain (figure 11a).

Green water assessment

Spatial agricultural management scope. Looking at the outcomes of the green water scarcity index, no single rotation scored the best in all watersheds (table 7 and table 3SI of the supporting information). Here, results vary among basins because the GWSI does not only depend on green water consumption by plants, but also on conditions of the location, such as the soil water availability, determined by soil properties (texture, slope, land use), and the amount and distribution of rainfall during the year. Irrigated rotations recorded very similar or even lower GWSI values (indicating lower aridity stress) than rainfed rotations in some water basins, which means that they do not use more soil water compared to non-irrigated rotations. The Mediterranean forest GWSI was the highest (most arid) in all basins (figure 10, GWSI). Watersheds where crop rotations grow in conditions with more soil green water availability are not clustered in one specific area of the country, but in several small basins in the northeast and south (figure 11b). In these basins, the GWSI score was ≤ 0.65, indicating less stress upon soil water reserves compared to other watersheds, where the GWSI reveals that nearly 90% of the soil reserves are consumed.

Productive scope. Delta green water deprivation values showed that all rotations affect green water resources per product yield less than the agricultural reference system B-W-F (negative values of weighted dGW in table 7, table 3SI and figure 10 dGW deprivation Agr. ref.). If the potential natural land situation is chosen as the reference, rainfed rotations caused higher deprivation impacts (positive values of weighted dGW in table 7, table 3SI and figure 10 dGW deprivation Nat. ref.). Irrigated crop rotations were those with the lowest impacts on green water. The optimal spatial distribution of rotations fits well with the distribution for minimising impacts of blue water consumption (figure 11c). Along with the northern and northeastern water basins recommended in the blue water assessment, some central and southern areas had the lowest weighted delta green water values. This is because most of these water basins have the lowest WSI values in Spain and high crop yields.

2.3.2 Land use assessment

A list of land use damages on the level of ecosystem quality and the aggregated EI99 single-scores is shown in table 7 and in table 4SI of the supporting information.

The natural reference system (MF-MF-MF) was by far the rotation with the least land use occupation impacts, having, as a country average, 97% and 82% less impacts than the other rainfed (including the agricultural reference state) and irrigated rotations, respectively.

Variation of aggregated impacts of land use occupation was more than 65% among watersheds. The lowest occupation impacts of growing rainfed rotations were in northern and central areas (figure 11d), whereas watersheds more suitable for irrigated rotations were found in central, southern and southeastern regions (figure 11e), some with the highest blue water use impacts. The lowest land use occupation impacts were found in the regions with the highest crop yields, usually with higher water consumption.

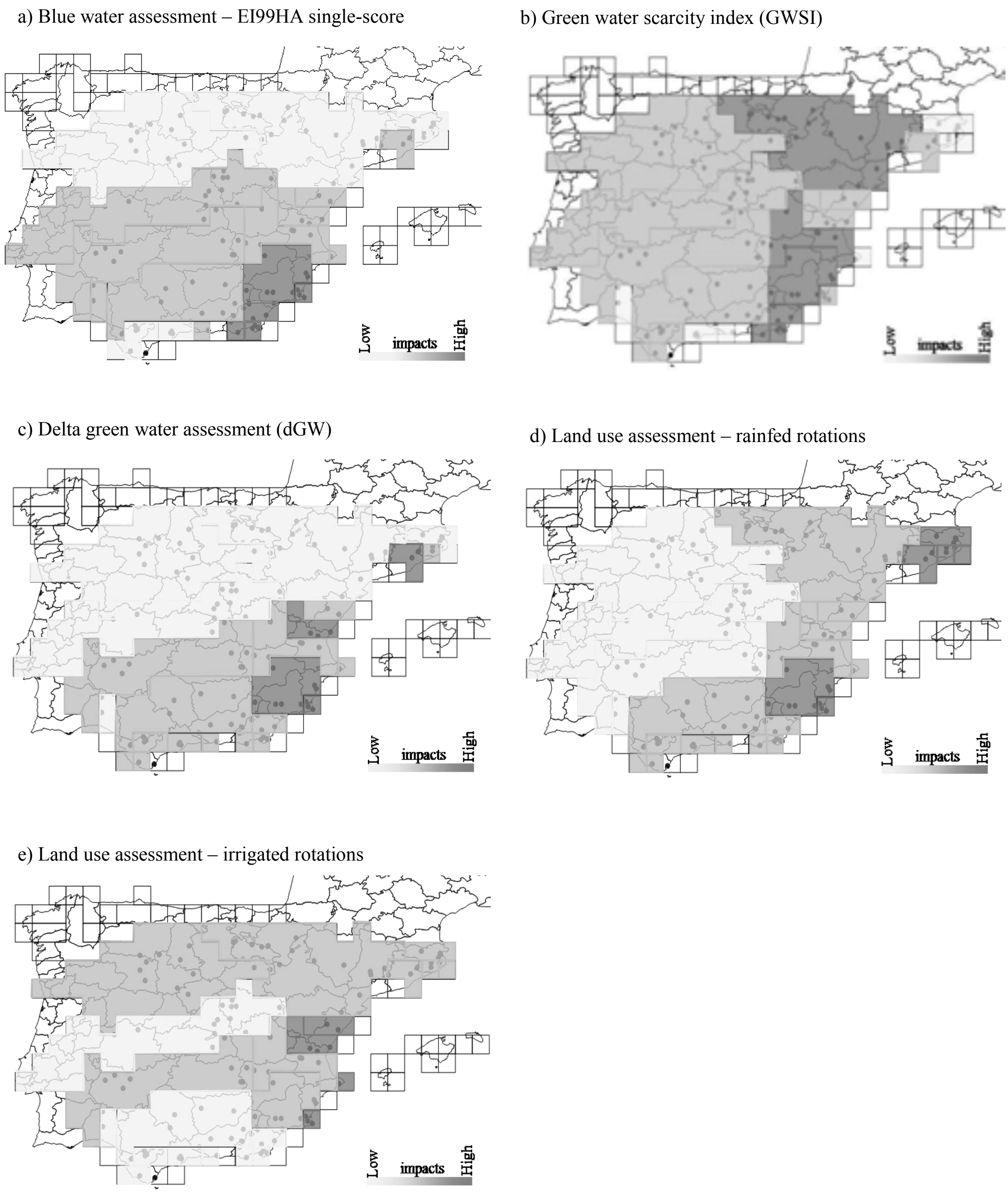


Figure 11 Location of the most and the least appropriate cultivation watersheds for a) blue water assessment - EI99HA single-score, b) green water scarcity index (GWSI), c) delta green water assessment (dGW), d) land use assessment for rainfed rotations, and e) land use assessment for irrigated rotations. The legend of the figures (high and low) is defined by ranking the impact scores of each watershed.

Table 7 Impacts of blue water and green water consumption for the assessments per m^2 and per tonne dry matter, and of land use per tonne dry matter for 5 relevant Spanish watersheds. Complete list of all basins in the supporting information.

Basin-ID[a]	Location in Spain	Crop rotation	Spatial agricultural management scope (m^2)					Productive scope (t)						Land use assessment
			Blue water				Green water	Blue water				Delta green water deprivation [m^3 t^{-1}]		
			Water deprivation [b] [m^3 m^{-2} $rotation^{-1}$]	Resources [MJm^{-2} $rotation^{-1}$]	Ecosystem quality [$PDFm^2yrm^{-2}$ $rotation^{-1}$]	EI99HA single-score [points m^{-2} $rotation^{-1}$]	GWSI[c] [-]	Water deprivation[b] [m^3 t^{-1}]	Resources [MJt^{-1}]	Ecosystem quality [$PDFm^2yrt^{-1}$]	EI99HA single-score [points t^{-1}]	Agricultural ref.[d]	Natural ref.[e]	EI99HA single-score [points t^{-1}]
31897	Northeast	B-W-R	0.000	0.000	0.000	0.00	0.62	0.00	0.00	0.00	0.00	-18.78	46.07	576.03
		B-SF-W	0.000	0.000	0.000	0.00	0.57	0.00	0.00	0.00	0.00	-19.52	45.32	628.52
		M-SG-M	0.037	0.000	0.171	0.013	0.53	13.99	0.00	64.28	5.013	-91.92	-27.07	100.98
		P-P-P	0.026	0.000	0.122	0.009	0.50	6.59	0.00	30.27	2.361	-94.42	-29.57	67.42
		B-W-F	0.000	0.000	0.000	0.00	0.58	0.00	0.00	0.00	0.00	0.00	-	793.53
		MF-MF-MF	0.000	0.000	0.000	0.00	0.78	0.00	0.00	0.00	0.00	-	0.00	15.18
32581	North	B-W-R	0.000	0.000	0.000	0.00	0.71	0.00	0.00	0.00	0.00	-22.29	64.29	377.82
		B-SF-W	0.000	0.000	0.000	0.00	0.68	0.00	0.00	0.00	0.00	-21.18	65.41	403.91
		M-SG-M	0.220	0.000	0.349	0.027	0.64	82.75	0.00	130.93	10.20	-164.55	-77.97	101.05
		P-P-P	0.154	0.000	0.244	0.019	0.55	38.59	0.00	61.06	4.759	-178.18	-91.60	67.42
		B-W-F	0.000	0.000	0.000	0.00	0.69	0.00	0.00	0.00	0.00	0.00	-	460.63
		MF-MF-MF	0.000	0.000	0.000	0.00	0.90	0.00	0.00	0.00	0.00	-	0.00	15.18
34697	Southeast	B-W-R	0.000	0.000	0.000	0.00	0.88	0.00	0.00	0.00	0.00	-534.80	1470.77	789.80
		B-SF-W	0.000	0.000	0.000	0.00	0.84	0.00	0.00	0.00	0.00	-493.63	1511.93	848.60
		M-SG-M	1.674	16.15	0.961	0.459	0.75	674.34	6508.22	387.28	185.11	-2271.44	-265.88	108.39
		P-P-P	0.900	8.69	0.517	0.247	0.79	225.56	2176.96	129.54	61.92	-2336.10	-330.53	67.42
		B-W-F	0.000	0.000	0.000	0.00	0.85	0.00	0.00	0.00	0.00	0.00	-	1065.79
		MF-MF-MF	0.000	0.000	0.000	0.00	0.98	0.00	0.00	0.00	0.00	-	0.00	15.18
35701	Southeast	B-W-R	0.000	0.000	0.000	0.00	0.87	0.00	0.00	0.00	0.00	-132.81	607.83	487.33
		B-SF-W	0.000	0.000	0.000	0.00	0.84	0.00	0.00	0.00	0.00	-196.80	543.84	474.44
		M-SG-M	1.780	14.15	1.225	0.432	0.82	601.48	4779.95	413.79	146.06	-967.85	-227.21	90.88
		P-P-P	0.900	7.45	0.638	0.227	0.81	225.56	1867.66	159.94	56.93	-1005.87	-265.23	67.42
		B-W-F	0.000	0.000	0.000	0.00	0.85	0.00	0.00	0.00	0.00	0.00	-	578.51
		MF-MF-MF	0.000	0.000	0.000	0.00	0.99	0.00	0.00	0.00	0.00	-	0.00	15.18
36039	South	B-W-R	0.000	0.000	0.000	0.00	0.65	0.00	0.00	0.00	0.00	-109.71	430.80	487.33
		B-SF-W	0.000	0.000	0.000	0.00	0.59	0.00	0.00	0.00	0.00	-274.09	266.42	474.44
		M-SG-M	1.748	0.000	0.196	0.015	0.56	590.49	0.00	66.13	5.137	-1292.19	-751.68	90.88
		P-P-P	0.900	0.000	0.101	0.008	0.43	225.56	0.00	25.26	1.962	-1347.26	-806.75	67.42
		B-W-F	0.000	0.000	0.000	0.00	0.62	0.00	0.00	0.00	0.00	0.00	-	578.51
		MF-MF-MF	0.000	0.000	0.000	0.00	0.93	0.00	0.00	0.00	0.00	-	0.00	15.18

B = barley; W = wheat; R = oilseed rape; SF = sunflower; M = maize; SG = sorghum; P = poplar; MF = Mediterranean forest.

[a] ID and name of each drainage basin in the supplementary material.

[b] WSI×BW consumption.

[c] Green water scarcity index: green water consumption to soil-water availability ratio (GW/Pr').

[d] dGW deprivation under agricultural reference: WSI×(GW consumption system studied - GW consumption B-W-F rotation).

[e] dGW deprivation under natural reference: WSI×(GW consumption system studied - GW consumption MF-MF-MF rotation).

2.4 Discussion

2.4.1 Choice of the basis for comparison

The selection of an appropriate functional unit for LCA of agricultural systems is not an easy task, being currently a key subject of discussion among experts in the field. In this study, the environmental burdens of water consumption were quantified per m^2 rotation and per tonne dry matter produced in each rotation. Whilst the m^2-based assessment is strictly for land management assessment, as a unit it provides relevant information to compare total water consumption and environmental impacts of a specific rotation among watersheds. However, for comparing alternative crops and rotations using LCA, neither the m^2 nor the tonne based assessments are appropriate, as the crops have different functions per m^2 and per tonne, and the definition of the functional unit is that all systems being compared have a common function. For example, wheat and barley share a similar function to make flour for producing bread, although they can be used for other purposes depending on the variety and where they are cultivated (e.g., wheat to make flour for biscuits or cakes, barley for animal feed and the elaboration of beer). On the other hand, oilseed rape is primarily cultivated for animal feed, producing vegetable oil for human consumption and biodiesel. Despite this, we chose tonne of harvested dry matter as the functional unit, rather than others that could better reflect the purpose of energy crops, such as the crop's calorific value, in order to avoid unfair comparisons between food and energy crops. For users who prefer another functional unit for an allocation we also provide the yield values of all crops in table 5SI of the supporting information.

2.4.2 Green water assessment

There was only partial agreement between the two methods applied to evaluate impacts of green water consumption. Whilst the best scores were with irrigated crop rotations for both approaches, the trends for the most suitable areas to grow energy crop rotations differed. Both assessments are still compatible if we apply a tiered approach.

Firstly, estimates of the use of green water additional to the reference situation have to be considered as the LCA base results so that water basins can be ranked according to their weighted green water impacts. This dGW deprivation indicator is useful for estimating the environmental burdens on the additional blue water resources of the watershed that the consumption of green water could contribute. The WSI used in the calculation of dGW deprivation (WSI×dGW consumption) is defined by Pfister et al. (2009) as a modified ratio of blue water withdrawal to blue water availability. While it is not entirely accurate to use this to make the assessment of the additional green water consumption LCA-compatible, since the ratio is based on blue water flows, we applied the WSI weighting, as dGW also influences the availability of blue water and thus may enhance water scarcity.

Secondly, once watersheds have been ranked, recommendations for agricultural land management and optimisation can be based on complementing the LCA results with the outcomes from the GWSI. This gives information on which watersheds are the most suitable in relation to the plant green water requirements and the soil green water availability.

Due to the difficulties of assessing green water in the water use impact category in LCA, it can be argued that its consumption should be evaluated when modelling the cause-effect chain of water use impacts. An alternative is to consider the change in green water flows as part of the land use impact category (Milà i Canals et al. 2007a), since land and water use impacts are closely linked. Green water assessment within the land use category has not yet been solved, and is a matter of further research.

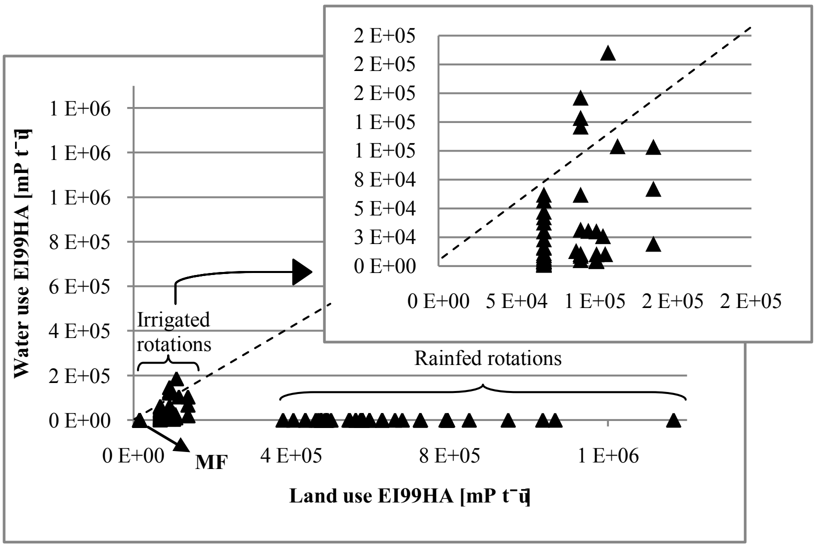
2.4.3 Comparing water consumption and land use of rainfed and irrigated energy crop rotations

It can be seen in figure 10 how blue and green water consumption (ET_c) vary depending on the type of assessment. While the m^2-based assessment indicates that attention must be paid to the total water consumption of the irrigated rotations M-SG-M (1.6-2.9 times higher than the other rotations) and P-P-P (1.2-2.1 times higher than the other rotations), these have the highest water-use efficiency per output function (M-SG-M: 0.43-0.52 and P-P-P: 0.21-0.49 times the consumption of the other rotations). These results indicate that both assessments are required for a sound evaluation and an efficient allocation of water resources, indicating the trade-off between both objectives. For regional resource management, the m^2-based assessment is helpful to alleviate water use intensity.

Considering the blue water assessment, rainfed rotations were the best option to minimise environmental burdens, as they are not irrigated (figure 10, BW deprivation). If irrigation efficiency had been considered in the inventory of blue water consumption, there would have been more differences between irrigated and non-irrigated rotations. It was not considered, as the irrigation system applied for each crop varies among, and within each, watershed. We still recommend that irrigation efficiency is considered for a particular case study, where excess irrigation water recharges soil, increasing its green water availability, and may also drain off to groundwater or be evaporated unproductively.

With green water, different trends in impacts were detected depending on the assessment method applied (GWSI and dGW deprivation, figure 10), albeit, with both indices, irrigated rotations had the lowest impacts on the additional blue water flows and on the soil green water availability of the plot.

The land use impact assessment per tonne of harvested dry matter, showed the land/water trade-off between irrigated and rainfed agriculture within watersheds (figure 12). While in rainfed agriculture there are no impacts on the blue water resources, there is 85% higher damage to ecosystems per product compared to irrigated rotations due to land use, on a nationwide average. This is because of the lower yields and, therefore, lower crop-water productivity of non-irrigated crops. In the vast majority of cases, the scores using EI99 methodology were higher for land use occupation impacts compared to water consumption impacts (figure 12, triangles below the line with slope = 1). The higher scores of the land use impacts in the EI99 method may be because the land use impact assessment was based on vascular plant species diversity in the Swiss lowlands and might therefore not be appropriate for Spain, with Mediterranean, not temperate forest, as the reference system.



MF: Mediterranean forest

Figure 12 Comparison of land use occupation impacts and blue water consumption impacts based on the aggregated EI99HA scores. Each triangle represents the combination of land/water use damages of a specific rotation in one of the studied watersheds. Triangles below the line (slope=1) indicate that land use impacts are greater than water use impacts for the rotation in the water basin.

2.4.4 Comparing green water consumption of energy crop rotations and reference systems

When the environmental impacts caused by green water consumption of the energy crops were evaluated against the reference agricultural and natural situations, the most harmful effects were generally obtained for the reference systems (figure 10, GWSI and dGW deprivation).

The agricultural reference system (B-W-F) gave the highest burdens per yield for change of green water consumption (dGW deprivation) when compared with the crop rotations. The reference rotation is under fallow during the third year, so the total water consumption is only allocated to the yield of the two years of low-productive, rainfed cereals.

Looking at the impacts on the on-site availability of green water (GWSI), the natural reference system (MF-MF-MF) was grown in some basins with up to 40% more severe aridity stress conditions than the other rotations. The higher green water consumption per area of Mediterranean forest compared to the four energy crop rotations is mainly due to the reduction of ET_c in cropstages with residue cover or bare soil in agricultural systems. Many studies have pointed out higher soil water retention capacity, higher water requirements and a decreasing runoff from areas under forest as compared with other plant communities in similar environmental conditions (Calder 2004; Valentíni 2003). This indicates that upstream forest cover does not generally enhance water availability downstream. In dry zone regions, as in most parts of Spain, soil moisture seldom reaches the field capacity necessary to percolate water to recharge aquifers, independently of the vegetation cover and the infiltration properties of soils (Bellot and Ortiz de Urbina 2008). Shorter agricultural rotations can therefore be considered beneficial as compared to the natural reference situation, since crop production creates more runoff that increases the streamflow of the river basin. Other environmental issues, for example, carbon sequestration, must be considered to fully compare agricultural and forestry systems.

Comparisons between the Mediterranean forest reference system and the crop rotations have to be interpreted with caution because: 1) Green water consumption (ET_c) of agricultural and natural systems was estimated using different methods, so our results should be compared with alternative methods for measuring green water flows (e.g., Lund-Potsdam-Jena model, LPJ, Gerten et al. 2005), 2) There is a lack of information on water consumption by forests, which should be thoroughly evaluated in comprehensive studies of their water balance, 3) We assigned the yearly net aboveground primary production of the Mediterranean forest as its energetic output for the ton-based comparison, even though the wood is not harvested.

As shown in figure 10, the use of the agricultural or the natural reference system does not influence the relative results of impact scores among rotations, but it may influence the importance attached to water use in the overall assessment. Therefore, choosing the most suitable and consistent reference is a crucial issue in the evaluation of agricultural system alternatives in each ongoing assessment of water use impacts.

Milà i Canals et al. (2007a) recommend using natural relaxation, that is, the potential natural vegetation, in attributional LCA. This type of LCA assessment aims to describe the overall system impacts relative to a situation where the human activities under study do not take place. On the other hand, if the study is aimed at evaluating the consequences of changes (consequential LCA), the alternative system, here the agricultural crop rotation without energy crops, may become the reference, assuming that farmers not producing energy crops will produce conventional grains. Despite these recommendations, the appropriateness of using the potential natural vegetation as the reference for attributional LCA in areas devoted to agricultural purposes for centuries is disputable. Moreover, energy crops, instead of occupying forest, can be grown in abandoned agricultural areas in Spain, thus an agricultural reference can be more appropriate.

2.5 Conclusions

The environmental impacts of freshwater consumption from growing agricultural products can be adequately assessed within LCA using the method presented in Pfister et al. (2009), provided there is blue water consumption. Up to now, emphasis has been given to the blue water consumption. Here, two methods were applied to also measure impacts from green water consumption.

If the aim is the production of agroenergetic crops and their transformation and utilisation locally, as well as not putting further pressure on water resources, basins in northeast Spain are suitable locations for energy crop rotations, whereas they should not be cultivated in some basins in the southeast. However, additional agricultural production in Spain should be carefully assessed and further alternatives for non-fossil energy production, such as solar power systems, should be analysed.

Together with the water assessment, studies covering land use impacts on biodiversity and ecosystem services (e.g., carbon sequestration potential, erosion regulation potential) must be measured, for a full comparison of natural and agricultural systems.

2.6 Supporting information

Table 1SI Life cycle inventory results for the assessments per m^2 and per tonne dry matter for the 23 studied watersheds.

Table 2SI Life cycle impact assessment results of blue water consumption for the assessments per m^2 and per tonne dry matter for the 23 studied watersheds.

Table 3SI Green water scarcity index for the assessment per m^2 and life cycle impact assessment of green water for the assessment per tonne dry matter of the green water consumption for the 23 studied watersheds.

Table 4SI Land use assessment per tonne dry matter for the 23 studied watersheds.

Table 5SI Harvested dry matter per crop for the 23 studied watersheds.

Table 1SI Life cycle inventory results for the assessments per m^2 and per tonne dry matter for the 23 studied watersheds.

