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ENVIRONMENTAL MANAGEMENT IN AGRICULTURE BASED ON WATER AND CLIMATE CHANGE ASSESSMENTS : TOOLS FOR DECISIONS MAKING BY LIFE CYCLE APPROACH.

Maria José Amores Barrero

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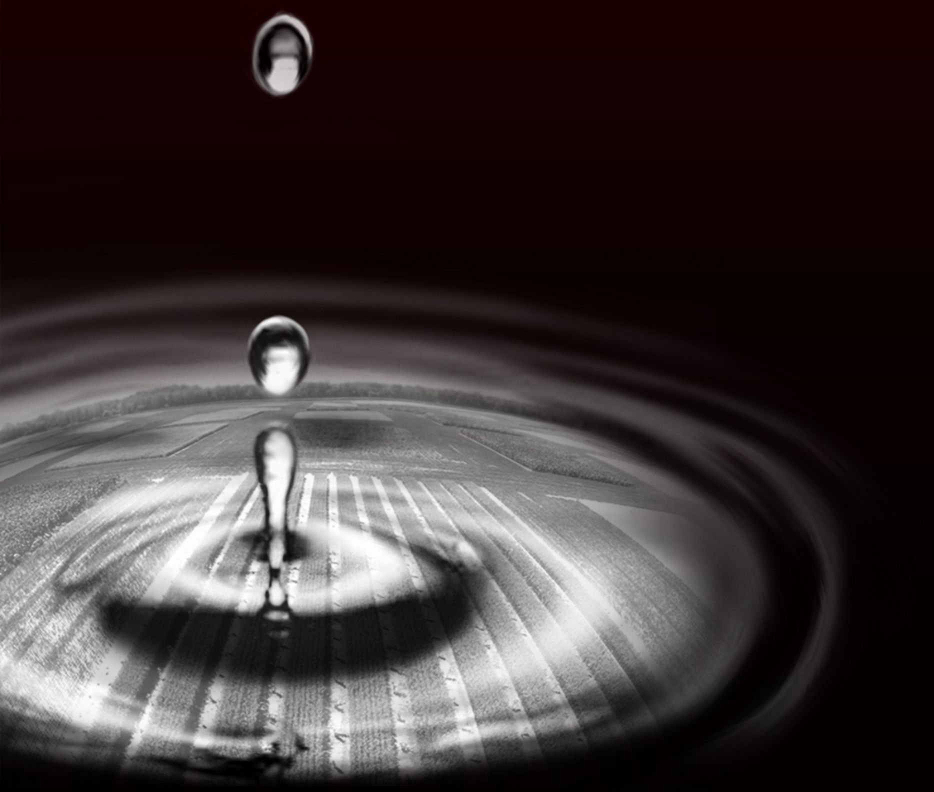

Environmental Management in Agriculture based on Water and Climate Change Assessments : Tools for Decision Making by Life Cycle Approach

Doctoral Thesis

Maria José Amores Barrero

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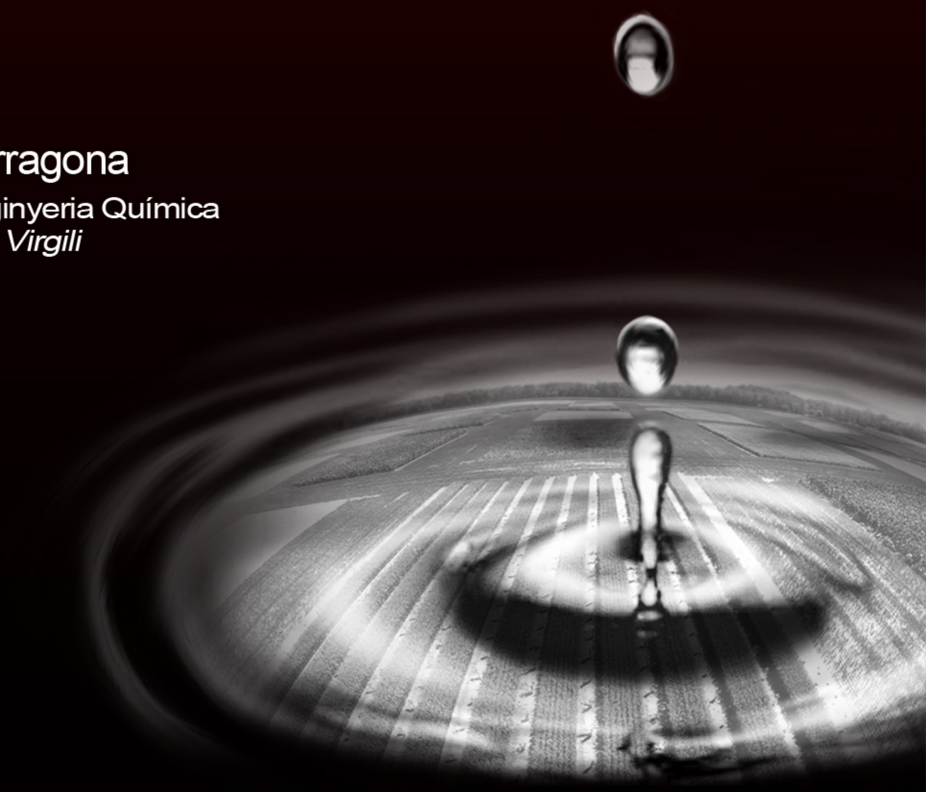

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ENVIRONMENTAL MANAGEMENT IN AGRICULTURE BASED ON WATER AND CLIMATE CHANGE ASSESSMENTS: TOOLS FOR DECISION MAKING BY LIFE CYCLE APPROACH

DOCTORAL THESIS

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The present thesis entitled "Environmental management in agriculture based on water and climate change assessments: tools for decision making by life cycle approach" was carried out by María José Amores Barrero at Environmental Management Group in Department of Chemical Engineering at the Universitat Rovira i Virgili (URV) under the supervision of Prof. Dr. Francesc Castells Piqué from the Department of Chemical Engineering at the URV and Dr. Assumpció Antón Vallejo, from the Environmental Horticulture Unit at the Institute of Agriculture and Food Research and Technology (IRTA) and the Department of Chemical Engineering at the Universitat Rovira i Virgili (URV). This doctoral thesis is submitted in fulfillment of the requirements for the International Mention of the PhD in Chemical, Environmental and Process Engineering.

Tarragona, April 29th of 2013

Supervisor of doctoral thesis	Co-supervisor of doctoral thesis
Dr. Francesc Castells Piqué	Dra. Assumpció Antón Vallejo

A Hugo y a mi familia.

Be the changes that you wish to see in the world.

Mahatma Gandhi

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List of acronyms, abbreviations and notation

ADP	Abiotic Depletion Potential
AGA	Anàlisis de Gestió Ambiental
AGIN	Accumulated Gross Irrigation Needed
ANIN	Accumulated Net Irrigation Needed
AP	Acidification Potential
ARD	Adra River Delta
AR_{eff}	Accumulated Effective Rainfall
BOD	Biological Oxygen Demand
C	Carbon
Ca	Calcium
C_{GW}	Salinity of the fresh groundwater
C_N	Salinity in Nueva
C_{Sea}	Salinity in the Mediterranean Sea
CAT	Water Consortium of Tarragona
CE	Electrical Conductivity of irrigation
CED	Cumulative Energy Demand
CF	Carbon Footprint
Cf	Characterization Factor
CFC	Chlorofluorocarbons
Cl_2	Chlorine
CML	Institute of Environmental Sciences (Leiden)
$CO_{2,eq}$	Carbon dioxide equivalent emissions
COD	Chemical Oxygen Demand
CWU	Consumptive Water Use
DALY	Disability Adjusted Life Years
DAR	Depletion of Abiotic Resources
DCB	Dichlorbenzene
DI	Deficit of Irrigation
EC50	50% Effective Concentration
EDF	Ecosystem Damage Factor
EESC	Equivalent Effective Stratospheric Chlorine
EF	Effect Factor
EP	Eutrophication Potential
EQ	Ecosystem Quality
eq	Equivalent
ET_0	Evapotranspiration Potential
ET_c	Crop Evapotranspiration
ET_N	Evapotranspiration from Nueva
ET_X	Nueva Evapotranspiration in wet months
ET_Y	Nueva Evapotranspiration in dry months

ETP	Ecotoxicity Potential
EUE	Energy Use Efficiency
EUPHOROS	Efficient Use of Inputs in Protected HORticulture Research project
EV	Expected value
EWR	Environmental Water Requirements
FAO	Food and Agricultural Organization of the United Nations
Fd	Flow diagram
FD	Freshwater Depletion
$FeCl_3$	Ferric chloride
FEI	Freshwater Ecosystem Impact
FF	Fate Factor
FGW	Fresh groundwater
FGW_x	Fresh groundwater in wet months
FGW_y	Fresh groundwater in dry months
FMU	Farm Management Unit
FU	Functional Unit
FWAET	Freshwater Aquatic Ecotoxicity
GH_{Area}	Greenhouse area
GHG	Greenhouse Gas Emissions
GIS	Geographic Information System
$GW_{o,y}$	Groundwater outflow from Nueva Lagoon in the dry months
$GW_{o,x}$	Groundwater outflow from Nueva Lagoon in the wet months
GWP	Global Warming Potential
HC50	50% Hazardous Concentration
HCl	Hydrochloric acid
HDF	Human Damage Factor
HH	Human Health
HT	Human Toxicity
I	Irrigation
ILCD	International Reference Life Cycle Data System
IPCC	Intergovernmental Panel on Climate Change
IRTA	Institute of Agriculture and Food Research and Technology
IS	Impact Score
ISO	International Organization for Standardization
IWC	Irrigation Water Cost
Kc	Crop Coefficient
$KMnO_4$	Potassium permanganate
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LCT	Life Cycle Thinking
LF	Leaching Fraction
LU	Land Use

MAE	Marine Aquatic Ecotoxicity
MAGRAMA	Spanish Ministry of Agriculture, Food and the Environment's
MCS	Monte Carlo Simulation
Mg	Magnesium
NaOH	Sodium hydroxide
NaClO	Sodium hypochlorite
NH_3	Ammonia
N_2O	Dinitrogen monoxide
NO_3^-	Nitrogen oxides
NOEC	No-Observed-Effect Concentration
NPP	Net Primary Production
NPV	Net Present Value
O_3	Ozone
ODP	Ozone Depletion Potential
OECD	Organisation for Economic Co-operation and Development
P	Precipitation
PAF	Potentially Affected Fraction of species
PDF	Potential Disappeared Fraction of species
PEC	Predicted Environmental Concentration
PNEC	Predicted No-Effect Concentration
PHO	Photochemical Oxidation potential
P_2O_5	Phosphate
PO_4^{-3}	Phosphate equivalent emissions
PWTP	Potable Water Treatment Plant
R	Rainfall
RAR	Risk Area Ratio
RC	Reference Case
RD	Resources Depletion
Sb_{eq}	Antimony equivalent emissions
SETAC	Society of Environmental Toxicology and Chemistry
SIA	Integrated Water Information database
SIGPAC	Geographic Information System of Spanish farms
SGW_Y	Sea Groundwater infiltration into Nueva Lagoon in dry months
SGW_X	Sea Groundwater infiltration into Nueva Lagoon in wet months
Si	Silicon
$SO_{2,eq}$	Sulfur dioxide equivalent emissions
SSD	Species Sensitivity Distribution
SWRO	Seawater Reverse Osmosis
TET	Terrestrial Ecotoxicity
UNEP	United Nations Environmental Program
UP	Upside Potential
UV	Ultra Violet
UWC	Urban Water Cycle

VaR	Value at Risk
V_N	Volumen Nueva
WA	Freshwater Availability
WAE	Water application efficiency of the irrigation system
WC	Worst Case
WF	Water Footprint
WF_b	Blue Water Footprint
WF_g	Green Water Footprint
WF_{gy}	Grey Water Footprint
WFIA	Water Footprint Impact Assessment
WFN	Water Footprint Network
WR	Renewable Water reserves
WSI	Water Stress Index
WTA	Withdrawal-to-Availavility
WU	Water use
WUE	Water Use Efficiency
WWTP	Wastewater Treatment Plant
X	Wet months
Y	Dry months
y	Yield
YLD	Years Lived Disabled
YLL	Years of Life Lost
y_c	Salinity concentration-response
yr	Year

Glossary

Effect factor	Ecotoxicological effect factor (exposure-response) is the change in PAF of species that experience an increase in stress for a change in contaminant exposure above a predefined effect level.
eFoodPrint®	Is a web-based calculator (www.efoodprint.com) whose goal is to account and assess on-farm environmental sustainability indicators in irrigated agriculture based on the three defined approaches (Agronomical, Water assessment and Global Warming Assessment) enabling to generate simple reports with the results of the analysis.
Fate factor	The fate of a chemical in the environment, hence ecosystem exposure (the fate factor, F), reflects the fraction of an emission that is transferred to water and the duration of the exposure (quantity x duration).
Global Warming Potential	It is a relative measure of how much heat a greenhouse gas traps in the atmosphere and compares the amount of heat trapped by a certain mass of the gas in question to the amount of heat trapped by a similar mass of carbon dioxide. It is calculated over a specific time interval, commonly 20, 100 or 500 years.
Groundwater	Water drawn from beneath the surface of the earth, such as from an aquifer.
Potable Water Treatment Plant	It treats water, groundwater or surface water, and produce potable water for public/human consumption. Normally its water line is based on pre-oxidation (pre-chlorination), physicochemical stage (coagulation/flocculation) and final conditioning (post-chlorination).

Reclaimed Water	The reclamation of water from tertiary treatment of wastewater treatment plant can be referred to as either direct or indirect non-potable reuse (e.g. agriculture, gardens).
Risk metrics	Techniques borrowed from financial risk management to analyze the probability curves associated with each assessed alternative. Value at Risk (VaR), Upside Potential (UP), Risk Area Ratio (RAR), and Worst Case (WC) are risk metrics to support risk-related decisions and properly manage the associated risk.
Seawater Reverse Osmosis	The desalination technologies consist of reducing the saline concentration of water to convert it into suitable water to be consumed by humans. Reverse osmosis membranes basically let water pass through semipermeable and ion-specific membranes to desalt seawater.
Species Sensitivity Distribution	It is an ecotoxicological tool that is useful for the derivation of environmental quality criteria and ecological risk assessment. It can help in the calculation of ecotoxicological effect factors (impact level).
Surface water	A water body on the earth's surface (e.g., a river, lake, or estuary) or water storage on the surface (e.g., cistern) where off-stream water is the source.
Statistical Uncertainty	Uncertainty concerns the absence of knowledge and the robustness of the data/facts on which knowledge is constructed and formulated. Statistical uncertainties can be described using probabilistic functions and can be handled in practice by means of two types of methods: *quantitative*, like Monte Carlo Simulation (MCS) and semi-*qualitative*

	methods, like the Pedigree matrix.
Water consumption or consumptive use	The off-stream use of water, where water release or return does not occur.
Water depletion	Withdrawal from a water source that is not replenished or recharged at approximately the same or greater rate than overall human withdrawal.
Water Footprint	It is defined as the volume of freshwater used to produce the product over the full supply-chain, which looks not only at the direct water use of a consumer or producer, but also at its indirect water use. It also distinguishes the volumes of water consumed by different 'water-colors' (green, blue and grey) depending on the type of water sourced and polluted.
Water-to-Availability	It represents the ratio between withdrawals for different users (WU) and annual freshwater availability (WA).
Water Stress Index	It indicates the portion of consumptive water use that deprives other users of freshwater. It ranges from 0.01 to 1 with 1 meaning a serious water stress in a basin.
Water use	Off-stream use where the water is released or returned to the original river basin, such as that portion of domestic, commercial, or industrial waters returned to a municipal sewer discharged to the original basin.
Wastewater Treatment Plant	It is a plant that "cleans" wastewater and rainwater. Made up of a succession of treatments (pre-, primary, secondary and tertiary) in which water is progressively rid of the pollutants it contains, the plant ultimately discharges clean but not drinking water into the natural environment. The treatment residues are recovered in the form of sludge.

Summary

The irrigated world area has been increased dramatically from the mid of 20th century. The rapid population growth and the resulting demand for food are the main drivers behind. Freshwater use and its consumption have emerged as areas of high environmental concern. Although agricultural lands represent only 12% of the world's land area, roughly 70% of water withdrawn from aquifers, streams and lakes are for irrigated agriculture.

Climate change is a truly global problem around the world and the contribution from the agricultural sector is significantly high. However, quite less attention has been given as it is not yet a subject in the international climate mitigation action such as Kyoto Protocol. Fossil fuels and mainly nitrogen fertilizers used in farming activities are the main contributors to GreenHouse Gas (GHG) emissions from the sector. Therefore, it is vital that the current climate change mitigation needs to consider the importance of the sector towards reducing the global carbon footprint.

Consequently, the environmental impacts from the use of water by agricultural activities and their relative contribution to GHG emissions should be properly addressed. During the last decade, several comprehensive impact assessment methods associated with freshwater use and climate change have been developed to improve water management and mitigate the Global Warming Potential (GWP). Life Cycle Assessment (LCA) methodology aims to measure and assess the environmentally relevant emissions and resources consumed, over the entire life cycle of a product or service (ISO 14040-14044). This analytical tool is increasingly applied and required by industry, authorities and consumers to make sustainability-based decisions. However, more developments are needed to be undertaken to adapt LCA to agricultural

systems as well. Unlike GWP which is assessed as global environmental impact category, the environmental impact of water can be different worldwide since it depends on the spatio-temporal variabilities. Hence, water impact methodologies in LCA are under development.

The present thesis aims mainly to assess the environmental performance of different agricultural systems through the application of LCA and other complementary methods. Two main environmental impacts were considered: Global Warming and Water Footprint. The objective is to provide farmers with methodological and practical decision- making tools to help them to practice in sustainable agriculture.

To provide clear guidance on the water consumption assessment, a first study was developed comparing three different local farms in Ebro basin (Spain). The spatio-temporal variability of the water resource was considered while performing an environmental assessment of freshwater use in agriculture. The Geographic Information System (GIS) have also been implemented as complementary tool for the case study located in Ebro Basin [Chapter 3]. In addition, a second study was carried out aiming at selecting the most environmental pathway for bioethanol production from molasses, honey and sugarcane in a case study located in Tucumán (Argentina) [Chapter 4]. LCA is applied as a methodological tool and different allocation rules such as economic, weight and calorific power were implemented to assess the different pathways to produce bioethanol and to identify the sensibility of the results. With the objective to guide farmers in better water management and reduce climate change associated with practices, a farm-to-gate environmental assessment indices based on the water footprint and GWP approach have been outlined [Chapter 5]. As a result of these water indices and the related Life Cycle Inventory (LCI), a farm-to-gate environmental decision tool (eFoodPrint®) has been developed and implemented [Chapter 6]. Two parallel applications related to water management in agriculture have been presented.

[Chapter 7] deals with the study of environmental impacts, with special attention to biodiversity, from salinity increase due to irrigation in a coastal wetland. A new characterization factor (Cf) has been developed. The Cf is defined as the change in potential affected fraction of species due to a change in salinity. In the second application, it has been considered the reuse of water from urban wastewater treatment plants. After a tertiary treatment this reclaimed water can be used in agriculture. In spite of the increasing values of carbon footprint by the energy consumption used in the tertiary treatment, reclaimed water is considered convenient in drought situations [Chapter 8]. LCA calculations are affected by different types of uncertainties, so the selection of the best option is far from being a straightforward task. A rigorous approach to aid practitioners in the selection of alternatives when the LCA analysis is affected by several sources of uncertainty is required for decision makers. Dealing with the best LCA choice when we have different alternatives, a framework for decision-making under probabilistic uncertainties in LCA has been developed using risk metrics [Chapter 9].

Among other important follow-up lines of research, future work should focus on calculating the uncertainties of the developed methods, proposing general characterization factors which can be applied in different areas with similar conditions and also, optimizing the full life cycles.

Resumen

La superficie mundial de regadío ha aumentado drásticamente desde la segunda mitad del siglo XX. El acelerado crecimiento de la población y la consecuente demanda de alimentos, son los principales conductores de ello. El uso de agua dulce y su consumo se han convertido en áreas de interés ambiental. Aunque las tierras agrícolas representan sólo el 12% de la superficie terrestre del mundo, aproximadamente el 70% del agua extraída de los acuíferos, ríos y lagos se utiliza para la agricultura de regadío.

El cambio climático es un conocido problema global en todo el mundo y la contribución del sector agrícola es significativamente alta. No obstante, no se ha prestado la atención necesaria en este sector ya que no ha sido sujeto todavía a las acciones internacionales como el Protocolo de Kyoto para la mitigación del cambio climático. Los combustibles fósiles y principalmente los fertilizantes nitrogenados utilizados en las actividades agrícolas son los principales contribuyentes en las emisiones de Gases de Efecto Invernadero (GEI) del sector. Por lo tanto, hay una necesidad vital en la mitigación del cambio climático para que se considere la importancia de este sector hacia una reducción de la huella de carbono global.

Consiguientemente, los impactos ambientales procedentes del uso del agua debido a las actividades agrícolas y su relativa contribución en la emisión de GEI deben ser tratados adecuadamente. Durante la última década, varios métodos globales de evaluación de impacto asociados con el uso de agua dulce y el cambio climático han sido desarrollados para mejorar la gestión del agua y mitigar el Potencial del Calentamiento Global (PCG). La metodología de Análisis del Ciclo de Vida (ACV) tiene como objetivo medir y evaluar las emisiones de relevancia ambiental y de los recursos consumidos, durante todo el ciclo de vida de un producto o servicio (ISO 14040 a 14044). Esta herramienta analítica se aplica cada vez más y es requerida también por la industria, las

autoridades y los consumidores para la toma de decisiones sostenibles. Sin embargo, hay una necesidad de emprender más desarrollos para adaptar dicha herramienta en sistemas agrícolas. Al contrario de lo que sucede con el PCG que se evalúa como una categoría de impacto global, el impacto ambiental relativo al agua puede ser diferente en todo el mundo ya que depende de las variabilidades espacio-temporal. Por lo tanto, las metodologías de impacto para el agua en ACV están en vías de desarrollo.

La presente tesis tiene como objetivo principal evaluar el perfil ambiental de los diferentes sistemas agrícolas a través de la aplicación del ACV y otras herramientas complementarias. Principalmente, se consideraron dos impactos ambientales: el Calentamiento Global y la Huella Hídrica. El objetivo es proporcionar a los agricultores herramientas metodológicas y prácticas para la toma de decisiones y poder así practicar una agricultura sostenible.

Para proporcionar una clara orientación sobre la evaluación del consumo de agua, un primer estudio se desarrolló comparando tres fincas diferentes localizadas en la Cuenca del Ebro (España). Se tuvo en cuenta la variabilidad espacio-temporal de los recursos hídricos para llevar a cabo una evaluación ambiental del uso de agua dulce en la agricultura. Se empleó el Sistema de Información Geográfica (SIG) como herramienta complementaria en el caso del estudio localizado en la cuenca del Ebro [Capítulo 3]. Un segundo estudio se llevó a cabo para seleccionar mediante ACV la vía más ambiental para producir bioetanol a partir de melaza, miel y caña de azúcar mediante un caso de estudio ubicado en Tucumán (Argentina) [Capítulo 4]. El ACV es aplicado como una herramienta metodológica y diferentes reglas de asignación como por ejemplo económica, másica, calorífica fueron implementadas para analizar diferentes vías de producción de bioetanol e identificar la sensibilidad de los resultados. Con el objetivo de orientar a los agricultores en mejorar la gestión del agua y la reducción del cambio climático asociadas en las prácticas agrícolas, se han propuesto índices ambientales a nivel de finca en base a la metodología de huella hídrica y el potencial de calentamiento global [Capítulo 5]. Como resultado de estos índices de agua y el consiguiente Inventario de Ciclo de Vida (ICV), se ha desarrollado e implementado una herramienta ambiental (eFoodPrint®) [Capítulo 6]. Adicionalmente, se han añadido dos aplicaciones paralelas relacionadas con la gestión del agua en la agricultura. El [Capítulo 7] se ocupa del estudio de impacto ambiental, con especial atención en la biodiversidad, a partir del aumento de la salinidad a causa del uso de agua para riego en un humedal costero, desarrollado un nuevo factor de caracterización (Fc). El Fc se define como la fracción de especies potencialmente afectada debido a un cambio en la salinidad. En la segunda aplicación, se ha considerado la reutilización de aguas residuales urbanas procedentes de depuradoras. Después de un tratamiento terciario esta agua recuperada se puede reutilizar en la agricultura. A pesar del incremento de la huella de carbono por el consumo de energía utilizada en el tratamiento terciario, se considera conveniente la reutilización de agua en situaciones de escasez hídrica [Capítulo 8]. Cualquier cálculo en ACV se ve afectado por diferentes tipos de incertidumbres, por lo que la selección de la mejor opción está lejos de ser una tarea sencilla. Así pues, los actores involucrados requieren una estrategia rigurosa en la selección de alternativas cuando un ACV es afectado por varias fuentes de incertidumbre. Para tratar la mejor opción de ACV cuando se tienen diferentes alternativas, se ha desarrollado un planteamiento para la toma de decisiones bajo incertidumbre probabilística en ACV utilizando métricas de riesgo [Capítulo 9].

Entre otras líneas de investigación importantes, se plantea un trabajo futuro que debe centrarse en el cálculo de las incertidumbres de los métodos desarrollados, en la propuesta de factores de caracterización generales que se puedan aplicar en diferentes áreas con condiciones similares y en la optimización de los ciclos de vida completos.

Resum

La superfície mundial de regadiu ha augmentat dràsticament durant la segona meitat del segle XX. L'accelerat creixement de la població i la conseqüent demanda d'aliments són els principals conductors. L'ús d'aigua dolça i el seu consum s'han convertit en àrees d'interès ambiental. Encara que les terres agrícoles representen només el 12% de la superfície terrestre del món, aproximadament el 70% de l'aigua extreta dels aqüífers, rius i llacs s'utilitza per a l'agricultura de regadiu.

El canvi climàtic és un conegut problema global a tot el món i la contribució del sector agrícola és significativament alta. No obstant això, no s'ha prestat l'atenció necessària en aquest sector ja que no ha sigut subjecte encara a les accions internacionals com el Protocol de Kyoto per a la mitigació del canvi climàtic. Els combustibles fòssils i principalment els fertilitzants nitrogenats utilitzats en les activitats agrícoles són els principals contribuents en les emissions de Gasos d'Efecte Hivernacle (GEH) del sector. Per tant, hi ha una necessitat vital en la mitigació del canvi climàtic perquè es consideri la importància d'aquest sector cap a una reducció de la petjada de carboni global.

Conseqüentment, els impactes ambientals procedents de l'ús de l'aigua a causa de les activitats agrícoles i la seva relativa contribució en l'emissió de GEH han de ser tractats adequadament. Durant l'última dècada, diversos mètodes globals d'avaluació d'impacte associats amb l'ús d'aigua dolça i el canvi climàtic han estat desenvolupats per millorar la gestió de l'aigua i mitigar el Potencial d'Escalfament Global (PEG). La metodologia d'Anàlisi del Cicle de Vida (ACV) té com a objectiu mesurar i avaluar les emissions de rellevància ambiental i dels recursos consumits, durant tot el cicle de vida d'un producte o servei (ISO 14040 a 14044). Aquesta eina analítica s'aplica cada vegada més i és requerida també per la indústria, les autoritats i els consumidors per a la presa de decisions sostenibles. No obstant això, hi ha una necessitat d'emprendre més

desenvolupaments per adaptar aquesta eina als sistemes agrícoles. Al contrari del que succeeix amb el PEG que s'avalua com una categoria d'impacte global, l'impacte ambiental relatiu a l'aigua pot ser diferent a tot el món ja que depèn de les variabilitats espai-temporal. Per tant, les metodologies d'impacte per a l'aigua en ACV estan en vies de desenvolupament.

La present tesi té com a objectiu principal avaluar el perfil ambiental dels diferents sistemes agrícoles a través de l'aplicació de l’ACV i altres eines complementàries. Principalment, es van considerar dos impactes ambientals: l'escalfament global i la petjada hídrica. L'objectiu és proporcionar als agricultors eines metodològiques i pràctiques per a la presa de decisions i poder així practicar una agricultura sostenible.

Per proporcionar una clara orientació sobre l'avaluació del consum d'aigua, un primer estudi es va desenvolupar comparant tres finques diferents localitzades en la Conca de l'Ebre (Espanya). Es va tenir en compte la variabilitat espai-temporal dels recursos hídrics per dur a terme una avaluació ambiental de l'ús d'aigua dolça en l'agricultura. Es va emprar el Sistema d'Informació Geogràfica (SIG) com a eina complementària en el cas de l'estudi localitzat en la conca de l'Ebre [Capítol 3]. Un segon estudi es va dur a terme per seleccionar mitjançant ACV la via més ambiental per produir bioetanol a partir de melassa, mel i suc de canya de sucre mitjançant un cas d'estudi localitzat al Tucumán (Argentina) [Capítol 4]. L’ACV és aplicat com una eina metodològica i diferents regles d'assignació com per exemple econòmica, màssica, calorífica van ser implementades per analitzar diferents vies de producció de bioetanol i identificar la sensibilitat dels resultats. Amb l'objectiu d'orientar als agricultors a millorar la gestió de l'aigua i la reducció del canvi climàtic associades en les pràctiques agrícoles, s'han proposat índexs ambientals a nivell de finca en base a la metodologia de la petjada hídrica i el potencial d'escalfament global [Capítol 5]. Com a resultat d'aquests índexs d'aigua i el conseqüent Inventari de

Cicle de Vida (ICV), s'ha desenvolupat i implementat una eina ambiental (eFoodPrint®) [Capítol 6]. Addicionalment, s'han afegit dues aplicacions paral·leles i relacionades amb la gestió de l'aigua en l'agricultura. El [Capítol 7] s'ocupa de l'estudi d'impacte ambiental, amb especial atenció en la biodiversitat, a partir de l'augment de la salinitat a causa de l'ús d'aigua per a reg en un aiguamoll coster desenvolupant un nou factor de caracterització (Fc). El Fc es defineix com la fracció d'espècies potencialment afectada a causa d'un canvi en la salinitat. En la segona aplicació, s'ha considerat la reutilització d'aigües residuals urbanes procedents de depuradores. Després d'un tractament terciari aquesta aigua recuperada es pot reutilitzar en l'agricultura. Malgrat l'increment de la petjada de carboni pel consum d'energia utilitzat en el tractament terciari, es considera convenient la reutilització d'aigua en situacions d'escassetat hídrica [Capítol 8]. Qualsevol càlcul en ACV es veu afectat per diferents tipus d'incerteses, per la qual cosa la selecció de la millor opció està lluny de ser una tasca senzilla. Així doncs, els actors involucrats requereixen una estratègia rigorosa en la selecció d'alternatives quan una ACV és afectada per diverses fonts d'incertesa. Per tractar la millor opció d'ACV quan es tenen diferents alternatives, s'ha desenvolupat un plantejament per a la presa de decisions sota incertesa probabilística en ACV utilitzant mètriques de risc [Capítol 9].

Entre altres línies de recerca importants, es planteja un treball futur que ha de centrar-se en el càlcul de les incerteses dels mètodes desenvolupats, en la proposta de factors de caracterització generals que es puguin aplicar en diferents àrees amb condicions similars i en l'optimització dels cicles de vida complets.

Preface

This thesis belongs to the Chemical Engineering, Environmental and Process PhD programme of the Universitat Rovira i Virgili in Tarragona (Spain). The work was carried out within the research group on Environmental Analysis and Management (AGA) at the Department of Chemical Engineering from January 2011 to March 2013.

The thesis wants to provide a methodology and water indices for agriculture through an environmental tool in order to improve the water management and mitigate the climate change from farm practices. Therefore, it aims to contribute to the methodological development of LCA providing a new method to evaluate the environmental impacts associated with salinity on biodiversity in a coastal wetland. A framework for decision-making under uncertainties in LCA and a proposal solution to reuse reclaimed water from Urban Water Cycle have been also addressed in the present dissertation.

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:

- **M.Amores,** A.Antón, F.Castells. "Comparing different methods to perform an environmental assessment of freshwater use in agriculture from Ebro Basin". International Journal of Life Cycle Assessment. (Last review from March 2013).
- **M.Amores**, F. Mele, L.Jimenez, F.Castells. "Life Cycle Assessment of fuel ethanol from sugarcane in Argentina". International Journal of Life Cycle Assessment, DOI 10.1007/s11367-013-0584-2.

- **M.Amores**, F.Verones, C.Raptis, R.Juraske, S.Pfister, F.Stoessel, A.Antón, F.Castells, S.Hellweg. "Biodiversity impacts from salinity increase in a coastal wetland". Environmental Science and Technology, dx.doi.org/10.1021/es3045423.

- **M. Amores**, A.Antón, F.Ferrer, O.Cabot, A.Baltiérrez, A.Duaigües, F. Castells. "Farm-gate environmental assessment indices based on the water footprint and life cycle approach". Journal of Cleaner Production. (Submitted in March 2013).
- **M.Amores**, G.Guillén., F.Castells "Framework for decision-making under probabilistic uncertainties in LCA studies". International Journal of Life Cycle Assessment. (Last review from January, 2013).
- **M.Amores**, M.Meneses, J.Pasqualino, A.Antón, F.Castells. "Environmental assessment of urban water cycle on Mediterranean conditions by LCA approach". Journal of Cleaner Production, 43: 84-92.

And the registered Software:

- Universitat Rovira i Virgili (**M.Amores**, F.Castells), Centre d'Assessoria Dr LabFerrer (F.Ferrer, O.Cabot, T. Baltiérrez) and Oleia S.L. (A.Duaigües) (2013): registered with the number T-0164-2012 in Registre Propietat Intel·lectual de Catalunya. December 18th, 2012. Registered Software "*Model d'Ús Sostenible de l'Aigua a l'Agricultura*" eFoodPrint® [in Spanish, Catalan and English].

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

- **M.Amores**, M.Meneses, J.Pasqualino, F.Castells. Análisis de Ciclo de Vida de l'Estació de Tractament d'Aigua Potable de l'Ampolla. En el marc del Projecte Sostaqua. Antiga Audiència de Tarragona, April 15, 2011.

- **M.Amores**, J.Pasqualino, M.Meneses, I.Butnar, F. Castells. Life Cycle Assessment on Catalonia coast: Integration of urban water cycle. 4th International Life Cycle Assesment Conference in Latin-America, CILCA 2011, April 4-7, 2011, Coatzacoalcos, Mexico.
- **M.Amores**, F.Ferrer, O.Cabot, A.Anton, A.Duaigües, F.Castells, ED.Gemechu. Implementing decision making in irrigation management based on productive and environmental indicators. VIIIth International Conference on Life Cycle Assessment in the Agri-Food Sector, October 2-4, 2012, Saint-Malo, France.
- **M. Amores**, M. Meneses, J. Pasqualino, F. Castells, ED. Gemechu "Life Cycle Assessment of Urban Water Cycle in Mediterranenan Cities". Society of Environmental Toxicology And Chemistry (SETAC) Europe 21st Annual Meeting, May 15-19, 2011 Milan, Italy.
- R.Aldaco, M. Margallo, **M.Amores**, M. Meneses, F.Castells, A. Irabien. Environmental performance of a photovoltaic solar electrooxidation (PSEO) process: comparison with a conventional biological treatment, 7th International Conference on Life Cycle Management, LCM2011, August 28-31, 2011, Berlin, Germany.
- **M.Amores**, A.Antón, F.Ferrer, O.Cabot, A.Baltiérrez, A.Duaigües, F. Castells. Evaluación de la sostenibilidad del uso del agua en fincas agrícolas. Modelo del Uso de Agua en Fincas agrícolas (MUSA). Congreso anual de la red española de Análisis de Ciclo de Vida, November 5, 2012, Móstoles, Madrid.
- **M.Amores**, A.Antón, F.Ferrer, O.Cabot, A.Baltiérrez, A.Duaigües, F. Castells Farm-gate environmental decision tool. 5th International Life Cycle Assessment Conference in Latin-America, CILCA 2013, March 24-27, 2013, Mendoza, Argentina.
- **M.Amores**, F.Verones, C.Raptis, R.Juraske, S.Pfister, F.Stoessel, A.Antón, F.Castells, S.Hellweg. Development of characterization factor to assess biodiversity damage due to salinity increase in a coastal wetland. Society

of Environmental Toxicology And Chemistry (SETAC) Europe 23rd Annual Meeting, May 12-16, 2013, Glasgow, United Kingdom.

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 the following conferences:

- **M. Amores**, I. Butnar, D. Bejarano, S. Laos, F. Castells, Y. Katakis. Carbon footprint comparison of several methods of Salmonella detection: método convencional (ISO 6579) vs método iMicroq. Confererencias sobre el Control de Salmonella, perspectivas de futuro y análisis de huella de carbono de diferentes métodos organizadas por IMICROQ en Fundació Rovira i Virgili (FURV), 2011.
- E.Demisse Gemechu, I.Butnar, M.Llop, **M.Amores**, F.Castells, "Environmental Tax On Products And Services Based On Life Cycle Analysis", SETAC Europe 21st Annual Meeting, May 14-17, 2011, Milan, Italy.
- E. Gemechu, I. Butnar, J. Recari, **M.Amores**, F.Castells. Comparison of life cycle inventory (LCI) methods for carbon footprint calculation:-the case of pulp and paper sector in Spain, 7th International Conference on Life Cycle Management, LCM2011, August 28-31, 2011, Berlin, Germany.
- **M.Amores**. Teacher in 34th Edition of Dow Chemical Company-Universitat Rovira i Virgili. "Sustainability tools and Life Cycle Assessment. Case studies", 2012.
- E.Demisse Gemechu, I.Butnar, J.Gomà-Camps, A.Pons, **M.Amores**, F. Castells, "GHG Emissions Comparison of Tissue Paper from Virgin Pulp vs. Recycled Waste Paper", SETAC Europe 22nd Annual Meeting, May 20-24, 2012. Berlin, Germany.
- E.Gemechu, I.Butnar, M.Llop, **M.Amores**, F.Castells. The impacts of environmental tax on production and consumption of goods in the

Spanish economy. The 20th International Input-Output Conference and the 2nd Edition of the International School of Input-Output Analysis, June 24-29, 2012 Bratislava, Slovakia.

The doctoral thesis is based on the methodology and results developed under the framework of the following projects:

- *Proyecto SOSTAQUA. Evaluación de la sostenibilidad del ciclo del agua en España* (TT07001C) (2008-2010). A Project funded by Spanish Ministry of Science and Technology (Spain). The project was coordinated by Aguas de Barcelona (AGBAR) and had the participation of several research institutions between them, Universitat Rovira i Virgili (URV).
- *Model de l'Ús Sostenible de l'Aigua a l'Agricultura* (2010VALOR00008) (2011- 2012). A project funded by Agència de Gestió d'Ajuts Universitaris i de Recerca (AGAUR), part of Catalan Government, Spain. The Project was coordinated by Universitat Rovira i Virgili (URV) and had the participation of the Centre d'Assessoria LabFerrer and Universitat de Lleida (UdL).
- *Técnicas estocásticas y deterministas de evaluación del impacto ambiental basadas en simulación de procesos y análisis de ciclo de vida"* (CTQ2009-14420-C02-01) (2009-2012). A project funded by Spanish Ministry of Science and Technology (Spain). The project was coordinated by Universitat Rovira i Virgili (URV) and it had the participation of Universidad de Alicante.

As well as, during the following research stay:

- Three-month stay (April – June 2012) at the Institute of Environmental Engineering of the Swiss Federal Institute of Technology (ETH), in Zürich (Switzerland). The hosting researcher was the Professor Dr. Stefanie Hellweg. Grant BE1-385 from Agència de Gestió d'Ajuts Universitaris i de Recerca (AGAUR), Catalan Government.

Structure of the thesis

The structure of the dissertation is organized into six main parts and eleven chapters. For clarity, the structure of the doctoral thesis is further outlined in Figure 0. This flow chart can be used throughout the reading of this manuscript as a *dissertation map*.

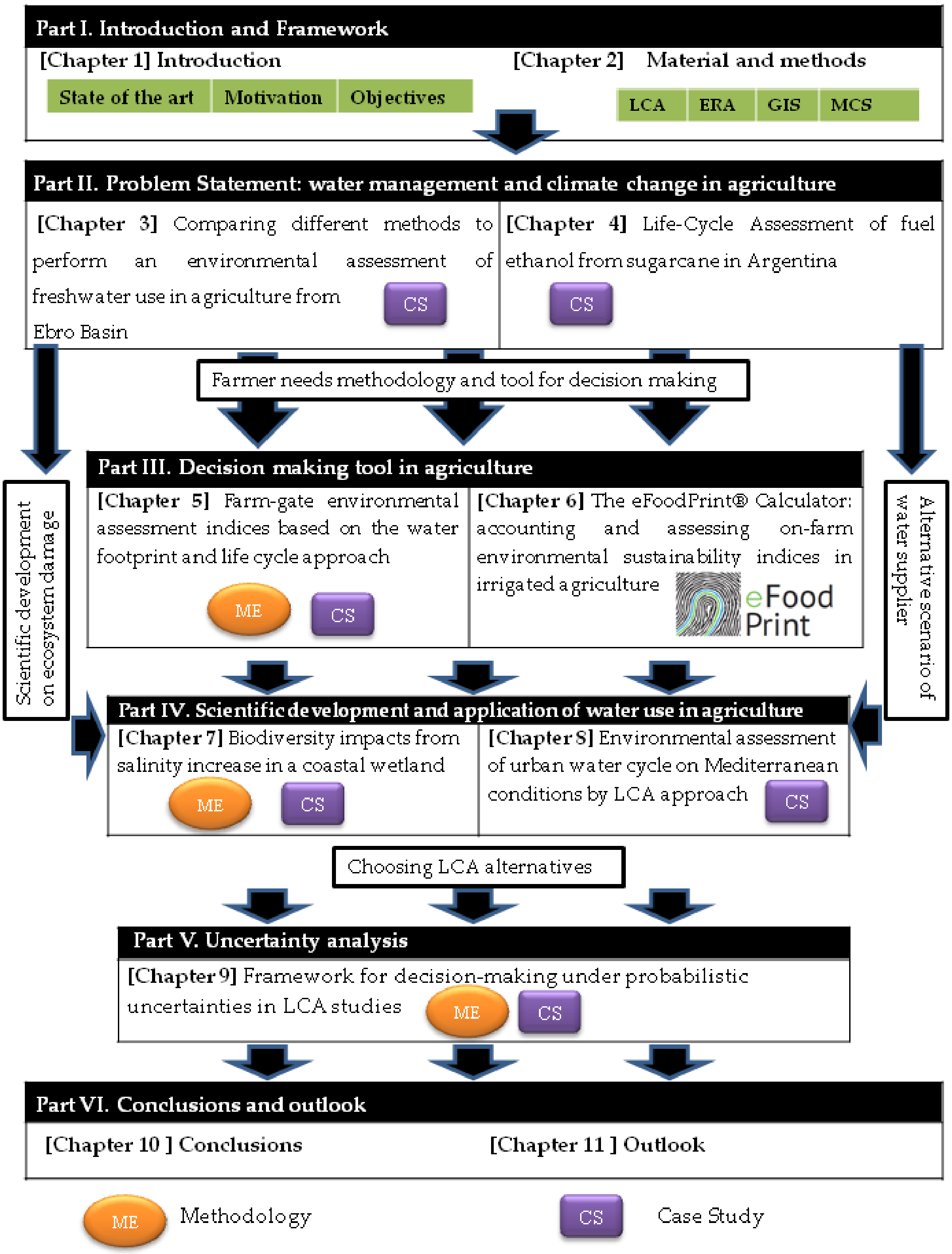


Figure 0. Map structure of methodologies applied in this dissertation.

Part I. Introduction and Framework

Part I is composed of two chapters. **Chapter 1** *[Introduction]* presents an overview of the topic of agriculture. The chapter is organized as follow: Section 1.1 and 1.2 deal about the evolution of climate change and water use/management, respectively. Section 1.3 outlines the applications of life cycle approach to agricultural systems. Section 1.4 explains how the correct decision making for the agricultural practices could be communicated to farmers. A State of the art of existing standards, protocols and methods proposed during the last decades in order to assess climate change and the water use/impact is highlighted in section 1.5. The motivation of this dissertation is presented in section 1.6, and finally, the objectives are enumerated in section 1.7. **Chapter 2** *[Material and methods]* describes the methodologies and tools applied in the dissertation: Life Cycle Assessment (LCA), Environmental Risk Assessment (ERA), Geographic Information System (GIS) and Monte Carlo Simulation (MCS).

Part II. Problem statement: water management and climate change in agriculture

Part II is composed of two chapters. **Chapter 3** *[Comparing different methods to perform an environmental assessment of freshwater use in agriculture from Ebro Basin]* addresses the issue of freshwater consumption through applying an LCA approach. To deal with this topic, water consumption and the environmental impacts of growing three experimental farms (grape, nectarine and corn) are evaluated in the Ebro watershed. Two origins of water are distinguished in the assessment, blue and green water. Therefore, they are separately evaluated. The blue water comes from surface and ground water bodies and it is used for irrigation. The green water is stored as soil moisture from rainfall that does not become runoff. This differentiation between the two water types results in different impacts for irrigated crops. In addition, different published water methodologies have been assessed at the midpoint and endpoint (ecosystem quality) levels, taking into account the spatial characterization factors of different models (country, river and watershed). The obtained results have been compared with the same crops in different Spanish watersheds. This study provides a response to the users (e.g. farmers, administration, certifiers) on the environmental assessment of water consumption at midpoint and endpoint level (focus on ecosystem quality) in order to assist beneficial and strategic decision-making. **Chapter 4** *[Life-Cycle Assessment of fuel ethanol from sugarcane in Argentina]* presents an LCA case study of ethanol fuel from sugarcane in Tucumán (Argentina). It assesses the potential environmental impacts associated with the full life of ethanol fuel from cradle-to-gate. It covers the typical emission-related impact categories at the midpoint Life Cycle Impact Assessment (LCIA). Several Argentinean industry subsystems have been assessed based on real data from three pathways to produce bioethanol (molasses, honey and sugarcane juice). Several sensitivity analyses have been performed to study the variability of the Global Warming Potential (GWP) according to different case studies. The results highlight that agriculture is the subsystem which shows the highest impact in almost all impact categories. This is mainly due to its high fossil fuel consumption. The total GWP impact is proven to be sensitive to changes in the economic, calorific and mass allocation for each pathway.

Part III. Decision making tool in agriculture

This part is composed of two chapters. **Chapter 5** *[Farm-gate environmental assessment indices based on the water footprint and life cycle approach]* presents a methodological approach based on water and agronomical indices. These indices were followed to develop an operational tool in order to generate an on-farm simple inventory to calculate environmental sustainability indicators regarding water consumption and GWP. In order to test the applicability & facility of the implementation of this procedure, three consecutive campaigns (2010-2012) have been considered for a pilot operational cases of farms located in the Ebro Basin (North-East Spain). The output of the study is the development of a web-based water, carbon and agronomical indices (so called, eFoodPrint®) which provides the farm managers information that helps them make strategic and tactical decisions. **Chapter 6** *[The eFoodPrint® calculator: accounting and assessing on-farm environmental sustainability indices in irrigated agriculture]* explains how the environmental support tool, eFoodPrint®, can be used to determine the water use management and environmental impacts of different agriculture crops by assessment of different indices. eFoodPrint® is an easy-to-use tool in order to help agricultural stakeholders to choose efficient options in order to mitigate the environmental impacts of crops and to improve the water management techniques.

Part IV. Scientific development and application of water use in agriculture

Part IV is composed of two chapters. **Chapter 7** *[Biodiversity impacts from salinity increase in a coastal wetland]* addresses a Life Cycle Impact Assessment method, which was developed to assess the salinity impacts in a coastal Spanish wetland called "Albufera de Adra". It is located in a semi-arid region in Almería (South-East of Spain), where agricultural activities require substantial irrigation and areas with native vegetation and fauna are restricted to some small patches and wetlands. This work derives the first Characterization Factor (Cf) for salinity impacts in a coastal wetland. It is defined as the change in the Potentially Affected Fraction (PAF) of species due to a change in salinity which is caused by the changed amounts of groundwater used for crop irrigation. Data describing the effect of salinity for various endpoints (e.g. survival, growth inhibition) on 18 species (plants, fish, algae and a crustacean) native to the "Albufera de Adra" wetland were collected from literature. In order to test the applicability of the Cf, an assessment of water consumption of greenhouse crops in the area was conducted as a case study. The results show that the impact of salinity due to water use may be significant in terms of damage to ecosystems. **Chapter 8** *[Environmental assessment of urban water cycle on Mediterranean conditions by LCA approach]* aims at carrying out an environmental analysis based on local operation data of the current urban water cycle (scenario I) in a Mediterranean city of Spain (Tarragona). Two additional scenarios have been outlined in order to improve the environmental performance and to reduce the high level of water stress as a result of an increasing demand or limited resources. Scenario II proposes the use of reclaimed water from tertiary treatment to non potable uses as agricultural irrigation and scenario III, additionally includes a desalinization plant to supply water during drought situations. The results from the LCA on the current situation (scenario I) show that distribution network, collection pumping and wastewater treatment plant are the main contributors for the GWP impact category. The results from scenario II (reclaimed water) highlights that there is no significant improvement in indicators is observed, whereas scenario III (the drought scenario) reveals increments in all impact categories compared with Scenario II. This study has significant contribution for the holistic conception of the urban water cycle demonstrated by the inclusion of the case study based on real operation data, taking into account a water supply system of a city. In addition, a reclaimed water option on agriculture irrigation could be considered as an alternative source of water supplier in order to avoid water consumption from natural sources.

Part V. Uncertainty analysis

Part V is composed of one chapter. **Chapter 9** *[Framework for decision-making under probabilistic uncertainties in LCA studies]* provides a framework to aid practitioners in the selection of alternatives when the LCA analysis is affected by several sources of uncertainty. A proposal methodology for decision-making under uncertainty in LCA is introduced in this chapter. It focuses on analyzing probabilistic uncertainties in which representative scenarios of the uncertain parameter spaces are generated via stochastic modeling methods and financial risk management tools. This chapter explains how to eliminate alternatives with poor performance, but which are difficult to discard in a straightforward manner using conventional approaches and illustrates the capabilities of the approach using an example where several alternatives for energy production are assessed under uncertainty in the Life Cycle Inventory (LCI) results.

Part VI. Conclusions and outlook

Finally, **Part VI** includes **Chapter 10** [*Conclusions*] and **Chapter 11** [*Outlook*] and it provides the general and specific conclusions of the dissertation and proposes future fields of research associated with the LCA methodologies in agricultural sector and some recommendations for further work.

Part I

Introduction and Framework

Chapter 1

Introduction

A journey of a thousand miles begins with a single step…

Chapter 1 introduces some relevant circumstances influencing the evolution of sustainable agriculture, explains the current situation regarding water management and climate change and the intergovernmental efforts to improve the environmental impacts caused for agriculture. Then, it presents the framework of how an environmental assessment tool (Life Cycle Assessment) could be used to improve and mitigate the problematic and how it could help in the decision making and communication to farmer community. All these topics are the cornerstones of this dissertation. Finally, it presents the justification and motivation of the dissertation and enumerates its objectives.

This chapter is structured as follows:

- Agriculture and Climate Change
- Water in agriculture
- Life cycle approach in agriculture assessment
- Decision making and communication to farmers
- State of the art of water and climate change standards, specifications and protocols
- Motivation of the dissertation
- Objectives of the dissertation

1.1. Agriculture and Climate Change

For more than a third of the world's population, **agriculture** is the main source of income while their products are needed to feed the whole population. Agriculture is a major user of natural resources and its environmental performance needs to be monitored and evaluated (OECD, 2010). At least 14% of global annual **greenhouse gas emissions** (GHG) come directly from farm sector (more than transport contribution and not far behind the contribution from industry) (Figure 1.1). Climate change is now largely accepted as a real, pressing and truly global problem. In accordance with IPCC (IPCC, 2007) climate change is defined as…

"…a change in the state of the climate that can be identified (e.g. using statistical tests) by changes in the mean and/or the variability of its properties and that persists for an extended period, typically decades or longer. It refers to any change in climate over time, whether due to natural variability or as a result of human activity…"

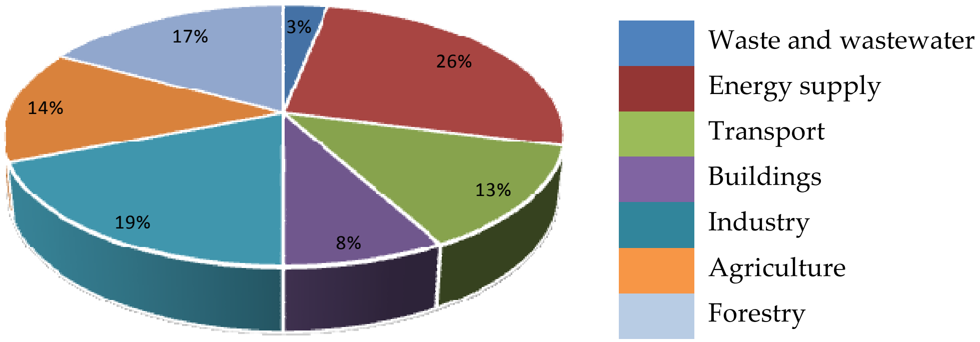


Figure 1.1. Distribution of global GHG emissions by sector in 2004 (IPCC, 2007; Burney et al. 2010). (*Source: Intergovernmental Panel on Climate Change www.ipcc.ch*).

Fossil fuels used in ploughing fields, transport, crop processing, pumping irrigation water and production of nitrogenous fertilizer releases **carbon dioxide (CO_2) emissions;** livestock breeding and wet rice cultivation emit large quantities of **methane (CH_4)** and excessive use of artificial N-fertilizer generates **nitrous oxide (N_2O).** Although in some cases agriculture through the photosynthesis and biological accumulation of carbon in plant matter and soil organic matter could contribute to store carbon (C), it is not usually accounted for edible human food and animal feed products because biogenic CO_2 balance is considered zero. Indeed, farming activities involve land-uses changes and become important changes on the earth's ability to absorb or reflect heat and emissions (OECD 2010). For instance, substitution of woods or rainforest by agricultural land involves usually a reduction in C fixation.

Despite its relatively high contribution to GHG, agriculture is not yet subject to emissions caps under the Kyoto agreement to mitigate climate change (FAO 2011a). Devising policies and undertaking commitments to **mitigate** climate change through agriculture clearly means knowing more about agriculture's **carbon footprint** (FAO 2011a). As in most other sectors this carbon footprint is increasing, since farming is set to expand to produce more **food** for a growing world population.

As the global population heads for more than 9 billion people (today's population of around 7 billion) by 2050 (under medium growth projections), the world is rapidly becoming urbanized and wealthier (more land-use change, more cultivation, more livestock and more use of fossil fuels) (OECD 2008). During next decades, world agriculture faces an enormous **challenge** in order to produce almost 50% more food up to 2030; to improve food security and livelihoods of the rural poor; to maintain the necessary **ecosystem** services; and to reconcile the use of land and water resources among competing uses (FAO 2011b). Future global **food demand** is expected to increase by some 70% by 2050 so food production will need to double from current levels and the amount of **water withdrawn** by **irrigated agriculture** will need to increase by 11% to match the demand for biomass production (FAO 2011b).

GHG emissions from human activity will need to decrease globally from 1990 levels by at least 50% by 2050 if future global warming is to be limited to a 2°C temperature increase, as currently recommended by the Intergovernmental

Panel on Climate Change (IPCC 2007). However, climate change is expected to alter the patterns of temperature, rainfalls and river flows (damaging aquatic ecosystems and the services) upon which the world's food production systems depend. Global atmospheric temperature is predicted to rise by approximately 4˚C by 2080, consistent with a doubling of atmospheric CO_2 concentration (mean temperature will have higher rates in upper latitudes and slower rates in equatorial regions) (FAO, 2011a).

Scientific evidence for global warming is now considered irrevocable (Allison et al. 2009). There is strong potential to mitigate GHG emissions from agriculture, and from other sectors (Paustian 1998). Adaptation and mitigation strategies should focus on increasing resilience, understood as robustness or resistance capacity, of farming systems to reduce current and likely risks such as droughts, excessive rainfall and other extreme events and so on, mitigate the negative impacts of climate change on agricultural production (FAO 2011a). Some strategies as energy saving, efficiency improvement and substitution of fossil fuels (using methane derived from bio-digestion and recycling of organic matter and **biofuels grown on farm**) will have direct benefits for farmers while reducing CO_2 load. Moreover, improvements in fertilizer efficiency through **better management** (precision application and slow-release formulations) can reduce N_2O losses from cropping.

The impacts of climate change on agricultural production and water resources remain highly **uncertain**, with potentially great **spatial variation**. Semi-arid and subtropical areas in the Mediterranean, the Near East, sub-Saharan Africa, and Latin America are likely to be affected most through higher temperatures, more rainfall variability, and greater frequency of extreme events (IPPC,2007; Kurukulasuriya et al. 2003). Figure 1.2 shows the most frequently used climate classification map which was proposed by Wladimir Köppen, and then, presented in its latest version 1961 by Rudolf Geiger. This world map of Köppen-Geiger climate classification was based on temperature and precipitation observations for the period 1951-2000 (Kottek et al. 2006). If we based on ensemble projections of global climate models provided by the Tyndall Centre for Climate Change Research, we can see how this world map could be affected by climate change for the period 2001-2100. Considering different IPCC scenarios A1, A2, B1, B2 (IPCC 2007) and focusing on the area of the study in this dissertation, the south of Spain will become in future projections from warm temperature-summer dry-hot summer (Csa in Figure 1.2) to more each time arid-steppe-hot arid (BSh in Figure 1.2) whereas the north will be fewer each time warm temperature-summer dry-cool summer in the west (Csc in Figure 1.2) and warm temperature-fully humid-warm summer east (Cfb in Figure 1.2) to become Csa (west) and BSh (east), respectively (Rubel and Kottek 2010).

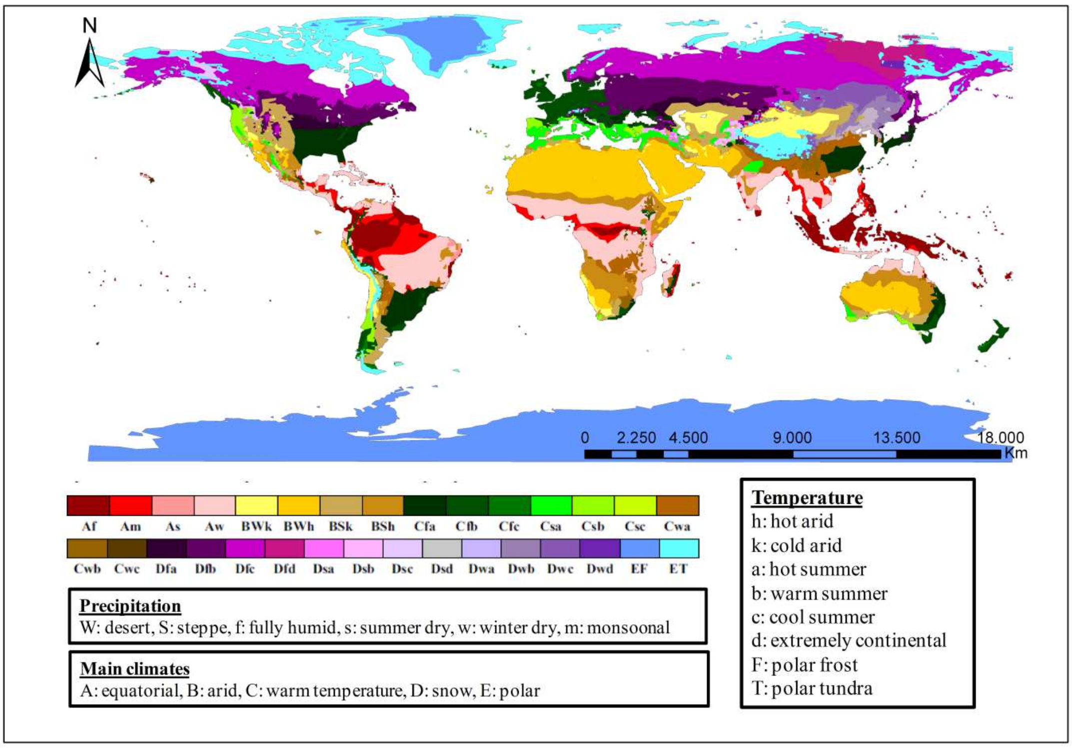


Figure 1.2. World map of the Köppen-Geiger climate classification (*Source: own elaboration based on Kottek et al. 2006; Climatic Research Unit (CRU) of the University of East Anglia and the Global Precipitation Climatology Centre (GPCC) at the German Weather Service.*)

Climate change will significantly impact agriculture by increasing water demand, limiting crop productivity and by reducing water availability in areas where irrigation is most needed or has comparative advantage (FAO 2011a). In addition, climate change brings an increase in risk and unpredictability for farmers from: the growing incidence of extreme weather events, warming and related aridity and shifts in rainfall patterns. Hence, this phenomena will have its greatest impact on agricultural water management in further sharpening the trade-offs between conservation and protection of natural ecosystems, which ultimately support agriculture, and the allocation of land and water to sustain productive agriculture.

A serious effort must be done not only to increase food production, but also to improve the efficiency of food production and collection and minimize losses in the distribution step to the consumer (FAO 2011b). Also, the gradual increase of temperature will have an influence, positive or negative, on the type of crops traditionally cultivated in the region.

1.2. Water in agriculture

Over the last 50 years, the world's cultivated area has grown by 12 percent. Land and water management has met rapidly rising demands for food since the global irrigated area has doubled over the same period, accounting for most of the net increase in cultivated land (FAO 2011b). Meanwhile, world's agricultural production has grown between 2.5 and 3 times thanks to significant increase in the yield of major crops (FAO 2011a). However, global achievements in production in some regions have been associated with degradation of land and water resources, and the deterioration of related **ecosystem** goods and services (biomass, carbon storage, soil health, water storage and supply, biodiversity, and social and cultural services) (MEA 2005).

Future food demand will probably have to be achieved with less water, mainly because of pressures from growing urbanization, industrialization and rise's climate change (FAO 2011a). Consequently, it will be important in future that **farmers** face the right signals to increase water use efficiency and improve **water management**, given that agriculture is the major user of water, **accounting** for about **70%** of the world's **freshwater withdrawals** from nature (aquifers, streams and lakes) with the remainder being for domestic, industrial and hydropower uses (FAO 2008).

The area of irrigation has grown massively in the twentieth century but has depleted **surface** and **groundwater** flows, often with severe consequences for aquatic eco-systems and those dependent on them (Emerton and Bos, 2004; FAO, 2004; Burke and Moench, 2000). Irrigation provides approximately 40 percent of the world's food, including most of its horticultural output. It is estimated that total water use in crop production (evapotranspiration) amounted to 7,130 km^3 in 2000 and is likely to rise to between 12,000 and 13,500 km^3 by 2050 (de Fraiture 2005) where irrigated area is forecast to grow by 33 percent (Bruinsma, 2009). Hence, with demand for food and water rising, it will require widespread adoption of sustainable management practices in agriculture to ensure that water resources are allocated efficiently and equitably and used to achieve socially, environmentally and economically beneficial outcomes (FAO, 2011b).

A great differentiation across different water basins is predominant from the local to international levels. For instance, there is a present heterogeneity of water sources (e.g. surface, groundwater, recycled wastewater, desalinated water) and a linkage between water resource (quantity) and water pollution (quality) issues as well as different consumptive uses (e.g. agriculture, domestic, industrial, power generation). Therefore, the different water sources, use and consumption entail different allocation of water. These varieties makes mandatory to meet environmental needs and the management of the complex institutional and property right arrangements associated with water (OECD, 2008).

Freshwater resources include **surface freshwater** systems (rivers, lakes and wetlands which occupy only 1% of the earth's surface) and **groundwater** systems (shallow aquifers- replenished by surface runoff- and fossil aquifers -non-renewable so not connected to runoff-) (Penning de Vries et al. 2003). The vulnerability of groundwater systems across different continents has recently been assessed in relation to existing utilization, the effects of climate change on recharge and sea-level rise, and wealth (World Bank, 2009). Agriculture has been quick to exploit groundwater circulation and now accounts for over 80 percent of all groundwater withdrawals (Siebert et al. 2007).

Some studies have already predicted 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). Many other recent reports on adaptation to climate change (Fischer et al. 2007, Nelson et al. 2009, Padgham, 2009) anticipate a substantial increase in irrigated area in response to global temperature rise, higher rates of crop water use, and declining and more variable rainfalls. The foregoing indicates, at least for many developing countries, that the options are limited and will need careful scrutiny.

Although freshwater is an abundant resource, now irrigated agriculture faces the problem that the availability of freshwater is decreasing around the world. **Water scarcity** is a reality in many regions of the world being major in regions with high **water consumption** (high population demand) and low **water availability** (reductions in river runoff and aquifer recharge). Water scarcity is defined by the United Nations Water Programme (United Nations 2006) as…

"…the point at which the aggregate impact of all users impinges on the supply or quality of water under prevailing institutional arrangements to the extent that the demand by all sectors, including the environment, cannot be satisfied fully."

In some areas, water scarcity is already a major problem and a serious limit to agricultural development. Farmers are under pressure to grow more "crop per drop" (FAO 2011b). Increasing water scarcity constrains irrigated production, particularly in the most highly stressed countries and areas (Figure 1.3).

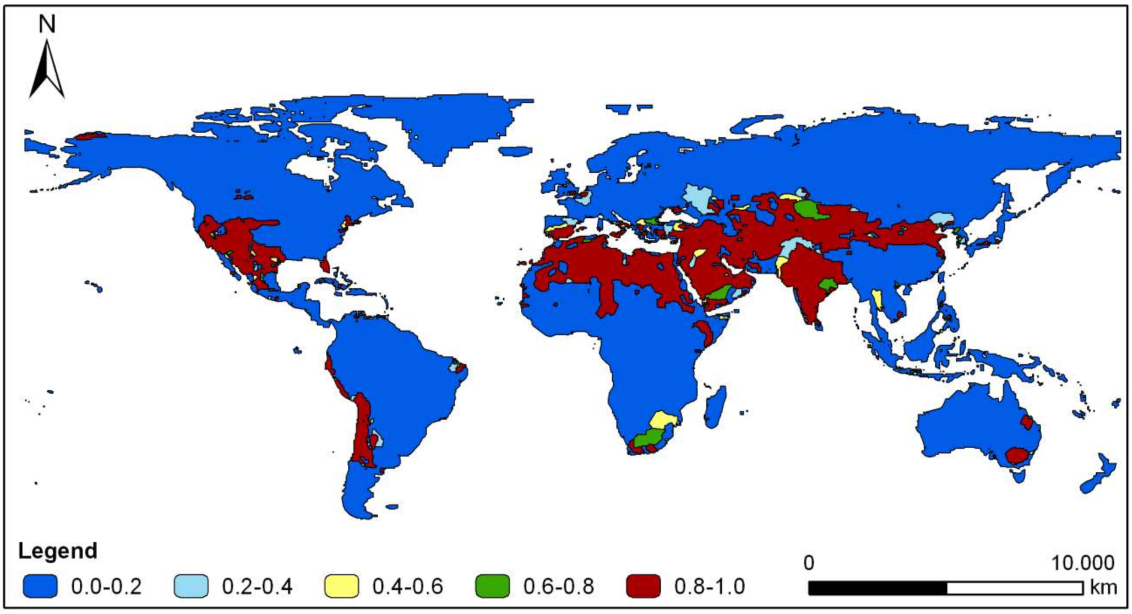


Figure 1.3. World's Water Stress Index (*Source: Own elaboration based on data from Pfister et al. 2009*).

Figure 1.3 shows that the future availability of water to match crop water requirements is confounded in areas with lower rainfall (**arid or semi-arid zones**) in addition to the southern, drier parts of Europe, America, Asia and North Africa. Runoff and groundwater recharge are both likely to decline dramatically in these areas.

Water scarcity is growing besides **salinization** and pollution of groundwater and degradation of water bodies and water-related **ecosystems** are rising. Impacts may include reduction in environmental flows, changes in downstream access to water, or many rivers do not reach their natural end points and **wetlands** are disappearing.

Agriculture's use of water, needs to be more precisely defined to better inform policy decision making. Several organizations have been raising awareness and prompting action on sustainable water management, and some have strengthened institutions and governance. Then, there is an urgent need for better and more effective integration of international initiatives dealing with water management to the lowest **geographic unit** and in involving stakeholders in **planning and decision making**.

Farmers also need to adopt more technical advice and education on best practices, especially when climate change may render past farm practices obsolete. Two important aspects of farm level irrigation efficiency need to be stated and understood by farmers (Seckler et al. 2003): irrigation management and soil moisture conservation. In order to establish mitigated and adapted strategies to achieve sustainable irrigation, farmers need to orientate to: *improve* food security and water use efficiency in areas of water scarcity; *develop* crops or change farm practices where climate change alters temperatures and precipitation (changing planting dates or switching to different crops) (Droogers and Aerts, 2005); *alter* (change or improve) management practices that can contribute to slowing water transport across farmland; and *integrate* sustainable water resource management in agriculture within the broader context of regional land use planning (FAO, 2011a).

To achieve all of these purposes, farmers need to follow some environmental indicators to assess their practices. Taking into account previous (and other) environmental impacts in decision-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. Among the tools offered by life cycle thinking, one of the disciplines that can be used to study the environmental impacts of human activities, such as those from agricultural systems is the methodology of Life Cycle Assessment (LCA). This environmental tool is further explained in the following section.

1.3. Life cycle approach in agriculture assessment

Life Cycle Assessment (LCA) is a systematic, analytic and environmental methodology that provides an environmental profile of all stages ("from the cradle to the grave") of a product, process or service taking into account all relevant flows from input side (use of resources) and on the output side (emissions to air, water and soil, including waste) (ISO 2006 a, b). Thereby, the assessment of the entire life cycle of the product, process or service, encompassing the extraction and processing of raw materials, manufacturing, transportation and distribution, use, re-use, maintenance, recycling, and final disposal. Likewise, the inputs and outputs amounts of the analyzed system are covered in the life cycle inventory phase (LCI), and characterized in the life cycle impact assessment phase (LCIA) which includes impact categories (*midpoints*) and/or damage categories (*endpoints*). **Midpoint** methods characterize impacts in terms of a common unit within their category based on modeled effects (e.g. **climate change** in units of $CO_{2\text{-equivalents}}$) whereas **endpoint** methods characterize a potential damage of the areas of protection (e.g. *human health, ecosystem quality and resource depletion*) (ISO 2006 a, b). LCA tool is becoming attractive for the different companies and organizations due to the great strengthens that it provides: useful information for the decision-maker in order to identify opportunities for environmental improvement during life cycle, an evaluation of trade-offs between different environmental concerns and a marketing differentiation between the companies.

LCA was traditionally applied to analyze industrial production systems, but has been adapted within the last 20 years to assess the environmental effects of agriculture (Ausdley 1997). For agricultural LCA's the majority of studies used cradle-to-farm gate as the system boundary (Cederberg and Stadig 2003). Agricultural LCA analyses all inputs and outputs (traced back to primary resources) that cross a specified system boundary for a certain agricultural system and can systematically identify key areas to improve environmental and economic performance. A standardized LCA methodology for agriculture will help practitioners undertake LCA studies and greatly increase their value by providing results that are comparable between other fields (Harris and Narayanaswamy 2009).

Agriculture is one of the most difficult sector in LCA since it is often complex because its variability, same inputs, different outputs, depending on external conditions (climate, pests), highly site-specific dependency and in addition to the main product, there are usually co-products, so that appropriate environmental impacts need to be assigned to each product (allocation). Additional complexities in agricultural LCA come from fertilizer application. Indeed, balances of soil nutrients such as N/P/K; Chemical fertilizers based on ammonium which use fossil feedstock and source of energy based on fossil fuels; and natural mineral fertilizers as sources of phosphate and potassium (require energy for extraction from the ground, processing, packing and delivery) need careful consideration (Harris and Narayanaswamy 2009).

The amount of GHG emissions emitted in agriculture depends mainly on fossil energy used and on N_2O emissions by soil. The uncertainty associated to this GHG gases, especially in the case of N_2O, is a clear source of variability in the value of GWP associated to an agricultural product.

The choice of indicators would need careful consideration and explanation within the individual LCA study. Traditional LCA studies examined a range of environmental impact categories based on the goals of the study such as commonly categories e.g. global warming potential (GWP), eutrophication potential (EP), air acidification potential (AP), depletion of abiotic resources (DAR), human (HT) and ecosystems toxicity (ET) (Williams et al. 2006). In previous agricultural LCA studies, **water** had not widely been considered as an **impact category** (depends on how the system boundaries are defined in time) which include water quantity impacts like flood and drought risk until Heuvelmans et al. (2005) developed a new impact category 'regional water balance' to address these risks in order to evaluate the impact on water table height (between planting and harvesting). Indicators for water resources based on water quantity (In/Off-stream water use/consumption/depletion indicators) and water quality were proposed by Owen (2001).

Other water concept used in agriculture which provides a bridge between LCA and resource efficiency and effectiveness frameworks, especially in accounting for fresh water flows associated with products and services, is the **water footprint**. It water′ indice does not provide an useful information of relative environmental impact it emits. Following the international LCA standards (ISO, 2006 a, b), this dissertation uses the Water Stress Index (Pfister et al. 2009) as a characterization factor in order to obtain the **water footprint impact assessment** from blue water footprint.

1.4. Decision making and communication to farmers

Climate change and water scarcity in agriculture is a global problem with **local solutions**. There is growing awareness of the need for methods and indicators to evaluate agri-environmental policies and for much more effective integration of international policies and initiatives dealing with climate change emissions and water management. Agriculture has a very close relationship with environment as far as they have common points of production (photosynthesis) and a linkage of where the ecosphere starts and where the technosphere finishes. The "greening farm" will involve stepping up production with **sustainable agriculture** (optimal relationship between agriculture and environment) so that "a given area of land yield more food, while conserving resources, reducing negative impacts on the environment and enhancing natural capital and the flow of ecosystem services" (European Comission, 2012). Only by these green changes can the world feed its citizens through a sustainable agriculture that produces within environmental limits. Sustainable agriculture development strategies should rely on holistic approaches for delineating the dimensions of various problems and finding out the possible solutions.

However, this statement is complex due to the site specificity of many environmental issues, difficulties in valuing and measuring environmental outcomes, and factors outside the control of farmers. Then, policymakers must work together to fill information gaps and ensure that farmers and managers have access to the information they need. By **encouraging** farmers to voluntarily protect and enhance the environment on their farm land, agri-environment measures play a crucial role in meeting society's demand for environmental outcomes provided by agriculture. Sustainable agriculture production should involve the successful management of resources to satisfy the changing human needs and this requires a systematic way of communication in which appropriate strategies and approaches could be planned accordingly.

Sustainable agriculture can be better produced at medium or large scale **farming** because a better efficiency in the use of resources and the easier availability of well prepared technical staff. When we are dealing with small scale farming, a necessary condition for sustainable agriculture is that large numbers of farming households must be motivated to use coordinated resource management (e.g. conserving soil and water resources and controlling the contamination of aquifers and surface water courses). The problem is that, in most places, platforms for collective **decision making** have not been established to manage such resources (Röling, 1994 a,b). So, the success of environmental **sustainable practices** therefore depends not just on the motivations, skills, and knowledge of individual farmers, but on action taken by groups or communities as a whole. In order to achieve this proposal, sustainable agriculture must enshrine new ways of **learning** *(what/how/with whom)* about the world to emphasize the transfer of knowledge by **teaching** farmers (FAO, 2012). This sustainable education involves a *transformation* in the fundamental objectives, strategies, theories, risk perceptions, skills, labour organization, and professionalism of farming; an *investment* in observation/monitoring procedures and life cycles of crops and; a *benchmark* with higher system-level management.