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop rotations	Harvested dry matter[b] [t ha^{-1}]	LCI: Spatial agricultural management scope: Blue water consumption [m^3 m^{-2} rotation^{-1}]	LCI: Spatial agricultural management scope: Green water consumption [m^3 m^{-2} rotation^{-1}]	LCI: Productive scope[c]: Blue water consumption [m^3 t^{-1}]	LCI: Productive scope[c]: Green water consumption [m^3 t^{-1}]
31897	Muga-Fluvià (n=4)	Internal watersheds of Catalonia	B-W-R	4.67	0.0000	0.9105	0.00	1949.68
			B-SF-W	4.28	0.0000	0.8272	0.00	1932.71
			M-SG-M	26.64	0.8506	0.7455	319.29	279.84
			P-P-P	39.90	0.6000	0.8890	150.38	222.81
			B-W-F	3.39	0.0000	0.8063	0.00	2378.47
			MF-MF-MF	16.95	0.0000	1.5220	0.00	897.94
32250	Tordera (n=2)	Internal watersheds of Catalonia	B-W-R	4.67	0.0000	1.0010	0.00	2143.47
			B-SF-W	4.28	0.0000	0.9109	0.00	2128.27
			M-SG-M	26.64	0.7346	0.8716	275.75	327.18
			P-P-P	39.90	0.3000	0.9766	75.19	244.76
			B-W-F	3.39	0.0000	0.8898	0.00	2624.78
			MF-MF-MF	16.95	0.0000	1.6300	0.00	961.65
32251	Ter (n=1)	Internal watersheds of Catalonia	B-W-R	4.67	0.0000	0.9534	0.00	2041.54
			B-SF-W	4.28	0.0000	0.8579	0.00	2004.44
			M-SG-M	26.64	0.8212	0.8045	308.26	301.99
			P-P-P	39.90	0.3000	0.8477	75.19	212.46
			B-W-F	3.39	0.0000	0.8477	0.00	2500.59
			MF-MF-MF	16.95	0.0000	1.5437	0.00	910.74
32581	Duero, Esla, Pisuerga, Tormes, Orbigo (n=13)	Duero	B-W-R	7.12	0.0000	0.7714	0.00	1083.43
			B-SF-W	6.66	0.0000	0.7259	0.00	1089.94
			M-SG-M	26.62	1.2875	0.6708	483.66	251.99
			P-P-P	39.90	0.9000	0.6876	225.56	172.33
			B-W-F	5.84	0.0000	0.7088	0.00	1213.70
			MF-MF-MF	16.95	0.0000	1.1995	0.00	707.67
32603	Besòs-Maresme, Llobregat, Anoia (n=3)	Internal watersheds of Catalonia	B-W-R	4.67	0.0000	0.8205	0.00	1756.96
			B-SF-W	4.28	0.0000	0.7586	0.00	1772.43
			M-SG-M	26.64	0.8448	0.7672	317.12	287.99
			P-P-P	39.90	0.7000	0.8181	175.44	205.04
			B-W-F	3.39	0.0000	0.7385	0.00	2178.47
			MF-MF-MF	16.95	0.0000	1.3210	0.00	779.35
32963	Ebro, Segre, Cinca, Arga (n=17)	Ebro	B-W-R	5.77	0.0000	0.8832	0.00	1530.68
			B-SF-W	5.57	0.0000	0.8160	0.00	1464.99
			M-SG-M	25.24	1.2332	0.7525	488.59	298.14
			P-P-P	39.90	0.9000	0.8122	225.56	203.56
			B-W-F	4.69	0.0000	0.7948	0.00	1694.67
			MF-MF-MF	16.95	0.0000	1.2345	0.00	728.32

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop rotations	Harvested dry matter[b] [t ha^{-1}]	LCI: Spatial agricultural management scope: Blue water consumption [m^3 m^{-2} rotation^{-1}]	LCI: Spatial agricultural management scope: Green water consumption [m^3 m^{-2} rotation^{-1}]	LCI: Productive scope[c]: Blue water consumption [m^3 t^{-1}]	LCI: Productive scope[c]: Green water consumption [m^3 t^{-1}]
33322	Cenia-Maestrazgo, Mijares-Plana de Castellón (n=2)	Júcar	B-W-R	4.79	0.0000	1.0110	0.00	2110.65
			B-SF-W	4.51	0.0000	0.8941	0.00	1982.48
			M-SG-M	19.58	1.1448	0.8562	584.68	437.28
			P-P-P	39.90	0.9000	0.9401	225.56	235.61
			B-W-F	3.71	0.0000	0.8881	0.00	2393.80
			MF-MF-MF	16.95	0.0000	1.5072	0.00	889.20
33676	Túria (n=3)	Júcar	B-W-R	4.79	0.0000	0.9481	0.00	1979.33
			B-SF-W	4.51	0.0000	0.8672	0.00	1922.84
			M-SG-M	19.58	1.1816	0.8477	603.47	432.94
			P-P-P	39.90	0.7000	0.9700	175.44	243.11
			B-W-F	3.71	0.0000	0.8566	0.00	2308.89
			MF-MF-MF	16.95	0.0000	1.4196	0.00	837.52
34026	Júcar, Marina Baja, Serpis (n=6)	Júcar	B-W-R	4.94	0.0000	0.8293	0.00	1678.74
			B-SF-W	4.64	0.0000	0.7717	0.00	1663.15
			M-SG-M	25.58	1.5615	0.6985	610.44	273.06
			P-P-P	39.90	0.9000	0.7467	225.56	187.14
			B-W-F	3.96	0.0000	0.7665	0.00	1935.61
			MF-MF-MF	16.95	0.0000	1.1758	0.00	693.69
34348	Tajo, Alagón, Guadarrama, Guadiela (n=12)	Tajo	B-W-R	5.72	0.0000	0.7585	0.00	1326.05
			B-SF-W	5.48	0.0000	0.7013	0.00	1279.74
			M-SG-M	30.56	1.4603	0.6356	477.85	207.98
			P-P-P	39.90	0.9000	0.6872	225.56	172.23
			B-W-F	4.92	0.0000	0.6970	0.00	1416.67
			MF-MF-MF	16.95	0.0000	1.2098	0.00	713.75
34367	Marina Alta (n=1)	Júcar	B-W-R	4.79	0.0000	0.8821	0.00	1841.54
			B-SF-W	4.51	0.0000	0.7792	0.00	1727.72
			M-SG-M	19.58	1.0559	0.6992	539.27	357.10
			P-P-P	39.90	0.3000	0.7676	75.19	192.38
			B-W-F	3.71	0.0000	0.7734	0.00	2084.64
			MF-MF-MF	16.95	0.0000	1.9703	0.00	1162.42
34697	Segura, Guadalentín, Mundo, Mula (n=8)	Segura	B-W-R	3.41	0.0000	0.6700	0.00	1964.81
			B-SF-W	3.18	0.0000	0.6379	0.00	2005.97
			M-SG-M	24.82	1.6737	0.5663	674.34	228.16
			P-P-P	39.90	0.9000	0.6524	225.56	163.51
			B-W-F	2.53	0.0000	0.6324	0.00	2499.60
			MF-MF-MF	16.95	0.0000	0.8374	0.00	494.04
35032	Mar Menor (n=3)	Segura	B-W-R	2.84	0.0000	0.4834	0.00	1702.11
			B-SF-W	2.60	0.0000	0.4628	0.00	1780.00
			M-SG-M	23.50	1.4777	0.4478	628.81	190.55
			P-P-P	39.90	0.9000	0.4741	225.56	118.82
			B-W-F	1.97	0.0000	0.4608	0.00	2339.09
			MF-MF-MF	16.95	0.0000	0.8359	0.00	493.16

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop rotations	Harvested dry matter[b] [t ha^{-1}]	LCI: Spatial agricultural management scope: Blue water consumption [m^3 m^{-2} rotation^{-1}]	LCI: Spatial agricultural management scope: Green water consumption [m^3 m^{-2} rotation^{-1}]	LCI: Productive scope[c]: Blue water consumption [m^3 t^{-1}]	LCI: Productive scope[c]: Green water consumption [m^3 t^{-1}]
35354	Guadiana (n=9)	Guadiana	B-W-R	6.20	0.0000	0.7795	0.00	1257.26
			B-SF-W	6.19	0.0000	0.7202	0.00	1163.49
			M-SG-M	28.08	1.7211	0.6326	612.93	225.28
			P-P-P	39.90	0.9000	0.6698	225.56	167.87
			B-W-F	5.39	0.0000	0.7133	0.00	1323.38
			MF-MF-MF	16.95	0.0000	1.2367	0.00	729.62
35355	Tinto, Odiel, Chanza, Piedras (n=2)	Atlantic of Andalucia	B-W-R	5.52	0.0000	0.8532	0.00	1545.65
			B-SF-W	5.67	0.0000	0.7708	0.00	1359.44
			M-SG-M	29.60	1.6428	0.6863	555.00	231.86
			P-P-P	39.90	0.9000	0.6967	225.56	174.61
			B-W-F	4.65	0.0000	0.7649	0.00	1644.95
			MF-MF-MF	16.95	0.0000	1.7670	0.00	1042.48
35365	Almanzora (n=4)	Mediterranean of Andalucia	B-W-R	5.52	0.0000	0.7363	0.00	1333.88
			B-SF-W	5.67	0.0000	0.6978	0.00	1230.69
			M-SG-M	29.60	1.6892	0.6652	570.68	224.73
			P-P-P	39.90	0.7500	0.7042	187.97	176.49
			B-W-F	4.65	0.0000	0.6905	0.00	1484.95
			MF-MF-MF	16.95	0.0000	1.0096	0.00	595.63
35693	Guadalquivir (n=11)	Guadalquivir	B-W-R	5.52	0.0000	0.7548	0.00	1367.39
			B-SF-W	5.67	0.0000	0.6998	0.00	1234.22
			M-SG-M	29.60	1.6707	0.6504	564.43	219.73
			P-P-P	39.90	0.8455	0.6788	211.90	170.13
			B-W-F	4.65	0.0000	0.6942	0.00	1492.90
			MF-MF-MF	16.95	0.0000	1.2925	0.00	762.54
35697	Guadalhorce, Guadalmedina (n=4)	Mediterranean of Andalucia	B-W-R	5.52	0.0000	0.6512	0.00	1179.71
			B-SF-W	5.67	0.0000	0.5849	0.00	1031.57
			M-SG-M	29.60	1.5759	0.5157	532.40	174.22
			P-P-P	39.90	0.9000	0.5451	225.56	136.62
			B-W-F	4.65	0.0000	0.5811	0.00	1249.62
			MF-MF-MF	16.95	0.0000	1.4668	0.00	865.37
35698	Algarrobo, Sayalonga, Torrox (n=1)	Mediterranean of Andalucia	B-W-R	5.52	0.0000	0.5864	0.00	1062.32
			B-SF-W	5.67	0.0000	0.5218	0.00	920.28
			M-SG-M	29.60	1.3312	0.4432	449.73	149.73
			P-P-P	39.90	0.9000	0.4470	225.56	112.03
			B-W-F	4.65	0.0000	0.5213	0.00	1121.08
			MF-MF-MF	16.95	0.0000	1.0533	0.00	621.42
35700	Guadalfeo (n=2)	Mediterranean of Andalucia	B-W-R	5.52	0.0000	0.6132	0.00	1110.87
			B-SF-W	5.67	0.0000	0.5892	0.00	1039.15
			M-SG-M	29.60	1.7147	0.5689	579.29	192.20
			P-P-P	39.90	0.9000	0.5926	225.56	148.52
			B-W-F	4.65	0.0000	0.5849	0.00	1257.85
			MF-MF-MF	16.95	0.0000	0.7291	0.00	430.15

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop rotations	Harvested dry matter[b] [t ha^{-1}]	LCI			
					Spatial agricultural management scope		Productive scope[c]	
					Blue water consumption [m^3 m^{-2} rotation^{-1}]	Green water consumption [m^3 m^{-2} rotation^{-1}]	Blue water consumption [m^3 t^{-1}]	Green water consumption [m^3 t^{-1}]
35701	Andarax, Adra (n=4)	Mediterranean of Andalucia	B-W-R	5.52	0.0000	0.5587	0.00	1012.14
			B-SF-W	5.67	0.0000	0.5376	0.00	948.15
			M-SG-M	29.60	1.7804	0.5242	601.49	177.09
			P-P-P	39.90	0.9000	0.5549	225.56	139.07
			B-W-F	4.65	0.0000	0.5324	0.00	1144.95
			MF-MF-MF	16.95	0.0000	0.6853	0.00	404.31
36038	Turón, Guadalteba (n=3)	Mediterranean of Andalucia	B-W-R	5.52	0.0000	0.7428	0.00	1345.65
			B-SF-W	5.67	0.0000	0.6822	0.00	1203.17
			M-SG-M	29.60	1.7527	0.6231	592.13	210.51
			P-P-P	39.90	0.9000	0.6784	225.56	170.03
			B-W-F	4.65	0.0000	0.6730	0.00	1447.31
			MF-MF-MF	16.95	0.0000	1.5099	0.00	890.80
36039	Guadalete, Barbate (n=2)	Atlantic of Andalucia	B-W-R	5.52	0.0000	0.7729	0.00	1400.18
			B-SF-W	5.67	0.0000	0.7007	0.00	1235.80
			M-SG-M	29.60	1.7478	0.6444	590.47	217.70
			P-P-P	39.90	0.9000	0.6489	225.56	162.63
			B-W-F	4.65	0.0000	0.7021	0.00	1509.89
			MF-MF-MF	16.95	0.0000	1.6431	0.00	969.38

B = barley; W = wheat; R = oilseed rape; SF = sunflower; M = maize; SG = sorghum; P = poplar; MF = Mediterranean forest.

[a] Basin code of each watershed according to the WaterGAP 2 global model (Alcamo et al. 2003).

[b] Grains for B, W, R, M and SF. Whole biomass for SG, P and MF.

[c] Water consumption allocated to grains or whole plant, depending on the product from each crop: see footnote “b”

Table 2SI Life cycle impact assessment results of blue water consumption for the assessments per m^2 and per tonne dry matter for the 23 studied watersheds.

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop rotations	LCIA ğ Blue water							
				Spatial agricultural management scope				Productive scope[b]			
				Water deprivation [m^3 m^{-2} $rotation^{-1}$]	Resources [MJ m^{-2} $rotation^{-1}$]	Ecosystem quality [$PDFm^2y$ m^{-2} $rotation^{-1}$]	EI99HA single-score [milipoints m^{-2} $rotation^{-1}$]	Water deprivation [m^3 t^{-1}]	Resources [MJt^{-1}]	Ecosystem quality [$PDFm^2y$ t^{-1}]	EI99HA single-score [points t^{-1}]
31897	Muga-Fluvià (n=4)	Internal watersheds of Catalonia	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	0.0373	0.0000	0.1712	13.35	13.99	0.00	64.28	5.01
			P-P-P	0.0263	0.0000	0.1218	9.42	6.59	0.00	30.27	2.36
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
32250	Tordera (n=2)	Internal watersheds of Catalonia	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	0.4225	0.0000	0.1590	12.41	158.59	0.00	59.70	4.66
			P-P-P	0.1725	0.0000	0.0650	5.07	43.24	0.00	16.28	1.27
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
32251	Ter (n=1)	Internal watersheds of Catalonia	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	0.0240	0.0000	0.1448	11.33	9.00	0.00	54.34	4.25
			P-P-P	0.0088	0.0000	0.0529	4.14	2.20	0.00	13.26	1.04
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
32581	Duero, Esla, Pisuerga, Tormes, Orbigo (n=13)	Duero	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	0.2203	0.0000	0.3485	27.17	82.75	0.00	130.93	10.21
			P-P-P	0.1540	0.0000	0.2436	18.99	38.59	0.00	61.06	4.76
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
32603	Besòs-Maresme, Llobregat, Anoia (n=3)	Internal watersheds of Catalonia	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	0.8445	2.4742	0.2732	80.17	316.99	928.74	102.56	30.09
			P-P-P	0.6997	2.0501	0.2264	66.43	175.37	513.81	56.74	16.65
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
32963	Ebro, Segre, Cinca, Arga (n=17)	Ebro	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	0.3197	0.0000	0.3335	26.02	126.64	0.00	132.12	10.31
			P-P-P	0.2333	0.0000	0.2434	18.99	58.47	0.00	60.99	4.76
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop rotations	LCIA ğ Blue water							
				Spatial agricultural management scope				Productive scope[b]			
				Water deprivation [m^3 m^{-2} $rotation^{-1}$]	Resources [MJ m^{-2} $rotation^{-1}$]	Ecosystem quality [$PDFm^2y$ m^{-2} $rotation^{-1}$]	EI99HA single-score [milipoints m^{-2} $rotation^{-1}$]	Water deprivation [m^3 t^{-1}]	Resources [MJt^{-1}]	Ecosystem quality [$PDFm^2y$ t^{-1}]	EI99HA single-score [points t^{-1}]
33322	Cenia-Maestrazgo, Mijares-Plana de Castellón (n=2)	Júcar	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	0.9372	0.0000	0.4845	37.78	478.64	0.00	247.45	19.30
			P-P-P	0.7367	0.0000	0.3809	29.70	184.65	0.00	95.46	7.44
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
33676	Túria (n=3)	Júcar	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	1.1816	6.5021	0.6116	202.41	603.47	3320.79	312.36	103.38
			P-P-P	0.7000	3.8520	0.3623	119.91	175.44	965.40	90.81	30.05
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
34026	Júcar, Marina Baja, Serpis (n=6)	Júcar	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	1.5488	0.0000	0.8511	66.36	605.48	0.00	332.74	25.94
			P-P-P	0.8927	0.0000	0.4906	38.25	223.74	0.00	122.95	9.59
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
34348	Tajo, Alagón, Guadarrama, Guadiela (n=12)	Tajo	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	0.7798	0.0000	0.5088	39.72	255.18	0.00	166.49	13.00
			P-P-P	0.4806	0.0000	0.3136	24.48	120.45	0.00	78.59	6.14
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
34367	Marina Alta (n=1)	Júcar	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	1.0559	4.0952	0.4323	131.15	539.29	2091.53	220.79	66.98
			P-P-P	0.3000	1.1635	0.1228	37.26	75.19	291.60	30.78	9.34
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
34697	Segura, Guadalentín, Mundo, Mula (n=8)	Segura	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	1.6737	16.1534	0.9612	459.44	674.34	6508.22	387.28	185.11
			P-P-P	0.9000	8.6861	0.5169	247.05	225.56	2176.96	129.54	61.92
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
35032	Mar Menor (n=3)	Segura	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	1.4777	12.3155	0.6173	244.29	628.80	5240.65	262.67	103.95
			P-P-P	0.9000	7.5009	0.4035	150.48	225.56	1879.92	101.13	37.71
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop rotations	LCIA ğ Blue water							
				Spatial agricultural management scope				Productive scope[b]			
				Water deprivation [m^3 m^{-2} rotation^{-1}]	Resources [MJ m^{-2} rotation^{-1}]	Ecosystem quality [PDFm^2y m^{-2} rotation^{-1}]	EI99HA single-score [milipoints m^{-2} rotation^{-1}]	Water deprivation [m^3 t^{-1}]	Resources [MJt^{-1}]	Ecosystem quality [PDFm^2y t^{-1}]	EI99HA single-score [points t^{-1}]
35354	Guadiana (n=9)	Guadiana	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	1.7086	0.0000	1.0986	85.71	608.47	0.00	391.24	30.52
			P-P-P	0.8934	0.0000	0.5745	44.82	223.92	0.00	143.98	11.23
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
35355	Tinto, Odiel, Chanza, Piedras (n=2)	Atlantic of Andalucia	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	0.4119	0.0000	0.3938	30.72	139.14	0.00	133.03	10.38
			P-P-P	0.2256	0.0000	0.2157	16.83	56.55	0.00	54.07	4.22
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
35365	Almanzora (n=4)	Mediterranean of Andalucia	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	1.6892	13.3474	0.8024	380.24	570.68	4509.24	271.07	128.46
			P-P-P	0.7500	5.9261	0.3563	168.83	187.97	1485.24	89.29	42.31
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
35693	Guadalquivir (n=11)	Guadalquivir	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	1.6707	4.9752	0.8278	182.94	564.43	1680.82	279.68	61.81
			P-P-P	0.8455	2.5177	0.4189	92.58	211.89	631.00	104.99	23.20
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
35697	Guadalhorce, Guadalmedina (n=4)	Mediterranean of Andalucia	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	1.5759	0.0000	0.3358	26.16	532.41	0.00	113.46	8.84
			P-P-P	0.9000	0.0000	0.1918	14.94	225.56	0.00	48.07	3.74
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
35698	Algarrobo, Sayalonga, Torrox (n=1)	Mediterranean of Andalucia	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	1.3312	2.6757	0.3790	93.19	449.75	903.94	128.04	31.48
			P-P-P	0.9000	1.8089	0.2562	63.00	225.56	453.36	64.22	15.79
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
35700	Guadalfeo (n=2)	Mediterranean of Andalucia	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	1.7147	11.9545	0.9417	358.04	579.31	4038.69	318.15	120.96
			P-P-P	0.9000	6.2744	0.4943	187.92	225.56	1572.54	123.88	47.10
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop rotations	LCIA ğ Blue water							
				Spatial agricultural management scope				Productive scope[b]			
				Water deprivation [m^3 m^{-2} $rotation^{-1}$]	Resources [MJ m^{-2} $rotation^{-1}$]	Ecosystem quality [PDFm^2y m^{-2} $rotation^{-1}$]	EI99HA single-score [milipoints m^{-2} $rotation^{-1}$]	Water deprivation [m^3 t^{-1}]	Resources [MJt^{-1}]	Ecosystem quality [PDFm^2y t^{-1}]	EI99HA single-score [points t^{-1}]
35701	Andarax, Adra (n=4)	Mediterranean of Andalucia	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	1.7804	14.1486	1.2248	432.32	601.48	4779.95	413.79	146.06
			P-P-P	0.9000	7.4520	0.6382	227.16	225.56	1867.66	159.94	56.93
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
36038	Turón, Guadalteba (n=3)	Mediterranean of Andalucia	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	1.7527	0.0000	0.3936	30.67	592.12	0.00	132.99	10.36
			P-P-P	0.9000	0.0000	0.2021	15.75	225.56	0.00	50.66	3.95
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
36039	Guadalete, Barbate (n=2)	Atlantic of Andalucia	B-W-R	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			B-SF-W	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			M-SG-M	1.7478	0.0000	0.1958	15.21	590.49	0.00	66.13	5.14
			P-P-P	0.9000	0.0000	0.1008	7.83	225.56	0.00	25.26	1.96
			B-W-F	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00
			MF-MF-MF	0.0000	0.0000	0.0000	0.00	0.00	0.00	0.00	0.00

B = barley; W = wheat; R = oilseed rape; SF = sunflower; M = maize; SG = sorghum; P = poplar; MF = Mediterranean forest.

[a] Basin code of each watershed according to the WaterGAP 2 global model (Alcamo et al. 2003).

[b] Water consumption allocated to grains or whole plant, depending on the product from each crop: grains for B, W, R, M and SF. Whole biomass for SG, P and MF.

Table 3SI Green water scarcity index for the assessment per m^2 and life cycle impact assessment of green water for the assessment per tonne dry matter of the green water consumption for the 23 studied watersheds.

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop rotations	GWSI and LCIA ğ Green water		
				Spatial agricultural management scope	Productive scope[b]	
				GWSI=GW/Pr' [dimensionless]	Agricultural reference (B-W-F) dGW deprivation [m^3 t^{-1}]	Natural reference (MF-MF-MF) dGW deprivation [m^3 t^{-1}]
31897	Muga-Fluvià (n=4)	Internal watersheds of Catalonia	B-W-R	0.62	-18.78	46.07
			B-SF-W	0.57	-19.52	45.32
			M-SG-M	0.53	-91.92	-27.07
			P-P-P	0.50	-94.42	-29.57
			B-W-F	0.58	0.00	-
			MF-MF-MF	0.78	-	0.00
32250	Tordera (n=2)	Internal watersheds of Catalonia	B-W-R	0.62	-276.80	679.66
			B-SF-W	0.58	-285.54	670.92
			M-SG-M	0.56	-1321.35	-364.89
			P-P-P	0.50	-1368.75	-412.28
			B-W-F	0.59	0.00	-
			MF-MF-MF	0.79	-	0.00
32251	Ter (n=1)	Internal watersheds of Catalonia	B-W-R	0.61	-13.40	33.02
			B-SF-W	0.55	-14.49	31.94
			M-SG-M	0.52	-64.20	-17.78
			P-P-P	0.44	-66.81	-20.39
			B-W-F	0.57	0.00	-
			MF-MF-MF	0.82	-	0.00
32581	Duero, Esla, Pisuerga, Tormes, Orbigo (n=13)	Duero	B-W-R	0.71	-22.29	64.29
			B-SF-W	0.68	-21.18	65.41
			M-SG-M	0.64	-164.88	-77.97
			P-P-P	0.55	-178.88	-91.60
			B-W-F	0.69	0.00	-
			MF-MF-MF	0.90	-	0.00
32603	Besòs-Maresme, Llobregat, Anoia (n=3)	Internal watersheds of Catalonia	B-W-R	0.66	-421.34	977.22
			B-SF-W	0.62	-405.87	992.68
			M-SG-M	0.63	-1889.72	-491.17
			P-P-P	0.56	-1972.64	-574.08
			B-W-F	0.63	0.00	-
			MF-MF-MF	0.84	-	0.00
32963	Ebro, Segre, Cinca, Arga (n=17)	Ebro	B-W-R	0.80	-42.51	207.97
			B-SF-W	0.74	-59.53	190.95
			M-SG-M	0.70	-361.98	-111.50
			P-P-P	0.64	-386.50	-136.02
			B-W-F	0.74	0.00	-
			MF-MF-MF	0.91	-	0.00

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop rotations	GWSI and LCIA ğ Green water		
				Spatial agricultural management scope	Productive scope[b]	
				GWSI=GW/Pr' [dimensionless]	Agricultural reference (B-W-F) dGW deprivation [m^3 t^{-1}]	Natural reference (MF-MF-MF) dGW deprivation [m^3 t^{-1}]
33322	Cenia-Maestrazgo, Mijares-Plana de Castellón (n=2)	Júcar	B-W-R	0.69	-231.79	999.87
			B-SF-W	0.62	-336.70	894.96
			M-SG-M	0.61	-1601.61	-369.94
			P-P-P	0.56	-1766.69	-535.03
			B-W-F	0.64	0.00	-
			MF-MF-MF	0.88	-	0.00
33676	Túria (n=3)	Júcar	B-W-R	0.76	-329.56	1141.81
			B-SF-W	0.70	-386.06	1085.32
			M-SG-M	0.69	-1875.95	-404.58
			P-P-P	0.62	-2065.79	-594.41
			B-W-F	0.72	0.00	-
			MF-MF-MF	0.83	-	0.00
34026	Júcar, Marina Baja, Serpis (n=6)	Júcar	B-W-R	0.80	-254.78	977.08
			B-SF-W	0.74	-270.25	961.61
			M-SG-M	0.68	-1649.07	-417.22
			P-P-P	0.60	-1734.30	-502.44
			B-W-F	0.76	0.00	-
			MF-MF-MF	0.94	-	0.00
34348	Tajo, Alagón, Guadarrama, Guadiela (n=12)	Tajo	B-W-R	0.75	-48.39	326.97
			B-SF-W	0.70	-73.12	302.24
			M-SG-M	0.64	-645.44	-270.08
			P-P-P	0.58	-664.53	-289.17
			B-W-F	0.72	0.00	-
			MF-MF-MF	0.91	-	0.00
34367	Marina Alta (n=1)	Júcar	B-W-R	0.63	-243.09	679.13
			B-SF-W	0.56	-356.92	565.30
			M-SG-M	0.50	-1727.54	-805.32
			P-P-P	0.42	-1892.26	-970.04
			B-W-F	0.57	0.00	-
			MF-MF-MF	0.77	-	0.00
34697	Segura, Guadalentín, Mundo, Mula (n=8)	Segura	B-W-R	0.88	-534.80	1470.77
			B-SF-W	0.84	-493.63	1511.93
			M-SG-M	0.75	-2271.44	-265.88
			P-P-P	0.79	-2336.10	-330.53
			B-W-F	0.85	0.00	-
			MF-MF-MF	0.98	-	0.00
35032	Mar Menor (n=3)	Segura	B-W-R	0.69	-636.97	1208.96
			B-SF-W	0.65	-559.09	1286.84
			M-SG-M	0.63	-2148.53	-302.60
			P-P-P	0.59	-2220.26	-374.33
			B-W-F	0.67	0.00	-
			MF-MF-MF	0.97	-	0.00

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop rotations	GWSI and LCIA ğ Green water		
				Spatial agricultural management scope	Productive scope[b]	
				GWSI=GW/Pr' [dimensionless]	Agricultural reference (B-W-F) dGW deprivation [m³ t⁻¹]	Natural reference (MF-MF-MF) dGW deprivation [m³ t⁻¹]
35354	Guadiana (n=9)	Guadiana	B-W-R	0.76	-65.64	523.79
			B-SF-W	0.71	-158.72	430.71
			M-SG-M	0.63	-1090.08	-500.65
			P-P-P	0.57	-1147.07	-557.65
			B-W-F	0.73	0.00	-
			MF-MF-MF	0.94	-	0.00
35355	Tinto, Odiel, Chanza, Piedras (n=2)	Atlantic of Andalucia	B-W-R	0.61	-24.89	126.15
			B-SF-W	0.56	-71.58	79.46
			M-SG-M	0.52	-354.26	-203.22
			P-P-P	0.40	-368.61	-217.57
			B-W-F	0.59	0.00	-
			MF-MF-MF	0.89	-	0.00
35365	Almanzora (n=4)	Mediterranean of Andalucia	B-W-R	0.84	-151.07	738.24
			B-SF-W	0.80	-254.26	635.05
			M-SG-M	0.79	-1260.22	-370.90
			P-P-P	0.71	-1308.46	-419.14
			B-W-F	0.82	0.00	-
			MF-MF-MF	0.95	-	0.00
35693	Guadalquivir (n=11)	Guadalquivir	B-W-R	0.72	-125.51	604.85
			B-SF-W	0.67	-258.69	471.68
			M-SG-M	0.64	-1273.17	-542.81
			P-P-P	0.55	-1322.78	-592.41
			B-W-F	0.69	0.00	-
			MF-MF-MF	0.94	-	0.00
35697	Guadalhorce, Guadalmedina (n=4)	Mediterranean of Andalucia	B-W-R	0.63	-69.97	314.34
			B-SF-W	0.58	-218.11	166.20
			M-SG-M	0.54	-1075.45	-691.15
			P-P-P	0.42	-1113.06	-728.75
			B-W-F	0.60	0.00	-
			MF-MF-MF	0.88	-	0.00
35698	Algarrobo, Sayalonga, Torrox (n=1)	Mediterranean of Andalucia	B-W-R	0.63	-58.76	440.90
			B-SF-W	0.57	-200.79	298.87
			M-SG-M	0.50	-971.35	-471.69
			P-P-P	0.41	-1009.05	-509.39
			B-W-F	0.59	0.00	-
			MF-MF-MF	0.94	-	0.00
35700	Guadalfeo (n=2)	Mediterranean of Andalucia	B-W-R	0.86	-146.98	680.72
			B-SF-W	0.83	-218.70	609.01
			M-SG-M	0.81	-1065.65	-237.95
			P-P-P	0.80	-1109.33	-281.63
			B-W-F	0.84	0.00	-
			MF-MF-MF	0.98	-	0.00

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop rotations	GWSI and LCIA ğ Green water		
				Spatial agricultural management scope	Productive scope[b]	
				GWSI=GW/Pr' [dimensionless]	Agricultural reference (B-W-F) dGW deprivation [$m^3 t^{-1}$]	Natural reference (MF-MF-MF) dGW deprivation [$m^3 t^{-1}$]
35701	Andarax, Adra (n=4)	Mediterranean of Andalucia	B-W-R	0.87	-132.81	607.83
			B-SF-W	0.84	-196.80	543.84
			M-SG-M	0.82	-967.85	-227.21
			P-P-P	0.81	-1005.87	-265.23
			B-W-F	0.85	0.00	-
			MF-MF-MF	0.99	-	0.00
36038	Turón, Guadalteba (n=3)	Mediterranean of Andalucia	B-W-R	0.73	-101.66	454.86
			B-SF-W	0.68	-244.14	312.38
			M-SG-M	0.65	-1236.81	-680.29
			P-P-P	0.54	-1277.29	-720.77
			B-W-F	0.70	0.00	-
			MF-MF-MF	0.93	-	0.00
36039	Guadalete, Barbate (n=2)	Atlantic of Andalucia	B-W-R	0.65	-109.71	430.80
			B-SF-W	0.59	-274.09	266.42
			M-SG-M	0.56	-1292.19	-751.68
			P-P-P	0.43	-1347.26	-806.75
			B-W-F	0.62	0.00	-
			MF-MF-MF	0.93	-	0.00

B = barley; W = wheat; R = oilseed rape; SF = sunflower; M = maize; SG = sorghum; P = poplar; MF = Mediterranean forest.

[a] Basin code of each watershed according to the WaterGAP 2 global model (Alcamo et al. 2003).

[b] Water consumption allocated to grains or whole plant, depending on the product from each crop: grains for B, W, R, M and SF. Whole biomass for SG, P and MF.

Table 4SI Land use assessment per tonne dry matter for the 23 studied watersheds.

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop rotations	Harvested dry matter[b] [t ha^{-1}]	LCIA ğ Land use	
					Ecosystem quality[c] [PDFm^{2}y t^{-1}]	I99HA single-score [points t^{-1}]
31897	Muga-Fluvià (n=4)	Internal watersheds of Catalonia	B-W-R	4.67	7387.58	576.03
			B-SF-W	4.28	8060.75	628.52
			M-SG-M	26.64	1295.05	100.98
			P-P-P	39.90	864.66	67.42
			B-W-F	3.39	10176.99	793.53
			MF-MF-MF	16.95	194.69	15.18
32250	Tordera (n=2)	Internal watersheds of Catalonia	B-W-R	4.67	7387.58	576.03
			B-SF-W	4.28	8060.75	628.52
			M-SG-M	26.64	1295.05	100.98
			P-P-P	39.90	864.66	67.42
			B-W-F	3.39	10176.99	793.53
			MF-MF-MF	16.95	194.69	15.18
32251	Ter (n=1)	Internal watersheds of Catalonia	B-W-R	4.67	7387.58	576.03
			B-SF-W	4.28	8060.75	628.52
			M-SG-M	26.64	1295.05	100.98
			P-P-P	39.90	864.66	67.42
			B-W-F	3.39	10176.99	793.53
			MF-MF-MF	16.95	194.69	15.18
32581	Duero, Esla, Pisuerga, Tormes, Orbigo (n=13)	Duero	B-W-R	7.12	4845.51	377.82
			B-SF-W	6.66	5180.18	403.91
			M-SG-M	26.62	1296.02	101.05
			P-P-P	39.90	864.66	67.42
			B-W-F	5.84	5907.53	460.63
			MF-MF-MF	16.95	194.69	15.18
32603	Besòs-Maresme, Llobregat, Anoia (n=3)	Internal watersheds of Catalonia	B-W-R	4.67	7387.58	576.03
			B-SF-W	4.28	8060.75	628.52
			M-SG-M	26.64	1295.05	100.98
			P-P-P	39.90	864.66	67.42
			B-W-F	3.39	10176.99	793.53
			MF-MF-MF	16.95	194.69	15.18
32963	Ebro, Segre, Cinca, Arga (n=17)	Ebro	B-W-R	5.77	5979.20	466.21
			B-SF-W	5.57	6193.90	482.95
			M-SG-M	25.24	1366.88	106.58
			P-P-P	39.90	864.66	67.42
			B-W-F	4.69	7356.08	573.57
			MF-MF-MF	16.95	194.69	15.18

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop rotations	Harvested dry matter[b] [t ha^{-1}]	LCIA ğ Land use	
					Ecosystem quality[c] [PDFm^{2}y t^{-1}]	I99HA single-score [points t^{-1}]
33322	Cenia-Maestrazgo, Mijares-Plana de Castellón (n=2)	Júcar	B-W-R	4.79	7202.51	561.60
			B-SF-W	4.51	7649.67	596.47
			M-SG-M	19.58	1762.00	137.39
			P-P-P	39.90	864.66	67.42
			B-W-F	3.71	9299.19	725.08
			MF-MF-MF	16.95	194.69	15.18
33676	Túria (n=3)	Júcar	B-W-R	4.79	7202.51	561.60
			B-SF-W	4.51	7649.67	596.47
			M-SG-M	19.58	1762.00	137.39
			P-P-P	39.90	864.66	67.42
			B-W-F	3.71	9299.19	725.08
			MF-MF-MF	16.95	194.69	15.18
34026	Júcar, Marina Baja, Serpis (n=6)	Júcar	B-W-R	4.94	6938.81	544.55
			B-SF-W	4.64	8476.66	660.95
			M-SG-M	25.58	1348.71	105.16
			P-P-P	39.90	864.66	67.42
			B-W-F	3.96	8712.12	679.31
			MF-MF-MF	16.95	194.69	15.18
34348	Tajo, Alagón, Guadarrama, Guadiela (n=12)	Tajo	B-W-R	5.72	6031.47	470.29
			B-SF-W	5.48	6295.62	490.89
			M-SG-M	30.56	1128.93	88.03
			P-P-P	39.90	864.66	67.42
			B-W-F	4.92	7012.20	546.76
			MF-MF-MF	16.95	194.69	15.18
34367	Marina Alta (n=1)	Júcar	B-W-R	4.79	7202.51	561.60
			B-SF-W	4.51	7649.67	596.47
			M-SG-M	19.58	1762.00	137.39
			P-P-P	39.90	864.66	67.42
			B-W-F	3.71	9299.19	725.08
			MF-MF-MF	16.95	194.69	15.18
34697	Segura, Guadalentín, Mundo, Mula (n=8)	Segura	B-W-R	3.41	10129.18	789.80
			B-SF-W	3.18	10883.28	848.60
			M-SG-M	24.82	1390.12	108.39
			P-P-P	39.90	864.66	67.42
			B-W-F	2.53	13668.78	1065.79
			MF-MF-MF	16.95	194.69	15.18
35032	Mar Menor (n=3)	Segura	B-W-R	2.84	12147.89	947.20
			B-SF-W	2.60	13269.23	1034.64
			M-SG-M	23.50	1468.09	114.47
			P-P-P	39.90	864.66	67.42
			B-W-F	1.97	17512.69	1365.51
			MF-MF-MF	16.95	194.69	15.18

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop rotations	Harvested dry matter[b] [t ha^{-1}]	LCIA ğ Land use	
					Ecosystem quality[c] [PDFm^{2}y t^{-1}]	I99HA single-score [points t^{-1}]
35354	Guadiana (n=9)	Guadiana	B-W-R	6.20	5564.52	433.88
			B-SF-W	6.19	5573.51	434.58
			M-SG-M	28.08	1228.63	95.80
			P-P-P	39.90	864.66	67.42
			B-W-F	5.39	6400.74	499.08
			MF-MF-MF	16.95	194.69	15.18
35355	Tinto, Odiel, Chanza, Piedras (n=2)	Atlantic of Andalucia	B-W-R	5.52	6250.00	487.33
			B-SF-W	5.67	6084.66	474.44
			M-SG-M	29.60	1165.54	90.88
			P-P-P	39.90	864.66	67.42
			B-W-F	4.65	7419.35	578.51
			MF-MF-MF	16.95	194.69	15.18
35365	Almanzora (n=4)	Mediterranean of Andalucia	B-W-R	5.52	6250.00	487.33
			B-SF-W	5.67	6084.66	474.44
			M-SG-M	29.60	1165.54	90.88
			P-P-P	39.90	864.66	67.42
			B-W-F	4.65	7419.35	578.51
			MF-MF-MF	16.95	194.69	15.18
35693	Guadalquivir (n=11)	Guadalquivir	B-W-R	5.52	6250.00	487.33
			B-SF-W	5.67	6084.66	474.44
			M-SG-M	29.60	1165.54	90.88
			P-P-P	39.90	864.66	67.42
			B-W-F	4.65	7419.35	578.51
			MF-MF-MF	16.95	194.69	15.18
35697	Guadalhorce, Guadalmedina (n=4)	Mediterranean of Andalucia	B-W-R	5.52	6250.00	487.33
			B-SF-W	5.67	6084.66	474.44
			M-SG-M	29.60	1165.54	90.88
			P-P-P	39.90	864.66	67.42
			B-W-F	4.65	7419.35	578.51
			MF-MF-MF	16.95	194.69	15.18
35698	Algarrobo, Sayalonga, Torrox (n=1)	Mediterranean of Andalucia	B-W-R	5.52	6250.00	487.33
			B-SF-W	5.67	6084.66	474.44
			M-SG-M	29.60	1165.54	90.88
			P-P-P	39.90	864.66	67.42
			B-W-F	4.65	7419.35	578.51
			MF-MF-MF	16.95	194.69	15.18
35700	Guadalfeo (n=2)	Mediterranean of Andalucia	B-W-R	5.52	6250.00	487.33
			B-SF-W	5.67	6084.66	474.44
			M-SG-M	29.60	1165.54	90.88
			P-P-P	39.90	864.66	67.42
			B-W-F	4.65	7419.35	578.51
			MF-MF-MF	16.95	194.69	15.18

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop rotations	Harvested dry matter[b] [t ha^{-1}]	LCIA ğ Land use	
					Ecosystem quality[c] [PDFm^{2}y t^{-1}]	I99HA single-score [points t^{-1}]
35701	Andarax, Adra (n=4)	Mediterranean of Andalucia	B-W-R	5.52	6250.00	487.33
			B-SF-W	5.67	6084.66	474.44
			M-SG-M	29.60	1165.54	90.88
			P-P-P	39.90	864.66	67.42
			B-W-F	4.65	7419.35	578.51
			MF-MF-MF	16.95	194.69	15.18
36038	Turón, Guadalteba (n=3)	Mediterranean of Andalucia	B-W-R	5.52	6250.00	487.33
			B-SF-W	5.67	6084.66	474.44
			M-SG-M	29.60	1165.54	90.88
			P-P-P	39.90	864.66	67.42
			B-W-F	4.65	7419.35	578.51
			MF-MF-MF	16.95	194.69	15.18
36039	Guadalete, Barbate (n=2)	Atlantic of Andalucia	B-W-R	5.52	6250.00	487.33
			B-SF-W	5.67	6084.66	474.44
			M-SG-M	29.60	1165.54	90.88
			P-P-P	39.90	864.66	67.42
			B-W-F	4.65	7419.35	578.51
			MF-MF-MF	16.95	194.69	15.18

B = barley; W = wheat; R = oilseed rape; SF = sunflower; M = maize; SG = sorghum; P = poplar; MF = Mediterranean forest.

[a] Basin code of each watershed according to the WaterGAP 2 global model (Alcamo et al. 2003).

[b] Grains for B, W, R, M and SF. Whole biomass for SG, P and MF.

[c] Land use allocated to grains or whole plant, depending on the product from each crop: see footnote “b”.

Table 5SI Harvested dry matter per crop for the 23 studied watersheds.

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop	Harvested grains[b] [t ha^{-1} y^{-1}]
31897	Muga-Fluvià (n=4)	Internal watersheds of Catalonia	Barley	1.20
			Wheat	2.19
			Oilseed rape	1.28
			Sunflower	0.89
			Maize	9.57
32250	Tordera (n=2)	Internal watersheds of Catalonia	Barley	1.20
			Wheat	2.19
			Oilseed rape	1.28
			Sunflower	0.89
			Maize	9.57
32251	Ter (n=1)	Internal watersheds of Catalonia	Barley	1.20
			Wheat	2.19
			Oilseed rape	1.28
			Sunflower	0.89
			Maize	9.57
32581	Duero, Esla, Pisuerga, Tormes, Orbigo (n=13)	Duero	Barley	3.29
			Wheat	2.55
			Oilseed rape	1.28
			Sunflower	0.82
			Maize	9.56
32603	Besòs-Maresme, Llobregat, Anoia (n=3)	Internal watersheds of Catalonia	Barley	1.20
			Wheat	2.19
			Oilseed rape	1.28
			Sunflower	0.89
			Maize	9.57
32963	Ebro, Segre, Cinca, Arga (n=17)	Ebro	Barley	2.83
			Wheat	1.86
			Oilseed rape	1.08
			Sunflower	0.88
			Maize	8.87
33322	Cenia-Maestrazgo, Mijares-Plana de Castellón (n=2)	Júcar	Barley	2.20
			Wheat	1.51
			Oilseed rape	1.08
			Sunflower	0.80
			Maize	6.04
33676	Túria (n=3)	Júcar	Barley	2.20
			Wheat	1.51
			Oilseed rape	1.08
			Sunflower	0.80
			Maize	6.04
34026	Júcar, Marina Baja, Serpis (n=6)	Júcar	Barley	2.41
			Wheat	1.55
			Oilseed rape	0.98
			Sunflower	0.68
			Maize	9.04
34348	Tajo, Alagón, Guadarrama, Guadiela (n=12)	Tajo	Barley	2.72
			Wheat	2.20
			Oilseed rape	0.80
			Sunflower	0.56
			Maize	11.53
34367	Marina Alta (n=1)	Júcar	Barley	2.20
			Wheat	1.51
			Oilseed rape	1.08
			Sunflower	0.80
			Maize	6.04
34697	Segura, Guadalentín, Mundo, Mula (n=8)	Segura	Barley	1.51
			Wheat	1.02
			Oilseed rape	0.88
			Sunflower	0.65
			Maize	8.66
35032	Mar Menor (n=3)	Segura	Barley	1.18
			Wheat	0.79
			Oilseed rape	0.87
			Sunflower	0.63
			Maize	8.00

Basin-ID[a]	Basin name (number of plots studied within the watershed)	Belonging to	Crop	Harvested grains[b] [t ha^{-1} y^{-1}]
35354	Guadiana (n=9)	Guadiana	Barley	2.61
			Wheat	2.78
			Oilseed rape	0.81
			Sunflower	0.80
			Maize	10.29
35355	Tinto, Odiel, Chanza, Piedras (n=2)	Atlantic of Andalucia	Barley	2.37
			Wheat	2.28
			Oilseed rape	0.87
			Sunflower	1.02
			Maize	11.05
35365	Almanzora (n=4)	Mediterranean of Andalucia	Barley	2.37
			Wheat	2.28
			Oilseed rape	0.87
			Sunflower	1.02
			Maize	11.05
35693	Guadalquivir (n=11)	Guadalquivir	Barley	2.37
			Wheat	2.28
			Oilseed rape	0.87
			Sunflower	1.02
			Maize	11.05
35697	Guadalhorce, Guadalmedina (n=4)	Mediterranean of Andalucia	Barley	2.37
			Wheat	2.28
			Oilseed rape	0.87
			Sunflower	1.02
			Maize	11.05
35698	Algarrobo, Sayalonga, Torrox (n=1)	Mediterranean of Andalucia	Barley	2.37
			Wheat	2.28
			Oilseed rape	0.87
			Sunflower	1.02
			Maize	11.05
35700	Guadalfeo (n=2)	Mediterranean of Andalucia	Barley	2.37
			Wheat	2.28
			Oilseed rape	0.87
			Sunflower	1.02
			Maize	11.05
35701	Andarax, Adra (n=4)	Mediterranean of Andalucia	Barley	2.37
			Wheat	2.28
			Oilseed rape	0.87
			Sunflower	1.02
			Maize	11.05
36038	Turón, Guadalteba (n=3)	Mediterranean of Andalucia	Barley	2.37
			Wheat	2.28
			Oilseed rape	0.87
			Sunflower	1.02
			Maize	11.05
36039	Guadalete, Barbate (n=2)	Atlantic of Andalucia	Barley	2.37
			Wheat	2.28
			Oilseed rape	0.87
			Sunflower	1.02
			Maize	11.05

[a] Basin code of each watershed according to the WaterGAP 2 global model (Alcamo et al. 2003).
[b] Assumed harvested whole biomass of sorghum: 7.50 t ha^{-1} $year^{-1}$. Assumed harvested whole biomass of poplar: 13.3 t ha^{-1} $year^{-1}$. Assumed aboveground net primary production of the Mediterranean forest: 5.65 t ha^{-1} $year^{-1}$.

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Chapter 3.

Assessing potential desertification environmental impact in life cycle assessment. Methodological aspects.

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Abstract

Background, aim, and scope LCA enables the objective assessment of global environmental burdens associated with the life cycle of a product or a production system. One of the main weaknesses of LCA is that, as yet, there is no scientific agreement on the assessment methods for land-use related impacts, which results in either the exclusion or the lack of assessment of local environmental impacts related to land use. The inclusion of the desertification impact in LCA studies of any human activity can be important in high-desertification risk regions.

Main features This paper focuses on the development of a methodology for including the desertification environmental impact derived from land use in LCA studies. A set of variables to be measured in the LCI, their characterisation factors (CFs), and an impact assessment method for the LCIA phase are suggested. The CFs were acquired using GIS.

Results For the LCI stage it is necessary to register information on: i) the four biophysical variables of aridity, erosion, aquifer overexploitation and fire risk, with a created scale of values; ii) the geographical location of the activity, and iii) the spatial and temporal extension of the activity. For the CFs, the four LCI biophysical variables in i) were measured for the main terrestrial natural regions (ecoregions) by means of GIS.

Discussion Using GIS, calculation of the CF for the aridity variable shows that 38% of the world area, in 8 out of 15 existing ecoregions, is at risk of desertification. The most affected is the tropical/subtropical desert. The LCIA model has been developed to identify scenarios without desertification impact.

Conclusions The developed method makes possible the inclusion of the desertification impact derived from land use in LCA studies, using data generally available to LCA users.

Recommendations and perspectives While this LCIA model may be a simplified approach, it can be calibrated and improved for different case studies. The model proposed is suitable for assessing the desertification impact of any type of human activity and may be complemented with specific activity indicators, and although we have considered biophysical factors, the method can be extended to socio-economic vectors.

Keywords Aridity index • Characterisation factors • Desertification • Geographical information system (GIS) • Land use impacts • Life cycle assessment (LCA) • Life cycle impact assessment (LCIA) • Life cycle inventory (LCI)

3.1 Introduction

Life cycle assessment methodology was initially developed for environmental assessments of industrial systems. It was later adapted to agricultural systems, where its use has gradually spread. Traditionally, LCA studies take a general approach that is spatially and temporally independent of the environmental impacts derived from a product or production system (ISO 14040 2006; ISO 14044 2006). However, as agricultural systems are closely related to local and temporal aspects, especially water consumption and land use, adjustments of the LCA methodology to take land use impacts into consideration are the subject of study (Guinée et al. 2006) for both the LCI and the LCIA.

Today it is acknowledged that land use should be assessed by LCA, but there is still no consensus on the parameters to consider and the methodology to follow, mainly due to the lack of available data (Cowell and Lindeijer 2000). Research has been carried out by a number of authors to identify the possible problems and propose solutions to address land use in LCA (Audsley 1997; Goedkoop et al. 2009; Koellner and Scholz 2007; Milà i Canals et al. 2007; Wegener Sleeswijk et al. 1996; Weidema and Meeusen 2000).

Apart from the area of land, land use quality and soil disturbance by the activity being developed have to be considered (Guinée et al. 2002; Mattsson et al. 2000; Milà i Canals et al. 2007).

Due to the complexity of the different factors affecting land use quality, most proposals have suggested distinguishing between different ecosystems (Heijungs et al. 1992; Steen and Ryding 1993) and using multiple indicators (Blonk et al. 1997; Cowell and Clift 2000; Mattsson et al. 2000).

3.1.1 Desertification as a life cycle assessment land use impact category

To date, no attempts to include desertification impact in LCA studies have been published, even though this is one of the main problems for sustainability in arid, semi-arid and dry sub-humid areas, especially in developing countries. The United Nations Convention to Combat Desertification (UNCCD) states that arid, semi-arid, and dry sub-humid areas include areas other than polar and sub-polar regions, in which the ratio of annual precipitation to potential evapotranspiration lies between 0.05 and 0.65 (United Nations 1994).

Irreversible soil degradation due to desertification is a concern in arid areas worldwide, as such, it is important to include desertification impact in LCA studies in these areas (Civit 2009). In order to assess such land use impact, it is first necessary to define the variables and to gather quality information about them in the LCI. Other elementary flows required in the LCI are the spatial and temporal extent, and the geographical location. Once the inventory data is gathered, the LCI results have to be characterised in the impact assessment phase.

At present, the international community has not agreed on the methodology that should be followed to select desertification indicators. The main obstacle is that not all indicators are suitable at all scales. Two main factors for selecting the appropriate scale to measure an indicator are the availability of data sets for the area over a sufficient length of time, and the possibility of using remote-sensing technologies to obtain information (DESERTLINKS 2004).

Desertification indicators are arranged according to the three dimensions of sustainable development: environmental (or biophysical), economic and social (MAM 2006). Most of the studies on desertification assessment (e.g., DESERTLINKS, DesertNet, DISMED, LADA, MEDALUS) have focused on the biophysical dimension of desertification, using several variables to measure it. Thus, desertification assessment using a multi-indicator approach appears to be an appropriate method for evaluating this land use impact.

3.1.2 Life cycle assessment and geographical information systems

Nowadays, most desertification indicators included in national action programmes and in studies to combat desertification are directly obtained from digitalized maps or by using GIS.

Despite the widespread employment of GIS, little use is made of combinations of GIS data with LCIA to obtain land use indicators. Examples of using GIS and LCA focus on handling the information acquired with GIS methods to produce site-specific information of the environmental effects of a product or production system, which allows for more accurate LCA (Bengtsson et al. 1998; Jäppinen et al. 2008).

Given the benefits of GIS to provide land information, and the weakness of LCA, lacking impact categories related to land use, the integration of GIS with LCA is a good tool to define site-dependent characterisation factors (CFs). This improves LCA adaptability to land use impacts, not only for desertification but also for other land use impacts such as biodiversity, water consumption, and erosion.

The aim of this research was to develop a methodology for including potential desertification from land use as an environmental impact in LCA studies. This study provides LCI data, CFs obtained by GIS analysis, and a LCIA model.

3.2 Life cycle inventory modelling

LCI considers the total consumptions and emissions of a system and quantifies them according to the functional unit established.

Inputs to the model proposed must reflect the causes of land degradation, specifically those that affect arid lands, i.e., the causes of desertification. The choice was carried out by overlapping indicators at the local, national, and global level, and selecting those indicators applicable within the LCA methodology. Only physical factors, belonging to the state and pressure framework, were taken into account, due to the disagreement related to social and economic vectors. The four selected physical variables state the desertification impact due to the different human activities that can occupy a portion of land during a certain period of time. The selected variables were: aridity index, erosion, aquifer overexploitation and fire risk. After an extensive review of action programmes to combat desertification from several countries and a number of international studies focused on assessment of desertification indicators, we concluded that these four variables adequately embrace the main factors that cause desertification. The four variables may be complemented with those specific for different kinds of human activity (e.g., salinity for agricultural activities, soil crusting for building activities). For LCA practitioners wishing to include a specific variable, because it is a basic indicator in a certain region under study, the same procedure must be followed as with the other variables.

Table 8 shows the variables selected for inclusion in the LCI and the possible values they can take. Each one must first be quantified then qualified following the scales of values proposed in sections 3.2.1 to 3.2.4. The proposed values were based on the reviewed desertification programmes and desertification studies.