The threat for farmers is that they cannot rely on routine, calendar-based activities if they engage in sustainable farming. For that, an important role of **effective communication** is to make visible the state of the environment and the extent to which present farming practices are untenable. About 2.5 billion smallholders will need the **know-how** and techniques to implement changes (European commission 2012). Hence, farmers require instruments and **easy-friendly tools** with indicators which make more visible the ecological relationships on and among farms in order to help individual farmers assess their own farms situations (Röling, 1993). Combining expert advice together farmers' knowledge (e.g. farmer experimentation) could help farmers overcome major hurdles in adapting their farms and then, it would demonstrate the feasibility of sustainable farming practices.

To sum up, sustainable agriculture offers solutions to the challenges we face worldwide. The farmers have a need of improving their environmental and water management in order to reach competitiveness, cost reduction and achieve environmental responsibility but they need high knowledge of the system and perceive relevant benefits from the implementation. So, communication needs to be planned in such a way that the information should reach to each and every farmer by the best use of various ways strategies to be used for fulfilling the needs and solving the problems of the farming community.

1.5. State of the art of water and climate change standards, protocols and methods

As it has been seen in previous sections, sustainable agriculture is dealing with two main challenges: **climate change mitigation** and **water management improving**. During the last two decades several water and carbon standards and specifications have been developed to assess and attempt these two fundamental issues.

The complexity of agriculture sector makes difficult the definition and use of environmental indicators. Although standard LCA becomes a powerful environmental tool, it lacks in suitable methodology to face issues from agricultural sector. LCA application in agricultural sector is still developing but has strongly progressed during last years in order to achieve a common understanding of the goals, structure, challenges and procedural issues. Among the different impact categories climate change is one of the most successful applied, several protocols, standards and normative have been constructed and demonstrated during last decade in order to account the GHG emissions (Figure 1.4).

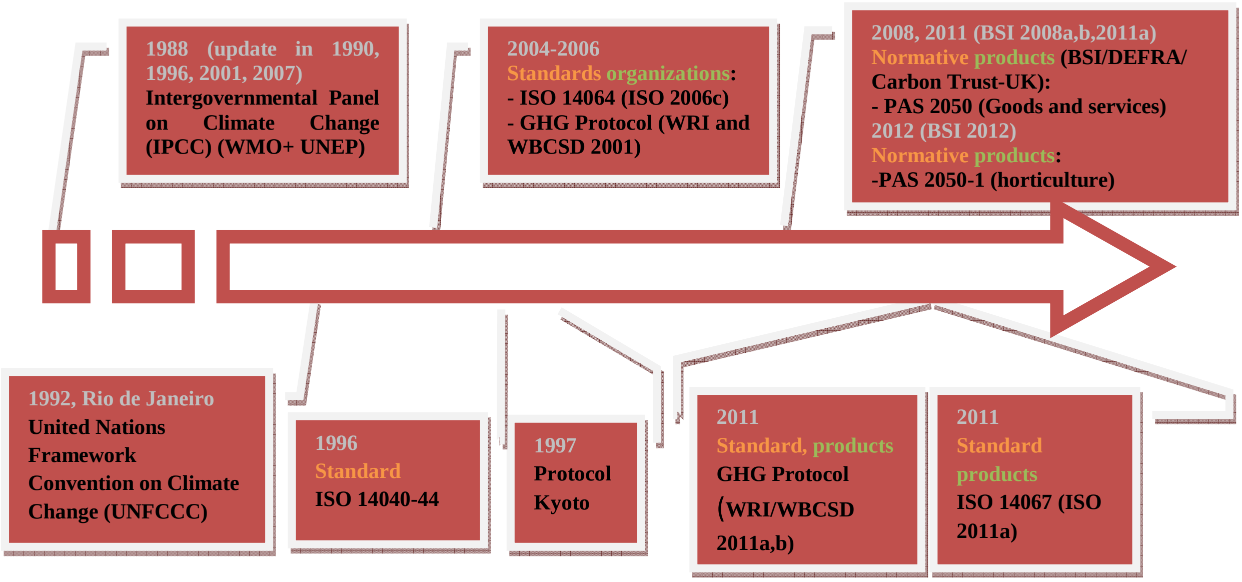


Figure 1.4. Several climate change /carbon footprint standards, protocols and specifications developed from 1992 to 2012.

The Intergovernmental Panel on Climate Change (IPCC) was established in 1988 by the World Meteorological Organization (WMO) and under the auspices of United Nations Environment Programme (UNEP) for the purpose of assessing “the scientific, technical and socioeconomic information relevant for the understanding of the risk of human-induced climate change (recognition of the problem of global warming). Through the IPCC, climate experts from around the world synthesize the most recent climate science findings every five to seven years and present their report to the world’s political leaders. The IPCC has issued comprehensive assessments in 1990, 1996, 2001 and most recently the Fourth Assessment Report (AR4) released in 2007 (IPCC, 2007).

It bases its assessment mainly to inform international policy and negotiations on climate-related issues. The First Assessment Report (FAR) of the IPCC (1990), as well as a supplemental report prepared in 1992, supported the establishment of the **United Nations Framework Convention on Climate Change (UNFCCC, 1997)** at the United Nations Conference on Environment and Development (UNCED, commonly known as "The Earth Summit") held in Rio de Janeiro, Brazil, in 1992.

The significantly strengthened Second Assessment Report (SAR, 1996), along with additional special materials on the implications of various potential emission limitations and regional consequences, provided key input to the negotiations that led to the adoption of the **Kyoto Protocol** to the UNFCCC in 1997. The Kyoto Protocol is an international agreement that establishes binding targets for reducing the heat-trapping emissions of developed countries. Given that this Protocol established a goal of an average reduction of 5.2% of world GHG emissions for the period 2008-2012, related to emissions in 1990 (United Nations Framework Convention on Climate Change (1997), every country depending on the developed level of the country, needs to achieve the fixed target.

This fact makes vital to know the produced emissions by country, and so on, which metrics and patterns are needed to follow them. At country level, several normative which explain how the GHG emissions have to be calculated have appeared (e.g. France (Bilan de Carbone launched by Agence d l'Environnement et de la Maîtrise de l'Energie, 2006 (ADEME, 2006)), United Kingdom (PAS 2050 established in 2008 (BSI 2008 a,b, 2011a, 2012)), Japan (Carbon Footprint Program in 2008 and launched the Labelling Pilot Project in April 2009)). Therefore, these normatives have been splitted up in two levels: *organization* and *product*. This dissertation focusses on the second level.

During 2011 and 2012, these normative have gained force:

- **Specification**:
    - PAS 2050 (BSI/DEFRA/CarbonTrust) (BSI 2008a,b, 2011a,2012). It is based on the LCA methodology ISO 14040-14044 (ISO 2006 a,b) and ecolabeling normative ISO 14021 (ISO 2011 c).

  - PAS 2060 (BSI/DEFRA/CarbonTrust) (BSI 2010). It is based on specifications to demonstrate the carbon neutrality in organizations.
- **Standards**:
  - for *products*, ISO 14067 (ISO 2011a) (scope 1 and 2) based on LCA methodology and GHG Protocol.
  - for *organizations,* ISO 14069 (ISO 2011b), carbon footprint.
- **Protocols**: GHG Protocol (a Corporate Accounting and Reporting Standard). International Protocol by WRI/WBCSD, to calculate the GHG emissions based on ISO 14064 (ISO 2006c).

On the other side impacts of water use have been usually of only minor concern to LCA practitioners and a comprehensive impact assessment has been lacking for decades (Jolliet et al. 2003, Koehler 2008). The main reasons have been that the emphasis is traditionally placed on environmental issues that reflect societal priorities in industrialized countries (e.g. global warming) and another reason for the disregard of water resources might be that water use requires regional distinction, and regionalization is the unsolved issue in LCA (Pfister 2011). However, several initiatives launched by Water Footprint Network, ISO, WULCA project (www.wulca-waterlca.org) from UNEP/SETAC Life Cycle Initiative among others have also been developed regarding water footprint (Figure 1.5). In practice, databases (Quantis water database, ecoinvent v3) can now support most methods, but stress assessment methods (including quality) are still lost between inventory databases and impact assessments software.

Water footprint (WF) in LCA includes both, inventory of water volumes and comprehensive impact assessment related to water. WF considers together water quantity and quality and can be assessed regionally. WF profile is the result of the combination of water availability with impacts from water pollution. The international standard for Water Footprint, ISO 14046, is still on development, and will specify the principles, requirements and guidelines of assessing and reporting WF.

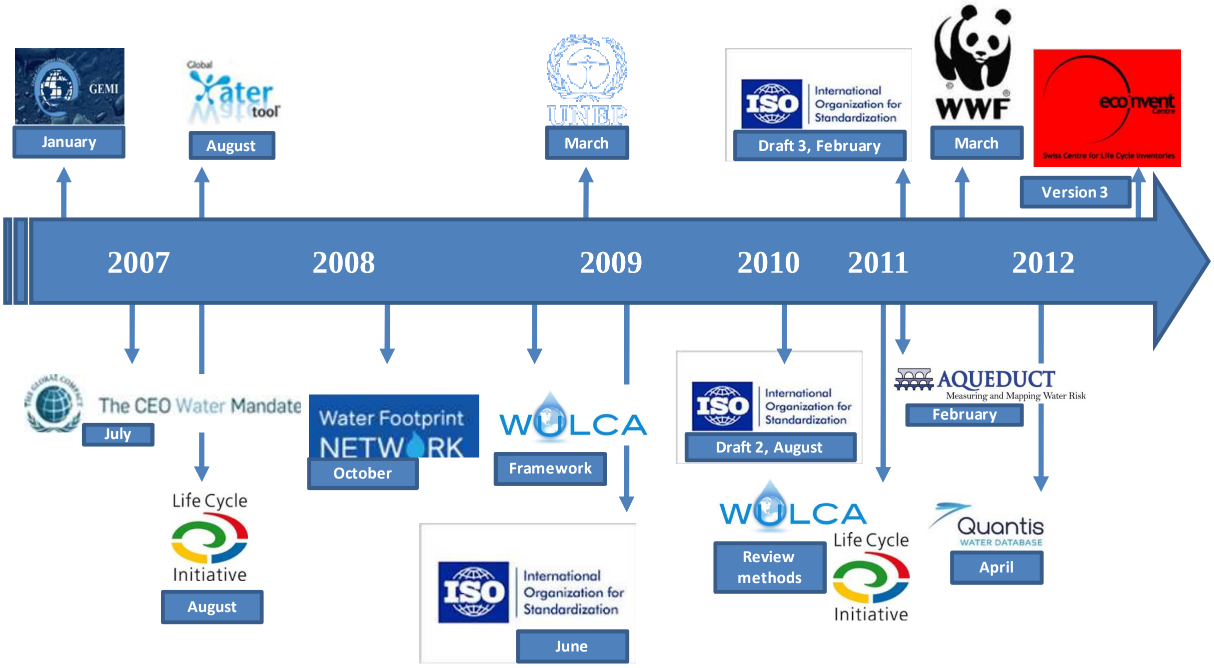


Figure 1.5. Several water footprint standards, database and normatives developed from 1992 to 2012.

Freshwater use has gained attention in LCA and some methods addressing water scarcity and use have been developed to evaluate. Recently, new methodologies were developed that propose freshwater use inventories (Boulay et al. 2011a; Peters et al. 2010, WBCSD 2010). Indeed, the use of freshwater can generate potential impacts to humans, the ecosystems and resources and these impacts can be related to water scarcity, water functionality, water ecological value and water renewability rate (Kounina et al. 2012). Several studies (Milà i Canals et al. 2009, Motoshita et al. 2010a, Pfister et al. 2009) have assessed the potential environmental impacts of freshwater use considering various cause-effect relationships.

Freshwater use ultimately leads to an aggregated impact on *human health* that depends on the level of economic development and welfare (Boulay et al. 2011b; Bayart 2008) and it is normally expressed in disability-adjusted life years

(DALY) (Boulay et al. 2011b; Motoshita et al. 2010a,b; Pfister et al. 2009). Freshwater use can also affect the ecosystem, for instance by changes in the river, lake or wetland flow quantity (e.g. surface water withdrawals), changes in the level of the groundwater table (e.g., groundwater withdrawal), changes in flow regimes (e.g., turbined water use) and loss of freshwater quality (Kounina et al. 2012). Impacts on *ecosystem quality* are commonly expressed in potentially disappeared fraction of species (PDF) on given surface or volume during a given time (PDF·m^2·yr or PDF·m^3·yr) (van Zelm et al. 2011, Hannafiah et al. 2011).

Consumption of all freshwater types as well as withdrawal and release of fossil groundwater can respectively lead to overuse of renewable water bodies or exhaustion of non-renewable fossil groundwater (Kounina et al. 2012). In order to characterize the impact on *resources*, different approaches expressing the resource damage have been developed at: *midpoint level* with abiotic depletion potential given in antimony equivalents (Sb-eq) from Milà i Canals et al. (2009) and *endpoint level*: in megajoules (MJ) surplus energy (Pfister et al. 2009) or exergy-based methods given in mega joules of exergy (MJex) (Boesch et al. 2007).

Figure 1.6 shows different methods developed at inventory level, some water indices (e.g.Withdrawal/consumption-to-availability) and at midpoint and endpoint categories classified in Human Health, Ecosystem Quality and Resources Depletion.

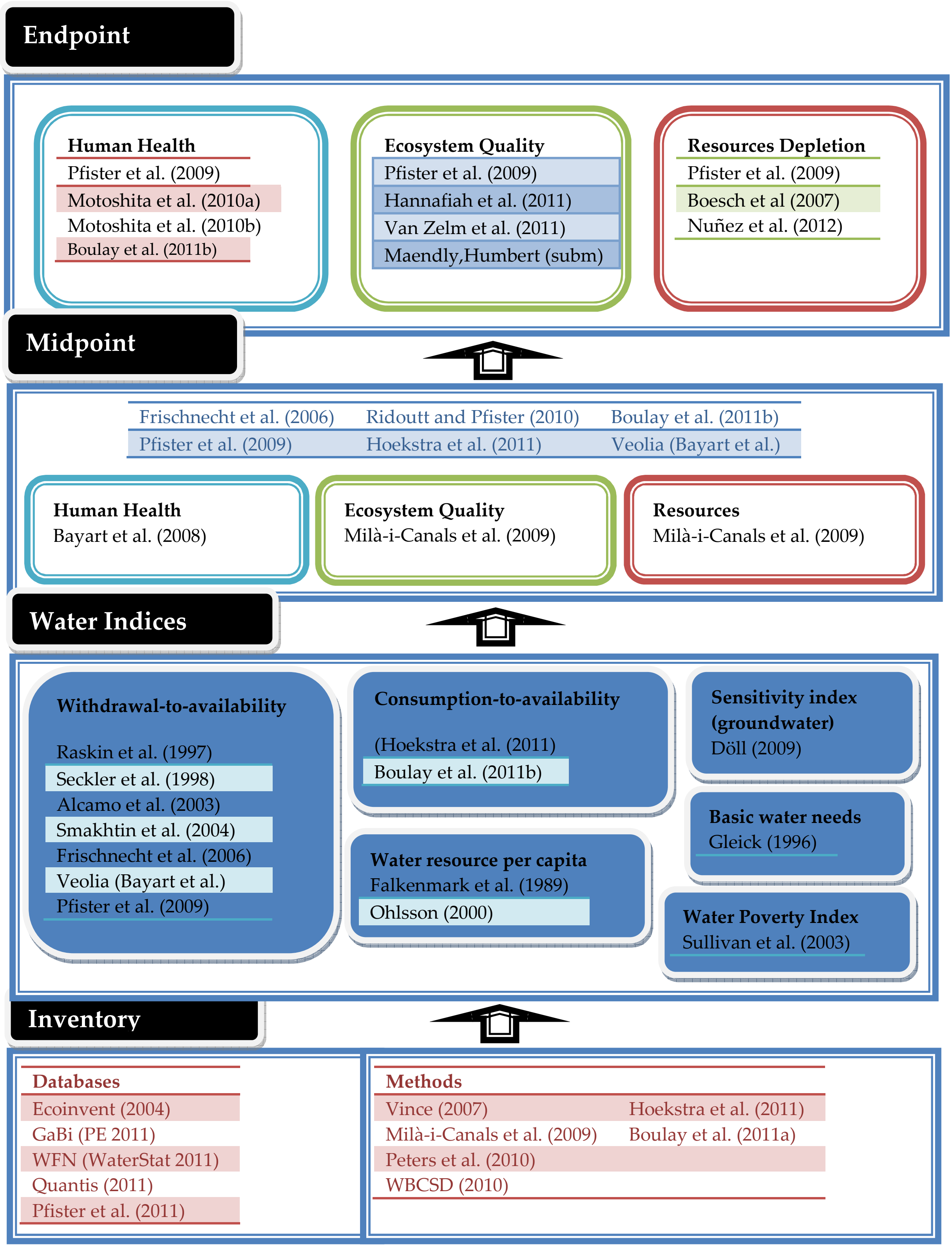


Figure 1.6. Available freshwater use inventory and impact assessment methods with classification for the three areas of protection. (*Source: Kounina et al. 2012*)

As a response of the political concern as well as the increasing consumer demand for environmentally friendly food products, a friendly-use tool becomes essential to assess the carbon and water footprint in the agriculture field. This dissertation deals with a solution to the current necessity of simplifying Life Cycle Impact Assessments water methodologies and proposes a robust and effective tool in agricultural sector.

1.6. Motivation of the dissertation

Taking into account the problem statement proposed in previous sections, this dissertation tries to deal with water management and climate change in agriculture. Regarding the water management, a comparison of different methods to perform an environmental assessment of freshwater use in agriculture was studied with Ebro Basin' cases study (Chapter 3). As a conclusion of this chapter, three main issues were proposed to research in the dissertation:

a) There are a wide range of water methodologies in order to assess the farmer's water management. However, choice of one method over the other has been the most difficult task for them. Therefore, there is a need to propose a clear methodology for agricultural stakeholders which combines water, carbon and agronomical assessment (Chapter 5) and a user-friendly tool to carry out a decision making easily (Chapter 6).

b) How the water irrigation used from natural resources (e.g. wetlands) for agriculture could damage in autochthon biodiversity (Chapter 7).

c) The concern to propose solutions in order to avoid the depletion of water resources for agriculture irrigation by using reclaimed water from Water Urban Cycle (Chapter 8).

Facing on the climate change, a LCA case study based on the production of ethanol fuel from sugarcane (Chapter 4) demonstrates the important contributors in Global Warming assessment from agriculture during all the processes. Therefore, the selection of the most suitable alternative in LCA taking into account the inputs of the process and the uncertainty of data motivated the proposal framework for decision making under probabilistic uncertainties in LCA (Chapter 9).

Given that the impacts from water use, water availability and many geographical factors are highly dependent on the location of the system under study. Spatial variability of water is assessed in this dissertation in order to determine the severity of the environmental effects from water.

Since the southern European countries are the major producers of horticultural products, several cases studies proposed in this dissertation have been located in Spain (a country located on the top of the major producers list). Furthermore, the current data provided in this dissertation came from the real farms and plants assessed in MUSA and SOSTAQUA projects which are located in North-East of Spain, mainly in Ebro watershed.

The cooperation in international projects between researches groups from Universitat Rovira i Virgili and Universidad Nacional del Tucumán has encouraged to propose a case study developed in Chapter 4.

The claim in addressing key aspects for developing and improving LCIA methods and Cf for water use promoted my research stay in Switzerland. A Chapter 7 of this thesis, devoted to improve impact assessment methodology, has been developed in the frame of the European "Life Cycle Impact Assessment Methods for Improved Sustainability Characterization of Technologies" (LC-IMPACT, http://www.lc-impact.eu/), Contract No. 243827.

Apart from these international initiatives, the thesis by Núñez (2011) is the other important background to this dissertation that carried out an environmental assessment of the environmental impacts of water consumption on energy crops grown in Spain as well as the potential desertification impacts and inclusion of soil erosion impacts.

Also, the thesis focus on the development of LCA methodology to be applied in agriculture by Antón (2004) has a huge contribution to the agronomical aspects for this dissertation.

In addition, the possibility of working together URV and professionals from Centre d'assessoria Dr Labferrer S.L., let to develop a robust quantitative tool and a keystone in decision making by producers and stakeholders.

The Environmental Management Group (AGA)'s background (Tarragona) has motivated the proposal solution to provide water source for irrigation in agriculture (Chapter 8). For instance, the collaboration in the national project "SOSTAQUA" during the period 2007-2010 which focused on life cycle assessments about different water technologies (potable water treatment plants, wastewater treatment plants and desalination plants) promoted to propose a holistic Urban Water Cycle with reclaimed water for irrigation.

All these issues have been taken into account in the methodological improvements proposed in this dissertation.

1.7. Objectives of the dissertation

The main objective of this dissertation is to account and assess on-farm environmental sustainability based on water footprint and LCA methodology. In addition, the present dissertation tries to contribute in the methodological development of LCA and to provide a tool to help agricultural stakeholders in their decision making.

In order to achieve these main aims, several goals are outlined:

- To provide a comprehensive **overview of impact assessment methods** to address the impact of **water consumption in ecosystems quality** through experimental case studies of different crops.
- To assess the life-cycle stages of **biofuel production** from sugarcane with **major contribution** in environmental impacts through LCA study taking as a case study different real pathways located in Argentina.
- To assess the **environmental performance of traditional farming systems** and other aspects of several agricultural cultivation options, focusing on water management and climate change.
- To provide a **user-friendly tool** based on LCA guidelines to map technological advancement and operational practices to improve eco-efficiency performance of food supply chains and gives a solution to sustainable management of water and carbon dioxide assessment in agriculture.
- To develop a **Life Cycle Impact Assessment** (LCIA) **method** to evaluate the environmental impacts associated with the consumption of groundwater on **salinity and biodiversity damage in a Spanish coastal wetland.**
- To address an **environmental solution** in order **to take advantage of using reclaimed water** from tertiary treatment in the Urban Water Cycle to be used as **irrigation sources in agriculture.**
- To propose a rigorous approach to aid practitioners in the selection of alternatives when the **LCA analysis** is **affected by** several sources of **uncertainty.**

Chapter 2

Material and Methods

Chapter 2 presents an overview of the dissertation's methodological aspects. First, the sustainable assessment tool applied in the dissertation, which is under a life-cycle-thinking approach, is presented: *Life Cycle Assessment (LCA)*, which is more broadly described. In addition, three other tools are briefly explained in combination with LCA: Environmental Risk Assessment (ERA) (mainly ecotoxicological impacts), Geographic Information System (GIS) and Uncertainty Assessment (mainly Monte Carlo Simulation (MCS)).

This chapter is structured as follows:

- Methodological approaches overview
- Life Cycle Assessment tool
- Combining LCA with other tools

2.1. Methodological approaches overview

Sustainable development is understood as satisfying the needs of the present generation without compromising the needs of future generations taking into account three aspects: economic, environmental and social. For the world to make substantial progress toward becoming a safer planet, it is necessary to introduce environmental considerations in all aspects of management practices for all phases of production, marketing, use and end of life of a product (Sonnemann et al. 2004). Based on this statement, different general objectives have been formulated as programs that intend to encompass the idea of good environmental management as shown in Figure 2.1.

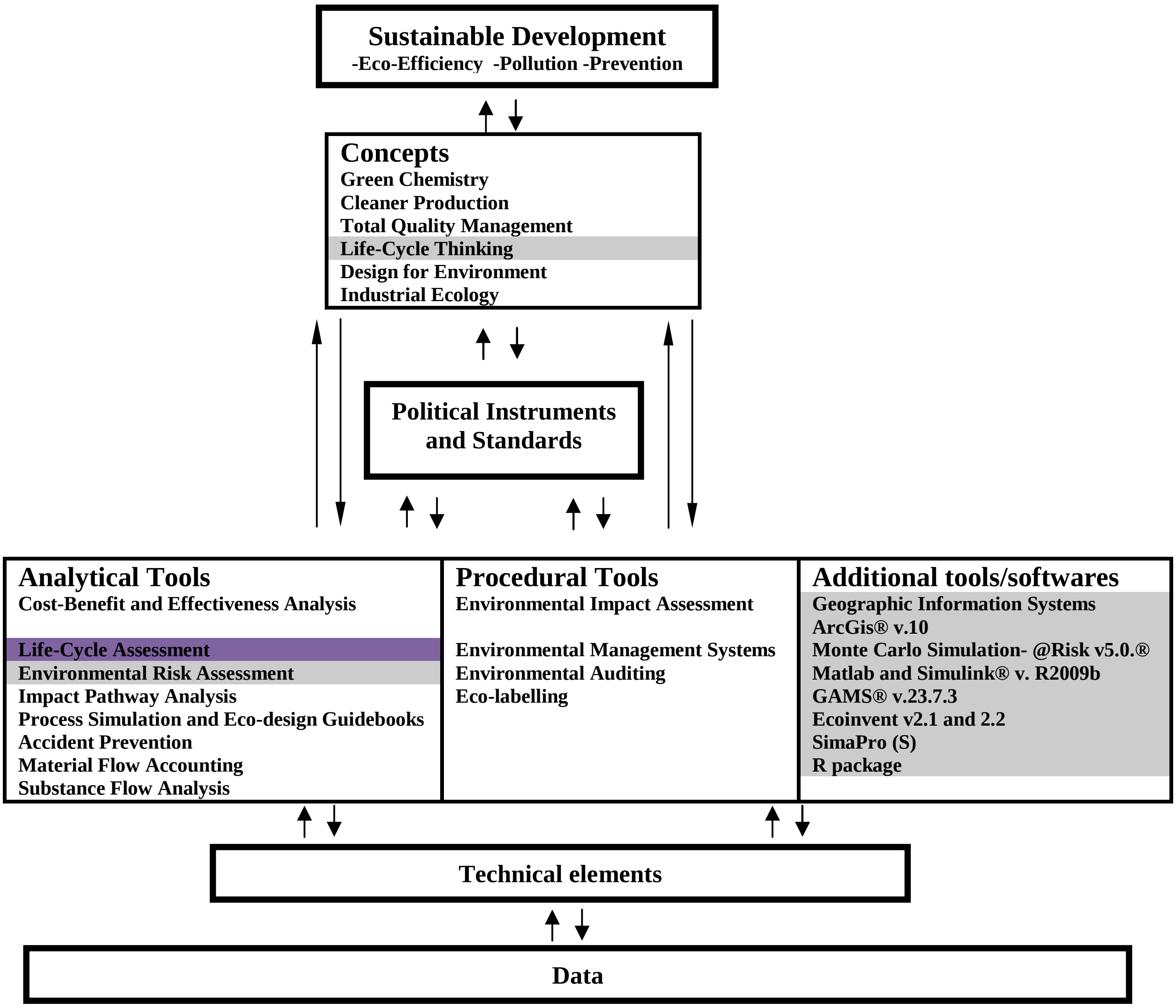
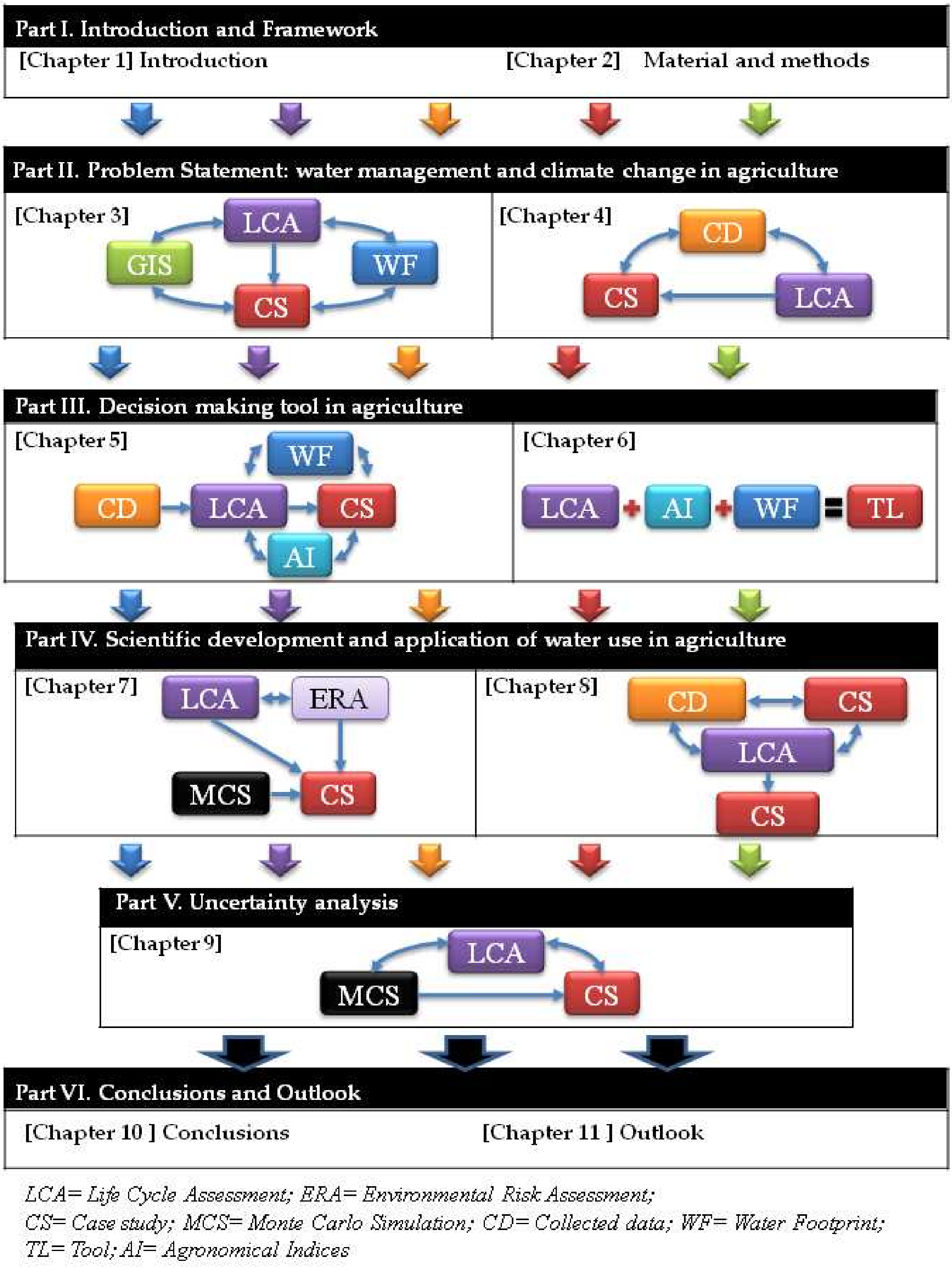


Figure 2.1 Conceptually related methods in environmental management. (*Source: Adapted from De Smet, B et al. 1996 and Sonnemann et al. 2004*).

In order to establish environmental strategies are reflected in and can be applied to various **concepts** between them **Life Cycle Thinking** (LCT) is located. LCT is a way of addressing environmental issues and opportunities from a system or holistic perspective and evaluating a product or service system with the goal of reducing potential environmental impacts over its entire life-cycle (UNEP 1999). The preceding concepts (Figure 2.1) including LCT, which have been developed to direct environmental management, need tools to transfer them into action and make environmental aspects more concrete, taking into account economical, social and technological information. In accordance with Society of Environmental Toxicology and Chemistry (SETAC) has distinguished the following types of tools (De Smet et al. 1996): political instruments and standards, analytical tools and procedural tools. Hence, the application of these tools provides consistent environmental information that facilitates adequate decision-making toward sustainable development. Based on the idea of the interaction between these methods in environmental management, this dissertation proposes the combination of them during the execution of different chapters.

Life Cycle Assessment (LCA) is a global and holistic analytical tool that can be combined with other analytical, procedural and instrumental tools. Although this dissertation is based on LCA methodology, it additionally is combined with other analytical tool as for example Environmental Risk Assessment (ERA) or other spatial and random tools as Geographic Information Systems (GIS) and Monte Carlo Simulation (MCS), respectively.

Figure 2.2 represents the methodologies integration during the present dissertation. Combining with LCA methodology, GIS was used in chapter 3, indirectly, ERA was used in terms of ecotoxicity during the execution of chapter 7, and the MCS was managed in chapter 7 and 9 in order to assess the uncertainty in LCA. In addition, the Water Footprint (WF) accounting was assessed in chapters 3, 5 and 6.



*LCA= Life Cycle Assessment; ERA= Environmental Risk Assessment;
CS= Case study; MCS= Monte Carlo Simulation; CD= Collected data; WF= Water Footprint;
TL= Tool; AI= Agronomical Indices*

Figure 2.2 Map structure of methodologies applied in the present dissertation.

2.2. Life Cycle Assessment tool

(i) Background

An overview of LCA history can be found in Assies (1992), Vigon et al. (1993), Pedersen (1993) and Boustead (1992). The first Life Cycle Assessment concept emerged during 1960s with H.Smith's calculations of energy requirements for manufacturing chemical products (Vigon et al. 1993). Later, other studies predicted the effects of an increase in population, energy and material resources (Club or Rome (1972); Meadows et al. (1972)) which together oil crisis of 1970s encouraged more detailed studies, focused on the optimum management of energy resources. But it would not be scientifically interesting until 80s to become an environmental topic for several pioneering companies (i.e. the Swiss Federal Laboratories for Materials Testing and Research (EMPA) and Institute of Environmental Sciences (CML) (Guinée et al. 1993). The maximum relevance and the fundamental basis for environmental LCA arrive in 1990 through Society of Environmental Toxicology and Chemistry (SETAC). From then, several international reference guides for LCA based on ISO 14040 series were proposed until current LCA framework from 2006 with two standards: ISO 14040:2006 (ISO 2006a) and the ISO 14044:2006 (ISO 2006b).

Nowadays, several international initiatives are ongoing to build a consensus and provide recommendations (i.e. Life Cycle Initiative of the United Nations Environment Programme (UNEP), Society of Environmental Toxicology and Chemistry (SETAC) and International Reference Life Cycle Data System (ILCD) of the European Commission) because LCA method is still under development.

(ii) Definition

LCA is a structured, comprehensive and internationally standardized method which is the compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle (ISO 2006a). It is a tool to evaluate the environmental performance of any goods or services (products) which quantifies all relevant emissions (e.g. air, water and soil), consumed resources (e.g. water) and the related environmental impacts (e.g. Global Warming Potential). LCA takes into account a product's full life cycle: from the extraction of resources and processing of raw materials, through production, use, maintenance, recycling, up to the disposal of remaining waste (JRC and IES 2010b).

(iii) Methodology

In accordance with ISO 14040 (2006) and 14044 (2006) standards and the international life cycle data system (ILCD) handbook (JRC-IES 2010a,b), the evaluation and standardized framework commonly applied in LCA involves four phases (Figure 2.3): a) Goal and scope definition, b) Inventory analysis, c) Impact assessment and d) Interpretation.

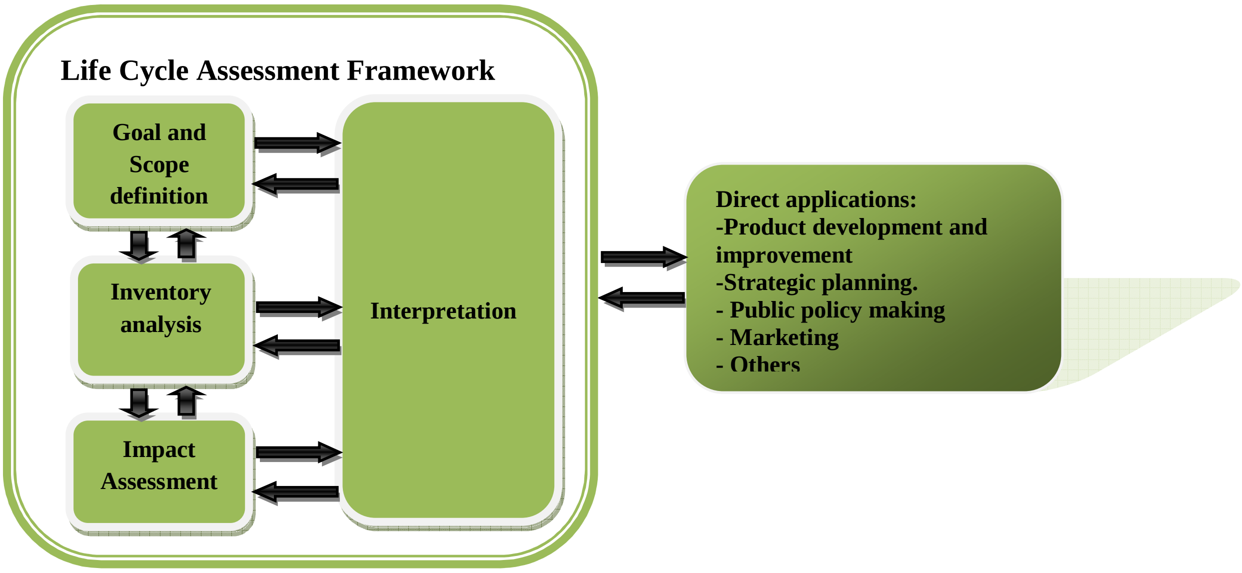


Figure 2.3 Phases of LCA according to ISO 14040 (2006a). (*Source JRC- IES* (2010b)).

Concerning these iterative LCA phases, firstly we need to define the goal of the work, then the initial scope settings are derived (sometimes it needs to be revised), secondly the life cycle inventory of data collection is accounted and the final complex phases are the subsequent impact assessment and interpretation (SETAC (1993); JRC- IES 2010b).

Goal and scope definition: The first stage of the LCA study based on the purposed and the scope of the study, the hypothesis considered and the detailed data requirements are the most important keys to successful assessment (Table 2.1). The goal of the LCA study is formulated in terms of the exact question, target audience and intended application. The scope of the study is defined in terms of temporal, geographical and technological coverage, and the level of sophistication of the study in relation to its goal (Guinée et al. 2001). A system may have a number of possible functions and the one(s) selected for a study depend(s) on the goal and scope of the LCA. The product(s) that are the object of the analysis are described in terms of function, functional unit (FU) and reference flows. The FU is a key element that defines the quantification of the identified functions (performance characteristics) of the product and the system boundaries delimit the unit processes and flows that are going to be included in the system.

Table 2.1 Required specifications to determine the purpose and scope of the study (continued in next page). *(Source: ISO 2006a, JRC- IES 2010b and Guinée et al. 2006).*

Definition of…	Required specifications
Purpose of the study	- Goal of the LCA study - Specifying the intended use of the results (application), the initiator (and commissioner) of the study, the practitioner, the stakeholders and the intended users (target audience) - Method assumptions and impact limitations - Reasons for carrying out the study Life Cycle Inventory decision-context: - Atributtional: specific supply chain+use+end-of-life value chain+ static technosphere - Consequential: generic supply-chain+ consequence of the analyzed decision+ dynamic technosphere

Scope of the study	- Description of the system under study and its functions - Functional unit (e.g. 1kg, 1 ha, 1 m^3)→ reference to which the inputs and outputs are related to ensure comparability of LCA results. - System boundaries: Input-Output flow diagram (elementary flows) - Technical, geographical and time-related representativeness - Allocation procedure rules - Methodology of impact assessment - Selected impact categories
Data requirements	- Data requirements: main types, quality and sources of data - Other information both for the LCI and LCIA steps. - Assumptions and limitations

Inventory analysis: The inventory analysis or the Life Cycle Inventory (LCI) is the phase in which the product system is defined. The LCI involves collecting all data to quantify relevant inputs and outputs of the unit processes of the product system and relating them to the FU of the study (e.g. consume draw materials and energy and produced solid waste). In addition, all emissions released into the environment (to air, water and soil) and resources extracted from the environment (e.g. abiotic/biotic resources and land transformation/occupation) along the whole life cycle of a product are accounted in LCI table. Other step in LCI phase is performing allocation steps for multifunctional processes. The multifunctional outflows (products) of a given process that may produce (co-products) who's environmental load needs to be allocated over the different functions and outputs.

Impact assessment: Life Cycle Impact Assessment (LCIA) is the phase in which the set of results of the inventory analysis is further processed and interpreted in terms of environmental impacts and societal preferences. To this end, a list of impact categories is defined, and models for relating the environmental interventions to suitable category indicators for these impact categories are selected. The actual modeling results are calculated in the characterization step, and an optional normalization serves to indicate the share of the modeled results in a worldwide or regional total. Finally, the category indicator results can be grouped and weighted to include societal preferences of the various impact categories (Guinée et al. 2001).

LCIA aims to connect LCI on the basis for modeling in LCA (impact pathways or environmental mechanism or cause-effect chain) with their potential environmental damages (Figure 2.4).

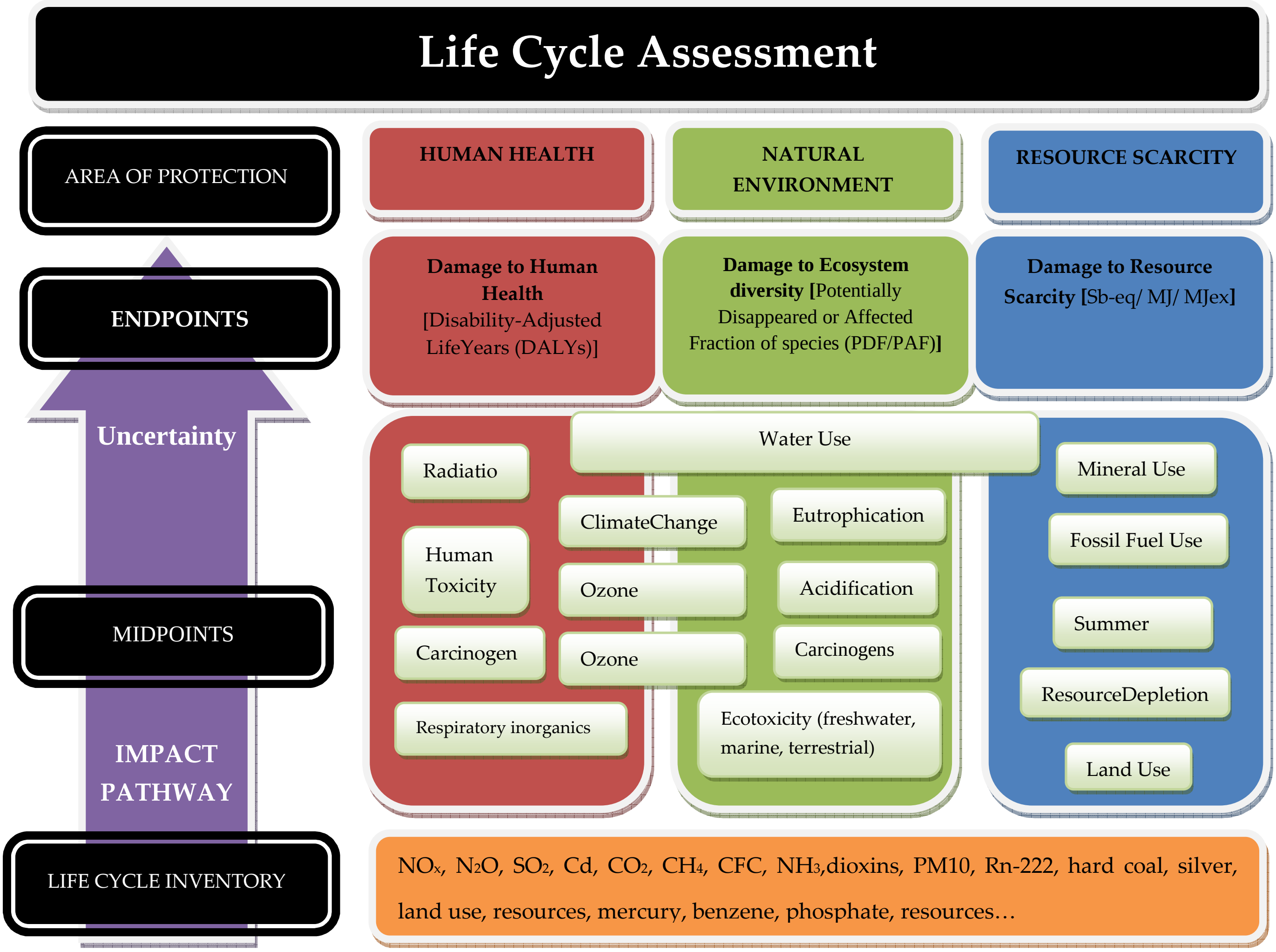


Figure 2.4. Environmental mechanism and schematic steps of the LCIA framework (JRC-IES 2010a).

Figure 2.4 represents the LCIA which consists of several steps that are briefly described below:

Selection of characterization methods: category indicators, characterization models and factors (mandatory):

a) Category indicators: The quantification of the environmental impacts could be mainly distinguished in two possible levels: the *midpoint* where the impact category indicator is defined close to the intervention (i.e. problem-oriented, such as global warming potential) and *endpoint approach* (i.e. damage-oriented) which model the cause-effect chain up to the environmental damages and indicators are close to recognizable values for society, also called areas of protection (AoP). The ILCD handbook (JRC-IES 2010 a,b) distinguishes three AoP: human health, natural environment and natural resources. Figure 2.5 shows a schematic illustration of the definition of midpoint and endpoint levels (Sonneman et al. 2004).

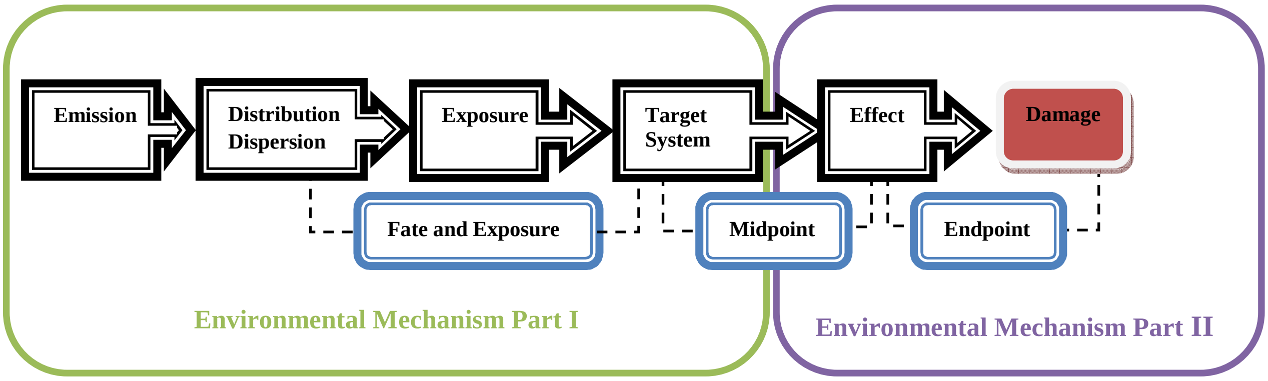


Figure 2.5 Schematic illustration of the definition of midpoint and endpoint levels.

(Source: Adapted from Olsen, SI et al. 2001).

The present doctoral thesis uses different impact categories (Table 2.2):

- *Midpoint categories* (Chapter 3, 4, 5, 7, 8 and 9): Climate change, Acidification potential, Eutrophication potential, Photochemical oxidation, Stratospheric ozone depletion, Depletion of abiotic resources, Freshwater aquatic ecotoxicity and Terrestrial ecotoxicity – which had been defined by the CML 2001 (Guinée et al. 2001) and cumulative energy demand (Frischknecht and Jungbluth 2003), as an energy flow indicator.

- *Endpoint categories* (Chapter 3, 7): Ecoindicator99 (Goedkoop and Spriensma (2000)) and ReCiPe (Goedkoop et al. (2009)).

Table 2.2 Environmental impact categories considered during the dissertation (continued in next pages). (*Source: Goedkoop and Spriensma (2000); Guinée et al.* (2001), *JRC-IES 2010 a,b and Frischknecht and Jungbluth* (2003)).

Category impact	Acronym	Description	Unit
Midpoint Categories			
Abiotic depletion potential	**ADP**	"Abiotic resources" are natural resources such as iron ore, crude oil and wind energy, which are regarded as non-living. It is concerned with the protection of human welfare, human health and ecosystem health. It is related to the extraction of minerals and fossil fuels due to inputs into the system.	kg Sb eq.
Acidification potential	**AP**	AP is caused by atmospheric deposition of acidifying substances generated largely from emissions of nitrogen oxides (NOx), sulphur dioxide (SO_2) and ammonia (NH_3), the latter contributing to acidification after it is nitrified (in the soil). Acidifying substances cause a wide range of impacts on soil, groundwater, surface water, organisms, ecosystems and materials (buildings).	kg SO_2 eq.
Eco-Toxicity	**ET**	Substances such as toxic heavy metals (Cd, Pb, Hg), persistent organic compounds (dioxins and furans, PCDD/Fs, polycyclic aromatic hydrocarbons, PCBs, etc.), and organic substances (PVC) that are emitted to the environment can accumulate in organisms and cause effect different types of damage. ET is expressed as the percentage of all species present in the environment living under toxic stress. The potentially affected fraction (PAF) is used (Van de Meent and Klepper, 1997) as an indicator and corresponds to the fraction of a species exposed to a concentration equal to or higher than the no-observed-effect concentration (NOEC).	PAF or NOEC
Eutrophication potential	**EP**	EP includes all impacts due to excessive levels of macro-nutrients in the environment caused by emissions of nutrients into the air, water and soil. It is based on the stoichiometric procedure of Heijungs et al. (1992). Fate and exposure is not included.	kg PO_4^{-3}eq.
Fresh water aquatic ecotoxicity potential	**FWAE**	FWAE refers to impact on freshwater ecosystems, as a result of emissions of toxic substances into to the air, water and soil. It is calculated with USES-LCA, describing fate, exposure and effects of toxic substances.	kg 1.4 DCB eq.
Freshwater Ecosystem Impact	**FEI**	FEI is a midpoint impact category proposed by Milà i Canals et al. (2009). It has been suggested, which is linked to ecosystem health at a damage level. It needs to be highlighted	m^3 o kg eco/kg crop

		that variation in seasonal flows is not considered in the FEI.	
Global warming potential	**GWP**	GWP is defined here as the impact of human emissions on the radiative forcing of the atmosphere. This may in turn have adverse impacts on ecosystem health, human health and material welfare. Climate change is related to emissions of greenhouse gases into air. The characterization model as developed by the IPCC is selected for development of characterization factors. Factors are expressed as for time horizon of 100 years.	kg CO_2 eq.
Land Use	**LU**	LU covers a range of consequences of human land use. It reflects the damage to ecosystems due to the effects of occupation and transformation of land. Occupation of land can be defined as the maintenance of an area in a particular state over a particular time period. Transformation is the conversion of land from one state to another state, e.g. from its original state to an altered state or from an altered state to another altered state.	m^2·year
Marine aquatic Ecotoxicity potential	**MAE**	MAE refers to impacts of toxic substances on marine ecosystems, as a result of emissions of toxic substances into the air, water and soil. It is calculated with USES-LCA, describing fate, exposure and effects of toxic substances.	kg 1.4 DCB eq.
Ozone depletion potential	**ODP**	ODP of a substance is a relative measure for the potency to form EESC (Equivalent Effective Stratospheric chlorine). The ODPs are equivalency factors that encompass the atmospheric residence time of ozone depleting substances, the formation of EESC and the resulting stratospheric ozone depletion. Because of stratospheric ozone depletion, a larger fraction of UV-B radiation reaches the earth's surface. This can have harmful effects upon human health, animal health, terrestrial and aquatic ecosystems, biochemical cycles and on materials. This category is output-related and at global scale.	kg CFC eq.
Photochemical oxidation potential	**PHO**	Photo-oxidant formation is the formation of reactive chemical compoundssuch as ozone by the action of sunlight on certain primary air pollutants. These reactive compounds may be injurious to human health and ecosystems and which also may damage crops.	kg O_3 eq.
Terrestrial ecotoxicity potential	**TE**	It refers to impacts of toxic substances on terrestrial ecosystems, as a result of emissions of toxic substances into the air, water and soil. It is calculated with USES-LCA, describing fate, exposure and effects of toxic substances.	kg 1.4 DCB eq.
Cumulative energy demand	**CED**	It aims to investigate the energy use throughout the life cycle of a good or a service. This includes the direct as well as the indirect uses. Characterization factors were given for the energy resources divided in: non renewable, fossil and nuclear, renewable, biomass, wind, solar, geothermal and water.	MJ eq.
Endpoint Categories			
Human Health	**HH**	Damages to human healthexpressed as DALY (disability adjusted life years). Four sub-steps are used to establish the link between the LCI table and the potential damages: a) *Fate analysis*, linking an emission to a temporary	DALY

		change in concentration. b) *Exposure analysis*, linking this temporary concentration to a dose. c) *Effect analysis*, linking the dose to a number of health effects. d) *Damage analysis*, linking health effects to the number of years lived disabled (YLD) and years of life lost (YLL).	
Ecosystem Quality	**EQ**	Damages to ecosystem quality expressed as PDF (potential disappeared fraction of species). Two different approaches are used: 1) 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: *a) Fate analysis*, linking emissions to concentration. *b) Effect analysis*, linking concentrations to toxic stress or increased nutrients or acidity levels. *c) Damage analysis*, linking the effects to the increased potential disappeared fraction of plants. 2) 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.	PDF(% plant species·m^{2}·yr)
Resource Depletion	**RD**	Damages to resources expressed as MJ of surplus energy required to extract an additional unit of the resource. Only fossil fuels and minerals are modeled. Two sub-steps are followed: a) *Resource analysis*, linking the resource extraction to the decrease of the resource concentration. b) *Damage analysis*, linking lower concentration to the increased efforts to extract the resource in the future.	MJ surplus energy)

b) Characterization methods (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 (Goedkoop and Spriensma 1999) and ReCiPe (Goedkoop et al. 2009) at the endpoint level. Other novel methods combine both approaches (e.g., IMPACT 2002+, Jolliet et al. 2003).

Following two steps are mandatory:

- *Classification*, corresponds to a process in which all the environmental interventions qualified and quantified in the inventory analysis (inputs and outputs) are assigned on a purely qualitative basis to the various pre-selected impact categories, according to the environmental effects they are expected to contribute. It answers to the question *What does this emission contribute to? (e.g.* global warming potential category includes CO_2, CH_4 and N_2O emissions).

- *Characterization,* where the environmental interventions assigned qualitatively to a particular impact category in classification are quantified in terms of a common unit to all contributions within a given impact category by applying the so-called characterization factors (CFs) (e.g. climate change is kg of $CO_{2\text{-equivalents}}$). Characterization factors are factors derived from characterization model which allows all substances that contribute to this category to be reduced to a single reference substance. This step answers to the question *How much may it contribute? (e.g.* global warming potential reduce CO_2, CH_4, N_2O emissions to an equivalent substance: $CO_{2,\text{equivalent}}$) (Guinée et al. 2002).

Other three steps are considered in this phase, *normalization, groupping and weighting* which are optional according to ISO 14040:2006 and ISO 14044:2006 (ISO 2006 a,b).

Interpretation: Life cycle interpretation is the phase in which the results of the analysis and all choices and assumptions made during the course of the analysis are evaluated in terms of soundness and robustness, and overall conclusions are drawn. The main elements of the interpretation phase are an evaluation of results (in terms of consistency and completeness), an analysis of results (for instance, in terms of robustness) and the formulation of the conclusions and recommendations of the study (Guinée et al. 2001):

- *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.

The holistic role of LCA try completing the life cycle, assessing all environmental effects, involving the need of turning to different tools in order to reach the risk assessment, specifying the local conditions and covering the uncertainties, all of them by choosing the simplest way.

2.3. Combining LCA with other tools

2.3.1 Environmental Risk Assessment and Life Cycle Assessment

2.3.1.1. Environmental Risk Assessment

Risk analysis is defined as a process for controlling situations where a target could be exposed to a hazard (OECD 2003). It consists of three parts: *risk management* is related to decision-making processes involving considerations of political, social, economic, and technical factors with relevant *risk assessment* as the process intended to calculate or estimate the probability of an adverse effect in an organism, system, or population under specified circumstances by exposure to an agent and *risk communication* related to the exchange of information about risks between risk assessors, public managers, policy makers, interested groups, and the general population) (Rovira et al. 2012) (Figure 2.6).

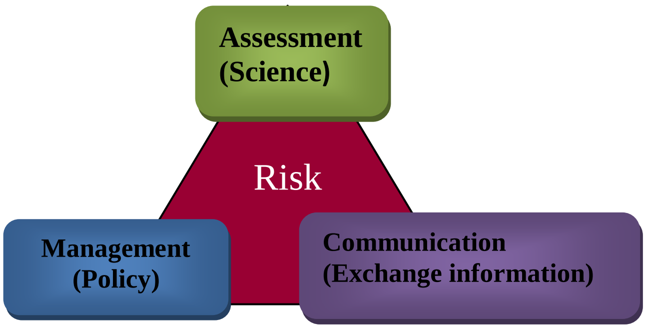


Figure 2.6 Risk analysis structure (adapted from FAO (Fazil 2005))

Risk assessment is a tool used to organize, structure and compily scientific information in order to help identify existing hazardous situations, anticipate potential problems, establish priorities and provide a basis for regulatory controls and /or corrective actions (Sonneman et al. 2004).

Environmental Risk Assessment (ERA) consists of evaluating the probability that adverse effects on the environment or human health occur on may occur as a consequence of exposure to physical, chemical or biological agents. This risk assessment process entails a sequence of four actions outlined following: a) *Hazard identification* consists in identifying the type and nature of adverse effects caused by the agent in the receptor; b)*Hazard characterization or dose–response assessment* is the qualitative and quantitative description of the inherent properties of an agent or situation having the potential to cause adverse effects including the severity of adverse effects related to the amount and condition of exposure to an agent/substance; c) *Exposure assessment* is the estimation of the concentrations/doses to which human populations or environmental compartments (aquatic, terrestrial and air) are or may be exposed and d) *Risk characterization* is the estimation of the incidence, severity and probability of occurrence the adverse effects (Rovira et al. 2012; Sonneman et al. 2004).

2.3.1.2. LCA Risk- Based Toxicological Indicators

In life-cycle assessment (LCA), it is desirable to compare quantities of chemicals released into the environment in terms of the risk and consequences of toxicological effects. LCA should provide indicators of toxicological effects based on the relative risk and associated consequences of chemicals that are released into the environment (Hogan et al. 1996; Assies 1997; Udo de Haes et al. 2002; Pennington et al. 2004a). LCA’s provide insights for products that are complementary to those of many regulatory-, site-, or process-oriented risk assessments (Udo de Haes et al. 2002; Pennington et al. 2004 b). Whether or not current regulatory limits will be exceeded at specific locations or points in time by these exchanges is not the focus of an LCA (Pennington et al. 2006).

In applications like LCA, the characterization factors should reflect cumulative risk, the risk integrated over time and space that is associated with the release of a quantity of chemical into the environment (full extent of risk). The desire to consider cumulative risk in LCA is a fundamental difference from many regulatory approaches (like Environmental Risk Assessment), which focus more on peak exposures compared to acceptable thresholds (Pennington et al. 2006).

Emissions in LCI are reported in terms of the mass of each chemical released at each stage of a life cycle to provide a specific amount of a product. For the impact analysis, the inventory emissions (mass of each chemical) are multiplied by a "characterization factor" (Cf) providing an impact indicator for an environmental class representing an environmental issue of concern (ISO 2006 a,b; Udo de Haes et al. 2002; Pennington et al. 2004 a,b).

Therefore, in LCA a Cf linearly expresses the contribution to an impact category of a quantity of a chemical (eg. 1 kg) released into the environment (a function of when and where an emission occurs).

Characterization factors for toxicological impacts are necessarily based on models that account for a chemical's fate in the environment, species exposure, and differences in toxicological response, as outlined in Figure 2.7 (Guinée et al. 1996; Goedkoop and Spriensma 1999; Huijbregts et al. 2000a; Udo de Haes et al. 2002).

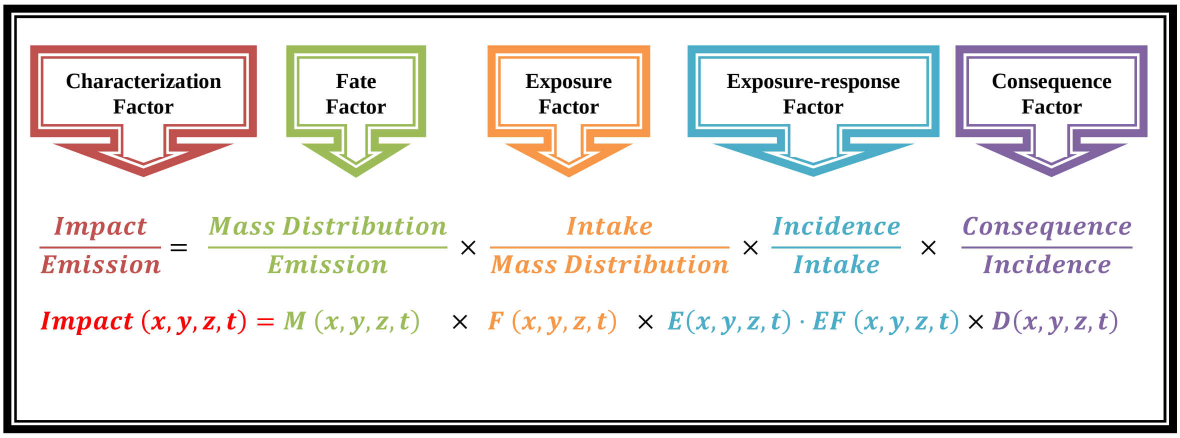


Figure 2.7 Key factors to obtain a characterization factor: fate, exposure, response and consequence (*Source: Own elaboration based on Pennington et al. 2006*).

As shown in figure 2.7 the impact assessment of chemicals in LCA can be divided into four parts. Then a *mass* of chemical released into the environment, M [kg] will distribute in time (t) and in location-space (x, y, z). Ecosystems and human populations will be exposed to a fraction of this mass, *fate* (F), at an *exposure rate* E [hour^{-1}]. Fate modeling estimates the increase in concentration in a given medium due to an emission in the LCI and the exposure model quantifies the amount of a compound transferred into a target organism based on the concentration in the different media. So, this exposure contributes to the risk of undesirable toxicological effects and the risk and potential consequences of *effects* are quantified by two terms: the exposure-response (EF, relates the amount absorbed to an effect on the organism) and the potential consequences or damage (D) related to the effect on the organisms in a change integrated over time and space for a group of organisms (e.g. human population or ecosystems).

These four parts are in common between Human Health and Ecosystems. The relationship between Human Damage Factors (HDFs) and Ecosystem Damage Factors (EDFs) are characterization factors for toxicological effects on human health and ecosystems, respectively (Figure 2.8).

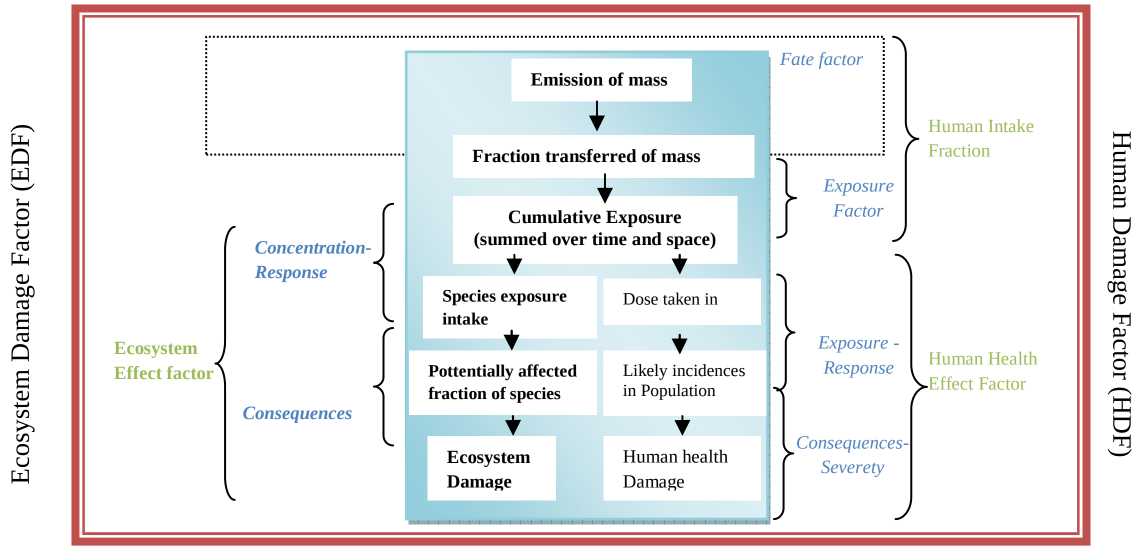


Figure 2.8. Framework and underlying methodological steps for calculating risk based characterization factors for toxicological effects in LCA. (*Source: Pennington et al. 2006, Jolliet et al. 2003*)

During this dissertation we are focus on EDFs, mainly in chapter 7.

2.3.1.3. Ecotoxicological impacts

EDFs were primarily estimated in current LCA practice for aquatic freshwater species, considering exposure to contaminants in surface waters at the pan-regional scale (Pennington et al. 2006). This focus reflects the availability of toxicological data associated with typical regulatory assessments (Udo de Haes et al. 2002). To estimate toxicological risk in any assessment, it is necessary to first consider how a chemical will distribute in the environment in time and space. In LCA, the mass of a chemical released into the environment is given in a life cycle inventory (Margni et al. 2004).

Fate factor

The fate of a chemical in the environment, hence ecosystem exposure (the fate factor, F), reflects the fraction of an emission that is transferred to water and the duration of the exposure (*quantity x duration*) (Udo de Haes et al. 2002; Pennington et al. 2004b). Steady-state fate models (Cowan et al. 1994; European

Commission 1996), similar to those commonly used to estimate concentrations in some regulatory assessments, provide a convenient way to estimate fate factors in support of LCA (Heijungs 1995; Jolliet 1995; Mackay and Seth 1999).

Effect factor

Various sets of characterization factors (Cf) have been proposed for the aggregation of pollutants causing ecotoxicological impacts. Dealing with ecotoxicological impacts is a common part of the environmental life cycle assessment (LCA) of products. Characterization factors based on the predicted environmental concentration/predicted no-effect concentration (PEC/PNEC) ratio have been widely used for ecotoxicological impact assessment in LCA (Guinée and Heijungs, 1993; Guinée et al., 1996; Huijbregts et al., 2000a). However, an index of toxic pressure based on the potentially affected fraction of species (PAF), as derived from species sensitivity distributions (SSDs), may allow a more proper aggregation of toxic impacts due to exposure to multiple substances. An advantage of the PAF concept is that it may be extended to the potentially disappeared fraction of species (PDF) to allow comparison of ecosystem damage of environmental stressors in general (Klepper et al. 1999). Then, Cf that allows a more proper aggregation of pollutants causing ecotoxicity would increase the validity of LCA.

The SSD concept was proposed two decades ago as an ecotoxicological tool that is useful for the derivation of environmental quality criteria and ecological risk assessment. Information on SSD (Figure 2.8) can be very useful in the calculation of ecotoxicological effect factors (impact level) and may serve as input for the calculation of effect factors toward ecosystem quality (damage level).

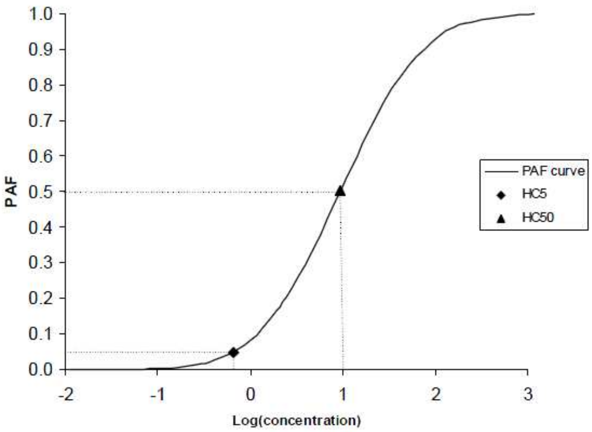


Figure 2.9. Potentially Affected Fraction of species (PAF) curve vs concentration.

Ecotoxicological effect factor (exposure-response) in LCA is commonly based on theory of SSDs and PAF of species (Posthuma et al. 2002; Udo de Haes et al. 2002; Pennington et al. 2004b). A straightforward approach used in LCA is expressed in equation 2.1. The resultant effect factor, EF, is interpreted in terms of the likely fraction of species experiencing an increase in exposure above a defined effect level.

$$EF = \frac{\Delta PAF_{ms}}{\Delta M} \cdot V = \frac{0.5PAF}{\mathrm{HC50_{EC50}}} \; [\mathrm{PAF_{ms}} \cdot \mathrm{m}^3 \cdot \mathrm{kg}^{-1}] \qquad \text{Equation (2.1)}$$

Where EF is the change in PAF of species that experience an increase in stress for a change in contaminant exposure above a predefined effect level [$\mathrm{PAFm^3kg^{-1}}$]; $\mathrm{PAF_{ms}}$ of species when exposed in the presence of multiple substances [dimensionless]; M is mass of contaminant [kg] in an environmental compartment; V is the volume of the environmental compartment [$\mathrm{m^3}$] and HC50 is the median hazardous concentration[1] of a chemical [$\mathrm{kg \cdot m^{-3}}$] affecting 50% of the species at which 50% of the species (in an aquatic ecosystem) are

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[1] Using the benchmark HC5 (concentration that is likely to affect 5% of species) the effect factor in LCA could be 0.05/HC5 instead of 0.5/HC50. However, reasons for adopting the HC50 included that the uncertainty of the estimate is lower than on the HC5 estimate particularly for small data sets of test results, and the HC50 is usually required anyway to estimate the HC5 (Payet 2004; Pennington et al. 2004).

exposed to a concentration above their EC50 (e.g. the concentration at which 50% of a population dies in a laboratory test). It is calculated by equation 2.2:

$$\log\mathrm{HC50} = \frac{1}{n_s}\sum \log\mathrm{EC50}_s \qquad \text{Equation (2.2)}$$

Where *n* is the number of species (or taxa or throphic levels) for which EC50 values are available.

USETox (Rosenbaum et al. 2008) proposes that at least three different EC50 values (species) from at least three different trophic levels are needed to construct the SSD curve. In addition, the estimated HC50 value is designated "interim" and the chronic (long-term) EC50s are preferred with population relevant endpoints, (e.g. reproduction, growth, mortality). However, if the numbers of EC50chronic values are insufficient (< 3) but sufficient EC50acute values (≥ 3) are available an HC50acute is calculated. On this basis an HC50chronic is calculated by use of an assessment factor of 2: HC50chronic = HC50acute/2.

2.3.2. Integration of Geographic Information Systems into LCA

A Geographic Information System (GIS) refers both to (1) the specific software and to (2) the data sets to be used with the software (Skidmore, 2002; Nuñez 2011).

1) GIS software contains subsystems for:
    a) Data input
    b) Data storage, retrieval, and representation
    c) Data management, transformation, and analysis
    d) Data reporting and product generation.
2) GIS supports spatial data collection, analysis, and decision making.

Due to its analytical capacity and the potential of managing a great amount of data, GIS have been increasingly used in environmental decision making which can be applied together with other environmental assessment tools such as LCA. They are different analytical tools but they can become a complementary and environmental decision support tool based on the holistic view of LCA with the visualization power of GIS.

Although the lack of spatial differentiation of LCA has been frequently criticized it was not until recently that LCA began to take advantage of GIS properties (Boulay et al. 2010; Gasol et al. 2011; Geyer et al. 2010; Martinez-Blanco 2012; Núñez 2011; Saad et al. 2011). Recent studies have concluded that the key to realizing the potential of regionalized LCA is to choose appropriate spatial scales for impact assessment methods (Manneh et al 2010; McKone et al. 2011; Mutel et al. 2012).

This dissertation use the coupling LCA and GIS in order to provide spatially-explicit LCA methods for water use environmental impacts. It relies on spatial and temporal conditions where the evaluated activity takes place, and current (site-generic) LCA methods do not consider this geographic information. Coupling LCA with GIS tool provides the distribution of impacts on environment (spatial and environmental keys).

Chapter 3 of this dissertation implemented two combinations LCA-GIS according to the LCA framework:

- LCI: Spatial-dependent inventory in order to plot blue water footprint applied in Spanish watersheds.
- LCIA: Spatially-dependent characterization factors to be applied in the regionalized impact models in Spanish watersheds, providing environmental damages for each Area of Protection.

The distribution of impacts on environment is classified in accordance with geographical dimension of the indicators (Figure 2.10).

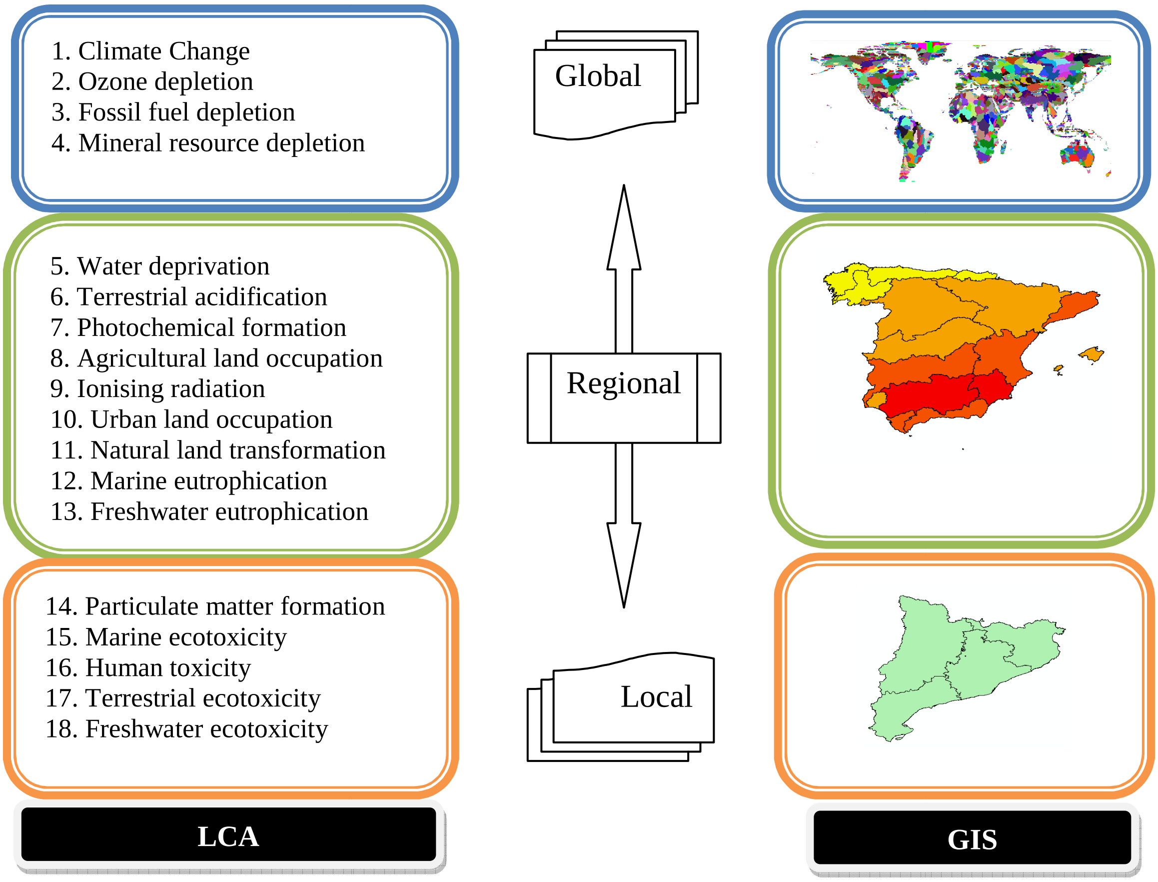

Figure 2.10. Geographical dimension of category impacts.

Potting and Hauschild (2006) distinguished among site-generic (globally valid), site-dependent (operates on the regional scale), and site-specific assessments (locally applicable). In the context of LCA *regionalization* is the recognition that industrial production characteristics and the environmental impact of environmental flows vary throughout space (Mutel et al. 2012). Many of regionalized impact assessment methods use a gridded spatial scale or country-based spatial units (Huijbregts et al. 2000b; Krewitt et al. 2001; Finnveden 2005; Hauschild and Potting 2005; Van Zelm and Huijbregts 2007; Humbert et al. 2009; Pfister et al. 2009; Manneh et al. 2010; Wegener and Heijungs 2008; Saad et al. 2011; Nuñez et al. 2010). Other authors have also used spatial units based on watersheds, ecoregions, or population density (Pfister et al. 2009; Manneh et al. 2010; Nansai et al. 2005). Despite the increased complexity of regionalized LCA, the benefits of reduced impact assessment uncertainty, better supply chain modeling, and geographic interpretation of results are widely recognized.