Once the individual value for each variable has been assigned, the LCI value ($LCI_{Desertification}$) can be calculated as shown in equation 4:

$$\begin{cases} \text{If } V_{Aridity} \leq 0 \quad LCI_{Desertification} = 0 \\ \text{If } V_{Aridity} > 0 \quad LCI_{Desertification} = V_{Aridity} + V_{Erosion} + V_{Aquifer\ overexploitation} + V_{Fire\ risk} \end{cases} \quad (4)$$

where $LCI_{Desertification}$ is the desertification index for the LCI phase, and $V_{Aridity}$, $V_{Erosion}$, $V_{Aquifer\ overexploitation}$ and $V_{Fire\ risk}$ are the individual values for each variable. The sum of the individual values for the four variables was used to express the desertification impact caused by an activity, following the United Nations definition of desertification (United Nations 1994), even though other mathematical formulae may also be appropriate. As a general rule, the higher the $LCI_{Desertification}$ value, the greater the desertification impact, however, section 3.2.1 reports an exception to this generalisation. This LCI framework allows for easy comparisons of the desertification impact of several activities under study.

As table 8 shows, the range between the higher and the lower values of each variable differs. While $V_{Aquifer\ overexploitation}$ and $V_{Fire\ risk}$ range from 1 to 2, the range for $V_{Aridity}$ is from 0 to 3 and $V_{Erosion}$ from 1 to 3. Only $V_{Aridity}$ can take a value of 0. A different weighting was assigned to the conditions considered in each variable, depending on their importance in determining desertification. Both $V_{Aquifer\ overexploitation}$ and $V_{Fire\ risk}$ have a lower weighting than $V_{Aridity}$ and $V_{Erosion}$. The aridity variable has a higher weighting as it is the criterion used by the United Nations (1994) to identify those zones where desertification could occur. The high weighting allocated to soil erosion is due to its major impact at a global level. In addition, both the aridity index and soil erosion are the two basic indicators in all national action programmes to combat desertification (DESERTLINKS 2004). Following the criterion established in the Spanish Desertification National Action Programme (DNAP-Spain), four desertification impact categories were distinguished in agreement with the $LCI_{Desertification}$ value: low ($LCI_{Desertification}$ from 4 to 5), medium ($LCI_{Desertification}$ from 5 to 6), high ($LCI_{Desertification}$ from 6 to 7), and very high ($LCI_{Desertification}$ from 7 to 10).

Apart from the individual values for each variable ($V_{Aridity}$, $V_{Erosion}$, $V_{Aquifer\ overexploitation}$ and $V_{Fire\ risk}$), to assess this impact of land use it is necessary to register the geographical location, and spatial and temporal extension of the activity in the LCI.

Table 8 Proposal for the estimation and evaluation of desertification variables for the inventory phase ($LCI_{Desertification}$) (dimensionless).

Desertification variables	
Estimation value	**Evaluation (LCI variable data, dimensionless).**
Aridity variable ($V_{Aridity}$)[a]	
Arid (0.05–0.20)	3
Semi-arid (0.20–0.50)	2
Dry sub-humid (0.50–0.65)	1
Humid sub-humid (0.65–0.75)	0
Humid (>0.75)	0
Erosion variable ($V_{Erosion}$)	
>25 t ha^{-1} year^{-1}	3
12–25 t ha^{-1} year^{-1}	2
<12 t ha^{-1} year^{-1}	1
Aquifer overexploitation variable ($V_{Aquifer\ overexploitation}$)	
W[b]>0.8R[c]	2
0.8R≥W>0.4R	1.6
0.4R≥W>0.2R	1.3
W≤0.2R	1
Fire risk variable ($V_{Fire\ risk}$)	
≥10% burned area in previous 10 years	2
<10% burned area in previous 10 years	1

[a] Ratio between precipitation (Pr) and evapotranspiration (ET_0) (Pr/ET_0), dimensionless
[b] Withdrawal
[c] Recharge

3.2.1 Aridity variable

The aridity factor was calculated considering the climatic surface map given by the ratio Pr/ET_0, where Pr is the precipitation and ET_0 the potential evapotranspiration. General criteria to characterise each geographic area are shown in table 8.

To assign values to the aridity variable ($V_{Aridity}$), the regions described within the United Nations definition of desertification (United Nations 1994) were considered. According to the UNCCD, humid sub-humid and humid areas are not at risk of desertification. The numerical values assigned were: arid regions, 3; semi-arid regions, 2; dry sub-humid regions, 1; and humid sub-humid and humid regions, 0. These numerical values were adapted from the DNAP-Spain proposal. This action programme gives arid regions a value of 2, semi-arid regions a value of 1, and the remainder (dry sub-humid, humid sub-humid and humid regions) a value of 0. However, it seems more appropriate to increase the value of all climates with desertification risk, from dry sub-humid to arid regions, by one unit. Regardless of the $V_{Erosion}$, $V_{Aquifer\ overexploitation}$ and $V_{Fire\ risk}$

values, the desertification impact in a portion of land exists only if $V_{Aridity}$ is not equal to 0, as shown in equation 4.

3.2.2 Erosion variable

Erosion is one of the main reasons for soil degradation and desertification. The LCI data for the erosion variable ($V_{Erosion}$) only requires an estimate of water erosion for the study area. Wind erosion is not included in the measurement because, by comparison, water erosion causes greater soil losses on a world scale (Oldeman et al. 1990; Reich et al. 2001). Water erosion is a biophysical indicator usually built within the national action programmes to combat desertification. The universal soil loss equation (USLE) (Wischmeier and Smith 1978) was used for the assessment, as it is the quantitative model of soil loss evaluation with the greatest agreement on an international level and widely applied (Boellstorff and Benito 2005; Nelson 2002; van der Knijff et al. 2000). The USLE predicts the average annual water erosion rate in the long-term on a field slope based on rainfall pattern, soil type, topography, crop system, and management practices. In this study, the five erosion intensity categories proposed by Stone (2000) were reduced to three: category 1 (<12 t ha^{-1} y^{-1}) combines the <7.5 and 7.5–12.5 t ha^{-1} y^{-1} categories, category 2 (12–25 t ha^{-1} y^{-1}) corresponds to the 12.5–25.5 t ha^{-1} y^{-1} category, and category 3 (>25 t ha^{-1} y^{-1}) combines the 25.5–37 and >37 t ha^{-1} y^{-1} categories. These three categories were given the numerical values of 1, 2, and 3 respectively, following the DNAP-Spain criteria. The established thresholds fit well within those adopted by DNAP-Spain and are similar to those suggested by other authors: Basic et al. (2004) considers 6 categories, from <2 t ha^{-1} y^{-1} (insignificant erosion) to >40.01 t ha^{-1} y^{-1} (disastrous erosion) and Kirkby et al. (2004) distinguishes 8 erosion limits, from ≤0.5 t ha^{-1} y^{-1} to >50 t ha^{-1} y^{-1}.

3.2.3 Aquifer overexploitation variable

Overexploitation may be defined as the situation in which, over a period of years, the average aquifer abstraction rate is greater than, or close to, the average recharge rate (RDPH 1986). The LCI data estimate for the aquifer overexploitation variable ($V_{Aquifer\ overexploitation}$) needs to take into account the hydrological balance of the aquifers located in the area under study. The variable can take 4 different values, between 1 and 2, depending on the degree of aquifer exploitation, calculated as withdrawal divided by recharge (water exploitation ratio). The aquifer exploitation thresholds established in this study are based on those suggested by Alcamo et al. (2000). For this author, an exploitation ratio above 0.8 indicates very high stress ($V_{\text{Aquifer overexploitation}}$ is 2); a ratio between 0.4 and 0.8 represents high stress ($V_{\text{Aquifer overexploitation}}$ is 1.6); a ratio between 0.2 and 0.4 indicates medium stress ($V_{\text{Aquifer overexploitation}}$ is 1.3); a ratio between 0.1 and 0.2 means low stress; and a ratio below 0.1 shows no stress ($V_{\text{Aquifer overexploitation}}$ is 1 in these last two cases). Currently, there is no objective basis for selecting a threshold for overexploitation ratio. The DNAP-Spain considers that aquifer overexploitation only takes place when water withdrawal is equal or higher than 0.8 times the recharge. However, other authors consider this exploitation rate shows high-stressed water resources. For this reason, the criterion of the DNAP-Spain was rejected in favour of more universal and conservative criteria for water requirements of ecosystems.

3.2.4 Fire risk variable

Forest fires, recognised as a cause of desertification (MAM 2005), are one of the main factors that influence the structure and function of terrestrial ecosystems all over the world. The LCI data for the fire risk variable ($V_{\text{Fire risk}}$) was obtained by quantifying the accumulated percentage of surface affected by forest fires during the last ten years in the geographical area under study. If these data are not available, statistical data over a period of ten years may be used. The selected geographical area must be equivalent to a regional administrative division (e.g., *departamentos*, *comarcas* in Spain and Argentina). The choice of the ten-year period for the fire statistics was based on the monitoring period of the Forest Resources Assessment 2005 (1988–1992 and 1998–2002) (FAO 2006). Fire intensity was classified into two groups, following the DNAP-Spain criterion: <10% of affected area and ≥10% of affected area. According to this criterion, when the area affected is <10%, the fire risk variable takes a value of 1, and when it is ≥10%, the variable takes a value of 2. The 10% threshold is also considered appropriate for regions outside Spain because this value was the result of a consensus agreed in an action programme to combat desertification in a Mediterranean region. Here, many studies on the consequences of fires have been carried out, as a consequence of high fire frequency and its derived economic and social problems.

3.3 Development of the characterisation factors for natural areas

As CFs of the LCIA phase for the four variables estimated in the LCI have been established for the large natural areas of the world, soil quality was taken into account in land use assessment, following the proposals by other authors (Heijungs et al. 1992; Steen and Ryding 1993). CFs for large natural areas that incorporate biodiversity impacts in LCA have also been developed (Cowell and Lindeijer 2000; Koellner 2000; Schmidt 2008), but no inclusion of desertification impacts has previously been published. One of the main contributions of this study is the establishment of desertification impact CFs for the large natural areas of land. The divisions between these areas are based on climatic and vegetative cover factors, both aspects having a major influence on soil desertification risk.

3.3.1 Choice of an ecosystem classification

The assessment of any impact category in LCA requires CFs that are unique on a global scale. To satisfy this premise, the classification of natural systems that is used in the LCIA must comply with the following criteria: i) it must be applicable worldwide, ii) it needs to be accepted by the scientific community and widely used, iii) the data must be available worldwide, and iv) a relationship between each natural system category and its desertification risk must be shown. Additionally, it is preferable that the classification is available in digital format, to enable work with GIS.

Many authors have developed classification systems of natural areas. For example, Begon et al. (1999) define 12 biomes, Folch et al. (1984) distinguish 12 physiognomic domains, Olson et al. (1983) identify 44 land ecosystem classes, and Bailey (1996, 1998) describes 15 ecosystem regions (or ecoregions). These authors all comply with the four requirements mentioned above, but the hierarchical classification of Bailey's ecoregions was used here as it is available in a GIS compatible format, while the other three have poor quality digitalized public maps. The lack of geo-referenced maps makes it difficult to determine the CFs in the analyses, and to combine the natural areas layer with other information layers (aridity index, erosion risk, aquifer overexploitation and fire risk).

Based on macroclimate conditions and the prevailing plant formations determined by these conditions, Bailey (1996, 1998) subdivided the continents into ecoregions with three levels of detail: domains (macroecosystems), divisions and climate subtypes, provinces or sites (microecosystems) (Bailey 2002). Table 6SI in the supporting information lists climate, vegetation, and surface area associated with each of the 4 domains and 15 divisions.

3.3.2 Calculation of the characterisation factors

The CFs must be calculated for each LCI variable and for each ecoregion. The assessment methodology and the possible values follow the same procedure used to obtain the values of the variables in the LCI (see sections 3.2.1 to 3.2.4). The CFs were calculated using GIS. The configuration of each layer (one for each variable) was based on the collection of maps and statistical data from several information sources:

$CF^{i}_{Aridity}$ (characterisation factor for the aridity variable for each ecoregion, i) was based on the global aridity index map of the Global Agro-Ecological Zoning 2000 from FAO and IIASA (Fischer et al. 2000).

$CF^{i}_{Erosion}$ was derived from the world map of the global assessment of human-induced soil degradation (GLASOD) (ISRIC 2008). In GLASOD, a total of 12 soil degradation types are recognised and mapped, of which two types are related to water erosion: loss of topsoil (Wt) and terrain definition/mass movement (Wd). Both were accounted for in each ecoregion in order to obtain the $CF^{i}_{Erosion}$.

$CF^{i}_{Aquifer\ overexploitation}$ was determined using statistical data on the recharge and withdrawal aquifer exploitation rates per country published by EEA (1999), EUWI (2007), FAO (2007a), UNEP (2002) and WRI (2007). These five groups collected their data for years between 1960 and 2007. However, the years covered for statistical analysis are variable and typically not available for a time series.

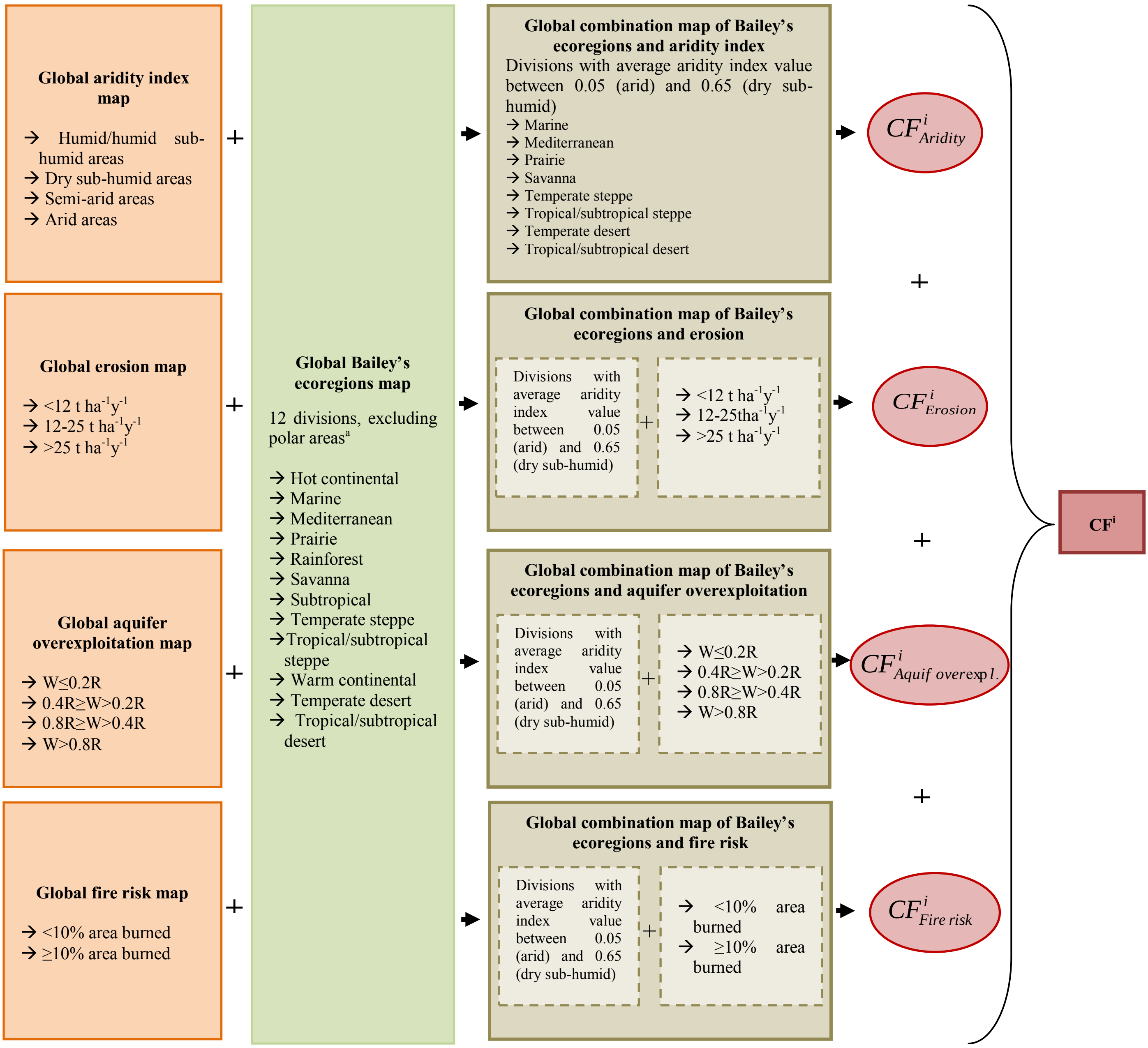
$CF^{i}_{Fire\ risk}$ was also derived from statistical data at the national level published by FAO (2006, 2007b, 2008) and UNEP (2002). The statistical period covered is from 1985 to 2004. In countries where data on the surface affected by forest fires for a period of ten years was not available, the average burned area per year was calculated from the available information.

Once the information on each of the four variables was compiled, geo-referenced layers were made using GIS. For this, we used two software programmes: MiraMon® 6.1 (Pons 2008) and ArcView 3.2®. The GIS allows the overlap of Bailey's ecoregions layer with the four desertification variable layers, to obtain surface statistics of each variable category for each ecoregion.

$CF^{i}_{Aridity}$ and $CF^{i}_{Aquifer\ overexploitation}$ were calculated as the average value for each ecoregion. $CF^{i}_{Erosion}$ is the average value of soil erosion risk in the ecoregion soils threatened by water erosion, weighted by the ecoregion surface area. Finally, $CF^{i}_{Fire\ risk}$ was calculated by a similar method to that applied in the LCI phase, in which an ecoregion is considered to have a fire risk (CF is 2) if, during the previous 10 years, a minimum of 10% of its surface area had been affected by fires, with each country affected having a ratio equal to or higher than 10% of area burned. If the burned area of the ecoregion is lower than 10%, its CF is 1.

The CF for a given ecoregion can be calculated as the sum of the CFs for each of the four variables. The methodology explained in this section is summarised in figure 13. As shown in this figure, after carrying out the overlap between the global aridity index map and the global Bailey's ecoregions, only 8 out of 15 ecoregions have arid, semi-arid, or dry sub-humid average aridity index. These ecoregions are: marine, Mediterranean, prairie, savanna, temperate steppe, tropical/subtropical steppe, temperate desert, and tropical/subtropical desert, which represent 38% of the total land surface. According to the United Nations (1994) criterion, desertification risk is only possible in these eight ecoregions. In the remaining seven ecoregions, $CF^{i}_{Aridity}$ is equal to 0 (see section 3.2.1 and table 8).

Information from GIS technology was directly applicable to the LCA in this research, allowing the calculation of the CFs for the LCIA phase. Therefore, the simultaneous use of these tools (LCA and GIS) has clear advantages for retrieving information on land use impact. Figure 14 shows one of the four GIS layers.



[a] Three divisions of the polar domain (icecap, tundra and subarctic) were not included as they are in a cold or antarctic climate (not arid, semi-arid or dry sub-humid areas) (Fischer et al. 2000)
i: ecoregion
CF: characterisation factor

Figure 13 Diagram of the methodology applied for obtaining the characterisation factors of desertification risk for each ecoregion (CF^i).

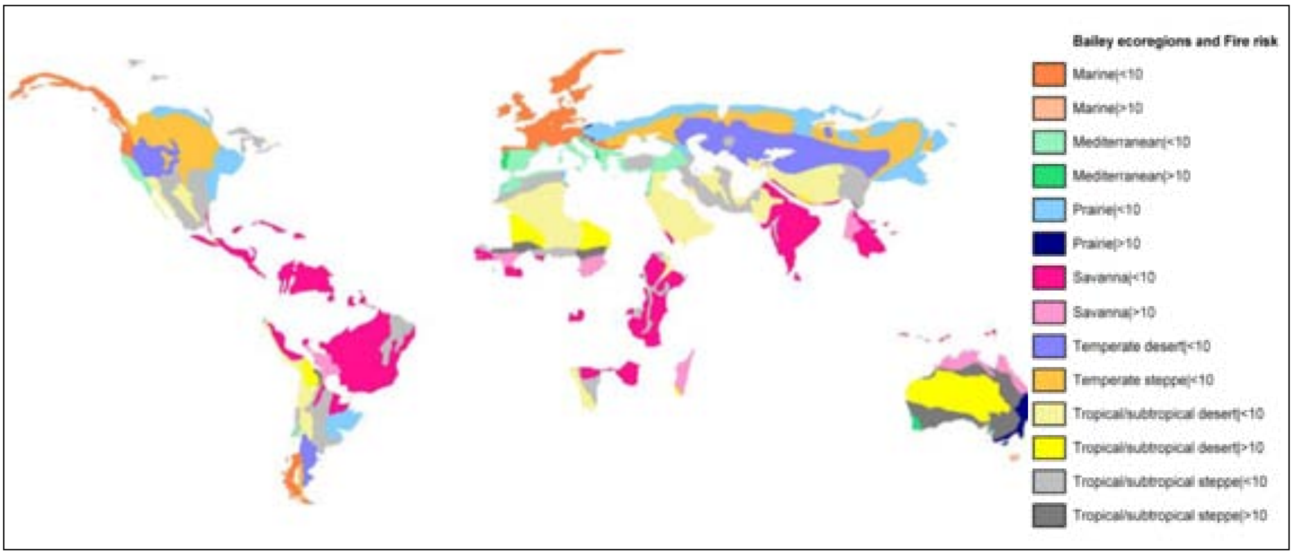


Figure 14 Global combination map of Bailey's ecoregions with fire risk.

3.4 Life cycle impact assessment modelling

3.4.1 Application of characterisation factors

Table 9 shows the CFs for each variable and ecoregion division with desertification risk. These CFs were obtained by applying the methodology described in figure 13 and the variable values estimated in the LCI data phase (section 3.2 and table 8).

The greatest desertification risk is found in the tropical/subtropical desert ecoregion, with a CF of 7.6 (out of 10). This ecoregion is mainly located in northern countries of Africa, some Arabian countries, Australia, the southwest of China, and the western edge of South America. Terrestrial deserts (hyper-arid areas where precipitation is lower than 25 mm y^{-1}) and semi-deserts (areas with imminent desertification risk) are located within this ecoregion.

Mediterranean and tropical/subtropical steppe (both with a value of 6.3) are the ecoregions with the next greatest desertification risk, while marine and prairie are, among all the ecoregions with desertification risk, the least susceptible (CF of 4).

Table 9 Characterisation factors of desertification risk for each ecoregion (dimensionless).

	Marine	Prairie	Temperate steppe	Temperate desert	Savanna	Mediterranean	Tropical/subtropical steppe	Tropical/subtropical desert
$CF_{Aridity}$	1	1	2	2	1	1	2	3
$CF_{Erosion}$	1	1	1	1	2	2	1	2
$CF_{Aquifer\ overexpl.}$	1	1	1	1.3	1	1.3	1.3	1.6
$CF_{Fire\ risk}$	1	1	1	1	2	2	2	1
$\mathbf{CF^i}$	**4.0**	**4.0**	**5.0**	**5.3**	**6.0**	**6.3**	**6.3**	**7.6**

i: ecoregion

3.4.2 Life cycle impact assessment model

In this study, we also propose a desertification impact assessment model. Equation 5 shows the model proposed for inclusion in LCA studies.

$$LCIA_{Desertification} = \left(LCI_{Desertification} \times CF^i\right) \times \frac{Area_{LCI\ activity}}{Log\ Area_{Ecoregion\ i}} \times t \qquad (5)$$

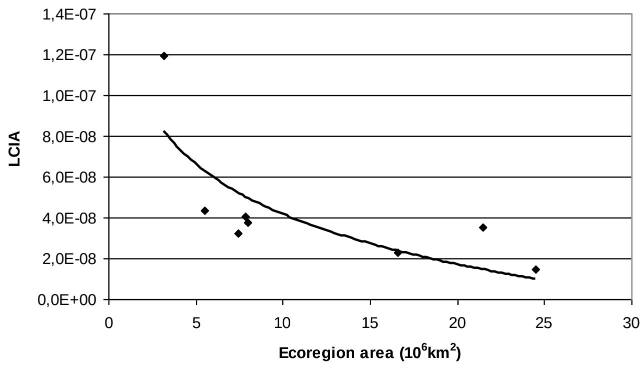
where $LCIA_{Desertification}$ is the desertification impact due to the assessed activity, in $km^2_{LCI\ activity} \times km^{-2}_{Ecoregion\ i}$ during the period of time (years) that the activity takes place; $LCI_{Desertification}$ is the inventory data (dimensionless) of this activity; CF^i is the characterisation factor of the ecoregion where the evaluated activity takes place (dimensionless, see table 9); $Area_{LCI\ Activity}$ is the spatial extension of the activity (in km^2); Log $Area_{Ecoregion\ i}$ is the decimal logarithm of the ecoregion area where the activity is located (in km^2); and *t* is the temporal extension of the activity (in years). High values of $LCIA_{Desertification}$ mean high soil desertification impact caused by the situation under analysis.

It should be noted that, when the variable $V_{Aridity}$ in the $LCI_{Desertification}$ is equal to 0 (humid climates, without desertification risk), $LCIA_{Desertification}$ is zero. In this case the desertification impact of the activity should not be integrated in LCA studies. This can be used to identify those cases without desertification impact. The $LCIA_{Desertification}$ value is also zero when CF^i or any other variable in equation 5 is zero. A value of zero for CF^i means that the activity being studied is in an ecoregion with no desertification risk (icecap, tundra, subarctic, warm continental, hot continental, subtropical, and rainforest).

The model suggested focuses on the desertification risk value of the ecoregions without comparing to a reference ecoregion (Blonk et al. 1997; Heijungs et al. 1992; Weidema et al. 1996). This approach works well, as there is no agreement about which reference system to select or how to measure it.

The LCIA model selected was that which fit best the models used to apply in LCA of those tested. Previous LCIA models tested were, for instance, the average $LCI_{Desertification}/CF^i$ and the weighted average $\sum_{i=1}^{n} V_{LCI\ variable} \Big/ \sum_{i=1}^{n} CF^{i}_{variable}$ approaches. They were not effective in cases of single land use (activities with the same LCI) in different ecoregions. In both approaches the greatest impact values were obtained in ecoregions with lower CF.