2.3.3. LCA uncertainties by Monte Carlo Simulation in environmental models

Uncertainty is a true jigsaw puzzle in Life Cycle Assessment (LCA) that has recently become a key challenge for integrated assessments (Van der Sluijs et al. 2005). According to Funtowicz and Ravetz (1990), uncertainty concerns the absence of knowledge and the robustness of the data/facts on which knowledge is constructed and formulated. Other authors have defined uncertainty as a deviation from the unachievable ideal of a deterministic knowledge in a system (Walker et al. 2003), or as incomplete information about any subject (Refsgaard 2007 and Ascough et al. 2008). Morgan and Henrion (1990) and Hofstetter (1998) proposed frameworks for describing different kinds of uncertainty that distinguish between uncertainty and variability. The main difference between both is that *uncertainty* can be reduced by additional research, while *variability* cannot. The latter is caused by differences between places (spatial), in time (temporal) and between individuals or processes (statistical).

Uncertainty analysis is gaining wider acceptance in LCA, with several tools being proposed to deal with it in the recent years (Finkel 1990; Huijbregts 1998; Hertwich et al. 2000; Warmink et al. 2010). Transparency of an LCA can be enhanced by giving more attention and detail procedures-methodologies to improve and quantify uncertainties (Huijbregts et al. 2001; Björklund 2002; Lewandowska et al. 2004; Geisler et al. 2005; Benetto et al. 2006; Lloyd and Ries 2007; Reap et al. 2008; Hong J et al. 2010). Huijbregts et al. (2003), Lewandowska et al. (2004), Geisler et al. (2005) and Hung and Ma (2009) studied imprecise measurements in LCA, while Hertwich et al. (2000) proposed several assumptions to simplify the LCA calculations when knowledge is uncertain or there is a lack of knowledge.

However, the application of an uncertainty analysis in the LCA studies is not common practice so a proper analysis will be facilitated when it is clear which types of uncertainties exist and which tools are available to deal with them. Both the credibility and transparency of an LCA can be enhanced by giving more attention to quantifying uncertainties (Geisler et al. (2005), Reap et al. (2008), Hung and Ma (2009)).

In this dissertation, we focus on statistical uncertainties, that is, on uncertainties that can be described using probabilistic functions (Warmink et al. 2010). Statistical uncertainties can be handled in practice by means of two types of methods: *quantitative*, like Monte Carlo Simulation (MCS) (Huijbregts et al. 2003) and semi-*qualitative* methods, like the Pedigree matrix.

Monte Carlo analysis is a statistic sampling technique that uses computer simulation to generate a great amount of input scenarios (samples), taking into account the information of the variability and uncertainty related to all input parameters (Passuello 2010). To perform a MCS' parameters, it has to be specified as uncertainty distributions. The MCS varies all the parameters at random, but the variation is restricted by the given uncertainty distribution for each parameter. Various parameter distributions, such as uniform, triangular, normal, or lognormal distributions, can be used in the model. The randomly selected values from all the parameter uncertainty distributions are inserted in the output-equation. Repeated calculations produce a distribution of the predicted output values, reflecting the combined parameter uncertainties (Huijbregts, 1998). Huijbregts et al. (2003) estimated default statistical uncertainties for a number of impact categories. Default uncertainty factors for physical flows were also reported by Frischknecht et al. (2007) in the Ecoinvent Database (Ecoinvent, 2010).

Much research effort has been devoted towards the identification and modelling of the main uncertainties sources in LCA analysis. In contrast, to the best of my knowledge, no work has addressed the underlying decision-making problem under problabilistic uncertainty. That is, given the results of an LCA study under uncertainty, this dissertation herein interested in developing tools to facilitate decision-making in the interpretation phase. Particularly, a comprehensive approach for decision making under uncertainty in LCA that is based on the analysis of probabilistic environmental data using risk-based assessment metrics is presented in Chapter 9.

Since it is not always possible to express statistical uncertainty in terms of measurable values, strategies to address uncertainties in a qualitative way have been developed. The pedigree matrix is the most widely used qualitative tool to handle uncertainties. It was originally developed by Weidema and Wesnaes (1996), and is based on five quality indicators that characterize the environmental data available by looking at their *reliability (sources), completeness* (statistical properties of the data), *temporal correlation, geographical correlation* and further *technological correlation* for the inventory of environmental interventions (emissions and extractions). This dissertation applies Pedigree Matrix in Chapters 3 and 9.

In addition, uncertainty ranges of Characterization Factor (Cf), deriving from the input data of impact assessment models, are not always quantified by the method developer (e.g. Heijungs et al., 1992, Goedkoop and Spriensma, 2000). Only few studies provide rough guidelines and rules of thumb to quantify uncertainty ranges of Cfs (Huijbregts et al., 2003, Geisler et al., 2005, Hung and Ma, 2009; Van Zelm et al. 2011; Mutel et al. 2012). In this dissertation, Chapter 7 presented a development of Cf for the salinity impact in a Spanish coastal wetland where the uncertainty assessment was carried out by propagating with Monte Carlo Simulation to quantify the 95% confidence interval.

Part II

Problem Statement: Water Management and Climate Change in Agriculture

Chapter 3

Comparing different methods to perform an environmental assessment of freshwater use in agriculture from Ebro Basin

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Abstract

Purpose: This work describes an experimental approach to farm performance within the framework of Life Cycle Assessment (LCA). Our primary goal is to test different inventory analysis and impact assessment methods applied to water use developed over the past decade using characterization models for LCA. This study aims to provide a beneficial response to users (farmers, administration, certifiers, and so on) on the consciousness of water consumption.

Methods: We first collected data for experimental case studies from three representative crops grown in the Ebro watershed: grape, nectarine and corn. We assessed the different methods (Hoekstra et al. 2009; Milà i Canals et al. 2009; Muñoz et al. 2010; Pfister et al. 2009) at midpoint and endpoint levels, taking into account the spatial characterization factors of the different models (country, river and watershed). Finally, we compared our results with those obtained by other authors through the application of these methods to crops studied in different Spanish watershed locations. We propose using different methods to determine the sustainability of farms at a country-wide level. Our approach provides a set of regional characterization factors that are obtained from information generated using geographic information systems.

Results: Our assessment of the different models suggests that they can be useful as initial screening methods. The comparison of absolute values of the measured parameters in different crops is difficult, not only because the diversity of the products themselves but also due to the variability and uncertainty associated with local conditions. At the endpoint level for the regionalized area, the production of one kg of crop causes a significant damage to the natural environment (ecosystem quality). In contrast, no damage to human health or natural resources was revealed through our case study that considers Spanish rural areas. Our results indicate that water use in the production of all three crops is adequate.

Conclusions: There is an inherent trade-off between the complexity of the analysis and the practicality and simplicity of the results (the parsimony principle: as simple as possible and as complex as necessary). Different methods cover different aspects related to water consumption. Hence, it is difficult to reach a consensus on their applicability and to generalize specific impacts through the use of tools like the water footprint database or global life cycle assessment methods. We found considerable differences between experimental and statistical data, emphasizing the high degree of variability in results depending on the information source, crop variety, and cultivation technique; therefore caution must be exercised when basing decisions on data from different types and sources.

Keywords: life cycle assessment, water footprint, water stress index, freshwater, agriculture, spatial regionalization.

3.1 Introduction

Freshwater is one of the planet's most valuable resource as it is an irreplaceable and essential component to the existence and maintenance of human life. Drinking water is used for hygiene purposes and lies at the foundation of our food supply. Because of this, it is vital to our survival, and at the same time ensures biodiversity and the pivotal ecosystem functions on which we all ultimately depend (Koehler 2008). Despite the critical importance of water resources for the continuation of life on earth, no consensus has been reached so far with regard to a standard indicator for measuring the impact of water use.

In many parts of the world today, competition for scarce water resources has become intense, as the water contained in many river basins is not sufficient to meet local demand. A lack of water is a major constraint to produce enough food. Agriculture is a central component to the water issue because the production of food and other agricultural products accounts for 70% of the freshwater withdrawals from rivers and groundwater sources (International Water Management Institute 2007).

Today's agriculture and the food systems that it underpins are at crossroads where food is a major sustainability challenge. Numerous recent reports have emphasized the need for drastic changes in the agri-food system in order to meet the double challenge of feeding a growing population with a rising demand for high-quality diets while minimizing the environmental impacts of food production (Foley et al. 2011). Nowadays, the Food Ethics Councils of different countries have looked at the value on food products for promoting sustainable water use. Bearing this in mind, they have assessed the effectiveness of the process of communicating to consumers the water footprints (WF) of products, that is, the impact that food production has on the world's water resources (Segal and MacMillan 2009).

Corporate water accounting would allow consumers, civil society groups and the investment community to compare different companies' social and environmental impacts. This information could be ultimately used to assess their actions and support decision-making in this context. In short, the ability to effectively account for corporate water use and impacts is essential in helping companies to promote improvements and become better aligned with external stakeholders' expectations, thereby directing their efforts towards advance sustainable water management.

Several methods have been used to assess the environmental sustainability on farms. Irrigated agriculture faces the need to improve water management practices from an environmental viewpoint. At a farm level, the farmer is facing strategic and tactical decisions that will have important impacts on the environment. Based on a realistic characterization of the farming system, gathering data and calculating some selected indicators (e.g. Water Use Efficiency) can be used to assess and benchmark the farmer's performance at the end of the season.

International standards for water footprint are being developed, so it is time to consider the comparative advantages of the different approaches that have been proposed to quantify the water footprint of food products. A comprehensive method for converting water use into a water footprint (WF) was developed by Hoekstra et al. (2009), who provide highly detailed spatial data applicable from the regional to the global sphere.

Other approaches, like those developed by Ridoutt (2011), have applied and developed a water footprint metric for agricultural products and the food industry. Antón et al. (2005) explored ways to account for water use in agricultural systems. In their method, evapotranspiration is considered in the inventory analysis phase, while water use and freshwater consumption are accounted for in the impact assessment stage. Heuvelmans et al. (2005) and Guinée et al. (2001) offer background information on the potential effects of mining on water resources, but do not provide a means to use this information in the characterization stage.

Several initiatives have recently been suggested in the LCA impact assessment framework, including the works by other authors such as Milà i Canals et al. (2009) and Bayart et al. (2010), among others, and several models are being developed to evaluate water footprint through life cycle assessment methods (WF-LCA) (Nuñez et al. 2012; Ridoutt and Pfister 2010). For the most part, the various methods for studying impact levels, water use and quality in life cycle inventories differ in the use of either midpoint or endpoint assessments.

From a global perspective, there is not a water scarcity problem. In contrast, water consumption is one of the main environmental problems linked to spatial definition due to the local availability of water resources. Regionalization, in the context of LCA, is the recognition that production characteristics and the environmental impact of environmental flows vary throughout space. Hence, it becomes essential to provide specific guidance for future improvements of inventory data sets and impact assessment methods (Mutel et al. 2012). Regionalized impact assessment methods show significant differences in terms of impacts and spatial scales: site-generic (globally), site-dependent (regional), and site-specific (locally) assessments (Potting and Hauschield 2006). Choosing an appropriate spatial scale for impact assessment methods is a key issue in regionalized LCA (Manneh et al. 2010; Mutel and Hellweg 2009). In the context of freshwater analysis, spatial regionalization differs from the previous presented WF and WF-LCA methods. Characterization factors published in the literature cite differentiated values by *country* (Bayart 2008; Boulay et al. 2011a, 2011b; Frischknecht et al. 2006; Milà i Canals et al. 2009; Pfister et al. 2009; Ridoutt and Pfister 2010), by *watershed* (Boulay et al. 2011b; Frischknecht et al. 2006; Pfister et al. 2009; Water Footprint Network 2011), or *grid cell* (Pfister et al. 2009; Ridoutt and Pfister 2010), by *river* (Hannafiah et al. 2011; Milà i Canals et al. 2009; Smakhtin et al. 2004) and more recent works have also differentiated values at the *wetland* level, such as in Verones et al. (2012) and Amores et al. (2012).

Various different trends have emerged in the assessment of freshwater use. Hence, the aim of this work is to compare the different results based on an experimental case study. At the field level, we aim to provide a comprehensive overview of impact assessment methods to address the impact of water use and consumption in ecosystems quality. Hence, in this work, we applied these methods at three different levels (WF or inventory indicator, and midpoint and endpoint LCA) through experimental case studies of three different crops (grape, nectarine and corn) considering the following scenarios.

1) Water footprint (WF): We conducted water footprint following the Water Footprint Assessment Manual developed by Hoekstra et al. (2011).
2) Midpoint level: We carried out the Freshwater Ecosystem Impact (FEI) developed by Milà i Canals et al. (2009) and the Water Stress Index developed by Pfister et al. (2009) and Smakhtin et al. (2004) for different spatial references (country, watershed, river).
3) Endpoint level: We used the endpoint method developed by Pfister et al. (2009) to assess damage impacts associated with freshwater use, such as harm to the natural environment (ecosystem quality), for each of the three crops in the Ebro Basin.

3.2 Methods

3.2.1 Experimental fields

To assess and benchmark farmers' performance with regard to water use, we employ data concerning farming systems (crop, irrigation system, soil type, climate and management variables) and calculate a set of selected indicators. A set of working farms were used as experimental cases to determine water footprints and conduct environmental impact assessments.

The data obtained from the monitoring systems was used to determine realistic index values that quantify the performance of the irrigation systems and that are employed in further LCAs and water footprint analyses.

As a practical assessment trial, a case study was carried out in the Ebro Basin near Lleida (a region in Catalonia, north-eastern Spain) where irrigated farms where studied during the 2010-2011 growing season. Farms either comply with some quality standards (examples Global G.A.P., Tesco Nature's choice, Integrated Agriculture) or show genuine interest in some kind of sustainability assessment. The assessed crops were:

- Grapes for winemaking: chardonnay grapes used in the production of high quality wine.
- Sweet fruit family: early harvest nectarines.
- Spring cereals: corn for silage watered by pivot irrigation.

Lleida is a location of interest due to the current construction of channels and infrastructure to improve irrigation systems in the region. These three crops were chosen because of their economic and harvested importance in the area and accurate (real-time) field data were available from farmers. Daily or weekly real-time field monitoring would be required to guide the tactical decisions and develop heuristic rules that would be used on the farmer's decision dashboard throughout the growing season. Therefore, data mining was undertaken at a farm level and within the watershed (Table 3.1). This was done through several ways: interviews with technicians and regional water management bodies, collecting the results of analyses already conducted at the farms (soil tests, irrigation water quality values), conducting field surveys of the hydraulic distribution network, using the on-farm registration diaries and production records, using the network of automatic weather stations in the area to retrieve evapotranspiration (ETo) and other relevant meteorological data, and finally, through field monitoring of precipitation. Weekly weather data (Table A3.1) in the local area (precipitation and reference ETo) was gathered from nearby automatic monitoring weather stations and characteristic data about crops and crop coefficients (kc) were obtained from local extension agents.

Table 3.1 Crop characteristics, climate and irrigation conditions during 2011 for the three crops assessed: grape, nectarine and corn.

Parameter	Unit	Grape	Nectarine	Corn
Variety	-	Chardonnay	Sweet beginner fruit	Cereal of summer
Location	m	X:289,915.3	X:288,542.5	X:314,747.5
	m	Y:24,617,359.8	Y:4,596,907.5	Y:4,610,982.5
Yield	$kg \cdot ha^{-1} \cdot yr^{-1}$	11,935	47,137	14,285
Climate				
Precipitation	$l \cdot m^{-2} \cdot yr^{-1}$	104.2	312.9	80.7
Evapotranspiration	$l \cdot m^{-2} \cdot yr^{-1}$	837.6	969.8	595.4
Crop coefficient	-	Min:0.1, Max:0.4	Min:0.3, Max:1.0	Min:0.3, Max:1.1
Irrigation				
Period	weeks	26 (28 March to 26 September 2011)	34 (7 March to 31 October 2011)	16 (23 May to 12 September, 2011)
Water applied	$m^3\ ha^{-1}$	2,137.3	5,123.7	4,645.8

3.2.2 Life Cycle Inventory

As a result of environmental concerns, several initiatives have recently been launched in order to develop and standardize analytical tools to measure and manage water resources (Water Footprint Network 2011). Following the work by Milà i Canals et al. (2009), direct water use was calculated using the water footprint approach (Hoekstra et al. 2011). To this end, we considered the main local crop characteristics as well as data on rainfall availability and irrigation water use during crop production. In addition, this method accounts for the volume of freshwater used directly, which is calculated from the evapotranspiration, as a first approach based on the climate contribution of rainfall and irrigation without considering soil parameters (Mekonnen et al. 2010).

From a life cycle perspective, water footprint (WF) is an accounting indicator of freshwater which neglects the availability of the resource. Hence, it provides no information on impact assessment. The concept of WF was introduced by Hoekstra et al. (2003) and subsequently developed by Hoekstra et al. (2009). WF provides a framework with which to analyze the link between human consumption and the appropriation of the globe's freshwater. According to the Water Footprint Network (WFN) method, the WF of a product is defined as the volume of freshwater used to produce the product over the full supply-chain (Hoekstra et al. 2011), which looks not only at the direct water use of a consumer or producer, but also at its indirect water use. WF identifies the importance of specifying water appropriation geographically and temporally. It also distinguishes the volumes of water consumed by different 'water-colors' (green, blue and grey) depending on the type of water sourced and polluted (Hoekstra et al. 2011). The green virtual-water content of a crop, considering agricultural raw material, refers to the total rainwater evapotranspiration from the field during the growing period; the blue virtual-water refers to the total volume of surface water or groundwater evapotranspiration during the growing period; and the grey virtual-water content of a crop is the theoretical volume required to dilute polluted water generated during growth to an 'unpolluted' condition. Other initiatives include the ISO 14046 standard proposed by the International Organization for Standardization (ISO) Water Footprint Working Group (ISO/TC207/SC5/WG8) and others, who have launched a global water tool to assess water consumption and use that is based on the life cycle assessment methodology developed by the UNEP/SETAC Life Cycle Initiative Working Group WULCA (Koehler and Aoustin 2008).

At the field level, crop water consumption is defined in terms of evapotranspiration, because it is difficult to separate the transpiration of the crop by evaporation from the soil surface between the plants. The total water footprint of the process of growing crops (WF_{tot}) is the sum of the green (WF_g), blue (WF_b) and grey (WF_{gy}) components (3.1):

$$WF_{tot} = WF_g + WF_b + WF_{gy} \; [\mathrm{m}^3 \cdot \mathrm{tonne}^{-1}] \qquad \text{Equation (3.1)}$$

We will express the water footprint cases in m^3tonne^{-1}, i.e. in water volume per mass. The blue and green component in the process of water footprint a growing crop (WF_b, $m^3 \cdot tonne^{-1}$ and WF_g, $m^3 \cdot tonne^{-1}$) are calculated in accordance with that developed by Hoekstra et al. (2011). The green and blue components in crop water use (CWU, m^3ha^{-1}) are calculated from the accumulation of daily evapotranspiration (ET, mm) over the complete growing period. During the irrigation period, the grey water corresponds to the leaching fraction to avoid salinity excess, which depends on fertilizers management. Baring this in mind, we focused on the blue and green water footprint (Table 3.2), leaving grey water out of the scope of this work.

Table 3.2 Water consumption calculated as a blue and green water footprint and yields for crops provided for two campaigns (2010 and 2011) and average reference values.

Water consumption		Grape			Nectarine			Corn		
	units	2010	2011	reference	2010	2011	reference	2010	2011	reference
Blue water footprint	m^3tonne^{-1}	227.0	179.1	66.6[1]	97.6	108.7	122.0[1]	306.9	325.2	307.4[1]
Green water footprint	m^3tonne^{-1}	32.3	28.7	203.9[1]	28.9	19.2	327.9[1]	31.7	43.5	403.1[1]
Yields	$tonne\ ha^{-1}$	8.19	11.9	5.53[2]	46.3	47.1	22.4[2]	15.0	14.3	8.50[2]

[1] Source (Mekonnen et al. 2010) Water footprint per tonne of crop or derived crop product (Code Faostat: 507 Grapefruit, 534 peaches and nectarines, 56 maize, corn) during (1996-2005) adapted for reference area (m^3 $tonne^{-1}$).

[2] Data available in Spanish Ministry of Agriculture, Food and Environment (MAGRAMA): Yield average for grape, nectarine and corn in area of study during 2005.

3.2.3 Midpoint and endpoint impact categories

Water indices can be used as characterization factors (Cf) for midpoint (Falkenmark et al. 1989; Raskin et al. 1997; Smakhtin et al. 2004) and endpoint (Döll 2009; Sullivan et al. 2003) impact assessment methods when applied to freshwater consumptive use.

Different authors have employed a water index called withdrawal-to-availavility (WTA) (Alcamo et al. 2003; Frischknecht et al. 2006; Raskin et al. 1997; Seckler et al. 1998; Smakhtin et al. 2004) or consumption-to-availability ratio (Boulay et al. 2011b; Hoekstra et al. 2011), which represents the ratio between withdrawals for different users ($WU_{i,j}$) and annual freshwater availability (WA_i) (3.2).

$$WTA_i = \frac{\sum_j WU_{i,j}}{WA_i} \qquad \text{Equation (3.2)}$$

The Water Stress Index (WSI) has been used for the assessment of the environmental impact due to water consumption at a midpoint level. WSI indicates the portion of consumptive water use that deprives other users of freshwater. In the context of LCA, some authors like Milà i Canals et al. (2009) used directly WTA as WSI, whereas others like Pfister et al. (2009) proposed performing a screening assessment using the WSI and adapting the freshwater WTA to account for precipitation variability. This latter approach may lead to increased water stress during specific periods, and thus differentiate watersheds with strongly regulated flows. So, WSI serves as a Cf for the midpoint category “water deprivation” considered in LCIA, and ranges from 0.01 to 1 (Pfister et al. 2009), with 1 meaning a serious water stress in a basin, e.g. in the south watersheds of Spain.

Milà i Canals et al. (2009) proposed ways of quantifying the impacts of water use. They identified two primary pathways through which freshwater use can impact available supply like freshwater ecosystem impact (FEI) and freshwater depletion (FD). As we have focused on ecosystem quality assessment, we have used FEI indicator in this work.

The FEI midpoint impact category aims to assess the ecological consequences of water use in a certain region measured as the volume of "ecosystem-equivalent" water, which refers to the volume of water that is likely to affect freshwater ecosystems. It is calculated using the equation (3.3):

$$FEI = \sum_{i=1}^{n} CWU_i \; x \; WSI_i \qquad \text{Equation (3.3)}$$

Where CWU_i is consumptive water use of a unit process (m^3) and the characterization factor WSI. The latter is defined considering WU as the total annual freshwater withdrawal in a river basin and WA as the annual freshwater availability in that same basin, while *i* to *n* are the set of unit processes involved in the product system. Milà i Canals et al. (2009) take into account the evaporative use of blue water (surface water and aquifers) as well as water use due to land use changes using countries as a spatial reference.

In order to obtain the value of Cf, we use the value of the WSI developed by Smakhtin et al. (2004), which is determined using equation (3.4). This indicator reflects the scarcity of water for human use by taking into account environmental water requirements (EWR). The method estimates the EWR for the most important world river basins and combines it with the water resources available and their use by subtracting EWR from the available resources, thereby obtaining the value of WSI. The WSI of region *i* is the ratio between the total water use (WU) and the difference between the renewable water reserves (WR) and the environmental water requirement (EWR) in that region. Depending on local water scarcity and the respective ecosystem demand, site specific characterization factors are obtained by assessing the severity of additional human water use.

$$WSI_i = \frac{WU_i}{WR_i - EWR_i} \qquad \text{Equation (3.4)}$$

In this work, we have used the WSI proposed for the Ebro River, including the environmental requirements defined by Smakhtin et al. (2004), and Muñoz et al. (2010) who adapted the methodology for the Ebro Basin according to the work by Milà i Canals et al. (2009) and the hydrological policies.

From a midpoint perspective, Pfister et al. (2009) propose performing a screening assessment using the WSI, and adapting at the same time the freshwater WTA to account for precipitation variability. This approach may lead to increased water stress during specific periods and consequently to different watersheds with strongly regulated flows. To assess the damage to the ecosystem quality at the endpoint level, we have followed the methodology proposed by Pfister et al. (2009).

Table 3.3 shows the characterization factors of WSI for the different geographical units according to different methodologies: at the country level (Spain) described by Milà i Canals et al. (2009), at the river level (Ebro River) as described by the same authors based on Smakhtin et al. (2004), and at the watershed level (Ebro Basin) based on the WSI of Muñoz et al. (2010) and Pfister et al. (2009). The application of these different methods provided impact values at the midpoint level (Table 3.3) for the different crops at different spatial scales (country, river, and watershed).

Table 3.3 Impact score using different Water Stress Index as characterization factors and results for case study crops: grape, nectarine and corn in the Ebro Basin during campaign 2011.

			Grape	Nectarine	Corn	Units	
Production			**11.9**	**47.1**	**14.3**	**tonne ha^{-1}**	
Water applied	**Reference area**	**WSI**	**2,137.3**	**5,123.7**	**4,645.8**	**m^3 ha^{-1}**	**References**
Spain	Country	0.320	57.3	34.8	104.1	m^3 tonne^{-1}	Milà i Canals et al. 2009
Ebro River	River	0.888	157.6	95.7	286.2	m^3 tonne^{-1}	Milà i Canals et al. 2009
Ebro Basin	Watershed	0.269	46.4	28.2	84.2	m^3 tonne^{-1}	Pfister et al. 2009
		0.460	82.4	50.0	149.6	m^3 tonne^{-1}	Muñoz et al. 2010

The variability of the characterization factor (WSI) is based on different scales and different approaches. Normally, the WSI has a spatial resolution of 0.5 degrees, which is more relevant for describing water stress at a local watershed level than indicators which are based on national or per capita statistics (Rijsberman 2006). For our reference area, the highest WSI occurs using Smakhtin et al. (2004), who took into account ecosystem water requirements in their approach.

There are two different WSIs concerning the watershed reference area (Ebro Basin). The first was published by Pfister et al. (2009) and is equal to 0.269. The second was defined by Muñoz et al. (2010), and equals 0.460. The WSI developed by Pfister et al. (2009) is lower because they introduce a correction factor to calculate a modified WTA/WSI which takes into account the seasonal availability of resources and differentiates watersheds with regulated flows (the Ebro River is assumed to be a regulated river).

3.2.4. Assessments of different Spanish watersheds

In order to provide reference values and illustrate environmental sustainability in our case study, we used the different Spanish watersheds as a framework to calculate environmental impacts associated with consumptive freshwater use for three crops. Average yields and ratio percentage harvested of grape, nectarine and corn for each watershed were calculated by means of production per surface area according to the Spanish Ministry of Agriculture, food and environment (MAGRAMA) for each province in Spain and the relative contribution of surface area in each watershed (SIA). In this work, we propose to calculate some indicators like watershed areas (km^2), water availability (km^3 $year^{-1}$), average yields (kg ha^{-1}) and ratio (%) of land use for each crop (grape, nectarine and corn) (Table 3.4). In order to calculate them, we collected data from various sources – the provincial statistics from MAGRAMA, the Geographic Information System (GIS) database of farming land (SIGPAC), the Integrated Water Information database (SIA) and the WaterGap database (Alcamo et al. 2003).

Firstly, we took the blue water footprint (m^3) for each crop in the Spanish autonomous communities provided by Mekonnen et al. (2010), and then calculated the associated water footprint for each Spanish watershed. Given that this database does not provide yields for water footprints, we had to employ the database 2005 developed by the Spanish Ministry of Agriculture, Food and the Environment's (MAGRAMA).

3.3 Results and discussion

3.3.1 Part I: Application of methods in the reference case study

3.3.1.1 Water footprint assessment

Table 3.2 shows the blue and green water footprint for grape, nectarine and corn based on experimental data in Lleida (Ebro Basin), where farms are located. These experimental cases are compared with the database made available by Mekonnen et al. (2010) which contains yields collected by the Spanish Ministry of Agriculture, Food and the Environment's (MAGRAMA).

Blue water footprint (WF) data from representative farms in 2011 show that corn needs 325.2 m^3tonne^{-1} of water, whereas according to the statistical data (Mekonnen et al. 2010), corn needs almost 307.4 m^3tonne^{-1} of water. In addition, according to our case study, 108.7 m^3tonne^{-1} of water are needed for nectarines, whereas the statistical data shows much blue WF (122.0 m^3 $tonne^{-1}$). However, the experimental data for grapes (179.1 m^3tonne^{-1}) is twice as high as the statistical data (66.6 m^3tonne^{-1}). These differences represent variations in water use as calculated with our experimental data compared to database sources of 6% for corn, 12% for nectarines and more than double in the case of grapes. Generally, it is clear that the blue WF will differ drastically depending on whether it is based on experimental data or database statistics. Both, grape and corn use higher volumes of blue WF according to experimental data, while compared to the reference database the opposite is true for nectarines. It is important to consider that this variation maybe due to differences during the growth stage, crop coefficient (kc), precipitation and production for each crop zone and each reference area (Appendix A3-Table A3.1). In order to assess the variability from different campaigns, these results have been compared with the previous campaign 2010 (Table 3.2). The production of grape was fewer in 2010 (8.19 tonne of grape·ha^{-1}) due to extremely bad climatic conditions and it consumed higher values of blue WF (227.0 m^3). Nectarine and corn farms produced practically the same yield that in previous campaign 2010 (46.3 tonne of nectarine·ha^{-1} and 15.0 tonne of corn·ha^{-1}) with almost the same blue WF (97.6 m^3·tonne nectarine^{-1} and 306.9 m^3·tonne corn^{-1}).

Green WF data from representative farms in 2011 show also that there is a huge difference between data from experimental and reference sources (Table 3.2). Whereas grape, nectarine and corn have contributions from green WF of 28.7 m^3tonne^{-1}, 19.2 m^3tonne^{-1} and 43.5 m^3tonne^{-1}, respectively for experimental data, they received 203.9 m^3tonne^{-1}, 327.9 m^3tonne^{-1} and 403.1 m^3tonne^{-1} in accordance with reference data. Comparing this campaign with the previous one (2010), we can observe that grape and nectarine farms received much green WF (32.3 m^3tonne^{-1} and 28.9 m^3tonne^{-1}, respectively) in 2010 whereas corn receives fewer (31.7 m^3tonne^{-1}). Following the same trend than blue WF assessment, these experimental data are extremely different from reference data (203.9 m^3tonne^{-1} for grape, 327.9 m^3tonne^{-1} for nectarine and 403.1 m^3tonne^{-1} for corn).

Part of the difference between the results of our experimental assessment and the database information can be attributed to the differences between yields or average climate conditions, crop evapotranspiration (ET_C) and precipitation (P) during the crop's period. However, a simple comparison with average data shows that in this case study, farm nectarines performed very well, while grapes demonstrated a high water-consumption-related impact and corn a medium impact considering their different yields. Comparisons between crops are also difficult because they have different crop cycles so those that grow in summer need more water than those that grow during the rainy season, which take advantage of green WF. Figure 3.1 shows precipitation, irrigation and the evapotranspiration of the three crops for the assessed campaign (2011) and the previous one (2010). The irrigation campaign for grapes goes in 2011 from the 10th week until the 40th, so because the precipitation for that year at that farm was higher during the first weeks of the season and it began decreasing from the 23rd week, crop evapotranspiration and irrigation increased progressively until the 36th week. If we compare these results with the results obtained in 2010, we observe the same trend than in 2011 given that the crop evapotranspiration and irrigation were increasing from 26th week whereas considerable precipitation took place until 23rd week. We can therefore conclude that growth of grapes take mainly place during summer so they have more difficultly to take advantage of the green WF. In the case of the nectarine, irrigation period in 2010 and 2011 started in the 10th week and continued to the 44th week. We can observe that in both campaigns, 2010 and 2011, crop evapotranspiration during the crop growth was supplied by constant precipitation and irrigation, making this crop suitable for that zone based on climatic parameters, and therefore the most suitable crop in the area due to rainfall conditions. Corn irrigation started in 2010 and 2011 approximately in the 21st week and continued to the 37th, when precipitation was very low for this area and high levels of irrigation in keeping with its evapotranspiration were required by the crop. Hence, we can foresee a priori in accordance with climatic parameters that in the area of study the best and worst suitable crops are nectarine and corn, respectively.

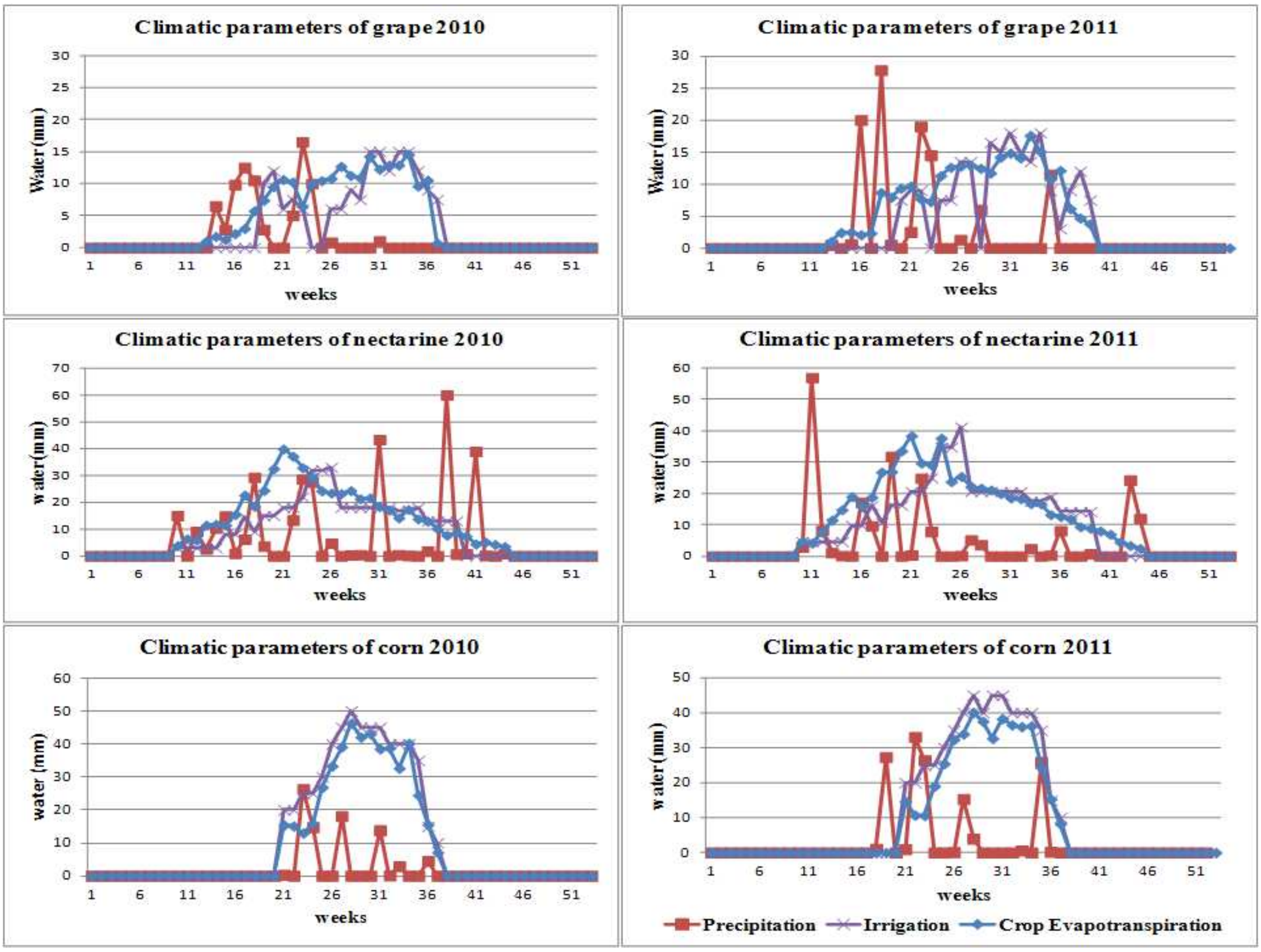

Figure 3.1 Precipitation, evapotranspiration and irrigation (l·m^{-2}) data for grape, nectarine and corn farms. (Climatic parameters are different for each crop because they are located in different fields in the Ebro Basin).

All of these differences illustrate the enormous divergence between statistical data and experimental data which could seriously affect the final impact results. As most of the literature is based on statistical data, this is a very important point to consider. Moreover, none of these results have provided information about the risk to ecosystems, but rather have simply provided and estimate of the quantity of water consumed. So, these values may be useful as a method of screening but are irrelevant as a means of assessing the environmental sustainability of the production of a crop.

3.3.1.2 Midpoint assessment

We assessed these experimental farms located in the Ebro Valley at the midpoint level using the Water Stress Index. We consider the WSI developed by Milà i Canals et al. (2009) for the Ebro River at 0.888, which had been previously proposed by Smakhtin et al. (2004). This high WSI is due to the fact that this basin is heavily exploited, and this method implicitly assumes that water has been reserved for ecological purposes and estimates a ratio (or a percentage) of total withdrawals to usable water. Therefore, a comparison of the damage to ecosystems according to the midpoint WSI proposed by Smakhtin et al. (2004) and the endpoint WSI proposed by Pfister et al. (2009) is an interesting topic for discussion.

Using the same considerations applied to WSI characterization factors, we compared WSI at the country level as developed by Milà i Canals et al. (2009) (WSI=0.32) and WSI at the watershed level as developed by Muñoz et al. (2010) (WSI=0.46). This suggests that, on average, the consumption of 1 l of water is more damaging at the level of the Ebro Basin than at the country level.

Concerning the two different WSIs according the watershed reference area (Ebro Basin), the first was published by Pfister et al. (2009) as 0.269 and the second by Muñoz et al. (2010) as 0.460. We consider the WSI from Pfister et al. (2009) the most suitable figure for our case study since they include the seasonality and the regulated flows. Nevertheless, because the Ebro watershed comprises a very large area with different climatic conditions, WSI at sub-basin level will be needed to obtain major detail in the area of study.

Based on these data, our case study shows that the freshwater used in nectarine production has the lowest values in all geographical scale levels (*country* 34.8 m^3tonne^{-1}, *river* 95.7 m^3tonne^{-1}, *watershed* 28.2/50.0 m^3tonne^{-1}) compared to the other two crops, grape (*country* 57.3 m^3tonne^{-1}, *river* 157.6 m^3tonne^{-1}, *watershed* 46.4/82.4 m^3tonne^{-1}) and corn (*country* 104.1 m^3tonne^{-1}, *river* 286.2 m^3tonne^{-1}, *watershed* 84.2/149.6 $m^3 tonne^{-1}$). Moreover, as mentioned in section 3.1.1, corn consumes the highest amounts of freshwater.

3.3.1.3 Endpoint assessment

Using the endpoint methodology developed by Pfister et al. (2009), damage impacts associated with freshwater use such as ecosystem quality (EQ) were assessed for each crop in the Ebro Basin. Pfister et al. (2009) take into account the effects of freshwater consumption on biodiversity in the Ebro Basin, adding a characterization factor of $2.70x10^{-01}$ for damage to ecosystem quality, which uses the vulnerability of vascular plant species biodiversity as representative of the net primary production (NPP) water-shortage vulnerability of an ecosystem. According to this system, the following values were obtained for each crop: $4.6x10^{-02}$ $m^2 \cdot yr \cdot kg^{-1}$ grape, $3.11x10^{-02}$ $m^2 \cdot yr \cdot kg^{-1}$ nectarine and $1.28x10^{-01}$ $m^2 \cdot yr \cdot kg^{-1}$ corn. Hence, as seen in previous sections, corn has the biggest impact calculated at midpoint level, but it is also more damaging at the endpoint level to ecosystem quality than the other two crops.

To date, endpoint level studies have scarcely been expanded upon. Pfister et al. (2009) provided a general approximation at the endpoint level, and Smakhtin et al. (2004) did the same at the midpoint level. However, a detailed approximation of number and variability of native species in different ecosystems at local level are needed, similar to what other authors have accomplished at river level (Hanafiah et al. 2010), at the groundwater level (Van Zelm et al. 2011) and with reference to wetlands (Amores et al. 2013; Verones et al. 2012).

3.3.2. Part II: Spanish watersheds comparison

To test environmental sustainability and differences at the georeference scale level, we compared the three farms in Ebro basin (grape, nectarine and corn) with the same crops grown in different regions of Spain using the same methods applied to assess the different watersheds.

Table 3.4 shows data on Spanish watersheds for the production of grapes, nectarines and corn in terms of surface area (km^2), water availability (km^{3}year^{-1}) and percentage (%) of occupation of the three crops in the watershed and their average yield (kg ha^{-1}). The data show that the most far-reaching watersheds are Ebro (85,939.3 km^2), Duero (78,859.7 km^2) and Tajo (55,764.5 km^2), which are also the most plentiful 16.1 km^3, 24.1 km^3 and 18.4 km^3, respectively. Although grape production is predominant in the Galicia-Costa watershed, with 8,421.4 kg ha^{-1}, this crop accounts for the highest harvested surface compared to the available watershed surface in Segura. Table 3.4 shows that the highest yield of nectarines occurs in the Ebro watershed (17,725.3 kg·ha^{-1}) but the amount of land harvested per watershed is higher in Segura. Finally, the Guadiana watershed has the highest yield of corn (10,753.8 kg·ha^{-1}) but the Galicia-Costa watershed has the highest ratio of land use to available watershed. These data show that grape, nectarine and corn production do not always follow the tendency of having higher yields in watersheds with the most water availability and the largest surface area. In addition, as shown in Table 3.4, although crop yield is high in one watershed (e.g. 8,421.4 kg·ha^{-1} for grapes in Galicia-Costa), the ratio of land use harvested to surface area in that watershed is higher than in other watersheds (2.7% in Segura).

Assessing the blue WF comparison of these three crops between Spanish watersheds, the results show that, by crop, the highest WF$_b$ corresponds to grapes and nectarines in the Segura watershed and corn in Guadalete and

Barbate, all three of which are southern basins. On the other hand, the lowest blue water footprints by crop are for nectarines and corn in the Cantabric watershed and for grapes in the inland watersheds of the Basque Country in the north of the country (Table 3.5).

Figure 3.2 compares the three crops grown throughout Spain and shows that grapes have the lowest blue water (m^3) consumption values per kg of product. In almost all basins, corn displays high and very high ratios of blue water consumption to product yield (Appendix A3- Table A3.2).

The southern basins have higher consumption values than northern basins. In any discussion about the blue water footprint of different watersheds, it is important to take the available water for each basin into account. For instance, the Duero with 24.1 km^3 (Number 5 in Figure 3.2), Tajo with 18.4 km^3 (Number 8 in Figure 3.2) and Ebro with 16.1 km^3 (Number 6 in Figure 3.2) basins in central-northern zone of the country are the most abundant watersheds in Spain, whereas the Guadalete-Barbate with 0.4 km^3 (Number 15 in Figure 3.2), Tinto, Odiel and Piedras with 0.6 km^3 (Number 12 in Figure 3.2) and Segura with 0.8 km^3 with (Number 14 in Figure 3.2) are the watersheds with few available water. Therefore, considering the blue water footprint for each crop studied as it corresponds to each watershed, it is clear that the southern basins will generate higher values than the northern basins.

Table 3.4 Total surface area and water availability of Spanish watersheds, percentage of watershed land use for grape, nectarine and corn acreages and their respective yields. Bold figure mean the highest value for each assessed parameter (surface, water availability, % land used, average yield) among the Spanish watersheds.

Code in Maps	Watershed	Surface (km^2)[1]	Water availability (km^3·year^{-1})[2]	Grape		Nectarine		Corn	
				% land used[3]	average yield[3] (kg·ha^{-1})	land[3] used (%)	average yield[3] (kg·ha^{-1})	land used[3] (%)	average yield[3] (%)
1	Galicia-Costa	16437.4	10.2	3.83x10^{-01}	**8421.4**	2.70x10^{-02}	8579.6	4.33x10^{-01}	7185.4
2	Miño-Sil	17619.2	10.9	5.05x10^{-01}	7203.6	8.20x10^{-03}	5938.8	**8.18x10^{-01}**	8840.2
3	Cantabric	19002.9	9.5	2.89x10^{-02}	4671.0	1.03x10^{-03}	757.6	6.64x10^{-02}	3392.7
4	Internal watersheds of Basque countries	6411.7	3.7	1.11x10^{+00}	7207.7	2.70x10^{-02}	9464.9	4.41x10^{-01}	4690.2
5	Duero	78859.7	**24.1**	1.10x10^{-01}	3128.3	1.92x10^{-04}	6053.8	1.79x10^{-01}	9676.6
6	Ebro	**85939.3**	16.1	1.86x10^{-01}	3628.4	6.33x10^{-02}	**17725.3**	2.25x10^{-01}	9209.4
7	Internal watersheds of Catalonia	18047.4	3.9	9.58x10^{-01}	6321.4	8.72x10^{-02}	13234.8	2.50x10^{-01}	6996.8
8	Tajo	55764.5	18.4	2.31x10^{-01}	2205.7	6.01x10^{-03}	2390.2	6.38x10^{-02}	6895.8
9	Júcar	45117.9	3.7	1.28x10^{+00}	3638.9	4.91x10^{-02}	7032.3	6.31x10^{-02}	9731.8
10	Balearic	8790.1	0.7	1.92x10^{-01}	3973.9	5.01x10^{-03}	9090.9	5.97x10^{-02}	5523.8
11	Guadiana	55468.8	6.8	1.98x10^{+00}	5040.5	4.02x10^{-02}	10056.8	2.46x10^{-01}	**10753.8**
12	Tinto, Odiel and Piedras	4940.4	0.6	1.26x10^{+00}	6878.5	4.24x10^{-01}	10597.5	9.21x10^{-02}	9833.7
13	Guadalquivir	57725.9	6.3	3.16x10^{-01}	4474.2	4.71x10^{-02}	9063.2	9.85x10^{-02}	10569.3
14	Segura	20115.2	0.8	**2.73x10^{+00}**	2432.4	**4.33x10^{-01}**	14770.5	1.49x10^{-01}	9941.3
15	Guadalete and Barbate	6474.0	0.4	1.58x10^{+00}	6821.1	5.96x10^{-02}	4498.1	5.79x10^{-01}	9256.6
16	Mediterranean Watersheds of Andalusia	20012.9	3.4	1.44x10^{-01}	4144.7	1.67x10^{-02}	7167.9	3.99x10^{-02}	5216.6

[1] Surface from SIA.

[2] Yearly water availability from WaterGap (Alcamo et al. 2003).

[3] Produced areas (ha) and yield for each crop by MAGRAMA and SIA.

Table 3.5 Blue water consumption (m^3 per kg of product) for grape, nectarine and corn among the different watersheds in Spain. Bold figures show the highest blue water footprint for each crop among Spanish watersheds.

	Watersheds in Spain	Location	grape[1,2] (m^3kg^{-1})	nectarine[1,2] (m^3kg^{-1})	corn[1,2] (m^3kg^{-1})
1	Galicia-Costa	Northern Basins	0.013	0.047	0.148
2	Miño-Sil	Northern Basins	0.025	0.073	0.173
3	Cantabric	Northern Basins	0.011	0.019	0.144
4	Internal watersheds of the Basque country	Northern Basins	0.019	0.034	0.210
5	Duero	Northern Basins	0.091	0.050	0.343
6	Ebro	Northern Basins	0.068	0.104	0.297
7	Internal watersheds of Catalonia	Northeastern Basins	0.059	0.244	0.475
8	Tajo	Central basins	0.182	0.162	0.504
9	Júcar	Eastern Basins	0.166	0.153	0.517
10	Balearic Islands	Northeastern Basins	0.087	0.131	0.370
11	Guadiana	Central basins	0.145	0.266	0.541
12	Tinto, Odiel and Piedras	Central basins	0.133	0.323	0.613
13	Guadalquivir	Southern basins	0.212	0.422	0.569
14	Segura	Southern basins	0.175	0.249	**0.748**
15	Guadalete and Barbate	Southern basins	0.132	**0.806**	0.646
16	Mediterranean Watersheds of Andalusia	Southern basins	**0.228**	0.364	0.599

[1] Source (Mekonnen et al. 2010) Water footprint per ton of crop or derived crop product (Code Faostat: 507 Grapefruit. 534 peaches and nectarines. 56 maize. corn) during (1996-2005) adapted for reference area (m^3 $tonne^{-1}$).

[2] Produced yields for each crop by MAGRAMA.

Figure 3.2 shows that grape production has low levels (<0.25 m^3kg^{-1}) of blue water consumption by product in all Spanish watersheds, whereas corn shows medium and high levels (among 0.25-0.75 m^3kg^{-1}) in almost all watersheds except for in the Galicia-Costa, Miño-Sil , Cantabric and the inland watersheds of the Basque Country, which have low levels. Nectarine production has low consumption levels in almost all central-northern basins, but in the south it has medium levels in the Segura, Tinto, Odiel and Piedras, Guadalquivir and Mediterranean watersheds of Andalusia. Furthermore, a considerably high level of consumption (>0.75 m^3kg^{-1}) predominates in the Guadalete and Barbate basin.

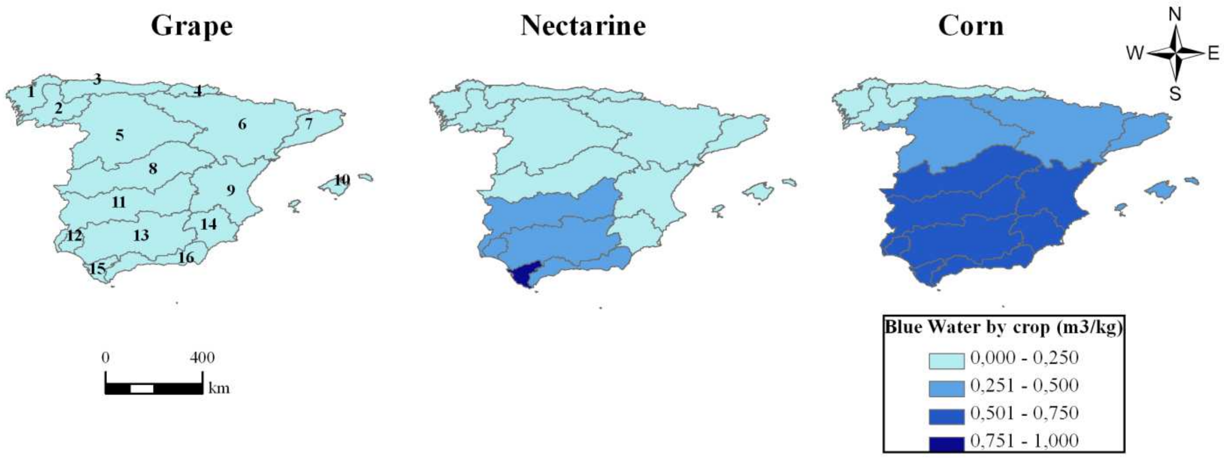

Figure 3.2 Water consumption expressed as blue water per kg of crop production (grape, nectarine, corn) (m^3kg^{-1}). Numbers in grape map refer to Spanish watersheds: (1) Galicia-Costa (2) Miño-Sil (3) Cantabric (4) Internal watersheds of the Basque countries (5) Duero (6) Ebro (7) Internal watersheds of Catalonia (8) Tajo (9) Júcar (10) Balearic Islands (11) Guadiana (12) Tinto, Odiel and Piedras (13) Guadalquivir (14) Segura (15) Guadalete and Barbate (16) Mediterranean Watersheds of Andalucia.

Table 3.6 shows the results of the water stress index (WSI) developed by Pfister et al. (2009) and Smakhtin et al. (2004) for each watershed in Spain. According to Pfister et al. (2009), the water stress indices (WSI) of Spanish watersheds are on the rise in a north-westerly to south-easterly direction. If we divided Spain into two halves (by drawing a diagonal line), the north-western (the central upper-left) watersheds (Duero, Ebro, Tajo, Galicia-Costa, Miño-Sil, Cantabric, the inland watersheds of the Basque Country, Tinto, Odiel and Piedras) have a WSI of below 0.55, while the south-eastern watersheds (the central lower-right) have high WSIs from 0.72 for the inland watersheds of Catalonia to 1.00 for Guadalquivir, Segura, and Guadalete and Barbate. This trend is thus closely related to a higher water demand and a shortage of availability from north-west to south-east.

Although the water stress index is always a ratio of water consumption and water availability, the absolute values of WSI proposed by these authors cannot be directly compared, since Milà i Canals et al.(2009) (based on the work of Smakhtin et al. (2004)) reported WSI values for Guadalquivir, Ebro and Duero (Guadalquivir (1.774)> Ebro (0.888) > Duero (0.554)) that were higher, for example, than Pfister et al. (2009)

(Guadalquivir (1.000)> Ebro (0.2592) > Duero (0.1711)), this is clearly due to the different approaches, as explained in section 3.2.

Table 3.6 Results for crops studied and the different Spanish watersheds applying the water stress index by Pfister et al. (2009) and Smakhtin et al. (2004) as characterization factors.

Spanish watersheds	Water Stress Index		grape [m^3 kg^{-1}]		nectarine [m^{3}kg^{-1}]		corn [m^{3}kg^{-1}]	
	(a)	(b)	(a)	(b)	(a)	(b)	(a)	(b)
Case Study in Ebro Valley	0.259	0.888[1]	4.60x10^{-02}	1.59x10^{-01}	2.50x10^{-02}	8.50x10^{-02}	8.80x10^{-02}	3.03x10^{-01}
Galicia-Costa	0.023	0.100[2]	2.86x10^{-04}	1.26x10^{-03}	1.07x10^{-03}	4.73x10^{-03}	3.36x10^{-03}	1.48x10^{-02}
Miño-Sil	0.020	0.100[2]	4.97x10^{-04}	2.49x10^{-03}	1.46x10^{-03}	7.30x10^{-03}	3.45x10^{-03}	1.73x10^{-02}
Cantabric	0.018	0.200[2]	2.06x10^{-04}	2.24x10^{-03}	3.42x10^{-04}	3.72x10^{-03}	2.66x10^{-03}	2.89x10^{-02}
Internal watersheds of the Basque countries	0.113	0.300[2]	2.15x10^{-03}	5.70x10^{-03}	3.80x10^{-03}	1.01x10^{-02}	2.37x10^{-02}	6.29 x10^{-02}
Duero	0.171	0.554[1]	1.56x10^{-02}	5.06x10^{-02}	8.57x10^{-03}	2.78x10^{-02}	5.87x10^{-02}	1.90x10^{-01}
Ebro	0.259	0.888[1]	1.75x10^{-02}	6.01x10^{-02}	2.71x10^{-02}	9.27x10^{-02}	7.70x10^{-02}	2.64x10^{-01}
Internal watersheds of Catalonia	0.724	0.900[2]	4.30x10^{-02}	5.35x10^{-02}	1.77x10^{-01}	2.20x10^{-01}	3.44x10^{-01}	4.27x10^{-01}
Tajo	0.534	0.600[2]	9.70x10^{-02}	1.09x10^{-01}	8.64x10^{-02}	9.70x10^{-02}	2.69x10^{-01}	3.02x10^{-01}
Júcar	0.992	1.000[2]	1.65x10^{-01}	1.66x10^{-01}	1.51x10^{-01}	1.53x10^{-01}	5.12x10^{-01}	5.17x10^{-01}
Balearic Islands	0.959	0.800[2]	8.38x10^{-02}	6.99x10^{-02}	1.26x10^{-01}	1.05x10^{-01}	3.55x10^{-01}	2.96x10^{-01}
Guadiana	0.993	0.800[2]	1.44x10^{-01}	1.16x10^{-01}	2.64x10^{-01}	2.13x10^{-01}	5.37x10^{-01}	4.33x10^{-01}
Tinto. Odiel and Piedras	0.251	0.500[2]	3.33x10^{-02}	6.63x10^{-02}	8.09x10^{-02}	1.61x10^{-01}	1.54x10^{-01}	3.06x10^{-01}
Guadalquivir	1.00	1.774[1]	2.12x10^{-01}	3.76x10^{-01}	4.22x10^{-01}	7.48x10^{-01}	5.69x10^{-01}	1.01x10^{+00}
Segura	1.00	1.000[2]	1.75x10^{-01}	1.75x10^{-01}	2.49x10^{-01}	2.49x10^{-01}	7.48x10^{-01}	7.48x10^{-01}
Guadalete and Barbate	1.00	0.700[2]	1.32x10^{-01}	9.25x10^{-02}	8.06x10^{-01}	5.65x10^{-01}	6.46x10^{-01}	4.52x10^{-01}
Mediterranean Watersheds of Andalusia	0.996	1.00	2.27x10^{-01}	2.28x10^{-01}	3.62x10^{-01}	3.64x10^{-01}	5.96x10^{-01}	5.99x10^{-01}
Percentil (90%)	1.00	1.00	0.19	0.20	0.62	0.67	0.39	0.46
Percentil (50%)	0.63	0.75	0.06	0.07	0.31	0.30	0.11	0.13

[1] WSI data from Milà i Canals et al. (2009)

[2] WSI data from Smakhtin et al. (2004)

The upper maps described in Figure 3.3 show a comparison of the impacts of the three crops based on the characterization factors provided by Pfister et al. 2009. It is clear that the impact of all three crops is higher in the southern basins, but this difference is most striking for corn (0.5-0.75 m^{3}kg^{-1}). In addition, the Guadalete and Barbate watershed show very high levels for nectarines (>0.75 m^{3}kg^{-1}). The lower maps in Figure 3.3

compare the three crops using the characterization factors proposed by Smakhtin et al. (2004).

Generally, grape production in Spain has low levels of impact in almost all basins, except for a few southern basins like Guadalquivir, where impact is at a medium level. The WSI in Spain for corn indicates that some northern basins (the Galicia Costa, Miño-Sil, Cantabric, and the inland watersheds of the Basque Country) have low impacts compared to the rest of the country, where medium (0.25-0.50 m^3 kg^{-1}) and high levels (0.50-0.75m^3kg^{-1}) were found. Special attention must be paid to Guadalquivir and Jucar, which were found to have very high levels of impact (>0.75 m^3kg^{-1}).

Finally, impacts in Spain for nectarines were found to be medium and high only in a few south-western basins, with the levels of impact found in Guadalete and Barbate, and Guadalquivir high and very high.

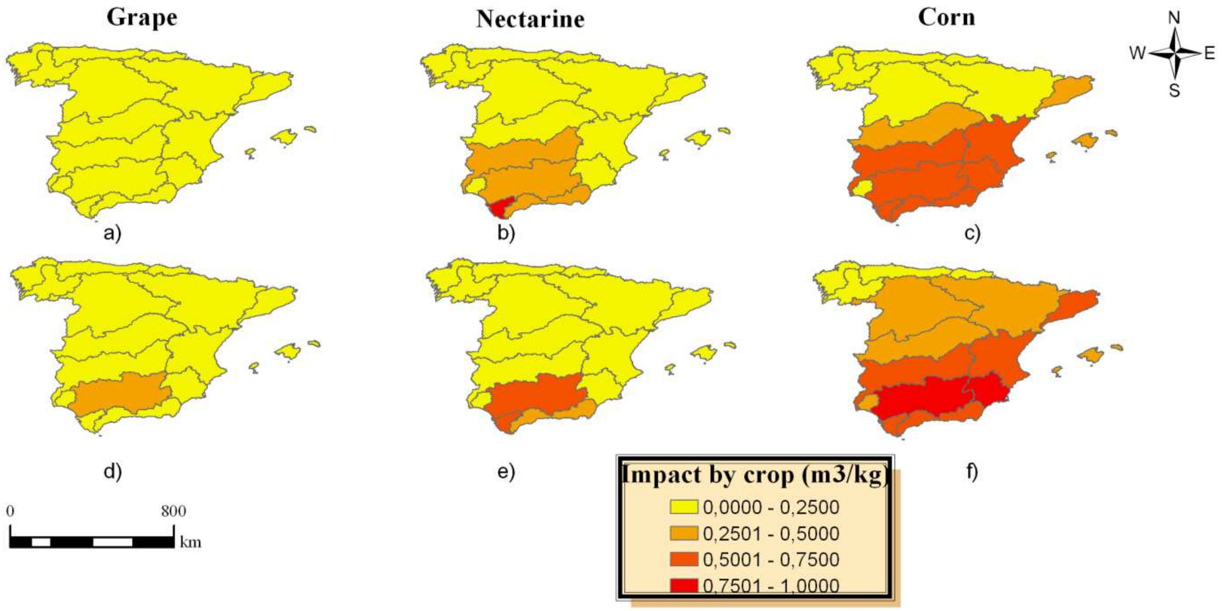

Figure 3.3 Midpoint impacts in Spain for grape, nectarine and corn using the WSIs of Pfister, Milà i Canals and Smakhtin: a) Impact for grape b) Impact for nectarine c) Impact for corn using the model by Pfister et al. (2009); c) Impact for grape d) Impact for nectarine e) Impact for corn using the model by Smakhtin et al. (2004).

Taking into account the characterization factors at the endpoint level proposed by Pfister et al. (2009), ecosystem quality damage is present in all Spanish basins, which led us to assess this endpoint impact through the three crops studied in this work (Appendix A3-Table A3.3).

As shown in Figure 3.4, ecosystem quality damage is very high for corn production in the Guadiana basin because Pfister et al. (2009) proposed the highest characterization factor of ecosystem quality in Spanish watersheds based on 6.38x10^{-01} m^{2}·yr·m^{-3}. An intermediate degree of damage occurs at Guadalquivir, Jucar and Segura in the south of the country. Grape and nectarine production causes little damage to ecosystem quality in all Spanish watersheds except for grape production in Guadiana, which was found to cause a mid-range impact.

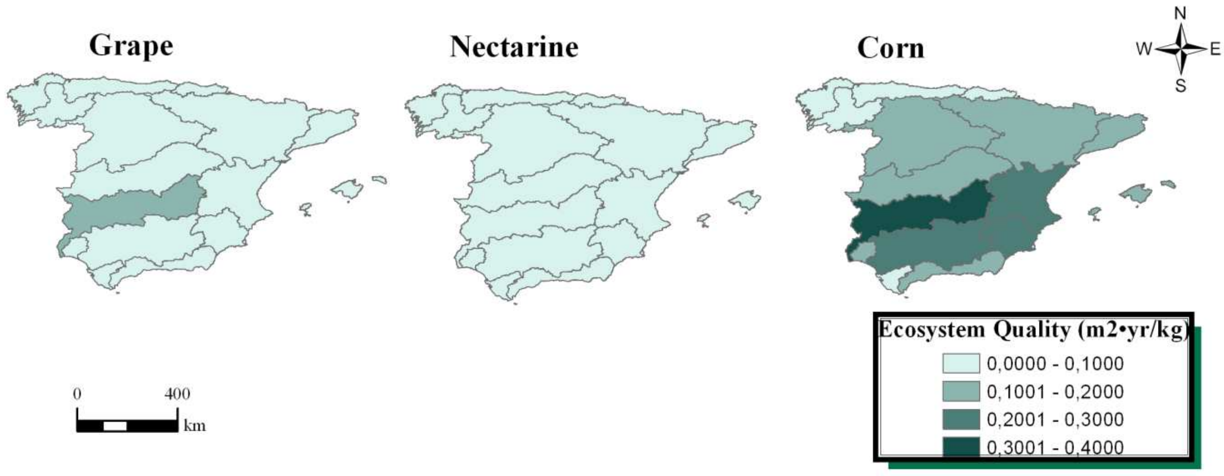

Figure 3.4 Damage to the natural environment at the endpoint level: ecosystem quality for grape, nectarine and corn and in Spain.

Getting an idea of farm environmental sustainability, we compared the impact scores resulting from our case studies in the framework of Spanish watersheds. Figure 3.5 shows the results of applying Pfister et al.'s (2009) WSI method for each Spanish watershed and each crop: grape, nectarine and corn. We considered anything below 50% of impact as having a good level of environmental sustainability. It could be argued that this 50% of impact is an arbitrary value that lower values may be more sustainable, and in fact this is true. As for the selection of the different watersheds to be compared, in our case we used the criteria of the country of Spain as a reference framework as most policy decisions are made at the country level, so the 50% represents a compromise figure. The grape farm from our case study is included in the 50th percentile so it should be comparable to the northern Cantabric, Duero and Tinto, Odiel and Piedras watersheds. The impacts in Spanish watersheds resulting from grape production are in Guadalquivir and the Mediterranean watersheds of Andalusia, which are above the 90th percentile, which can be explained by a low yield and high WSI due to low availability and high consumption (Figure 3.5). The experimental case study of nectarine production is also within the 50th impact percentile and is comparable to the same watersheds as grape production. Of the impacts to Spanish watersheds from corn, the Segura and Guadalete and Barbate watersheds were found to have the highest impact due to high WSIs, low yield (compared to the production in other watersheds) and high water consumption. Finally, the experimental case study of corn is below the 50th percentile, although it is important to point out that the impact of corn in Spanish watersheds falls within the 90th percentile, with Guadalquivir and Guadalete and Barbate exceeding this percentage due to high water consumption, (Table 3.5) and high WSIs (Table 3.6).

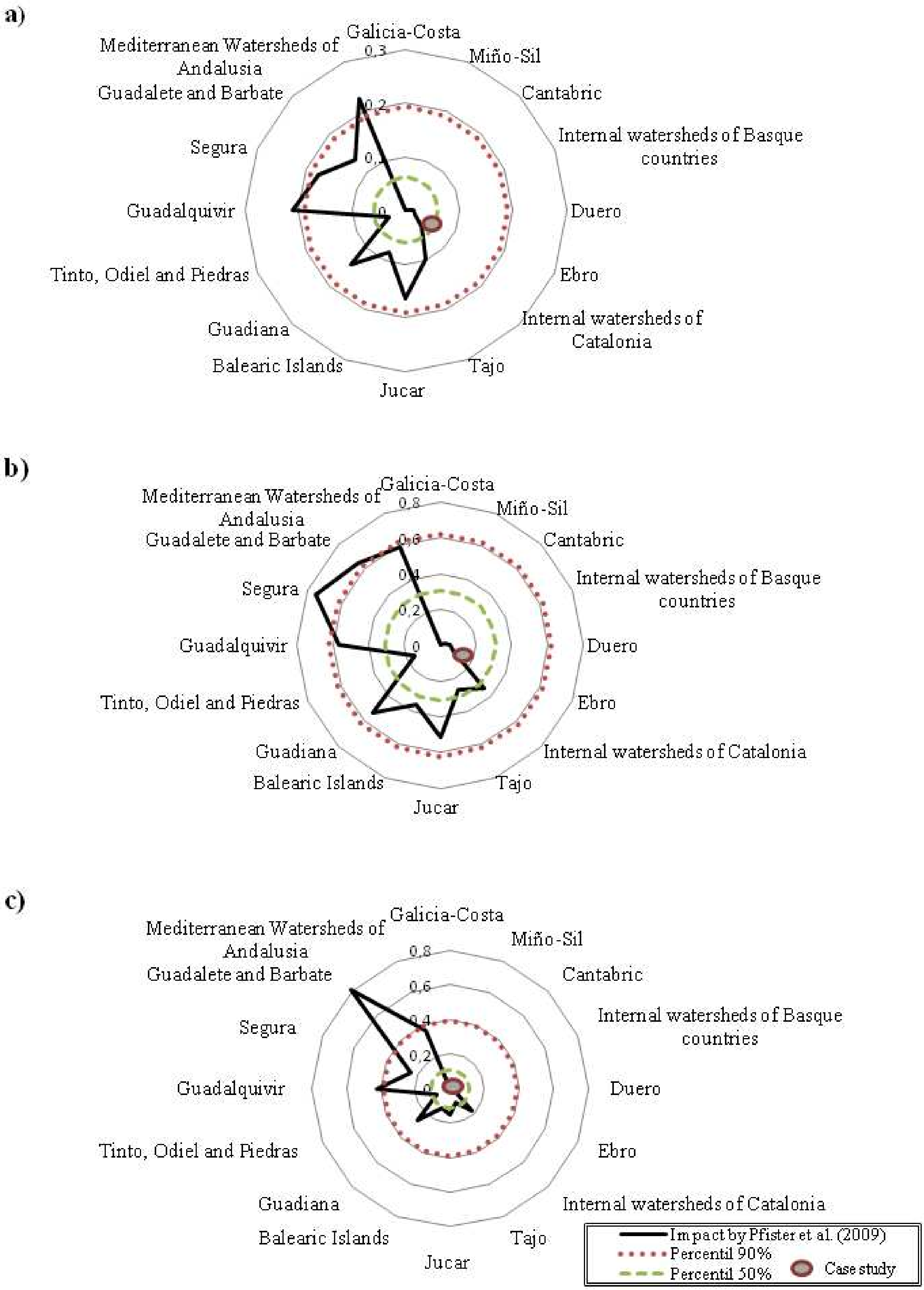

Figure 3.5 Environmental indicator of water consumption in different watersheds for a) grape, b) nectarine and c) corn.

This approach can provide an overview of the sustainability of this farm in this year with the information now available. It is also important to be aware that this is a relative comparison, rating impacts as better (or worse) than the rest of Spain. Obviously, characterization factors for such a local and complex process as water consumption need to be further researched to expand the body of knowledge on the influence of water use on ecosystems.

Generally, comparing all these results with those already published in other works provides insight into the position of these farms. The amount of blue water needed (Table 3.2) for the three crops planted in the Ebro Basin (179.1 m^3tonne^{-1} for grapes, 108.7 m^3tonne^{-1} for nectarines and 325.2 m^3tonne^{-1} for corn) is lower for nectarines and corn and higher for grapes compared to the average (Mekonnen et al. 2010) amount of blue water for these three crops in Spain (116 m^3tonne^{-1} grape, 224 m^3tonne^{-1} nectarine and 406 m^3tonne^{-1} corn).

The results of our case study with regard to midpoint impacts (46 m^3tonne^{-1} for grapes, 25 m^3tonne^{-1} for nectarines and 88 m^3tonne^{-1} for corn) can be compared to others studies carried out in the Ebro watershed, like those of Milà i Canals et al. (2009), who found an impact of 56.5 m^3tonne^{-1} for broccoli. Comparing all these crops, we can conclude that the most suitable crop for cultivation in this area is the nectarine, as its agro-climatic needs coincide well with the characteristics of the Ebro Basin.

3.4 Conclusions

This work presents an attempt to assess the sustainability of irrigation water consumption and to integrate it into managers' decision dashboards to assist in strategic decision-making by means of the life cycle impact assessment methods developed up to now.

We compared different methods of characterizing the water stress index (Milà i Canals et al. 2009; Pfister et al. 2009; Smakhtin et al. 2004) with three crops (grapes, nectarines and corn). We also discussed the findings of our experimental crop assessments and their shortcomings and identified some of the resulting research gaps.

A practical trial of sustainability assessment in the three case studies was conducted in the Ebro Basin near Lleida (north-eastern Spain), where three irrigated farms where chosen during the 2011 growing season. The crops were grape, nectarine and corn for silage. Because the information is useful to the irrigator, assessments were calculated for each farm management unit, which may correspond to a particular field where water consumption, potential water crop requirements and yield could be assigned.

We discussed the problems and limitations discovered when these methods were applied to our case study in the Ebro Basin. To summarize, the existing methods were tested in three crop case studies but none of them yielded complete and precise information. Although all these methods are complementary, the method proposed by Pfister et al. (2009) provided the most accurate results, as it considers seasonality and water reservoirs. However, a very interesting issue that remains relatively unexplored is how the native species in different ecosystems are affected by water use.

We found considerable differences between experimental and statistical data, emphasizing the high degree of variability in results depending on the information source, crop variety, and cultivation technique; therefore caution must be exercised when basing decisions on data of one type or another.

We also suggest a first approach for measuring the sustainability of water use by watersheds at the country level, looking at water availability, the harvested surface area of the crops studied and yields.

Therefore, it is our belief that further research is needed to shed light into how ecosystems are affected by water use. Focusing on damage to ecosystems is crucial to make suitable decisions on sustainability and to understand the variability of certain types of data. It is essential to work towards standardizing water use assessment methods to develop a holistic method for evaluating water management.

Therefore, two topics for future research stem from this work: (1) the need to address the uncertainty in the input data and in the different models of the midpoint and endpoint methods studied and (2) the need to implement decision-making in irrigation management based on productive and environmental indicators in order to achieve sustainability.

Appendix A3

Table A3.1 Climatic parameters of three crops studied by week: a) grape, b) nectarine c) corn.

Crop	GRAPE (a)				
Parameter	Crop	2011		2010	
Week	Coefficient	Precipitation	Evapotranspiration	Precipitation	Evapotranspiration
Units	Kc	P (l m^{-2})	Eto (l m^{-2})	P (l m^{-2})	Eto (l m^{-2})
1-12	0	0	0	0	0
13	0.1	0.5	10.4	0	9.6
14	0.1	0	24.2	6.5	17.3
15	0.1	0.6	24.6	2.8	12.3
16	0.1	20	20.6	9.8	21.6
17	0.1	0	23.3	12.5	29.7
18	0.3	27.8	28.9	10.5	19.1
19	0.3	0.5	26.4	2.8	24.6
20	0.3	0	31.1	0	31.7
21	0.3	2.5	32.3	0	35.4
22	0.3	19	25.6	5	34.1
23	0.3	14.5	24.2	16.5	21.3
24	0.3	0	37.7	10	33.3
25	0.3	0	42.1	0	34.8
26	0.3	1.3	42.7	0.8	36
27	0.3	0	43.2	0	42.3
28	0.3	6	41.6	0	37.5
29	0.3	0	39.1	0	36.3
30	0.4	0	35.5	0	35.5
31	0.4	0	37.1	1	30.6
32	0.4	0	35.2	0	32.1
33	0.4	0	44	0	32.2
34	0.4	0	37.7	0	36.5
35	0.4	11.5	26.8	0	24
36	0.4	0	30.3	0	26.3
37	0.2	0	30.8	0	3.3
38	0.2	0	23.4	0	0
39	0.2	0	18.8	0	0
40-52	0	0	0	0	0

Crop		NECTARINE (b)			
Parameter		2011		2010	
Week	Crop Coefficient	Precipitation	Evapotranspiration	Precipitation	Evapotranspiration
Units	Kc	P (l m^{-2})	ETo (l m^{-2})	P (l m^{-2})	ETo (l m^{-2})
1-9	0	0	0	0	0
10	0.3	2.8	15.3	15	12.6
11	0.3	56.9	11.3	0	17.4
12	0.4	8.2	17.7	9.1	13.6
13	0.4	1.1	23.3	2.5	23.3
14	0.5	0.1	26.5	10.3	21.2
15	0.6	0	30.5	14.8	18.5
16	0.6	17.1	23.1	0.9	22.5
17	0.7	9.6	25.0	6.2	30.3
18	0.8	0	32.9	29.3	22.7
19	0.8	31.8	30.7	3.6	27.8
20	0.9	0	35.6	0	34.6
21	1	0.3	38.3	0	39.8
22	1	24.8	29.6	13.3	37.1
23	1	7.8	29.0	28.5	32.8
24	1	0	37.5	27.4	29.4
25	0.6	0	39.5	0	40.3
26	0.5	0.1	43.4	4.7	40.0
27	0.5	5.1	39.1	0	40.8
28	0.5	3.6	39.2	0.3	44.0
29	0.5	0	39.2	0.4	39.6
30	0.5	0	38.2	0	41.5
31	0.5	0	36.6	43.3	36.3
32	0.4	0	37.3	0	34.8
33	0.4	2.3	35.3	0.4	29.9
34	0.4	0	36.4	0.1	37.7
35	0.4	0.2	29.9	0	31.1
36	0.4	8.1	29.5	1.7	30.3
37	0.4	0	28.9	0	24.7
38	0.3	0	23.6	59.9	19.4
39	0.3	0.7	23.3	0.6	22.8
40	0.3	0	22.0	0.8	20.4
41	0.3	0	20	38.9	12.6
42	0.3	0	13.4	0.1	15.5
43	0.3	24.2	10.6	0	13.4
44	0.3	11.9	8.1	0.8	11.5
45-52	0	0	0	0	0

Crop	CORN (c)				
Parameter	Crop Coefficient	2011		2010	
Week		Precipitation	Evapotranspiration	Precipitation	Evapotranspiration
Units	Kc	P (l m^{-2})	Eto (l m^{-2})	P (l m^{-2})	Eto (l m^{-2})
1-17	0	0	0	0	0
18	0	1.1	29.4	0	0
19	0	27.3	29.8	0	0
20	0	0	34	0	0
21	0.4	1	36.7	0.3	38.5
22	0.4	33.1	26.8	0	37.7
23	0.4	26.4	26.5	26.3	32.4
24	0.5	0	35.2	14.7	29.4
25	0.6	0	37.4	0	39.5
26	0.8	0.1	39.4	0	40.6
27	0.9	15.3	35.3	18.1	40.7
28	1.1	4	36.5	0	42.0
29	1.1	0	34.1	0	38.2
30	1.1	0	29.7	0	39.1
31	1.1	0	34.8	13.8	35.0
32	1.1	0	33.2	0.1	35.3
33	1.1	0.6	32.8	2.9	29.7
34	1.1	0	32.9	0	36.5
35	0.8	25.9	28.6	0	28.8
36	0.5	0.2	27.8	4.5	28.2
37	0.3	0	27.4	0	23.7
38	0	0	23.1	0	0
39	0	0	21.8	0	0
40-52	0	0	0	0	0

Table A3.2. Production, surface, yield and blue water footprint of Spanish provinces: a) grape, b) nectarine, c) corn.

		GRAPE (a)			
Autonomous community	**Province**	**Production (kg)**	**Surface (ha)**	**Yield ($kg \cdot ha^{-1}$)**	**Blue water ($m^3 \cdot kg^{-1}$)**
Galicia	A Coruña	$2.58\text{x}10^{+07}$	$2.97\text{x}10^{+03}$	$8.69\text{x}10^{+03}$	$1.22\text{x}10^{-02}$
	Lugo	$2.46\text{x}10^{+07}$	$2.55\text{x}10^{+03}$	$9.64\text{x}10^{+03}$	$1.10\text{x}10^{-02}$
	Ourense	$8.27\text{x}10^{+07}$	$1.19\text{ x}10^{+04}$	$6.93\text{x}10^{+03}$	$1.53\text{x}10^{-02}$
	Pontevedra	$1.19\text{x}10^{+08}$	$1.62\text{x}10^{+04}$	$7.35\text{x}10^{+03}$	$1.44\text{x}10^{-02}$
Asturias	Asturias	$6.00\text{x}10^{+05}$	$1.25\text{x}10^{+02}$	$4.80\text{x}10^{+03}$	$7.34\text{x}10^{-03}$
Cantabria	Cantabria	$1.00\text{x}10^{+05}$	$4.20\text{x}10^{+01}$	$2.38\text{x}10^{+03}$	$1.67\text{x}10^{-02}$
Basque country	Álava	$8.73\text{x}10^{+07}$	$1.31\text{x}10^{+04}$	$6.68\text{x}10^{+03}$	$2.20\text{x}10^{-02}$
	Gipuzkoa	$2.10\text{x}10^{+06}$	$2.20\text{x}10^{+02}$	$9.55\text{x}10^{+03}$	$1.54\text{x}10^{-02}$
	Vizcaya	$1.40\text{x}10^{+06}$	$2.20\text{x}10^{+02}$	$6.36\text{x}10^{+03}$	$2.31\text{x}10^{-02}$
Navarra	Navarra	$1.56\text{x}10^{+08}$	$2.57\text{x}10^{+04}$	$6.08\text{x}10^{+03}$	$5.50\text{x}10^{-02}$
La Rioja	La Rioja	$3.19\text{x}10^{+08}$	$0\text{x}10^{+00}$	$0\text{x}10^{+00}$	$0\text{x}10^{+00}$
Aragon	Huesca	$2.27\text{x}10^{+07}$	$6.12\text{x}10^{+03}$	$3.71\text{x}10^{+03}$	$6.54\text{x}10^{-02}$
	Teruel	$7.50\text{x}10^{+06}$	$4.10\text{x}10^{+03}$	$1.83\text{x}10^{+03}$	$1.33\text{x}10^{-01}$
	Zaragoza	$1.16\text{x}10^{+08}$	$3.87\text{x}10^{+04}$	$3.00\text{x}10^{+03}$	$8.08\text{x}10^{-02}$
Catalonia	Barcelona	$1.72\text{x}10^{+08}$	$2.34\text{x}10^{+04}$	$7.34\text{x}10^{+03}$	$5.01\text{x}10^{-02}$
	Girona	$1.55\text{x}10^{+07}$	$2.72\text{ x}10^{+03}$	$5.71\text{x}10^{+03}$	$6.45\text{x}10^{-02}$
	Lleida	$3.20\text{x}10^{+07}$	$5.79\text{ x}10^{+03}$	$5.53\text{x}10^{+03}$	$6.66\text{x}10^{-02}$
	Tarragona	$1.65\text{x}10^{+08}$	$3.41\text{x}10^{+04}$	$4.84\text{x}10^{+03}$	$7.61\text{x}10^{-02}$
Balearic Islands		$6.70\text{x}10^{+06}$	$1.69\text{x}10^{+03}$	$3.97\text{x}10^{+03}$	$8.74\text{x}10^{-02}$
Castilla and Leon	Ávila	$6.50\text{x}10^{+06}$	$3.55\text{x}10^{+03}$	$1.83\text{x}10^{+03}$	$1.40\text{x}10^{-01}$
	Burgos	$5.20\text{x}10^{+07}$	$1.77\text{x}10^{+04}$	$2.94\text{x}10^{+03}$	$8.69\text{x}10^{-02}$
	León	$4.90\text{x}10^{+07}$	$1.25\text{x}10^{+04}$	$3.92\text{x}10^{+03}$	$6.53\text{x}10^{-02}$
	Palencia	$1.30\text{x}10^{+06}$	$6.09\text{x}10^{+02}$	$2.13\text{x}10^{+03}$	$1.20\text{x}10^{-01}$
	Salamanca	$4.80\text{x}10^{+06}$	$2.63\text{x}10^{+03}$	$1.82\text{x}10^{+03}$	$1.40\text{x}10^{-01}$
	Segovia	$7.30\text{x}10^{+06}$	$1.34\text{x}10^{+03}$	$5.44\text{x}10^{+03}$	$4.70\text{x}10^{-02}$
	Soria	$4.20\text{x}10^{+06}$	$1.42\text{x}10^{+03}$	$2.96\text{x}10^{+03}$	$8.63\text{x}10^{-02}$
	Valladolid	$6.28\text{x}10^{+07}$	$1.85\text{x}10^{+04}$	$3.39\text{x}10^{+03}$	$7.54\text{x}10^{-02}$
	Zamora	$4.71\text{x}10^{+07}$	$1.44\text{x}10^{+04}$	$3.26\text{x}10^{+03}$	$7.85\text{x}10^{-02}$
Madrid	Madrid	$3.75\text{x}10^{+07}$	$1.60\text{x}10^{+04}$	$2.34\text{x}10^{+03}$	$1.22\text{x}10^{-01}$
Castilla La Mancha	Albacete	$4.28\text{x}10^{+08}$	$1.19\text{x}10^{+05}$	$3.61\text{x}10^{+03}$	$2.00\text{x}10^{-01}$
	Ciudad Real	$1.22\text{x}10^{+09}$	$1.93\text{x}10^{+05}$	$6.31\text{x}10^{+03}$	$1.15\text{x}10^{-01}$
	Cuenca	$4.77\text{x}10^{+08}$	$9.77\text{x}10^{+04}$	$4.88\text{x}10^{+03}$	$1.48\text{x}10^{-01}$
	Guadalajara	$1.01\text{x}10^{+07}$	$2.51\text{x}10^{+03}$	$4.03\text{x}10^{+03}$	$1.79\text{x}10^{-01}$
	Toledo	$9.12\text{x}10^{+08}$	$1.41\text{x}10^{+05}$	$6.45\text{x}10^{+03}$	$1.12\text{x}10^{-01}$
Valencia	Alicante	$3.89\text{x}10^{+07}$	$1.49\text{x}10^{+04}$	$2.62\text{x}10^{+03}$	$2.06\text{x}10^{-01}$
	Castellón de la Plana	$2.30\text{x}10^{+06}$	$1.07\text{x}10^{+03}$	$2.15\text{x}10^{+03}$	$2.51\text{x}10^{-01}$
	Valencia	$2.91\text{x}10^{+08}$	$6.02\text{x}10^{+04}$	$4.84\text{x}10^{+03}$	$1.12\text{x}10^{-01}$

Murcia	Murcia	7.07x10^{+07}	4.11x10^{+04}	1.72x10^{+03}	1.51x10^{-01}
Extremadura	Badajoz	4.23x10^{+08}	8.49x10^{+04}	4.98x10^{+03}	1.47x10^{-01}
	Cáceres	1.18x10^{+07}	4.50x10^{+03}	2.62x10^{+03}	2.80x10^{-01}
Andalucia	Almería	4.10x10^{+06}	9.86x10^{+02}	4.16x10^{+03}	2.18x10^{-01}
	Cádiz	7.51x10^{+07}	1.08x10^{+04}	6.98x10^{+03}	1.30x10^{-01}
	Córdoba	4.79x10^{+07}	8.58x10^{+03}	5.58x10^{+03}	1.63x10^{-01}
	Granada	1.71x10^{+07}	4.83x10^{+03}	3.54x10^{+03}	2.57x10^{-01}
	Huelva	4.38x10^{+07}	6.34x10^{+03}	6.91x10^{+03}	1.32x10^{-01}
	Jaén	1.60x10^{+06}	5.25x10^{+02}	3.05x10^{+03}	2.98x10^{-01}
	Málaga	1.09x10^{+07}	2.82x10^{+03}	3.86x10^{+03}	2.35x10^{-01}
	Sevilla	4.90x10^{+06}	1.02x10^{+03}	4.80x10^{+03}	1.89x10^{-01}

NECTARINE (b)					
Autonomous community	**Province**	**Production (kg)**	**Surface (ha)**	**Yield (kg·ha^{-1})**	**Blue water (m^3·kg^{-1})**
Galicia	A Coruña	5.50x10^{+06}	6.03x10^{+02}	9.12x10^{+03}	2.78x10^{-02}
	Lugo	1.10x10^{+06}	1.82x10^{+02}	6.04x10^{+03}	1.71x10^{-01}
	Ourense	1.70x10^{+06}	1.85x10^{+02}	9.19x10^{+03}	2.76x10^{-02}
	Pontevedra	1.90x10^{+06}	2.15x10^{+02}	8.84x10^{+03}	2.87x10^{-02}
Asturias	Asturias	3.00x10^{+05}	0.00x10^{+00}	0.00x10^{+00}	0.00x10^{+00}
Cantabria	Cantabria	0.00x10^{+00}	1.00x10^{+00}	0.00x10^{+00}	0.00x10^{+00}
Basque country	Álava	0.00x10^{+00}	1.00x10^{+00}	0.00x10^{+00}	0.00x10^{+00}
	Gipuzkoa	0.00x10^{+00}	0.00x10^{+00}	0.00x10^{+00}	0.00x10^{+00}
	Vizcaya	1.00x10^{+05}	9.00x10^{+00}	1.11x10^{+04}	4.07x10^{-02}
Navarra	Navarra	1.91x10^{+07}	8.49x10^{+02}	2.25x10^{+04}	8.76x10^{-02}
La Rioja	La Rioja	2.67x10^{+07}	9.48x10^{+02}	2.82x10^{+04}	8.70x10^{-02}
Aragon	Huesca	1.67x10^{+08}	8.56x10^{+03}	1.95x10^{+04}	1.20x10^{-01}
	Teruel	2.49x10^{+07}	2.06x10^{+03}	1.21x10^{+04}	1.92x10^{-01}
	Zaragoza	1.08x10^{+08}	6.14x10^{+03}	1.76x10^{+04}	1.32x10^{-01}
Catalonia	Barcelona	8.80x10^{+06}	1.08x10^{+03}	8.16x10^{+03}	3.36x10^{-01}
	Girona	9.00x10^{+06}	4.88x10^{+02}	1.84x10^{+04}	1.48x10^{-01}
	Lleida	2.78x10^{+08}	1.24x10^{+04}	2.24x10^{+04}	1.22x10^{-01}
	Tarragona	4.44x10^{+07}	3.00x10^{+03}	1.48x10^{+04}	1.85x10^{-01}
Balearic Islands		4.00x10^{+05}	4.40x10^{+01}	9.09x10^{+03}	1.31x10^{-01}
Castilla and Leon	Ávila	4.00x10^{+05}	5.50x10^{+01}	7.27x10^{+03}	1.63x10^{-01}
	Burgos	8.00x10^{+05}	3.20x10^{+01}	2.50x10^{+04}	4.75x10^{-02}
	León	0.00x10^{+00}	5.00x10^{+00}	0.00x10^{+00}	0.00x10^{+00}
	Palencia	0.00x10^{+00}	0.00x10^{+00}	0.00x10^{+00}	0.00x10^{+00}
	Salamanca	1.00x10^{+05}	1.00x10^{+01}	1.00x10^{+04}	1.19x10^{-01}
	Segovia	0.00x10^{+00}	0.00x10^{+00}	0.00x10^{+00}	0.00x10^{+00}
	Soria	0.00x10^{+00}	0.00x10^{+00}	0.00x10^{+00}	0.00x10^{+00}
	Valladolid	0.00 x10^{+00}E+00	3.00x10^{+00}	0.00x10^{+00}	0.00x10^{+00}
	Zamora	2.00 x10^{+00}E+05	2.10x10^{+01}	9.52x10^{+03}	1.25x10^{-01}

Madrid	Madrid	0.00 x10^{+00}E+00	1.00x10^{+00}	0.00x10^{+00}	0.00x10^{+00}
Castilla La Mancha	Albacete	5.30 x10^{+00}E+06	4.35x10^{+02}	1.22x10^{+04}	1.14x10^{-01}
	Ciudad Real	2.00 x10^{+00}E+05	4.80x10^{+01}	4.17x10^{+03}	3.33x10^{-01}
	Cuenca	0.00 x10^{+00}E+00	5.00x10^{+00}	0.00x10^{+00}	0.00x10^{+00}
	Guadalajara	0.00 x10^{+00}E+00	1.00x10^{+00}	0.00x10^{+00}	0.00x10^{+00}
	Toledo	3.00 x10^{+00}E+06	7.55x10^{+02}	3.97x10^{+03}	3.49x10^{-01}
Valencia	Alicante	5.90 x10^{+00}E+06	6.31x10^{+02}	9.35x10^{+03}	1.56x10^{-01}
	Castellón de la Plana	3.70 x10^{+00}E+06	5.53x10^{+02}	6.69x10^{+03}	2.18x10^{-01}
	Valencia	4.09 x10^{+00}E+07	6.91x10^{+03}	5.92x10^{+03}	2.46x10^{-01}
Murcia	Murcia	2.59 x10^{+00}E+08	1.45x10^{+04}	1.79x10^{+04}	2.60x10^{-01}
Extremadura	Badajoz	1.02 x10^{+00}E+08	5.76x10^{+03}	1.78x10^{+04}	2.66x10^{-01}
	Cáceres	5.70 x10^{+00}E+06	3.00x10^{+02}	1.90x10^{+04}	2.49x10^{-01}
Andalucia	Almería	3.00 x10^{+00}E+05	4.90x10^{+01}	6.12x10^{+03}	5.57x10^{-01}
	Cádiz	6.00 x10^{+00}E+05	1.45x10^{+02}	4.14x10^{+03}	8.25x10^{-01}
	Córdoba	8.10 x10^{+00}E+06	6.96x10^{+02}	1.16x10^{+04}	2.93x10^{-01}
	Granada	1.44 x10^{+00}E+07	1.67x10^{+03}	8.63x10^{+03}	3.95x10^{-01}
	Huelva	2.09 x10^{+00}E+07	1.98x10^{+03}	1.06x10^{+04}	3.23x10^{-01}
	Jaén	3.00 x10^{+00}E+05	6.60x10^{+01}	4.55x10^{+03}	7.51x10^{-01}
	Málaga	1.30 x10^{+00}E+06	1.57x10^{+02}	8.28x10^{+03}	4.12x10^{-01}
	Sevilla	8.75 x10^{+00}E+07	7.41x10^{+03}	1.18x10^{+04}	2.89x10^{-01}

		CORN (c)			
Autonomous community	**Province**	**Production (kg)**	**Surface (ha)**	**Yield (kg·ha^{-1})**	**Blue water (m^3·kg^{-1})**
Galicia	A Coruña	6.56x10^{+07}	8.85x10^{+03}	7.42x10^{+03}	1.39x10^{-01}
	Lugo	1.69x10^{+07}	1.71x10^{+03}	9.87x10^{+03}	1.05x10^{-01}
	Ourense	1.72x10^{+07}	2.10x10^{+03}	8.20x10^{+03}	1.26x10^{-01}
	Pontevedra	3.27x10^{+07}	6.05x10^{+03}	5.41x10^{+03}	1.91x10^{-01}
Asturias	Asturias	8.00x10^{+05}	3.00x10^{+02}	2.67x10^{+03}	1.21x10^{-01}
Cantabria	Cantabria	2.00x10^{+05}	9.20x10^{+01}	2.17x10^{+03}	2.02x10^{-01}
Basque country	Álava	0.00x10^{+00}	1.30x10^{+01}	0.00x10^{+00}	0.00x10^{+00}
	Gipuzkoa	1.00x10^{+06}	3.00x10^{+02}	3.33x10^{+03}	1.83x10^{-01}
	Vizcaya	4.00x10^{+05}	1.40x10^{+02}	2.86x10^{+03}	2.14x10^{-01}
Navarra	Navarra	1.43x10^{+08}	1.34x10^{+04}	1.06x10^{+04}	2.99x10^{-01}
La Rioja	La Rioja	6.20x10^{+06}	7.47x10^{+02}	8.30x10^{+03}	2.90x10^{-01}
Aragon	Huesca	4.14x10^{+08}	3.81x10^{+04}	1.08x10^{+04}	3.36x10^{-01}
	Teruel	3.59x10^{+07}	3.86x10^{+03}	9.30x10^{+03}	3.92x10^{-01}
	Zaragoza	2.37x10^{+08}	2.41x10^{+04}	9.83x10^{+03}	3.71x10^{-01}
Catalonia	Barcelona	6.20x10^{+06}	1.64x10^{+03}	3.77x10^{+03}	6.92x10^{-01}
	Girona	8.78x10^{+07}	7.95x10^{+03}	1.10x10^{+04}	2.36x10^{-01}
	Lleida	2.43x10^{+08}	2.85x10^{+04}	8.50x10^{+03}	3.07x10^{-01}
	Tarragona	3.00x10^{+05}	4.10x10^{+01}	7.32x10^{+03}	3.57x10^{-01}

Balearic Islands		$2.90x10^{+06}$	$5.25x10^{+02}$	$5.52x10^{+03}$	$3.70x10^{-01}$
Castilla and Leon	Ávila	$1.40x10^{+06}$	$1.76x10^{+02}$	$7.95x10^{+03}$	$4.18x10^{-01}$
	Burgos	$3.20x10^{+06}$	$3.15x10^{+02}$	$1.02x10^{+04}$	$3.27x10^{-01}$
	León	$5.43x10^{+08}$	$5.72x10^{+04}$	$9.50x10^{+03}$	$3.50x10^{-01}$
	Palencia	$2.54x10^{+07}$	$2.82x10^{+03}$	9.0010^{+03}	$3.70x10^{-01}$
	Salamanca	$1.50x10^{+08}$	$1.32x10^{+04}$	$1.13x10^{+04}$	$2.94x10^{-01}$
	Segovia	$1.60x10^{+06}$	$2.00x10^{+02}$	$8.00x10^{+03}$	$4.16x10^{-01}$
	Soria	$5.50x10^{+06}$	$5.06x10^{+02}$	$1.09x10^{+04}$	$3.06x10^{-01}$
	Valladolid	$1.04x10^{+08}$	$1.03x10^{+04}$	$1.00x10^{+04}$	$3.32x10^{-01}$
	Zamora	$1.76x10^{+08}$	$1.85x10^{+04}$	$9.48x10^{+03}$	$3.51x10^{-01}$
Madrid	Madrid	$6.71x10^{+07}$	$5.8x10^{+03}$	$1.15x10^{+04}$	$4.37x10^{-01}$
Castilla La Mancha	Albacete	$1.50x10^{+08}$	$1.15x10^{+04}$	$1.30x10^{+04}$	$4.97x10^{-01}$
	Ciudad Real	$5.27x10^{+07}$	$4.43x10^{+03}$	$1.19x10^{+04}$	$5.43x10^{-01}$
	Cuenca	$9.90x10^{+06}$	$9.35x10^{+02}$	$1.06x10^{+04}$	$6.11x10^{-01}$
	Guadalajara	$4.59x10^{+07}$	$3.83x10^{+03}$	$1.20x10^{+04}$	$5.39x10^{-01}$
	Toledo	$1.58x10^{+08}$	$1.14x10^{+04}$	$1.39x10^{+04}$	$4.67x10^{-01}$
Valencia	Alicante	$1.20x10^{+06}$	$1.67x10^{+02}$	$7.19x10^{+04}$	$5.77x10^{-01}$
	Castellón de la Plana	$1.00x10^{+05}$	$1.60x10^{+01}$	$6.25x10^{+04}$	$6.64x10^{-01}$
	Valencia	$5.80x10^{+06}$	$5.75x10^{+02}$	$1.01x10^{+04}$	$4.11x10^{-01}$
Murcia	Murcia	$1.10x10^{+06}$	$1.12x10^{+02}$	$9.82x10^{+04}$	$5.17x10^{-01}$
Extremadura	Badajoz	$2.90x10^{+08}$	$2.75x10^{+04}$	$1.05x10^{+04}$	$5.36x10^{-01}$
	Cáceres	$1.60x10^{+08}$	$1.55x10^{+04}$	$1.03x10^{+04}$	$5.46x10^{-01}$
Andalucia	Almería	$1.00x10^{+05}$	$7.30x10^{+01}$	$1.37x10^{+03}$	$4.39x10^{+00}$
	Cádiz	$3.27x10^{+07}$	$3.52x10^{+03}$	$9.28x10^{+03}$	$6.48x10^{-01}$
	Córdoba	$5.32x10^{+07}$	$4.31x10^{+03}$	$1.23x10^{+04}$	$4.87x10^{-01}$
	Granada	$2.16x10^{+07}$	$2.33x10^{+03}$	$9.27x10^{+03}$	$6.49x10^{-01}$
	Huelva	$2.00x10^{+06}$	$2.04x10^{+02}$	$9.80x10^{+03}$	$6.13x10^{-01}$
	Jaén	$1.04x10^{+07}$	$9.70x10^{+02}$	$1.07x10^{+04}$	$5.61x10^{-01}$
	Málaga	$3.70x10^{+06}$	$5.16x10^{+02}$	$7.17x10^{+03}$	$8.39x10^{-01}$
	Sevilla	$1.30x10^{+08}$	$1.27x10^{+04}$	$1.03x10^{+04}$	$5.86x10^{-01}$

Table A3.3. Impacts at endpoint level of Natural Environment (Ecosystem Quality).

Map ID	Watershed	Characterization factor	Grape	Nectarine	Corn
				m^2 yr^1 m^{-3}	
1	Galicia-Costa	$1.15x10^{-01}$	$1.97x10^{-02}$	$1.33x10^{-02}$	$5.44x10^{-02}$
2	Miño-Sil	$1.07x10^{-01}$	$1.83x10^{-02}$	$1.23x10^{-02}$	$5.06x10^{-02}$
3	Cantabric	$1.21x10^{-01}$	$2.07x10^{-02}$	$1.39x10^{-02}$	$5.71x10^{-02}$
4	Internal watersheds of the Basque countries	$1.59x10^{-01}$	$2.72x10^{-02}$	$1.83x10^{-02}$	$7.52x10^{-02}$
5	Duero	$2.71x10^{-01}$	$4.63x10^{-02}$	$3.11x10^{-02}$	$1.28x10^{-01}$
6	Ebro	$2.70x10^{-01}$	$4.62x10^{-02}$	$3.11x10^{-02}$	$1.28x10^{-01}$
7	Internal watersheds of catalonia	$3.14x10^{-01}$	$5.37x10^{-02}$	$3.61x10^{-02}$	$1.48x10^{-01}$
8	Tajo	$3.48x10^{-01}$	$5.96x10^{-02}$	$4.01x10^{-02}$	$1.65x10^{-01}$
9	Júcar	$5.45x10^{-01}$	$9.32x10^{-02}$	$6.27x10^{-02}$	$2.58x10^{-01}$
10	Balearic Islands	$2.76x10^{-01}$	$4.72x10^{-02}$	$3.18x10^{-02}$	$1.31x10^{-01}$
11	Guadiana	$6.38x10^{-01}$	$1.09x10^{-02}$	$7.35x10^{-02}$	$3.02x10^{-01}$
12	Tinto, Odiel and Piedras	$2.40x10^{-01}$	$4.10x10^{-02}$	$2.76x10^{-02}$	$1.13x10^{-01}$
13	Guadalquivir	$4.95x10^{-01}$	$8.47x10^{-02}$	$5.70x10^{-02}$	$2.34x10^{-01}$
14	Segura	$5.74x10^{-01}$	$9.82x10^{-02}$	$6.61x10^{-02}$	$2.71x10^{-01}$
15	Guadalete and Barbate	$1.12x10^{-01}$	$1.91x10^{-02}$	$1.29x10^{-02}$	$5.29x10^{-02}$
16	Mediterranean Watersheds of Andalucia	$3.98x10^{-01}$	$6.80x10^{-02}$	$4.58x10^{-02}$	$1.88x10^{-01}$

Chapter 4

Life-Cycle Assessment of fuel ethanol from sugarcane in Argentina

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Abstract

Purpose: The production of bioethanol in Argentina is based on sugarcane plantation, with extensive use of agricultural land, scarce use of fertilizers, pesticides and artificial irrigation, and burning of sugarcane prior to harvesting. The objective of this chapter is to develop the Life Cycle Assessment (LCA) of the fuel ethanol from sugarcane in Tucumán (Argentina), assessing the environmental impact potentials to identify which of them cause the main impacts.

Methods: Our approach innovatively combined knowledge about the main impact pathways of bioethanol production with LCA which covers the typical emission-related impact categories at the midpoint LCIA. Real data from the Argentinean industry subsystems have been used to perform the study: S1-sugarcane production, S2-milling process, S3-sugar production and S4-ethanol production from molasses, honey or sugarcane juice.

Results: The results are shown in the three alternative pathways to produce bioethanol. Different impact categories are assessed, being Global Warming Potential (GWP) which has had the highest impact. So, the production of 1kg of ethanol from molasses emitted 22.5 kg CO_2 (pathway 1), 19.2 kg CO_2 from honey (pathway 2) and 15.0 kg CO_2 from sugarcane juice (pathway 3). Several sensitivity analyses to study the variability of the GWP according to different cases studies have been performed (changing the agricultural yield, including economic and calorific allocation in sugar production and modifying the sugar price).

Conclusions: Agriculture is the subsystem which shows the highest impact in almost all the categories due to the fossil fuel consumption. When an economic and calorific allocation is considered to assess the environmental impact, the value is lower than when mass allocation is used because ethanol is relatively cheaper than sugars and it has higher calorific value.

Keywords: Sugarcane, bioethanol, Life Cycle Assessment, Argentina.

4.1. Introduction

The concern about 'sustainability' (Brundtland 1987) has increased in the last decades. Nowadays, there is a rising awareness about the future reduction of fossil energy resources, such as those coming from oil. Thus, renewable fuels have gained wider interest in the recent past, bioethanol being one of the most successful examples of a shift from fossil fuels to bio-based fuels.

The use of ethanol in vehicles was first proposed by H. Ford (1896). After the oil crisis, ethanol became more popular and oil-importing countries were forced to develop alternative fuel programs to reduce their dependence on oil. Over the last decades, vast investments, government sponsorship, and tax incentives made Brazil and United States the world leaders in ethanol production. Nowadays, many countries have launched programs to replace gasoline by ethanol in the midterm (Olsson 2007). Following this trend, Argentina published law #26.093 in 2006, which provides the framework for investment, production, and marketing of biofuels, which focused primarily on conventional biofuels: biodiesel from soybean oil and bioethanol from sugarcane. This framework, which became active in 2010, establishes 5 % as the minimum bioethanol content in gasoline. The main goal is to reduce emissions of carbon dioxide and other greenhouse gases (GHG), to diversify the supply of energy and to promote the development of rural areas, especially in benefit of small- and medium-sized agricultural producers.

So far, the Argentinean ethanol industry is based exclusively on sugarcane, and its primary focus is the domestic market. The sugar industry welcomed the alternative to direct sugarcane surpluses to the profitable local ethanol market rather than exporting sugar. Moreover, Argentina has abundant natural resources, a very efficient agricultural production sector, good processing and export infrastructures and a suitable human capital (Joseph 2010).

The fuel ethanol production model is based mostly on a sugarcane plantation system with extensive use of agricultural land, scarce use of fertilizers, pesticides and artificial irrigation, and burning of sugarcane prior to harvesting. A schematic of the production process from sugarcane to ethanol can be seen in Figure 4.1. Ethanol in Argentina is currently produced by 15 sugar mills located in the northwest of the country that use sugar molasses as main feedstock. Almost all sugar mills produce electricity from bagasse for their own use, but there are four sugar mills which co-generate electricity and sell the excess to the electrical grid. Argentina needs to expand its sugarcane industry in order to meet the official requirements. Production in 2011 is projected at 280,000 cubic meters of bioethanol (Joseph 2010).

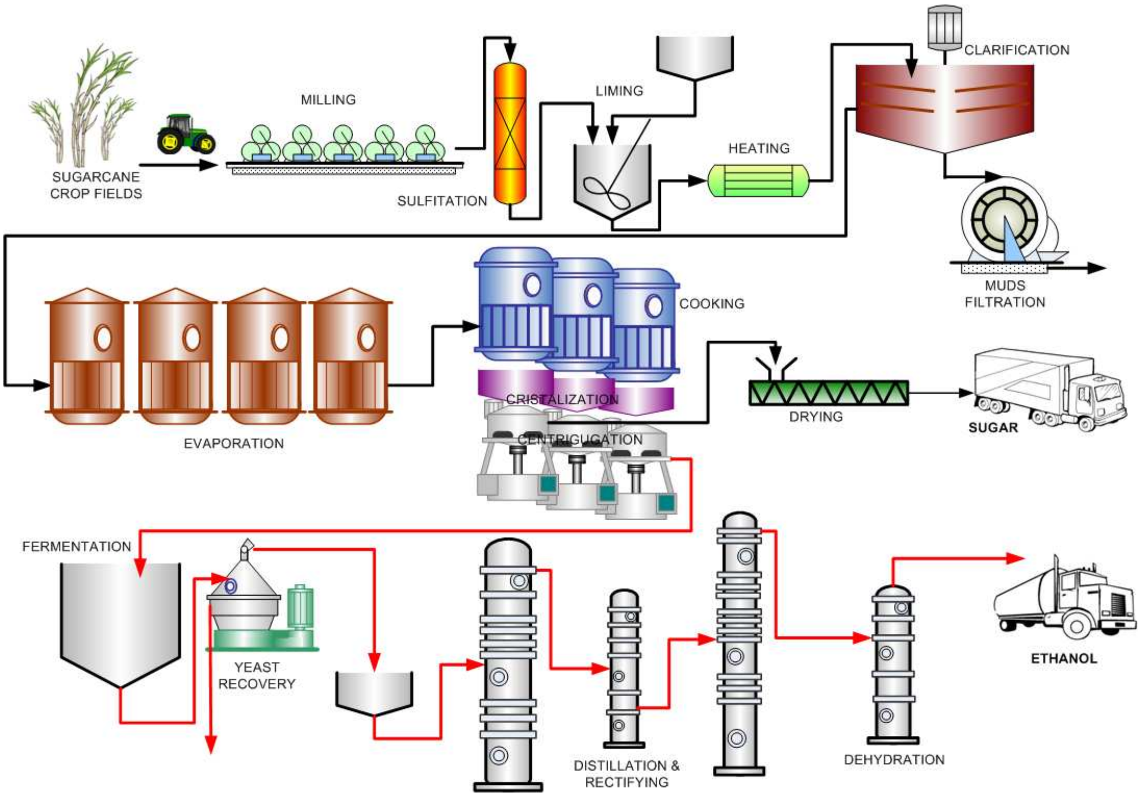


Figure 4.1 Productive system used in Argentina to produce ethanol from sugarcane.

Argentinean bioethanol producers will need to study the environmental performance of their product in order to comply with sustainability criteria (Farrell et al. 2006). Due to the potential of Argentina to produce bioethanol, this work constitutes a baseline to analyze the environmental impact and country-specific pathways by estimating the GHG emissions, and other impact potentials. The sugarcane will play an important role given its resistance, rapid growth, and uptake capacity for atmospheric carbon. However, Argentina as a sugarcane-based biofuel producer is likely to be not competitive from the environmental point of view unless specific measures are implemented. These measures include avoiding deforestation, applying reduced tillage and crops successions, using low ecotoxicity pesticides and improving the sustainable methods to treat the huge volumes of vinasses generated. Moreover, there are other arising drawbacks such as the land competition with food and the environmental impact associated with the transport sector. In addition to this, the rapid expansion of ethanol production/consumption has affected the international market of sugar, an important co-product of ethanol. Such a complex environment poses significant challenges for practitioners and researchers.

In particular, one of the key issues that have not been tackled so far and it is very timely to perform is the environmental impact assessment of the bioethanol production in Argentina from a LCA perspective, such as it has been done in the cases of the Argentinean biodiesel production (Asal et al. 2006; Panichelli et al. 2009; Tomei and Upham 2009). The LCA approach will shed light on some of these problems, but not on all of them. For example it will be able to show improvement opportunities on greenhouse gases, acidification and eutrophication emissions, however land and water use and issues related to indirect land use changes and competition with food products do not fit well into the LCA framework. These last categories require a broader approach (Luo et al. 2009). The LCA studies conducted in Argentina on the bioethanol production have been incipient as can be confirmed from the review article by Chauhan et al. (2011). With regards to the LCA applied to the sugarcane-based ethanol, some relevant contributions appear in recent literature referred to production in other countries, for instance Australia (Renouf et al. 2011), Brazil (Cavalett et al. 2012; Luo et al. 2009; Pereira and Ortega 2010; Ometto et al. 2009; Seabra et al. 2011), Mexico (García et al. 2011) and Thailand (Nguyen et al. 2008). None of these studies are comparable enough to the case of Argentina as they analyze specific geographic situations and practices, however, some general trends can be derived from the results of these studies, which will be mentioned in the conclusions section.

There are a number of reasons to assess the Argentinean sugarcane based ethanol from a LCA perspective: (i) Fuel ethanol made from sugarcane is based on renewable resources in contrast to other types of fuel; (ii) The sugarcane leads to one of the main economic activities in the rural areas of the

Argentinean Northwest, with many environmental and social implications; (iii) A country-specific approach to LCA is crucial when evaluating the environmental impacts of bioenergy systems: local conditions, such as agricultural practices, land use changes and transport infrastructures, will have a major impact on the environmental performance of the system (Panichelli et al. 2009); (iv) Many steps in the lifecycle of Argentinean fuel ethanol have remained unchanged for a long time so the improvement potential is substantial; it is possible to co-generate electricity from the solid waste, and to recycle some of the waste streams; (v) Argentinean ethanol producers will need to evaluate the environmental performances of their product in order to comply with sustainability criteria being developed.

Therefore, the goal of this chapter is to present the LCA of the fuel ethanol from sugarcane in Tucumán (Argentina), assessing the environmental impact potentials, in order to identify the activities of the lifecycle, which cause the main impacts. Also, some guidelines for environmental improvements will be indicated. Different scenarios will be analyzed (the current situation and two hypothetical scenarios) and a sensitivity analysis of some parameters will be performed.

4.2 Methods

4.2.1 Overview

The methodological structure for this LCA study was built according to the International Standardization Organization (ISO) series 14040 and 14044 on LCA (ISO 2006 a, b). We perform a 'cradle to gate' analysis that embraces all the activities of the network, starting from the extraction of raw materials (agricultural stages) and ending with the products at the 'gate' of the manufacturing plants (sugar mills and distilleries). The system description and inventory data are valid for sugarcane-based bioethanol in Argentinean

Northwest (country-specific approach), and for a time framework 2000–2005, as technologies, prices and production methods are assumed to change in the midterm.

The Life Cycle Inventory (LCI) data for our problem were obtained from different sources. With regard to the agricultural stages, we collected data from local agricultural companies and governmental organizations (Pérez et al. 2007). For the industrial stages, we considered standard mass and energy balance coefficients taken from typical sugar mills and distilleries. Data gaps have been covered using specialized literature, handbooks, and databases; all cited when showing the LCI results. These data have been partially used in other works of the authors (Mele et al. 2011; Kostin et al. 2011a,b) in which the objective is rather to optimize the ethanol supply chain than perform a detailed LCA study.

The Life Cycle Impact Assessment (LCIA) was performed on Ecoinvent® 2.2 database (Swiss Centre for Life-Cycle Inventories 2010). The LCIA of this study has been done using the CML2001 (Centre for Environmental Studies 2001) the most widely applied midpoint LCIA method, covering the following emission-related impact categories: Acidification Potential (AP), Global Warming Potential (GWP), Eutrophication Potential (EP), Photochemical Oxidation (PHO), Depletion of Abiotic Resources (DAR) and Ozone Depletion Potential (ODP), Freshwater Aquatic Ecotoxicity (FWAET) and Terrestrial Ecotoxicity (TET) all available in the Ecoinvent® 2.2 database.

The functional unit (FU) of this study is 1 kg of fuel anhydrous ethanol produced. The results are calculated assuming the average sugarcane productivity of 67 tonne sugarcane·ha^{-1} in one year crop cultivation. This value is based on the average yield from 2000 to 2005 harvesting periods in the province of Tucumán (Pérez et al. 2007).

The system boundaries have been expanded where possible including the impact associated with the production of raw materials (e.g., fertilizers, lime, sulfuric acid, etc.). In line with common LCA practice, we have not considered the impact associated with the production of capital equipment. Moreover, as the study focuses on the production stages, the system boundaries do not include storage and transportation tasks after production.

Although the ISO norm recommends subdividing the system or performing system expansion so as to avoid allocation, some allocation has been used when it was strictly necessary. Sensitivity analysis was performed for allocation based on mass, price and calorific value to evaluate the variability of results with regard to this key methodological parameter. The allocation method based on price has been considered in accordance with the Ecoinvent database for bioenergy products (Jungbluth et al. 2007) where allocation of environmental impacts between co-products is based on the respective prices of co-products. Different authors of comparable studies also conducted mass allocation by mass (Luo et al. 2009; Ometto et al. 2009; Seabra et al. 2011), energy content (García et al. 2009) and price (Cavalett et al. 2012; García et al. 2011; Nguyen et al. 2008).

Emissions from land use other than direct deforestation are excluded due to lack of data. Agricultural and industrial data are, as much as possible, specific to the Argentinean context. However, since the production of bioethanol in Argentina so far remains a secondary activity, data are not fully available nor gathered into a unified database. Therefore, some data are based on average technologies and they are opportunely referenced.

4.2.2 System description and inventory data

The system under study is the Argentinean standard fuel anhydrous ethanol industry (Figure 4.1). The overall system has been divided into four subsystems: (S1) Agriculture, (S2) Milling, (S3) Sugar Production, and (S4) Ethanol Production, which are combined to give the production pathways considered in this study.

The reference case studied in this paper includes three main pathways according to the technology used. These pathways have in common that they include the same type of subsystems (Agriculture and Milling), but they differ in the technologies used to produce sugar and ethanol, as shown in Figure 4.2.

- Pathway 1 (P1) produces ethanol from molasses. The sugarcane is milled. Then, the cane juice is processed with technology T1, and finally the resulting molasses are converted into ethanol in a distillery type T3.
- Pathway 2 (P2) produces ethanol from honey obtained in a manufacturing process that uses technology T2. Honey is converted into ethanol in a distillery using technology T4.
- Pathway 3 (P3) produces ethanol directly from sugarcane juice coming from sugarcane milling. Ethanol is obtained through technology T5. This pathway does not produce sugar.

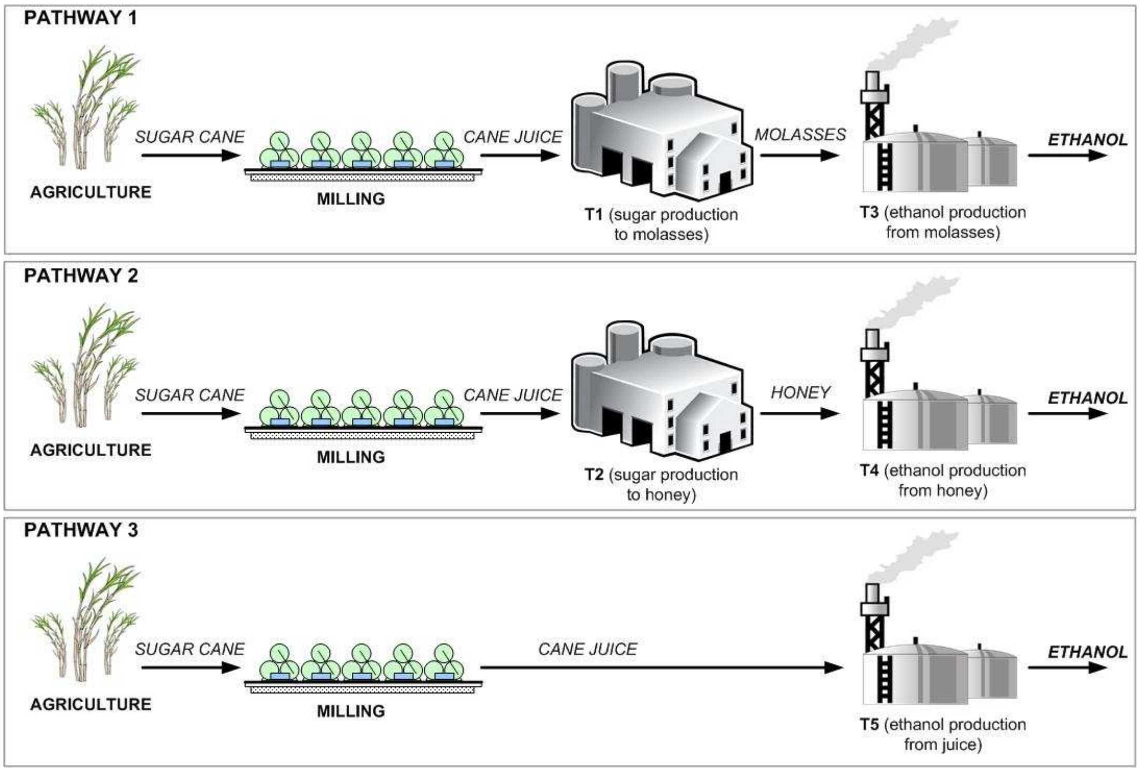


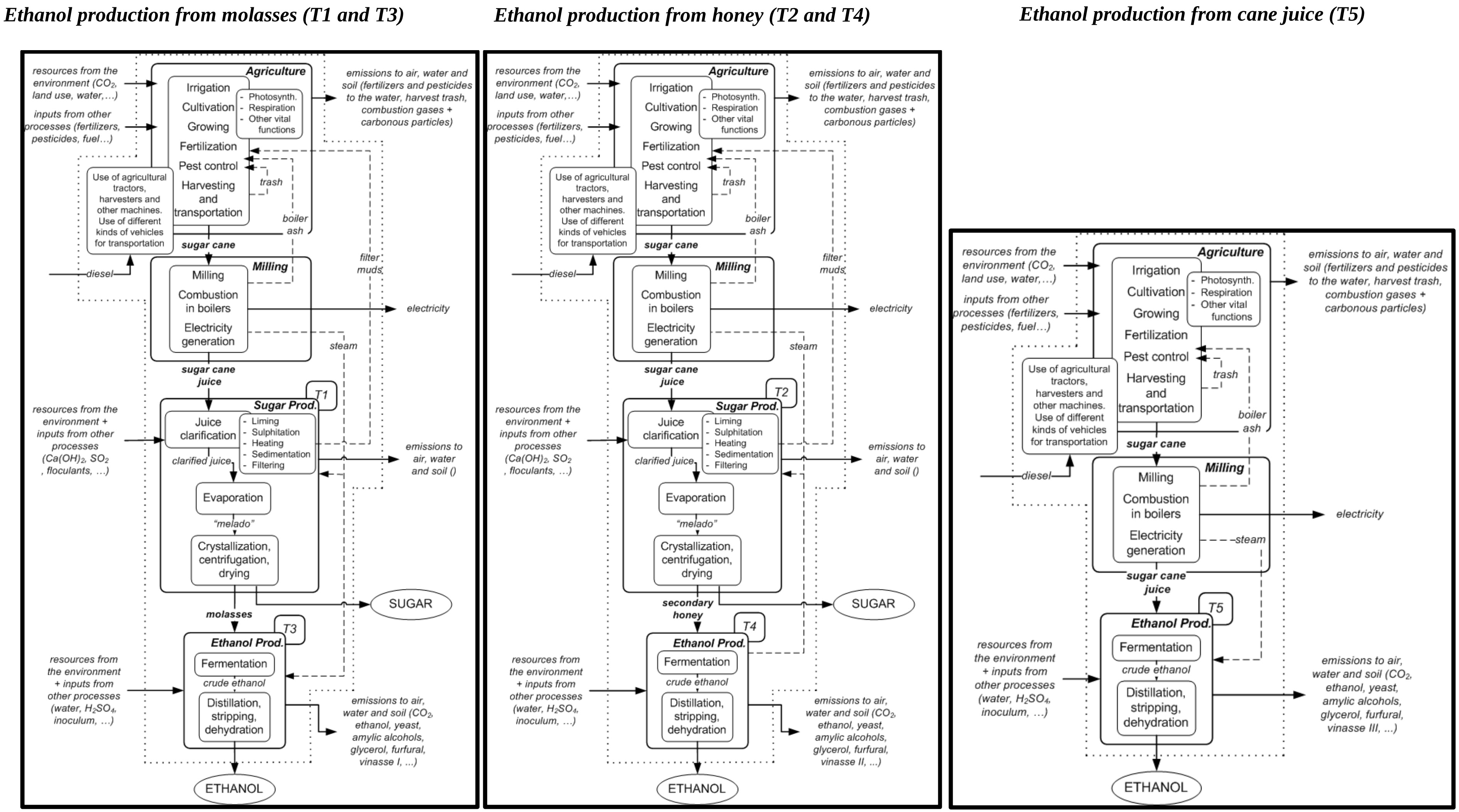
Figure 4.2 Schematic of the three pathways considered in the reference case.

It is worth mentioning that these pathways are standard in the world-wide sugarcane industry. Countries apply one of them according to their preferences. In Brazil, P3 –autonomous distillery- is very common (Cavalett et al. 2012; Seabra et al. 2011) whereas in Argentina this pathway is practically inexistent. In Mexico, García et al. (2011) considered the three options in the study, while in Thailand, Nguyen et al. (2008) claim that ethanol from molasses (P1) is the most common pathway. Apart from that, there is a big variability in the way the countries implement these pathways which is mostly related to the way in which each country drives the agriculture practices and supplies energy to the process (e.g. bagasse, crop trash, natural gas, fuel oil, different mixtures, etc.).

Figure 4.3 (a,b,c) shows a detailed input-output diagram for the three pathways described above. The subsystem *Agriculture* involves all the activities related to sugarcane planting, growing and harvesting, as well as transportation to sugar mills. Sugarcane is regarded as the main product of these activities. The sugarcane production in the province of Tucumán (Argentina) is characterized by the partial use of synthetic fertilizers, pesticides, and semi-mechanized cultivation and harvesting. Although sugarcane is allowed to grow with the same stalk five times after cut, the annual renewal percentage, as well as the involved activities, has not been taken into account in this study. We have taken the worst case of planting for the first time, as described by Caro et al. (2009). The equipments used for the manual plantation are trucks to transport ratoons and tractors to open trenches on the field.

Regarding to agrochemicals applications, tractors are used for pesticide and fertilizer applications in the total sugarcane area. For the fertilizer and pesticide application emissions, the consideration that the soil is part of the technosphere was used, in which only emissions that come out of the production system ground level and that interfere with the air or water quality were assessed. In Argentina, most of the pesticides and fertilizers are imported. The production processes for these inputs have been included into the system boundaries. Sugarcane nitrogen fertilizer is applied as urea (200 $kg \cdot ha^{-1}$), and phosphorus fertilizer is applied as triple super phosphate in low doses of about 24 $kg \cdot ha^{-1}$. Other fertilizers have been not included due to lack of data and low use. No K fertilizer is applied in sugarcane production in Tucumán. Among the pesticides, the most common are insecticides like permethrin and herbicides like glyphosate. Due to the lack of regional information, nitrogen oxides and ammonia emissions to air, nitrate and phosphorous emissions to groundwater and phosphorous emissions to surface water all they are estimated from Renouf et al. (2008). The ash from bagasse combustion and the filter cake from juice clarification (see subsystems Milling and Sugar Production in Figure 4.3) are transported to the fields by truck and disposed in the soil to substitute some of the synthetic fertilizers used.

In Tucumán, the artificial irrigation is not significant, while the use of vinasses, a residue of the ethanol distillation, for irrigation is not a generalized practice. Harvesting is carried out every year in the total area from May to November (*zafra*). Sugarcane harvesting in the province of Tucumán is carried out mainly with machines. Although it is currently penalized by law, burning before harvesting to facilitate cutting, has been considered to be used on 50% of the total area. After being cut, the sugarcane is transported with trucks to the industrial process. In the agricultural stages, transports distances are based on an average distance of 50 km. Diesel consumption in agricultural processes was converted into inputs of the subsystem Agriculture according to Ecoinvent® (Nemecek et al. 2007). Diesel consumption for agricultural activities: plantation, cultivation, harvesting and transport have been calculated based on data from Caro et al. (2009). Biogenic CO_2 uptake (0.819 $kg \cdot kg^{-1}$ sugarcane) is estimated from the carbon balance. Table 4.1 shows the inventory values for the subsystem Agriculture. All the entries for this table are referred to 1 kg of sugarcane as a reference flow.



Ethanol production from molasses (T1 and T3)

Ethanol production from honey (T2 and T4)

Ethanol production from cane juice (T5)

Figure 4.3 System diagrams for ethanol production from different production pathways.

Table 4.1 Summary of the inventory data for subsystem Agriculture yearly (reference flow: 1 kg sugarcane harvested).

Inputs	Value	Unit	Comments
Land use (Perez et al. 2007)	1.49×10^{-05}	ha	
Urea (Universidad Nacional de Tucumán 2010)	2.99×10^{-03}	kg	Urea, as N, at regional storehouse.
Superphosphate (Universidad Nacional de Tucumán 2010)	3.58×10^{-04}	kg	Triple superphosphate, as P_2O_5, at regional storehouse.
Pesticides (Universidad Nacional de Tucumán 2010)	2.69×10^{-04}	kg	Nitro-compounds, at regional storehouse
Filter muds (Universidad Nacional de Tucumán 2010)	4.00×10^{-02}	kg	3.5 kg filter mud/100 kg cane. It contains colloidal organic matter, 15-30% of fiber, 5-15% proteins, 5-15% crude wax and fats, 10-20% ash including oxides of Si, Ca, Mg and K
Fuel oil (Ecoinvent Database 2007).	1.78×10^{-03}	kg	Diesel, at regional storage
Outputs			
Ash (Universidad Nacional de Tucumán 2010)	5.67×10^{-03}	kg	Avoided credit: Disposal, wood ash mixture, pure, 0% water, to land farming
Emissions air [a] (Renouf et al. 2008)	1.00×10^{-03}	kg	7.76×10^{-02}kg NH_3, -6.27×10^{-02}kg CO_2, $2.17 \times 10^{+01}$ kg CO, 2.00×10^{-01}kg N_2O, $1.02 \times 10^{+00}$ kg NO_x, 6.20×10^{-02} kg SO_x, $1.50 \times 10^{+01}$ kg particulates >10 um
Emissions water[a] (Renouf et al. 2008)	1.00×10^{-03}	kg	1.94×10^{-01} kg NO_3^- 3.82×10^{-01} kg PO_4^{3-}
Sugarcane harvested	**1**	**kg**	

[a] *Obtained through carbon balance calculations (photosynthesis + respiration + combustion).*

The subsystem *Milling* involves washing of the harvested sugarcane, transport by conveyor belts to the mills, milling to extract sugarcane juice and burning of the lignocellulosic residue of the sugarcane (bagasse) in boilers to generate steam and electricity in cogeneration plants. The electricity production satisfies all the sugar mill requirements. The excess of energy is exported to the public network. Sugarcane juice is the main product of this subsystem. Exported electricity has been regarded as a co-product whose allocation has been solved by expanding the system boundaries, and retrieving the necessary data from the Ecoinvent® Database. In that case, we have used the electric mix from Argentina according to the relative contribution of 45% natural gas, 11% Oil, 33% hydropower, 6% nuclear, 1% hard coal, 1% biomass and 3% imported energy from Brazil. Table 4.2 shows the inventory values for the subsystem Milling.

Table 4.2 Summary of the inventory data for subsystem Milling yearly (reference flow: 1 kg sugarcane juice).

Inputs	Value	Unit	Comments
Sugarcane harvested	**$1.11\text{x}10^{+00}$**	**kg**	
Water	$4.54\text{x}10^{-01}$	kg	Tap water, at user
Outputs			
Avoided electricity 132.1 MJ (Universidad Nacional de Tucumán 2010)	$4.09\text{x}10^{-02}$	kWh	The subsystem produces 132.1 MJ for export to the network. Electricity mix in Argentina[a]
Avoided electricity 54.0 MJ (Universidad Nacional de Tucumán 2010)	$1.67\text{x}10^{-02}$	kWh	The subsystem produces 54 MJ used in the process operation. Electricity mix in Argentina[a]
Avoided steam 0.504 t (Universidad Nacional de Tucumán 2010)	$5.61\text{x}10^{-01}$	kg	The subsystem produces 0.504 t used in the process operation
Avoided bagasse 0.02 t (Universidad Nacional de Tucumán 2010)	$2.23\text{x}10^{-02}$	kg	The subsystem produces 0.02 t used in the process operation. Bagasse, from sugarcane, at sugar refinery
Emissions air (U.S. Environmental Protection Agency 1996)	$1.11\text{x}10^{-03}$	kg	$2.58\text{x}10^{+02}$ kg CO_2, $2.00\text{x}10^{-01}$ kg NO_x, 1.00 $\text{x}10^{-01}$ kg SO_2, $2.00\text{x}10^{-01}$ kg particulates >10 um
Emissions water (U.S. Environmental Protection Agency 1996)	$1.11\text{x}10^{-03}$	kg	$1.90\text{x}10^{+00}$ kg dissolved solids
Sugarcane juice	**1**	**kg**	

[a] *Secretaría de Energía de la Nación: Informe del Sector Eléctrico 2009 (Part 1).*

The subsystem *Sugar Production* includes the purification and concentration of sugarcane juice to obtain dry sugar crystals. First, the juice is acidified with SO_2 and neutralized with lime slurry to precipitate the impurities. Then, it is heated to diminish juice viscosity and the solubility of the calcium salts generated. Finally, the juice enters a series of decanters where solid materials are separated from the juice. The solid residue of the clarification is filtered to recover some juice producing a filter mud (*cachaza*), which is transported to the crop fields as a fertilizer. Clarified juice is concentrated in multi-effect continuous evaporators followed by discontinuous multi-stage crystallization (cooking). It has been considered only one process to carry out the juice purification (sulphitation, liming, heating, sedimentation and filtering) although there are two ways to concentrate the clarified juice to produce sugar. The first technology yields white sugar, raw sugar and molasses (technology T1), while the second one produces white sugar, raw sugar and a secondary honey (technology T2). These two byproducts differ in their sucrose content. Molasses is a viscous dark honey whose low sucrose content cannot be recovered by crystallization, while secondary honey is a liquid with a higher amount of sucrose that leaves the sugar mill before being exhausted by crystallization. White sugar has been taken as the main product for both technologies in this subsystem. Allocation between sugars and molasses in T1 and sugars and honey in T2 has been done in a mass basis. Table 4.3 shows results for the inventory phase of the subsystem Sugar Production.

Table 4.3 Summary of the inventory data for sugar production to molasses (T1) and honey (T2) (reference flow: 1 kg molasses or honey, respectively).

Inputs	Value T1	Value T2	Unit	Comments
Sugarcane juice	**$4.13\ x10^{+00}$**	**$4.13\ x10^{+00}$**	**kg**	
Steam(Universidad Nacional de Tucumán 2010)	$2.04x10^{+00}$	$1.71x10^{+00}$	kg	Steam, for chemical processes, at plant
Lime (Universidad Nacional de Tucumán 2010)	$9.32x10^{-03}$	$9.20x10^{-03}$	kg	Lime, algae, at regional storehouse
Sulphur (Universidad Nacional de Tucumán 2010)	$1.40x10^{-03}$	$1.38x10^{-03}$	kg	Sulphuric acid, liquid, at plant
Oxygen (Universidad Nacional de Tucumán 2010)	$2.52x10^{-03}$	$2.48x10^{-03}$	kg	Oxygen, liquid, at plant
Flocculant (Renouf et al. 2008)	$5.59x10^{-07}$	$5.52x10^{-07}$	kg	Iron (III) chloride, 40% in H_2O, at plant
NaOH (Universidad Nacional de Tucumán 2010)	$7.82x10^{-04}$	$7.36x10^{-04}$	kg	Sodium hydroxide, 50% in H_2O, production mix, at plant
HCl (Universidad Nacional de Tucumán 2010)	$1.51x10^{-03}$	$1.47x10^{-03}$	kg	Hydrochloric acid, 30% in H_2O, at plant
Outputs				
Emissions air (Renouf et al. 2008)	$5.59x10^{-04}$	$4.60x10^{-04}$	kg	T1: $5.00x10^{-01}$kg SO_2 T2: $6.00x10^{-01}$ kg SO_2
Emissions water (Renouf et al. 2008)	$5.59x10^{-04}$	$4.60x10^{-04}$	kg	T1: $2.10x10^{-03}$ kg BODs $6.96x10^{+01}$ kg dissolved solids $3.20x10^{-03}$ kg suspended solids T2: $6.00x10^{-01}$kg SO_2 $2.60x10^{-03}$ kg BODs $8.35x10^{+01}$ kg dissolved solids $3.90x10^{-03}$ kg suspended solids
Filters muds (Hugot 1986)	$8.09x10^{-01}$	$5.33x10^{-01}$	kg	Avoided credit: 3.5% of filter mud (contains colloidal organic matter, 15-30% of fiber, 5-15% proteins, 5-15% crude wax and fats, 10-20% ash including oxides of Si, Ca, Mg and K) per weight of cane
Sugars (Universidad Nacional de Tucumán 2010)	$6.57x10^{-01}$	$5.43x10^{-01}$	kg	T1: 1000 kg white sugar and 176 kg raw sugar. T2: 1000 kg white sugar and 180 kg raw sugar.
Molasses (T1) or Honey (T2)	**1**	**1**	**kg**	

In the subsystem *Ethanol Production*, a yeast prepared substrate is inoculated (*Saccharomyces cerevisiae*), which converts saccharose ($C_{12}H_{22}O_{11}$) and other fermentable carbohydrates into ethanol and CO_2. The process takes place in open discontinuous fermentation tanks. According to the most common practice in Tucumán, the fermented product is transported to a train of three distillation columns to obtain the desired alcohol concentration. The products of the distillation are hydrated alcohol 97 °Gl, low grade ethanol and fusel oil. Three different technologies can be distinguished depending on the raw material arriving to the distillery: molasses (technology T3), honey (technology T4), or sugarcane juice (technology T5). All three technologies consume the same inputs (e.g., water, yeasts, etc.), but the consumption rates differ in each case. These technologies lead in turn to different amounts of nearly the same emissions (i.e., CO_2, VOCs, fusel oil, etc.). The hydrated ethanol, which is close to its azeotropic composition, is dehydrated through molecular sieves. Each of the three technologies generates a harmful liquid residue called vinasses in a ratio of 15 l of vinasse/l of ethanol produced. Vinasse properties depend on the raw material used in the process. Currently, each ethanol company in Argentina implements a different waste disposal option. In this study, we have considered an average impact for disposing vinasses in soil and surface watercourses. Table 4.4 shows results for the inventory phase of the subsystem Ethanol Production.

Table 4.4 Summary of the inventory data for ethanol production from; molasses distillery (T3), honey distillery (T4) and cane juice distillery (T5) yearly (reference flow: 1 kg ethanol).

Inputs	Value T3	Value T4	Value T5	Unit	Comments
Molasses (T3), Honey (T4), or Sugarcane juice (T5)	**4.00x10^{+00}**	**3.35x10^{+00}**	**1. 19x10^{+01}**	**kg**	
Steam	3.85x10^{+00}	2.97x10^{+00}	4.99x10^{+00}	kg	Steam, for chemical processes, at plant
Urea	4.26x10^{-03}	4.26x10^{-03}	4.26x10^{-03}	kg	Urea, as N, at regional store-house
H_3PO_4	3.55x10^{-04}	3.55x10^{-04}	3.55x10^{-04}	kg	Phosphoric acid plant, fertilizer grade
Water	1.63x10^{-03}	1.63x10^{-03}	1.63x10^{-03}	kg	Tap water, at user
H_2SO_4	1.15x10^{-02}	1.15x10^{-05}	1.15x10^{-05}	kg	Sulphuric acid, liquid, at plant
Outputs					
Emissions air (Cortez 1997)	8.88x10^{-04}	8.88x10^{-04}	8.88x10^{-04}	kg	T3: 3.37x10^{+03}kg CO_2 T4: 3.37x10^{+03} kg CO_2 T5: 3.37x10^{+03} kg CO_2
Emissions water (Cortez 1997)	8.88x10^{-04}	8.88x10^{-04}	8.88x10^{-04}	kg	T3: 1.45x10^{+03} kg COD, 5.28x10^{+02}kg DBO$_5$, 1.58x10^{+03}kg total solids, 5.65x10^{+02} kg inorganic solids, 2.97x10^{+01} kg Ca, 6.4 x10^{+00} kg Mg, 7.8x10^{+00} kg Na, 1.95x10^{+02} kg K, 2.19x10^{+01} kg N, 1.6x10^{+00} kg P T4: 1.45x10^{+03} kg COD, 5.28x10^{+02} kg DBO$_5$, 1.58x10^{+03} kg total solids, 5.65 x10^{+02}kg inorganic solids, 2.97 x10^{+01} kg Ca, 6.4 x10^{+00} kg Mg, 7.8x10^{+00} kg Na, 1.95x10^{+02} kg K, 2.19x10^{+01} kg N, 1.6x10^{+00} kg P T5: 1.45x10^{+03} kg COD, 5.28x10^{+02} kg DBO$_5$, 2.21x10^{+02} kg total solids, 4.35x10^{+02}kg inorganic solids, 1.69x10^{+01} kg Ca, 3.2x10^{+00} kg Mg, 2.7x10^{+00} kg Na, 6.82x10^{+01} kg K, 5.5 x10^{+00} kg N, 1.1x10^{+00} kg P
Ethanol LG	1.11x10^{-01}	1.11x10^{-01}	1.11x10^{-01}	kg	
Fusel oil	1.51x10^{-03}	1.51x10^{-03}	1.51x10^{-03}	kg	
Steam excess			1.69x10^{+00}	kg	Steam excess produced in T5 which is used as feedback.
Ethanol	**1.00**	**1.00**	**1.00**	**kg**	

Moreover, only technologies T1+T3 are applied in Argentina, but technologies T2+T4 and T5 are promising candidates due to the increasing demand of ethanol and the good performance of these technologies in other countries, e.g. Brazil. Previous works of the authors also consider these options (Mele et al. 2011).

Steam generation and use deserve a special digression. In the subsystem Milling, the bagasse obtained from sugarcane juice extraction is burnt in boilers to produce steam in a close loop to cover both power and heating needs of the sugar plants and distilleries. Moreover, electricity is produced by steam-driven generators. In this study, for pathways 1 and 2 we considered that all the steam generated is consumed by the sugar plant and distillery, and extra steam is considered to enter distilleries in order to cover its needs. However, in the case of the pathway 3, distillery T5 does not consume all the steam produced, and therefore there is a steam surplus that is considered as a co-product, since it can be used in other processes out of the system boundaries. In this case its environmental impacts have been avoided by including the steam production process within the system boundaries.

Regarding CO_2, the worst case has been taken, that is, the complete mass balance of CO_2 taking all the CO_2 inputs and outputs has been considered, using the impact factors corresponding to fossil CO_2.

4.3 Results and discussion

4.3.1 Life Cycle Impact Assessment (LCIA) of the reference case

For the reference case, the eight impact categories considered in the LCIA phase of the study are: AP (Acidification Potential, global, kg SO_2 equivalent), GWP (Global Warming Potential, kg CO_2 equivalent), EP (Eutrophication Potential, global, kg PO_4 equivalent), PHO (Photochemical Oxidation, kg ethylene equivalent), DAR (Depletion of Abiotic Resources, kg Sb equivalent), ODP (Ozone Depletion Potential, kg CFC-11 equivalent), FWAET (Freshwater Aquatic Ecotoxicity, kg 1,4-DCB equivalent) and TET (Terrestrial Ecotoxicity, kg 1,4-DCB equivalent). All the impact values for each subsystem (Agriculture, Milling, Sugar Production and Ethanol Production) are shown in Table 4.5. The impacts are referred to 1 kg of ethanol as the FU. From Table 4.5 it can be deduced that P1 has the highest impact, except for AP, followed by P2, and finally P3, which is the most environmentally convenient path.

Table 4.5 Impact categories in different subsystems to produce ethanol for each Pathway (FU = 1 kg of ethanol).

		Subsystem1	**Subsystem2**	**Subsystem 3: Sugar Production**		**Subsystem 4: Ethanol Production**			**Total**
	Pathways	**Agriculture**	**Milling**	**T1**	**T2**	**T3**	**T4**	**T5**	
Global Warming Potential, kg CO_2 eq.	P1	$1.33x10^{+01}$	$3.95x10^{+00}$	$1.36x10^{+00}$	-	$3.91x10^{+00}$	-	-	$2.25x10^{+01}$
	P2	$1.11x10^{+01}$	$3.31x10^{+00}$	-	$1.05x10^{+00}$	-	$3.70x10^{+00}$	-	$1.92x10^{+01}$
	P3	$9.54x10^{+00}$	$2.84x10^{+00}$	-	-	-	-	$2.61x10^{+00}$	$1.50x10^{+01}$
Acidification Potential, kg SO_2 eq.	P1	$1.93x10^{-02}$	$2.67x10^{-03}$	$-9.70x10^{-03}$	-	$2.39x10^{-03}$	-	-	$1.46x10^{-02}$
	P2	$1.61x10^{-02}$	$2.23x10^{-03}$	-	$-4.72x10^{-03}$	-	$1.72x10^{-03}$	-	$1.54x10^{-02}$
	P3	$1.39x10^{-02}$	$1.92x10^{-03}$	-	-	-	-	$-8.78x10^{-04}$	$1.49x10^{-02}$
Eutrophication Potential, kg PO_4 eq.	P1	$2.12x10^{-02}$	$3.27x10^{-04}$	$-4.49x10^{-03}$	-	$4.11x10^{-02}$	-	-	$5.82x10^{-02}$
	P2	$1.78x10^{-02}$	$2.73x10^{-04}$	-	$-2.40x10^{-03}$	-	$4.10x10^{-02}$	-	$5.67x10^{-02}$
	P3	$1.53x10^{-02}$	$2.35x10^{-04}$	-	-	-	-	$4.08x10^{-02}$	$5.63x10^{-02}$
Photochemical Oxidation, kg ethylene eq.	P1	$1.35x10^{-02}$	$3.22x10^{-05}$	$2.70x10^{-04}$	-	$1.25x10^{-04}$	-	-	$1.39x10^{-02}$
	P2	$1.13x10^{-02}$	$2.69x10^{-05}$	-	$1.84x10^{-04}$	-	$9.14x10^{-05}$	-	$1.16x10^{-02}$
	P3	$9.70x10^{-03}$	$2.31x10^{-05}$	-	-	-	-	$-4.80x10^{-05}$	$9.68x10^{-03}$
Ozone Depletion Potential, kg CFC-11 eq.	P1	$3.82x10^{-03}$	$-5.09x10^{-08}$	$3.39x10^{-07}$	-	$1.77x10^{-07}$	-	-	$3.82x10^{-03}$
	P2	$3.19x10^{-03}$	$-4.26x10^{-08}$	-	$2.47x10^{-07}$	-	$1.37x10^{-07}$	-	$3.19x10^{-03}$
	P3	$2.74x10^{-03}$	$-3.66x10^{-08}$	-	-	-	-	$-7.35x10^{-08}$	$2.74x10^{-03}$
Depletion of Abiotic Resources, kg Sb eq.	P1	$4.12x10+^{00}$	$-2.15x10^{-03}$	$1.34x10^{-02}$	-	$7.37x10^{-03}$	-	-	$4.14x10^{+00}$
	P2	$3.44x10+^{00}$	$-1.80x10^{-03}$	-	$9.78x10^{-03}$	-	$5.72x10^{-03}$	-	$3.46x10^{+00}$
	P3	$2.96x10^{+00}$	$-1.55x10^{-03}$	-	-	-	-	$-3.05x10^{-03}$	$2.96x10^{+00}$
Freshwater Aquatic Ecotoxicity, kg 1,4-DCB eq.	P1	$2.60x10^{+02}$	$-9.14x10^{-03}$	$-2.30x10^{-02}$	-	$2.88x10^{-02}$	-	-	$2.60x10^{+02}$
	P2	$2.17x10^{+02}$	$-7.64x10^{-03}$	-	$-1.25x10^{-03}$	-	$2.23x10^{-02}$	-	$2.17x10^{+02}$
	P3	$1.87x10^{+02}$	$-6.57x10^{-03}$	-	-	-	-	$-9.05x10^{-03}$	$1.87x10^{+02}$
Terrestrial Ecotoxicity, kg 1,4-DCB eq.	P1	$8.11x10^{+01}$	$-2.43x10^{-03}$	$-9.37x10^{-03}$	-	$5.04x10^{-03}$	-	-	$8.11x10^{+01}$
	P2	$6.78x10^{+01}$	$-2.03x10^{-03}$	-	$-3.5x10^{-03}$	-	$3.91x10^{-03}$	-	$6.78x10^{+01}$
	P3	$5.83\ x10^{+01}$	$-1.75x10^{-03}$	-	-	-	-	$-2.07\ x10^{-03}$	$5.83x10^{+01}$

Global Warming Potential: Agriculture is the unit process that most contributes to Global Warming Potential with 13.3 kg CO_2 in pathway 1 (59.0%), 11.1 kg CO_2 in pathway 2 (57.9%) and 9.54 kg CO_2 in pathway 3 (63.6%) mainly due to the fossil fuel consumption in trucks and machines used in cultivation, harvesting and transportation activities (Figure 4.4.). Ethanol production by distillery T4 has 19.3 % of the total impact in pathway 2, because of the emissions of 3.7 kg CO_2 to the air during fermentation. Similar impacts are obtained for T3 being 3.91 kg CO_2 (17.4%), as they have similar processes. T5 has lower impact than the ones before with 2.61 kg CO_2 because this technology shows an avoided impact charge due to the exceed steam production from the subsystem Milling.

Pathway 1 shows the highest impact with 22.5 kg CO_2 compared to pathways 2 and 3 with 19.2 and 15.0 kg CO_2, respectively. This difference of the three pathways lies in the conversion factors in subsystem Agriculture which is higher in pathway 1 to produce 1 kg of ethanol taking into account the 0.72 kg CO_2 per kg of sugarcane harvested. Pathways 1 and 2 exhibit similar values of impact because the Sugar Production results are quite similar using either technology T1 or technology T2.

Acidification Potential: the subsystem Agriculture becomes the most important contributor in this impact category with $1.93\text{x}10^{-02}$, $1.61\text{x}10^{-02}$ and $1.39\text{x}10^{-02}$ kg SO_2 equivalent (Table 4.5) with a relative contribution of 131.7%, 105 % and 93.0% for each pathway (Figure 4.4) of the total impact in this category. It can be attributed to the NO_x emitted during cane burning and fossil fuel combustion, and also to the manufacturing and decomposition in the soil of fertilizers. Distilleries T3 and T4 have also high impact values around 16.4% and 11.2%, respectively, due to the combustion of fossil fuels during steam generation, whereas T1 and T2 show avoided amounts of SO_2 due to the use of the filter muds as raw material for other process. Also T5 shows avoided credits due to the excess of steam coming from Milling.

Eutrophication Potential: Overall, the three distilleries (T3, T4 and T5) have the most important impact in this indicator, with 4.11x10^{-02}, 4.10x10^{-02} and 4.08x10^{-02} kg PO_4 (Table 4.5) equivalent with 70.6%, 72.3% and 72.5% (Figure 4.4) in each technology. It is due to the emission of a high volume of vinasses to the water courses with high content of chemical oxygen demand (COD), nitrogen and phosphorus. This indicator shows also some negative values due to the avoided impact charges of reusing filter muds in T1 and T2.

Photochemical Oxidation: Agriculture is the subsystem that shows the highest impact in this category with 1.35x10^{-02} and 1.13x10^{-02} kg ethylene for pathways 1 and 2 and in pathway 3 is 9.70x10^{-03} kg ethylene (Table 4.5) due to the emission of carbon monoxide and sulfur dioxide to air when the fossil fuel and the sugarcane are burned. Hence, Agriculture shows in three pathways approximately 96-100% of the total impact.

Almost 100% of the impact in categories such as *Ozone Depletion Potential, Depletion of Abiotic Resources, Freshwater Aquatic Ecotoxicity and Terrestrial Ecotoxicity* can be attributed to the subsystem Agriculture because of the use of pesticides, which contain methane and halocarbon compounds. This value can be adjusted accurately provided that pesticides composition is known. Water consumption due to artificial irrigation in the aforementioned subsystem is not an issue unlike the case, for instance, of Brazil (Faist et al. 2011).

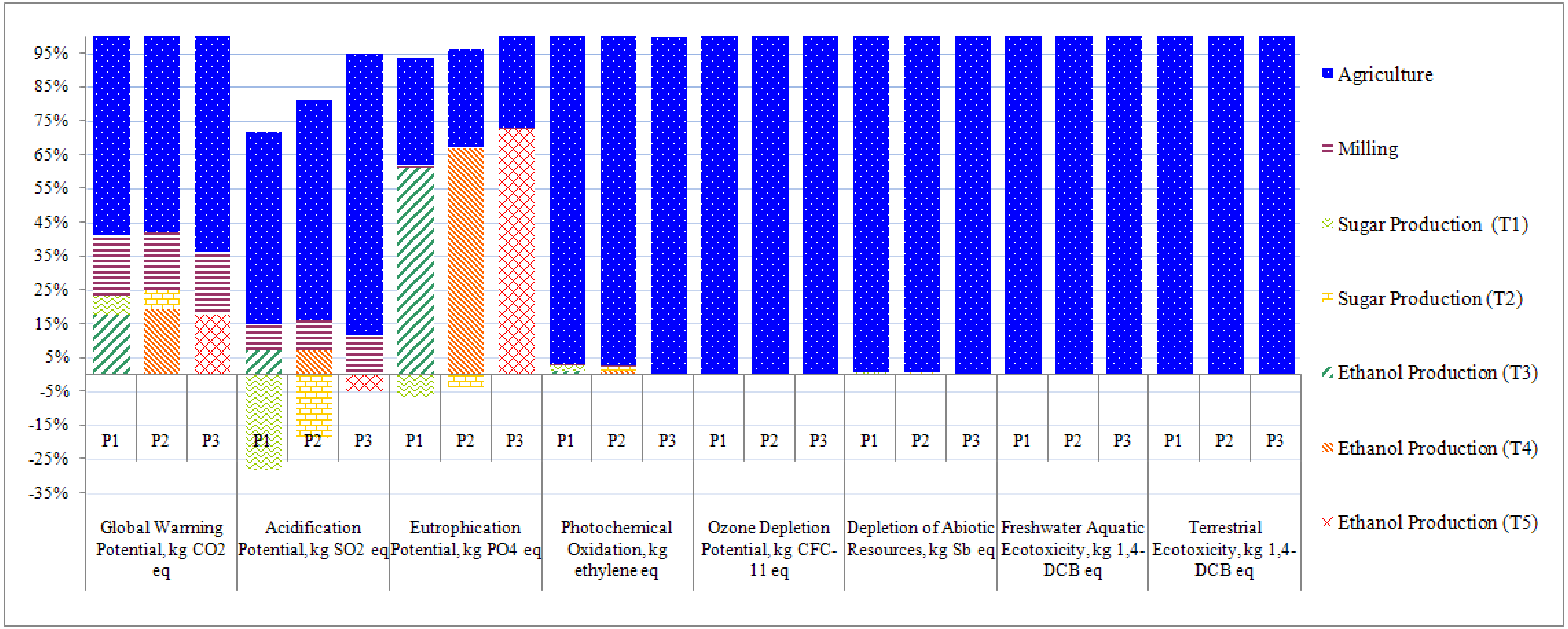
In addition, we have assessed the Land Use indicator which only has been considered in Agriculture subsystem. The life cycle inventory provides the land use which is 2.74x10^{-04} ha, 2.30x10^{-04} ha and 1.97x10^{-04}ha for pathway 1, 2 and 3, respectively.