The area of the ecoregion ($Area_{Ecoregion\ i}$) is expressed in equation 5 by a logarithmic relationship with the resulting impact. The aim was to represent the likelihood of an area-desertification relation, with smaller surface areas having more desertification risk than larger areas (figure 15). Logarithmic correlations between area-species interactions have been reported for several groups of organisms (theory of insularity), where the drop in the number of species is faster in smaller areas (Begon et al. 1999). It could be argued that the potential impact does not follow an area-desertification relation in a logarithmic way. However, unlike the linear model, the model proposed shows the growing marginal effects with successive surface area reductions.



$$LCIA = [\,LCI_{Desertification} \times CF^{i}\,] \times \frac{Area_{LCI\ activity}}{Area_{Ecoregioni}} \times t$$

Figure 15 Relationship between the desertification impact (LCIA, in $km^2_{LCI\ activity} \times km^{-2}_{Ecoregion\ i} \times y$) and the ecoregion area for the 8 ecoregions with desertification risk. Hypothetical case where $LCI_{Desertification}$=6, $Area_{LCI\ activity}$= 10,000 m^2, t=1 year.

3.5 Conclusions

In this research, an LCA methodology for assessing the environmental local impact of desertification was developed and adapted. The approach adds an innovative contribution, since previous LCA methodological studies consider water consumption, erosion, and biodiversity impacts but not the desertification impact.

Four biophysical variables belonging to the state and pressure frameworks were selected for the model: aridity, erosion, aquifer overexploitation and fire risk. The desertification impact evaluation of any human activity in a LCA should include these common, basic four variables. The $LCI_{Desertification}$ value of the activity being assessed is determined by the addition of the individual values given to each of the four variables, according to a scale of values.

Following methodologies proposed by other authors to include local biodiversity impacts in LCA, we established CFs of desertification impact for the large divisions of the terrestrial ecological regions (ecoregions). This study is the first we are aware of, at an international level, to include desertification impact in LCA based on a classification of natural areas. GIS technology has facilitated the development of this study. The simultaneous use of LCA and GIS is a major advantage for gathering information that can be applied to decision-making in land management. The calculation of the CF for the aridity index shows that only 8 out of 15 terrestrial ecoregions have desertification risk, as their prevailing climate is arid, semi-arid, or dry sub-humid. These 8 ecoregions are: marine, prairie, Mediterranean, savanna, temperate steppe, temperate desert, tropical/subtropical steppe, and tropical/subtropical desert, which represent 38% of the terrestrial surface. The greatest desertification risk is found in the tropical/subtropical desert ecoregion and the lowest in marine and prairie divisions.

All the information required for a desertification impact assessment in the LCA is generally available. This paper provides CFs for including desertification impact in LCA studies, and the variables suggested allow the comparison of the benefits and threats posed by different human activities.

3.6 Recommendations and perspectives

Although the LCIA model developed as a product of factors may be a simplified approach, it can be calibrated and improved when applied to specific case studies.

The proposed LCI variables are appropriate for assessing the desertification impact of any human activity (agriculture, industry, mining, etc.). The scheme proposed could be complemented with specific LCI factors for different human activities (e.g., salinity for agricultural activities, soil crusting for building activities).

Even though we have considered biophysical variables, the method could be extended to social and economic vectors. However, this is a long-term task due to its magnitude and difficulty.

3.7 Supporting information

Table 6SI Climate, vegetation, and surface area for each ecoregion division. The identification code for each domain and division is shown in brackets.

Name of domain	Name of division	Equivalent Köppen-Trewartha climates	Zonal vegetation	Surface area (10^6 km^2)
Polar (100)	Ice cap (110)	Polar climate: ice cap		15.40[a]
	Tundra (120)	Polar climate: tundra	Ice and stony deserts: tundras	14.90
	Subarctic (130)	Boreal climate: subarctic	Forest-tundras and open woodlands; taiga	16.80
Humid temperate (200)	Warm continental (210)	Temperate climate: temperate continental, cool summer	Mixed deciduous-coniferous forests	4.34
	Hot continental (220)	Temperate climate: temperate continental, warm summer	Broad-leaved forests	2.56
	Subtropical (230)	Subtropical climate: humid subtropical	Broad-leaved coniferous evergreen forests; coniferous broad-leaved semi-evergreen forests	4.91
	Marine (240)	Temperate climate: temperate oceanic	Mixed forests	4.78
	Prairie (250)	Subtropical climate: humid subtropical Temperate climate: temperate continental, warm summer Temperate climate: temperate continental, cool summer	Forest-steppes and prairies; savannas	3.90
	Mediterranean (260)	Subtropical climate: subtropical dry summers	Dry steppe; hard-leaved evergreen forests; open woodlands and shrub	2.83
Dry (300)	Tropical/subtropical steppe (310)	Dry climate: tropical/subtropical semiarid	Open woodland and semi-deserts; steppes	13.94
	Tropical/subtropical desert (320)	Dry climate: tropical/subtropical arid	Semi-deserts; deserts	17.98
	Temperate steppe (330)	Dry climate: temperate semiarid	Steppes; dry steppes	6.84
	Temperate desert (340)	Dry climate: temperate arid	Semi-deserts and deserts	6.78
Humid tropical (400)	Savanna (410)	Tropical and humid climate: tropical wet-dry	Open woodlands, shrubs and savannas; semi-evergreen forest	20.54
	Rainforest (420)	Tropical and humid climate: tropical wet	Evergreen tropical rain forest	12.50

Source: Bailey (2002)
[a] Includes Antarctica.

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Chapter 4.

Assessment methodology to establish new USLE cover and management C-factors. Case study on food and energy crops.

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Abstract

The USLE equation is one of the most widely used methods to estimate soil losses by water erosion. Datasets of the cover and management factor (C-factor) in the USLE equation were developed for specific cropping systems so they are not always applicable, or a generalised C-factor must be used.

To overcome this, we developed a methodology to determine new USLE C-factors. Periodical photographs of the crop (per month or per cropstage), specific software for vegetative cover assessment, monthly data of local rainfall and tables and figures of vegetative cover proposed by the authors of the USLE are the parameters needed to apply this C-factor assessment scheme. The method was successfully used in a case study with a two-year rotation of winter wheat and oilseed rape. The C-factor for the complete winter wheat cycle was 0.222 and that for oilseed rape, 0.400, indicating that soil losses due to the cover and management factor are higher during oilseed rape cultivation than when winter wheat is grown. Using this methodology, a C-factor data set for different crops and locations can be built, facilitating further studies on soil erosion estimates.

Keywords Agricultural systems • Oilseed rape • Soil erosion • Universal soil loss equation (USLE) • Vegetative cover

4. 1 Introduction

Soil erosion is an issue of major interest, especially in arid and semi-arid countries, where rainfall intensities are high and the mean annual precipitation is low (Cerdà et al. 2009). It is one of the main reasons for soil degradation and desertification (United Nations 1994). Soil erosion involves the detachment and removal of topsoil from one site, due to wind or flowing water, and its transport to another location. As topsoil is normally rich in nutrients, a relatively large amount of nutrients is lost together with the topsoil (Quinton et al. 2006; Ramos and Martínez-Casasnovas 2006). This erosion results in a rapid decrease in levels of soil organic matter, a necessary component of soil, because it facilitates the formation of soil aggregates and increases soil porosity, thereby improving soil structure, water infiltration, and overall productivity (Pimentel et al. 1995). The soil surface is susceptible to erosion when the live plant or residue cover is inadequate. Of the world area affected by human-induced soil degradation, water erosion has the greatest impact, affecting around 1,100 million ha. (Oldeman et al. 1990).

One of the methods most widely applied for estimating water erosion is the universal soil loss equation (USLE, Wischmeier and Smith 1978) or models based on it, such as in the erosion evaluation of the Dutch National Institute for Public Health and the Environment (RIVM 1992), that of van der Knijff et al. (2000), and the INRA estimations (le Bissonnais et al. 2002). Due to its modest data requirements compared to other models and its transparent structure, USLE is the most internationally accepted quantitative model of soil loss evaluation on an international level (Panicker et al. 2004). USLE and its updates (Revised USLE, RUSLE, Renard et al. 1997) predict long-term average soil losses in runoff from a single plot under specific cropping and management systems, with a specific soil type, rainfall pattern and topography. The soil loss is calculated as shown in equation 6:

$$A = R \times K \times LS \times C \times P \qquad (6)$$

where

A is the average annual soil loss by water erosion [t ha^{-1} y^{-1}].

R is the rainfall and runoff factor, which increases with rain intensity and duration.

K is the soil erodibility factor, which depends on soil texture and structure.

LS is the topographic factor combining the effects of the slope (S) and the slope-length (L), with the value of LS increasing for steeper and longer plots.

C is the cover and management factor, which determines the effectiveness of the vegetative cover and management practices in preventing soil losses.

P is the support practice factor, which reflects the positive effects of the practices in reducing runoff and erosion, such as contour farming or terracing.

Despite its virtues, a significant shortcoming of the USLE is that only rill and sheet erosion are estimated, thus other types of water erosion and deposition processes are not modelled. However, rill and sheet erosion are the most important types of water erosion in agricultural land (Durán Zuazo and Rodríguez Pleguezuelo 2008; Morgan 1997). USLE and other models also over-predict small soil losses and under-predict large soil losses (Nearing 1998). Therefore, long-term estimations are best as these factors balance one another out (Gabriels et al. 2003).

Apart from these model deficiencies, the authors of USLE supply C-factors for a limited number of cropping systems and management practices, with specific C-factor values for a number of crops and management practices not yet calculated. Nevertheless, C values are essential for estimating the impact of land use on soil erosion.

The C-factor, dimensionless, is one of the most important parameters of the USLE, since it takes into account all the variables influenced by agricultural practices, so it is the factor most affected by human activity. It may be used as a conservation planning criterion, to select a suitable rotation system that reduces soil erosion on site. The C-factor is based on the concept of deviation from a standard, an area under clean-tilled continuous-fallow management conditions (Institute of Water Research 2002), and ranges from 0 (very strong cover effect, no erosion) to 1 (no cover effect, erosion comparable to tilled bare soil).

Crop canopy protection depends on the type of vegetation, the stand, and the quality of growth, but also varies greatly in different months or seasons (Wischmeier and Smith 1978), depending on the erosivity of rainfall. Therefore, as the C-factor is different in each region or locality for a particular cropping system during the year, for determining accurate C-factors it is important to know the distribution of rainfall throughout the 12 months of the year.

Although the C-subfactor approach of RUSLE better reflects current management practices, a new method for determining USLE C-factors is advantageous as it requires less extensive data.

C-factors can be measured in field experiments, but this is very time-consuming (Gabriels et al. 2003) and expensive. C-factor mapping methods based on the use of remote sensing technologies reduce the time and cost, but there is poor correlation with C-factors in the literature (Erencin 2000). All this creates the need for an easy to apply and time-efficient method for obtaining new and accurate C-factors that facilitates further studies on soil erosion estimates.

The aims of this study were i) to develop a methodology to determine new USLE cover and management factors, necessary for estimating soil losses by water erosion using the equation, ii) to check the applicability of the method, defining C-factors for a two-year rotation of food and energy crops. With a new methodology such as this, requiring less and more accessible data, a C-factor data set for different crops and places can be constructed for erosion assessments.

4.2 Materials and methods

4.2.1 Procedure to derive cover and management factors

The methodology applied by Morgan (1997) in an area with a rainforest climate and by van der Knijff et al. (1999) in an area with a Mediterranean climate was followed here (equation 7). According to this equation, an individual C value for each period (cropstage or month, C_i, ranging from 0 to 1) is weighted with the erosivity of rainfall for that period (R_i), and the annual C-factor (C) obtained by the addition of the partial C values is defined by:

$$C=\sum C_i \times R_i \qquad (7)$$

As previously mentioned, erosivity of rainfall, the erosion potential which depends on intensity and duration, has to be taken into account as the erosion control effectiveness of a cropping system in a particular field depends on the distribution of erosive rainfall during the year (Wischmeier and Smith 1978). Morgan (1997) and van der Knijff et al. (1999) assumed that erosivity can be directly related with the amount of rainfall. This simplification makes the calculation of C-factors straightforward and may be especially useful for those who do not have access to comprehensive rainfall datasets. It is advisable that rainfall data to calculate R_i refer to a time period of several years in order to avoid bias due to drought and excessively wet years.

For an annual crop, USLE distinguishes six cropstage periods: (1) seedbed, from sowing to 10% canopy cover (2) establishment, from 10% to 50% canopy cover (3) development, from 50% to 75% canopy cover (4) maturing crop, from 75% to harvest (5) residue or stubble, from harvesting to ploughing and (6) rough fallow, from ploughing to new seeding. C_i and R_i parameters can be calculated either for cropstage periods or monthly periods. Within each period, both parameters are considered uniform. Whenever possible, it is recommended that C_i and R_i are calculated on a monthly basis, for more accurate C-factors.

Calculation of C_i requires tables and plots from Wischmeier and Smith (1978), where C_i-factors are derived from information on:

(1) Type and height of vegetative cover

(2) Percentage vegetative cover

(3) Crop type

(4) Percentage ground cover in contact with the soil surface

Combining these four subfactors gives C_i values for each *i* period from sowing to harvest, whereas cropstages 5 and 6 are post-harvest. Data for (1), (2) and (4) must be gathered from fieldwork in the plot under study. C_i-factors for cropstages from sowing to harvest are derived from Wischmeier and Smith (1978, table 10 pp.32), C_i -factors for the residue or stubble cropstage come from figure 16, and values for the fallow period from the literature.

Type and height of the vegetative cover

Four different vegetative covers were distinguished by Wischmeier and Smith (1978, table 10 pp.32), according to their type and height: (a) no appreciable canopy (b) tall weeds or short brush with average drop fall height of 0.45 meters (c) appreciable brush or bushes, with average drop fall height of 1.65 meters and (d) trees, but no appreciable low brush, average drop fall height of 3.30 meters. Intermediate categories can also be defined.

Percentage vegetative cover

Digital photographs for each *i* period (cropstage or month) are necessary for calculating the percentage vegetative cover per *i* period, with photos taken on or around the same day every month, using a zenithal orientation and avoiding shadows for high quality photos. The percentage of vegetative cover can be worked out from a photo using specialized software to count the number of green pixels, or any specified colour. Examples of this kind of software are

GreenPix 0.3® (Casadesús et al. 2007) and CobCal v.1.1® (Ferrari et al. 2007). Percentage vegetative cover that differs from 0, 25, 50 and 75 per cent coverage predefined by the USLE's authors can be added to derive intermediate C_i-factors.

Crop type

Two types of vegetative cover were distinguished by Wischmeier and Smith (1978, table 10 pp.32). Type G coverage is grass or grasslike plants on the surface and type W, mostly broadleaf herbaceous plants on the surface.

Percentage ground cover in contact with the soil surface

This subfactor must not be mistaken for percentage vegetative cover. While percentage cover refers to the portion of total surface area that would be hidden from view by the canopy in a vertical projection (a bird's eye-view), percentage ground cover measures the part of this canopy in contact with the soil: the higher the ground cover percentage, the smaller the C_i-factors and therefore the soil losses. This parameter is calculated by visualisation in the field, although photographs taken to calculate the percentage vegetative cover may help in this calculation. In Wischmeier and Smith (1978, table 10 pp.32), six percent ground cover categories were differentiated, from 0 to >95%. Intermediate percentages may again be added.

Residue or stubble stage

The expected effects of mulch and canopy combinations have been calculated in Wischmeier and Smith (1978, see figure 15). We used this figure to account for the residue or stubble cropstage soil losses, by taking the curve that corresponds to zero per cent canopy cover (highlighted in figure 16). The percentage cover by mulch on the x-axis can be found by processing the digital photographs of the residue or stubble period with software such as GreenPix 0.3®, or CobCal v.1.1®, which count green pixels, moving vertically to the zero per cent canopy curve to read the soil loss ratio on the y-axis.

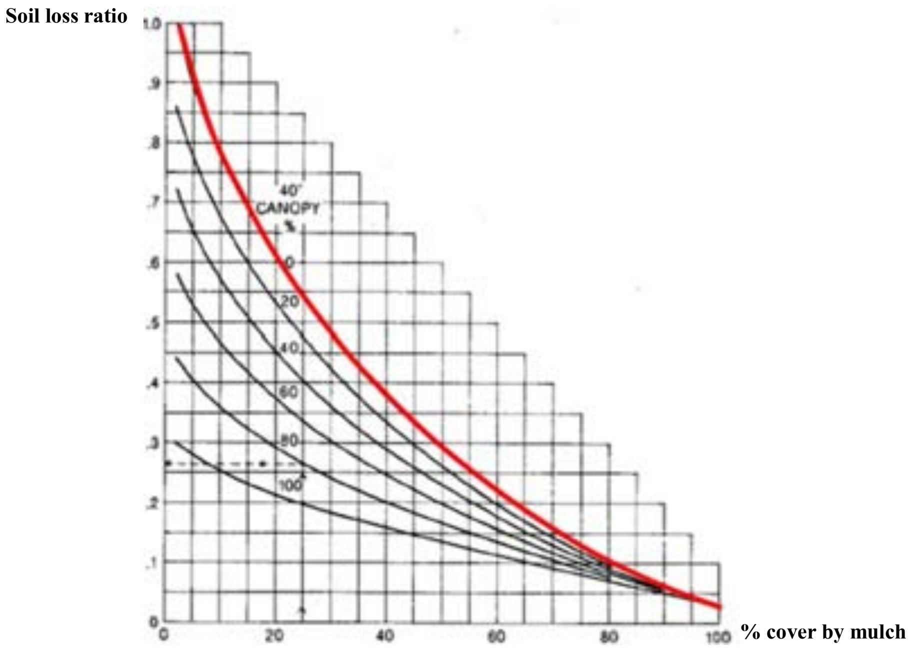

Figure 16 Plotted C_i values for deriving cover and management factors for the residue or stubble cropstage, shown in a thicker red line width (0% canopy). Adapted from Wischmeier and Smith (1978).

Rough fallow period

This C_i-value is derived from the literature, which gave a value between 0.90 (Boellstorff and Benito 2005) to 1.00 (Kooiman 1987).

4.2.2 Study area

The methodology to calculate C-factors was applied to one plot located in Catalonia, Spain (geographic coordinates 2º46’30’’East, 41º49’45’’North, Datum WGS1984). A two year rotation of winter wheat (*Triticum aestivum*) and oilseed rape (*Brassica napus*), both winter annual crops, was grown over the 2006-2008 period. Winter wheat is one of the most common crops in Spain, usually grown under rainfed conditions. The herbaceous crop, oilseed rape, is also usually grown without irrigation and its cultivation has increased over the last few years for the production of biodiesel. This energy crop can make a substantial contribution to increasing the share of renewable energy consumption in the country. In the plot under study, winter wheat was sown on 1st November 2006 and harvested on 26th June 2007. It was left with a mulch of straw residues from the harvest until the end of August, when the plot was ploughed to keep it fallow. On 28th September 2007, oilseed rape was sown and the crop was harvested on 1st July 2008. Again, the soil was left with straw residues until tillage. The rough fallow period lasted from this tillage to the next seedbed preparation, which could, for example, be used for sowing another cereal crop in November 2008.

C_i and R_i for wheat and oilseed rape were calculated on a monthly basis, as suggested in section 4.2.1. The distribution of rainfall during the year, R_i, was calculated based on detailed information from the Spanish Ministry of the Environment and Rural and Marine Affairs (MARM 2010), which recorded monthly average rainfall over a period of 36 years (1960-1996) in 4,189 weather stations scattered throughout the country. The nearest weather station to the plot was located at approximately 4 km. (geographic coordinates 02º44'East, 41º48'North, Datum WGS1984).

4.3 Results

The outcomes of the erosivity of rainfall in the analyzed plot and of the cover and management factor for *i* period and for the complete cycles of winter wheat and oilseed rape are stated below.

4.3.1 Erosivity of rainfall

The adjusted rainfall factor, R_i, for each month according to rainfall data from the weather station set out in subsection 4.2.2 are shown in table 10.

Table 10 Average monthly precipitation (Pr) over the period 1960-1996 and adjusted rainfall factors (R_i) for each month in the plot under study.

Month	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Pr (mm.)	53.9	49.7	54.7	57.7	71.1	46.0	26.3	52.3	70.9	116.2	69.1	62.3	**730.2**
R_i	0.07	0.07	0.07	0.08	0.10	0.06	0.04	0.07	0.10	0.16	0.09	0.09	**1.00**

4.3.2 Cover and management factor for *i* period

C_i-monthly-factors for winter wheat and oilseed rape were calculated as stated in subsection 4.2.1.

Type and height of vegetative cover

The height of the vegetative cover was measured each month, on the same day when digital photos of the canopy cover were taken. According to the categories of Wischmeier and Smith (1978, table 10 pp.32), winter wheat fitted into category (a), no appreciable canopy, from the seedbed in November until February, and into category (b), tall weeds or short brush with average drop fall height of 0.45m, in March and April. An intermediate category between (b) and (c) was defined for the final maturing stage in May and June, which includes brush of 1.00 m average drop fall height. C_i-values were estimated as the average between C_i-factors of categories (b) and (c). For example, the value of C_i for 25% vegetative cover, crop type G and 0% ground cover will be equal to 0.38 (0.36 plus 0.40 divided by 2).

For oilseed rape, in September and October the type and height of vegetative cover corresponded to category (a), and to category (b) from November to March. In April, when the flowers appear, oilseed rape grows fast, so the final cropstages before harvest fell into category (c), with appreciable brush or bushes, and an average drop fall height of 1.65 m.

Percentage vegetative cover

The GreenPix 0.3® (Casadesús et al. 2007) software was used for the percentage of vegetative cover pixels from digital photographs taken monthly for both crops. Monitoring took place from the date of winter wheat sowing, in November 2006, until the rough fallow after the oilseed rape crop, in October 2008, prior to the seedbed of the next crop. Percentage vegetative cover of the winter wheat and oilseed rape crops, dates and photographs for each month are shown in table 11.

Table 11 Digital photographs of winter wheat and oilseed rape by month *i* and the corresponding vegetative cover (VC, %) obtained with GreenPix 0.3® (Casadesús et al. 2007). Note the zenithal orientation of images and the same or similar dates when the photos were taken each month.

Winter wheat (2006-2007)					
					
Nov. 1 (sowing)[a] VC: 0%	Dec. 1 VC: 19%	Jan. 2 VC: 19%	Feb. 3 VC: 26%	Mar. 1 VC: 77%	Apr. 3 VC: 85%
					
May 2 VC: 99%	Jun 1 VC: 99%	Jun 26 (harvest) VC: 99%	Jul 3 (residue) VC: 97%	Aug 6 (residue) VC: 16%	Sep. 2 (fallow)[a] VC: 0%

Oilseed rape (2007-2008)					
					
Sep.28 (sowing)[a] VC: 0%	Oct. 25 VC: 7%	Nov. 24 VC: 88%	Dec. 23 VC: 82%	Jan. 25 VC: 78%	Feb. 25 VC: 78%
					
Mar. 23 VC: 87%	Apr. 25 VC: 92%	May 23 VC: 99%	Jun. 26 VC: 93%	Jul. 1 (harvest) VC: 90%	Jul.25 (residue)[b] VC: 11%
					
Aug. (residue)[b] VC: 11%	Sep. (residue)[b] VC: 11%	Oct. (fallow)[a] VC: 0%			

[a] Same photographs used for sowing and fallow stages for both crops, due to their high quality.
[b] Same photographs applied, as no fieldwork was carried out during these months.

Crop type

Wheat was considered to be plant type G (grass or grasslike plants) and oilseed rape plant type W (broadleaf herbaceous plants).

Percentage of ground cover in contact with the soil surface

This subfactor was found by the observation of the photographs and the plot, with the ground cover percentage for each *i* period selected using a conservative approach, i.e., choosing the worst possible scenario, where soil losses could be higher. This gave 0% ground cover in contact with the soil for the seedbed cropstage, 20% for the establishment and development stages, and 40% for the final maturity cropstage. This was applied for both winter wheat and oilseed rape, and may also be applied to other crops with similar cycles and cover canopies (e.g., barley, maize, sorghum).

Residue or stubble stage

The winter wheat and oilseed rape cropping systems studied were left with residues on the surface to prevent soil, nutrient and moisture losses from harvest to ploughing prior to the new seedbed (FAO 2011). The percentage residue cover during these months was estimated by processing the digital photographs with GreenPix 0.3® and entering the percentage on the x-axis of figure 16, calculating the corresponding C_i value on the y-axis by moving to the 0% canopy curve.

Rough fallow period

The 0.9 C_i-factor was chosen, as it has been applied in previous studies of soil losses in Spain (Boellstorff and Benito 2005).

Table 12 shows the C_i values for each month of this two-year-rotation cropping system as well as the parameters used in each month to derive them.

Table 12 C_i-factors for winter wheat and oilseed rape for each *i* period of the complete cycle on a monthly basis and the parameters used.

Winter wheat						**Oilseed rape**					
Month	Type & height[a]	VC[b] (%)	Type[c]	GC[d] (%)	**C_i-factor**	Month	Type & height[a]	VC[b] (%)	Type[c]	GC[d] (%)	**C_i-factor**
Nov 06	1	0	G	0	**0.45**	Oct 07	1	3.5[e]	W	0	**0.43**
Dec 06	1	19	G	20	**0.16**	Nov 07	2	88	W	40	**0.05**
Jan 07	1	19	G	20	**0.16**	Dec 07	2	82	W	40	**0.06**
Feb 07	1	26	G	20	**0.15**	Jan 08	2	78	W	40	**0.08**
Mar 07	2	77	G	40	**0.06**	Feb 08	2	78	W	40	**0.08**
Apr 07	2	85	G	40	**0.03**	Mar 08	2	87	W	40	**0.05**
May 07	3	99	G	40	**0.00**	Apr 08	4	92	W	40	**0.04**
Jun 07	3	99	G	40	**0.00**	May 08	4	99	W	40	**0.00**
Jul 07	RC	97			**0.04**	Jun 08	4	93	W	40	**0.03**
Aug 07	RC	16			**0.67**	Jul 08	RC	11			**0.75**
Sep 07	RF	0			**0.90**	Aug 08	RC	11			**0.75**
						Sep 08	RC	11			**0.75**
						Oct 08	RF	0			**0.90**

[a] 1: no appreciable canopy. 2: average drop fall height of 0.45 m. 3: average drop fall height of 1.00 m. 4: average drop fall height of 1.65 m. RC: residue cover. RF: rough fallow.
[b] Vegetative cover.
[c] G: grass or grasslike plants. W: broadleaf herbaceous plants.
[d] Ground cover.
[e] Average of the September VC (0%) and the October VC (7%).