4.3.2 Sensitivity analysis

Figure 4.4 shows the relative contribution of each impact category at each subsystem. Because of the relatively high contribution of GWP, a detailed assessment for this category impact is developed upon the reference case. Related to that, under Law 26,190 of 2006 and its regulatory framework, the Argentinean Government created the program Genren (Renewable Generation). Its main objective is to reduce emissions of CO_2 and other GHG. In addition, Argentina in early 2010 joined the Global Research Alliance, established to increase international cooperation, collaboration and investment to help reducing the GHG emissions of agricultural production and increasing its potential for soil carbon sequestration (Joseph 2010).

A sensitivity analysis to study the variability of the GWP according to different criteria has been performed. Case study 1 considers changes in the agricultural yield; Case study 2 includes the economic allocation in subsystem Sugar Production (technologies T1 and T2); Case study 3 takes into account the effect of changes in the sugar price, and finally, Case study 4 shows the effect of using energy-based allocation in subsystem Sugar Production (technologies T1 and T2).

Figure 4.4 Relative contributions of environmental impacts categories for each pathway.



4 3.2.1 Variation in the agricultural yield (*Case study 1*)

The agricultural yield regarded in subsystem Agriculture is an important source of exogenous uncertainty, which strongly depends on weather conditions. The system considered so far is called the Reference Case (RC), and three additional scenarios have been studied. The objective is to know the behavior of the GWP when agricultural yield increases 10% and decreases 10% from the RC value. Therefore, the yield values considered in the three scenarios are 73.7 t/ha (scenario 1, +10%), 67 t/ha (scenario 2, RC), and 60.3 t/ha (scenario 3, -10%). Note that, on agricultural yield variations, some inputs and outputs from the reference inventory are modified. For example, the consumption of fuel oil, urea, superphosphate, and pesticides changes. The same occurs with the emissions to the water (pesticides, PO_4^{3-}, NO_3^-) and to the air (NO_x, N_2O and NH_3).

As expected, results indicate that in all pathways, when the yield decreases in the subsystem Agriculture, the GWP impact raises and, when the yield increases the GWP impact do the opposite. Comparing the three pathways shown in Table 4.6, it is possible to identify, in pathway 1, that at 60.3 tonne·ha^{-1} yield (10% below RC value) the total impact of 25.3 kg CO_2 is the highest due to the huge impact in the subsystem Agriculture. It is interesting to take into account that the relative contribution from the reference case is plus 12.3% whereas we only have decreased 10% of yield, given that there is not a direct relation of percentage. On the other hand, the least harmful scenario for the environment is scenario 1 (maximum yield), shown in pathway 3 with 13.1 kg CO_2.

Table 4.6 Global Warming Potential (kg $CO_{2\ eq}$) for the three pathways, in Case study 1 (FU = 1 kg ethanol).

	Pathway 1			Pathway 2			Pathway 3		
	Scenario 1	Scenario 2	Scenario 3	Scenario 1	Scenario 2	Scenario 3	Scenario 1	Scenario 2	Scenario 3
Agriculture	10.7	13.3	16.0	8.95	11.1	13.4	7.69	9.54	11.5
Milling	3.95	3.95	3.95	3.31	3.31	3.31	2.84	2.84	2.84
T1 - Sugar production to molasses	1.36	1.36	1.36				-	-	-
T2 - Sugar Production to honey	-	-	-	1.05	1.05	1.05			
T3 - Distillery to ethanol (molasses)	3.91	3.91	3.91	-	-	-	-	-	-
T4 - Distillery to ethanol (honey)	-	-	-	3.70	3.70	3.70	-	-	-
T5 - Distillery to ethanol (cane juice)	-	-	-	-	-	-	2.61	2.61	2.61
Total	19.9	22.5	25.3	17.0	19.2	21.5	13.1	15.0	17.0
Change Relative to the RC, %	-11.4	0.0	12.3	-11.2	0.0	12.1	-12.3	0.0	13.3

4.3.2.2 Economic allocation in subsystem Sugar Production using technologies T1 and T2 (*Case study 2*)

The objective of this subsection is to quantify the total GWP changes when economic allocation is considered in the subsystem Sugar Production, using either T1 or T2 (scenario 1), instead of using mass allocation as in the reference case (scenario 2). The economic allocation has been calculated after reviewing the sugar and ethanol prices from Centro Azucarero (2010) and the Federal Ministry of Planning, Public Investment and Services (2010), respectively. These amounts have been converted into dollars for the study (exchange rate of October 2010).

Table 4.7 shows that, through pathways 1 and 2, the total GWP impact with economic allocation is lower than this impact with mass allocation (RC). This is because ethanol is relatively cheaper than sugars, thus ethanol production carries a lower amount of environmental burden. It is interesting to remark that changes are higher in pathway 1 (from scenario 1 to 2 is 10.7 kg CO_2) than in pathway 2 (from scenario 1 to 2 is 7.1 kg CO_2). Hence, results point out the importance of selecting the allocation for each study, due to the different sensitivity of the process studied. This had been already brought up in previous works on ethanol LCA (García et al. 2011, Ometto et al. 2009).

Table 4.7 Global Warming Potential (kg $CO_{2\ eq}$) in pathways 1 and 2 considering economic allocation versus mass allocation in subsystem Sugar Production.

	Pathway 1		**Pathway 2**	
	Scenario 1	**Scenario 2**	**Scenario 1**	**Scenario 2**
Agriculture	5.76	13.3	6.16	11.1
Milling	1.72	3.95	1.84	3.31
T1- Sugar Production to molasses	0.266	1.36	-	-
T2- Sugar Production to honey	-	-	0.442	1.05
T3- Distillery to ethanol (molasses)	4.06	3.91	-	-
T4- Distillery to ethanol (honey)	-	-	3.91	3.70
Total	11.8	22.5	12.1	19.2
Change Relative to the RC (%)	-47.5	-	-35.5	-

To sum up, from the environmental point of view, scenario 1 (economic allocation) in pathway 1 (Agriculture + Milling + Sugar Production-T1 + Ethanol Production-T3) is the best suitable scenario to produce ethanol with 11.8 kg CO_2 of total impact, being scenario 2 in that pathway (mass allocation) the worst one with 22.5 kg CO_2.

4.3.2.3 Variation in the sugar price (*Case study 3*)

As the ethanol price is fixed by the Argentinean Government and the sugar price changes according to the international market, the objective of this case study is to see how the total GWP is affected if the sugar price increases or decreases 10% from the reference case.

This case study works with pathways 1 and 2 given that only these pathways produce sugar through technologies T1 and T2, respectively. This case study is based on modifying the price of sugar used in Case study 2, i.e. modifying the economic allocation factors.

Scenario 2 in both pathways considers the reference case with economic allocation. Hence, scenario 1 corresponds to an increase of 10% on the sugar price while scenario 3 represents a decrease of 10%.

Table 4.8 shows that when the sugar price decreases, total GWP also increases, since the ethanol allocation factor turns to be higher than the sugar factor.

Table 4.8 Global Warming Potential (kg $CO_{2\ eq}$) in pathways 1 and 2 taking into account economic allocation in the reference case versus economic allocation with modified sugar price ±10%.

	Pathway 1			Pathway 2		
	Scenario 1	**Scenario 2**	**Scenario 3**	**Scenario 1**	**Scenario 2**	**Scenario 3**
Agriculture	5.29	5.76	6.33	5.72	6.16	6.69
Milling	1.58	1.72	1.89	1.70	1.84	1.99
T1 - Sugar Production to molasses	0.198	0.266	0.348	-	-	-
T2 - Sugar Production to honey	-	-	-	0.387	0.442	0.507
T3 - Distillery to ethanol (molasses)	4.07	4.06	4,05	-	-	-
T4 - Distillery to ethanol (honey)	-	-	-	3.93	3.70	3.89
Total	11,1	11.8	12.6	11.7	12.1	13,1
change relative to the RC, %	-5,63	-	6.81	-3.34	-	7.71

Finally, comparing scenarios 1 and 3 some conclusions about changes in the GWP impact corresponding to changes in the sugars price can be derived. By and large, the tendency in both pathways is as follows. When the reference price of sugar increases 10%, the total GWP impact slightly decreases, whereas if the sugar price is reduced by 10%, the impact also increased, but in a higher percentage.

Regarding the results from Table 4.8, the most favorable scenario to produce ethanol is through pathway 1 (Agriculture + Milling + Sugar Production-T1 + Ethanol Production-T3) using economic allocation with 10% high prices for sugar. This scenario impacts with 11.1 kg CO_2. On the other hand, the worst case is though pathway 2 (Agriculture + Milling + Sugar production-T2 + Ethanol Production-T4) using economic allocation, with 10% less prices for sugars in scenario 3 (13.1 kg CO_2).

4.3.2.4 Calorific Value allocation in subsystem Sugar Production using technologies T1 and T2 (*Case study 4*)

The objective of this subsection is to quantify the total GWP changes when calorific allocation is considered in the subsystem Sugar Production, using either T1 or T2 (scenario 1), instead of using mass allocation as in the reference case (scenario 2). Even though sugar is not an energy product (i.e, not used as a fuel), it is interesting to complete the allocation analysis with some conclusions related to the energy content of the products, being an antecedent the work by García et al. (2011). The calorific allocation has been calculated after reviewing the sugar (FDDB 2012) and ethanol (Edwards et al. 2011) calorific values. If we compare the total GWP values obtained from calorific allocation (scenario 1) with the mass allocation (RC) (scenario 2), Table 4.9 provides lower impacts for scenario 1. Pathway 1 in scenario 1 is reduced from scenario 2 in a 40.3% and pathway 2 is reduced in a 27.9%. This is because ethanol's calorific value is relatively higher than sugars. Likewise, this case study shows again the importance to choose a suitable allocation. Analogously to section 4.3.2.2 the best scenario to produce ethanol at the environmental impact point of view is scenario 1 (calorific allocation) in pathway 1 (Agriculture + Milling + Sugar Production-T1 + Ethanol Production-T3).

Table 4.9 Global Warming Potential (kg $CO_{2\ eq}$) in pathways 1 and 2 considering calorific value allocation versus mass allocation in subsystem Sugar Production.

	Pathway 1		**Pathway 2**	
	Scenario 1	**Scenario 2**	**Scenario 1**	**Scenario 2**
Agriculture	6.90	13.3	7.22	11.1
Milling	2.06	3.95	2.15	3.31
T1- Sugar Production to molasses	0.432	1.36	-	-
T2- Sugar Production to honey	-	-	0.571	1.05
T3- Distillery to ethanol (molasses)	4.04	3.91	-	-
T4- Distillery to ethanol (honey)	-	-	3.87	3.70
Total	13.4	22.5	13.8	19.2
Change Relative to the RC (%)	-40.3	-	-27.9	-

4.4 Conclusions

Law 26,093 in Argentina is an opportunity for bioethanol to use and mobilize the human and natural resources. Argentinean bioethanol producers will need to study the environmental performance of their product in order to comply with sustainability criteria because the LCIA conclusions are that the ethanol life-cycle contributes to all the impacts analyzed. The sugarcane industry does not only affect the environment in terms of global warming but also it contributes to other impacts like acidification and eutrophication (Chauhan et al. 2011).

The reference case study to produce 1 kg of ethanol shows: (i) pathway 1 emits 22.5 kg CO_2, which relative contribution are structured in S1-agriculture (59.0%), S2-milling (17.6%), S3-sugar production T1 (6.00%) and S4-distillery T3 (17.4%); (ii) pathway 2 emits 19.2 kg CO_2, with a relative contribution of main impacts are S1-agriculture (57.9%), S2-milling (17.3%) and S3-sugar production T2 (5.50%) and S4-distillery T4 (19.3%); and (iii) pathway 3 emits 15.0 kg CO_2, which relative contribution is S1-agriculture (63.6%), S2-milling (19.0%) and S5-distillery T5 (17.4%).

Generally, all pathways have similar relative distributions being Agriculture the highest impact subsystem. This conclusion is reinforced by LCA studies conducted in other countries (García et al. 2011; Pereira and Ortega 2010; Ometto et al. 2009). However, pathways 1 and 2 have similar impacts because they have similar technologies, having both pathways a higher impact than pathway 3. Hence, the influence of the cane processing system and agricultural yield variability suggests that studies should be specific, concentrating at the regional scope. The recommendations that can be drawn are strongly related with those set by other authors in other countries (Nguyen et al. 2008; Ometto et al. 2009): (i) substituting fossil for biomass-based fuels in agriculture labour; (ii) wastewater treatment at distilleries; (iii) using cane trash for energy instead of open burning in fields; and (iv) water recycling systems during the industrial processing.

The sensitivity analysis allows drawing conclusions in three respects. (i) As the subsystem Agriculture includes the sections with a higher contribution to the environmental impact of the ethanol production, this impact is strongly influenced by agricultural yields: (ii) The LCA results for the Argentinean bioethanol are strongly dependent of the process pathway selected; (iii) The environmental impact calculated on a financial allocation basis can mask the impact of bioethanol production as it becomes dependent on the relative market price of sugar/ethanol.

Impact allocation to the multiple products system from sugarcane is an important issue (García et al. 2011; Renouf et al. 2011). When co-products appear (e.g. molasses, electricity, ethanol), economic allocation is less useful due to uncertainty in the mechanism for assigning impacts. To enable the consistent representation of impact across the full range of sugarcane products, mass allocation combined with energy allocation for the energy obtained from bagasse combustion and cogeneration is preferable.

Although so far the uncertainty associated with the Argentinean ethanol production is very high, the consideration of the water treatment subsystem within the system boundaries of the study is an issue of enormous importance that should be considered in future works.

According to the large amount of variables and process options that participate in the environmental evaluation of products like bioethanol the results presented in this paper demonstrate the need of progress in the standardization of protocols to calculate impacts based on LCA of products. The international market of products considered as commodities, requires common frames to evaluate the sustainability of products addressed to final consumer.

Part III

Decision Making Tool in Agriculture

Chapter 5

Farm-gate environmental assessment indices based on the water footprint and life cycle approach

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Chapter 5 is based on the following paper:

Amores MJ, Antón A, Ferrer F, Cabot O, Baltiérrez A, Duaigües A, Castells F. Farm-gate environmental assessment indices based on the water footprint and life cycle approach. Journal of Cleaner Production (Submitted in March 2013).

Abstract

Irrigated agriculture faces the challenge of improving farming tasks by carrying out an on-site performance evaluation and strategic assessment of farm water management practices. A methodological approach is adopted to develop an operational tool that makes it possible to generate an on-farm simple inventory that may be used to calculate environmental sustainability indicators regarding water consumption and global warming potential, as well as providing assessment for farm managers to make strategic and tactical decisions.

The purpose of the work is to set up an operational procedure at a farm level to evaluate water management practices, including water consumption and environmental impact through the water footprint and life cycle approach. The global Life Cycle Inventory includes data such as water consumption, rainfall, evapotranspiration, irrigation, farm tasks, fertilizers and chemical treatments, transport and distribution.

The end products are water, carbon and agronomic indices to construct a web-based application in which the user enters field-measured data to obtain a set of indicators from three different approaches: agronomic, water-use sustainability and global warming impact. As regards water and environmental assessment, the results are shown in three sections: (I) environmental water impact assessment, (II) environmental carbon impact assessment, and (III) agronomic indices. Results are evaluated for three consecutive campaigns (2010-2012) in two case studies: grape and nectarine. Highlight differences are obtained between real and theoretical blue water footprint for a grape farm, being more uniform for a nectarine farm. Regarding the Global Warming Potential assessment, the fertilizer stage has the highest contribution to the total environmental impact.

In order to test the calculations and facilitate the implementation of this procedure, a set of commercial farms located in the Ebro Basin (NE Spain) are used as pilot operational cases.

This work constitutes an attempt to implement an inventory procedure at a farm management unit level, with the aim of quantifying and assessing the water consumption impact and the contribution to global warming by greenhouse gases and to enable them to be integrated into the managers' dashboard for making strategic decisions. This tool is intended to be user-friendly and sufficiently generic to be applied in different cropping systems and agricultural areas of the world, enabling simple reports to be generated using the results of the analysis. This application has the potential to become a useful tool in the supply chain for evaluating and benchmarking fresh product providers (agro-producers), based on objective parameters.

Keywords: life cycle assessment, water footprint, water management, agriculture, environmental tool, decision making.

5.1. Introduction

Societal concern about environmental problems has increased the demand for reliable information and tools for understanding and mitigating environmental damage. One of the biggest water problems around the world is scarcity; in many regions, water supplies are insufficient to satisfy all agricultural, industrial, urban and environmental requirements (Jefferies et al. 2012). Irrigated agriculture faces the challenge of improving water management practices in order to meet requirements of consumption (quantity and quality), available resources (energy and water) and of course the environmental, social and economic sustainability of farming activity. In the process of providing humanity with food, agriculture places a serious burden on the environment (FAO, 2003). On average, it is the largest consumer of water (almost 86% of all water used in the world is to grow food (UNESCO-WWAP, 2009)), a main source of water pollution such as nitrate leaking into groundwater and surface water, and the principal source of ammonia as well as contributing to the phosphate pollution of waterways (OECD, 2001). Moreover, agriculture is a significant contributor to land degradation and the release of powerful anthropogenic global greenhouse gas emissions (GHG). It is responsible for 25% of carbon dioxide, 50% of methane and more than 75% of nitrous oxide emitted annually by human activities (Tubiello et al. 2007) due to emissions from nitrogen fertilizers and fossil resource consumption (IPCC, 2007). Global warming potential is the most widely studied impact category as a result of the initiatives taken by the agri-food business (i.e. supermarket chains), which advocates the use of carbon footprints (CF), since agricultural activities contribute 10-12% of GHG globally (Smith et al. 2007). The agri-food sector should assume responsibility for implementing environmental action regarding freshwater conservation and reduction of greenhouse gas emissions, as well as achieving the productive goal of providing enough food.

Several environmental indicators are developed to assess farm practices. The Water Stress Index (WSI) (Pfister et al. 2009) is an approach to measure the impact of water consumption (water footprint impact assessment) by means of the blue water footprint, while the Global Warming Potential (GWP) is one approach to measure the environmental impact from a life-cycle perspective (Morrison et al. 2010). The concept of 'water footprint' was introduced by Hoekstra et al. (2003) and subsequently developed by (Hoekstra et al. 2009, 2011). It provides a framework for analyzing the link between human consumption and the appropriation of freshwater resources, incorporating both direct and indirect water use by a consumer or producer. Nowadays, this water footprint approach is further used by several practitioners.

An international standard for water footprint is currently being drawn up by the International Organization for Standardization (ISO) 14046 (under development). The GWP refers to the sum of greenhouse gas (GHG) emissions caused by an organization, event or product, and is expressed in terms of CO_2 equivalents (IPCC, 2007). International standards such as the ISO 14047 (ISO 2012) provide guidelines within the framework of Life Cycle Assessment (LCA) (ISO14040-44, 2006a,b) for calculating carbon footprints. Furthermore, specific standards such PAS 2050.1 (BSI, 2011a) or those from the World Resources Institute (WRI, 2012) and the World Business Council for Sustainable Development (WBCSD), provide specific guides for measuring the carbon footprints of horticultural and agricultural products, respectively.

Many studies on food and agricultural products have been published to account for water footprint: Chapagain and Orr, 2009; Ercin et al.2011; Jefferies et al. 2012; Mekonnen and Hoekstra, 2010a, 2011; Ridoutt et al. 2009) and for GWP or carbon footprint: Torrellas et al. 2013; Gan et al. 2012a,b; Ma et al. 2012; Ingram and Fernandez 2012; Cellura et al. 2012; Hillier et al. 2009; Maheswarappa et al. 2011; Pathak et al. 2010; Röös et al. 2010; Vázquez-Rowe et al. 2012; Cerutti et al. 2010; Bosco et al. 2011; Point et al. 2012; Beccali et al. 2010), as well as for both water and carbon footprints: Page et al. 2012; Stoessel et al. 2012; Chapagain et al. 2006. Some calculators have recently been proposed for the assessment of water and carbon. The Water Footprint Network published an Extended calculator (Hoekstra et al. 2005), which helps users to gauge how much water they expend on a daily basis (water requirements per unit of product). In the field of agricultural practices, the most relevant published tools for sustainability assessment are the Fieldprint calculator (Alliance, 2012), which evaluates how crop production operations affect sustainability at farm level, and the Cool farm tool calculator (Unilever, 2012), a greenhouse gas calculator for farming, which takes into account emissions from fields, inputs, livestock, land use and land use change and primary processing.

The aim of the present study is to develop and validate a farm-gate accounting method for irrigated crops in order to assess on-farm environmental and sustainable indices based on the Water Footprint manual and the LCA methodology, together with the calculation of efficiency productive performance indices using as much realistic in-situ recorded data from the field as possible. This approach goes beyond the previously cited tools and considers a life-cycle approach of operations on a farm, including the direct and indirect uses of water and input and output carbon emissions. The organization of the data inventory, the output reports and their interpretation (impact assessment) has been performed by taking the practice of the farming decision-making scheme into account. The environmental impact approach to water consumption is carried out on the basis of the consumptive-based volumetric water footprint calculations (WF network Manual, Hoekstra et al. 2011) and the WSI approach (stress-weighted water footprint) according to Pfister et al. 2009, Ridoutt and Pfister, 2010). Agronomic performance and productivity indices related to water use (irrigation, energy use efficiency, water use efficiency, etc) and irrigation scheduling management (uniformity coefficient irrigation, effective precipitation) are also calculated based on ISO. Finally, the global warming contribution is calculated based on the LCA approach (ISO 14040-14044, 2006 a,b).

Following the principle of parsimony "as simple as possible and as complex as necessary" (Pidd, 1996), and in order to obtain a simplified and combined calculation tool for water footprint and GWP assessment plus productivity of crop products, the selected reference is the yield of the product per unit of surface (kg·ha^{-1}) of representative Farm Management Units (FMU) within the farm selected by the manager, while the time unit is the growing season (from harvest to harvest) and the spatial limit of the system is the farm barn. One of the key aspects of the study is to provide the inventory with realistic on-site data rather than using estimated values from generic databases. This is especially important where yield and water consumption are concerned. The assumption is that this simplified calculator may be integrated into the farm manager's dashboard to make strategic decisions and to benchmark the FMU performance, bearing in mind changes or improvements that may have a beneficial impact on crop performance and environmental impact at a farm level.

In this study, the combined indices are used to analyze two case studies taken from real irrigated commercial farms located in the Ebro Basin (NE Spain) and corresponding to two crops, grape and nectarine. The specific goals of the study are as follows:

- Combine and validate the calculations based on the WF and GWP methodologies, together with a productive evaluation of the crop performance, using the most realistic on-farm inventory data.
- Compare indices from different consecutive campaigns for the same field in order to check the calculator response to variability over different years.
- Evaluate the integral results of the indices from a methodological point of view and propose a way for future improvements to make a more useful tool for farm managers.

5.2. Methods

5.2.1. Case Studies

Case studies were applied to two representative crops in the area, grape and nectarine. Two farms, both in compliance with some quality standards (e.g. Global G.A.P.), and their managers expressed genuine interest in making some type of sustainability assessment of their fields and farms. To conduct the study, Farm Management Units (FMUs) on each farm located in the irrigated area around Lleida (Ebro Basin, NE Spain) were selected (Table 5.1). The growing seasons that came under assessment belong to the period 2010 - 2012.

Table 5.1 The main agronomic characteristics from the 2010 to 2012 growing season located in the Ebro Basin.

Item	Unit	Grape			Nectarine		
		2010	2011	2012	2010	2011	2012
Variety	-		Chardonnay			Sweet Beginner fruit	
Farm location[2]	Lat /Long (°)		41.7/ 0.48			41.5/ 0.46	
Irrigation period	weeks	24	26	28	35	34	37
Irrigation	$m^3 \cdot ha^{-1}$	854	753	1,420	5,852	5,424	6,934
Yield	$kg \cdot ha^{-1}$	15,294	19,144	13,149	46,663	44,780	50,445
Density of plantation	$unit \cdot ha^{-1}$		2,088			800	
Surface	ha		60.2			12.5	
First production after plantation	years		2nd (70%)			2nd (66%)	
Previous crop before plantation	-		corn			barley	
Soil texture	-		loam			clayloam	

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[2] The selected farms are the same than Chapter 3 but the assessed fields are different.

5.2.2. System boundary

In the frame of Life Cycle Thinking, it is necessary to define the system boundary. It is important that the scope of reported emissions is both comprehensive and consistent between different crops. The system boundary was defined from raw material extraction to farm gate. Pre-farm processes (often referred to as 'cradle') such as the extraction of raw materials, production, and transport of inputs used on the farm are also included in the assessment. The production system (farm-gate) is structured in different stages (production of packaging and fertilizers; transport of materials to the closest cooperatives; use; energy; irrigation; auxiliary equipment and waste management). The results are expressed in the form of key life-cycle stages for each crop, as shown in Figure 5.1.

WFIA and GWP are assessed by using kilograms of commercial production as a functional unit for grapes and nectarines.

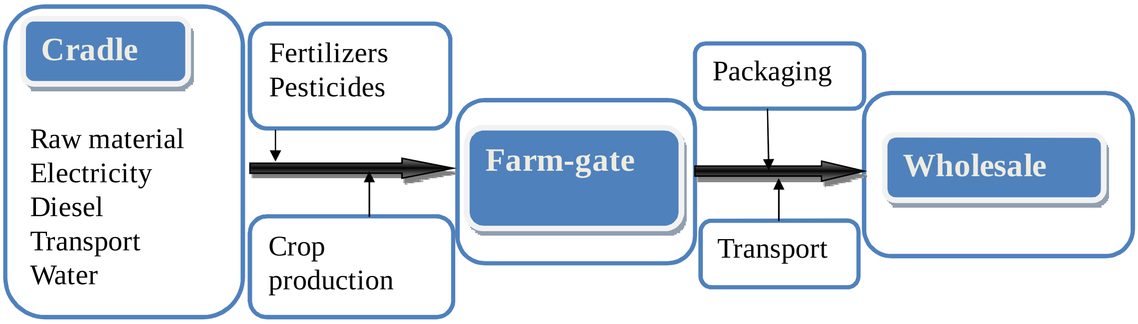

Figure 5.1. Flowchart from cradle to farm-gate to wholesale.

5.2.3. Water footprint assessment

The water footprint differentiates the volumes of water consumed by assigning 'water-colors' (green, blue and grey) depending on the source of the water consumed and its relative level of pollution (Hoekstra et al. 2011). The water footprint of a crop is the volume of water used to grow it; that is: the sum of the rainfall stored in the soil that is consumed by the crop (evapotranspiration) during growth (green water), the total volume of surface water or groundwater resources used during the growing period (blue water), plus the theoretical volume required to dilute polluted water (generated during growth) to an "unpolluted" condition (grey water) (Hoekstra et al. 2011). In this work, the total water footprint of the process of growing crops (WF_{tot}) is the sum of the green (WF_g) and blue ($WF_{b,\ actual}$) components, grey water being left out of the scope (5.1):

$$WF_{tot} = WF_g + WF_{b,a} \ (\mathrm{m^3 \cdot tonne^{-1}}) \qquad \text{Equation (5.1)}$$

This differentiated assessment is very useful for determining the origin of water consumption as well as enabling an objective comparison to be made of data coming from different climates and conditions.

5.2.3.1. Green water footprint (WF_g)

The green water footprint (WF_g) is calculated according to the following equation (5.2):

$$WF_g = \frac{AR_{eff}}{Y} \ (\mathrm{m^3 \cdot tonne^{-1}}) \qquad \text{Equation (5.2)}$$

where AR_{eff} is the sum of monthly effective rainfall during the crop season ($\mathrm{m^3 \cdot ha^{-1}}$). The AR_{eff} is calculated from the monthly precipitation value, according to the following algorithm (5.3):

If monthly rainfall (R) <=70 $\mathrm{l \cdot m^{-2}}$; $R_{eff} = 0.6 * R - 10$.

If monthly rainfall (R) >70 $\mathrm{l \cdot m^{-2}}$; $R_{eff} = 0.8 * R - 24$ (Smith, 1988) Equation (5.3)

Rainfall data (R) are obtained from public weather stations (www.ruralcat.net) (Table 5.2).

5.2.3.2. Blue water footprint (WF_b)

The blue water footprint is calculated from real consumptive water ($WF_{b,actual}$, $\mathrm{m^3 \cdot tonne^{-1}}$). Since in many cases this data is not usually available, the blue water footprint is estimated from the crop Evapotranspiration (ETc) ($WF_{b,theoretical}$, $m^3 \cdot tonne^{-1}$).

In this work, the blue water footprint is calculated using both real consumptive water ($WF_{b,a}$) (5.4, 5.5) and theoretical consumptive water $WF_{b,t}$ (5.6). A comparative analysis between these values is then conducted in order to identify the differences:

$$WF_{b,a} = \frac{I}{Y} \quad (m^3 \cdot tonne^{-1}) \qquad \text{Equation (5.4)}$$

where I is the net irrigation in $m^3 \cdot ha^{-1}$ and Y is the yield in $tonne^{-1} \cdot ha^{-1:}$

$$I = I_b \cdot WAE \cdot (1 - LF) \quad (m^3 \cdot ha^{-1}) \qquad \text{Equation (5.5)}$$

In turn, *I* (5.5) is equal to the total irrigation amount (I_b) in $m^3 \cdot ha^{-1}$, multiplied by the water application efficiency of the irrigation system *(WAE)* and the leaching fraction *(LF)*, respectively, as explained in detail in Section 5.2.5.

$$WF_{b,t} = \frac{ET_c - AR_{eff}}{Y} \quad (m^3 \cdot tonne^{1}) \qquad \text{Equation (5.6)}$$

where *ETc* is the accumulated crop evapotranspiration ($m^3 \cdot ha^{-1}$) and AR_{eff} the Accumulated Effective Rainfall ($m^3 \cdot ha^{-1}$). ET_c is obtained as a result of the product between the locally adapted crop coefficient (K_c) from FAO references (Allen et al. 2006) and the potential evapotranspiration (ET_o, $l \cdot m^{-2}$) (5.7).

$$ET_c = kc \cdot ET_o \quad (m^3 \cdot ha^{-1}) \qquad \text{Equation (5.7)}$$

Kc values are based on the generic FAO values (Smith,1988) adjusted to the specific cropping systems by the extension service of the Catalan Government Department of Agriculture, while ET_o is obtained from meteorological data bases belonging to public weather stations near the farms under study (www.ruralcat.net). Table 5.2 shows the monthly Kc, ET_o and rainfall (R) data for each crop studied in the cultivation area (Ebro Basin).

5.2.3.3. Water Footprint Impact Assessment ($WFIA_b$)

The WF indicator provides an account of freshwater consumption, but neglects the availability of the resource and it provides no information on impact assessment. Thus, other indicators are required to provide information on the impact of watering systems on the crop area, taking into account the overall water resources.

To assess the impact of water, we introduce the concept of Water Footprint Impact Assessment ($WFIA_b$) as an indicator of the effect of irrigation water related to the availability of water in the cultivation basin. The relative water availability can be measured by the Water Stress Index (WSI). WSI indicates the portion of consumptive water use that deprives other users of freshwater. It has been used for the assessment of the environmental impact due to water consumption at a midpoint level considered in LCIA and ranges from 0.01 to 1, with 1 indicating serious water stress in a basin. WSI values are taken from (Pfister et al. 2009).

The water footprint impact assessment ($WFIA_b$) for the blue-water footprint is given in equation 5.8. The water indices are understood to the consumptive-based volumetric blue WF estimated by using the water footprint manuals (Hoekstra et al. 2011) and the stress-weighted water footprint developed by (Ridoutt and Pfister, 2010) as the impact of water use employing the WSI proposed by Pfister et al. (2009).

$$WFIA_b = WF_{b,a} \cdot WSI \ (m^3 \cdot tonne^{-1}) \qquad \text{Equation (5.8)}$$

where $WFIA_b$ is the Water Footprint Impact Assessment ($m^3 \cdot t^{-1}$) by the blue-water footprint ($WF_{b,a}$, $m^3 \cdot tonne^{-1}$), and WSI is the Water Stress Index [-] applied (Pfister et al. 2009) in the watershed where the crop is growing; e.g. WSI=0.259 for the Ebro basin.

Table 5.2. Grape and nectarine crop characteristics and climatic parameters for campaigns 2010 to 2012 in farms located in the Ebro Basin.

Crops	Grape							Nectarine								
Months	Kc	Rainfall (R, l·m^{-2})			Evapotranspiration (ETo, l·m^{-2})			kc	Rainfall (R, l·m^{-2})			Evapotranspiration (ETo, l·m^{-2})		
		2010	2011	2012	2010	2011	2012		2010	2011	2012	2010	2011	2012
January	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	24.7
February	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	47.7
March	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	24.1	67.9	15.7	43.6	44.4	89.0
April	0.1	19.1	21.1	52.0	60.8	79.8	90.5	0.6	28.5	18.3	54.7	85.4	103.6	96.4
May	0.3	25.8	28.3	21.0	105.1	109.7	123.0	0.8	39.1	41.4	10.4	115.3	124.4	151.4
June	0.3	31.5	36.0	27.0	124.1	119.8	146.5	1.0	69.2	32.9	16.3	139.1	134.6	167.1
July	0.3	0.8	7.3	26.5	150.6	169.6	134.5	0.6	5.0	8.8	12.9	165.2	161.3	164.9
August	0.4	1.0	0.0	57.0	134.5	146.9	135.7	0.5	43.7	0.0	10.6	152.2	151.3	154.0
September	0.4	0.0	11.5	34.5	119.0	138.8	77.0	0.4	2.2	10.6	50.7	128.9	131.1	95.6
October	0.2	0.0	0.0	0.0	3.3	73.0	0.0	0.4	61.3	0.7	108.6	87.2	97.8	63.1
November	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	39.8	36.1	32.7	53.0	52.1	27.5
December	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	23.4

5.2.4. Calculation of global warming potential

In accordance with ISO 14040-44 (2006 a,b), the Global Warming Potential (GWP) is a measure of the impact that human activities have on the environment in terms of the amount of greenhouse gas emissions (GHG) emitted over the full life-cycle of a process or product measured in units of carbon dioxide (CO_2) equivalents. The GWP is expressed in $CO_{2\text{-eq}}$ and its value also depends on the degree to which gas concentration decays over time in the atmosphere; a time horizon of 100 years is commonly used in accordance with the United Nations Framework Convention on Climate Change and the Kyoto Protocol (IPCC 2007).

It is calculated as shown in equation 5.9.

$$GWP = Input_{flow} \cdot Cf \quad (\text{kg } CO_{2,eq}) \qquad \text{Equation (5.9)}$$

where GWP is provided in units of kg $CO_{2,eq}$, the input flows refer to the Life Cycle Inventory (LCI) of the crop's life cycle (e.g. unit of energy), and Cf is the characterization factor per unit flow (kg $CO_{2,eq}\cdot$ process unit^{-1}). LCI was built from primary data from farmers themselves, while secondary and environmental data were obtained from the Ecoinvent database 2.2 (Ecoinvent, 2010).

5.2.5. Calculation of agronomic indices

In order to support and complement water indices, the following agronomic equations were also calculated, thereby contributing to a better assessment of water use from an agronomic approach. The theoretical irrigation requirements or accumulated gross irrigation needed (AGIN) were calculated from crop evapotranspiration (ET_c), effective rainfall (R_{eff}), water application efficiency of the irrigation system (WAE) and the required leaching fraction (LF) (Allen et al. 2006).

AGIN indicates the total amount of water needed to be applied by the irrigation system in order to reach the theoretical demand (ET_c). Since some water may be provided by rainfall, the irrigation system is not 100% efficient and sometimes the irrigation water may contain salts that need to be leached from the root area. The irrigation system defines the WAE value and standard values exist based on the irrigation system type (60% for flood irrigation, 75% for sprinklers, 90% for drip irrigation).

The LF is the percentage of extra irrigation water needed for leaching irrigation water salts (5.10).

$$LF = \frac{CE_a}{2 \cdot CE_{max}} \text{ (-)} \qquad \text{Equation (5.10)}$$

where CE_a ($dS \cdot m^{-1}$) is the Electrical Conductivity of irrigation and CE_{max} is the Maximum Electrical Conductivity ($dS \cdot m^{-1}$), based on the bibliography. CE max values for the main crops in the world can be found in FAO nº56 (Allen et al. 2006).

In a first step, the accumulated net irrigation needed (ANIN) (5.11) can be calculated by simply taking into account the ET_c and AR_{eff}:

$$ANIN = K_c \cdot ET_o - AR_{eff} \text{ } (m^3 \cdot ha^{-1}) \qquad \text{Equation (5.11)}$$

By taking into account the WAE, LF and ANIN calculated previously, we are able to define AGIN according to equation 5.12:

$$AGIN = \frac{ANIN/WAE}{(1-LF)} \text{ } (m^3 \cdot ha^{-1}) \qquad \text{Equation (5.12)}$$

The water use efficiency (WUE) expressed in equation 5.13 represents the product yield (kg) by cubic meter (m^3) of water applied (by both irrigation and rainfall):

$$WUE = \frac{Y}{I + AR_{eff}} \text{ } (kg \cdot m^{-3}) \qquad \text{Equation (5.13)}$$

A complementary analysis of WUE performed in the study consisted of tracking water consumption throughout the season in order to determine the differences between the AGIN and the net irrigation (I).

Energy use efficiency (EUE) represents the product yield (kg·ha^{-1}) by unit of energy used for pumping and distributing the irrigation water (kWh·ha^{-1}) (5.14).

$$EUE = \frac{Y}{E} \text{ (kg·kWh}^{-1}\text{)} \qquad \text{Equation (5.14)}$$

Irrigation water cost (IWC) (5.15) represents the cost (€)by product yield (kg·ha^{-1}) derived from irrigation, taking into account the price of water (PW, €·ha^{-1}) in addition to the price of the energy (EC, €·ha^{-1}) for pumping and distributing this water:

$$IWC = \frac{EC + I \cdot PW}{Y} \text{ (€·kg}^{-1}\text{)} \qquad \text{Equation (5.15)}$$

The Deficit of Irrigation (DI) is the difference between applied water (I_b, m^3·ha^{-1}) and gross irrigation needed (AGIN, m^3·ha^{-1}). Positive values of DI mean that the real irrigation water applied is higher than theoretical gross irrigation water needed (5.16).

$$DI = I_b - AGIN \text{ (m}^3\text{·ha}^{-1}\text{)} \qquad \text{Equation (5.16)}$$

5.3. Results and discussion

5.3.1 On-farm recorded data of the growing season and LCA inventory for period 2010-2012

Table 5.3 shows the data that was obtained directly from the farm managers' records for each of the Farm Management Units (FMUs) at each farm (grape and nectarine). These include commercial yield (kg·ha^{-1}); the total accumulated amount of applied water (l·m^{-2}); the total amount of consumed energy used for pumping the irrigation system (kWh·ha^{-1}); fuel consumption for the machinery (l); the amount of fertilizers, pesticides, insecticides and herbicides involved (kg); transport (tkm) and packaging (kg). All input flows from the life cycle inventory of different farms are normalized by the Functional Unit (FU) (kg of produced crop).

5.3.2. Results of water footprint assessment

The results for water footprint indices are shown in Table 5.4. Both grape and nectarine crops present different scenarios for the water footprint. On the one hand, grapes are a crop in which irrigation is a key management factor for yield quality; differences between the irrigation applied in 2012 compared with that 2010 and 2011 can be observed. In the 2012 campaign, both the productive target and weather conditions were very different from the periods. Furthermore, it is not simply a matter of the total amount of water applied to the vineyard, but also how this water was distributed throughout the season. Although the $WF_{b,t}$ is more or less the same throughout the seasons, one may observe that the real irrigation applied is in fact different from and independent of rainfall. It is for this reason that almost no information can be taken from WF indices. However, it is clear is that there is no correlation between the amount of irrigation and volume of yield obtained, since many other factors need to be taken into account (productive target, number of grapes by vine, harvesting data, grape chemical traits, among others).

On the other hand, the situation with nectarines is different; the $WF_{b,t}$ and $WF_{b,a}$ are similar and constant throughout the different seasons, since these values only differ between 3% and 15%. The farmer attempted to fulfill the theoretical water demand of the crop with irrigation. As regards rainfall, it is very low compared with irrigation water, so the WF_g has very little significant weight as productive index. For the Nectarine crop, the WF_{tot} shows that in 2012 the farmer obtained a higher yield, but there was a higher proportion of irrigation water applied in comparison with previous years, resulting in an increase in the WF_{tot} increased, mainly in the the $WF_{b,a}$ fraction, thereby leading to a higher WFIA (30 $m^3 \cdot tonne^{-1}$).

Results from the two representative farms show that nectarine has a higher water footprint impact assessment than the grape farm for the period 2011 to 2012. WFIA is directly proportional to $WF_{b,a,}$ since both farms are located in the same watershed (Ebro basin), so if the real water footprint is higher then so is the respective impact. The WSI proposed by Pfister et al., (2009) may be divided into four categories, taking into account that 0.01 is the most suitable value and 1 is the worst: good WSI (0-0.25), regular (0.25-0.5), bad (0.5-0.75), very bad (0.75-1.0). In order to improve the farmer's water assessment, the authors propose giving variability between real and theoretical data of 15% if the farm is located in areas with good WSI; 10% in areas with regular range; 5% in areas with bad range and only 1% in areas with very bad range. The grape and nectarine farms are located in the Ebro Basin, where the WSI is equal to 0.259, so a variability of 10-15% between real and theoretical values would be acceptable for the farmer's water management.

The division of WF into blue and green provides a much better contribution to data analysis. The differences in these examples may not be very high, since rainfall is scarce during the growing season, and the WFIA is equivalent because both farms are irrigated from the same basin. However, should these contexts be different, this set of indices would have great value for a further analysis.

Table 5.3 Data obtained directly from the farm managers' records for each Management Unit (FMU), including the Life cycle inventory of three crops (FU= kg of produced crop).

Category	Item	Unit	Grape			Nectarine		
			2010	2011	2012	2010	2011	2012
Fuel	Diesel	kg· FU^{-1}	5.74×10^{-03}	4.65×10^{-03}	6.82×10^{-03}	5.95×10^{-03}	5.95×10^{-03}	5.44×10^{-03}
Energy	Electricity	kWh ·FU^{-1}	1.77×10^{-03}	9.15×10^{-04}	1.48×10^{-03}	5.14×10^{-03}	5.60×10^{-03}	4.07×10^{-03}
Fertilizer	N	kg·FU^{-1}	3.20×10^{-03}	2.04×10^{-03}	1.03×10^{-03}	2.41×10^{-03}	2.17×10^{-03}	2.18×10^{-03}
	P_2O_5	kg·FU^{-1}	2.58×10^{-03}	1.49×10^{-03}	1.79×10^{-03}	6.00×10^{-04}	4.25×10^{-04}	6.74×10^{-04}
	K_2O	kg·FU^{-1}	5.43×10^{-03}	3.89×10^{-03}	3.80×10^{-03}	3.60×10^{-03}	2.55×10^{-03}	4.06×10^{-03}
Pesticide	Fungicide	kg·FU^{-1}	6.06×10^{-04}	5.75×10^{-04}	1.22×10^{-03}	8.87×10^{-04}	8.26×10^{-04}	9.28×10^{-04}
	Herbicide	kg·FU^{-1}	2.19×10^{-04}	2.09×10^{-04}	3.04×10^{-04}	2.57×10^{-04}	2.55×10^{-04}	2.38×10^{-04}
	Insecticide	kg·FU^{-1}	0	0	0	7.98×10^{-04}	6.24×10^{-04}	8.07×10^{-04}
Water	Irrigation	m^3·FU^{-1}	5.58×10^{-02}	3.93×10^{-02}	1.08×10^{-01}	1.25×10^{-01}	1.21×10^{-01}	1.37×10^{-01}
Transport	Lorry	tkm·FU^{-1}	3.00×10^{-03}	3.00×10^{-03}	3.00×10^{-03}	3.12×10^{-02}	3.12×10^{-02}	3.12×10^{-02}
Packaging	Plastic pallet	kg·FU^{-1}	0	0	0	5.86×10^{-04}	5.94×10^{-04}	5.96×10^{-04}
Machinery	95 % Steel	kg·FU^{-1}	6.62×10^{-04}	5.29×10^{-04}	7.70×10^{-04}	1.74×10^{-03}	1.72×10^{-03}	1.61×10^{-03}
Infraestructure	5% Caux	kg·FU^{-1}	1.11×10^{-05}	8.88×10^{-06}	1.29×10^{-05}	3.45×10^{-05}	3.42×10^{-05}	3.19×10^{-05}

Table 5.4 Results of water footprints, blue (real and theoretical); green, total and impact assessment for the campaigns 2010 to 2012 for grape and nectarine on farms located in the Ebro basin.

Water indices	Units	Grape			Nectarine		
		2010	2011	2012	2010	2011	2012
$WF_{b,a}$	$m^3 \cdot tonne^{-1}$	48.2	33.9	129.7	104.2	100.6	114.2
$WF_{b,t}$	$m^3 \cdot tonne^{-1}$	124.2	116.2	108.4	101.1	119.4	101.2
WF_g	$m^3 \cdot tonne^{-1}$	10.4	11.1	53.8	24.3	15.2	22.9
WF_{tot}	$m^3 \cdot tonne^{-1}$	58.5	45.0	183.5	128.5	115.8	137.1
$WFIA_{b,a}$	$m^3 \cdot tonne^{-1}$	12.5	8.8	33.6	27.0	26.1	29.6

Footnote: Blue Water Footprint, actual ($WF_{b,a}$), Blue Water Footprint, theoretical ($WF_{b,t}$), Green Water Footprint (WF_g), WF_{tot} Water Footprint Impact Assessment ($WFIA_{b,a}$)

5.3.3. Results of global warming potential

The results in Table 5.5 are expressed as a functional unit (1 kg of crop) and are presented as absolute values of carbon dioxide emissions. Figure 5.2 shows their relative impact contribution in the defined system boundary (from raw material extraction to the farm gate) for grape and nectarine farms during the campaigns 2010 to 2012. The total results for the production system are based on the farmer's data, and they are calculated in the life-cycle impact assessment phase of the LCA, with the aim of identifying the main contributing impact stages as well as obtaining the total Global Warming Potential for each campaign by farm. The carbon assessment is conducted on the basis of the attributional LCA for the environmental assessment in accordance with the ISO 14040-14044 standard (2006a,b).

The main total GWP contribution comes from the fertilizer stage for both grape and nectarine farms. The results for fertilizer contribution (Figure 5.2) show that emissions from their production and their application to the farms have the highest contribution in this category impact (~40-76%). Emissions are evaluated according to Bentrup et al. (2000) and are calculated on the basis of total N applied: N_2O-N as 1.25% of total N applied. This high impact contribution is due to the high level of emissions released during their manufacture and field application. The contribution of fertilizer use may be satisfactorily estimated on the basis of chemical consumption and the generally agreed-upon emission factors for chemical production.

Table 5.5 Emissions of Global Warming Potential by crop production for different percentages.

Stages and units	**Grape**			**Nectarine**		
	2010	**2011**	**2012**	**2010**	**2011**	**2012**
Fuel (kg $CO_{2,eq}$·kg crop^{-1})	2.94x10^{-03}	2.38x10^{-03}	3.49x10^{-03}	3.04x10^{-03}	3.04x10^{-03}	2.79x10^{-03}
%	5.1	5.9	9.1	3.9	4.3	3.7
Transport (kg $CO_{2,eq}$·kg crop^{-1})	7.74x10^{-04}	7.74x10^{-04}	7.74x10^{-04}	8.04x10^{-03}	8.04x10^{-03}	8.04x10^{-03}
%	1.3	1.9	2.0	10.3	11.4	10.8
Energy (kg $CO_{2,eq}$·kg crop^{-1})	7.43x10^{-04}	3.85x10^{-04}	6.22x10^{-04}	2.16x10^{-03}	2.36x10^{-03}	1.71x10^{-03}
%	1.3	1.0	1.6	2.8	3.3	2.3
Fertilizer (kg $CO_{2,eq}$·kg crop^{-1})	4.36x10^{-02}	2.79x10^{-02}	1.65x10^{-02}	3.37x10^{-02}	2.97x10^{-02}	3.11x10^{-02}
%	75.6	68.9	42.8	43.3	42.2	41.6
Treatments (kg $CO_{2,eq}$·kg crop^{-1})	8.66x10^{-03}	8.23x10^{-03}	1.60x10^{-02}	2.53x10^{-02}	2.18x10^{-02}	2.57x10^{-02}
%	15.0	20.3	41.5	32.6	30.9	34.4
Packaging (kg $CO_{2,eq}$·kg crop^{-1})	0.0x10^{+00}	0.0x10^{+00}	0.0x10^{+00}	2.90x10^{-03}	2.94x10^{-03}	2.95x10^{-03}
%	0.0	0.0	0.0	3.7	4.2	3.9
Infrastructure (kg $CO_{2,eq}$·kg crop^{-1})	1.01x10^{-03}	8.07x10^{-04}	1.18x10^{-03}	2.67x10^{-03}	2.64x10^{-03}	2.47x10^{-03}
%	1.7	2.0	3.0	3.4	3.7	3.3
Total (kg $CO_{2,eq}$·kg crop^{-1})	0.058	0.040	0.039	0.078	0.071	0.075

The second major contribution to the total GWP of the grape and nectarine farms is the production of insecticides, fungicides and herbicides and their application on the farms (~15-43%). The assessment of different treatments is calculated according to the total amount of each, since there is an appreciable variability in the active ingredient formulations. Minor burdens come from transport (~1-11%), fuel (~3-6%), infrastructure (~1-4%), packaging (~0.1-5%)

and energy consumption (~0.1-3%). The transport stage includes the production transport from farm to wholesale, the transport of materials within the farm and the transport of waste materials to a treatment plant. A particular type of lorry was selected from Ecoinvent database, -lorry 3.5-16t, fleet average- for all transports considered. The Spanish mix is used in order to calculate the environmental impact of electricity consumption for each campaign. The type of packaging used to carry the nectarine production was polyethylene pallet. No packaging was used for grapes. The machinery infrastructure was mainly composed of a 90 kW tractor for the grape farm and 60 kW for the nectarine farm, with different complements (subsoiler, plough, slurry spreader and fertilizer sprayer) being employed for farm tasks.

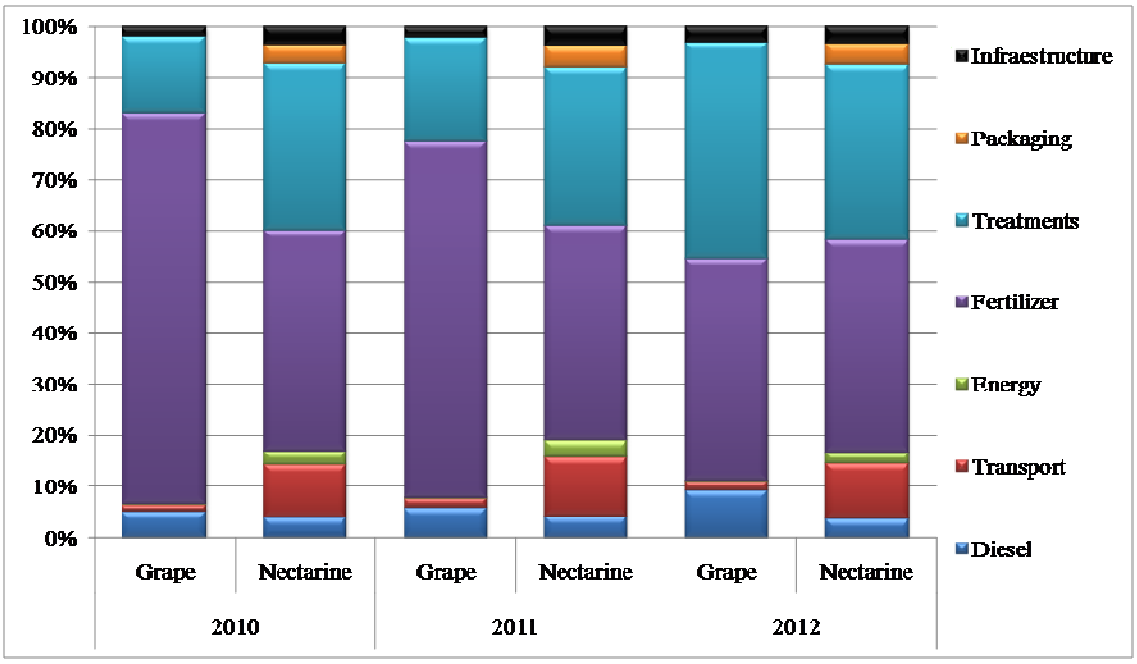

Figure 5.2. Relative contribution of GWP impact for different stages from farm-gate to wholesale.

Overall, total GWP emissions are higher in the nectarine farm than grape farm due to its higher contributions from transport, energy, treatments and packaging stages. The maximum GWP value was registered on the nectarine farm in the 2010 campaign 2010 (0.078 kg of $CO_{2,eq}\cdot FU^{-1}$), corresponding to the lowest production for that year. The grape farm registered a maximum $CO_{2,eq}{\cdot}FU^{-1}$ emission of 0.057 kg in the 2010 campaign. As a result of improvements made by farmers, who adjusted the amount of inputs to actual crop requirements, both grape and nectarine farms have shown a decrease in their emissions between 2010 and 2012.

5.3.4. Results of agronomic assessment

Table 5.6 shows the different water indicators that may help farmers in their water management. The WUE indicator shows that the maximum production by irrigated water volume on grape farm took place during the 2011 campaign, when the maximum production was obtained (~19 tonne·ha^{-1}) and minimum irrigation was applied (~750 m^3·ha^{-1}). The nectarine farm shows a maximum WUE in the 2010 campaign.

With regard to the EUE (kg·kWh^{-1}), the use of energy was more efficient for grapes in 2011 and for nectarines in 2012.

Table 5.6 Water indices related to agronomic efficiency of water use and irrigation scheduling management.

Indices	Units	Grape			Nectarine		
		2010	2011	2012	2010	2011	2012
WUE	kg·m^{-3}	5.64x10^{+00}	6.66x10^{+00}	4.30x10^{+00}	4.89x10^{+00}	4.57x10^{+00}	4.64x10^{+00}
EUE	kg·kWh^{-1}	9.40x10^{+00}	1.82x10^{+01}	1.12x10^{+01}	1.56x10^{+01}	1.36x10^{+01}	1.97x10^{+01}
ANIN	l·m^{-2}	1.97x10^{+02}	2.23x10^{+02}	1.47x10^{+02}	4.67x10^{+02}	5.29x10^{+02}	5.10x10^{+02}
AGIN	l·m^{-2}	2.29x10^{+02}	2.58x10^{+02}	1.71x10^{+02}	5.62x10^{+02}	6.37x10^{+02}	6.14x10^{+02}
IWC	€·kg^{-1}	6.71x10^{-03}	4.73x10^{-03}	1.81x10^{-02}	1.51x10^{-02}	1.45x10^{-02}	1.65x10^{-02}

Footnote: Water Use Efficiency (WUE), Energy Use Efficiency (EUE), Accumulated Net Irrigation Needed (ANIN), Accumulated Gross Irrigation Needed (AGIN), Irrigation Water Cost (IWC).

WUE and EUE have proved to be good indices for management and stewardship, since the farmer is able to estimate how much water and energy will be required for a potential yield, taking into account all the efficiency factors such as the WAE, LF and an estimation of the rainfall.

The ANIN indicator shows a maximum value for both grape and nectarine farms during the 2011 campaign, while both farms also registered similar climatic parameters during 2011 (high volume of ET_o and low values of R_{eff}). Considering that in both farms the water application efficiency (WAE) of the irrigation system is the same (90% drip irrigation) and that the leaching fraction is 4% in grape and 8% in nectarine, AGIN values are higher on the nectarine farm, reaching a maximum value in the 2011 campaign. AGIN and ANIN have proved useful for determining differences between the theoretical water needs and the real water applied, since ET_c is not the main factor for on-farm irrigation decisions. The grape crop is a good example of this, while for the nectarine crop one may state that ET_c has more specific weight for irrigation decisions, as is also shown by the DI indice.

5.4. Conclusions

A web-based application is proposed for calculating water, carbon and agronomic indices/impacts to assist farmers in their decision-making processes for sustainable agriculture.

Based on actual field data, the tool provides a full inventory and calculates a set of indices for building a sound environmental and agronomic profile of agricultural products.

WF, GWP and agronomic indices are essential for enabling farmers to fulfill their agronomic and environmental needs. In addition, the use of actual data from farms provides an accurate evaluation that may differ considerably from theoretical data.

A comparison using these water and carbon indices over three consecutive campaigns (2010-2012) for a same field enables farmers to assess management and provides them with a guide for the following campaign. Moreover, assessment of practices to reduce the environmental impact of irrigation (mainly by avoiding consumption in excess of real necessities and reducing the impact of leaching and erosion) should be conducted at a Farm Management Unit scale and should be integrated into the manager's dashboard. Therefore, the quantification of these indices should be based on a solid and simple conceptual model, enabling it to be integrated into the farmer's decision-making processes.

By means of these midpoint indices (e.g. GWP) it is possible to identify hotspots across and within the life-cycle stages studied on farm gate. The research reveals that measures which address hotspots such as fertilization and treatment in crop production is a priority for reducing carbon dioxide emissions into the atmosphere.

Additional on-site agronomic issues should be taken into account in the future, such as grey water footprint; the estimation of realistic on-site leaching fractions; the inclusion of information on the concentration of solutes in irrigation and drainage water; the aridity of the climate and the adjustment of the crop coefficient to the specific conditions of the farm management unit.

Chapter 6

The eFoodPrint® Calculator: accounting and assessing on-farm environmental sustainability indices in irrigated agriculture

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eFood
Print
Agrifood Environmental Sustainability

Chapter 6 is based on the following software:

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Abstract

The objective of chapter 6 is to transmit the scientific knowledge and the environmental assessment to the users through easy but rigorous way with the development of the environmental support tool to determine the water use management and environmental impacts of different agriculture crops by assessment of different indices. An effort was made to provide an easy-to-use tool in order to reach a wide audience and help agriculture stakeholders choose efficient options to mitigate the environmental impacts of producing crops and to improve the water management techniques.

Farmers or other users can estimate the environmental performance of their crops by entering a limited amount of data and following a few easy steps. A questionnaire must be answered with data on the crop, life cycle stages from cradle to gate (production of fertilizers, pesticides, transport at farm, use, waste management, energy, auxiliary equipments, water consumption).

The calculator was structure in two main assessments: I) water footprint and II) carbon footprint based on a simplified LCA result in Global Warming Potential indicator. They were analyzed in detail by proposed methodology in chapter 5 within the context of the *Model de l'Ús Sostenible de l'aigua a l'Agricultura* Project (AGAUR, 2011).The used farms for the study either comply with some quality standards (ex: Global G.A.P., Tesco Nature's choice) or their managers show genuine interest to make some kind of sustainability assessment of their fields and farms. Concerning Life cycle assessment, the selected functional unit was one kilogram of

crop. Default data were given for each reference system (e.g. climatic weather stations) for users who did not have complete specific data and to provide results for comparison with users' own results. Then, the results were presented by means of water and agronomical indices, water footprint assessment as well as the impact category of global warming and water impact assessment.

Multiple web-based calculators have come on the market as tools to support sustainable decision making, but few are available to agriculture. The resulting calculator is an useful tool to simulate the environmental performance of horticultural production systems and it is also helpful to growers and advisers for evaluating the efficiency of input reduction options through proposal water indices.

6.1. Introduction

Nowadays, agriculture faces the need to improve management practices from an environmental point of view. That refers to its impact on the environment (emissions to air, water and soil) and the consumption (quantity and quality) of resources (energy, land and water, among others). This, of course, has to be in accordance with the economic and social sustainability of the farming activity. All these aspects could be considered at a farm level or taken a larger spatial domain (local, catchment…).

From the water use assessment point of view, about 2/3 of the worlds available freshwater is used to produce food. However, the production of raw agricultural products is a complex process that does not have globally accepted indicators to quantify and assess the use of water. This makes quite difficult defining strategies related to water use in agriculture as required by the current environmental agendas (Agenda 21 and DMA 2000/60/CE). The final goal is to have some numbers that can help farmers, technicians and planners (among others) to make strategic decisions towards a more sustainable use of water.

From a management point of view, making strategically and tactical decisions in farming is difficult, as the system is highly complex, external factors with unknown effects, with interactions and non-linearity (Stirzaker, 2011).

There is a need to assess management practices at a farm level in order to improve water use and global warming impacts in conjunction with optimization of production, yield quality and cost reduction. Whatever method or tool that claims to be useful for that purpose should be integrated into the farm's technical, social and economical idiosyncrasy, and as simple and meaningful as possible.

The present thesis should be considered the core a project that aims at developing and implementing a software tool to calculate environmental sustainability indices to assess water use and global warming impacts in irrigated agriculture. The goal is to offer a solution for the stakeholders located at the beginning of the food-supply chain (managers, planners etc.), that are required to offer environmental and production efficiency true values. The chapter is focused on:

- The production of agricultural raw products from irrigated agriculture.
- Farm-gate level: the boundaries of the system are limited to the farm gate.
- The structure of the software (data input and result reports) has to be in concordance with the farm's management organization, in order to facilitate the adoption of the tool by the potential users.
- The data that is required as input will be as much on-site as possible. Special care is taken to select parameters that are accessible from records that are already available in many farms.
- Strategic and benchmarking multi-purpose tool: from the management and decision making point of view, the new tool is intended to fill the blank of a strategic analysis tool of the farm's performance from season to season and among its various management units.

- The software will tackle water use and global warming impacts as well as productive farm's performance indices.
- Calculation and impact of water use and global warming is done through the Footprint and LCA methodologies, and referred to product unit.
- The eFoodPrint® calculator will give more weight to water use than to global warming. The reason is that water management is the main factor in irrigated agriculture in semi-arid regions, where the project's participants have their main area of influence.

The description of the calculation algorithms is done in Chapter 5. Chapter 6 explains how the input data and the output reports are structured, and thus how the calculation algorithms are implemented to become an on-farm's accounting and assessment strategic tool. This is done by showing the screen shots as they follow the software's map.

6.2. State of the art

Given the increasing concern about environmental issues, and particularly water use due to human consumption, the demand for environmental information has increased (Halberg, 2004). In order to respond to this societal concern and the recognition that these issues are better understood if a life-cycle approach is taken, several public and private companies, administrations and universities have developed multiple web-based environmental-impact calculators as simplified life cycle management tools to support decision making and to find more environmentally sustainable solutions. These web-based calculators are based on methods to account for the water footprint and global warming potential or even web-calculators to give a quick estimation of water use and environmental performance in different environments (personal use, industrial, agriculture, etc.). Life Cycle Assessment (LCA) provides quantitative and qualitative environmental information and it could be use as management tool (and in combination with other environmental tools) for calculating the life cycle's environmental behavior of products, services and activities (Finnveden et al. 2009).

Several web-calculators have been developed in different fields: *ecodesign* (Okala Life Cycle Analysis calculator), *construction* (Athena EcoCalculator), *energy* (3GSolar) and *waste* management (Farreny and Gasol, 2012), *personal consumption habits in food industry* (Epp and Reichenback, 1999), *sustainable shopping* (Pasqualino et al., 2009) and *industrial activities* (Azapagic et al., 2010).

Agriculture is a tricky field which needs to pay attention towards more sustainable practices to reduce associated environmental impacts and water management. Pivotal sources such as energy consumption, fertilizers and pesticide application, water and land use and machinery become the major issues to be reduced. During the last time, the following four calculators have been explored in this field.

- The Water Footprint Network published an *extended calculator* which can be accessed at: http://www.waterfootprint.org/?page=cal/WaterFootprintCalculator. This calculator helps the user to know how much water he/she uses day by day. The calculations are based on the water requirements per unit of product as in user's country of residence.
- The *Fieldprint calculator,* published by the Field to Market, the Keystone Alliance for Sustainable Agriculture in the US: http://www.fieldtomarket.org/fieldprint-calculator, evaluates how crop production operations affect the sustainability at farm level. It focuses on four crops (wheat, corn, soybean and cotton) and assesses energy use, climate impact, soil loss, and water use.
- The *Cool farm tool calculator*, commissioned by Unilever from the University of Aberdeen, http://www.growingforthefuture.com/content/Cool+Farm+Tool, is a GHG calculator for farming. It calculates the greenhouse gas balance of farming,

including emissions from fields, inputs, livestock, land use and land use change and primary processing.

- Recently, Torrellas et al. (2013) has been published an environmental impact calculator for greenhouse production systems. This tool enclosed in the framework of Efficient Use of Inputs in Protected HORticulture Research project (EUPHOROS, 2008-2012) is a contribution in the protected horticultural production in order to develop an environmental support tool to determine the environmental impacts of protected crops.

All these calculators are good tools to orientate farmers to more sustainable practices. Our approach for eFoodPrint® goes beyond these tools and considers a life-cycle approach of operations within a farm, including the direct and indirect uses of water. EFoodPrint® will also allow the differentiation between blue and green water footprint, as described by the Water Footprint Manual, published by the Water Footprint Network (Hoekstra et al. 2009, 2011), other water and agronomical indices and a Global Warming Potential. Furthermore, eFoodPrint® tries giving a response to the possible requirements from supermarkets and wholesalers about risky, healthy and security aspects through a Global G.A.P. (Good Agricultural Practice) questionnaire.

6.3. The eFoodPrint® calculator

From the very initial stages of the eFoofPrint® project, the farm manager and the technicians have been integrated into the eFoodPrint® working group, in order to help defining an operational method to conduct the accounting and to obtain a meaningful tool at the end. To satisfy the user' interests, the eFoodPrint® application is "easy-to-use", "simple as possible" (required minimum data to do the assessment), "complicated as necessary", "robustness" and "accessible".

The end product is a web-based calculator (www.efoodprint.com), where the user would enter field-measured data and the eFoodPrint's algorithms would conduct water sustainability calculations based on the three defined approaches (Agronomical, Water assessment and Global Warming Assessment) and indices. The eFoodPrint® calculator will be user-friendly and generic enough to be applied in different cropping systems and agricultural areas, enabling to generate simple reports with the results of the analysis. Apart from giving information about the environmental impacts of products and activities, this life cycle tool aim to provide advice on how to reduce the impacts in the agricultural production system.

6.3.1. The conceptual blocks

6.3.1.1. Water Footprint

Water footprint (green and blue) has been adapted from the water footprint (Hoekstra et al. 2011).

Taking into account the Water Stress Index provided by Pfister et al. (2009) for worldwide watersheds, the tool assesses the Water Footprint Impact Assessment at watershed level.

6.3.1.2.Farm productive performance indices

In order to support and complement water indices several farm performance indices such as water application efficiency, accumulated effective rainfall, leaching fraction, accumulated gross and net irrigation needed, water and energy use efficiency, irrigation water cost and deficit of irrigation have been assessed in this tool. To evaluate the performance from a productive and water use efficiency point of view, some indices have been traditionally used (Hoffman et al., 2007; Fessehaziona et al., 2011 and Stirzaker, 2011). They could be classified in three categories: *first*, water productivity or irrigation requirements based on water consumption, yield and evapotranspiration (ET) estimations; *second*, leaching fraction or drainage requirements for soil and water salinity control; and *third*, irrigation system performance.

6.3.1.3.Global G.A.P. water management requirements

eFoodPrint® also includes Global G.A.P. questionnaire which is the worldwide standard that assures it. The aim of this questionnaire is to connect agricultural stakeholders in the production and marketing of safe food to achieve: sustainable food for everyone everywhere today and in the future, responsible use of water resources, easier certification and wider markets for producers, welfare of workers and animals and minimizing the negative impact to the environment.

6.3.1.4.Carbon footprint

Carbon footprint assessment has been done based on attributional LCA for the environmental assessment in accordance with the ISO 14040-14044 standard (ISO-14040-14044, 2006 a,b). Since the goal of the eFoodPrint® is to improve the production management, the defined system boundary was from raw material extraction to the farm gate leaving out of scope the subsequent stages, processing and commercialization.

The total carbon results show total results from the most relevant stages from the agriculture production system for Global Warming Potential impact category (Guinée et al. 2002) in emissions of kg $CO_{2,eq}$ by functional unit (FU) i.e. kilogram of crop. The detailed results could split up with the contribution of each source. Environmental data are obtained from Ecoinvent database 2.2 (Ecoinvent, 2010).

6.3.2. Scope of eFoodPrint®

6.3.2.1. Farm Management Unit (FMU)

To be useful to the farm manager and to facilitate data mining, eFoodPrint® works at a Farm Management Unit (FMU) level, which may correspond to a particular field where water consumption, potential water crop requirements, yield, manpower, machinery and other parameters could be assigned. At the end of the season, these indices could help assessing its strategy, by benchmarking the FMUs performance, think about changes or improvements that may have a beneficial impact on the crop and water use performance of the farm. To be able to steer irrigation management along the growing season and make tactical decisions, reliable information should be obtained at different scales (FMUs, farm and watershed).

6.3.2.2.Data mining

Quality of the data used is basic to calculate reliable indicators. Data mining should be obtained from real monitored data or field observations, as much as possible. The data required to run eFoodPrint® calculator will be sourced from different spatial domains: watershed, irrigation community farm and plot/field. The use of on-farm sensors (such as soil moisture, wetting front detectors and rain gauge, among others) will not be necessary to provide input variables to the calculator, but will be a recommended as complementary tools to guide the farmer’s tactic decisions and to make a better adjustments and back-proof of the parameters and indices used by the program. The farm-management records (ex.: hours of cultural practices, etc) are also crucial.

6.3.2.3. Management and strategically decision making

This tool presents an attempt to implement an inventory procedure at a management unit level with the aim of calculating production, water use and environmental indices to quantify and assess the impact of irrigation, and to integrate it into the managers' dashboard to make strategically decisions. The results from eFoodPrint® could help orienting the application of the much more complex Life Cycle Assessment (LCA) to quantify and assess the environmental impact of irrigation considering a system beyond the limits of the farm.

6.3.3. Structure of the software:

6.3.3.1.Diagram of the software

The calculator consists of 6 blocks: 1.Cover, 2.registration, 3.Home page, 4. Water Input Data, 5. Carbon Input Data, 6. Reports (Figure 6.1).

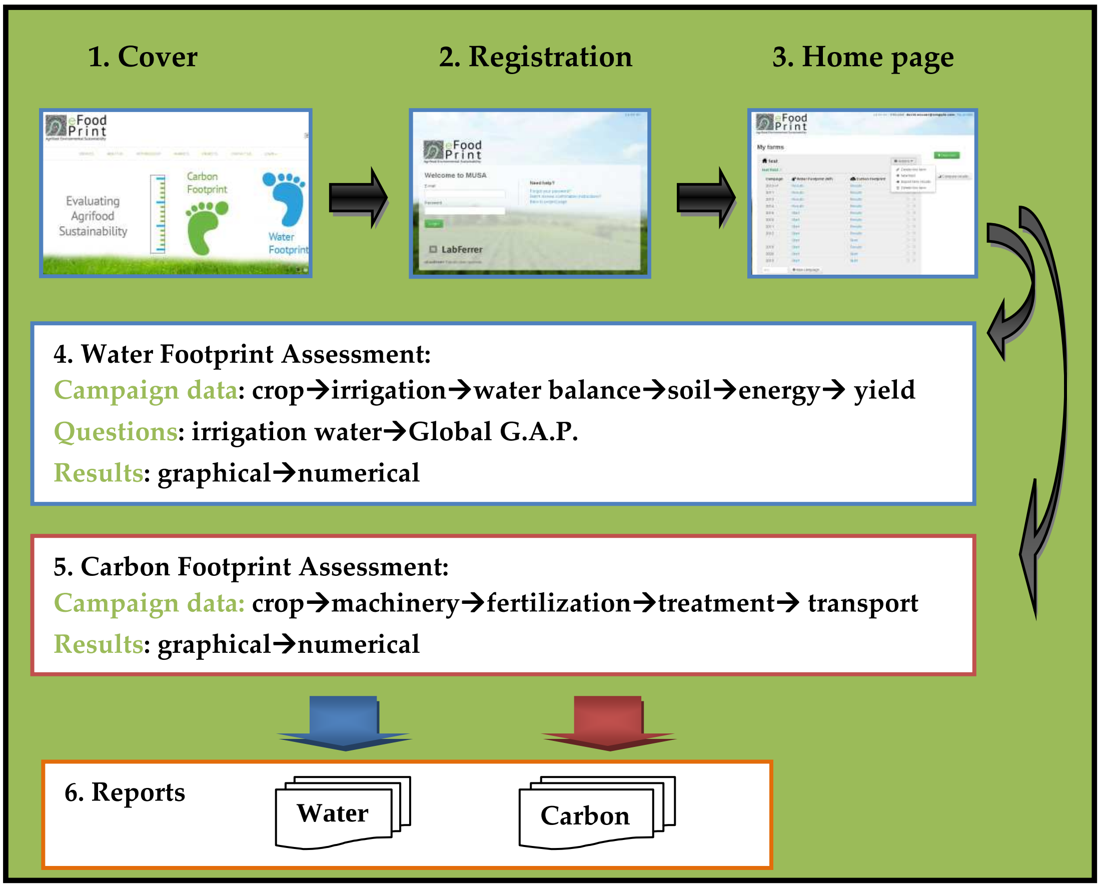


Figure 6.1 Diagram of the eFoodPrint®.

The ***eFoodPrint® Cover*** (Figure 6.2.) is a presentation to eFoodPrint® tool. Find enclosed an introduction to the environmental calculator with some instructions to use and the main steps to register new users. In addition, the *services* that the tool can offer, the *team* who has participated in the development of the tool, the future *stakeholders* for who is focus on, some *successful cases* and also, several published news in *press* are explained in different sections of the cover page. The last link is the login access where the practitioners only have to follow few easy steps to conduct an environmental simulation. Finally, this environmental calculator is available in different languages: English, Catalan and Spanish.

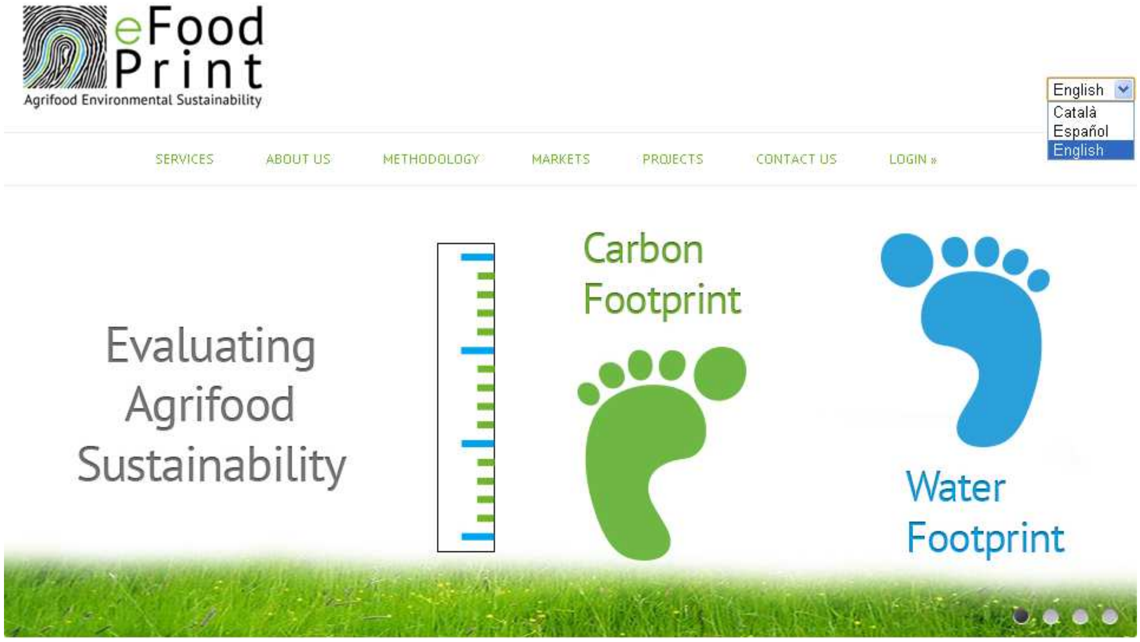


Figure 6.2 Cover slide.

Through the *login* link in cover, the new user can access to the registration form to obtain a login, if the user is not yet registered, he/she can access directly to the application by means of e-mail and password (Figure 6.3.).

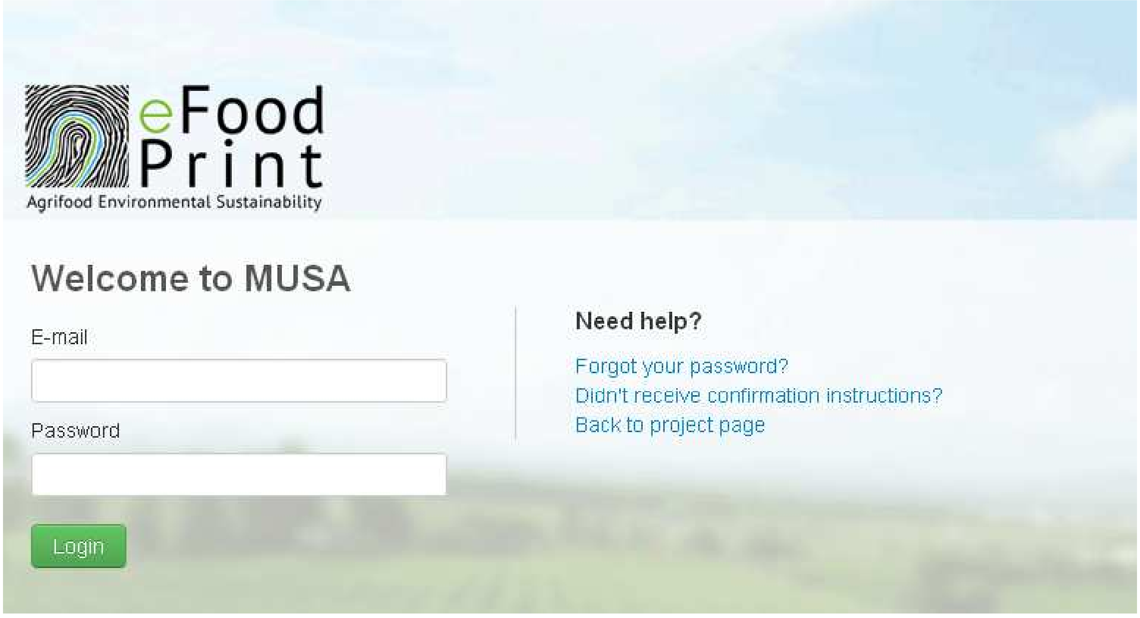
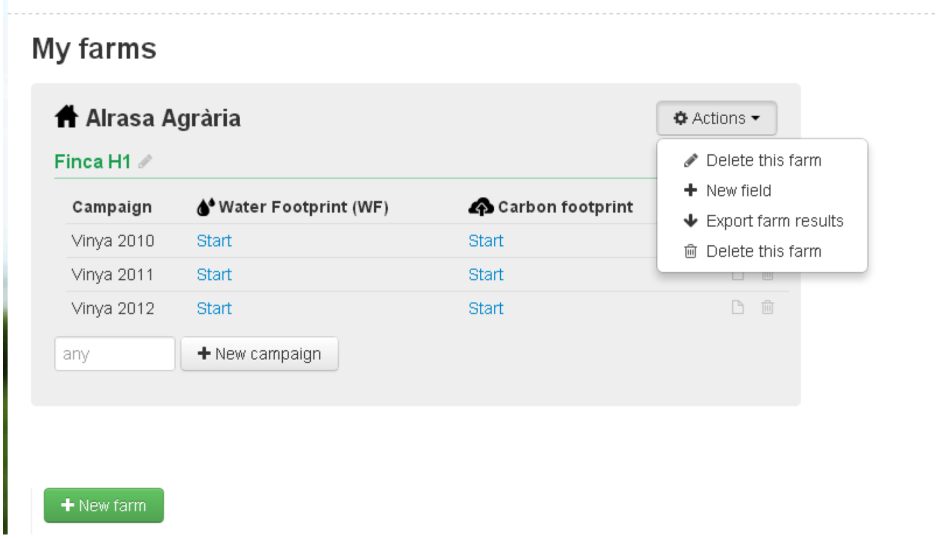


Figure 6.3 Registration form or login to access to the eFoodPrint®.

Then, the user can access to the **Home page** (Figure 6.4.). There are two assessments which can be assessed in eFoodPrint®: water and carbon footprint. The agronomic performance analysis is included inside the Water Footprint side. Here, the home page organizes the data by farms (e.g. nectarine, grape, corn) and each farm is also organized by campaigns (e.g. grape 2010, grape 2011, grape 2012) based on the different crops. In addition, some actions can be done in each campaign as for example: a) adding a new farm, b) editing the existed data, c) export the results and d) deleting a campaign.



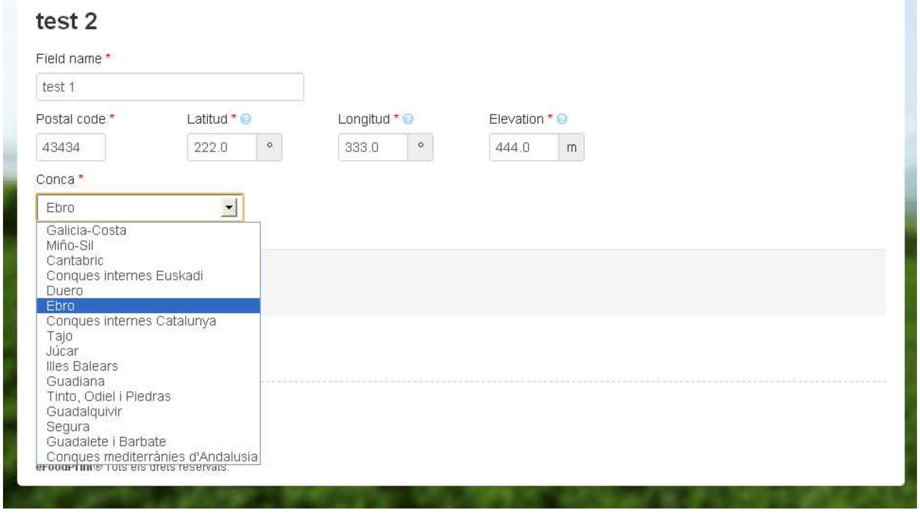
Campaign	Water Footprint (WF)	Carbon footprint
Vinya 2010	Start	Start
Vinya 2011	Start	Start
Vinya 2012	Start	Start

Figure 6.4 The Home page of eFoodPrint®.

6.3.3.2.Definition of the system: farm and field

The scope of the enclosed calculator was to set an operational procedure at a field and farm level to conduct water efficiency and sustainability analysis.

Different farms can be added through the hyperlink "New farm". Then, some information regarding farm such as name, postal code, latitude, elevation and the watershed where it takes place, are required by the user (Figure 6.5).



test 2

Field name *: test 1

Postal code *: 43434

Latitud *: 222.0 °

Longitud *: 333.0 °

Elevation *: 444.0 m

Conca *: Ebro

Figure 6.5 Information required for adding a new farm.

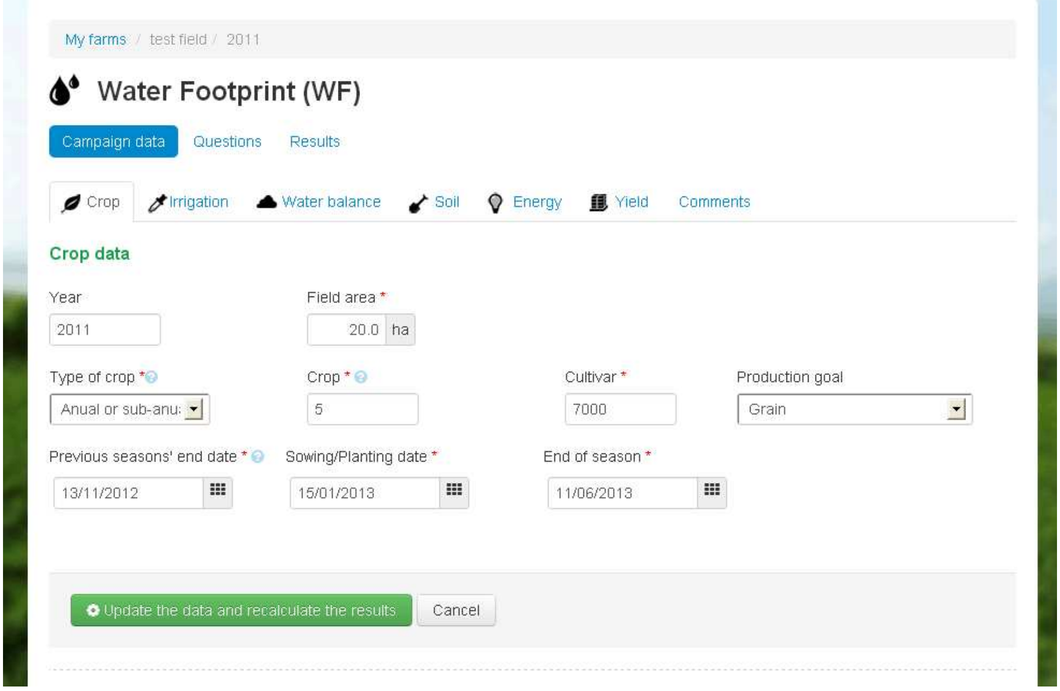
When a campaign has been constructed, the user can access inside to introduce his/her data through "start" hyperlink.

A set of real farms have been used as pilot operational cases to test the operability & easiness of implementation of the proposed data mining&accounting procedures and the robustness of the calculations (conversion factors, indicators) and further environmental impact assessments. The idea behind was to validate the results for the studied FMU in order to test if this procedure can be used at a larger farm scale (with many FMUs) and to assess the eventual insertion into the software of environmental sustainability indices. The tool has evolved and now, it is ready to introduce inputs from different crops.

6.3.3.3. Campaign data input for the water footprint, questionnaire and agronomic performance

To assess the Water Footprint assessment, the user needs to fill out an inventory enclosed in the Campaign Data related to different inventories related to different parameters: crop, irrigation, water balance, soil, energy and harvest. Every cell has a *help button* in order to clarify the required data, also several drop-down lists are included in several cells to facilitate data entry and some comments lead to understand every step by the user. Collected data need to be entered in the specific unit indicated in the unit cell.

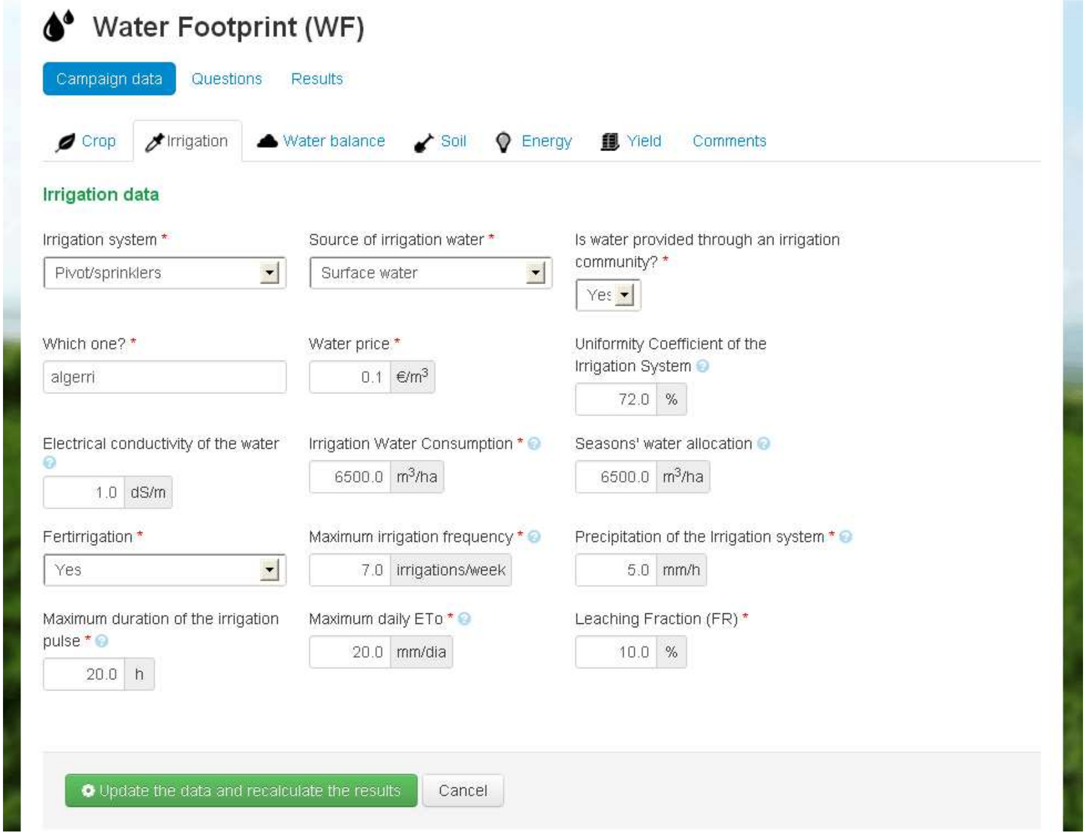
Figure 6.6 shows the first step "crop data" required by the user to do a water footprint assessment.



Year: 2011
Field area *: 20.0 ha
Type of crop *: Anual or sub-anu:
Crop *: 5
Cultivar *: 7000
Production goal: Grain
Previous seasons' end date *: 13/11/2012
Sowing/Planting date *: 15/01/2013
End of season *: 11/06/2013

Figure 6.6 Crop data inventory in campaign data in order to assess the water evaluation.

The second step is related to the irrigation data at farm and it is presented in Figure 6.7.

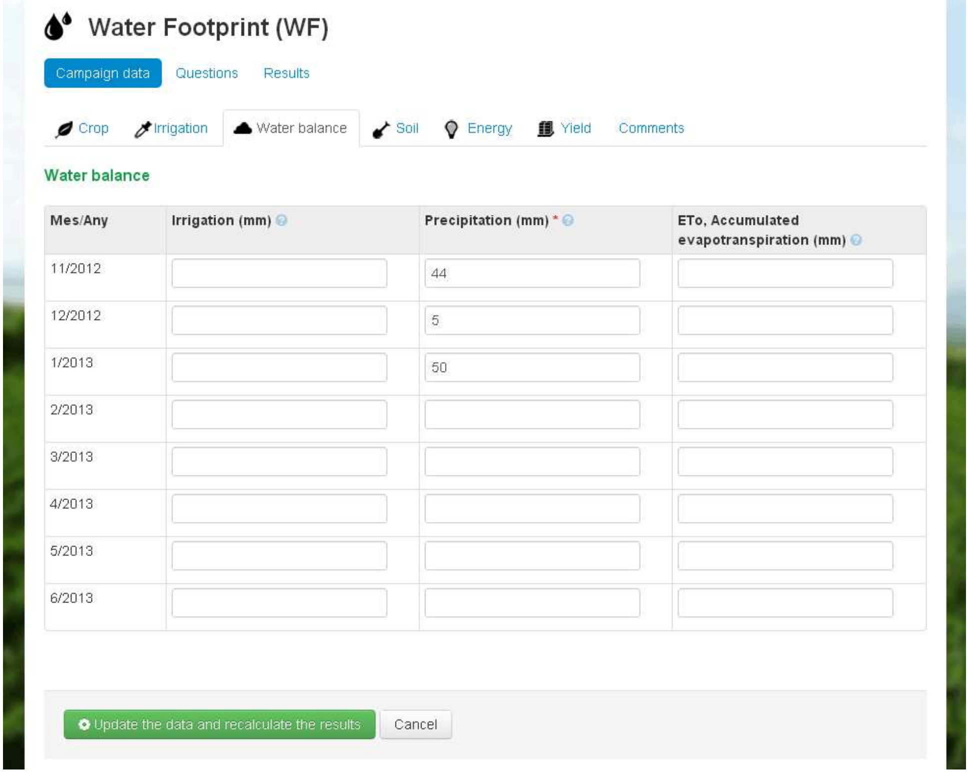


Irrigation data

Irrigation system * Pivot/sprinklers	Source of irrigation water * Surface water	Is water provided through an irrigation community? * Yes
Which one? * algerri	Water price * 0.1 €/m³	Uniformity Coefficient of the Irrigation System 72.0 %
Electrical conductivity of the water 1.0 dS/m	Irrigation Water Consumption * 6500.0 m³/ha	Seasons' water allocation 6500.0 m³/ha
Fertirrigation * Yes	Maximum irrigation frequency * 7.0 irrigations/week	Precipitation of the Irrigation system * 5.0 mm/h
Maximum duration of the irrigation pulse * 20.0 h	Maximum daily ETo * 20.0 mm/dia	Leaching Fraction (FR) * 10.0 %

Figure 6.7 Irrigation inventory in campaign data in order to assess the water evaluation.

The third step is the water balance and the data required is shown in Figure 6.8.

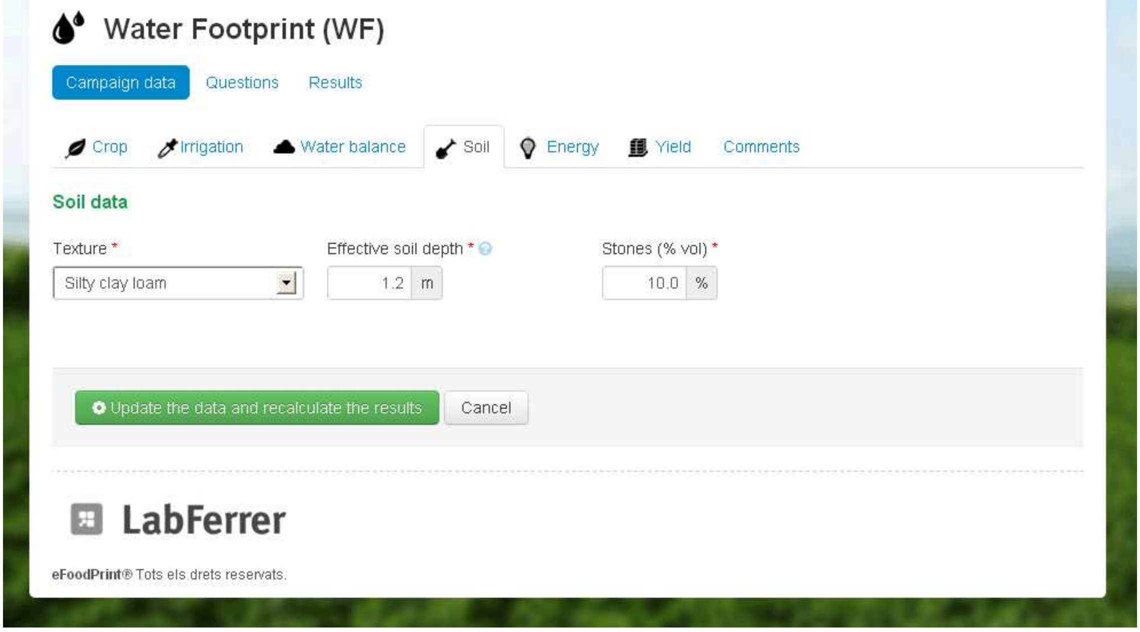


Water balance

Mes/Any	Irrigation (mm)	Precipitation (mm) *	ETo, Accumulated evapotranspiration (mm)
11/2012		44	
12/2012		5	
1/2013		50	
2/2013			
3/2013			
4/2013			
5/2013			
6/2013			

Figure 6.8 Water balance inventory in campaign data in order to assess the water evaluation.

The fourth step is the soil characteristics presented in Figure 6.9.



Soil data

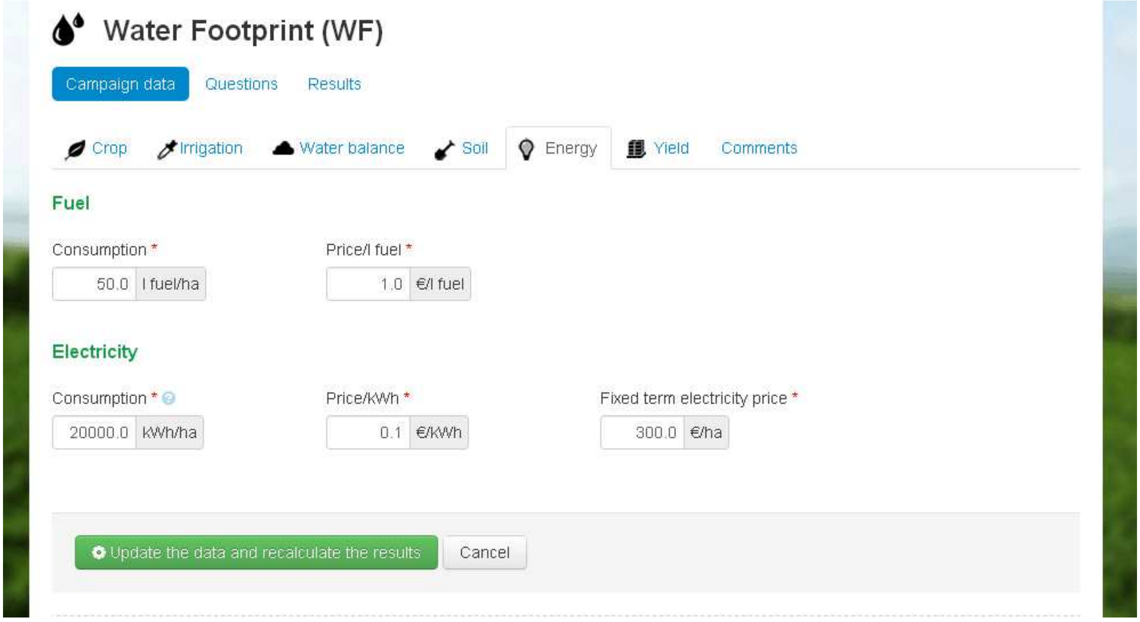
Texture *: Silty clay loam

Effective soil depth *: 1.2 m

Stones (% vol) *: 10.0 %

Figure 6.9 Soil inventory in campaign data in order to assess the water evaluation.

The fifth step is the energy resources used at farm (Figure 6.10.)



Fuel

Consumption *: 50.0 l fuel/ha

Price/l fuel *: 1.0 €/l fuel

Electricity

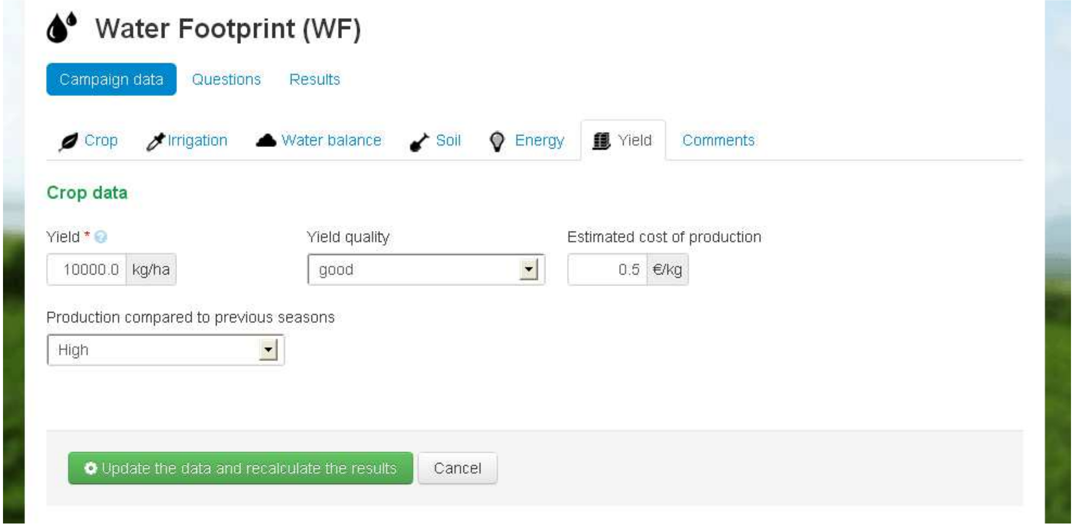
Consumption *: 20000.0 kWh/ha

Price/kWh *: 0.1 €/kWh

Fixed term electricity price *: 300.0 €/ha

Figure 6.10 Energy inventory in campaign data in order to assess the water evaluation.

The sixth step is the yield data used at farm (Figure 6.11)



Crop data

Yield *: 10000.0 kg/ha

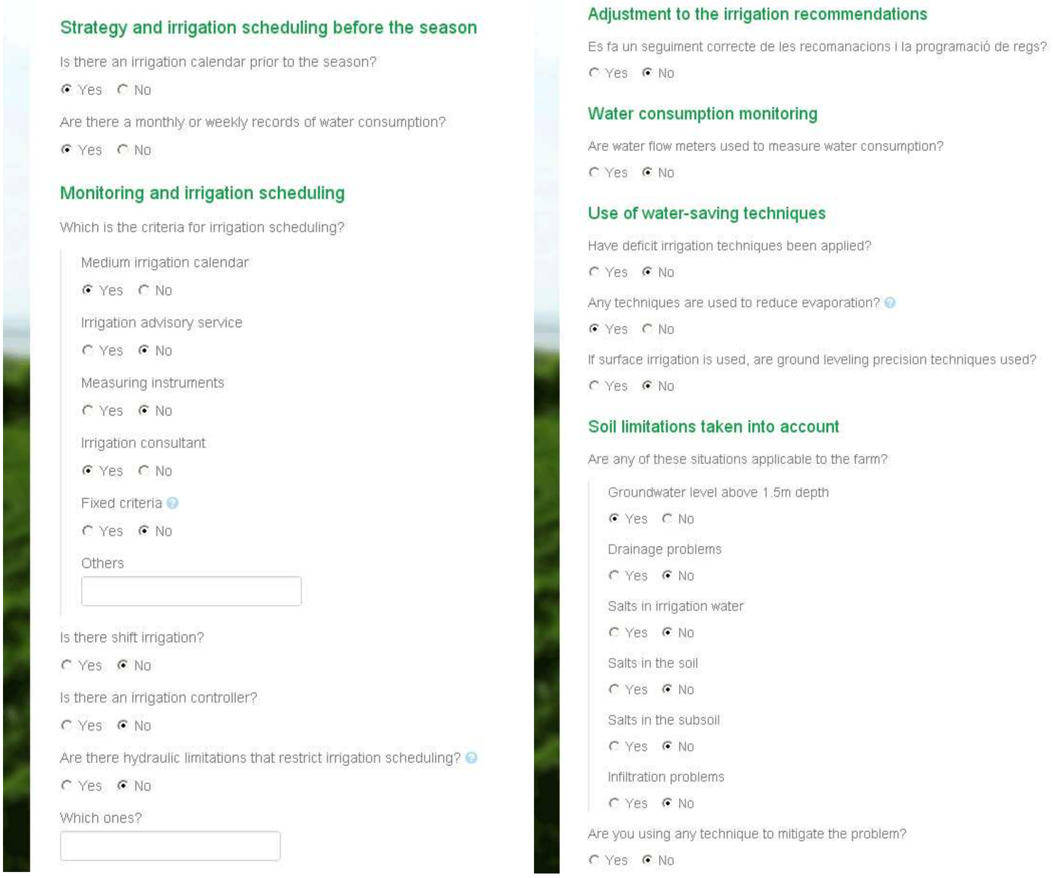
Yield quality: good

Estimated cost of production: 0.5 €/kg

Production compared to previous seasons: High

Figure 6.11 Harvest inventory in campaign data in order to assess the water assessment.

When each step in this section has been full out by the user, a qualitative questionnaire (answering yes or not) regarding water is asked (Figure 6.12) in order to evaluate the irrigation management previous to the campaign, the management and programming of irrigation and water consumption, and the adaptation of soils.



Strategy and irrigation scheduling before the season

Is there an irrigation calendar prior to the season?
● Yes ○ No

Are there a monthly or weekly records of water consumption?
● Yes ○ No

Monitoring and irrigation scheduling

Which is the criteria for irrigation scheduling?

Medium irrigation calendar
● Yes ○ No

Irrigation advisory service
○ Yes ● No

Measuring instruments
○ Yes ● No

Irrigation consultant
● Yes ○ No

Fixed criteria
○ Yes ● No

Others

Is there shift irrigation?
○ Yes ● No

Is there an irrigation controller?
○ Yes ● No

Are there hydraulic limitations that restrict irrigation scheduling?
○ Yes ● No

Which ones?

Adjustment to the irrigation recommendations

Es fa un seguiment correcte de les recomanacions i la programació de regs?
○ Yes ● No

Water consumption monitoring

Are water flow meters used to measure water consumption?
○ Yes ● No

Use of water-saving techniques

Have deficit irrigation techniques been applied?
○ Yes ● No

Any techniques are used to reduce evaporation?
● Yes ○ No

If surface irrigation is used, are ground leveling precision techniques used?
○ Yes ● No

Soil limitations taken into account

Are any of these situations applicable to the farm?

Groundwater level above 1.5m depth
● Yes ○ No

Drainage problems
○ Yes ● No

Salts in irrigation water
○ Yes ● No

Salts in the soil
○ Yes ● No

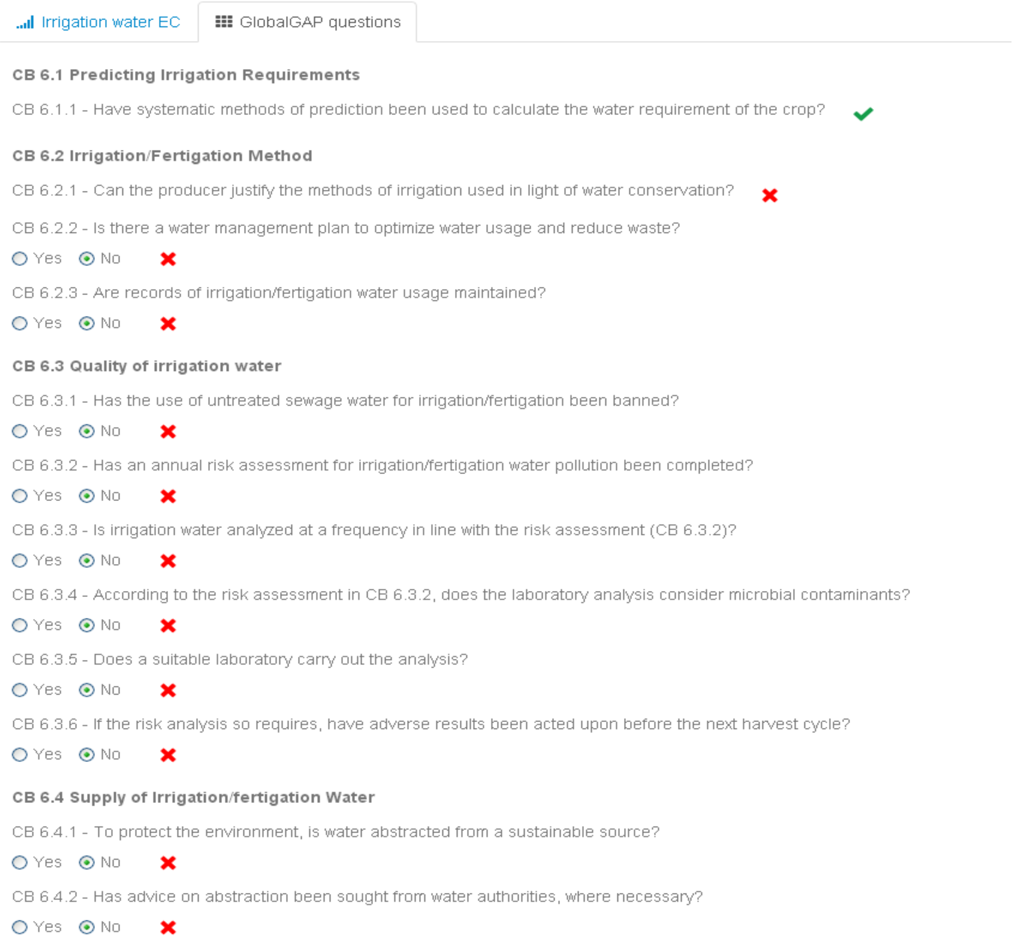
Salts in the subsoil
○ Yes ● No

Infiltration problems
○ Yes ● No

Are you using any technique to mitigate the problem?
○ Yes ● No

Figure 6.12 Water questionnaire in order to assess the water evaluation.

Additionally, the user can evaluate his/her water management in order to follow the global G.A.P. questionnaire (Figure 6.13).



CB 6.1 Predicting Irrigation Requirements

CB 6.1.1 - Have systematic methods of prediction been used to calculate the water requirement of the crop? ✓

CB 6.2 Irrigation/Fertigation Method

CB 6.2.1 - Can the producer justify the methods of irrigation used in light of water conservation? ✗

CB 6.2.2 - Is there a water management plan to optimize water usage and reduce waste?
○ Yes ● No ✗

CB 6.2.3 - Are records of irrigation/fertigation water usage maintained?
○ Yes ● No ✗

CB 6.3 Quality of irrigation water

CB 6.3.1 - Has the use of untreated sewage water for irrigation/fertigation been banned?
○ Yes ● No ✗

CB 6.3.2 - Has an annual risk assessment for irrigation/fertigation water pollution been completed?
○ Yes ● No ✗

CB 6.3.3 - Is irrigation water analyzed at a frequency in line with the risk assessment (CB 6.3.2)?
○ Yes ● No ✗

CB 6.3.4 - According to the risk assessment in CB 6.3.2, does the laboratory analysis consider microbial contaminants?
○ Yes ● No ✗

CB 6.3.5 - Does a suitable laboratory carry out the analysis?
○ Yes ● No ✗

CB 6.3.6 - If the risk analysis so requires, have adverse results been acted upon before the next harvest cycle?
○ Yes ● No ✗

CB 6.4 Supply of Irrigation/fertigation Water

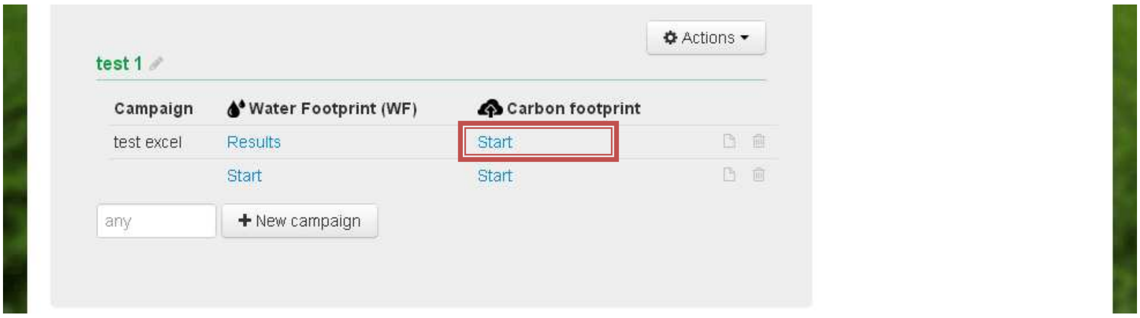
CB 6.4.1 - To protect the environment, is water abstracted from a sustainable source?
○ Yes ● No ✗

CB 6.4.2 - Has advice on abstraction been sought from water authorities, where necessary?
○ Yes ● No ✗

Figure 6.13 Global G.A.P. questionnaire in order to assess the water evaluation.

6.3.3.4.Campaign data input for the Carbon Footprint

Once the water assessment has been carried out, the user can follow with the carbon assessment through link "start" in Home page (Figure 6.14).



Campaign	Water Footprint (WF)	Carbon footprint
test excel	Results	Start
	Start	Start

Figure 6.14 Starting carbon assessment in the eFoodPrint® home page.

Given that constructing a full LCA is a complex task due to huge amount of data required and methodological needed requirements (Mourad et al. 2007), the carbon inventory is structured in several understandable topics enclosed in Campaign Data: users enter primary data related to crop, machinery including electricity and fuel consumption, fertilizers, treatments and transport during the whole life-cycle at farm gate.

The input data slides are questionnaires for each parameter evolved in the life-cycle emissions where users are asked to fill in cells with their campaign data to model the production system under study. As in water assessment, some confused cell has a *help link* in order to clarify the required data, also several drop-down lists are included in several cells to facilitate data entry and some comments lead to understand every step by the user. Collected data need to be entered in the specific unit indicated in the unit cell.

Crop data

General data as kind of crop, farm surface, commercial production and density of plantation are required in the first stage. Figure 6.15 shows the first step “crop data” required by the user to do a global warming assessment.

Crop and farm data.

Type of crop * Stable

Crop * nectarina

Density of plantation * 800.0 pl/m^2

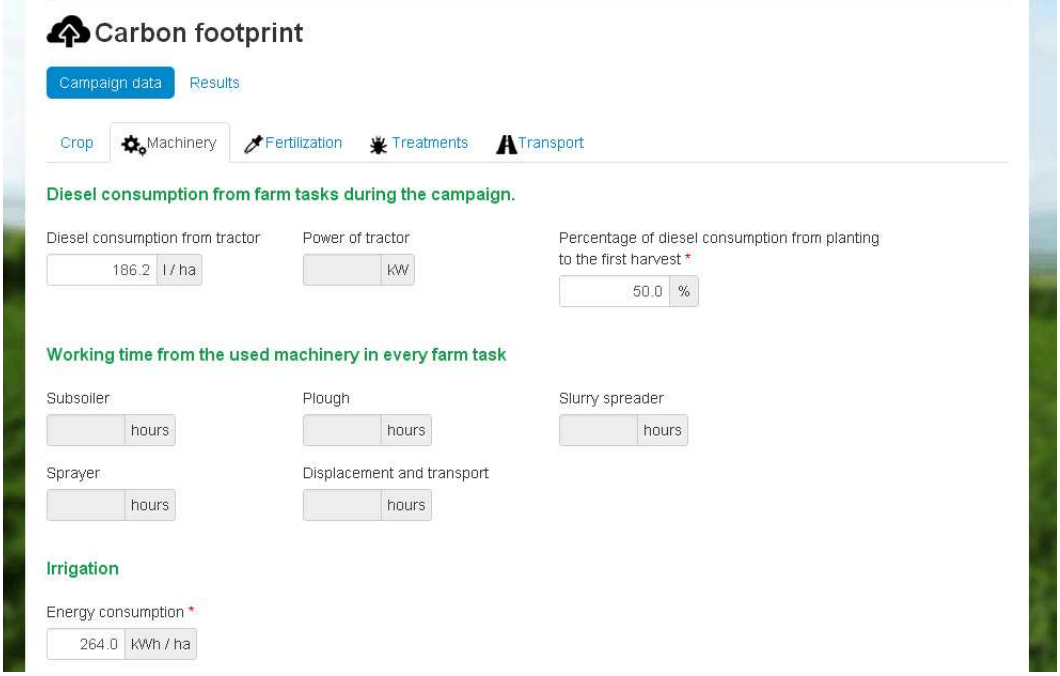
Yield * 47137.0 kg / ha

Farm area * 12.5 ha

Figure 6.15 Crop data inventory to evaluate the carbon dioxide emissions during whole life-cycle.

Machinery

The second step is addressed to assess the carbon dioxide emissions from the diesel consumption in farm tasks and energy consumption in irrigation pumping (Figure 6.16).The user needs to introduce the energy consumption of pumping irrigation in farm. The contribution of electricity consumption can differ depending on the electricity production mix of the country where the production system is analyzed. Hence, Spanish mix electric has been used in order to calculate the environmental impact of electricity consumption. Diesel consumption and power of tractor is also required to enter in. Therefore, machinery's time operation is asked for several farm tasks.



Diesel consumption from farm tasks during the campaign.

Diesel consumption from tractor: 186.2 l / ha
Power of tractor: KW
Percentage of diesel consumption from planting to the first harvest *: 50.0 %

Working time from the used machinery in every farm task

Subsoiler: hours
Plough: hours
Slurry spreader: hours
Sprayer: hours
Displacement and transport: hours

Irrigation

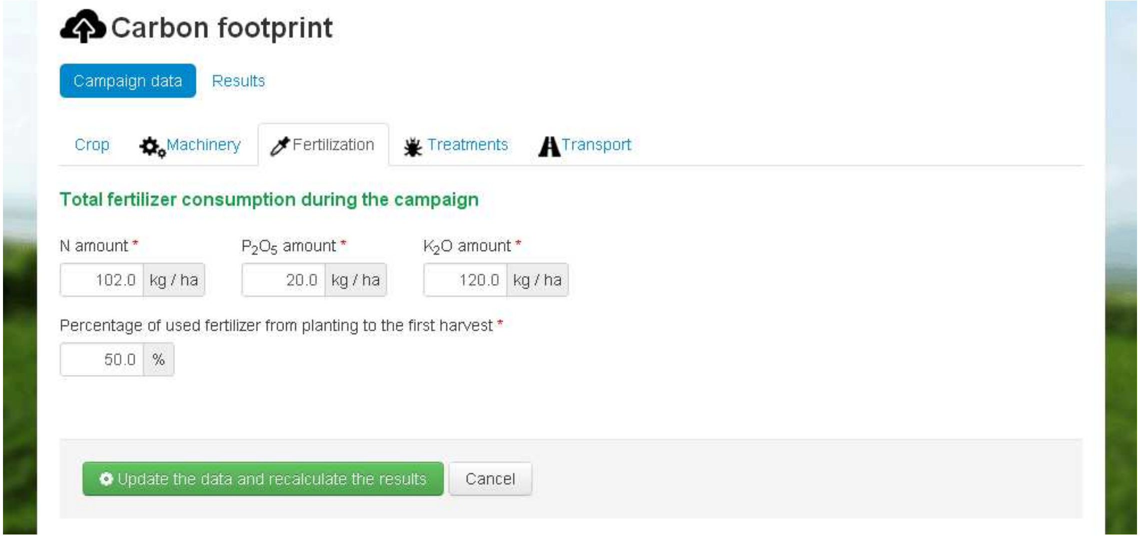
Energy consumption *: 264.0 KWh / ha

Figure 6.16 Diesel and energy inventory in farm tasks to evaluate the carbon dioxide emissions during whole life-cycle.

Fertilization

The environmental performance of fertilizers is an important issue in agricultural production systems, because of the emissions released during their production and N_2O emissions during application. In order to simplify the requirements for the environmental assessment of fertilizers, users are asked to enter the total amount of

N, P_2O_5 and K_2O on the carbon questionnaire (Figure 6.17). Therefore, the production of all the fertilizers applied to the crop is included. Datasets for emissions from the production and application of specific fertilizers were taken from the Ecoinvent database v.2.2 (Ecoinvent, 2010).



Total fertilizer consumption during the campaign

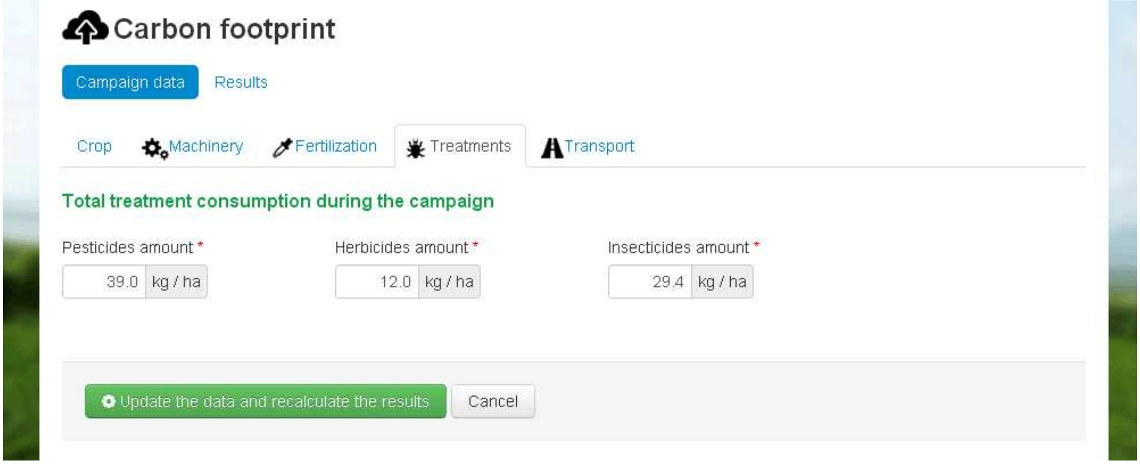
N amount * 102.0 kg / ha
P_2O_5 amount * 20.0 kg / ha
K_2O amount * 120.0 kg / ha

Percentage of used fertilizer from planting to the first harvest * 50.0 %

Figure 6.17 Fertilization inventory in farm to evaluate the carbon dioxide emissions during whole life-cycle.

Treatments

The user is asked to enter the total amount of fungicides, insecticides and herbicides consumed during the campaign (Figure 6.18). In addition, the production of all the pesticides applied to the crop is included, as well as the machinery for their application.



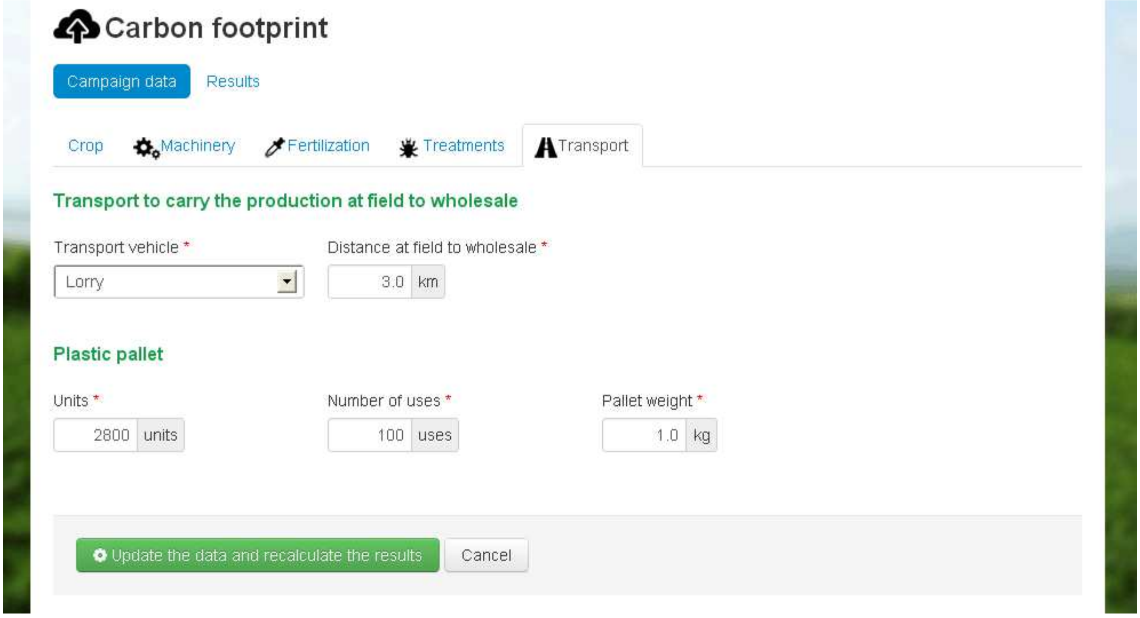
Total treatment consumption during the campaign

Pesticides amount *	Herbicides amount *	Insecticides amount *
39.0 kg / ha	12.0 kg / ha	29.4 kg / ha

Figure 6.18 Treatment inventory (pesticides, insecticides and fungicides) in farm tasks to evaluate the carbon dioxide emissions during whole life-cycle.

Transport

The contribution of transport in the total environmental impact can change depending on two factors, a) the distance from origin to destination and b) the types of road transport that the user selects, which could lead to make considerable variability in the emissions they release into the atmosphere. Several datasets on types of lorries are available in the Ecoinvent database. However, in order to simplify the calculator according with the stakeholders' consent, two lorries processes have been selected for the application, "van 3.5-16t" and "lorry 16t-24t" and additionally, a tractor. In addition, pallets to carry the production from farm to wholesale, the life use (number of uses) and the weight of these, are data asked to the user (Figure 6.19).



Transport to carry the production at field to wholesale

Transport vehicle *: Lorry
Distance at field to wholesale *: 3.0 km

Plastic pallet

Units *: 2800 units
Number of uses *: 100 uses
Pallet weight *: 1.0 kg

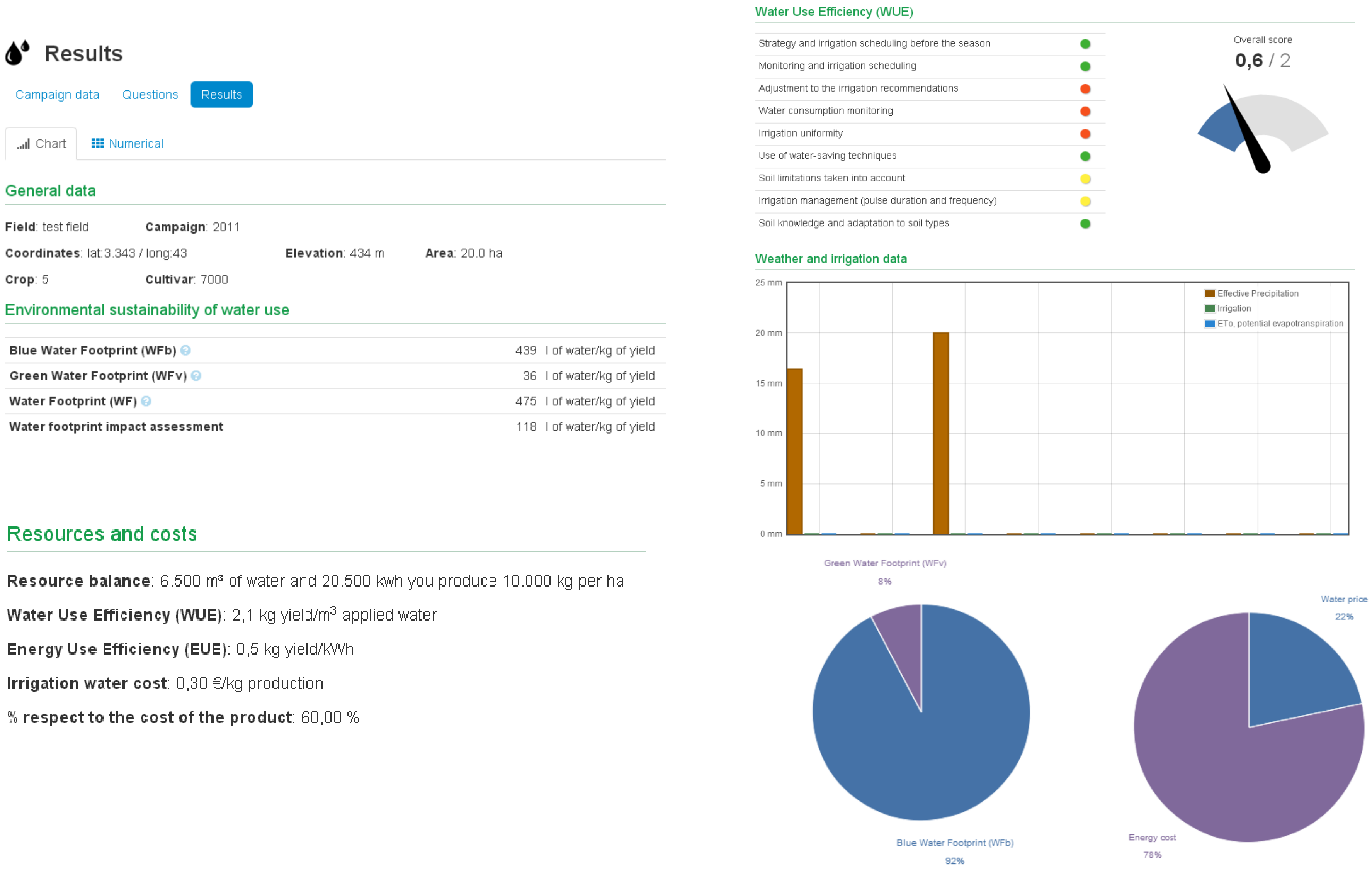
Figure 6.19 Transport and packaging to evaluate the carbon dioxide emissions during whole life-cycle.

6.3.3.5. Structure of the output reports:

The final step is to obtain the results from previous assessments. Each analysis is divided in two blocks, being the first one, graphical and the second, numerical.

Water Footprint Output

Figure 6.20 present the graphical results of water footprint assessment which are structured in 5 subsections: general data, environmental sustainability of water use, resources and costs, efficiency of water use, climatic data and irrigation.



Results

General data

Field: test field **Campaign**: 2011

Coordinates: lat:3.343 / long:43 **Elevation**: 434 m **Area**: 20.0 ha

Crop: 5 **Cultivar**: 7000

Environmental sustainability of water use

Blue Water Footprint (WFb)	439	l of water/kg of yield
Green Water Footprint (WFv)	36	l of water/kg of yield
Water Footprint (WF)	475	l of water/kg of yield
Water footprint impact assessment	118	l of water/kg of yield

Resources and costs

Resource balance: 6.500 m³ of water and 20.500 kwh you produce 10.000 kg per ha

Water Use Efficiency (WUE): 2,1 kg yield/m^3 applied water

Energy Use Efficiency (EUE): 0,5 kg yield/kWh

Irrigation water cost: 0,30 €/kg production

% **respect to the cost of the product**: 60,00 %

Water Use Efficiency (WUE)

Strategy and irrigation scheduling before the season	●
Monitoring and irrigation scheduling	●
Adjustment to the irrigation recommendations	●
Water consumption monitoring	●
Irrigation uniformity	●
Use of water-saving techniques	●
Soil limitations taken into account	●
Irrigation management (pulse duration and frequency)	●
Soil knowledge and adaptation to soil types	●

Overall score
0,6 / 2

Weather and irrigation data

Figure 6.20 Graphical results in order to assess the water assessment.

Numerical data is structured in three subsections: general data, environmental and agronomical sustainability of water use (Figure 6.21).

General data

Field: test field **Campaign**: 2011
Coordinates: lat:3.343 / long:43 **Elevation**: 434 m **Area**: 20.0 ha
Crop: 5 **Cultivar**: 7000

ENVIRONMENTAL SUSTAINABILITY OF WATER USE

Blue Water Footprint (WFb)	439	l of water/kg of yield
Green Water Footprint (WFv)	36	l of water/kg of yield
Water Footprint (WF)	475	l of water/kg of yield
Water footprint impact assessment	118	l of water/kg of yield

AGRONOMICAL SUSTAINABILITY OF WATER USE

Yield	10.000	kg/ha
Irrigation Water Consumption	6.500	m^3/ha
Energy Consumption	20.500	KWh
Water Use Efficiency (WUE)	2,10	kg yield/m^3 applied water
Energy Use Efficiency (EUE)	0	kg yield/kWh
Water price	0,07	€/kg production
Energy cost	0,24	€/kg production
Irrigation water cost	0,30	€/kg production
% respect to the cost of the product	60,00	%
Application Efficiency (Ea)	75	%
Irrigation Uniformity (CU)	72	%
Irrigation Water Consumption	650	mm
Accumulated Effective Rainfall (Pe)	36	mm
Accumulated evapotranspiration (ETo)	0	mm
Estimated soil water holding capacity	184	mm
Strategy and irrigation scheduling before the season	adequate	
Monitoring and irrigation scheduling	adequate	
Adjustment to the irrigation recommendations	inadequate	
Water consumption monitoring	inadequate	
Irrigation uniformity	inadequate	
Use of water-saving techniques	yes	
Soil limitations taken into account	improvable	
Irrigation management (pulse duration and frequency)	improvable	
Soil knowledge and adaptation to soil types	adequate	

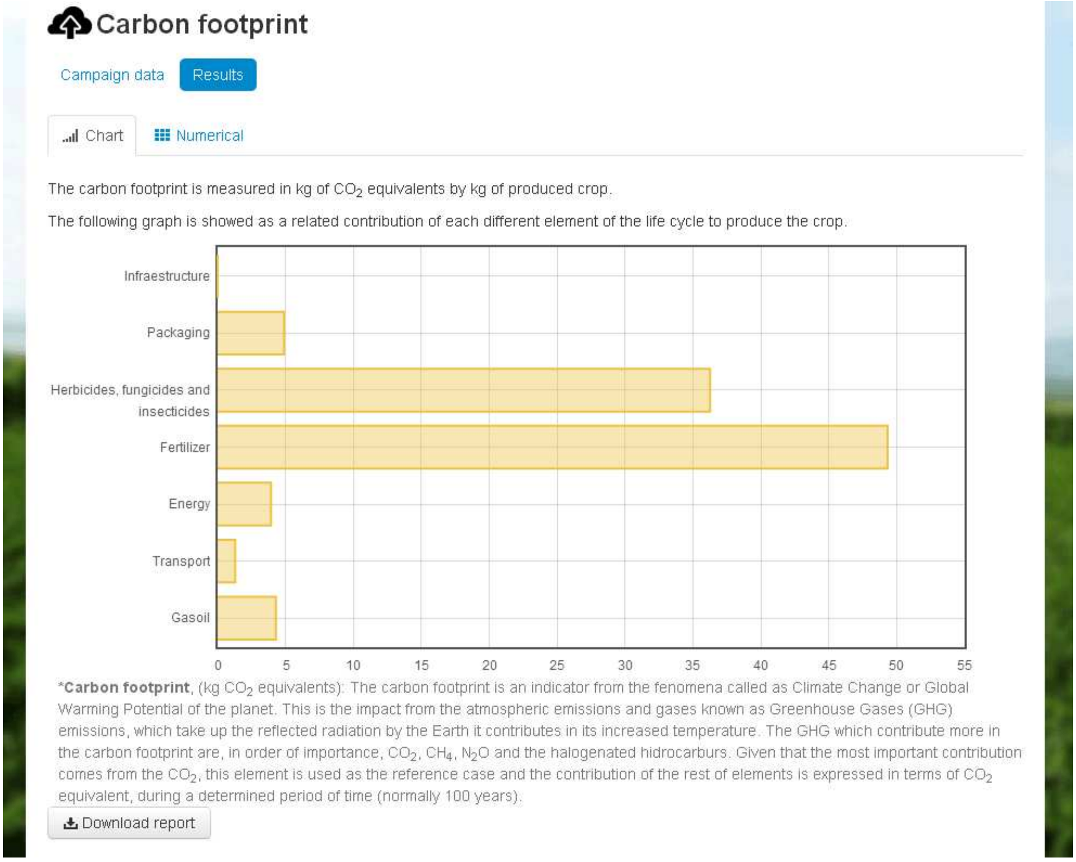
COMMENTS

Figure 6.21 Numerical results in order to assess the water evaluation.

Carbon Footprint output

Results are presented as on graph *charts* (Figure 6.22) to facilitate user comprehension and as absolute values in *numerical* sheet (Figure 6.23) available to be export in Excel format.

For instance, the results from fertilizer assessment showed that emissions from the production of them made higher contributions in Global Warming Potential (GWP). In the other hand, emissions from pesticides (caused by the production of insecticides, fungicides and herbicides) and transport were a minor burden in the total crop production system.

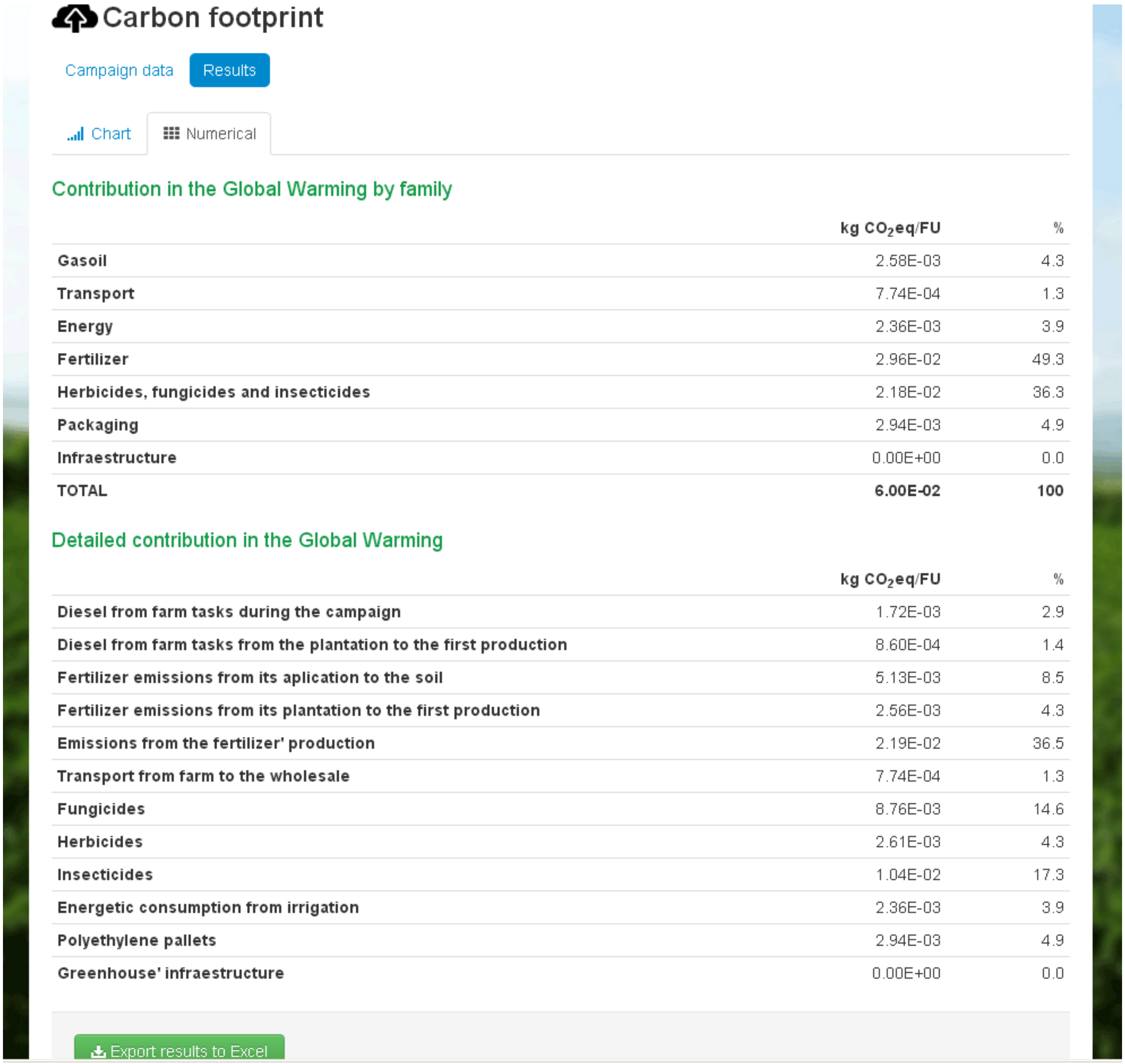


The carbon footprint is measured in kg of CO_2 equivalents by kg of produced crop.

The following graph is showed as a related contribution of each different element of the life cycle to produce the crop.

***Carbon footprint**, (kg CO_2 equivalents): The carbon footprint is an indicator from the fenomena called as Climate Change or Global Warming Potential of the planet. This is the impact from the atmospheric emissions and gases known as Greenhouse Gases (GHG) emissions, which take up the reflected radiation by the Earth it contributes in its increased temperature. The GHG which contribute more in the carbon footprint are, in order of importance, CO_2, CH_4, N_2O and the halogenated hidrocarburs. Given that the most important contribution comes from the CO_2, this element is used as the reference case and the contribution of the rest of elements is expressed in terms of CO_2 equivalent, during a determined period of time (normally 100 years).

Figure 6.22 Graphical results from the carbon dioxide emissions during whole life-cycle.



Contribution in the Global Warming by family

	kg CO_2eq/FU	%
Gasoil	2.58E-03	4.3
Transport	7.74E-04	1.3
Energy	2.36E-03	3.9
Fertilizer	2.96E-02	49.3
Herbicides, fungicides and insecticides	2.18E-02	36.3
Packaging	2.94E-03	4.9
Infraestructure	0.00E+00	0.0
TOTAL	**6.00E-02**	**100**

Detailed contribution in the Global Warming

	kg CO_2eq/FU	%
Diesel from farm tasks during the campaign	1.72E-03	2.9
Diesel from farm tasks from the plantation to the first production	8.60E-04	1.4
Fertilizer emissions from its aplication to the soil	5.13E-03	8.5
Fertilizer emissions from its plantation to the first production	2.56E-03	4.3
Emissions from the fertilizer' production	2.19E-02	36.5
Transport from farm to the wholesale	7.74E-04	1.3
Fungicides	8.76E-03	14.6
Herbicides	2.61E-03	4.3
Insecticides	1.04E-02	17.3
Energetic consumption from irrigation	2.36E-03	3.9
Polyethylene pallets	2.94E-03	4.9
Greenhouse' infraestructure	0.00E+00	0.0

Figure 6.23 Numerical results from the carbon dioxide emissions during whole life-cycle.

6.3.4. Example of the output reports

Communicating the calculation and interpretation of results to non-specialists is an added difficulty of the Life Cycle Assessment, carbon and water footprints. The total results of the crop production, based on user data are shown on the *Download Report* (report' link can be founded at the bottom of graphical results section). The results of the production system by stage are also provided on. In order to understand the Global Warming category impact for non-familiar users, a brief definition of this impact category is included in carbon footprint report so users can acquire knowledge about the damage that can be caused to the environment. Figure 6.24 displays the water footprint report and figure 6.25 shows the carbon footprint report.

General data

Field: test field **Campaign**: 2011

Coordinates: lat:3.343 / long:43 **Elevation**: 434 m **Area**: 20.0 h

Crop: 5 **Cultivar**: 7000

Environmental sustainability of water use

	439 l of water/kg of yield
Blue Water Footprint (WFb)	36 l of water/kg of yield
Green Water Footprint (WFv)	475 l of water/kg of yield
Water Footprint (WF)	118 l of water/kg of yield
Water footprint impact assessment	

Resources and costs

Resource balance: 6.500 m³ of water and 20.500 kwh you produce 10.000 kg per ha

Water Use Efficiency (WUE): 2,1 kg yield/m³ applied water

Energy Use Efficiency (EUE): 0,5 kg yield/kWh

Irrigation water cost: 0,30 €/kg production

% respect to the cost of the product: 60,00 %

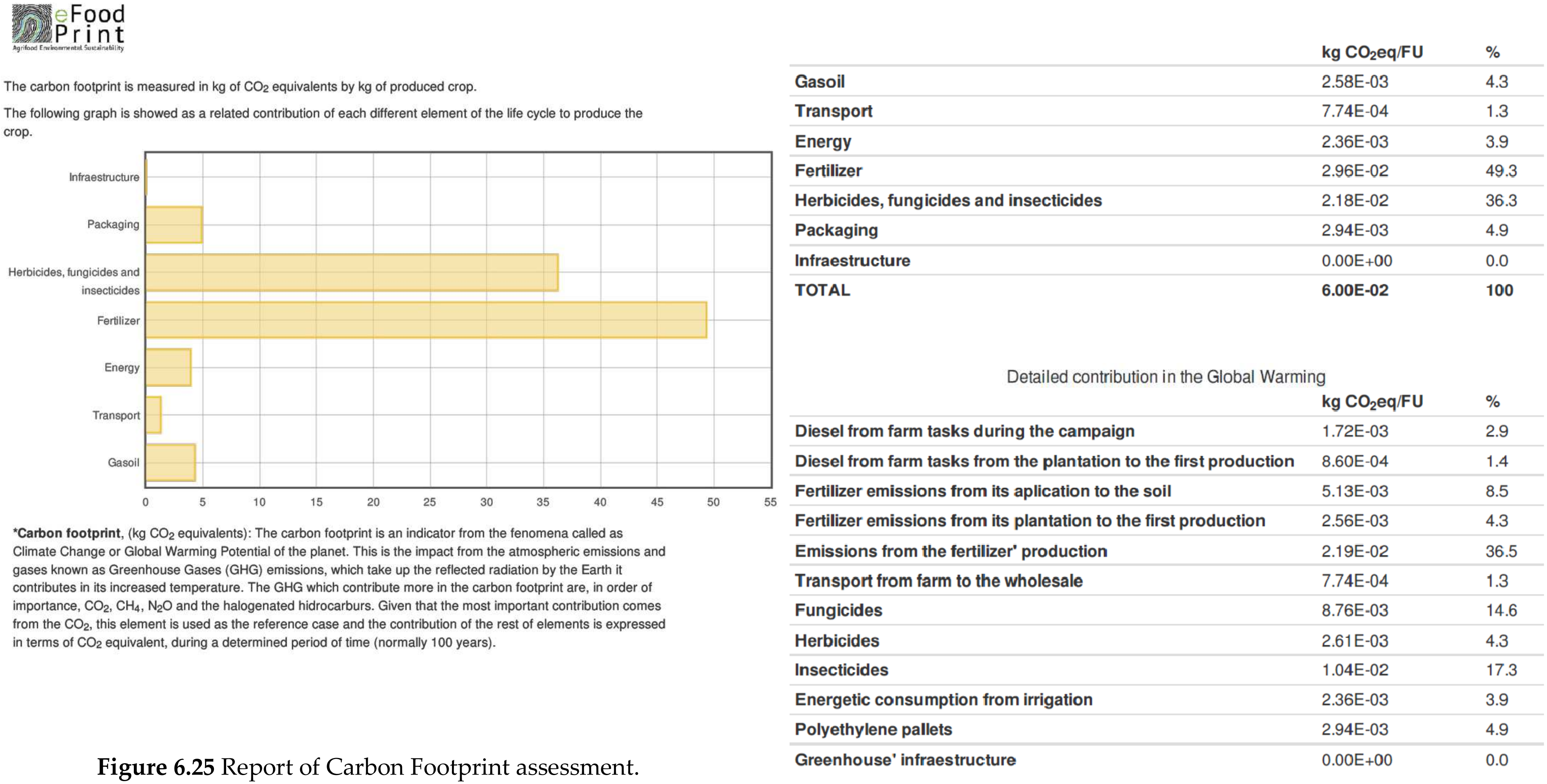
Water Use Efficiency (WUE)

Strategy and irrigation scheduling before the season	●
Monitoring and irrigation scheduling	●
Adjustment to the irrigation recommendations	●
Water consumption monitoring	●
Irrigation uniformity	●
Use of water-saving techniques	●
Soil limitations taken into account	●
Irrigation management (pulse duration and frequency)	●
Soil knowledge and adaptation to soil types	●

Overall score
0,6 / 2

Weather and irrigation data

Figure 6.24 Report of Water footprint.



The carbon footprint is measured in kg of CO_2 equivalents by kg of produced crop.

The following graph is showed as a related contribution of each different element of the life cycle to produce the crop.

***Carbon footprint**, (kg CO_2 equivalents): The carbon footprint is an indicator from the fenomena called as Climate Change or Global Warming Potential of the planet. This is the impact from the atmospheric emissions and gases known as Greenhouse Gases (GHG) emissions, which take up the reflected radiation by the Earth it contributes in its increased temperature. The GHG which contribute more in the carbon footprint are, in order of importance, CO_2, CH_4, N_2O and the halogenated hidrocarburs. Given that the most important contribution comes from the CO_2, this element is used as the reference case and the contribution of the rest of elements is expressed in terms of CO_2 equivalent, during a determined period of time (normally 100 years).

	kg CO_2eq/FU	**%**
Gasoil	2.58E-03	4.3
Transport	7.74E-04	1.3
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Packaging	2.94E-03	4.9
Infraestructure	0.00E+00	0.0
TOTAL	**6.00E-02**	**100**

Detailed contribution in the Global Warming

	kg CO_2eq/FU	**%**
Diesel from farm tasks during the campaign	1.72E-03	2.9
Diesel from farm tasks from the plantation to the first production	8.60E-04	1.4
Fertilizer emissions from its aplication to the soil	5.13E-03	8.5
Fertilizer emissions from its plantation to the first production	2.56E-03	4.3
Emissions from the fertilizer' production	2.19E-02	36.5
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Fungicides	8.76E-03	14.6
Herbicides	2.61E-03	4.3
Insecticides	1.04E-02	17.3
Energetic consumption from irrigation	2.36E-03	3.9
Polyethylene pallets	2.94E-03	4.9
Greenhouse' infraestructure	0.00E+00	0.0

Figure 6.25 Report of Carbon Footprint assessment.

6.4. Discussion, limitations and further research

The present chapter explains how the eFoodPrint® calculator is structured, as to be used by all stakeholders (growers, producers, advisers, consultants and planners) in agricultural production systems (fruits, vegetables, grain and forages) as a support tool to take decisions (decision making) and reduce the environmental impacts of farm crops (improving environmental concern). The calculation of environmental impacts is provided by including appropriate formulas for two main assessments, both water and carbon footprint in agriculture.

The provided results in eFoodPrint® allow establishing an initial situation of the farm management during an agronomical campaign. Therefore, it also is expected to be really helpful to users to simulate the own environmental performance of farms for following campaigns with prevision and so on, choosing the most appropriate decisions (e.g. reduce the amount of fertilizer).

This tool is different from other calculators on the market because it combines agronomical and water indices to assess the water management and also, it gives insight into environmental damage by providing results in Global Warming Impact category as well as risky and security damage by global G.A.P. questionnaire.

In addition, while environmental impact of water consumption on available freshwater resources, ecosystem quality and human health is forgotten in most LCA studies (Núñez, 2012), this calculator includes this, besides the water use and consumption' evaluation following models that take into account models based on the spatial differentiation of impacts at the watershed level (Pfister et al. 2009). Moreover, eFoodPrint® was finalized, validated and subsequently tested by a group of agriculture support technicians.

Although the proposal calculator has several advantages respect others already published (e.g. Cool Farm Tool, Fieldtomarket), the tool also show some limitations. Hence, several lines for future research could become needed in order to improve the calculator:

- This tool could be also improved by including practitioner's feedback. For that, tool appraisal is available in the calculator so users' opinions will be valuable given that the tool can be adapted to their needs.
- This calculator does not include an economic and social appraisal, but a Business Plan has been developed by some expertise people in marketing field from the MUSA project in order to justify the viability of the product in the market.
- Although this tool does not include an assessment of a greenhouse protected crops, an environmental assessment of greenhouse production system can be carried out using a separate tool developed for this purpose by Torrellas et al. (2013). The tool is available on the EUPHOROS website (EUPHOROS, 2008-2012).
- Water footprint assessment has not been assessed by three water colours (green, blue, grey). Only green and blue water footprints have been carried out in this calculator being grey water footprint out of scope.
- There is the possibility to obtain some important input variables from on-site monitoring. For instance, continuous on-site recording of soil water content in the soil profile and precipitation, can give a very realistic value for the Effective Precipitation (Pe).
- The environmental assessment is done at a farm level. For water use, the SWI is incorporated as related to water availability in the watershed. However, up-scaling is not considered and a good decision at a farm level cannot be the adequate at a larger scale, from an environmental point of view. Therefore, a more up-scaled ecological & economical domain should be further considered.