4.3.3 Cover and management factors for the complete cycles

Equation 7 was applied to obtain monthly C-factors adjusted for rainfall erosivity for winter wheat and oilseed rape, and to get a complete cycle C-factor for both crops (i.e., from seedbed to the fallow period before the subsequent seedbed, see table 13). The C-factor for the complete winter wheat cycle was 0.222 and for oilseed rape, 0.400, indicating that soil losses due to the cover and management factor are higher during oilseed rape cultivation than when winter wheat is grown. Comparison of C_i values for cropstage periods from sowing to harvest with those of the residue or stubble stage showed that residues are less effective than equivalent percentages of canopy cover from sowing to harvest.

Table 13 C-factors for winter wheat and oilseed rape for each *i* period on a monthly basis and for the complete cycle.

Winter wheat				**Oilseed rape**			
Month	R_i	C_i	**C**[a]	Month	R_i	C_i	**C**[a]
Nov 06	0.09	0.45	**0.041**	Oct 07	0.16	0.43	**0.069**
Dec 06	0.09	0.16	**0.014**	Nov 07	0.09	0.05	**0.005**
Jan 07	0.07	0.16	**0.011**	Dec 07	0.09	0.06	**0.005**
Feb 07	0.07	0.15	**0.011**	Jan 08	0.07	0.08	**0.006**
Mar 07	0.07	0.06	**0.004**	Feb 08	0.07	0.08	**0.006**
Apr 07	0.08	0.03	**0.002**	Mar 08	0.07	0.05	**0.004**
May 07	0.10	0.00	**0.000**	Apr 08	0.08	0.04	**0.003**
Jun 07	0.06	0.00	**0.000**	May 08	0.10	0.00	**0.000**
Jul 07	0.04	0.04	**0.002**	Jun 08	0.06	0.03	**0.002**
Aug 07	0.07	0.67	**0.047**	Jul 08	0.04	0.75	**0.030**
Sep 07	0.10	0.90	**0.090**	Aug 08	0.07	0.75	**0.053**
				Sep 08	0.10	0.75	**0.075**
				Oct 08	0.16	0.90	**0.144**
Complete cycle C-factor			**0.222**	**Complete cycle C-factor**			**0.400**

[a] $C = \sum R_i \times C_i$

4.4 Discussion

Rainfalls recorded in the Mediterranean study area in autumn (October to December) have the highest erosive potential of the year, accounting for 34% of the annual rainfall erosive factor (R). These seasonally concentrated large erosive rainfalls for the Mediterreanean region were also measured in previous studies with Mediterranean climate (Diodato and Bellocchi 2009; Usón and Ramos 2001).

C-factors obtained for winter wheat and oilseed rape corresponded to soil losses suffered during the complete cycle of each crop in the rotation system, 11 months for winter wheat and 13 months for oilseed rape. Considering this length of time, oilseed rape was less effective than winter wheat at controlling soil losses (C-factors of 0.400 and 0.222 for oilseed rape and winter wheat, respectively). However, if C-factors are calculated on a yearly basis, soil losses due to the cover and management factor are higher for winter wheat than for oilseed rape (C-factors of 0.271 for oilseed rape and 0.343 for winter wheat, if a period of 12 months is considered from the time of seeding). This difference between the resulting C-factors is because of the rainfall distribution during the cropstages as well as the consideration of winter wheat and oilseed rape as single crops or as crops embedded in a rotation system.

For example, if C-factors are calculated on a yearly basis for winter wheat, being sown in November of the first year and again in the following November, there is high rainfall erosivity (0.16) and a high C_i value (0.90, fallow stage) in October, giving a C value of 0.144. However, if winter wheat is part of a rotation in which oilseed rape is cultivated after wheat, soil losses in October correspond to the oilseed rape crop, considering it is sown at the end of September. Hence the comparison of crop C-factors must be handled with caution, always bearing in mind exactly what is being compared.

Another factor affecting the results was the assumption that there would be another cycle of winter wheat grown after oilseed rape, with the new winter wheat being sown on the same date as the former (November, 1). This means there was a four-month period without canopy cover between harvesting the oilseed rape on July 1, until the sowing of winter wheat, namely the three summer months (July, August and September) with residue cover and the first month of autumn (October) with rough fallow (tables 12 and 13). The C-factor during these months added up to 0.302, 75.5% of the total oilseed rape C-factor (0.400).

Good management practices to reduce soil erosion on site would be to raise the amount of residues left on the soil surface between harvest and the subsequent seeding, and to promote direct seed cropping systems, i.e., planting with no prior tillage to prepare the soil, retaining most of the crop residues on the soil surface. Another good soil conservation practice includes sowing quick-growing catch crops between successive planting of the main crops. These crops take up nutrients from the root zone, preventing them from leaching, so improving soil fertility when the catch crop is ploughed in (Rinnofner et al. 2008).

4.5 Conclusions

A user-friendly assessment methodology to derive new cover and management factors of the USLE equation has been developed. The method is more time-efficient and inexpensive than other already existent procedures, as it was based on published literature, simple fieldwork and available weather records and easy-to-use specialized software. The method was tested in a plot with a two-year rotation of winter wheat and oilseed rape. Of the two crops, the complete cycle of oilseed rape had a higher C-factor, with higher soil losses due to crop management. This is mainly because of the longer cycle of oilseed rape compared to that of winter wheat. From an agronomic perspective, the proposed scheme was used to easily characterize the performance of the analysed plot for reducing soil loss rates. It is concluded that, with this standardized method, a comprehensive data set of local C-factors can be built for further studies on environmental impacts derived from soil erosion.

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Chapter 5.

Inclusion of soil erosion impacts in life cycle assessment: application to energy crops in Spain.

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Abstract

A regionalized method for assessing the environmental impacts of soil erosion in LCA was developed. Life cycle inventory data of topsoil and topsoil organic carbon (SOC) losses is interpreted at the endpoint level in terms of ultimate damages to resources and ecosystem quality. Human health damages were excluded from the assessment. The method was tested on a case study of five three-year agricultural rotations, two of them with energy crops, grown in several locations in Spain. A great variability of soil and SOC losses was recorded in the inventory step, depending on climatic and edaphic conditions. The importance of using a spatially-explicit model and characterisation factors is shown in the case study. This regionalized assessment covers the differences in soil erosion-related environmental impacts caused by the great variability of soils. Taking this regionalized framework as the starting point, further research should focus on determining the most relevant and feasible scale at which to address the spatial differentiation for the inventory and the impact factors, to adequately capture the highly diverse nature of soils.

Keywords Agricultural systems • Ecosystem services • Life cycle impact assessment (LCIA) • Soil loss • Spatial differentiation • Universal soil loss equation (USLE)

5.1 Introduction

Life cycle assessment aims to provide a general picture of the environmental impacts of resource consumption and emissions during the entire cycle of products and systems. Despite the fundamental role of ecosystem goods and services in sustaining human activities, there is no harmonized and internationally agreed method for including them in LCA. According to a key framework for land use impact assessment in LCA (Milà i Canals et al. 2007a), ecosystem goods and services that should be integrated within LCA are impacts on biodiversity and, at least, impacts on the following five major ecosystem services: biotic production potential, carbon sequestration potential, freshwater regulation potential, water purification potential and erosion regulation potential. Operational characterisation factors and methods covering impacts on some of these ecosystem goods and services have been recently proposed (Müller-Wenk and Brandão 2010; Schmidt 2008) and others are currently being developed (Beck et al. 2010; Saad et al. 2010).

Because land use impacts of an activity depend on the location, methods focusing on these impacts should include geospatial information both in the LCI and the LCIA phases. Different levels of regionalization (e.g., countries, ecoregions, biomes) and ecological units classifications (e.g., life zones in Holridge 1947, ecoregions in Olson et al. 2001) in the LCIA are presently used, without a clear recommendation on a standardized approach to address the spatial differentiation. Although the use of ecological or geographical units instead of administrative limits give better estimates of the site-dependency of land use impacts, especially in countries with a high degree of variability, it is generally easier to find information at a country scale.

Our objective was to go a step further toward the integration of ecosystem services in LCA, developing a globally applicable and spatially resolved method to include land use occupation impacts on the erosion regulation potential ecosystem service. Indicators of the impact category were defined on the endpoint level, so modelling impacts on the entities described by the AoP, i.e., human health, natural environment and natural resources. The case study conducted to demonstrate the applicability of the method, focused on the impacts of agricultural rotations with energy crops in Spain as compared to the cultivation of traditional crops.

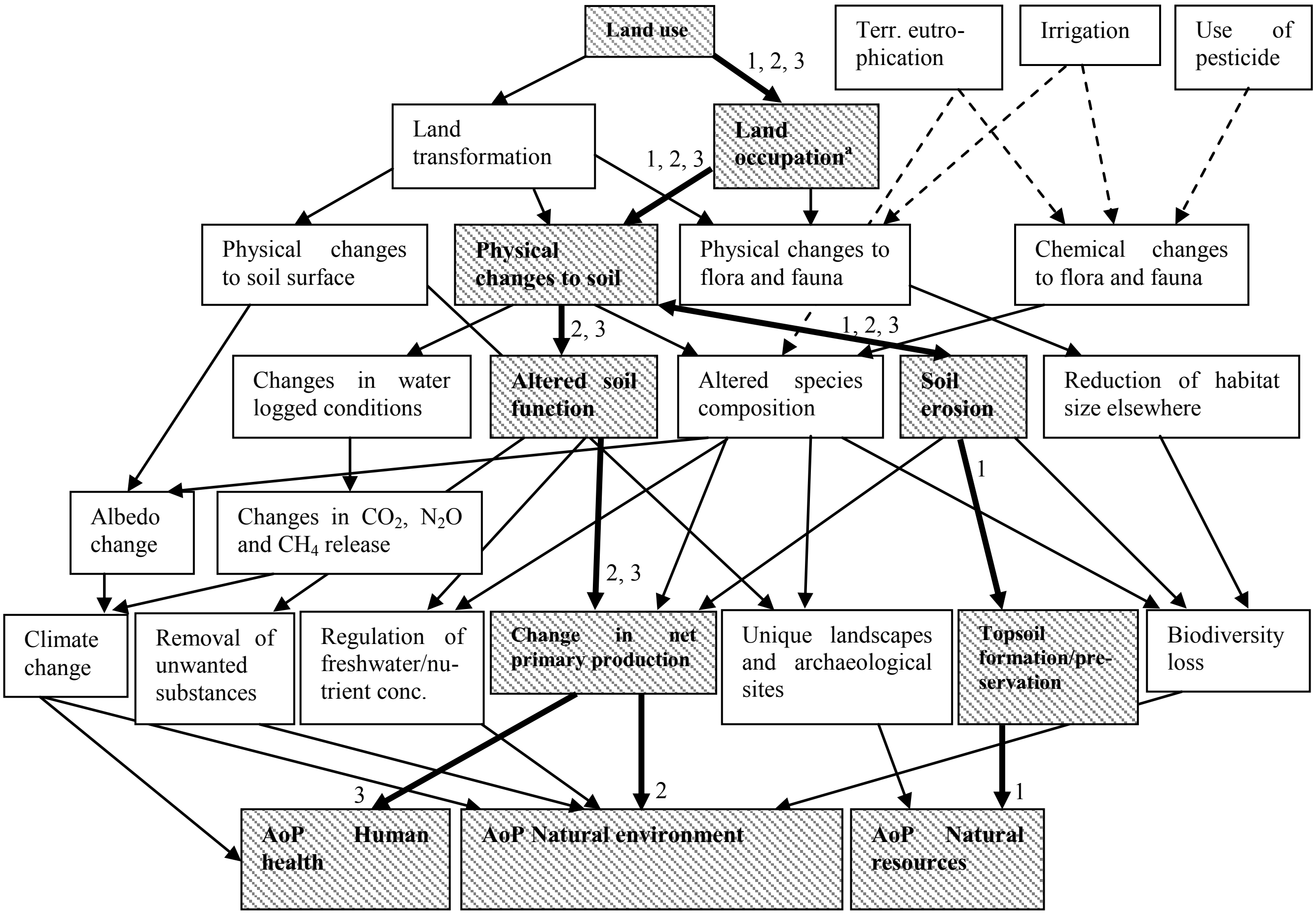
5.2 Materials and methods

Figure 17 illustrates the general land use impact mechanism, and shows the three impact pathways studied, as follows:

i. Land use occupation leads to soil erosion and this leads to loss of topsoil reserves, leading to damages on natural resources.

ii. Land use occupation leads to soil erosion and this to physical changes in the soil quality which affects the biotic production leading to damages on the natural environment.

iii. Land use occupation leads to soil erosion so to physical changes in the soil quality which affects food production leading to damages on human life.



[a] Land occupation does not “cause changes” but contributes to prolong altered conditions.

Figure 17 Main impact pathways related to land use. The pathways discussed in the paper are shown with cross-hatching and thicker arrows. Numbers 1, 2, 3 refer to impact pathways described in sections 5.2.1, 5.2.2 and 5.2.3 respectively (adapted from JRC 2010).

5.2.1 Resource depletion impact pathway

The AoP of natural resources is concerned with the removal of resources from the environment (impact pathway 1 in figure 17). Annually, humans cause the loss of 50-75 billion tons of soil (Harvey and Pimentel 1996). More than 75% of the arable soils of the world suffer from moderate to very high soil losses (Reich et al. 2001), well above the average rate of soil formation of 0.5-1 t $ha^{-1}y^{-1}$ (Mann et al. 2002). Though agricultural land accounts for 75% of the soil erosion worldwide, it also occurs in other human-modified ecosystems, such as the construction of roads and buildings. Current soil losses due to land use (land occupation) reduce soil availability as a future resource.

Life cycle inventory data requirements

The type of land use occupation has a determining role in the quantity of soil loss, as specific direct physical interventions are often related to the land management. The classification most used to register the type of land use occupation in the LCI is that of the Ecoinvent database (Frischknecht et al. 2007). The type and intensity of land use, e.g., natural or intensively used forest, fallow or irrigated arable crops, should be recorded in the LCI. In the method drawn here, these variables are considered within the calculation of soil losses in the USLE model (Wischmeier and Smith 1978). There is a consensus that the USLE equation and its update (revised USLE, Renard et al. 1997) are valid methods to estimate soil losses by water at the inventory stage (Beck et al. 2010; Muys and García Quijano 2002). The use of both methods has spread worldwide to predict soil erosion losses thanks to the growing availability and accessibility to climatic, edaphologic and land use and land cover data at the local and regional level. This spatial information can be increasingly found in a geo-referenced format, allowing data processing and visualization in GIS software. Soil losses are expressed in the LCI in soil loss mass.

As land occupation impacts are recorded in square metres times time (e.g., m^2 y) the area as well as the duration required for the production of a certain amount of products and services have to be gathered in the LCI. The georeferenced location of land use (longitude/latitude) should also be included if available. Failing that, a broader resolution (e.g., region, country) can be used, although this reduces the quality of the LCI data.

Impact assessment model

In line with other endpoint methods (EI99, Goedkoop and Spriensma 2001), damage to resources is expressed in surplus energy needed to make the resource available at some point in the future. Here, instead of using energy units (MJ), we used the Ecological Cumulative Exergy Consumption (ECEC) transformity value, which represents the exergy required to replace the soil loss during land occupation. With ECEC values flows expressed in raw units (e.g., g, m^3) can be converted to solar equivalent joules (sej), allowing for comparisons among different types of resources. We assumed an ECEC transformity value of 2.39E+07 sej g^{-1} soil loss (Odum 1996). The effects of soil erosion on soil resources (ΔR) are expressed as follows with units of sej soil loss per unit of area and time of land occupation:

$$\Delta R = \underbrace{\text{Soil loss}}_{\text{LCI}} \times \underbrace{\frac{SD_{ref} - SD_i}{SD_{ref}} \times ECEC_{transformity\ soil} = \text{sej m}^2\text{y}}_{\text{CF}} \qquad (8)$$

The endpoint indicator for resource impacts combines the inventory flow (i.e., soil loss) with the ECEC soil transformity value and the local available soil reserves on a spatial resolution of 5 arc-minutes (approximately 10×10 km^2, FAO/UNESCO 2007), as the characterisation factor (CF). This method relates impact assessment to biogeographical conditions in each grid cell *i* (SD_i), without any further aggregation of land use type or land use cover. Values were normalized with a reference soil depth (SD_{ref}), being the worldwide maximum locally available soil reserves (SD_{ref}= 3m). Choosing the maximum soil depth as the reference was judged to be a more objective correction of the site dependency of the characterisation factor, rather than the soil reserves of a particular region (e.g., Swiss lowlands for ecosystem quality assessment in the EI99 methodology, Goedkoop and Spriensma 2001). Systems with a lower value in the resource depletion category are less harmful for the environment than those with a high value.

5.2.2 Ecosystem quality impact pathway

Soil erosion contributes to damage of the natural environment. This AoP is concerned with negative effects on the function and structure of natural ecosystems (impact pathway 2 in figure 17). In the erosion process, soil quality declines, essential plant nutrients are lost and soil depth is reduced. As a result, biomass productivity diminishes. Ultimately, this has a profound effect on the diversity of plants, animal, microbes and other forms of life present in the ecosystems (Pimentel and Kounang 1998).

One of the methods used for measuring environmental impacts on ecosystem functions is based on the soil organic matter (SOM) content (Milà i Canals et al. 2007b). Here, we assessed the effects of soil erosion on the terrestrial ecosystem quality (ΔEQ) by modelling the change of biomass production due to changes in soil organic carbon (SOC) content. Biomass production of ecosystems was assessed as a function of the net primary production of potential vegetation (NPP_0).

Life cycle inventory data requirements

The quantity of SOC in the topsoil can be determined by direct measurements, calculated using models or estimated from figures in the literature. While site-specific measurements are more accurate, this is unlikely to be achievable in an LCA study (Milà i Canals et al. 2007b). The SOC content is generally about 58% of the SOM content (Buringh 1984). Erosion is considered to occur when SOC falls below 2%, while there may not be noticeable yield reductions unless SOC falls below 1% (Holland 2004). These are average values relying on site-specific conditions.

In the method outlined here, we calculated SOC losses from topsoil losses estimated for the LCI step in the resource depletion impact pathway by multiplying topsoil losses with the percentage of topsoil OC content, which must be reported in the inventory. The percentage of SOC can be determined from SOM, input data needed to estimate soil losses using USLE.

The soil unit where the activity is developed should also be included in the LCI, using the most recent FAO classification (Harmonized World Soil Database, HWSD, FAO/UNESCO/ISRIC 1990), which distinguishes 28 different soil units (table 14), each with harmonized soil parameters (pH, SOC content and textural class, among others).

As a rough estimate of the SOC of each soil unit, we determined the content of the 28 soil units of the HWSD (table 14). This was done by averaging topsoil OC content of over 16,000 soil mapping units in this database, which holds, on a resolution of 30 arc-seconds (approximately $1 \times 1\ km^2$), information on selected soil parameters of soil units in the entire land area of the world.

The 28 soil units were grouped into seven major categories (table 14). In 24 out of 28 soil units there was less than 2% SOC, the threshold considered by the European Commission for defining soils in phase of pre-desertification (COM 2002), and in 12 out of 28 less than 1% SOC.

As for the impact assessment of resource depletion, information on the type and intensity of land use, area size (m^2), duration (y) and location of the occupation should be recorded in the LCI.

Table 14 Topsoil organic carbon (% weight) of the 28 soil units of the Soil map of the world (FAO/UNESCO/ISRIC 1990) and linear equations used in the impact model. Soil units within the same soil category are arranged in increasing SOC content. The LCI is expressed as gC $m^{-2}y^{-1}$ in the $\%NPP_0$ depletion equations.

Soil units HWSD	Topsoil organic carbon (% weight)	Equation aLCI+b ($\%NPP_0$ depletion)
Gypsisols Arenosols Calcisols Solonchaks	0 - < 0.5	$4.0906 \times LCI+2.6568$ for $LCI < 23.80$ $gCm^{-2}y^{-1}$ 100 for $LCI \geq 23.80$ $gCm^{-2}y^{-1}$
Lixisols Luvisols Solonetz Plinthosols Planosols Fluvisols Regosols Leptosols	0.5 - < 1.0	$1.9583 \times LCI+2.6568$ for $LCI < 49.71$ $gCm^{-2}y^{-1}$ 100 for $LCI \geq 49.71$ $gCm^{-2}y^{-1}$
Acrisols Vertisols Cambisols Anthrosols Kastanozems Ferralsols	1.0 - < 1.5	$1.3157 \times LCI+2.6568$ for $LCI < 73.99$ $gCm^{-2}y^{-1}$ 100 for $LCI \geq 73.99$ $gCm^{-2}y^{-1}$
Greyzems Podzoluvisols Alisols Nitisols Phaeozems Chernozems	1.5 - < 2.0	$0.8768 \times LCI+2.6568$ for $LCI < 111.03$ $gCm^{-2}y^{-1}$ 100 for $LCI \geq 111.03$ $gCm^{-2}y^{-1}$
Gleysols Podzols	2.0 - < 2.5	$0.6885 \times LCI+2.6568$ for $LCI < 141.39$ $gCm^{-2}y^{-1}$ 100 for $LCI \geq 141.39$ $gCm^{-2}y^{-1}$
Andosols	4.86	$0.3137 \times LCI+2.6568$ for $LCI < 310.31$ $gCm^{-2}y^{-1}$ 100 for $LCI \geq 310.31$ $gCm^{-2}y^{-1}$
Histosols	34.60	$0.0441 \times LCI+2.6568$ for $LCI < 2207.33$ $gCm^{-2}y^{-1}$ 100 for $LCI \geq 2207.33$ $gCm^{-2}y^{-1}$

Impact assessment model

When modelling damage to natural ecosystems, biodiversity is the most common indicator implemented in endpoint LCA methodologies (e.g., EI99, ReCiPe). In these methods, the effects on biodiversity are quantified using the Potentially Disappeared Fraction of vascular plant species concept (PDF, Koellner 2000). However, according to the ILCD (JRC 2010), function-related parameters, such as the biomass production of the ecosystem, might also be good endpoint indicators.

A very limited number of studies have focused on accounting for current or potential NPP losses caused by soil erosion, and most estimate productivity losses according to qualitative degrees of erosion (light/slight, moderate, strong/severe, extreme/very extreme erosion) at the local or regional level (Mann et al. 2002; Mokma and Sietz 1992). We assigned to each qualitative degree of erosion an average soil and SOC losses, which in turns correspond to a percentage loss of NPP_0 (table 15). The values were modelled by linear regression (R^2=0.93) for each soil unit to give a function for each soil unit group (table 14).

Table 15 Productivity losses (% NPP_0) based on soil erosion (t ha^{-1} y^{-1}).

Literature	Description	Soil loss range (t ha^{-1} y^{-1})	Average soil loss (t ha^{-1} y^{-1})	SOC losses	NPP depletion (% NPP_0)
Zika and Erb (2009)	light/slight	≤ 2[a]	1		5[c]
	moderate	2-12[b]	7		18 [c]
	strong/severe	12-25 [b]	18.5	Function	38 [c]
	extreme/very extreme	> 25 [b]	40	of the	63 [c]
FAO/UNEP (1984)	light/slight	≤ 2	1	topsoil	1[c]
	moderate	2-12	7	OC	15 [c]
	strong/severe	12-25	18.5	content of	35 [c]
	extreme/very extreme	> 25	40	each soil	75 [c]
Dregne and Chou (1992)	light/slight	≤ 2	1	unit (table	1[c,d]
	moderate	2-12	7	14)	10 [c,d]
	strong/severe	12-25	18.5		25 [c,d]
	extreme/very extreme	> 25	40		50 [c,d]

[a] Soil formation ranges from 0.5 to 1 t ha^{-1} y^{-1} (Mann et al. 2002). Productivity losses occur where soil erosion is higher than soil formation. The first intensity category distinguished by Basic et al. (2004) is 2 t ha^{-1} y^{-1}.
[b] Thresholds used in Núñez et al. (2010).
[c] Choice of the minimum estimate of NPP_0 losses, as the reported productivity losses are due to soil degradation, which includes factors other than water erosion.
[d] Degradation in croplands.

In the impact assessment model, the inventory flow (i.e., SOC losses) should be used as an input parameter in the equation in table 14 for the soil unit where the land use activity is developed. Note that for SOC losses equal to or above a predetermined threshold for each soil unit group, NPP_0 is completely lost and the soil is unlikely to be able to recover. Soils with low SOC content, such as those in arid and semi-arid areas, are less resilient than soils rich in SOC, usually in wet regions. These SOC loss thresholds are equivalent to a soil depletion of 63.9 t $ha^{-1}y^{-1}$, which means an extreme/very extreme soil loss. Most agricultural land in the world lose soil at rates between 13 and 40 t $ha^{-1}y^{-1}$ (Pimentel and Kounang 1998) and losses of more than 100 t $ha^{-1}y^{-1}$ only occur in extreme events (Morgan 1992).

The effects of soil erosion on ecosystem quality (ΔEQ) are expressed as follows with units of NPPD (potential net primary production depletion) ranging from 0 to 1per unit of area and time of land occupation:

$$\Delta EQ=\frac{aLCI+b}{100}\times\frac{NPP_{0,i}}{NPP_{0,ref}}=NPPD\ m^2y \qquad (9)$$

This endpoint model combines the inventory flow (aLCI+b, i.e., SOC losses transformed into $\%NPP_0$ losses) with NPP_0 values spatially resolved for each grid cell ($NPP_{0,i}$) at 5 arc-minutes (approximately 10×10 km^2, Haberl et al. 2007). Normalisation was with an NPP_0 value corresponding to that of the ecosystem with the highest biotic productivity worldwide as an objective reference ($NPP_{0,\ ref}$ = 1496 g C $m^{-2}y^{-1}$). As for the resource impact assessment, regionalisation was at the grid cell level, without aggregating values on broader scales.

The lower the value in the ecosystem quality damage category, the less harmful the system is to the environment. Note that, for complete losses of NPP_0 (aLCI+b=100), damages only depend on the $NPP_{0,i}/ NPP_{0,ref}$ ratio. The highest value has been allocated to the most productive soils, as richer and scarcer ecosystems are generally more protected than those which are poorer and more abundant. The most productive lands can also be used for a greater diversity of purposes.

5.2.3 Human health impact pathway

The human health AoP is concerned with quantifying the changes in both mortality and morbidity caused by various types of stressors (impact pathway 3 in figure 17). One main soil erosion-related impact pathway for human health was identified: soil erosion affects productivity of agricultural and pasture lands, leading to a reduction in food availability. This, in turn, results in one of two scenarios, depending on the regional context: (1) increase in malnutrition or undernutrition in the so-called deficiency scenarios (i.e., developing countries) or (2) food importation or changes in food production in the so-called compensation scenarios (i.e., wealthy countries), using the same terminology agreed for water use in LCA (Bayart et al. 2010). Deficiency scenarios are unable to adapt to productivity losses and this generates an impact on human health, which results in a loss of quality of life or longevity, usually measured in endpoint methods with the unit of DALYs. In contrast, compensation scenarios are wealthy enough to offset a lack of food, so human health impacts due to soil erosion are avoided. Many regional socio-economic parameters may influence definition as a compensation or deficiency scenario, such as the gross domestic product (GDP), the percentage of malnutrition and the human development index. The relationship between soil erosion and human health damages is highly complex and dependent on many regional conditions difficult to reflect in the LCA methodology. We therefore focused on impacts on natural resources and natural environment, leaving the human health area of protection for future research.

5.2.4 Case study on energy crops

Water erosion is one of the main causes of land degradation in Spain, and a considerable threat to maintaining agricultural productivity (EEA 2005). An increase in energy crop production is forecasted in Spain (EEA 2006), so, given the effect of agriculture on the environment, it is necessary to assess the environmental cost of this alternative. To illustrate the outlined method, we analyzed the environmental impacts of growing five three-year agricultural rotations, two of them with energy crops, on the erosion regulation potential ecosystem service.

Study area

The soil erosion and environmental impacts were estimated in 120 agricultural plots covering the main Spanish water basins (figure 18). The area has a Mediterranean climate which is characterised by mild temperatures (10-19ºC of annual average) and two periods of maximum precipitation, one in spring and one in autumn. Autumn is when higher intensity rainfalls are recorded, likely to have a higher impact on the erosion processes (Usón and Ramos 2001).

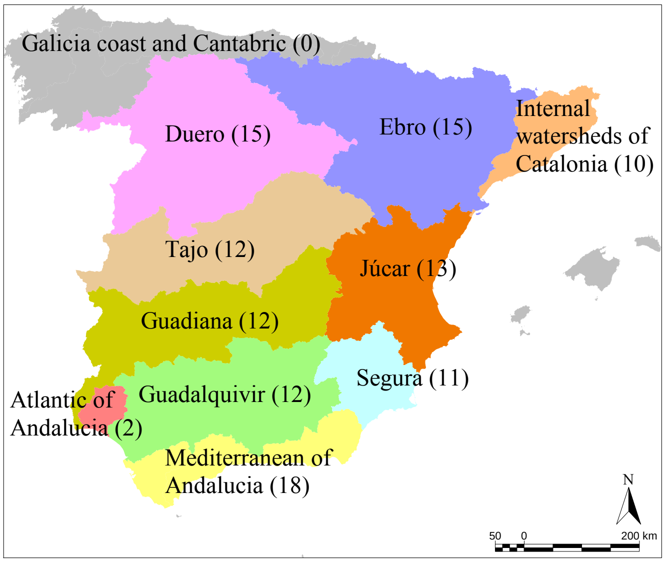


Figure 18 Geographical distribution of the plots in the Spanish water basins. The number in brackets indicates the quantity of plots studied within the watershed.

Life cycle inventory data

Data for the erosion-related LCI was gathered from several sources:

- To calculate soil losses with the USLE equation, we used national meteorological institute rainfall statistics for the period 1960-1990, specified in (MARM 2010) at a town resolution level. Raw data of the edaphic and geographic properties of the plots, namely soil texture, soil structure, soil organic matter content, soil permeability and soil slope gradient, was from the Spanish Soils Edaphic Properties database (Trueba et al. 2000) or measured specially for a current project studying the viability of energy crops in Spain (SSP On Cultivos, http://www.oncultivos.es). We assumed a common slope length of 100 metres for all the plots and no support practices such as terracing. The method applied to work out the soil protection provided by the vegetative cover of crop rotations and management practices (C-factor of USLE) is given in chapter 4.

- Soil types were identified according to the classification of FAO/UNESCO/ISRIC (1990) from FAO/IIASA/ISRIC/ISSCAS/JRC (2009).

- The georeferenced location of each plot was from (Trueba et al. 2000) or calculated in the case of the plots from the SSP On Cultivos project.

A land occupation of $1m^2$ during the three years of a complete crop rotation system was considered.

The five crop production systems assessed were rotations with food and energy purposes. Three of these were traditional rainfed rotations of annual crops grown in the Mediterranean region: i) winter barley-winter wheat-rye, ii) winter barley-winter wheat-pea, and iii) winter barley-winter wheat-unseeded fallow. Another was a rainfed rotation where a bioenergy crop was introduced: iv) winter barley-winter wheat-oilseed rape; and finally, a deficit-irrigated short rotation coppice of a perennial crop: v) poplar-poplar-polar. All are low-input systems, as the economic income from non-irrigated and bioenergy agriculture is low.

5.3 Results

5.3.1 LCIA characterisation factors

Regional characterisation factors for the resource depletion and ecosystem quality impact pathways of the erosion regulation potential ecosystem service are shown in figure 18. For resource depletion, the ratio is $((SD_{ref.} - SD_{i})/ SD_{ref})*ECEC_{transformity\ soil}$. Lower soil depths, and therefore higher damage factors, are found at high northern latitudes and over wide areas of Asia. For ecosystem quality, figure 19 shows the $NPP_{0,\ i}/NPP_{0,\ ref.}$ ratio. Higher biomass productivities, so damage factors, are found in biodiversity hotspots, such as the Mesoamerican forests in Central America, the Amazon rainforest and the Indo-Malayan archipelago. For both the resource depletion and ecosystem quality indicators, low values of CF indicate less environmental impacts.

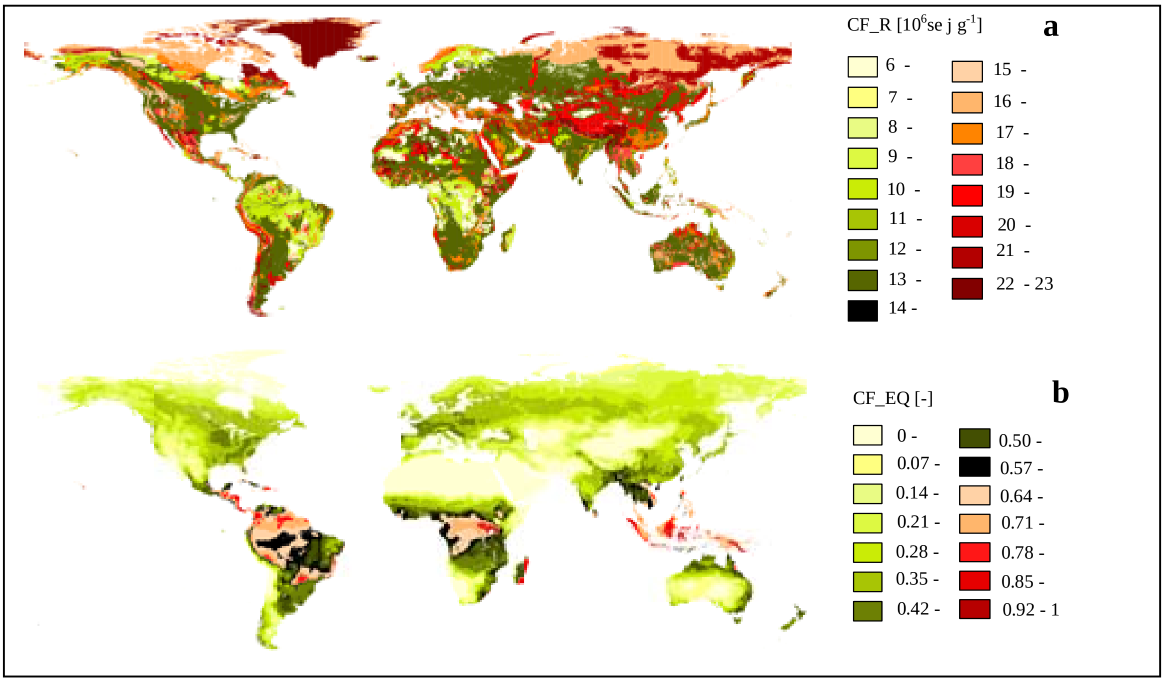


Figure 19 Characterisation factors for (a) resource depletion and (b) ecosystem quality.

5.3.2 Case study on energy crops

There was a great variability of soil and SOC losses in the LCI depending on the climatic and edaphic conditions of each water basin.

The crop rotation with the greatest erosion rate (table 16) is where the field is laid to fallow in the last year (B-W-F). These soil losses were around 10 times higher than the crop rotation with the lowest erosion rate, the energy profitable poplar short-forestry rotation (PP-PP-PP). For the cereal (B-W-R), pulse (B-W-P) and the energy (B-W-OR) annual crop rotation systems, similar soil losses were recorded, with rates about 40% lower than for the fallow rotation. Similar trends were found in the LCIA.

The hydrological Spanish basins with the lowest water erosion rates and environmental damages, and therefore most appropriate for growing poplar and oilseed rape energy crop rotations, are the Duero (northern Spain) and the Guadiana (central Spain), while those with the highest water erosion rates and environmental impacts are the internal watersheds of Catalonia (northeast Spain) and the Júcar (eastern Spain) basins.

Table 16 Life cycle inventory and life cycle impact assessment results per m^{2}y of land occupation.

	Internal watersheds of Catalonia	Ebro	Duero	Júcar	Tajo	Guadiana	Segura	Guadalquivir	Mediterranean of Andalucia	Atlantic of Andalucia
LCI – soil erosion [kg m^{-2}y^{-1}]										
B-W-R[a]	1.58	0.43	0.27	1.13	0.56	0.31	0.39	0.68	0.69	0.59
B-W-P[b]	1.63	0.45	0.29	1.16	0.58	0.31	0.39	0.68	0.68	0.58
B-W-F[c]	2.44	0.70	0.46	1.78	0.90	0.50	0.60	1.07	1.01	0.85
B-W-OR(*)[d, f]	1.51	0.41	0.24	1.03	0.54	0.28	0.38	0.63	0.63	0.56
PP(*)-PP(*)-PP(*)[e, f]	0.28	0.09	0.05	0.20	0.10	0.06	0.07	0.11	0.09	0.08
Resources [sej m^{2}y]										
B-W-R	2.5 E+10	6.4 E+09	4.1 E+09	1.9 E+10	9.0 E+09	5.0 E+09	5.8 E+09	9.9 E+09	9.7 E+09	9.9 E+09
B-W-P	2.6 E+10	6.6 E+09	4.4 E+09	2.0 E+10	9.3 E+09	5.2 E+09	5.8 E+09	9.8 E+09	9.7 E+09	9.9 E+09
B-W-F	3.9 E+10	1.0 E+10	7.0 E+09	3.0 E+10	1.4 E+10	8.2 E+09	9.2 E+09	1.6 E+10	1.5 E+10	1.4 E+10
B-W-OR(*)	2.4 E+10	6.0 E+09	3.6 E+09	1.8 E+10	8.7 E+09	4.6 E+09	5.6 E+09	9.2 E+09	9.2 E+09	9.4 E+09
PP(*)-PP(*)-PP(*)	4.5 E+09	1.3 E+09	7.5 E+08	3.4 E+09	1.6 E+09	9.0 E+08	9.7 E+08	1.6 E+09	1.4 E+09	1.3 E+09
Ecosystem quality [NPPD m^{2}y]										
B-W-R	0.12	0.03	0.02	0.07	0.04	0.03	0.02	0.05	0.04	0.05
B-W-P	0.12	0.04	0.02	0.07	0.04	0.03	0.02	0.05	0.04	0.05
B-W-F	0.18	0.05	0.03	0.11	0.06	0.04	0.03	0.07	0.06	0.07
B-W-OR(*)	0.11	0.03	0.02	0.07	0.04	0.02	0.02	0.05	0.04	0.05
PP(*)-PP(*)-PP(*)	0.03	0.02	0.01	0.02	0.01	0.01	0.01	0.02	0.02	0.02

[a] winter barley-winter wheat-rye; [b] winter barley-winter wheat-pea; [c] winter barley-winter wheat-unseeded fallow; [d] winter barley-winter wheat-oilseed rape; [e] poplar-poplar-poplar.

[f] Asterisks indicate crops for energy use.

5.4 Discussion

5.4.1 Soil erosion impact assessment model

Analysis of the method followed the general evaluation criteria and specific sub-criteria relevant for land use impacts of the ILCD handbook (JRC 2010). The aim of the analysis was to address model uncertainty qualitatively, to facilitate comparison with other soil erosion impact assessment models and to identify strengths and weaknesses of our method.

Completeness of scope

Two relevant impact pathways leading to the AoP natural resources and natural environment were addressed at the endpoint level. A general description of the impact mechanisms linking human interventions to the human health AoP is provided, but the impact assessment model still needs to be developed. The methods used by Motoshita et al. (2011) and Pfister et al. (2009), in which human damages due to water consumption are partially covered, are a helpful starting point to model soil erosion-related human health impacts. We express ecosystem damages in growth-based units (NPPD), while the majority of endpoint methods quantify ecosystem impacts in PDF (Koellner 2000). A significant correlation has been found (Pfister et al. 2009) between vascular plant species biodiversity and net primary productivity of the actual vegetation, which led the authors to consider NPP as a proxy for ecosystem quality. The same proxy could be applied to transform NPPD to PDF, taking into consideration that we used the potential instead of the actual vegetation. Globally, the NPP_0 to NPP_{act} ratio is between 0.9-1.2 in 81% of the terrestrial area. Similarly, for the indicator of resource depletion, we used exergy units, as in (Bösch et al. 2007), while most endpoint methods assess damages on abiotic resources with energy units (MJ). In reality, any energy measure to reconstruct a renewable resource such as soil lacks meaning. To allow for comparisons with other methods and impact categories, energy values can be transformed into exergies using the resource specific ECEC transformity value. Doing so, results can be compared to or aggregated with surplus exergy demands of energy-carriers (e.g., crude oil: 9.07E+04 sej g^{-1}) as well as non-energetic resources (e.g., water supply: 2.03E+05 sej g^{-1}). We assumed an ECEC transformity value of 2.39E+07 sej g^{-1} soil loss (Odum 1996). This value can be substituted by another conversion exergy ratio when more up-to-date information is available.

The characterisation model and factors are globally applicable and spatially defined, taking into account the ecosystem biomass productivity and soil characteristics at grid level. How to aggregate these factors on a wider scale (e.g., ecoregions, land cover, water basins) is an unresolved complex issue, due to the huge variability of soil types even at the landscape scale. This variability of soil types makes it difficult to identify a standardised approach to address spatial differentiation.

Environmental relevance

The method adequately assesses land occupation impacts (i.e., the use of a land area for a specific human purpose) of any type of human activity, but discounts impacts due to land transformation (i.e., change of a land area to make it suitable for a specific use). For occupation, it is debatable what period of occupation should be considered in the agricultural LCA inventory (i.e., duration of the crop, duration of the crop plus fallow period, crop rotation). For transformation, the choice of the reference situation to measure the magnitude of the change as well as the time needed to recover this reference situation are two fundamental issues to be agreed upon to properly include impacts from transformation. So far, the most applied reference situation is the so-called potential natural vegetation after land occupation, which differs from the natural situation because nature rarely recovers to its original after being disturbed.

Of the overall cause-effect chain for land use (figure 17), we focused on the soil erosion impact mechanism by quantifying changes in the topsoil formation/preservation and changes in the NPP_0 due to altered soil function. For a complete evaluation of land use impacts, the soil erosion assessment must be complemented with indicators that consider effects on climate change, biodiversity loss, water and nutrient regulation and unique landscapes.

Scientific robustness and certainty

The two indicators reflect the cause-effect chain from the interventions to the latest environmental damages. For resources, the impact can be expressed at the midpoint level if soil losses of the LCI are weighted by the available reference-corrected soil depth, without further transformation to exergy units. The cause-effect chain for ecosystem quality directly links soil erosion with the endpoint indicator (depletion of NPP_0), which can also be seen as an indicator at the midpoint level.

A linear regression equation relating SOC losses and NPP_0 losses was used for the ecosystem quality impact assessment model. This was used instead of other non-linear models with high coefficient of determination because there is insufficient knowledge of the type of relation between the two variables and the possible interference of other variables.

The geographical differentiation of the model has good potential for being improved and further developed when more detailed global maps of soil properties and biomass productivity of ecosystems become available. The current large scale maps do not fully resolve the real diversity of soil and ecosystems.

The indicator for ecosystem quality can be partially verified against monitoring data, measuring the loss of NPP at different degrees of soil erosion. Note that we used potential NPP, so the model can be tested in areas with potential or near potential vegetation. The indicator for resources cannot be verified (sej), as surplus exergy is an abstract concept.

The only measurement of statistical uncertainty was through the correlation coefficient R^2 in the SOC losses *vs* NPP_0 depletion linear regression analysis. Model uncertainties were qualitatively evaluated with the set of criteria listed in JRC (2010), although without using the scoring procedure to evaluate the model agreement against each criterion (i.e., scores from A-full compliance to E-no compliance). Other types of uncertainties are still to be estimated. An important statistical uncertainty of the LCI and LCIA results to be reported is that arising from the type of spatial aggregation applied.

Documentation, transparency and reproducibility

Documentation used for the model is published and readily accessible. The maps used to derive characterisation factors are available online. This availability of input data allows for third parties to further develop and improve the impact factors and the model.

Applicability

The characterisation factors are applicable for general LCA practitioners. Incorporation in current LCA software will require adaptation of the software to tackle spatially explicit data in GIS format. Estimations of the soil erosion needed as LCI data and measured with USLE require some knowledge of soil sciences and experience in the application of the equation.

Stakeholder acceptance

To be demonstrated.

5.4.2 Case study on energy crops

The results of the case study showed that the implementation of energy crop rotation systems in Spain has the potential to reduce erosion rates and related environmental impacts, compared to traditional crop rotations. The rotation system with fallow (B-W-F) had the highest soil losses because of the lack of live plant or residue cover throughout the third year. Bare soil causes the detachment of soil particles with the impact of rain on the topsoil, which is then lost in runoff. The cereal (B-W-R), pulse (B-W-P) and the energy (B-W-OR) annual crop rotation systems had similar soil losses because they have similar sowing and harvesting dates, and are without vegetative cover for the same period of the year. Soil losses from the poplar rotation (PP-PP-PP) are negligible, as the soil is protected with adventitious vegetation after the first months of crop planting.

Most cropping systems and watersheds recorded soil losses between 0.4 and 1.1 kg $m^{-2}y^{-1}$. These are tolerance values to maintain productivity but not to avoid the loss of soil. Other publications have reiterated the importance of erosion control (Cerdà et al. 2009; Kuderna 2004), considering these tolerances excessively high and inappropriate, especially taking into account that natural soil formation ranges from 0.05 to 0.10 kg $m^{-2}y^{-1}$ (Mann et al. 2002).

Both soil and organic carbon losses and impact factors vary as a function of location, leading to high differences in the environmental damages from soil erosion in different water basins. The average results are highly variable within a water basin due to the disparity of soils, climates and ecosystem biomass productivities (see the supporting information, at the end of this chapter). There is a need for statistical analyses combining basins and geospatial features to show to what extent the average results can be extrapolated across the watershed. Aggregation at the water basin level was not specific enough to reflect a common trend.

Apart from taking into account environmental impacts from soil erosion, other impact category indicators should be also accounted for in the selection of the most adequate watersheds for growing energy crops in Spain. For water consumption and environmental damages, it has been reported (chapter 2) that the most suitable locations for energy crop rotations are basins in northeast Spain, while they should not be cultivated in some southeast basins. According to these results, there is not a specific water basin capable of minimizing both water consumption and soil erosion impacts at the same time.

5.5 Supporting information

Table 7SI Life cycle inventory results – soil erosion (kg $m^{-2}y^{-1}$)

Table 8SI Life cycle inventory results – SOC losses (g $m^{-2}y^{-1}$)

Table 9SI Life cycle impact assessment results – Resource damages (sej m^2y)

Table 10SI Life cycle impact assessment results – Ecosystem quality damages (NPPD m^2y)

Table 7SI Life cycle inventory results – soil erosion (kg $m^{-2}y^{-1}$)

River basin	Subbasin	B-W-R	B-W-P	B-W-F	B-W-OR$^{(*)a}$	PP$^{(*)}$-PP$^{(*)}$-PP$^{(*)}$
Internal watersheds of Catalonia		**1.58**	**1.63**	**2.44**	**1.51**	**0.28**
	Besòs, Anoia	1.97	2.02	2.92	1.88	0.33
	Tordera	1.87	1.93	2.97	1.78	0.35
	Ter	0.62	0.64	0.96	0.58	0.10
	Muga, Fluvià	1.87	1.92	2.93	1.79	0.35
Ebro		**0.43**	**0.45**	**0.70**	**0.41**	**0.09**
	Ebro, Segre	0.43	0.45	0.70	0.41	0.09
Duero		**0.27**	**0.29**	**0.46**	**0.24**	**0.05**
	Duero, Pisuerga	0.27	0.29	0.46	0.24	0.05
Júcar		**1.13**	**1.16**	**1.78**	**1.03**	**0.20**
	Cenia, Maestrazgo	1.36	1.44	2.26	1.28	0.27
	Turia	1.71	1.67	2.52	1.52	0.29
	Júcar, Marina Baja	0.62	0.63	1.00	0.58	0.11
	Marina Alta	0.84	0.88	1.32	0.74	0.14
Tajo		**0.56**	**0.58**	**0.90**	**0.54**	**0.10**
	Tajo, Alagón	0.56	0.58	0.90	0.54	0.10
Guadiana		**0.31**	**0.31**	**0.50**	**0.28**	**0.06**
	Guadiana	0.31	0.31	0.50	0.28	0.06

River basin	Subbasin	B-W-R	B-W-P	B-W-F	B-W-OR(*)[a]	PP(*)-PP(*)-PP(*)
Segura		**0.39**	**0.39**	**0.60**	**0.38**	**0.07**
	Mar Menor	0.27	0.27	0.38	0.25	0.03
	Segura, Guadalentín	0.51	0.51	0.82	0.50	0.10
Guadalquivir		**0.68**	**0.68**	**1.07**	**0.63**	**0.11**
	Guadalquivir	0.68	0.68	1.07	0.63	0.11
Mediterranean of Andalucia		**0.69**	**0.68**	**1.01**	**0.63**	**0.09**
	Almanzora	0.24	0.24	0.38	0.23	0.04
	Andarax, Adra	0.38	0.37	0.60	0.35	0.06
	Gudalfeo	0.16	0.16	0.25	0.15	0.02
	Guadalmedina, Guadalhorce	1.32	1.31	1.93	1.19	0.18
	Turón, Guadalteba	1.11	1.10	1.62	1.04	0.15
	Gudalete, Barbate	0.90	0.89	1.30	0.81	0.11
Atlantic of Andalucia		**0.59**	**0.58**	**0.85**	**0.56**	**0.08**
	Tinto, Odiel	0.59	0.58	0.85	0.56	0.08

[a] Asterisks indicate crops for energy use.

Table 8SI Life cycle inventory results – SOC losses (g $m^{-2}y^{-1}$)

River basin	Subbasin	B-W-R	B-W-P	B-W-F	B-W-OR$^{(*)a}$	PP$^{(*)}$-PP$^{(*)}$-PP$^{(*)}$
Internal watersheds of Catalonia		**15.2**	**15.6**	**23.4**	**14.5**	**2.7**
	Besòs, Anoia	18.9	19.3	27.9	18.0	3.2
	Tordera	16.9	17.5	26.9	16.1	3.2
	Ter	5.5	5.6	8.4	5.1	0.9
	Muga, Fluvià	19.3	19.9	30.2	18.6	3.6
Ebro		**4.4**	**4.6**	**7.1**	**4.1**	**0.9**
	Ebro, Segre	4.4	4.6	7.1	4.1	0.9
Duero		**2.7**	**2.9**	**4.6**	**2.3**	**0.5**
	Duero, Pisuerga	2.7	2.9	4.6	2.3	0.5
Júcar		**11.2**	**11.4**	**17.5**	**10.2**	**2.0**
	Cenia, Maestrazgo	13.2	14.0	21.9	12.4	2.6
	Turia	16.2	15.8	23.9	14.4	2.7
	Júcar, Marina Baja	6.1	6.2	9.9	5.8	1.1
	Marina Alta	9.2	9.6	14.4	8.1	1.6
Tajo		**5.8**	**5.9**	**9.2**	**5.6**	**1.0**
	Tajo, Alagón	5.8	5.9	9.2	5.6	1.0
Guadiana		**3.0**	**3.1**	**4.9**	**2.8**	**0.6**
	Guadiana	3.0	3.1	4.9	2.8	0.6

River basin	Subbasin	B-W-R	B-W-P	B-W-F	B-W-OR(*)[a]	PP(*)-PP(*)-PP(*)
Segura		**4.2**	**4.2**	**6.5**	**4.0**	**0.7**
	Mar Menor	3.0	2.9	4.2	2.7	0.3
	Segura, Guadalentín	5.4	5.4	8.7	5.3	1.0
Guadalquivir		**6.9**	**6.8**	**10.8**	**6.4**	**1.1**
	Guadalquivir	6.9	6.8	10.8	6.4	1.1
Mediterranean of Andalucia		**7.0**	**6.9**	**10.4**	**6.4**	**1.0**
	Almanzora	1.7	1.7	2.8	1.6	0.3
	Andarax, Adra	3.3	3.3	5.3	3.1	0.6
	Gudalfeo	1.5	1.5	2.4	1.4	0.2
	Guadalmedina, Guadalhorce	13.8	13.7	20.2	12.5	1.9
	Turón, Guadalteba	12.1	12.0	17.7	11.4	1.7
	Gudalete, Barbate	9.4	9.3	13.7	8.5	1.1
Atlantic of Andalucia		**5.3**	**5.3**	**7.7**	**5.1**	**0.7**
	Tinto, Odiel	5.3	5.3	7.7	5.1	0.7

[a] Asterisks indicate crops for energy use.