For above mentioned, eFoodPrint® is a robustness environmental tool but it could be improved for future research in farm production systems by including the limitations discussed above.

Part IV

Scientific Development and Application of Water Use in Agriculture

Chapter 7

Biodiversity impacts from salinity increase in a coastal wetland

María José Amores, Francesca Verones, Catherine Raptis,

Ronnie Juraske, Stephan Pfister, Franziska Stoessel, Assumpció Antón,

Francesc Castells, Stefanie Hellweg

Chapter 7 is based on the following paper:

Amores MJ, Verones F, Raptis C, Juraske R, Pfister S, Stoessel F, Antón A, Castells F, Hellweg S. Biodiversity impacts from salinity increase in a coastal wetland. Environmental Science and Technology, dx.doi.org/10.1021/es3045423.

Abstract

A Life Cycle Impact Assessment (LCIA) method was developed to evaluate the environmental impacts associated with salinity on biodiversity in a Spanish coastal wetland. The developed characterization factor consists of a fate and an effect factor and equals $3.16\text{x}10^{-1} \pm 1.84\text{x}10^{-1}$ PAF·m^3·yr·m^{-3} (PAF: Potentially Affected Fraction of species) indicating a “potential loss of 0.32 m^3 ecosystem” for a water consumption rate of 1 m^3yr^{-1}. The fate factor was calculated from seasonal water balances of the wetland Albufera de Adra (Figure 7.0). The effect factor was obtained from the fitted curve of the potentially affected fraction of native wetland species due to salinity and can be applied to other wetlands with similar species composition. In order to test the applicability of the characterization factor, an assessment of water consumption of greenhouse crops in the area was conducted as a case study. Results converted into ecosystem quality damage using the ReCiPe method were compared to other categories. While tomatoes are responsible for up to 30 % of the impact of increased salinity due to water consumption on ecosystem quality in the studied area, melons have the largest impact per tonne produced.

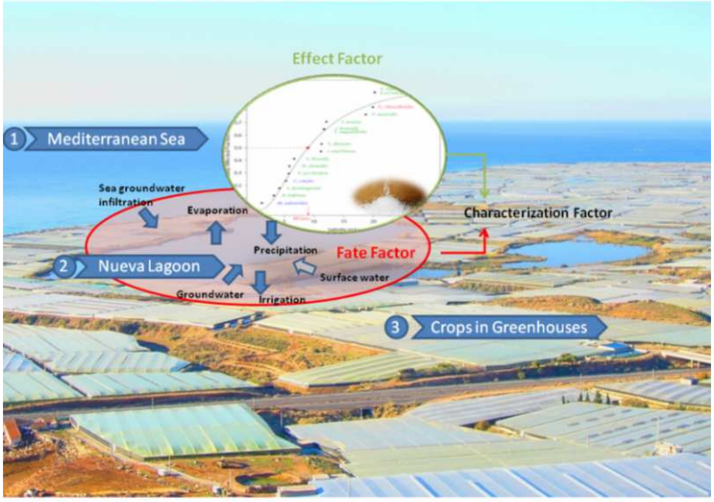

Figure 7.0. Greenhouses close to Albufera de Adra wetland in Adra (Almería-Spain).

7.1. Introduction

In 2006 the Food and Agricultural Organization of the United Nations (FAO) estimated the global water withdrawals to be 3,830 $km^3 \cdot yr^{-1}$, of which 70% was used for agriculture (FAO 2006a). Additionally, water availability and use related to agricultural production are faced with important challenges such as growing population, climate change and changing dietary patterns. It is thus fundamental to assess impacts and changes in agriculture, in order to respond to challenges in the near future.

Wetlands are important and vulnerable ecosystems. More than 50% of all wetlands worldwide were destroyed during the twentieth century, more than 60% in Spain and Greece, and more than 70% in Italy (Hollis 1992), mostly due to agricultural drainage (OECD 1996). During the last decades, the use of greenhouses instead of traditional farming systems has often been accompanied by additional groundwater withdrawal for irrigation (Rodríguez et al. 2011). In coastal areas, over-pumping of aquifers leads to sea water intrusions, thus increasing the salinity in aquifers. At the same time, coastal wetlands, where fresh water and salt water are often mixed, are among the most productive, valuable, and yet most threatened ecosystems in the world (Agardy et al. 2005).

Coastal wetlands in arid and semi-arid zones experience periods of increasing salinity as a consequence of high evaporative conditions, variability of inflows, their proximity to the sea but also due to impacts of human pressures (Jolly et al. 2008), such as overpumping of aquifers. Due to the inflow of salty water, coastal wetlands might experience an increase in salinity, which could potentially be detrimental for the wetland's specific ecological system.

With an increasing awareness of the value and importance of wetlands, fostered by the Ramsar Convention (Ramsar 1971), numerous coastal wetlands have been designated as wetlands of international importance. Still, environmental impacts due to agricultural practices and dependencies upon wetlands are becoming increasingly significant (Molden et al. 2011). Hence, balancing wetland conservation and wise water use, as well as assessing the prevalent impacts is important for the preservation of the remaining wetlands.

Life Cycle Assessment (LCA) is a method for evaluating the total environmental impact throughout the life cycle of a product or process (ISO 14040, 2006a). The ISO 14044 standard defines Life Cycle Impact Assessment (LCIA) as the phase of LCA aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts of a product system with the purpose of interpreting the life cycle emissions and resource consumption inventory in terms of indicators for the three Areas of Protection (resources, human health, ecosystem quality) (JRC, 2010). Several methods have been developed for assessing damages from water use on ecosystems, such as the decrease of terrestrial biodiversity due to freshwater consumption (Pfister et al. 2009), the disappearance of terrestrial plant species due to a change in extraction of groundwater (van Zelm et al. 2011), and the effects on freshwater fish species from water consumption in rivers (Hanafiah et al. 2011). We understand water consumption in this case study as general water consumption in the wetland (evapotranspiration, product integration, and discharge into the sea or into areas outside the wetland). Recently, a case study dealing with the effects of changes in water temperature and salinity on freshwater molluscs in the river Rhine has been published (Verbrugge et al. 2012). For wetlands in particular, a case study in the coastal arid area of Peru concerning the local plant biodiversity impacts of agricultural water use has been published (Verones et al. 2012). However, so far no LCIA methodology has taken into account salinity impacts in coastal wetlands.

In order to develop a methodology for salinity impacts, we selected a coastal wetland in Spain called "Albufera de Adra" as an exploratory case study in order to learn (since water-related impacts are quite complex) how to include the impacts from salinity increases due to water consumption. It is located in a semi-arid region in Almería (South-East of Spain), where agricultural activities require substantial irrigation and areas with native vegetation and fauna are restricted to small patches and wetlands (Molina 2006). The aims of this study were (a) to develop a characterization factor (Cf) in terms of potentially affected fraction of species (PAF) (Pennington et al. 2003) for salinity impacts based on a new effect factor and a locally specific new fate factor, and (b) to apply it to a local case study and compare the impact of salinity with commonly used ecosystem quality impact categories.

7.2. Materials and Methods

7.2.1. Description of the wetland Albufera de Adra

The case study area is located in a semi-arid and mountainous area of South-Eastern Spain, in the province of Almería (N36º 45′ 16′′ / W 2º 57′ 0′′). Albufera de Adra contains two lagoons which together occupy 36.4 ha. Nueva lagoon is situated closer to the sea than Honda lagoon. The wetland is located at the south-eastern edge of the Adra River Delta (Paracuellos 2009) area close to the Mediterranean Sea (Figure 7.1). Only Honda lagoon is recharged with surface water (ephemeral streams) while Nueva lagoon is predominantly fed by groundwater (Rodriguez et al. 2011). From 2003 to 2010, modifications in the surrounding agricultural practices led to differences in the hydrological dynamics in both lagoons (Rodriguez et al. 2011; Rodriguez 2007). Specifically, an extension of irrigated areas and more efficient irrigation techniques have resulted in a reduced natural and irrigation return-flow to the aquifer. Consequently, the electrical conductivity (as proxy for salinity) (Slinger and Tension 2005) in Nueva lagoon has increased from 6 to 13 mS·cm^{-1} due to an increase in sea water intrusion (Rodriguez et al. 2011), while in the Honda Lagoon the conductivity has decreased from 6 to 1 mS·cm^{-1} due to an increase in surface water return flow.

Albufera de Adra is protected as a nature reserve by the Andalusian Autonomous Government and additionally classified as a wetland of international importance under the Ramsar convention. It harbors a large variety of fauna and flora like plants, fishes and algae and is especially important for waterfowl and autochthonous ichthyofauna (Rodriguez et al. 2011). In this study we focused on plants, fishes, algae and a crustacean.

Figure 7.1. Albufera de Adra (Spain) composed by the larger, coastal Nueva Lagoon and inland Honda Lagoon enclosed in the blue circle. The red line delimits the agricultural area of the study which consists of 899.2 ha (Fernández and Pérez 2004) of greenhouses area (white areas). The main economic activity in that area is protected horticulture (Junta de Andalucía 2011; 2012; Tolón and Lastra 2010). (*Source Google Earth*).

7.2.2. Developing the Characterization Factor

As commonly done in LCIA, Characterization Factors (Cf, Equation 7.1) are calculated as the product of a Fate Factor (FF, Equation 7.2) and an Effect Factor (EF, Equation 7.9). The FF models the salinity increase in the wetland due to increased water consumption rate (in g· l^{-1}·m^3·yr·m^{-3}) and the EF relates an ecological damage to the increased salinity measured as Potentially Affected Fraction (PAF) of species (in PAF·l·g^{-1}). The Cf for the salinity impact in this coastal wetland is therefore defined as the change in PAF of species due to a change in groundwater consumption, which is affecting the salinity content via altered amounts of groundwater and seawater infiltration into the wetland. This can be translated into the effect per m^3 of water consumed.

$$Cf = FF \cdot EF \; [\; \mathrm{m}^3 \cdot \mathrm{PAF} \cdot \mathrm{yr} \cdot \mathrm{m}^{-3}] \qquad \text{Equation (7.1)}$$

The uncertainties from FF and EF were propagated with Monte Carlo simulation to quantify the 95% confidence interval of the characterization factor. The assessment was performed with the probabilistic risk assessment simulation software @risk, version 5.0. (Palisade 2012) Normal distributions were applied for the FF and EF. The sampling method applied was Latin Hypercube and the number of iterations was 10,000.

Fate Factor. The FF was developed for the Nueva Lagoon since it is closer to the sea and thus affected by sea water intrusions. Moreover, recharge to the wetland from the Adra River Delta aquifer is predominant in the Nueva lagoon, while Honda is mainly fed by surface water (Rodriguez et al. 2011). The FF was based on a salt and a water balance, and we split each up into wet (X) and dry (Y) months, since there is a natural seasonal cycle of salinity. According to Rodriguez et al. (2011) there are 3 dry months (June, July, August) with almost no precipitation and 9 wet months. Due to the precipitation in the wet months, the salinity in the wetland decreases, since freshwater leads to a dilution and an exfiltration of saline water. Salinity increases during the dry period, when evaporation and saline water infiltration increase the salt concentration. Monthly water and salt balances were calculated for 1983, 2003 and 2008, respectively. For the FF calculation (Equation 7.2) the monthly results were aggregated to yearly values.

$$FF = \frac{\Delta C_N.V_N}{\Delta FGW}\frac{\Delta FGW}{\Delta \mathrm{ETcrop}}\left[\mathrm{g}\cdot\mathrm{l}^{-1}\cdot\mathrm{m}^3\cdot\mathrm{yr}\cdot\mathrm{m}^{-3}\right] \qquad \text{Equation (7.2)}$$

The first ratio defines the net salinity change due to change in freshwater inflow ($\Delta C_N \cdot V_N/\Delta FGW$) while the second part is related to the change in fresh groundwater infiltration due to changes in crop water consumption ($\Delta FGW/\Delta ET_{crop}$). FGW ($m^3 \cdot yr^{-1}$) is the total fresh groundwater inflow to Nueva Lagoon in the dry and wet seasons (FGW_x+FGW_y). C_N symbolizes the salinity, V_N the volume of the Nueva lagoon and ET_{crop} is crop evapotranspiration. Δ symbolizes the change between years. We consider water consumption as crop evapotranspiration. There is no runoff since everything is taken up by plants/evaporated.

Irrigation on the fields of Almería province and close to Albufera de Adra uses 80% groundwater (Céspedes et al. 2009) and 20% surface water. For more details and explanations of the other constant parameters see Table 7.1.

Equation 7.2 is calculated for three time spans, 1983-2003, 2003-2008 and 1983-2008. We focused on these years because between 2003 and 2008 the salinity constantly increased from 4.5 to 7.5 g/l, and changes in irrigation techniques occurred, along with a trend of greenhouse extension out of the delta valley. We took 1983 as a year with a situation as natural as possible in order to compare to 2003 and 2008 since from 1975 until 1983 the salinity remained constant in Albufera de Adra (Pulido et al. 1988).

Table 7.1. Constant parameters in Nueva Lagoon for the years 1983, 2003 and 2008. Climatic parameters were provided by Adra weather station (IFAPA 2012) and water bodies, salinities and morphometric characteristics by existing literature (Rodriguez et al. 2011). (*continued in next pages*)

Parameter	Definition	Unit	Value 1983	Value 2003	Value 2008	Comments
X	Wet months in a year[4]	month	9	9	9	Number of wet months: January, February, March, April, May, September, October, November and December (Rodriguez et al. 2011)
Y	Dry months in a year[4]	month	3	3	3	Number of dry months: June, July and August. (Rodriguez et al. 2011)
$\Delta C_{N,Y}$	Change in salinity from Nueva in dry months	g/l	0.6	0.6	0.6	Increase in salinity due to evapotranspiration and almost no precipitation assuming steady state in the individual years. This value was taken as assumption.
$\Delta C_{N,X}$	Change in salinity from Nueva in wet months	g/l	-0.6	-0.6	-0.6	Decrease in salinity due to higher precipitation levels and lower evapotranspiration during the wet months[4]. It is assumed that the salt balance must be zero in order to get the natural behavior of a stable situation over the year, so we assume that ΔC_{N2}=$-\Delta C_{N1}$ as a simplifying assumption in steady state although is an evidence that salinity is increasing over the years.
V_N	Volume of Nueva taking area and depth from literature[4]	m^3	316,667	316,667	316,667	Assuming the volume as a cone with a maximum depth of 3.8 m and area of 25 ha took from literature. The water level did not change much between 2006 and 2008 according to the Rodriguez et al. (2011)
C_{GW}	Salinity of the fresh groundwater[16]	g/l	1.6	1.6	1.6	Considering the salinity of the groundwater in the Adra River Delta Aquifer from the literature (Molina 2006).

C_{Sea}	Salinity in the Mediterranean Sea	g/l	37.6	37.6	37.6	Assuming an average of the Mediterranean Sea from literature.
C_N	Salinity in Nueva lagoon	g/l	2.6	4.50	7.50	Salinity averages from 2003 and 2008 according to Rodriguez et al.[4] using as conversion 1mS=0.64 $g \cdot l^{-1}$
ET_Y	Nueva evapotranspiration in dry months from IFAPA weather station[29]	m^3/month	59,325	39,432	41,098	Average evapotranspiration from dry months between 2003 to 2008. The values were taken from literature per area of 25 ha. (IFAPA 2012)
ET_X	Nueva evapotranspiration in wet months from IFAPA weather station[29]	m^3/month	24,331	21,337	23,737	Average evapotranspiration from wet months between 2003 to 2008. The values were taken from literature per area of 25 ha. (IFAPA 2012)
P_Y	Precipitation in 3 dry months[29]	m^3/month	1,308	183.3	33.3	Average precipitation from dry months between 2003 to 2008 considering 25 ha of surface. The values were taken from literature. (IFAPA 2012)
P_X	Precipitation in 9 wetmonths[29]	m^3/month	5,419	8,933	9,911	Average precipitation from wet months between 2003 to 2008 considering 25 ha of surface. The values were taken from literature. (IFAPA 2012)
ET_N	Evapotranspiration from Nueva[29]	m^3/month	33,079	25,860	28,076	Evapotranspiration's average from 2003 to 2008 taking into account the 25 ha surface of the lagoon. (IFAPA 2012)
P_N	Precipitation in Nueva[29]	m^3/month	4,392	6,745	7,442	Precipitation's average from 2003 to 2008 taking into account the 25 ha of surface of the lagoon. (IFAPA 2012)
ET_{crop}	Crop Evapotranspiration[31]	m^3/month	146,368	306,849	320,984	Taking into account the harvested areas for 2003 and 2008 from literature[23]

						(Junta de Andalucía 2012; Fernández et al. 2001)
SW_Y	Surface water inflows in dry	m^3/month	0	0	0	In all equations SWY is the surface water inflows in dry months but we neglected it since it is irrelevant for the Nueva Lagoon
SW_X	Surface water inflows in wet	m^3/month	0	0	0	In all equations SWX is the surface water inflows in wet months but we neglected it since it is irrelevant for the Nueva Lagoon

In order to obtain the unknown variables, FGW_Y, FGW_X, SGW_Y and SGW_X, we developed several equations for salt and water balances considering wet (X) and dry (Y) seasons. Equation 7.3 shows the salt balance for dry months, taking into account fresh groundwater inflows (FGW), subterranean sea water intrusions (SGW, henceforward called sea groundwater inflows) and groundwater outflow (GW_o) from Nueva Lagoon to the neighboring aquifer. Equation 7.4 shows the salt balance for wet months with inflows from fresh groundwater and sea groundwater, as well as groundwater outflow. Equation 7.5 is the yearly salt balance, incorporating both dry (Y) and wet (X) months and reflects the steady-state assumption, that the yearly balance is equal to zero. Equations 7.6, 7.7 and 7.8 are the water balances for dry (Y) and wet (X) months, respectively, as well as the yearly balance. The difference between evapotranspiration (ET) and precipitation (P) in dry months is greater than or equal to the sum of the respective inflows, which consists of fresh and sea groundwater inflows. Concerning the wet months, Equation 7.7 shows the difference between evapotranspiration and precipitation being lower than or equal to the sum of inflows and the groundwater outflow. To solve the system of equations we assume the algebraic sign in Equation 7.6 and Equation 7.7 to be equality instead of inequality. The yearly water balance (Equation 7.8) is zero due to the steady-state assumption.

Salt Balances

$$\frac{\Delta C_{N1} \cdot V_N}{Y} = SW_Y \cdot C_{SW} + FGW_Y \cdot C_{GW} + SGW_Y \cdot C_{Sea} - GW_{o,y} \cdot C_N \qquad \text{Equation (7.3)}$$

$$\frac{\Delta C_{N2} \cdot V_N}{X} = SW_X \cdot C_{SW} + FGW_X \cdot C_{GW} + SGW_X \cdot C_{Sea} - GW_{o,x} \cdot C_N \qquad \text{Equation (7.4)}$$

$$0 = Y \cdot SW_Y \cdot C_{SW} + X \cdot SW_X \cdot C_{SW} + Y \cdot FGW_Y \cdot C_{GW} + X \cdot FGW_X . C_{GW} + Y \cdot SGW_Y \cdot C_{Sea} + X \cdot SGW_X \cdot C_{Sea} - X \cdot GW_{o,x} \cdot C_N - Y \cdot GW_{o,y} \cdot C_N \qquad \text{Equation (7.5)}$$

Water Balances

$$ET_Y - P_Y \geq SW_Y + FGW_Y + SGW_Y \ - GW_{o,y} \qquad \text{Equation (7.6)}$$

$$ET_X - P_X \leq SW_X + FGW_X + SGW_X - GW_{o,x} \qquad \text{Equation (7.7)}$$

$$0 = -ET_N \cdot 12 + P_N \cdot 12 + Y \cdot SW_Y + \ X \cdot SW_X + Y \cdot FGW_Y + \ X \cdot FGW_X + Y \cdot SGW_Y + X \cdot SGW_X - X \cdot GW_{o,x} - Y \cdot GW_{o,y} \qquad \text{Equation (7.8)}$$

SGW_Y(m^3·month^{-1}) and SGW_X(m^3·month^{-1}) are the sea groundwater inflow into Nueva Lagoon in dry and wet months, respectively and $GW_{o,y}$ (m^3·month^{-1}) and $GW_{o,x}$ (m^3·month^{-1}) are the groundwater outflow from Nueva Lagoon in the dry and wet months, respectively. The values of the unknown variables of Equations 7.3 to 7.8 were obtained with the help of the solver GAMS© (GAMS 2012) using non-linear programming through the BARON optimizer (GAMS 2012) and the equation system was solved by minimizing the balance error. We established Equation 7.5 as the objective function which is to be minimized (to get close to zero) in that solver.

A sensitivity analysis was carried out by changing different constant parameters, such as salinities ($C_{N,1983}$, $C_{N,2003}$, $C_{N,2008}$,), the number of wet (X) and dry (Y) months and the amount of precipitation and evapotranspiration. Several assumptions were made in this section (see Appendix A7.2.1.). The confidence intervals for the FF were calculated by taking into account the maximum and minimum FF from the sensitivity analyses and assuming a normal distribution.

Effect Factor. Data describing the effect of salinity for various endpoints (e.g. survival, growth inhibition) on 18 species (plants, fish, algae and a crustacean) native to the “Albufera de Adra” wetland were collected from literature (Glenn et al.1995; Ahmed et al. 2008; Mufarrege et al. 2011; Van Wijk et al. 1988; Calheiros et al. 2011; Espinar et al. 2005; Cambrollé et al. 2011; Lissner and

Schierup 1997; Lillebø et al. 2003; Bartolomé et al. 2009; Hutchinson; Guma et al. 2010; Wang et al. 1997; Barman et al. 2005; Bright and Addison 2002; Greenwood; Schuytema et al. 1997) and are shown in the Appendix A7 (Table A7.1). This work focused on the indigenous species and the associated damages only. As is common in LCA, only negative impacts are considered and that potential benefits and changes in species composition are not included.

The use of EC50s from bioassays with different endpoints is the norm in the calculation of effect factors in LCA. A prime example for such practice can be found in USEtox, the LCIA toxicity model recommended by UNEP-SETAC: one of the two ecotoxicity EF' databases used in this model (Payet 2004), explicitly states that the EC50s, employed in the construction of SSDs and subsequently the derivation of EFs, can come from numerous different endpoints. Indeed it would be much more consistent to could construct SSDs based on EC50s describing the exact same effect (e.g. death or growth) of a stressor on an organism. However, in light of the absence of identical endpoints measured in bioassays, in LCIA the aggregation of many different endpoints is preferred over the use of much fewer data in the calculation of the EF.

The 50% effective concentration (EC50) due to salinity is the concentration where a 50% reduction in a given endpoint (e.g. growth) is observed compared to the control. EC50s were either calculated by fitting the log-logistic function to the salinity concentration-response plots (see figure A7.1), or were taken directly from literature.

Species Sensitivity Distribution (SSD) is an ecotoxicological tool that has been employed to calculate effect factors for different impacts (e.g. ecotoxicity, thermal pollution, eutrophication) in life cycle impact assessment. Several studies have been published in literature using SSD to obtain an EF, such as van de Meent and Huijbregts (2005), who proposed a multisubstance potentially affected fraction (msPAF)-based method for calculating ecotoxicological effect factors for LCA, Verones et al. (2010), who used SSD to calculate an EF for thermal pollution in freshwater aquatic environments and Struijs et al. (2011), who constructed a field sensitivity distribution of macroinvertebrates in inland waters to derive an EF for eutrophication due to phosphorus.

The SSD for salinity was constructed by fitting the log-normal cumulative distribution function (Equation A7.1) to the EC50s for native species (see Table A7.1). Equation 7.9 describes the effect factor (EF_{Sal}), which is the average change in the potentially affected fraction of freshwater aquatic species (ΔPAF_{Sal}) due to the change in salinity (ΔSal). The effect factor is calculated as the average gradient at the 50% hazardous concentration ($HC50_{Sal}$), defined as the concentration at which 50% or more of the species included in the SSD are exposed to concentrations above their EC50 (Rosenbaum et al. 2008).

$$EF_{Sal} = \frac{\Delta PAF_{Sal}}{\Delta Sal} = \frac{0.5}{HC50_{Sal}} \ [\mathrm{PAF} \cdot \mathrm{l} \cdot \mathrm{g}^{-1}] \qquad \text{Equation (7.9)}$$

A 95% confidence interval for the hazardous concentration was estimated by parametric bootstrapping and this uncertainty was propagated to the Effect Factor by taking Equation 7.9 into account.

Calculation of impact scores. The required amount of consumptive irrigation water (CW, m^3), resulting from the inventory of input/output data, is expressed by a functional unit (quantified performance of a product system for use as a reference unit) (JRC-IES 2010a,b). In our case, the functional unit is a tonne of tomato, pepper, cucumber, zucchini, watermelon, melon, aubergine or green bean harvested in greenhouses close to Albufera de Adra (Tolón and Lastra 2010). The impact score (IS, $m^3 \cdot PAF \cdot yr$) is the product of the Cf (Equation 7.1) and the CW. IS shows the impact of increasing salinity on aquatic species in the Nueva lagoon caused by the use of groundwater for agriculture.

Water consumption was calculated as the average evapotranspiration for the different cultivation periods for each crop following the irrigation crop management practices recommended by the experimental station of Cajamar research institute (Fernández et al. 2001) to improve the efficiency of agriculture production close to the wetland.

We considered the area of greenhouses from the municipality of Adra (Fernández and Pérez 2004) for the 8 main crops from 26 different vegetables that are produced in Almería. Amounts of production from the province of Almería (Junta de Andalucía) in 2008 were downscaled to the area around Albufera de Adra.

In order to compare the impact due to water consumption to that of other categories, results were converted to species per year following the recommendations of the ReCiPe method (species density for freshwater, 7.89×10^{-10} species·m^{-3}) (Goodkoop et al. 2009).

7.3. Results and discussion

7.3.1 Fate Factor

There was fresh groundwater inflow (FGW_Y and FGW_X) from the Adra River Delta (ARD) aquifer in wet and dry seasons in all three years. Sea groundwater inflow occurs in dry months only, while groundwater outflow occurs in wet seasons only. This shows that there is less recharge to the wetland from the aquifer in wet months. We further simplified the groundwater system presented by Rodriguez et al. (2011) by neglecting the blurred transition zone between low and high salinity parts of the aquifer, which is not a subsystem in our model. Since only salinities of the fresh groundwater and the sea water are known we assumed sharply separated sections of fresh and saline groundwater. Also, the location of the brackish-freshwater transition zone in the aquifer fluctuates during the seasons, which we did not take into account (Table 7.2.).

Table 7.2. Unknown variables in Nueva Lagoon for the years 1983, 2003 and 2008. The values of the variables are obtained by solving the equation system of Equation 7.3 to Equation 7.8 with GAMS (GAMS 2012).

Parameter	Definition	Unit	Value 1983	Value 2003	Value 2008	Comments
FGWY	Fresh groundwater inflow to Nueva Lagoon in dry months	$m^3 \cdot month^{-1}$	$3.96 \times 10^{+4}$	$3.92 \times 10^{+4}$	$3.96 \times 10^{+4}$	Calculated from Equation 7.3 to Equation 7.8
FGWX	Fresh groundwater inflow to Nueva Lagoon in wet months	$m^3 \cdot month^{-1}$	$8.63 \times 10^{+4}$	$2.65 \times 10^{+4}$	$2.18 \times 10^{+4}$	Calculated from Equation 7.3 to Equation 7.8
SGWY	Sea groundwater infiltration into Nueva Lagoon in dry months	$m^3 \cdot month^{-1}$	0	$1.49 \times 10^{+1}$	0	Calculated from Equation 7.3 to Equation 7.8
SGWX	Sea groundwater infiltration into Nueva Lagoon in wet months	$m^3 \cdot month^{-1}$	0	0	0	Calculated from Equation 7.3 to Equation 7.8
GWo,y	Groundwater outflow from Nueva Lagoon in the dry months	$m^3 \cdot month^{-1}$	0	0	0	Calculated from Equation 7.3 to Equation 7.8
GWo,x	Groundwater outflow from Nueva Lagoon in the wet months	$m^3 \cdot month^{-1}$	$6.12 \times 10^{+4}$	$1.41 \times 10^{+4}$	$7.46 \times 10^{+3}$	Calculated from Equation 7.3 to Equation 7.8

In dry periods, groundwater from ARD aquifer enters the lagoon. For the years 2003 and 2008 the rate throughout the months of June to August was similar (Rodriguez et al. 2011). During wet periods the wetland and aquifer produce additionally a groundwater outflow to the sea, as has been found previously by Alcalá et al. (2008) Still, the presence of the wetland at the south-eastern edge of the ARD aquifer reduced potential groundwater discharge to the sea because of the high evapotranspiration rates in the surface area of this wetland (Rodríguez et al. 2004).

Comparing the three years (1983, 2003, 2008), the ratio between outflow and inflow from and to the wetland is constantly reduced. In 1983, the outflow was 82% of the inflow, in 2003 it was 48% while in 2008 it was only 28%. We suggest as principal FF the one between 2003 and 2008 due to the best quality of the data and the stabilization of crop extension.

Results for the sensitivity analysis are shown in the Appendix A7 Table A7.2. The maximum value of the FF occurs between 2008 and 2003 when the wetland salinity is increased by 20% and the other parameters are kept constant, which gives a $FF_{2008\text{-}2003}$ of 6.72 $g·l^{-1}·m^3·yr·m^{-3}$. On the other hand, the minimum value of the FF results by decreasing the salinity by 20% for the period between 2003-1983 with a FF of $2.50×10^{-1} g·l^{-1}·m^3·yr·m^{-3}$. These two extreme scenarios were taken as minimum and maximum for the uncertainty assessment and for establishing the distribution function.

7.3.2. Effect Factor

Figure 7.2 shows the species sensitivity distribution (SSD) for salinity, for native species identified in the Nueva lagoon. The log-normal cumulative distribution function was fitted to ordered EC50 values, and the $HC50_{Sal}$ was found to be equal to 8.87 $g·l^{-1}$ from the fitted curve. The 95% confidence interval for the $HC50_{Sal}$ was calculated as 6.29 – 12.5 $g·l^{-1}$. The EF was then found to be $5.64×10^{-2}$ $PAF·l·g^{-1}$ with a standard error of ± $0.76×10^{-2}$ $PAF·l·g^{-1}$, calculated via propagating the error from the HC50 to the EF.

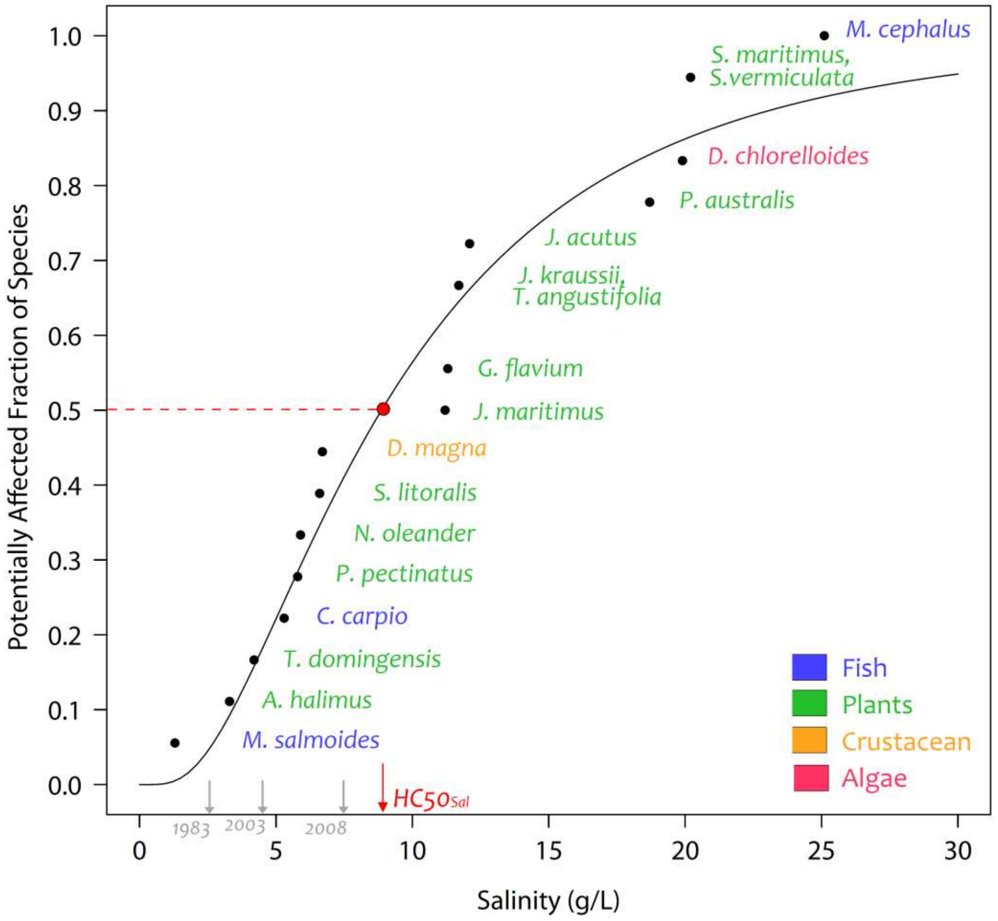

Figure 7.2. Species Sensitivity Distribution (SSD) for salinity for 18 species native to Nueva Lagoon. The grey arrows indicate the salinity in 1983 (2.6 g·l^{-1}), 2003 (4.5 g·l^{-1}) and 2008 (6.5 g·l^{-1}).

Salinity in the Nueva Lagoon increased from 4.5 g·l^{-1} in 2003 to 7.5 g·l^{-1} in 2008 (Table 7.1), which, according to Figure 7.2, corresponds to an increase of approximately 20% of species potentially affected in this period. The absolute number of species estimated to be found in the wetland is 30 (Waste ideal, 2012), taking into account plants, fish, algae and crustaceans. The increase in salinity and eutrophication over the past years has already resulted in the disappearance of a specie in Albufera de Adra, namely *Scirpus lacustri*, which was cited by Losa and Rivas Godo in 1968 (Losa and Rivas, 1968) and Sagredo in 1987 (Sagredo 1987) but is no longer found in the lagoon today. From Figure 7.2 we observe that two fish species, *M. salmoides* (EC50 = 1.3 g·l^{-1}) and *C. carpio*

(EC50 = 5.3 g·l^{-1}), are particularly sensitive to salt stress, and another fish species, *M. cephalus* (EC50 = 25.1 g·l^{-1}), is the least sensitive of all the native species included in this study. The EC50 of the algae included in the SSD, *D. chlorelloides* is 19.9 g·l^{-1} and is above $HC50_{sal}$ but the crustacean, *D.magna* is located below $HC50_{sal}$ with an EC50 of 6.6 g·l^{-1}. Plants, the most abundant taxonomic group are distributed throughout the SSD, with 5 species lying below $HC50_{sal}$ and 8 species lying above it.

Data availability permitted the consideration of 18 species from Nueva Lagoon (13 plants, 3 fish, 1 algae and 1 crustacean) in this work. So, given a total of 30 species reported to be found in the wetland (Waste Ideal 2012), with these 18 species we cover 60% of the species in the wetland. The calculated EF could, with some caution, be applied to other wetlands, assuming their native species composition is not entirely dissimilar to the one encountered in Nueva Lagoon.

For instance, Punta Entinas is an endorheic wetland, located in the arid southeast of Spain very close to Albufera de Adra surrounded by Mediterranean ecosystems. Both wetlands, Nueva Lagoon and Punta Entinas (focus on salt marsh and marshland) have species in common, for instance *A. halimus, S. vermiculata, S. maritimus, J. maritimus, J. acutus, P. australis, N. oleander, P. pectinatus* and *D. chlorelloides* (Waste ideal, 2012), amounting to 50% of the species included in the EF for Albufera de Adra. Therefore, we might consider the EF to be applicable for the marshland of Punta Entinas.

7.3.3. Characterization Factor

According to Equation 7.1 the Cf for Nueva Lagoon is 3.16 x10^{-1} with a standard error of ±1.84 x10^{-1} PAF·m^{3}·yr·m^{-3}, and a 95% confidence interval of 8.30x10^{-2}–7.83x10^{-1} PAF·m^{3}·yr·m^{-3}.

7.3.4. Impact Score

The impact score is calculated as a product of the characterization factor (Cf) developed for Nueva Lagoon ($3.16\text{x}10^{-1}$ $PAF{\cdot}m^3{\cdot}yr{\cdot}m^{-3}$) and the crop evapotranspiration (ET_{crop}) (Equation 7.10). Crop evapotranspiration was obtained from "Las Palmerillas" experimental station close to Albufera the Adra (constructed to improve the efficiency in agricultural production) (Fernández et al. 2001) and was converted to $m^3{\cdot}yr^{-1}$ taking into account the cultivation area from the study (Fernández and Pérez 2004) close to Adra. Table 7.3 shows the Impact Score ($m^3{\cdot}PAF{\cdot}yr$) for each crop per unit of area and per ton of production ($m^3{\cdot}PAF{\cdot}tonne^{-1}{\cdot}yr$) considering the local productions in that area (Junta de Andalucía 2012).

$$IS = CF \cdot ET_{crop} \qquad \text{Equation (7.10)}$$

Table 7.3. Characteristics and impact scores for the 8 main crops in the area of study: greenhouse area (GH_{Area}), crop evapotranspiration (ET_{crop}), impact score per area ($IS_{1,per\ area}$) and its assigned percentage ($IS_{1,\%}$), crop's yield (Y_c) and Impact Score per tonne ($IS_{2,tonne}$) .

Crops	GH_{Area}	ET_{crop}	$IS_{1,\ per\ area}$	$IS_{1,\%}$	Y_c	$IS_{2,tonne}$
	ha	$m^3{\cdot}ha^{-1}$	$m^3{\cdot}PAF{\cdot}yr$	%	$tonne{\cdot}ha^{-1}$	$m^3{\cdot}PAF{\cdot}tonne^{-1}{\cdot}yr$
Tomato	216.1	$7.50\text{x}10^{+5}$	$1.98\text{x}10^{+4}$	29.6	100.6	$9.08\ \text{x}10^{-1}$
Pepper	180.5	$6.82\text{x}10^{+5}$	$1.80\ \text{x}10^{+4}$	26.9	61.7	$1.61\ \text{x}10^{0}$
Cucumber	95.1	$1.53\text{x}10^{+5}$	$4.03\ \text{x}10^{+3}$	6.0	87.0	$4.87\ \text{x}10^{-1}$
Zucchini	97.9	$2.75\text{x}10^{+5}$	$7.24\ \text{x}10^{+3}$	10.9	54.5	$1.36\ \text{x}10^{0}$
Watermelon	101.1	$1.81\text{x}10^{+5}$	$4.76\ \text{x}10^{+3}$	7.1	69.6	$6.76\ \text{x}10^{-1}$
Melon	115.0	$3.42\text{x}10^{+5}$	$9.02\ \text{x}10^{+3}$	13.5	36.0	$2.18\ \text{x}10^{0}$
Aubergine	33.7	$9.58\text{x}10^{+4}$	$2.52\ \text{x}10^{+3}$	3.8	72.6	$1.03\ \text{x}10^{0}$
Green bean	59.7	$5.19\text{x}10^{+4}$	$1.37\ \text{x}10^{+3}$	2.1	15.3	$1.50\ \text{x}10^{0}$

For the cultivated area considered, tomato is the crop that shows the highest impact score, with approximately 30% of the overall impact, because tomato is the most produced crop in the province, whereas green bean shows the smallest impact with around 2% due to a relatively small cultivated area. However, when we consider the total impact score per tonne of production, we obtain different results due to different crop yields. The low yield of melon leads to the highest impact per tonne, while cucumber with a higher yield leads to the lowest impact score per tonne for the crops studied (Junta de Andalucía 2011).

7.4. Application in LCA studies

This work derived the first Cf for salinity impacts in a coastal wetland defined as the change in the Potentially Affected Fraction (PAF) of species due to a change in salinity related to the extraction of groundwater for crop irrigation. This case study takes the expectation away that this is a fully applicable approach for the whole world and it proved to be very relevant indeed. The impacts on wetland biodiversity due to the irrigation of the existing crops close to the study area, were calculated using the proposed Cf. A comparison between the salinity impacts of the main crops tomato, cucumber, zucchini, melon and aubergine with other impact categories was carried out in order to investigate the relative importance of salinity impacts. For this comparison we used the endpoints of several categories within the area of protection of "ecosystem quality" of the ReCiPe methodology (Goedkoop et al. 2009). Experimental data for these crops were adapted from Stoessel et al. (2012) taking into account a local yield (Table 7.3) in Adra greenhouses. The crop-specific impact scores presented in Table 7.3 ($PAF \cdot m^3 \cdot yr \cdot tonne^{-1}$) were converted into $species \cdot yr \cdot kg^{-1}$ considering the recommended freshwater species density (7.89×10^{-10} $species \cdot m^{-3}$) and the conversion (Goedkoop et al. 2009) dPDF/dPAF = 1 (Table 7.4).

Table 7.4. Endpoint Impacts (*species·yr·kg*$^{-1}$) according to the ReCiPe methodology and the contribution of each category to the total ecosystem quality impact. No data was available for green beans and watermelon, and thus these crops are neglected in this comparison.

Category impact	**Tomato**		**Cucumber**		**Zucchini**		**Melon**		**Aubergine**	
Unit	**species·yr·kg^{-1}**	**%**	**species·yr·kg^{-1}**	**%**	**species·yr·kg^{-1}**	**%**	**species·yr·kg^{-1}**	**%**	**species·yr·kg^{-1}**	**%**
Salinity impact due to water use	7.16 x10^{-13}	0.08	3.84 x10^{-13}	0.05	1.07 x10^{-12}	0.02	1.72 x10^{-12}	0.09	8.14E-13	0.02
Climate change	6.16 x10^{-10}	69.7	5.85 x10^{-10}	73.2	1.96 x10^{-9}	37.4	7.07 x10^{-10}	38.3	3.67 x10^{-9}	70.4
Terrestrial acidification	2.32 x10^{-12}	0.26	2.12 x10^{-12}	0.27	9.88 x10^{-12}	0.19	3.38 x10^{-12}	0.18	1.43 x10^{-11}	0.28
Freshwater eutrophication	3.85 x10^{-13}	0.04	3.46 x10^{-13}	0.04	5.63 x10^{-13}	0.01	3.82 x10^{-13}	0.02	2.56 x10^{-12}	0.05
Terrestrial ecotoxicity	2.40 x10^{-11}	2.71	2.10 x10^{-12}	0.26	4.73 x10^{-12}	0.09	3.08 x10^{-12}	0.17	7.38 x10^{-11}	1.42
Freshwater ecotoxicity	2.59 x10^{-14}	0.00	1.10 x10^{-14}	0.00	1.39 x10^{-13}	0.00	7.65 x10^{-14}	0.00	1.14 x10^{-13}	0.00
Marine ecotoxicity	1.21 x10^{-16}	0.00	4.86 x10^{-17}	0.00	1.86 x10^{-16}	0.00	8.28 x10^{-17}	0.00	5.33 x10^{-16}	0.00
Agricultural land occupation	2.15 x10^{-10}	24.4	1.82 x10^{-10}	22.7	3.12 x10^{-09}	59.5	1.08 x10^{-9}	58.5	1.25 x10^{-9}	24.1
Urban land occupation	8.11 x10^{-12}	0.92	1.29 x10^{-11}	1.61	4.48 x10^{-11}	0.86	2.18E x10^{-11}	1.18	7.62 x10^{-11}	1.46
Natural land transformation	1.68 x10^{-11}	1.90	1.52 x10^{-11}	1.90	9.87 x10^{-11}	1.88	2.72 x10^{-11}	1.47	1.20 x10^{-10}	2.30
Total	8.84 x10^{-10}	100	7.99 x10^{-10}	100	5.24 x10^{-9}	100	1.84 x10^{-9}	100	5.21 x10^{-9}	100

The results for all crops show that, if the generic freshwater species density from ReCiPe is used (Table 7.4), the impact of salinity (due to water use) for the total damage to ecosystems is in the range of freshwater eutrophication for tomato ($7.16x10^{-13}$ species.yr·kg^{-1}) and cucumber ($3.84x10^{-13}$ species·yr·kg^{-1}), in the range of terrestrial ecotoxicity for zucchini ($1.07x10^{-12}$ species·yr·kg^{-1}) and melon ($1.72x10^{-12}$ species·yr·kg^{-1}) and in the range of freshwater ecotoxicity for aubergine ($8.14x10^{-13}$ species·yr·kg^{-1}). The relative contribution to the total impact score is dominated by climate change for all the crops with approximately 70% in tomato, cucumber and aubergine and around 40% in zucchini and melon. The climate change ecosystems impact category considers fertilizing (ammonium nitrate, single superphosphate and potassium sulphate) and electricity consumption for the irrigation system, which is the highest contributing impact when capital goods are not considered.

Note that if a specific freshwater species density ($9.47x10^{-5}$ species·m$^{-3)}$ from the wetland had been used instead of generic freshwater species density from ReCiPe, the salinity impact for the total damage to ecosystems would represent the major contribution (94-99%) between all categories impact (tomato $8.60x10^{-08}$ species.yr·kg^{-1}, cucumber $4.61x10^{-08}$ species.yr·kg^{-1}, zucchini $1.29x10^{-07}$ species.yr·kg^{-1}, melon $2.07x10^{-07}$ species.yr·kg^{-1} and aubergine $9.77x10^{-08}$ species.yr·kg^{-1}). Hence, the difference between them shows that taking a global average value can be misleading since local species richness can be very different.

7.5. Outlook

Future efforts should be undertaken in order to further methodological development and make global characterization factors available. It is a broader approach (exploratory case study) with a more global perspective shall be developed. This will close an important gap in the LCIA methodology regarding the relevant impacts on coastal wetlands.

In this work we used the freshwater species density from the ReCiPe model, acknowledging that freshwater species density can greatly vary depending on local conditions (e.g. the freshwater species density in Albufera de Adra is 9.47×10^{-5} species·m^{-3}). Hence, further improvements in LCIA endpoint methodologies could be considered given that species density for freshwater was shown to be higher than terrestrial species density (Vörösmarty et al. 2010), in contrast to estimates proposed by ReCiPe.

Appendix A7

A7.1. Effect Factor

Figure A7.1 shows the plots of salinity concentration-response curves for the various endpoints (see references for further information). The 4-parameter log-logistic function (Equation A7.1) was fitted to all data with the aid of the R package 'drc'[1]:

$$y_c = c + \frac{d-c}{1+(\frac{x}{e})^b} \qquad \text{Equation (A7.1)}$$

Where y_c is the response , x the salinity concentration, c and d are the lower and upper horizontal asymptotes respectively, b is the curvature and e is the inflection point[2]. The EC50 was estimated from each fitted curve and is displayed for each species in Table A7.1, which also displays the EC50s gathered from literature.

Table A7.1. EC50's from species in Albufera de Adra.

Species	EC50	Endpoint	Comments	References
Micropterus salmoides	1.3	Survival	Calculated	Meador, MR. and Kelso, WE (1989)
Atriplex halimus	3.3	Germination	Calculated	Ahmed, HB. et al.(2008)
Typha domingensis	4.2	Height, shoots and dry matter	Calculated	Glenn, E. et al. (1995)
Ciprinus carpio	5.3	Length and weight	Calculated	Wang, JQ. et al. (1997)
Potamogeton pectinatus	5.8	Biomass	Calculated	Van Wijk, R.J. et al. (1988)
Nerium oleander	5.9	Growth	From literature	Bright, DA and Addison, J (2002)
Daphnia magna	*6.6	Survival	From literature	Schuytema, GS et al. (1997)
Scirpus litoralis	6.7	Germination	Calculated	Espinar, JL and Garcia, LV (2005)
Juncus maritimus	11.2	Germination	Calculated	Boscaiou, M et al. (2007)
Glacium flavium	11.3	Growth	Calculated	Cambrollé, J. et al. (2011)
Typha angustifolia	11.7	Survival	Calculated	Hutchinson, I. (1988)
Juncus kraussii	11.7	Germination	From literature	Greenwood, M. (2008)
Juncus acutus	12.1	Germination	EC50 calculated = 10.6; final EC50 calculated by taking geometric mean of extracted EC50 plus one more value found in literature: 13.9	Boscaiou, M et al. (2007) Greenwood, M. (2008)
Phragmites australis	18.7	Survival	EC50 extracted from plot = 16.4; final EC50 calculated by taking geometric mean of extracted EC50 plus two more values found in literature: 18.9 and 21	Lissner, J. and Schierup, HH (1997) Greenwood, M. (2008)
Dictyosphaerium chlorelloides	19.9	Growth	From literature	Bartolomé, MC (2009)
Scirpus maritimus	20.2	Survival	Calculated	Lillebø, AI et al. (2003)
Salsola vermiculata	20.2	Germination	Calculated	Guma IR. et al. (2010)
Mugil cephalus	25.1	Growth	Calculated	Barman, UK.et al.(2005)

**LC50 mean value for the LC50 for Daphnia magna, as calculated by Schuytema et al. (1997)*

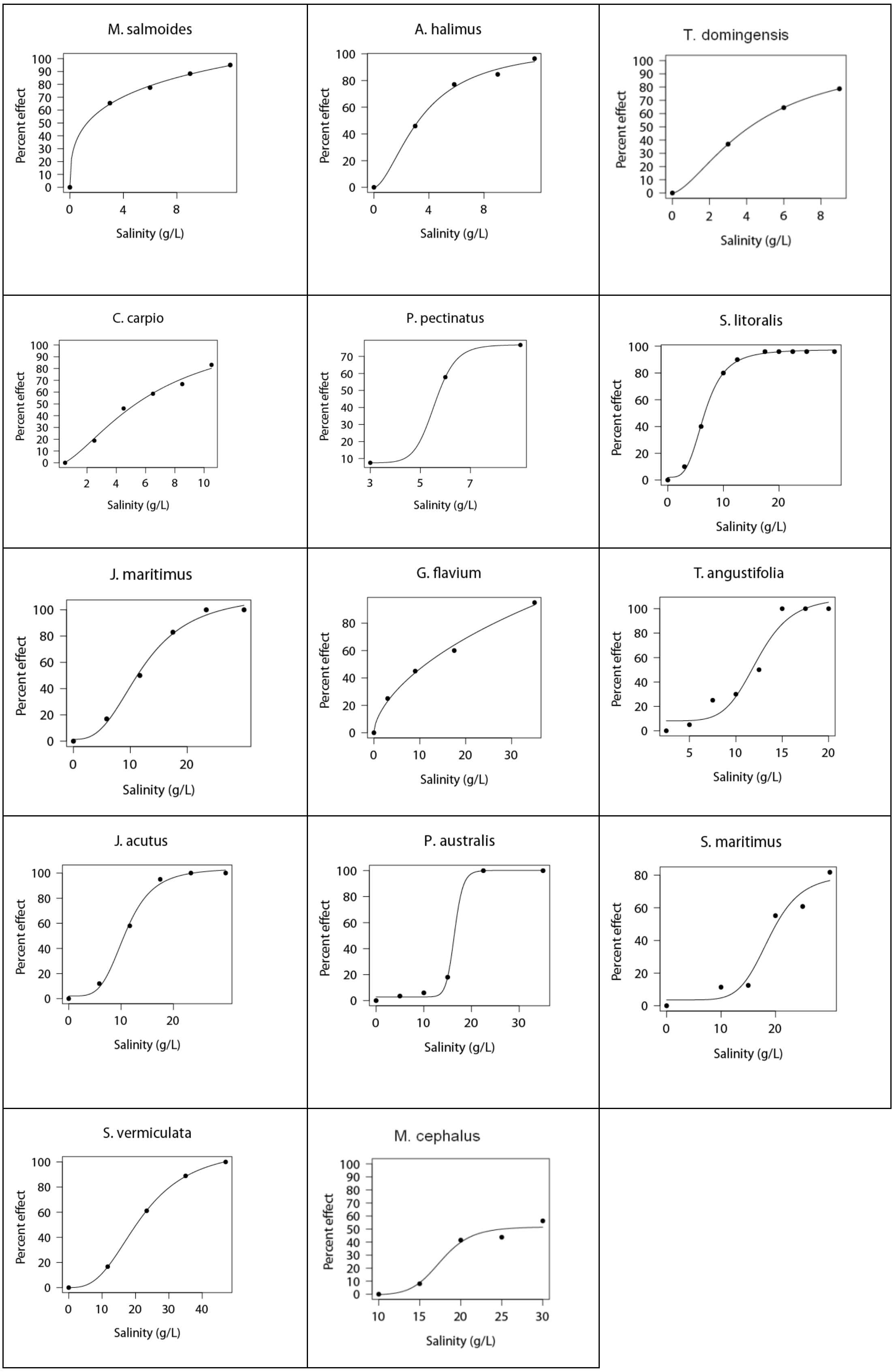

Figure A7.1. Salinity concentration-response curves for species in Albufera de Adra.

A7.2. Fate Factor

A7.2.1. General assumptions for the Fate Factor

Regarding the FF, we developed a model which only distinguished fresh- and sea- groundwater (the main inputs in the Nueva Lagoon Water balance) with their respective salinities. The FF is typically quantified as a product of residence time, concentration and area or volume. In our case, the FF would basically be the fresh water that lacks in the wetland system leading to increased concentration of salt due to increased sea water infiltration into the wetland. The reality is a much more complex situation; hence our model results provide only a simplified solution.

A7.2.2. Sensitivity analysis for the Fate Factor

Table A7.2: Sensitivity scenarios for the Nueva lagoon of Albufera de Adra. Parameters which were varied are the salinity (C_N) in the wetland in 2003 and 2008, the number of wet (Y) and dry (X) months, as well as the amounts of precipitation (P_N) and evapotranspiration (ET_N) on the Nueva lagoon.

	$FF_{2008-2003}$	$FF_{2003-1983}$	$FF_{2008-1983}$	$C_{N,1983}$	$C_{N,2003}$	$C_{N,2008}$	X	Y	P_N	ET_N
Scenarios	**($g \cdot m^3 \cdot month \cdot l^{-1} \cdot m^{-3}$)**			**($g \cdot l^{-1}$)**			**month**		**($m^3 \cdot yr^{-1}$)**	
Original scenario	$5.61x10^{+00}$	$3.19x10^{-01}$	$7.47x10^{-01}$	2.6	4.5	7.5	9	3	$8.39x10^{+04}$	$3.17x10^{+05}$
Increasing the salinity by 20%	$6.72x10^{+00}$	$3.75x10^{-01}$	$8.89x10^{-01}$	3.1	5.4	9	9	3	$8.39x10^{+04}$	$3.17x10^{+05}$
Decreasing the salinity by 20%	$4.48x10^{+01}$	$2.50x10^{-01}$	$5.92x10^{-01}$	2.1	3.6	6	9	3	$8.39x10^{+04}$	$3.17x10^{+05}$
Increasing nº of wet months	$5.61x10^{+00}$	$3.19x10^{-01}$	$7.47x10^{-01}$	2.6	4.5	7.5	10	2	$8.39x10^{+04}$	$3.17x10^{+05}$
Increasing nº of dry months	$5.61x10^{+00}$	$3.19x10^{-01}$	$7.47x10^{-01}$	2.6	4.5	7.5	8	4	$8.39x10^{+04}$	$3.17x10^{+05}$
Increasing precipitation and evapotranspiration by 20%	$5.61x10^{+00}$	$3.19x10^{-01}$	$7.47x10^{-01}$	2.6	4.5	7.5	9	3	$5.27x10^{+03}$ (1983) $8.10x10^{+03}$ (2003) $8.93x10^{+03}$ (2008)	$3.97x10^{+04}$ (1983) $3.10x10^{+04}$ (2003) $3.37x10^{+04}$ (2008)
Decreasing precipitation and evapotranspiration by 20%	$5.61x10^{+00}$	$3.19x10^{-01}$	$7.47x10^{-01}$	2.6	4.5	7.5	9	3	$3.51x10^{+03}$ (1983) $5.95x10^{+03}$ (2003) $5.95x10^{+03}$ (2008)	$2.65x10^{+04}$ (1983) $2.07x10^{+04}$ (2003) $2.25x10^{+04}$ (2008)

Chapter 8

Environmental assessment of urban water cycle on Mediterranean conditions by LCA approach

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Abstract

Urban water authorities in many countries are struggling to satisfy the increasing demand for both potable and non-potable water whilst also improving the environmental profile of the urban water system.

The main goal of this study is to use the Life Cycle Assessment methodology to carry out an environmental analysis of every stage of the urban water cycle in Tarragona, a Mediterranean city of Spain (scenario I). These stages are: water abstraction, potable water treatment, intermediate pumping, distribution network, sewage collection and wastewater treatment. This study also proposes possible scenarios for improving the environmental performance reducing the high level of water stress resulting from increasing demand and limited resources: using reclaimed water (scenario II) and using desalination plants and reclaimed water to supply water during a drought (scenario III). Inventory analysis was performed using local operation data.

In the current situation (scenario I) the main environmental impacts on urban water cycle were caused by 35.2% of distribution network , 20.5% of collection pumping and 13.8% of wastewater treatment plant for the Global Warming Potential impact category because these stages have high energy consumption. In Reclaimed water (scenario II), no significant improvement in indicators is observed, because the addition of environmental loads introduced by the tertiary treatment, except for the reduction in freshwater consumption due to the net water saving. In the drought scenario (scenario III), all impact categories have been increased from Scenario II e.g. 18% in Photochemical Oxidation, 21% in Eutrophication Potential, 30% Ozone Depletion Potential and Ecotoxicity Potential, 36% in Depletion of Abiotic Resources and Cumulative Energy Demand, 38% in Global Warming Potential and 42% in Acidification Potential.

Keywords: Life Cycle Assessment (LCA), urban water, potable water, wastewater, desalination, tertiary treatment.

8.1. Introduction

The European Union Water Framework Directive (2000) is a European regulation that requires EU Member States to ensure that all their water bodies have 'good ecological and chemical status' by 2015. In Spain, where water demand and degradation has increased in recent decades, this legislation could be a key instrument for bringing about an important qualitative change in water management and a transition from an old to a new water culture. Integrated management is needed to ensure that water is used rationally, and this means assessing the entire water management life cycle including water abstraction, potable water treatment, distribution, water consumption, sewage collection, wastewater treatment and water reclamation.

Proper management of the urban water cycle should include environmental criteria to respect the natural environment and return water to the environment in an acceptable condition. In addition, potable water should be of sufficient quality and meet the needs of the population. Water conservation requires action across the whole urban water cycle. A large number of methods and tools have been developed to describe the environmental implications of urban water systems.

Life Cycle Assessment (LCA) is a standard international tool, ISO 14040-44, (ISO 2006 a,b) used to evaluate the environmental impact of products, processes, and services. It considers the entire life cycle, from raw material extraction, to manufacture, distribution, use, end of life treatment, recycling and eventually, disposal.

In addition, there have been several studies based on Life Cycle Assessment (LCA) in recent decades to determine the environmental impacts resulting from water production, water transport to the customer and wastewater treatment; however, few studies have assessed the whole urban water cycle. The literature contains studies on water classifications and definitions (Owens 2001), different elements of water cycle like potable water treatment plants (Barrios et al. 2008; Dixon et al. 2003; Friedrich 2001 and Hospido et al. 2004), wastewater treatment plants (Hospido et al. 2004; Lundin et al. 2000; Palme et al. 2005; Pasqualino et al. 2009; Renou et al. 2008 and Tidaker et al. 2006) and the environmental assessment of urban wastewater reuse (Asano et al. 2007; Beavis and Lundie, 2003; Lim and Park 2008; Meneses et al. 2010a; Muñoz et al. 2009 and Pasqualino et al. 2011) and wastewater sludge treatment (Suh and Rousseaux 2001). In addition, some papers have taken into account increased awareness of scarcity scenarios and have focused on studying alternative methods of supply water to the population. Some of these papers assess the treatment of potable water from desalination plants (Bonton et al. 2011; Meneses et al. 2010b; Muñoz and Rodríguez Fernández-Alba, 2008; Pankratz et al., 2004; Raluy et al. 2004; Sombekke et al. 1997; Vince et al. 2008). Others studies have proposed sustainable metropolitan/urban water systems planning (El Sayed Mohamed et al. 2010; Lundie et al. 2004; Lundie et al. 2005; Lundin and Morrison, 2002; Mohapatra et al. 2002; Starkt et al. 2005; Stokes and Horvath, 2006) or water cycle from the pumping stations to the wastewater treatment plants (Lassaux et al. 2007).

In addition, Muñoz et al. (2010) purposed a Life Cycle Assessment comparison between a case study concerning a Spanish plan based on increasing desalination capacity as well as regeneration of wastewater in order to complement the natural water resources and augment the available freshwater in the Mediterranean basins (Program AGUA 2010 (Ministerio de Medio Ambiente, 2010)) and the predecessor program which is the Ebro river water transfer program. During the last years, some developments in water assessment methodologies have been made to include the water use impact taking into account the water availability (Milà i Canals et al. 2009; Pfister et al. 2009; Smathkin et al. 2004).

All these authors have carried out LCA studies on different stages of the urban water cycle; however, this work would be the first case in a Spanish context including several stages of water of a Mediterranean Case study.

The main objective of this work is to carry out an environmental life cycle assessment of the Urban Water Cycle (UWC) in a Mediterranean area based on actual operation data. We chose Tarragona as a case study and representative city due to its typical Mediterranean climate of low rainfall and frequent droughts. Tarragona has a natural fortress, of surrounding hill overlooking the sea and sheltering the calm waters of the port. Its exact geographic coordinates are 41º 07′ N and 0º 14′ E with an extension of 57.9 km^2 and it has a population of 134,085 inhabitants. The distribution network provided an average of 145.000 inhabitant's population by year with annual supplied of 11,368,859 m^3 potable water. The sewerage network consists of 37 piping km and 4 wastewater treatment plants which treat the collected wastewater of 9,499,894 m^3 in 2008 and generates 8,461 tones of sludge intended for agriculture and composting (Ematsa, 2009).

The system considered, namely Scenario I, involves all the main steps of a UWC: water abstraction, potable water treatment plant (PWTP), intermediate pumping, distribution network, sewerage and wastewater treatment plant (WWTP).

Further two alternative scenarios have also been analysed:

- Scenario II (*reclaimed water*): Part of the water treated in a wastewater treatment Plant (WWTP) is reused as reclaimed water. It includes an additional tertiary treatment in the wastewater treatment stage and considers the non-potable application of the reclaimed water.
- Scenario III (*drought scenario*): This includes the water reclamation, and the addition of a desalination plant within the cycle, as an additional source of potable water.

8.2. Description of an Urban Water Cycle in the Mediterranean area

In all the scenarios water resources are supplied by the Water Consortium of Tarragona (CAT) which treats water from the Ebro river (75%) and water from other sources such as mines and wells near the city (25%). However, water from mines is not sent through the Potable Water Treatment Plant (PWTP) because only chlorine needs to be added for the water to reach the mandatory quality.

8.2.1. Scenario I: Urban Water Cycle

The urban water cycle considered in scenario I (Figure 8.1) follow the entire life cycle of water from its origin as a natural resource until it is released again into the environment. The following steps have been considered:

1. *Water abstraction:* This is the extraction of water resources (Ebro River) directly from the environment by pumping.

2. *Potable Water Treatment Plant*: This involves conditioning the water before it reaches the distribution network. Drinking water needs to meet specific health

requirements. Treatment starts with sieving, settling and filtering in order to separate the suspended particles and remove turbidity (Pre-oxidation, physical-chemical treatment, decantation), and continues with a chemical disinfection process (activated carbon treatment and post-chlorination). (See flowchart in Figure A8.1). This eliminates microorganisms from the water and also disinfects the piping-in network.

3. *Distribution Network*: Potable water is supplied from the PWTP to the population through the distribution network. At this stage, groundwater is added into the water supply.

4. *Sewerage:* This includes wastewater collection and its passage through the sewage network to the wastewater treatment plant.

5. *Wastewater Treatment Plant*: Wastewater treatment is divided into two main steps; the water line and the sludge line. The water line is structured into the pre-treatment (desanding), primary treatment (primary decanter), secondary treatment (biologic reactor, secondary reactor) and tertiary treatment (sand filtration, UV treatment, chlorination). Data used in this study are drawn from a previous study (Pasqualino et al., 2009), of a wastewater treatment plant located in the Mediterranean area (See flowchart in Figure A8.2).

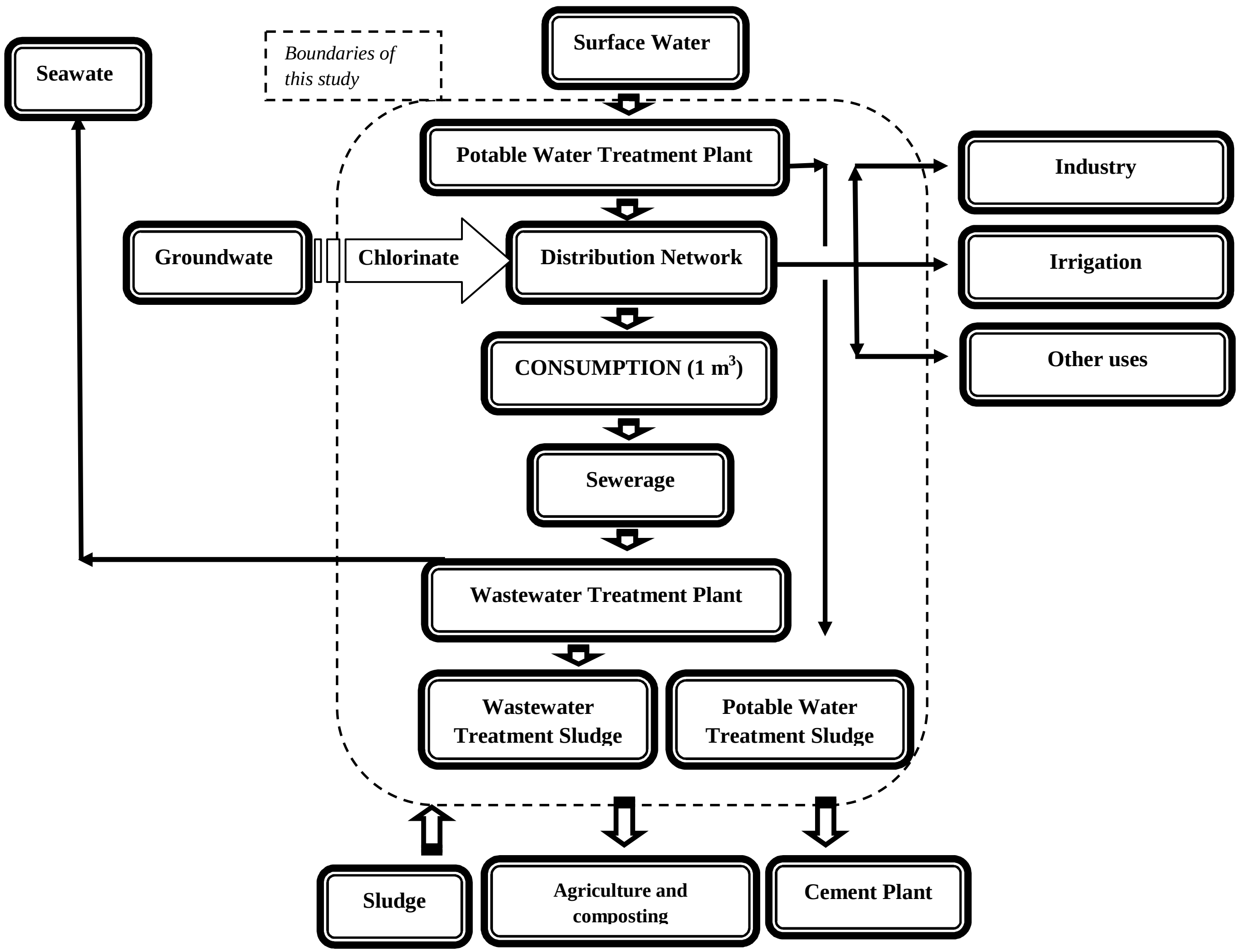


Figure 8.1. Urban water cycle in Mediterranean area.

8.2.2. Scenario II: Reclaimed water

This case study is based on scenario I, but includes a tertiary treatment in the wastewater treatment plant to reclaim water for use in non-potable applications such as irrigation (Figure 8.2). Tertiary treatment consists of physico-chemical treatment (Pre-chlorination with NaClO (15%) and sand filter) followed by ultraviolet (UV) disinfection (Post-chlorination with cationic electrolyte in a tank of accumulation and dosage). These processes allow improvement of physical, chemical and biological properties of wastewater similar to its natural state and to be returned to natural ecosystems (rivers or the sea). Water that has undergone a tertiary treatment in the wastewater treatment plant (reclaimed water) can be used for irrigation and moreover, it has a nutrient content (ammonia nitrogen, nitrates and nitrites, potassium and phosphorus) that can replace chemical fertilizers such as ammonium sulphate. Obviously different fertilizers can replace the nitrogen supplied by treated wastewater. However, following the LCA worst case criteria, we have used ammonium sulphate due to its lowest impact among the nitrogen fertilizers. The amount of the fertilizer replaced is calculated on the basis of the nitrogen content in the sludge and reclaimed water. It should be noted that this study does not take into account the energy needed to send water from the tertiary treatment to agricultural applications. This is because fields need to be irrigated regardless of where the water comes from. The traditional way of irrigating is to pump groundwater into the field. We assume that the impact of the pumping energy needed for irrigation can be compensated by impact of the energy needed to pump water from the tertiary treatment. The main difference between both irrigation options is the amount of fertilizer needed. Data for tertiary treatment operation and reclaimed water application are taken from a previous study (Meneses et al. 2010a; Pasqualino et al. 2011).

8.2.3. Scenario III: Drought scenario

Although water reuse strategies are intended to address the problem of water scarcity, sometimes other water sources such as seawater can help to solve the problem. Thus, Seawater Reverse Osmosis (SWRO) is introduced as an alternative technology (Figure 8.2).

This case study is based on scenario I, considering the wastewater treatment plant with tertiary treatment and reclaimed water from scenario II in conjunction with a seawater reverse osmosis desalination plant (SWRO) as an extra source of freshwater supply. It has been assumed that the water supply is composed of 75% from surface water treated at the potable water treatment plant and 25% from seawater treated at the SWRO plant. The SWRO operation also generates some brine which is returned to the sea. Data for the SWRO plant are taken from a previous study (Meneses et al. 2010b).

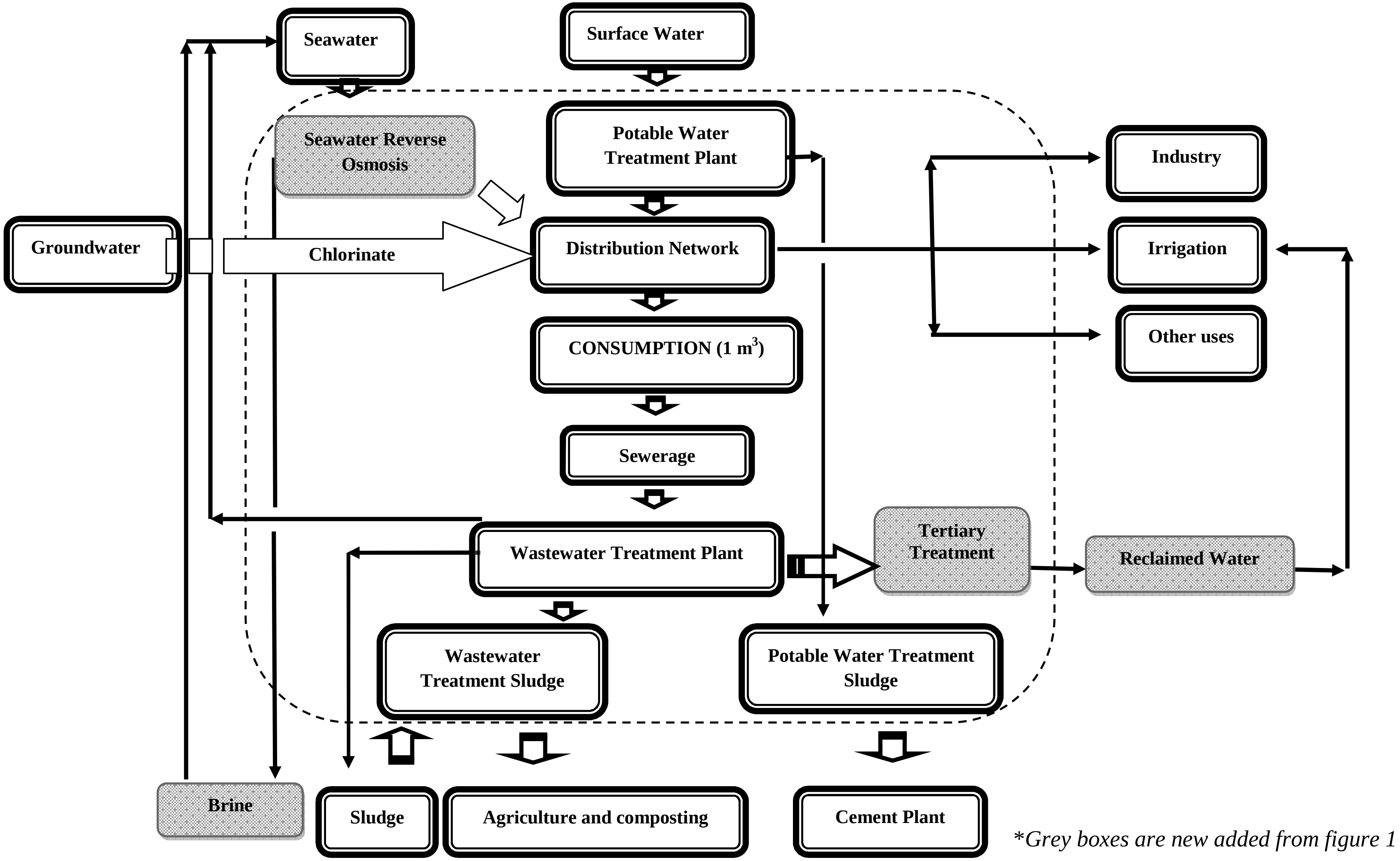


Figure 8.2. Urban water cycle with reclaimed water (scenario II) and seawater reverse osmosis (scenario III).

8.3. Material and methods

8.3.1. Life Cycle Assessment

The environmental assessment technique applied in this study is Life Cycle Assessment (LCA) which is an international standard methodology ISO14040 and ISO14044 (ISO 2006 a,b) for assessing the environmental impact of systems throughout their entire life cycles, beginning with the extraction of raw materials, continuing with product development and manufacturing, and ending with material recycling or disposal.

The LCA study is conducted in four steps: definition of goal and scope, inventory analysis, impact assessment and interpretation of results.

8.3.2. Definition of Goal and Scope

The main goal of this study is to assess the environmental profile of an urban water cycle in a Mediterranean city. The secondary objective is to compare the current scenario with two alternative scenarios (water reclamation and drought).

The Functional Unit (FU) used is 1m^3 of potable water supplied to the consumers in the Spanish Mediterranean area. A water intake of 1.2 m^3 has been calculated based on the 20% of water losses (Ramirez, 2009) which includes the water lost due to break downs, water leaks, maintenance operations in the distribution network, etc. All the processes, from water abstraction to final treatment are included within the system boundaries.

The infrastructure of the treatment plants' processes and products was not taken into account because the results of other studies indicate that the environmental impact of constructing and dismantling the infrastructure materials is negligible when compared to the operation phase (Raluy et al. 2006;

Muñoz and Rodríguez Fernández-Alba, 2008). However, the infrastructure (steel, fibre concrete, cast iron, reinforced concrete, galvanized iron, HDPE, LDPE and PVC) of the distribution network and sewage collection has been considered due to its relative significance.

8.3.3. Inventory Analysis

Inventory data, including energy and materials consumption, are presented in table 8.1. Data were obtained from site specific operating data collected from internal reports and personal interviews with the different plant staff, as well as some data from previous studies (Pasqualino et al., 2009; Meneses et al. 2010a, 2010b; Pasqualino et al. 2011). Data quality is assured by the accuracy of the plant's operating data and the reported deviation of values from the database.