Table 9SI Life cycle impact assessment results – Resource damages (sej m^{2}y)

River basin	Subbasin	B-W-R	B-W-P	B-W-F	B-W-OR$^{(*)a}$	PP$^{(*)}$-PP$^{(*)}$-PP$^{(*)}$
Internal watersheds of Catalonia		**2.5 E+10**	**2.6 E+10**	**3.9 E+10**	**2.4 E+10**	**4.5 E+09**
	Besòs, Anoia	3.2 E+10	3.3 E+10	4.8 E+10	3.1 E+10	5.5 E+09
	Tordera	2.7 E+10	2.8 E+10	4.3 E+10	2.6 E+10	5.1 E+09
	Ter	8.7 E+09	8.9 E+09	1.3 E+10	8.1 E+09	1.4 E+09
	Muga, Fluvià	3.2 E+10	3.3 E+10	5.0 E+10	3.1 E+10	5.9 E+09
Ebro		**6.4 E+09**	**6.6 E+09**	**1.0 E+10**	**6.0 E+09**	**1.3 E+09**
	Ebro, Segre	6.4 E+09	6.6 E+09	1.0 E+10	6.0 E+09	1.3 E+09
Duero		**4.1 E+09**	**4.4 E+09**	**7.0 E+09**	**3.6 E+09**	**7.5 E+08**
	Duero, Pisuerga	4.1 E+09	4.4 E+09	7.0 E+09	3.6 E+09	7.5 E+08
Júcar		**1.9 E+10**	**2.0 E+10**	**3.0 E+10**	**1.8 E+10**	**3.4 E+09**
	Cenia, Maestrazgo	2.2 E+10	2.3 E+10	3.6 E+10	2.1 E+10	4.3 E+09
	Turia	2.9 E+10	2.8 E+10	4.3 E+10	2.6 E+10	4.8 E+09
	Júcar, Marina Baja	1.0 E+10	1.1 E+10	1.7 E+10	9.8 E+09	1.9 E+09
	Marina Alta	1.5 E+10	1.6 E+10	2.3 E+10	1.3 E+10	2.5 E+09
Tajo		**9.0 E+09**	**9.3 E+09**	**1.4 E+10**	**8.7 E+09**	**1.6 E+09**
	Tajo, Alagón	9.0 E+09	9.3 E+09	1.4 E+10	8.7 E+09	1.6 E+09
Guadiana		**5.0 E+09**	**5.2 E+09**	**8.2 E+09**	**4.6 E+09**	**9.0 E+08**
	Guadiana	5.0 E+09	5.2 E+09	8.2 E+09	4.6 E+09	9.0 E+08

River basin	Subbasin	B-W-R	B-W-P	B-W-F	B-W-OR(*)[a]	PP(*)-PP(*)-PP(*)
Segura		**5.8 E+09**	**5.8 E+09**	**9.2 E+09**	**5.6 E+09**	**9.7 E+08**
	Mar Menor	3.8 E+09	3.8 E+09	5.4 E+09	3.5 E+09	4.3 E+08
	Segura, Guadalentín	7.8 E+09	7.8 E+09	1.3 E+10	7.7 E+09	1.5 E+09
Guadalquivir		**9.9 E+09**	**9.8 E+09**	**1.6 E+10**	**9.2 E+09**	**1.6 E+09**
	Guadalquivir	9.9 E+09	9.8 E+09	1.6 E+10	9.2 E+09	1.6 E+09
Mediterranean of Andalucia		**9.7 E+09**	**9.7 E+09**	**1.5 E+10**	**9.2E+09**	**1.4E+09**
	Almanzora	3.8 E+09	3.8 E+09	6.0 E+09	3.6 E+09	6.6 E+08
	Andarax, Adra	6.7 E+09	6.6 E+09	1.1 E+10	6.2 E+09	1.1 E+09
	Gudalfeo	2.8 E+09	2.8 E+09	4.3 E+09	2.6 E+09	4.3 E+08
	Guadalmedina, Guadalhorce	1.8 E+10	1.8 E+10	2.7 E+10	1.7 E+10	2.6 E+09
	Turón, Guadalteba	1.5 E+10	1.5 E+10	2.3 E+10	1.5 E+10	2.1 E+09
	Gudalete, Barbate	1.2 E+10	1.2 E+10	1.8 E+10	1.1 E+10	1.5 E+09
Atlantic of Andalucia		**9.9 E+09**	**9.9 E+09**	**1.4 E+10**	**9.4 E+09**	**1.3 E+09**
	Tinto, Odiel	9.9 E+09	9.9 E+09	1.4 E+10	9.4 E+09	1.3 E+09

[a] Asterisks indicate crops for energy use.

Table 10SI Life cycle impact assessment results – Ecosystem quality damages (NPPD m^2y)

River basin	Subbasin	B-W-R	B-W-P	B-W-F	B-W-OR$^{(*)a}$	PP$^{(*)}$-PP$^{(*)}$-PP$^{(*)}$
Internal watersheds of Catalonia		**0.12**	**0.12**	**0.18**	**0.11**	**0.03**
	Besòs, Anoia	0.15	0.15	0.22	0.14	0.03
	Tordera	0.14	0.15	0.22	0.14	0.04
	Ter	0.05	0.05	0.08	0.05	0.02
	Muga, Fluvià	0.12	0.13	0.19	0.12	0.03
Ebro		**0.03**	**0.04**	**0.05**	**0.03**	**0.02**
	Ebro, Segre	0.03	0.04	0.05	0.03	0.02
Duero		**0.02**	**0.02**	**0.03**	**0.02**	**0.01**
	Duero, Pisuerga	0.02	0.02	0.03	0.02	0.01
Júcar		**0.07**	**0.07**	**0.11**	**0.07**	**0.02**
	Cenia, Maestrazgo	0.10	0.10	0.16	0.09	0.03
	Turia	0.10	0.10	0.14	0.09	0.02
	Júcar, Marina Baja	0.05	0.05	0.07	0.04	0.02
	Marina Alta	0.04	0.04	0.06	0.04	0.01
Tajo		**0.04**	**0.04**	**0.06**	**0.04**	**0.01**
	Tajo, Alagón	0.04	0.04	0.06	0.04	0.01
Guadiana		**0.03**	**0.03**	**0.04**	**0.02**	**0.01**
	Guadiana	0.03	0.03	0.04	0.02	0.01

River basin	Subbasin	B-W-R	B-W-P	B-W-F	B-W-OR(*)[a]	PP(*)-PP(*)-PP(*)
Segura		**0.02**	**0.02**	**0.03**	**0.02**	**0.01**
	Mar Menor	0.01	0.01	0.02	0.01	0.01
	Segura, Guadalentín	0.03	0.03	0.04	0.03	0.01
Guadalquivir		**0.05**	**0.05**	**0.07**	**0.05**	**0.02**
	Guadalquivir	0.05	0.05	0.07	0.05	0.02
Mediterranean of Andalucia		**0.04**	**0.04**	**0.06**	**0.04**	**0.02**
	Almanzora	0.02	0.02	0.02	0.02	0.01
	Andarax, Adra	0.02	0.02	0.03	0.02	0.01
	Gudalfeo	0.01	0.01	0.02	0.01	0.01
	Guadalmedina, Guadalhorce	0.08	0.08	0.11	0.07	0.02
	Turón, Guadalteba	0.07	0.07	0.10	0.07	0.02
	Gudalete, Barbate	0.06	0.06	0.08	0.06	0.02
Atlantic of Andalucia		**0.05**	**0.05**	**0.07**	**0.05**	**0.02**
	Tinto, Odiel	0.05	0.05	0.07	0.05	0.02

[a] Asterisks indicate crops for energy use.

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Chapter 6.

Conclusions.

This chapter contains the overall conclusions of the thesis, based on the objectives outlined in the introduction (chapter 1) and the specific conclusions of chapters 2 to 5.

6.1 About the methodology

Improvement of the LCIA methods for water and land use

In this thesis, the life cycle assessment methodology was ameliorated with the development of assessment frameworks to incorporate water and land use environmental impacts. Improvements dealt with the following issues:

- For **water use**, the focus was on the impact assessment stage of the off-stream green water consumption. Many and important advances have been made in recent years towards the integration of water consumption in LCA. Particularly, special emphasis has been given to better report inflows and outflows in the inventory step and to develop methods to integrate the consumption of blue water. Thus, while prior to this thesis the matter of study has been centred on these topics, here **two screening frameworks to also measure impacts of green water consumption** were proposed. The first accounts for the difference in green water demand of the system under study (here agricultural crop rotations) and a reference system (here the potential natural vegetation in Spain and an agricultural crop rotation of reference in the country). The second is an additional indicator –the green water scarcity index– that supplements the first approach and that is designed to be used in LCA studies of agricultural and forestry systems, which are those that consume green water. By complementing the existing indicators of blue water with those of green water proposed in this thesis, LCA can provide a more comprehensive picture of the real impacts of water use.

- For **land use**, the focus was on developing impact assessment methods to include desertification and soil erosion impacts in LCA.

  For desertification, a multi-indicator approach of four variables –aridity, erosion, aquifer overexploitation and fire risk– was developed, and characterisation factors were proposed for the eight natural ecoregions with desertification risk. This method is applicable to calculate the desertification impact of any kind of human activity in arid areas. The framework proposed adds an innovative contribution to LCA, as this is an impact category not modelled prior to this thesis.

For soil erosion, a method that quantifies damages to soil resources and to ecosystems was developed. The method for calculating damage to ecosystems relates soil losses with soil organic carbon losses and the subsequent impacts on ecosystem biomass (NPP). This approach connects in one single indicator the ecosystem services of erosion regulation, carbon sequestration and biomass production, providing an integrative approach to quantify the effects of human activities on these ecosystem services.

These models were built using the best available digitized and georeferenced information at the global level. However, these global maps are currently resolved at too large scales, meaning they do not capture the large diversity of water and land use impacts at the local scale. Yet, the models would be improved and further developed when more detailed global maps become available.

Improvements in the LCI step for desertification and soil erosion

At the inventory level, there were two main contributions:

- **For desertification, the inventory flows to be collected** in order to apply the developed desertification impact assessment model were set up. Information relates to the four variables relevant for the desertification phenomenon –aridity, erosion, aquifer overexploitation and fire risk–, and the location, time span and area occupied for the activity.

- **For soil erosion, the development of a standardised method to determine USLE cover and management factors** (C-factor) for crops and management practices not yet calculated. This C-factor is indispensable to obtain, in the LCI step, estimates of soil erosion with the USLE equation. The method can be used to create a C-factor data set for different crops and locations in order to facilitate the incorporation of soil erosion impacts in LCA studies. The main strong points of the proposed procedure are that it is easy to apply, time-efficient and requires less extensive data that other available methods. The only data requirements are published literature, digital photographs of the crop, a user-friendly specialised software to calculate the vegetative cover of the digital photographs and publicly available rainfall records.

Simultaneous use of LCA and GIS

This thesis opened a brand-new research field regarding the methodological development of the regional impact categories in LCA. This new line of investigation focuses on the combination of LCA and GIS tools. Namely, in the thesis, **LCA-GIS coupling proved to be very helpful to develop methods to assess the environmental impacts of using water and land**. As observed, the quantity and quality of two resources that depend greatly on the spatial and temporal conditions of the location where the evaluated activity takes place. GIS facilitated the gathering of location-specific LCI data to complete the inventory table for the systems of study of the thesis. Besides, GIS facilitated the definition of spatially differentiated characterisation factors, which have proven to be a key issue to developing improved methodologies for the site-dependent impact categories of water and land use.

A lot of information that may be useful to model non-global impact categories in LCA has still not been released in GIS format. Also, the current accessible maps are in a very rough scale, which hampers the development of more precise local models. The release of free and more accurate maps will enable a better representation of local-dependent environmental impacts in LCA.

Selection of the reference system

The choice of one reference system (e.g., pristine vegetation before any land use, potential natural vegetation after environment recovery, most likely land use to be in a given location at the expense of the studied system) as the baseline scenario against which the land and water use impacts of a system under study can be compared is not evident. In fact, a scientific debate to come to an agreement regarding the most appropriate reference system is currently underway. In this thesis, **two reference situations, one agricultural** –crop rotations without energy crops– **and the other one natural** –potential natural vegetation–, were used. Both of them were apparently equally suitable. However, some shortcomings were identified when they were applied:

For the **water impact assessment**, **the concerns** to be highlighted are:

- Both the natural and the agricultural reference situations only consume green water –they are never irrigated–, and LCA still lacks an agreed and robust method accounting for this type of water consumption impacts. Therefore, **comparisons** of these baseline systems and the activities under study, as carried out in this thesis, **should be interpreted with caution until green water does not be soundly integrated in LCA**.

- **The appropriateness of using the potential natural vegetation** as the reference situation in areas that have been occupied with arable and pasture lands from a long time ago **is debatable** for agricultural LCAs.

For the **soil erosion assessment, the aspect** to be highlighted is:

- The net primary productivity of the potential natural vegetation (NPP_0) was used as a rough proxy for the NPP of the actual vegetation (NPP_{act}) to quantify the effects of soil erosion on current biomass productivity. However, the annual biomass production of the potential vegetation is not necessarily equivalent to the production potential of the current vegetation (NPP_0 to NPP_{act} ratio is between 0.9-1.2 in 81% of the terrestrial area). Land use does not necessarily reduce NPP. Irrigated land as well as intensively used agricultural areas can have a higher productivity than NPP_0. Also, the reduction of the NPP_{act} of cropland ecosystems might be due to the shortening of the vegetation period, without being related to soil erosion. These **alterations of NPP_0 relative to NPP_{act} should be considered if the potential natural vegetation is used as the reference system**.

6.2 About the systems of study: energy crops grown in Spain

Appropriateness of the energy crop rotation systems as the systems of study

The selection of energy crop rotations as the system of study to check the performance of the land and water LCA methods was very relevant. In recent years, annual and short forestry energy crop rotations have been promoted as an alternative to fossil fuels. However, little is currently known about the impacts on land and water resources of many crops used for energy purposes. **With this thesis, some insight into the existing trade-offs between energy**

production and impacts on water and land resources, **has been gained**. Due to the forecasted impacts that climate change could have on water resources, the water-energy relationship will be a keystone for sustainable energy production.

Site and temporal dependency of water and land use impacts of the energy crop rotations

Water consumption, soil erosion, and related environmental impacts of the studied crop rotations **varied greatly among water basins and also inside one specific water basin**. For example, taking the watersheds with the higher and the lower water consumption, the difference in the overall consumption per tonne yield (ET_c, m^3t^{-1}) for the rotation with oilseed rape (winter barley–winter wheat–oilseed rape) was around a rate of two. And the variation of water consumption within the most demanding watershed reached 18%. For this same oilseed rape crop rotation, soil losses were up to about six times superior in the watershed more prone to erosion, compared to that with the minimum losses. This disparity of results in the LCI is also supported by the LCIA results. The variation is due to the diversity of climates, soil types and landscape properties among locations. These results justify the importance of a regionalised assessment of water and land use, both in the LCI and the LCIA steps.

Green water and land use impacts of energy crop rotations versus reference systems

When green water consumption and soil erosion impacts of the energy crop rotations were compared to specific agricultural and natural reference systems in Spain, results indicated that **rotations with energy crops can maintain or even reduce the environmental impacts**.

As such, the agricultural reference system composed of cereals plus fallow (winter barley–winter wheat–fallow) led to a green water deprivation of up to 1200 m^3t^{-1} yield more than the energy rotation with the highest green water use efficiency, the short forestry rotation of poplar. Also, according to the green water scarcity index, the natural reference system (Mediterranean forest) was grown in some basins with up to 40% more severe aridity stress conditions than the studied energy crop rotations.

Results of the soil erosion impact assessment showed a similar trend. For example, there is a NPP depletion of ecosystems about 1.5 times higher for the agricultural reference rotation with cereals plus fallow (winter barley–winter wheat–fallow) than for the energy crop rotation with oilseed rape (winter barley–winter wheat–oilseed rape). This same oilseed rape rotation resulted in the same or very similar damages to ecosystems as other cereal (winter barley–winter wheat–rye) and pulse (winter barley–winter wheat–pea) rotations without energy crops.

Therefore, energy crops can be cultivated without putting further pressure on water and soil resources, whenever the environmental constraints of each locality are taken into account.

Suitable Spanish areas in relation to water use impacts

The **most appropriate locations** to cultivate the studied energy crop rotations to reduce water consumption and resulting ecological damages are water basins **in the northeast of Spain**, namely in some internal watersheds of Catalonia (figure 20a). There, compared to other watersheds, crops are grown without blue water scarcity (average WSI=0.2) and have more soil green water availability coming from rainfall in the warm months of spring and summer (average spring and summer rainfall about 300 mm), which contributed to saving irrigation water. On the contrary, **cultivation is not recommended in some southeastern watersheds** such as the Segura basin, due to the combination of high blue water scarcity (average WSI=1) and low soil moisture (average spring and summer rainfall about 110 mm).

Suitable Spanish areas in relation to soil erosion impacts

Looking at the soil erosion rates and impacts, the **watersheds more prone to soil erosion are located in the northeast** (internal watersheds of Catalonia) **and east** (some watersheds of the Júcar basin) of Spain (figure 20b), due to the high erosivity of rainfall characteristic of the Mediterranean climate in this area (2290 MJmm $h^{-1}ha^{-1}$). Special care in applying sustainable management practices should therefore be taken in these watersheds when the analysed energy crop rotations, and in general, any crop rotation, is cultivated. On the contrary, the **lowest erosion risk was for water basins located in northern** Spain (Duero basin) **and central Spain** (Guadiana), where the average erosivity of rainfall was about 770 MJmm $h^{-1}ha^{-1}$.

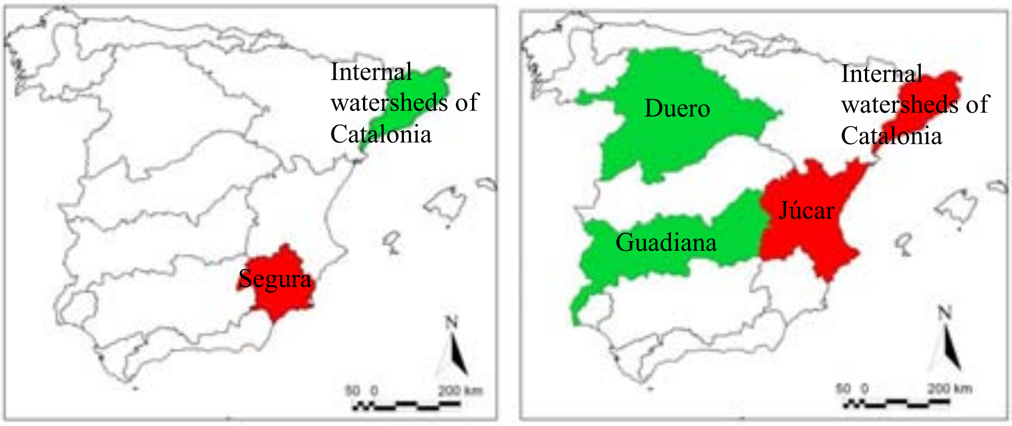

Figure 20 Location of the most (in green) and the least (in red) appropriate cultivation watersheds for a) water use environmental impacts, and b) soil erosion environmentalimpacts.

Trade-off between land and water use impacts

The holistic perspective given by the application, at the same time, of the impact categories land and water use has enabled the transfer of impacts from one category to the other to be revealed. This means that irrigated crops consume blue water resources while rainfed crops do not, but, in contrast, irrigated crops have a more environmentally efficient use of land per unit of product, with an average of 85% less ecosystem damage, at the nationwide scale. Also, the side effects are observed from one water basin to the other: impacts due to water scarcity are reduced at the expense of a higher amount and intensity of rainfall, which led to higher soil losses. Therefore, no water basin is capable of minimising water and soil erosion impacts simultaneously. In this regard, the case of the internal watersheds of Catalonia is worthy of mention. There, the most favourable and unfavourable scores for water and land use impacts, respectively, are found. All these results show that **indicators of land and water use should be jointly considered for a fair evaluation of the alternatives for growing energy crops, and in general, for growing any crop** in an environmentally friendly way.

Chapter 7.

Outlook.

Future lines of research that may follow this thesis are mentioned here. Some of them relate to one specific chapter of the dissertation, and others present general lines of action towards integrating spatial differentiation in the life cycle assessment methodology.

7.1 About the methodology

Further study on suitable functional units for agricultural LCA

This is a very important line of research in order to have the LCA methodology adapted to the study of agricultural systems. Two main questions need to be resolved:

- **Time span of the functional unit**. Traditionally, the **crop growing period** from sowing to harvest and the **yearly period** (crop growing period + fallow land) have been the time frames most used in the allocation of the environmental burdens of the system. However, lately, the **crop rotation period** has gained importance. This temporal boundary expansion enables the advantages of well-planned crop rotations to be taken into account, with positive consequences on soil physical-chemical and biological properties arising from specific crops grown in the same plot year after year. Even though this latter option would seem the most suitable one, the three given are actually in use. Similar problems arise when selecting an appropriate time span to compare perennial crops.

- **Service provided for the functional unit**, that is, in which one of the multiple functions of agriculture the functional unit should be expressed. As outlined in chapter 2.4.1, surface-based functional units do not really reflect the function of the crop/crop rotation, which are cultivated with the aim of obtaining an output (e.g., food, fibre). Yield and energy based functional units require comparing crops/crop rotations with strictly identical functions, which rarely occurs except for comparisons of the same crop or crop rotation. Therefore, an **adequate LCA comparison of alternative crop/crop rotations currently presents many troubles**.

Development of location-dependent characterisation factors for regional impact categories

Some developed spatial differentiated impact assessment methods give very accurate results within a specific region but they are imprecise outside this geographical boundary. Upcoming research in site-dependent LCIA models and characterisation factors should examine globally applicable methods. Current spatial differentiated characterisation factors are developed according to different types of boundaries (generally, political or biogeographical) and in different levels of detail (country, administrative region, water basin, ecozone, ecoregion, etc.). **A standardised approach to address the spatial differentiation over the different impact categories in a consistent way would be recommended**. Facing this challenge means identifying a common relevant scale among the impact categories, which is not an easy task. For example, an appropriate aggregation system for erosion resistance, biomass production, carbon sequestration and water purification impact categories would be soil type-based, as the performance of such ecosystem services depends on the soil ecological quality. However, aggregation per soil type may not be as relevant for biodiversity, where ecological-based units seem, *a priory*, to fit better.

Challenges to face regarding incorporating green water in LCA

Though a screening method to include green water consumption impacts has been proposed in the thesis, there are still many essential open questions to be solved for a consistent integration of green water in the LCA methodology. Some of these are:

- To clarify **what part of the water consumed by the plant is water from the soil and what part is water taken up from aquifers with deeper roots**, and if this root groundwater use is green or blue water. Solving this latter aspect is not straightforward, as there is an interaction between both kinds of water.

- **To investigate how to minimise problems related to the definition of spatial differentiated characterisation factors** for a type of water consumption which varies even at plot scale. Green water consumption strongly depends on many locally-related parameters (climate, soil properties, soil-water availability, land slope) as well as on the cultivated crop (type of crop, crop variety, length of growing periods). This high variability makes it more difficult to find a relevant and feasible level for the spatial aggregation of characterisation factors.

Challenges to face regarding including land transformation impacts in LCA

Most of the land use LCIA methods focus the attention on evaluating occupation impacts, while **a lot of research is still needed to account for transformation impacts** as well. The main important methodological gaps for the quantification of transformation impacts are:

- Definition of an adequate reference situation to compare the land quality
- Agreement on the relaxation time (time of recovery) needed to re-establish the reference state
- Specification of the relaxation rate at which the reference system is recovered

Calibration of the desertification impact assessment model

The scientific robustness, environmental relevance, applicability and reproducibility of the **desertification impact model** (chapter 3**) should be calibrated with case studies**. In this way, the desertification impact of growing several food and energy crops as well as a reference fallow land situation is currently under study. Two ecoregions, more specifically the Mediterranean and the tropical/subtropical desert, which have different desertification risks, are being compared.

Impact assessment methods for combining midpoint and endpoint indicators and harmonisation of endpoint indicator results with standard units

The use of endpoint impact assessment methods has been gradually increasing in LCA. Currently, there is a consensus on the need to merge midpoint and endpoint models in a consistent framework to combine the advantages of both concepts. One of the main strengths of the endpoint methods is that they group in only three AoP the midpoint level impact results of many impact categories. However, in practice, endpoint indicators contributing to one specific AoP are presently expressed with different units, which hinder the comparison between impact categories and the grouping at the damage level. For example, the use of PDF m^2yr to measure effects on biodiversity and the use of NPP m^2yr to measure effects on the soil erosion regulation ecosystem service prevents comparability and aggregation of damages to ecosystems. **Further research on endpoint methods should therefore tend to use common units**.

Another important point to be addressed to improve the consistency of endpoint methods is related to the **definition of easier to measure and certain transformation factors to pass from midpoint to endpoint indicators**, as for example, to pass from the midpoint water stress index to the endpoint damages to natural resources.

Calculation of uncertainties

The impact assessment method for soil erosion (chapter 5) is based on a linear relationship between soil organic carbon losses and loss of potential net primary productivity. Large uncertainties are associated with the quantification of NPP effects of soil erosion as well as with the linear correlation established between both variables. Similarly, the impact assessment method for desertification (chapter 3) is highly uncertain. It was built as a product of factors where the area of the ecoregion and the desertification impact result follows a logarithmic correlation. This area-desertification relation is inspired in the well-established area-species interaction for some groups of organisms. However, little is known about the validity of this principle for the desertification phenomenon. More research on these two relationships should be carried out to enhance the scientific soundness of the models. Apart from these **model uncertainties, statistical and decision rule uncertainties**, the latter related to value choices, **should be quantitatively or qualitatively addressed**.

7.2 Databases

Setup of new databases and homogenisation of the information

- Two main problems related to databases were identified: In many locations, there was a **lack of databases** with detailed information about variables that are essential to apply and validate the models proposed in the thesis. For instance, neither an USLE R-factor (rainfall and runoff factor) data set nor hydrological balances of aquifers are always available. Also, it is important to mention the need to **set up an USLE C-factor database for different crops and places** in order to include land erosion in LCA studies. Such a database would facilitate the estimation of soil losses with the USLE equation in the inventory phase.

- **Little homogenisation of the information of databases** for a particular type of data collected. Some of the inconsistencies detected are the use of several estimation models or measurement methods to calculate a specific variable, the application of incompatible systems of classification, and the different spatial and temporal spans of the data gathered. Two examples are the use of a multiplicity of ET_0 estimation methods or the uneven temporal periods for forest fires statistics.

These issues should be resolved by means of new studies, to obtain more detailed life cycle inventories and characterisation factors –if needed–, as well as to foster the combined use of LCA and GIS in the modelling of site-dependent environmental impact categories.

7.3 Software

Incorporation of GIS tools into LCA software

The use of GIS technologies has proven to be a very good tool complementing LCA, as GIS provides a spatially explicit interpretation of the location-generic LCA results. GIS facilitates the development of new and improved current inventory data and impact assessment methods. It also assists in the research of what might be the most appropriate aggregation level for the characterisation factors of regional impact categories. Therefore, the **integration of GIS tools into LCA software packages opens new perspectives in the frame of the life cycle assessment methodology** to spatially represent site and temporal specific impacts, such as land and water use, eutrophication or noise impacts.