All the energy and material flows were related to the inventory data provided by the Ecoinvent V2.1 database (Swiss Centre for Life-Cycle Inventories, 2009). These data were adapted to the Spanish energy consumption mix (Red Eléctrica Española, 2008).

The most significant energy and mass, input and output flows were considered in the inventory. The following items were considered: reagents used during different treatment stages, including packaging and transport of these reagents, treatment and/or final disposal of waste generated by water treatment plants, transport of this final waste to the treatment plants and the consumption of energy and materials for maintenance.

Table 8.1 Inventory Analysis of Urban Water Cycle (continued in next pages).

	Consumptions	Processes and Materials considered	Volume related to FU (m^3/m^3)
Water Abstraction	Water abstraction: 72.8x10^{+06} m^3 /year Energy: 2.94x10^{-01} kWh/m^3 Infrastructure: 1.25x10^{-03} kg steel/m^3, 1.51 x10^{-05} m^3 concrete/m^3	Pumping and piping materials from Ebro River	1.2*
Potable Water Treatment Plant	Potable water produced: 71.9x10^{+06} m^3/year Sludge produced: 4.09x10^{-02} kg/m^3 Energy Consumption: 7.10x10^{-02} kWh/m^3 Chemicals: 5.03x10^{-04} kg $KMnO_4$/m^3, 7.69x10^{-03} kg CO_2/m^3, 3.70 x10^{-03} kg $NaClO_2$/m^3, 1.44x10^{-02} kg $FeCl_3$/m^3, 7.69x10^{-04} kg Polyelectrolyte PoliDadmac/m^3, 1.51x10^{-04} kg sand/m^3, 1.65x10^{-02} kg granulate active carbon/m^3, 2.31x10^{-03} kg Cl_2/m^3	Water line: Preoxidation, Coagulation, Flocculation, Decantation, Sand filters, Active Carbon filters, Post-chloration Sludge line: Collection, Thickening, Sludge drying, Drying zone 100% sludge sent to cement plant	1.2*
Intermediate pumping	Energy Consumption: 1.87x10^{-01} kWh/m^3 Infrastructure: 3.76x10^{-03} kg steel/m^3, 4.50x10^{-05} kg concrete/m^3	Pumping from PWTP to consumer	1.2*
Distribution network	Water: 9.52x10^{+06} m^3/year Energy Consumption: 2.94x10^{-01} kWh/m^3 Chemicals: 4.90x10^{-03} kg Cl_2/m^3 Infrastructure: 4.23x10^{-03} kg steel/m^3, 2.03x10^{-01} kg fibre concrete/m^3, 8.88x10^{-01} kg cast iron/m^3, 1.06x10^{-04} kg reinforced concrete/m^3, 1.06x10^{-04} kg galvanized iron/m^3, 2.25x10^{-03} kg HDPE/m^3, 2.24x10^{-04} kg LDPE/m^3, 2.82x10^{-05} kg PVC/m^3.	354 km piping and 6 supply tanks	1
Sewerage	Wastewater: 9.50x10^{+06} m^3/year Sludge Waste: 8.90x10^{-01} kg/m^3 Infrastructure: 6.71x10^{-04} kg steel/m^3, 4.92x10^{-02} kg fibre concrete, 1.44x10^{-01} kg cast iron/m^3, 1.31x10^{-02} kg PVC/m^3.	37 km piping	1

Wastewater Treatment Plant [Pasqualino et al. 2009]	Wastewater treated: $10.2\text{x}10^{+06}$ m^3/year Sludge: $6.67\text{x}10^{-01}$ kg/m^3 Energy Consumption: 1.09 kWh/m^3 Chemicals: $2.45\text{x}10^{-02}$ kg $FeCl_3$/m^3, $1.76\text{x}10^{-03}$ kg polyelectrolite /m^3.	Water Line: bar screen, sand chamber/degreaser, primary settler, anaerobic reactor, aerobic reactor. secondary settler Sludge Line: primary sludge sieve, gravity thickener, flotation thickener, mixing chamber, anaerobic digester, tampon storage, centrifuge dehydration, final storage, final disposal 99.86% of the sludge is allocated for agricultural uses, 0.14% is allocated to a compost plant	1
Scenario II: Reclaimed water			
Tertiary Treatment [Meneses et al. 2010a; Pasqualino et al. 2011]	Water Treatment: 4.33 $\text{x}10^{+06}$ m^3/year Chemicals: $1.18\text{x}10^{-02}$ kg/m^3 de NaClO, $1.99\text{x}10^{-03}$ kg/m^3 cationic electrolyte	Water Line: Pumping, sand filters, UV Treatment, coloration	1
Reclaimed water [Meneses et al 2010a; Pasqualino et al 2011]	Water Application from Tertiary Treatment: $1.37\text{x}10^{+05}$ m^3/year Energy Consumption: $7.91\text{x}10^{-01}$ kWh/ m^3	Irrigation in agriculture with Tertiary Treatment water: 31 ha	1
Scenario III: Drought Scenario			
Seawater Reverse Osmosis (SWRO) Desalination Plant [Meneses et al 2010b]	Seawater Treatment: $9.24\text{x}10^{+06}$ m^3/year Water Production: $4.58\text{x}10^{+06}$ m^3/year Brine: $4.66\text{x}10^{+06}$ m^3 brine/year Energy Consumption: 5.69 kWh/m^3	Water Line: Water collection, Potable Water Treatment Plant and Seawater Reverse Osmosis, Storage tank Pretreatment, Reagents dosage, Reverse osmosis, Post-treatment. Final brine Destination: Send to seawater Services and maintenance	0.9 from Potable Water Treatment Plant 0.3 from Seawater Reverse Osmosis

Note: Functional Unit (FU) = 1m^3 water supplied to consumers. *= The 20% water losses include the water lost due to break downs, water leaks, maintenance operations in the distribution network, etc. The distribution losses are included in the study.

8.3.4. Impact Assessment at midpoint level

CML2001 (Centre for Environmental Studies, 2001) was the chosen impact assessment method. The impact categories considered are: AP (Acidification Potential, global, kg SO_2-eq.), GWP (Global Warming Potential, kg CO_2-eq.), EP (Eutrophication Potential, global, kg PO_4-eq.), PHO (Photochemical Oxidation, kg formed ozone), DAR (Depletion of Abiotic Resources, kg Antimony- Eq.), ODP (Ozone Depletion Potential, kg CFC-11 eq.) and ETP (Ecotoxicity Potential). Two additional environmental indicators have also been used: Freshwater Use (FWU, m^3) and Cumulative Energy Demand (CED, MJ). The Freshwater use indicator shows the amount of water consumed from the different sources (surface and groundwater) throughout the whole life cycle. This indicator is especially significant in Spain because of the problems of local water availability. The cumulative energy demand indicator is a direct and easily understood indicator of the environmental implications of energy consumption, especially fossil fuels. This work also assesses other impact category developed by Muñoz et al. (2010) who propose ways of quantifying the impacts of water use. The Freshwater Ecosystem Impact (FEI) aims to assess the ecological consequences of water use in a certain region measured as the volume of "ecosystem-equivalent" water, which refers to the volume of water likely to affect freshwater ecosystems. FEI has been calculated using the equation (8.1):

$$FEI = \sum_{i=1}^{n} (CWU_i \cdot WTA_i) \qquad \text{Equation (8.1)}$$

Where CWU_i is consumptive water use of a unit process (m^3), WTA_i is the withdrawal to availability ratio of the river basin where water is consumed, and and n is the set of unit processes involved in the product system. The unit of Freshwater Ecosystem Impact refers to volume (m^3) of ecosystem equivalent per unit volume (m^3) of available water. The characterization factor, WTA, is defined for Ebro Basin by Milà i Canals et al. (2009) with a value of 0.46 (m^3 ecosystem equivalent/ m^3 water availability).

8.4. Results and discussion

8.4.1. Scenario I

The scenario I shows the environmental impact of Urban Water Cycle for a Mediterranean city. All these calculations have been done based on operational data therefore results of scenario I reflects the current situation.

Table 8.2 present the environmental evaluation of the scenario I (current situation) for the steps considered in the urban water cycle.

Table 8.2 Scenario I: Environmental impacts and contribution in different stages of Urban Water Cycle in Mediterranean area.

	Water Abstraction	Potable Water Treatment Plant	Intermediate Pumping	Distribution Network	Sewerage	Wastewater Treatment Plant	Total
AP (kg SO$_2$-Eq)	$1.40x10^{-03}$	$6.72x10^{-04}$	$1.04x10^{-03}$	$1.89x10^{-03}$	$3.33x10^{-05}$	$1.58x10^{-03}$	$6.62x10^{-03}$
%	**21.2**	**10.2**	**15.7**	**28.6**	**0.5**	**23.9**	**100**
GWP (kg CO$_2$-Eq)	$1.77x10^{-01}$	$1.01x10^{-01}$	$1.54x10^{-01}$	$3.04x10^{-01}$	$8.91x10^{-03}$	$1.19x10^{-01}$	$8.64x10^{-01}$
%	**20.5**	**11.7**	**17.8**	**35.2**	**1.0**	**13.8**	**100**
EP (kg PO$_4$-Eq)	$1.08x10^{-04}$	$6.87x10^{-05}$	$8.02x10^{-05}$	$1.57x10^{-04}$	$4.29x10^{-06}$	$1.27x10^{-03}$	$1.69x10^{-03}$
%	**6.4**	**4.1**	**4.8**	**9.3**	**0.3**	**75.2**	**100**
PHO (kg O$_3$)	$1.05x10^{-05}$	$1.17x10^{-05}$	$1.10x10^{-05}$	$4.15x10^{-05}$	$3.45x10^{-06}$	$2.16x10^{-05}$	$9.98x10^{-05}$
%	**10.5**	**11.7**	**11.0**	**41.6**	**3.5**	**21.7**	**100**
DAR (kg Antimony-Eq)	$1.33x10^{-03}$	$7.21x10^{-04}$	$9.41x10^{-04}$	$1.96x10^{-03}$	$6.90x10^{-05}$	$1.74x10^{-03}$	$6.76x10^{-03}$
%	**19.7**	**10.7**	**13.9**	**29.0**	**1.0**	**25.7**	**100**
ODP (kg CFC-11-Eq)	$1.55x10^{-08}$	$2.68x10^{-08}$	$9.74x10^{-09}$	$2.81x10^{-08}$	$4.12x10^{-10}$	$3.64x10^{-08}$	$1.17x10^{-07}$
%	**13.3**	**22.9**	**8.3**	**24.0**	**0.4**	**31.1**	**100**
ETP (kg 1.4-DCB-Eq)	$3.04x10^{-01}$	$2.44x10^{-01}$	$2.52x10^{-01}$	$7.88x10^{-01}$	$5.76x10^{-02}$	$6.29x10^{-01}$	$2.27x10^{+00}$
%	**13.4**	**10.7**	**11.1**	**34.6**	**2.5**	**27.7**	**100**
FEI (m^3)	$5.40x10^{-01}$	$9.77x10^{-04}$	$4.55x10^{-04}$	$9.41x10^{-04}$	$2.97x10^{-05}$	$6.08x10^{-04}$	$5.43x10^{-01}$
%	**99.4**	**0.2**	**0.1**	**0.2**	**0.0**	**0.1**	**100**
CED (MJ)	$3.90x10^{+00}$	$2.45x10^{+00}$	$2.63x10^{+00}$	$5.08x10^{+00}$	$1.41x10^{-01}$	$5.82x10^{+00}$	$2.00x10^{+01}$
%	**19.5**	**12.2**	**13.1**	**25.4**	**0.7**	**29.1**	**100**

Note: Functional Unit (FU) = 1m^3 water supplied to consumers. AP = Acidification Potential; GWP = Global Warming Potential; EP = Eutrophication Potential; PHO = Photochemical Oxidation; DAR = Depletion of Abiotic Resources; ODP = Ozone Depletion Potential; ETP = Ecotoxicity Potential; FEI= Freshwater Ecosystem Impact; CED= Cumulative Energy Demand.

Overall, the main environmental impact of the urban water cycle occurs during the distribution network stage (25 to 42% of the total impact), except for Eutrophication Potential (EP) and Freshwater Ecosystem Impact (FEI). Nevertheless, if we add the contribution from abstraction and intermediate pumping, it also can be observed that, except EP and FEI, the impacts are in the 21 to 37% range. In all the cases, these high impacts are due to energy consumption, mainly in pumping processes. These results reflect the fact that impacts are highly dependent on the orography and on the distance from abstraction sites to the consumer.

The category of Eutrophication Potential has the most environmental impact during the WWTP (~75%) due to the high amount of nutrients from use of sewage sludge on arable land. The use of sewage sludge in agriculture avoids the production of synthetic fertilizer (Pasqualino et al. 2009) and save energy, raw materials and emissions which provides potential environmental benefits. However, applying nutrient rich sewage sludge in soils has an effect on EP due to the composition of the sludge (nutrient excess). In addition, other impact categories such as Stratospheric Ozone Depletion (~31%) and Cumulative Energy Demand (~29%) have a high impact in this stage due to the chemicals and energy needed to treat wastewater.

The high impact of Freshwater Ecosystem impact in water abstraction is caused by direct consumption of 1.2 m^3 of water. The impacts of other stages are negligible.

8.4.2. Scenario II: Reclaimed water

This scenario evaluates an alternative for saving potable water consumption considering a reclaimed water application. The main function of a wastewater treatment plant is to minimize the environmental impact of discharging untreated water into natural water systems. This scenario also shows the use of the tertiary treatment to reclaim water for application in agriculture.

Table 8.3 presents the values of the different environmental indicators for the steps considered in the Urban Water Cycle with the reclaimed water scenario and it also presents the relative contributions of these steps.

Scenario I goes from water abstraction in the Potable Water Treatment Plant (PWTP) to secondary treatment in Wastewater Treatment Plant (WWTP). While scenario II covers the same boundaries as scenario I but additionally, includes reuse phase with a tertiary treatment and irrigation in agriculture.

Except for the impact of Freshwater Ecosystem in water abstraction, the stages with the most impact in this scenario are WWTP (22 to 71%) and distribution network (25 to 30%), mainly because of energy consumption. In the case of Eutrophication Potential, the highest contribution of 71% is due to the impact of WWTP sludge sent for agricultural applications.

The impacts related to tertiary treatment have the same order of magnitude as potable water treatment, we obtained the same results in previous work (Meneses et al. 2010a). A comparison of the results in Table 8.2 (scenario I) and Table 8.3 (scenario II) shows that only the impact of Freshwater Use is significantly reduced when water is reclaimed (from 1.21 m^3 to 0.21 m^3). The values for all other impacts are mostly similar to scenario I.

For the Freshwater Use indicator, the impact of 1.2 m^3 in the water abstraction is balanced by the -1 m^3 for reclaimed water used in agriculture, with a final net result of 0.2 m^3 of freshwater consumption. Table 8.3 shows the relative contribution of the different stages of the water urban cycle with tertiary treatment and reclaimed water. Thus, when local freshwater is scarce, the use of reclaimed water is the best option as the investment in the tertiary treatment and slight increase in other environmental indicators, there is a net saving of freshwater. This option saves groundwater abstracted from wells used in common practices for irrigation.

Table 8.3 Scenario II: Environmental impacts and contribution in different stages of Urban Water Cycle with reclaimed water application.

	Water Abstraction	Potable Water Treatment Plant	Intermediate Pumping	Distribution Network	Sewerage	Wastewater Treatment Plant	Reuse		Total
							Tertiary Treatment	Application in Agriculture	
AP (kg SO_2-Eq)	$1.40x10^{-03}$	$6.72x10^{-04}$	$1.04x10^{-03}$	$1.89x10^{-03}$	$3.33x10^{-05}$	$1.58x10^{-03}$	$9.08x10^{-04}$	$-1.22x10^{-05}$	$7.51x10^{-03}$
%	**18.6**	**8.9**	**13.8**	**25.2**	**0.4**	**21.0**	**12.1**	**-0.2**	**100**
GWP (kg CO_2-Eq)	$1.77x10^{-01}$	$1.01x10^{-01}$	$1.54x10^{-01}$	$3.04x10^{-01}$	$8.91x10^{-03}$	$1.19x10^{-01}$	$1.88x10^{-01}$	$-2.99x10^{-03}$	$1.05x10^{+02}$
%	**16.9**	**9.6**	**14.7**	**29.0**	**0.8**	**11.3**	**17.9**	**-0.3**	**100**
EP (kg PO_4-Eq)	$1.08x10^{-04}$	$6.87x10^{-05}$	$8.02x10^{-05}$	$1.57x10^{-04}$	$4.29x10^{-06}$	$1.27x10^{-03}$	$9.43x10^{-05}$	$-2.38x10^{-06}$	$1.78x10^{-03}$
%	**6.1**	**3.9**	**4.5**	**8.8**	**0.2**	**71.3**	**5.3**	**-0.1**	**100**
PHO (kg O_3)	$1.05x10^{-05}$	$1.17x10^{-05}$	$1.10x10^{-05}$	$4.15x10^{-05}$	$3.45x10^{-06}$	$2.16x10^{-05}$	$4.77x10^{-05}$	$-3.27x10^{-07}$	$1.47x10^{-04}$
%	**7.1**	**8.0**	**7.5**	**28.2**	**2.3**	**14.7**	**32.4**	**-0.2**	**100**
DAR (kg Antimony-Eq)	$1.33x10^{-03}$	$7.21x10^{-04}$	$9.41x10^{-04}$	$1.96x10^{-03}$	$6.90x10^{-05}$	$1.74x10^{-03}$	$2.29x10^{-03}$	$-2.75x10^{-05}$	$9.02x10^{-03}$
%	**14.7**	**8.0**	**10.4**	**21.7**	**0.8**	**19.3**	**25.4**	**-0.3**	**100**
ODP (kg CFC-11-Eq)	1.5510-08	$2.68x10^{-08}$	$9.74x10^{-09}$	$2.81x10^{-08}$	$4.12x10^{-10}$	$3.64x10^{-08}$	$1.57x10^{-08}$	$-4.55x10^{-10}$	$1.32x10^{-07}$
%	**11.7**	**20.3**	**7.4**	**21.3**	**0.3**	**27.5**	**11.9**	**-0.3**	**100**
ETP (kg 1.4-DCB-Eq)	$3.04x10^{-01}$	$2.44x10^{-01}$	$2.52x10^{-01}$	$7.88x10^{-01}$	$5.76x10^{-02}$	$6.29x10^{-01}$	$2.88x10^{-01}$	$-6.51x10^{-03}$	$2.56x10^{+00}$
%	**11.9**	**9.5**	**9.9**	**30.8**	**2.3**	**24.6**	**11.3**	**-0.3**	**100**
FEI (m^3)	$5.40x10^{-01}$	$9.77x10^{-04}$	$4.55x10^{-04}$	$9.41x10^{-04}$	$2.97x10^{-05}$	$6.08x10^{-04}$	$4.55x10^{-04}$	$-4.50x10^{-01}$	$9.35x10^{-02}$
%	**577.8**	**1.0**	**0.5**	**1.0**	**0.0**	**0.6**	**0.5**	**-481.5**	**100**
CED (MJ)	$3.90x10^{+00}$	$2.45x10^{+00}$	2.63x10+00	$5.08x10^{+00}$	$1.41x10^{-01}$	$5.82x10^{+00}$	$5.52x10^{+00}$	$-5.99x10^{-02}$	$2.55x10^{+01}$
%	**15.31**	**9.61**	**10.32**	**19.94**	**0.55**	**22.84**	**21.66**	**-0.24**	**100**

Note: Functional Unit (FU) = 1m³ water supplied to consumers. AP = Acidification Potential; GWP = Global Warming Potential; EP = Eutrophication Potential; PHO = Photochemical Oxidation; DAR = Depletion of Abiotic Resources; ODP = Ozone Depletion Potential; ETP = Ecotoxicity Potential; FEI= Freshwater Ecosystem Impact; CED= Cumulative Energy Demand.

8.4.3. Scenario III: Drought Scenario

Scenario III is the extension of scenario II with an addition of water production alternative for extreme water scarcity situation. A desalination plant is considered as an alternative water source to obtain potable water. As in scenario II, this scenario also considers reclaimed water as away for saving potable water consumption.

The results in table 8.4 show that the Seawater Reverse Osmosis (SWRO) desalination plant has the highest impact in all categories except in Eutrophication Potential (EP) and Photochemical Oxidation (PHO). This is because of the high energy consumption throughout the desalination process (5.69 $kWh \cdot m^{-3}$). Wastewater treatment has the maximum impact in EP due to nutrient content, as in previous scenarios, and the highest impact for PHO in tertiary treatment, due to the use of chlorine for water disinfection. It should be noted that brine from SWRO and sludge from wastewater and potable treatment plants have been accounted in the total impact. Also, Table 8.4 shows the relative contributions of different processes in scenario III to the corresponding impacts.

Due to locational advantage of the city of Tarragona and its proximity to Mediterranean Sea, the most suitable alternative was the SWRO. However, Scenario III is the worst option from an environmental point of view because of high environmental cost of SWRO technology as it has high energy consumption per m^3 of water produced (Meneses et al. 2010b; Muñoz and Rodríguez Fernández-Alba, 2008). Nevertheless, in a region with severe problem of water scarcity, water desalination could be the only technology for ensuring that water is delivered to the population and industry. In this instance, increasing process efficiency and reducing the environmental impact of energy consumption by including a higher proportion of renewable energy will reduce the environmental impact of desalination. It is necessary to take into account that the technology applied is currently used in Telde (Meneses et al. 2010b).

Three scenarios have been assessed taking into account the damage in the ecosystems due to use of freshwater. Milà i Canals et al. (2009) and Muñoz et al. (2010) developed a characterization factor for Ebro basin which is 0.46. Then, taking into account this characterization factor and the freshwater consumption in each scenario, the impacts on ecosystems has been assessed.

Scenario I has a total freshwater consumption of 1.21 m^3 while scenario II represents 2.08E-01 m^3 of total freshwater. The impacts on ecosystems are 5.55E-01 m^3 for scenario I and 9.55E-02 m^3 for scenario II. So, scenario II represents approximately an 83% of impact reduction from scenario I due to the water reutilization in this scenario.

Scenario III also has beneficial values due to saving of freshwater consumption by the incorporation of desalination plant as additional source of potable water. However, this measure means an increase in energy consumption which results in a growth of the Global Warming Potential.

Table 8.4 Scenario III: Environmental impacts and contribution in different stages of Urban Water Cycle with reclaimed water application and Desalination.

	Water Abstraction	Potable Water Treatment Plant	Seawater Reverse Osmosis	Intermediate Pumping	Distribution Network	Sewerage	Wastewater Treatment Plant	Reuse		Total
								Tertiary Treatment	Application Agriculture	
AP (kg SO_2-Eq)	1.05x10^{-03}	5.04x10^{-04}	6.30x10^{-03}	7.77x10^{-04}	1.89x10^{-03}	3.33x10^{-05}	1.58x10^{-03}	9.08x10^{-04}	-1.22x10^{-05}	1.30x10^{-02}
%	**8.1**	**3.9**	**48.3**	**6.0**	**14.5**	**0.3**	**12.1**	**7.0**	**-0.1**	**100**
GWP (kg CO_2-Eq)	1.33x10^{-01}	7.58x10^{-02}	7.58x10^{-01}	1.15x10^{-01}	3.04x10^{-01}	8.91x10^{-03}	1.19x10^{-01}	1.88x10^{-01}	-2.99x10^{-03}	1.70x10^{+00}
%	**7.8**	**4.5**	**44.6**	**6.8**	**17.9**	**0.5**	**7.0**	**11.1**	**-0.2**	**100**
EP (kg PO_4-Eq)	8.06x10^{-05}	5.15x10^{-05}	5.41x10^{-04}	6.02x10^{-05}	1.57x10^{-04}	4.29x10^{-06}	1.27x10^{-03}	9.43x10^{-05}	-2.38x10^{-06}	2.26x10^{-03}
%	**3.6**	**2.3**	**24.0**	**2.7**	**7.0**	**0.2**	**56.3**	**4.2**	**-0.1**	**100**
PHO (kg O_3)	7.89x10^{-06}	8.81x10^{-06}	4.15x10^{-05}	8.23x10^{-06}	4.15x10^{-05}	3.45x10^{-06}	2.16x10^{-05}	4.77x10^{-05}	-3.27x10^{-07}	1.80x10^{-04}
%	**4.4**	**4.9**	**23.0**	**4.6**	**23.0**	**1.9**	**12.0**	**26.4**	**-0.2**	**100**
DAR (kg Antimony-Eq)	1.00x10^{-03}	5.41x10^{-04}	5.73x10^{-03}	7.06x10^{-04}	1.96x10^{-03}	6.90x10^{-05}	1.74x10^{-03}	2.29x10^{-03}	-2.75x10^{-05}	1.40x10^{-02}
%	**7.1**	**3.9**	**40.9**	**5.0**	**14.0**	**0.5**	**12.4**	**16.3**	**-0.2**	**100**
ODP (kg CFC-11-Eq)	1.16x10^{-08}	2.01x10^{-08}	7.02x10^{-08}	7.31x10^{-09}	2.81x10^{-08}	4.12x10^{-10}	3.64x10^{-08}	1.57x10^{-08}	-4.55x10^{-10}	1.89x10^{-07}
%	**6.1**	**10.6**	**37.1**	**3.9**	**14.8**	**0.2**	**19.2**	**8.3**	**-0.2**	**100**
ETP (kg 1.4-DCB-Eq)	2.28x10^{-01}	1.83x10^{-01}	1.30x10^{+00}	1.89x10^{-01}	7.88x10^{-01}	5.76x10^{-02}	6.29x10^{-01}	2.88x10^{-01}	-6.51x10^{-03}	3.66x10^{+00}
%	**6.2**	**5.0**	**35.6**	**5.2**	**21.6**	**1.6**	**17.2**	**7.9**	**-0.2**	**100**
FEI (m^3)	4.05x10^{-01}	7.34x10^{-04}	-1.50x10^{-01}	3.40x10^{-04}	9.41x10^{-04}	2.97x10^{-05}	6.08x10^{-04}	4.55x10^{-04}	-4.50x10^{-01}	-1.91x10^{-01}
%	**-212.0**	**-0.4**	**78.3**	**-0.2**	**-0.5**	**0.0**	**-0.3**	**-0.2**	**235.2**	**100**
CED (MJ)	2.92x10^{+00}	1.84x10^{+00}	1.67x10^{+01}	1.97x10^{+00}	5.08x10^{+00}	1.41x10^{-01}	5.82x10^{+00}	5.52x10^{+00}	-5.99x10^{-02}	3.99x10^{+01}
%	**7.3**	**4.6**	**41.8**	**4.9**	**12.7**	**0.4**	**14.6**	**13.8**	**-0.2**	**100**

Note: Functional Unit (FU) = 1m^3 water supplied to consumers. AP = Acidification Potential; GWP = Global Warming Potential; EP = Eutrophication Potential; PHO = Photochemical Oxidation; DAR = Depletion of Abiotic Resources; ODP = Ozone Depletion Potential; ETP = Ecotoxicity Potential; FEI= Freshwater Ecosystem Impact; CED= Cumulative Energy Demand

8.4.4. Comparative Scenarios Analysis

In summary, Table 8.5 shows the quantitative comparison of the total impact assessment for each impact category in scenarios I, II and III. The overall category impacts increased in scenario II (e.g. Global Warming Potential at 21.5%) and increased even more in scenario III, which has a 96.8% higher impact than scenario I. However, the freshwater use indicator shows a higher impact in scenario I than in scenarios II and III because the use of reclaimed water in these two latter scenarios prevents water from being taken from natural resources for use in common non-potable practices. This relative contribution is lower in scenario III than in scenario II because in scenario III only 75% of freshwater comes from surface and groundwater, the other 25% coming from seawater treated at the SWRO plant.

Table 8.5 also includes the impacts of proposed mix of electric use in 2050 for each scenario with an assumption of 30% renewable energy (15% of photovoltaic and 15% of wind power). We can observe that scenario *Ib* decreases in relation to *Ia* in all the impact categories approximately in the range 18 – 47 % except for Freshwater Ecosystem Impact, FEI, indicator, where the reduction is only of 0.3%. Results indicate the high influence of energy consumption and how environmental impacts are dependent of the quality of electricity. In the case of FEI, the type of electricity used has much less influence on the water consumption along the life cycle. The same trend is observed in Scenario II where impacts, decreases from 22 to 54% (except for FEI) when the electricity mix is improved by increasing the proportion of renewable. In Scenario III one-fourth of water supply comes from desalination units with a substantial increase of electricity consumption, thus all the absolute value of impacts are higher than scenarios I and II, except for FEI, with a negative value because there is a net saving of potable water. In this case, the reduction in the value of environmental impacts (again except for FEI) by the use of renewable electricity mix is obviously higher with values from about 55 to 103 %.

Table 8.6 shows a qualitative comparison of the advantages and disadvantages of three scenarios. The most important advantage of scenario I is its lower energy impact than scenarios II and III. However, this scenario assumes sufficient rainfall to meet water demands. On the other hand, scenario II includes a tertiary treatment and uses reclaimed water in agricultural irrigation. Consequently, its main advantage is that it avoids excess use of freshwater. The worst disadvantage in scenario III is its high energy consumption, although this scenario is an alternative to satisfy the water demand in situations of extremes scarcity. Furthermore, an improvement in the yield from seawater reverse osmosis may helps to decrease energy consumption in this scenario.

Table 8.5 Quantitative comparison from Scenarios I, II and III.

	AP (kg SO_2-Eq)	GWP (kg CO_2-Eq)	EP (kg PO_4-Eq)	PHO (kg O_3)	DAR (kg Antimony-Eq)	ODP (kg CFC-11-Eq)	ETP (kg 1.4-DCB-Eq)	FEI (m^3)	CED (MJ)
Scenario Ia [a]	6.61x10^{-03}	8.64x10^{-01}	1.68x10^{-03}	9.98x10^{-05}	6.76x10^{-03}	1.17x10^{-07}	2.28x10^{+00}	5.43x10^{-01}	2.00x10^{+01}
Scenario Ib [b]	4.23x10^{-03}	4.63x10^{-01}	1.38x10^{-03}	7.71x10^{-05}	3.58x10^{-03}	8.15x10^{-08}	1.43x10^{+00}	5.42x10^{-01}	1.41x10^{+01}
% Decreased	36.0	46.4	18.1	22.7	47.0	30.3	37.0	0.3	29.6
Scenario IIa [a]	7.50x10^{-03}	1.05x10^{+00}	1.78x10^{-03}	1.47x10^{-04}	9.02x10^{-03}	1.32x10^{-07}	2.56x10^{+00}	9.35x10^{-02}	2.55x10^{+01}
Scenario IIb [b]	4.15x10^{-03}	4.84x10^{-01}	1.35x10^{-03}	1.15x10^{-04}	4.55x10^{-03}	8.22x10^{-08}	1.37x10^{+00}	9.14x10^{-02}	1.71x10^{+01}
% Decreased	44.7	53.8	24.1	21.7	49.6	37.8	46.4	2.2	32.8
Scenario IIIa[a]	1.30x10^{-02}	1.70x10^{+00}	2.25x10^{-03}	1.80x10^{-04}	1.40x10^{-02}	1.89x10^{-07}	3.65x10^{+00}	-1.91x10^{-01}	3.99x10^{+01}
Scenario IIIb[b]	2.67x10^{-03}	-4.35x10^{-02}	9.30x10^{-04}	8.17x10^{-05}	1.95x10^{-04}	3.52x10^{-08}	-1.56x10^{-03}	-1.98x10^{-01}	1.42x10^{+01}
% Decreased	79.5	102.6	58.8	54.7	98.6	81.4	100.0	-3.3	64.5

Note: Functional Unit (FU) = 1m^3 water supplied to consumers. AP = Acidification Potential; GWP = Global Warming Potential; EP = Eutrophication Potential; PHO = Photochemical Oxidation; DAR = Depletion of Abiotic Resources; ODP = Ozone Depletion Potential; ETP = Ecotoxicity Potential; FEI= Freshwater Ecosystem Impact; CED= Cumulative Energy Demand.

[a] Results with Spanish Electric Mix

[b] Results with a Renewable Spanish Electric Mix

Table 8.6 Qualitative comparison of Scenarios I, II and III.

Scenarios	Advantages	Disadvantages
Scenario I	Fewer environmental impacts than Scenarios II and III	Requires high water reserves in order to always satisfy demand
Scenario II	Net saving of freshwater (reclaimed water used for non-potable uses)	Energy for tertiary treatment
Scenario III	Ensures water in situations of scarcity Net saving of freshwater	Energy for tertiary treatment and desalination

8.5. Conclusions

This chapter presents a case study of an urban water cycle as a possible methodology for calculating the environmental impacts of water supply and treatment systems using LCA approach. Such a study should take into account the needs of the population, the availability of resources and the environmental loads of the different stages of the urban water cycle. Furthermore, other future research would be a truly integrated system including the use of water saving technology in households, industry and agriculture.

The most important contribution of this study is a holistic conception of the urban water cycle demonstrated by the inclusion of the case study based on real operation data, taking into account a water supply system of a city considering water abstraction, potable water treatment, distribution network, wastewater treatment, reclaimed water and desalination. Generally, it has been observed that energy consumption is the main cause of the impacts.

In all three scenarios the main source of impact is the energy consumed through the collection and intermediate pumping of freshwater. Because of this, local orography and the distance from treatment to consumer are important aspects to take into account when designing the urban water cycle.
Scenario III, drought scenario, also considered a hypothetical drought situation where, in addition to reclaiming water, a seawater desalination plant was

included to guarantee the water supply. A desalination plant should be only used when surface and groundwater are extremely scarce due to high energy demand. However, if we use a renewable energy, a practically neutral carbon option (-4.35x10^{-02} kg CO_2 eq·m^{-3} water) is obtained. In that case, water reuse in agriculture with consumption of electricity from more renewable resources compensates the high impact of desalination process. So, in times of drought or water shortage, it is necessary to supply the population with water from desalination plants, in spite of their high energy consumption and high impact.

Appendix A8

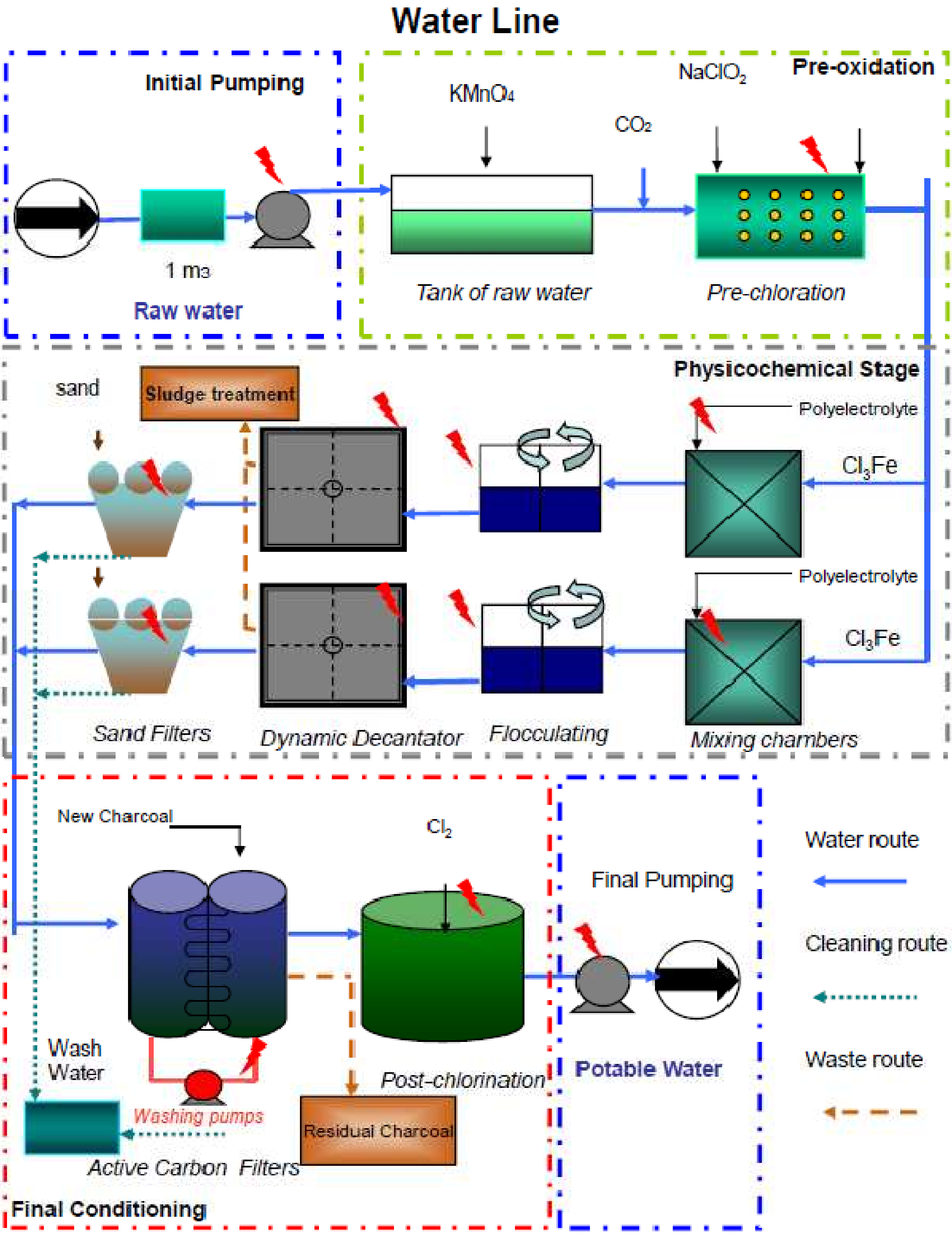


Figure A8.1 Water line in Potable Water Treatment Plant (SOSTAQUA 2010).

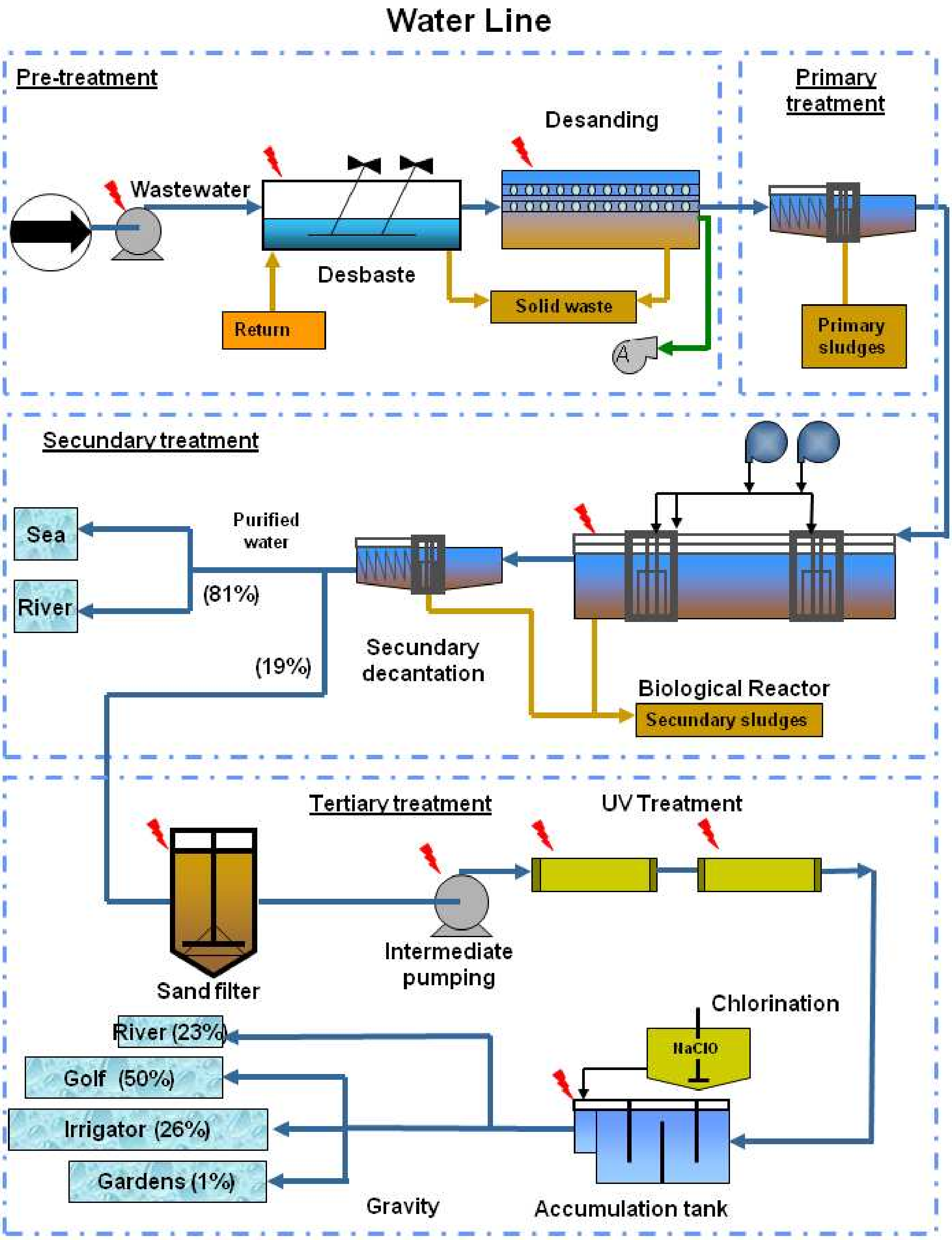


Figure A8.2 Water line in Wastewater Treatment Plan (SOSTAQUA 2010).

Part V

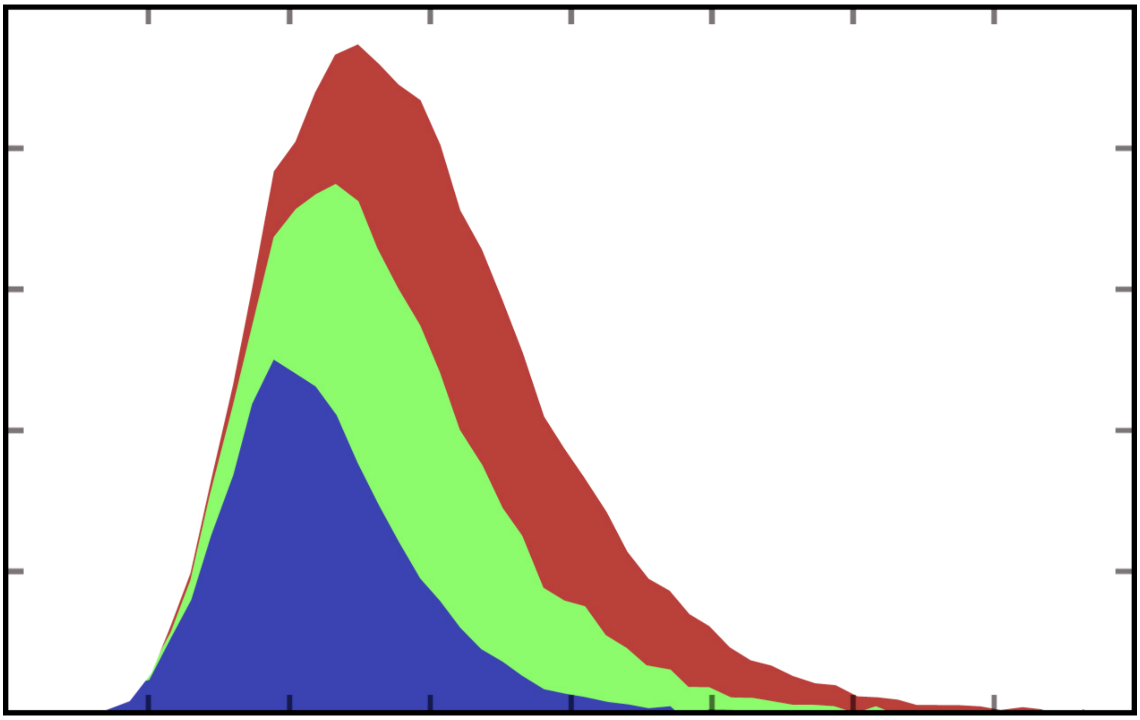
Uncertainty Analysisis

Chapter 9

Framework for decision-making under probabilistic uncertainties in LCA studies

María José Amores Barrero, Gonzalo Guillén Gosálbez,

Francesc Castells Piqué



Chapter 9 is based on the following paper:

Amores MJ, Guillén G, Castells F. Framework for decision- making under probablistic uncertainties in LCA studies. International Journal of Life Cycle Assessment. (Under review from January 2013).

Abstract

Purpose: Uncertainty analysis has recently gained wider interest in LCA. Most of the work on LCA under uncertainty has focused on quantifying its effect on the outcome of the LCA analysis. In contrast, little has been published on how to address the associated decision-making process under uncertainty. This work proposes a rigorous approach to aid practitioners in the selection of alternatives when the LCA analysis is affected by several sources of uncertainty. Given a set of options aiming at reducing the environmental impact of a process, our goal is to develop a systematic tool for facilitating the identification of those showing the best performance in the space of uncertain parameters.

Methods: We propose a methodology for decision-making under uncertainty in LCA that comprises several steps. We focus herein on analyzing probabilistic uncertainties, that is, uncertain data that can be described through probabilistic functions. First, the main sources of uncertainty are identified and characterized using probability distributions. Representative scenarios of the uncertain parameters space are generated next from these distributions via stochastic sampling techniques. Using these scenarios, we construct probability curves that describe the LCA performance under uncertainty. These curves are filtered using concepts from multi-objective optimization in order to discard suboptimal solutions. Several risk metrics widely used in operations and finances are then calculated from the curves kept for further inspection after the filtering step. The analysis of these risk metrics allows identifying robust solutions showing low probabilities of unfavourable outcomes.

Results: As a proof-of-concept, we have applied our methodology to assess several technologies for energy generation. The uncertainty sources were described by lognormal distributions using the Pedigree matrix. The methodology proposed allows eliminating suboptimal alternatives with poor performance, but which are difficult to discard in a straightforward manner using conventional approaches. Furthermore, by analyzing the risk metrics, we get valuable insight into the LCA results in the face of uncertainty.

Conclusions: The analysis of cumulative curves allows discarding suboptimal alternatives in LCA studies with imprecise information. Furthermore, by using risk metrics, we can identify robust solutions that minimize the impact of unfavourable scenarios.

Keywords: risk management, probabilistic uncertainty, decision-making, life cycle assessment, LCIA, framework

9.1 Introduction

Standard LCA studies assess the environmental performance of a set of alternatives with the goal to identify the one with the best performance. In practice, LCA calculations are affected by different types of uncertainties, so the selection of the best option is far from being a straightforward task. Several methods are available in the literature for identifying and characterizing the main sources of uncertainty in LCA analysis (see, for instance, Benetto et al. 2006; Björklund et al. 2002; Ciroth et al. 2004; Frischknecht et al. 2007; Hertwich et al. 2000; Hong et al. 2010; Huijbregts 1998; Huijbregts et al. 2001, 2003; Hung and Ma 2009; Lloyd and Ries 2007; Warmink et al. 2010). However, very little has been published on decision-making under uncertainty in LCA studies, that is, on how to assess these alternatives in the face of uncertainties.

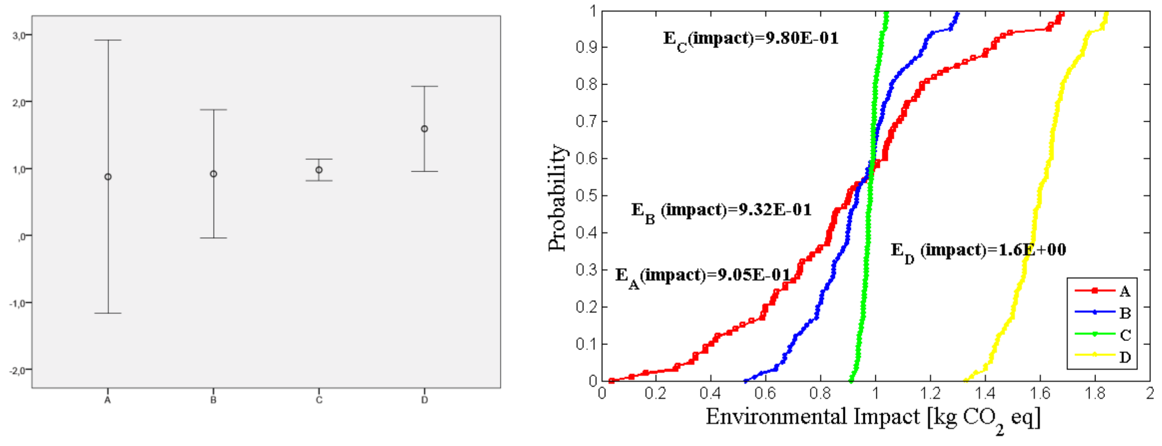
Decision-making under uncertainty is a very broad topic for which a wide variety of strategies and methods have been proposed (the reader is referred to other works on this topic by Ascough et al. (2008), Cheng et al. (2003), Finkel (1990), Geisler et al. (2005), Ierapetritou and Pistikopoulos (1994), Riggs (1968), Walker et al. (2003) for further details. Optimization under uncertainty is one such strategy that seeks optimal decisions using mathematical models with uncertain parameters (see Sahinidis, 2004). One key issue here is the selection of an appropriate objective to be optimized. The overwhelming majority of approaches attempt to optimize the expected value of the objective distribution. This approach guarantees the best performance on average, but provides no control on the variability of the objective function distribution. In other words, we can identify solutions that imply large risk levels, that is, large probabilities of unfavourable outcomes despite behaving well on average. A possible manner to overcome this limitation is to employ risk metrics, which are calculated from the outcomes of the model in the space of uncertain parameters. Risk metrics have been used in a wide variety of fields, like safety analysis, operations, and finances (Aseeri and Bagajewicz 2004; Barbaro and Bagajewicz 2004; Guldimann 2000).

In this work, we focus on decision-making under uncertainty in the context of LCA. Hence, given the results of an LCA study under uncertainty, we are herein interested in developing tools to assists practitioners in the interpretation phase. Particularly, we present here a comprehensive approach for decision-making under uncertainty in LCA that is based on the use of probabilistic risk metrics widely employed in financial management. Furthermore, we will focus on statistical uncertainties, that is, on uncertainties that can be described using probabilistic functions.

The article is organized as follows. We introduce first a simple example to motivate our approach. Our methodology for decision-making under uncertainty follows. A case study based on energy generation is then introduced and discussed.

9.1.1. Motivating example

To motivate our approach, let us consider the results of an LCA study where 4 alternatives are assessed in the face of uncertainties in the LCI data (i.e., the LCI data are imprecise and cannot be fully known with accuracy). Our goal is to make sure that the final technology to be implemented in practice will not surpass an environmental limit Ω_k of 0.8 kg CO_2 equivalent points of impact. We assume that the entries of the LCI are uncertain and can be described using probabilistic functions. This allows for the application of a stochastic sampling method on the LCI data, which provides a set of representative scenarios of the uncertain parameters space. For each of these scenarios, we can obtain the LCIA results by multiplying the emissions by the associated damage factors. Figure 9.1 (a) shows the maximum and minimum impacts along with the expected impact of each alternative, while Figure 9.1 (b) displays the cumulative probability curves associated with each solution. These curves are obtained in a straightforward manner from the scenarios of the LCA calculations. Given this probabilistic information, we aim to identify the best alternative (i.e., technology for energy generation) considering the aforementioned uncertainties.



(a) (b)

Figure 9.1 LCA alternatives assessed in the face of uncertainties in LCI data: (a) environmental impact (kg $CO_{2,eq}$) by alternative A, B, C and D and (b) by probability.

A standard approach to rank alternatives under uncertainty is to select the one showing the best expected performance, which in our case corresponds to the technology with minimum expected impact. While this strategy guarantees the best performance on average, it provides no control on the probabilities of undesirable outcomes. As an example, the minimum impact solution (i.e., solution A) has a 37% probability of surpassing the target value (Ω_k) of 0.8 kg CO_2 equivalent, whereas in solution B, this probability is reduced to 23.0%. This trade-off can also be observed in Figure 9.1(a,b), where it is clear that alternative A leads to the lowest expected impact but at the expense of the largest variability.

Thus, a trade-off tends to exist in practice between expected performance and risk level. In this context, selecting a final solution is far from being a trivial task. When deciding on the best alternative to select, we are in fact facing a multi-criteria decision-making problem, in which the expected performance and the risk level must be balanced to reach a final solution. In the section that follows, we describe a systematic methodology to support decision-making in this context.

9.2 Methods

We propose herein a methodology for assessing alternatives in LCA studies under uncertainty. We assume that the uncertain parameters can be described by means of probabilistic functions, which makes it possible to generate scenarios using sampling techniques.

Our methodology, which makes use of concepts and tools widely used in risk management, is based on a detailed analysis of the probabilistic information obtained from the LCA study under uncertainty. This analysis makes use of several risk metrics that are calculated from the cumulative probability curves of the LCI and LCIA results. These metrics are employed to quantify the potential of undesirable outcomes.

Let us consider a set *i* of alternatives taken as a basis in an LCA study. For simplicity, we will from here on focus on controlling the variability of the impact. Note, however, that our approach could be easily extended to deal with the variability of the LCI results as well. The environmental impact associated with each alternative can be determined from the LCI results and damage factors as follows (9.1):

$$LCIA_{i,k} = \sum_{j} LCI_{i,j} df_{j,k} \qquad \forall i,k \qquad \text{Equation (9.1)}$$

Where $LCIA_{i,k}$ is the impact of alternative *i* in damage category *k*, $LCI_{i,j}$ is the life cycle inventory entry associated with input/output *j* and alternative *i*, and $df_{j,k}$ is the damage factor that translates the LCI into the corresponding impact. In what follows, we assume that both, *LCI* and *df* are stochastic variables that follow known probability functions. Hence, we have:

$$\tilde{L}CIA_{i,k} = \sum_{j} \tilde{L}CI_{i,j} \tilde{d}f_{j,k} \qquad \forall i,k$$

Where variables $\tilde{L}CIA, \tilde{L}CI, \ \tilde{d}f$ denote the stochastic counterparts of the deterministic variables. We assume that these stochastic variables can be described through probability functions (either continuous or discrete).

Remark: The expected value of the impact distribution is the same as the impact in the mean scenario. In other words, there is no need to perform any stochastic analysis if the alternatives are ranked according to the expected performance, since this probabilistic metric can be obtained from the mean values of the uncertain parameters.

9.2.1 Proposed methodology

We describe next our methodology for evaluating LCA alternatives under uncertainty. Our approach comprises three steps (Figure 9.2). In step one, which is widely used in any standard probabilistic analysis applied in LCA, we identify and characterize the main sources of uncertainty using probability functions. We then apply sampling techniques to generate scenarios from these distributions. Note that the LCI values may differ from one scenario to another. The LCA calculations must be therefore repeated for each of them, obtaining values of environmental impact in each of these scenarios that are then used to construct probability curves of the LCIA results. In step two, we filter the curves based on the dominance concept in order to discard suboptimal alternatives. An option A is considered as suboptimal if it is dominated by at least another alternative B. Furthermore, A is said to be dominated by another solution B if the probability curve of A lies entirely above that of B. Note that if this property holds, then the environmental impact of solution A will be lower than that of B regardless of the probability level of choice. Hence, dominated solutions can be removed from further inspection, as they show poor performance. In step three, a set of risk metrics are calculated from the probabilistic LCIA results. Some rules of thumb and guidelines are provided at this point to facilitate decision-making. The steps of our approach are shown in Figure 9.2 and described in detail next.

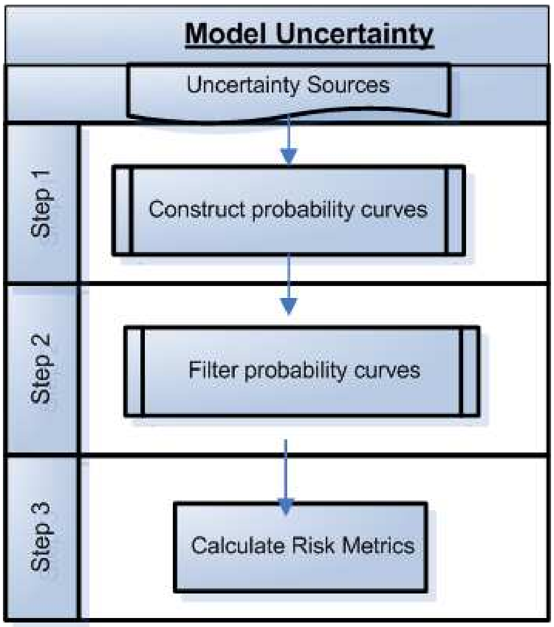


Figure 9.2. Steps of our Model uncertainty.

Step 1: Construction of the probability curves of the impact

In this phase, we identify the main sources of uncertainty affecting the calculations. We consider herein uncertainties that can be described through probability functions and that propagate in the LCA calculations, ultimately affecting the LCIA results. We place particular interest in the entries of the LCI parameters, which represent (arguably) the main source of impact. Nevertheless, the approach presented here is general enough to handle any type of uncertainty source as long as it can be described via probabilistic functions. These probabilistic models should be ideally obtained from the data available. When historical data are lacking, we may construct these functions using the Pedigree matrix (Weidema & Wesnaes 1996), which takes into account aspects related to the reliability, completeness, and temporal, geographical, and technological aspects of the data. Further details on this approach are provided in the Appendix A9.

After describing the uncertain parameters, we apply a sampling method (e.g., Monte Carlo simulation, Latin Hypercube, etc.) that generates scenarios from the distributions of the uncertain parameters. Each scenario is defined by a unique set of values of the uncertain parameters. By performing the LCA calculations for every scenario, we can obtain the uncertainty distribution of the output variable (Huibrejts 2001; Huibrejts 2003). This output variable typically corresponds to a given environmental impact. Note that the minimum number of scenarios required to ensure that the uncertain parameter space is well characterized can be determined using statistical methods like the one developed by Law and Kelton (2000). Hence, the sampling method provides as output a set of scenarios with known probability of occurrence, each one characterized by a different impact. The probability of not exceeding an environmental limit omega is therefore expressed as follows (9.2):

$$\Pr\left(\tilde{L}CIA_{i,k} \leq \Omega_k\right) = \sum_s \Pr_s z_s \quad \forall i,k$$

$$z_s \begin{cases} 1, \text{if } LCIA_{i,k,s} \leq \Omega_k \\ 0, \text{otherwise} \end{cases} \qquad \text{Equation (9.2)}$$

Where $LCIA_{i,k,s}$ is the value of impact $\underline{i}$ in scenario *s*, Pr_s is the probability of scenario *s*, and z_s is a binary variable that takes the value of 1 if the impact in that scenario is below the target and zero otherwise. That is, the probability of exceeding a given environmental target is obtained from the number of scenarios in which such a limit is surpassed and the probability associated with each of them.

Step 2: Filtering of the stochastic curves

The probability curves obtained in step 1 are next filtered prior to the calculation of the risk metrics. To this end, we apply concepts borrowed from multi-objective optimization. The ultimate goal of our analysis is to identify solutions with good performance for a wide range of probability levels while discarding others that perform poorly. We pose this filtering step in mathematical terms as a multi-criteria problem, where the impacts of each alternative associated with every probability level (9.3) are the objectives sought. That is, we are interested in minimizing simultaneously the probabilities of exceeding a set of environmental target values. Hence, the multi-criteria objective function to be optimized takes the following form:

$$\min\left\{\Pr\left(\tilde{L}CIA_{i,1} \leq \Omega_1\right), \ldots, \Pr\left(\tilde{L}CIA_{i,K} \leq \Omega_K\right)\right\} \qquad \text{Equation (9.3)}$$

That is, we deal with a multi-dimensional objective function that accounts for the probabilities of exceeding a set of environmental targets Ω_k simultaneously. Solution A is said to be weakly Pareto efficient if there is no other solution B that is better than A simultaneously in all of the objectives. Based on this dominance concept, it is possible to discard those alternatives that are dominated by any of the others, that is, those with larger probability levels for the same environmental targets (or equivalently, with larger impacts for the same probability levels). To illustrate this approach, let us consider a case with 4 curves: A, B, C and D depicted in Figure 9.1(a,b). As observed, alternative D shows worst probability levels for any possible environmental target than A, B and C, as its cumulative curve lies entirely on the right hand side of those corresponding to the other alternatives. Hence, solution D should be removed. Note that ruling out alternative D is not clear at all when we only look at the maximum and minimum impact values in Figure 9.1(b). In contrast, solutions A, B and C intersect in one point with each other. This implies that there are probability levels for which one alternative performs better than the others. We describe in the section that follows a systematic method for further assessing the performance of these non-dominated solutions.

We should note that the filtering step depends on the number of target values. In practice, we may choose a large enough set of targets so this phase does not heavily depend on the number of them.

Step 3: Calculation of risk metrics

The pivotal idea of our approach is to apply techniques borrowed from financial risk management to analyze the probability curves associated with each alternative (see Aseeri and Bagajewicz, 2004). Particularly, we explore the use of the following risk metrics: Value at Risk (VaR), Upside Potential (UP) or Opportunity Value (OV), Risk Area Ratio (RAR), and Worst Case (WC), as a means to support risk-related decisions and properly manage the associated risk (Figure 9.3).

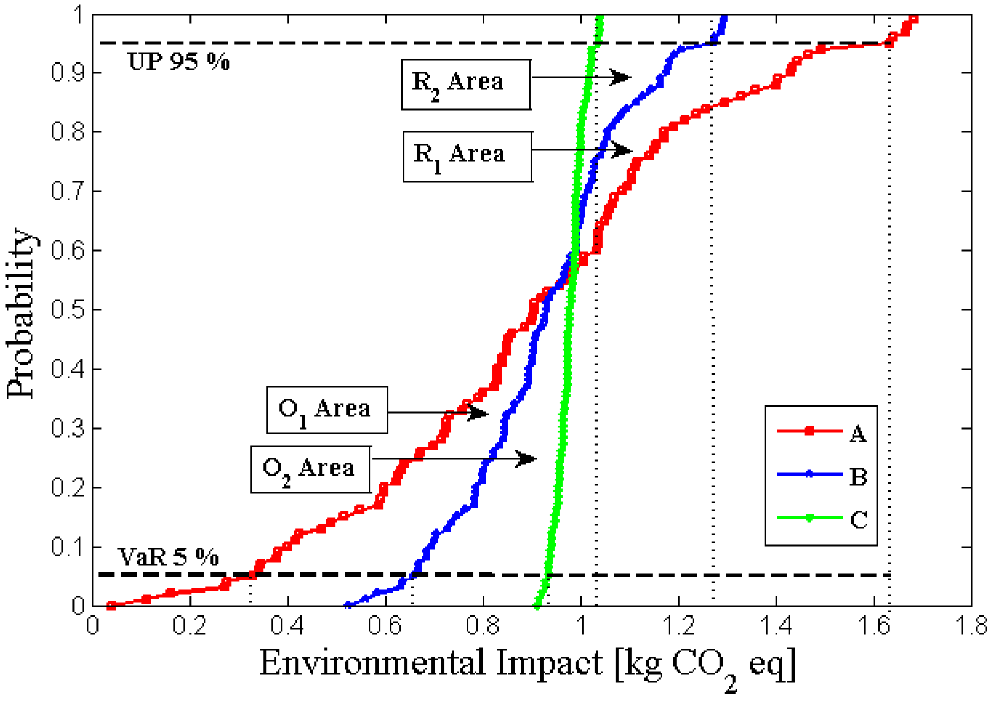


Figure 9.3 Cumulative probability curves.

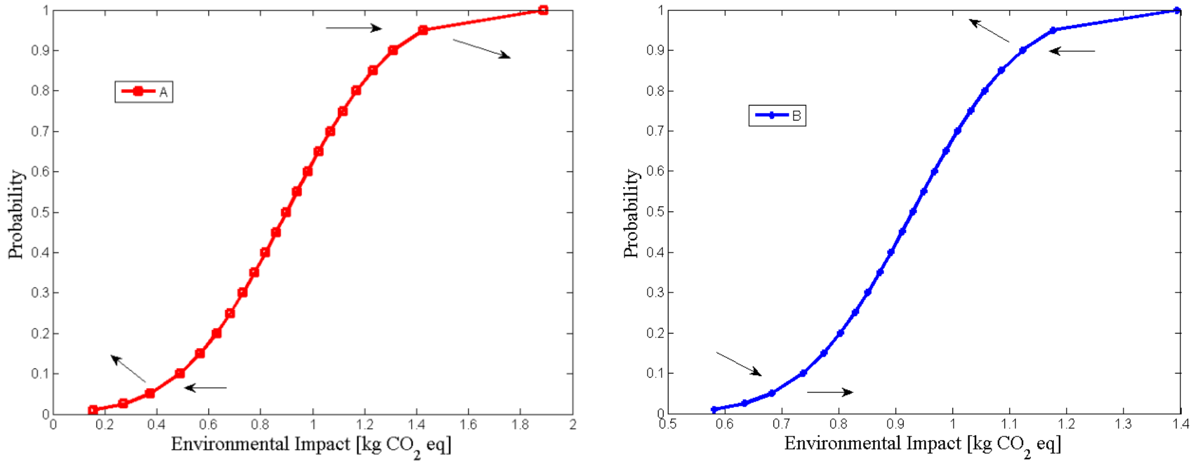
Note that these metrics are adequately modified in order to be adapted to the particular features of an LCA study. Particularly, the metrics are typically used for the case when we aim to maximize either the profit or the Net Present Value (NPV). In LCA studies, we seek to minimize the environmental impact. Hence, as we will discuss next, the interpretation of the metrics is done in an opposite manner as in the standard case. The risk metrics are described in detail next.

The Value at Risk (VaR) is defined as the expected loss for a certain confidence level, usually set at 5% probability (Linsmeier & Pearson 2000). The Upside Potential (UP) or Opportunity Value (OV) is a measure of risk-adjusted returns defined in a similar way to VaR, but at the other end of the risk curve for a quantile of (1-p), typically set at 95% probability. Hence, these metrics are obtained by calculating the difference between the mean value of the distribution and the corresponding probability level (the percentile covered by each metric). Figure 9.3 illustrates how these two metrics are determined. VaR and UP are single metrics that do not represent the behavior of the entire curve, yet inform about how the alternative performs for large and small impacts. The RAR is calculated as shown in Figure 9.3, where the key idea is to compare the areas between the three curves. The proposed ratio takes as reference a given curve, and it is obtained from the ratio between the opportunity area (O_Area), which is enclosed by the two curves above their intersection, to the risk area (R_ Area), which is enclosed by the two curves below their intersection. Note that even though solution A shows the minimum expected impact, it might not be selected if the decision-maker is risk-averse, as he/she might prefer to have a more robust solution with lower probabilities of large impacts even if the expected impact of such solution is (reasonably) larger than that of the others. Finally, the worst case is the impact in the worst scenario possible.

We now discuss how these metrics can support decision-making. In general, we will be interested in solutions with large VaR values and low UPs. This is because we want to maximize the probability levels of low impacts and minimize at the same time the probability of large impacts. Solution A is the best in terms of VaR but the worst according to UP, while solution C behaves in an opposite manner (i.e., it shows the best UP and worst VaR). If the UP is used as the only means of evaluating the solutions, then solution C would be selected since it shows lower UP and therefore lower probabilities of large environmental impacts. However, looking at the low part of the cumulative probability curve, one can see how this solution shows the best performance. In terms of worst case, solution D shows the worst performance, followed by solutions A, B and C, respectively. Finally, the best alternative in terms of expected impact is solution A, followed by B, C and D. The question then is how a decision-maker should assess the trade-off between risk and expected impact by just looking at these measures.

It is clear that VaR and UP are unable to represent the behaviour of the entire probability curve, as they cover only its extreme sides. To get further insight into the problem, we propose to compute the Risk Area Ratio (RAR), an alternative risk metric originally introduced by Aseeri and Bagajewicz (2004) and Barbaro and Bagajewicz (2004) that compares the areas between any two cumulative curves (Figure 9.3). In our case, the RAR is defined as the ratio between the areas enclosed between two curves below and above their intersection.

When comparing two cumulative probability curves, two possible cases may arise (see Figure 9.4). On the one hand, Figure 9.4 (a) shows a curve A that rotates clock wise with respect the reference curve, while Figure 9.4 (b) depicts a curve B rotating counter clock wise. In the first case, the probabilities of low impacts are increased at the expense of increasing as well the probabilities of large impacts. In the second case, the probabilities of large impacts are reduced at the expense of decreasing the probabilities of small impacts. Note that ideally we would like to improve the upper and lower parts of the curve simultaneously. Unfortunately, this might be impossible in practice, because the common situation is that alternatives with low probabilities of low impact show large probabilities of high impacts and the other way round. A curve is “well balanced” if what it is gained in the upper part is compensated by what it is lost in the lower part and vice-versa. Hence, the area ratio provides information on to what extent an alternative is “well balanced” compared to another one (usually, but not always, the one with the best expected performance). The RAR metric is therefore defined as the ratio between the area “gained” divided by the area "lost". When there is a rotation of the curve to the left with respect the reference alternative, then the RAR is defined as the ratio between the areas above and bellow the intersection. In contrast, when the curve rotates to the right, then the RAR is defined as the ratio between the areas bellow and above the intersecting point. A key point here concerns the curve taken as reference to calculate the RAR metrics. As a general curve, the minimum expected impact curve will be taken as reference in the calculations. Note that we will look for alternatives with large RAR values.



(a)

(b)

Figure 9.4 Calculation of RAR values: a) Right rotation b) Left rotation.

In the example given before, the area ratio between B-A (RAR_{BA} =0.74) and C-A (RAR_{CA}=0.60) reveals that solution B is more appealing than C. Hence, taking into account all the risk metrics simultaneously, VaR, UP, RAR, worst case and expected value, it seems that the best balanced alternative is solution B.

9.3 Results and discussion

9.3.1 Case study: Motivating example revisited

We illustrate the capabilities of the approach presented using a proof-of-concept example where we assess several technologies for energy production in terms of contribution to global warming. Particularly, we study five cogeneration technologies for energy production (1 MJ). The environmental data (i.e., GHG emissions) associated with each of them are retrieved from Ecoinvent v2.2 (Ecoincent, 2010). We assume that these life cycle emissions associated with the given functional unit (1 MJ of energy generated) are uncertain.

To obtain the impact distribution, we proceed as follows. We first obtain the probability functions of the single emissions using the Pedigree matrix. We consider only the main GHG emissions: carbonate dioxide fossil (CO_2 kg equivalent), methane fossil (kg CH_4) and nitrogen oxide (kg N_2O), and neglect the others. We assume that these emissions can be modelled using stochastic variables that follow lognormal distributions. A lognormal distribution is characterized by two parameters; "ξ" and "Ø" (Heijungs and Frischknecht 2005). In our case, the values of these parameters for the main GHG emissions, which are shown in Table 9.1, are obtained using equation (9.4) and equation (9.5), respectively:

$$\xi = \ln\left(1 + \frac{Var[X]}{(E[X])^2}\right) \qquad \text{Equation (9.4)}$$

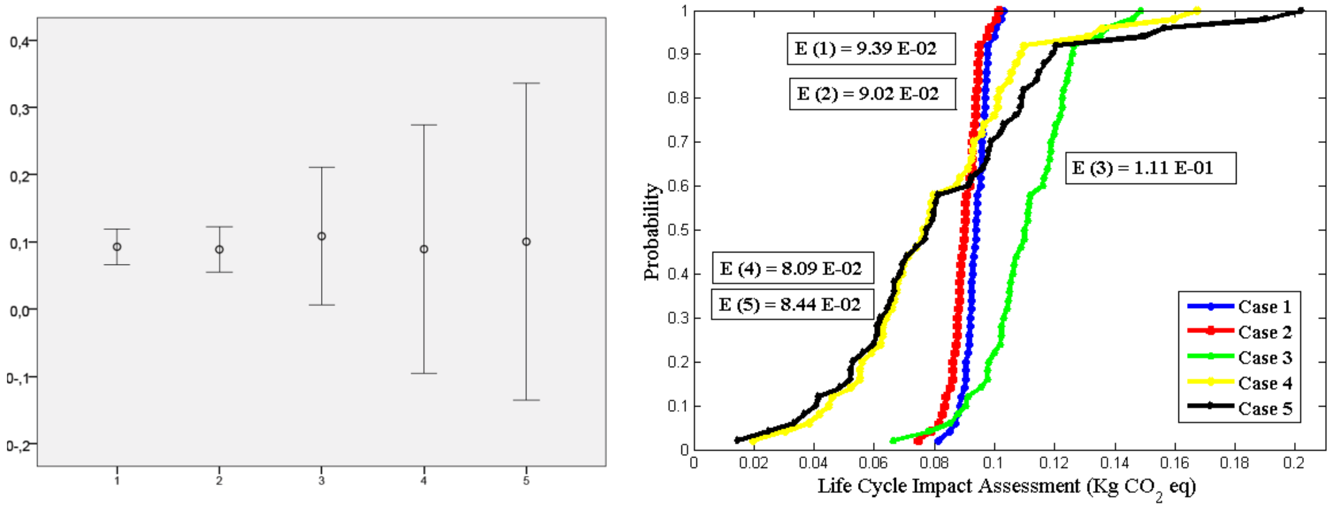
$$\varnothing = \ln\left(E[X] - \frac{1}{2}\xi\right) \qquad \text{Equation (9.5)}$$

Where E[X] is the expected value (i.e., mean value) of the LCI entry, which is retrieved from Ecoinvent v2.2, VaR [X] is its variance, and ξ and Ø are the mean and standard deviation of the natural logarithm of the stochastic variable (which by definition follows a normal distribution). To determine the variance, we make use of the Pedigree Matrix, which builds the lognormal distribution according to the quality of the data available. This methodology relies on formulating a set of questions regarding the reliability, completeness, temporal correlation, geographical correlation, further technological correlation, and sample size of the data employed in the LCA study. We assume hypothetical values of these factors (Table 9.1).

Table 9.1 ξ and $\varnothing$ for each GHG emissions.

Cases	Source from Ecoinvent	CO_2 emissions (ξ /$\varnothing$)	CH_4 emissions (ξ /$\varnothing$)	N_2O emissions (ξ /$\varnothing$)
1	heat, at cogen 160kWe lambda=1, allocation energy, [MJ]	$2.44\text{x}10^{-02}$/ $-2.52\text{x}10^{+00}$	$9.12\ \text{x}10^{-02}$/ $-1.04\ \text{x}10^{+01}$	$1.68\ \text{x}10^{-01}$/ $-1.01\ \text{x}10^{+01}$
2	heat, at system cogen 160kWe Jakobsberg, allocation energy, [MJ]	$4.04\text{x}10^{-02}$/ $-2.55\text{x}10^{+00}$	$9.67\ \text{x}10^{-02}$/ $-1.10\ \text{x}10^{+01}$	$1.71\ \text{x}10^{-01}$/ $-1.02\ \text{x}10^{+01}$
3	heat, at cogen 1MWe lean burn, allocation energy, [MJ]	$1.21\text{x}10^{-01}$/ $-2.59\ \text{x}10^{+00}$	$1.50\ \text{x}10^{-01}$/ $-9.23\ \text{x}10^{+00}$	$2.06\ \text{x}10^{-01}$/ $-9.17\ \text{x}10^{+00}$
4	heat, at local distribution cogen 160kWe Jakobsberg, allocation energy, [MJ]	$3.88\ \text{x}10^{-01}$/ $-2.68\ \text{x}10^{+00}$	$4.79\ \text{x}10^{-01}$/ $-1.06\ \text{x}10^{+01}$	$4.22\ \text{x}10^{-01}$/ $-1.03\ \text{x}10^{+01}$
5	heat, at cogen 160kWe Jakobsberg, allocation energy, [MJ]	$4.71\ \text{x}10^{-01}$/ $-2.76\ \text{x}10^{+00}$	$4.79\ \text{x}10^{-01}$/ $-1.51\ \text{x}10^{+00}$	$5.00\ \text{x}10^{-01}$/ $-1.52\ \text{x}10^{+00}$

We generated 50 scenarios from the lognormal distributions that characterize the uncertain emissions using Monte Carlo sampling. Figure 9.5 (a, b) shows the cumulative probability curves associated with each alternative. As observed, it is difficult to identify the solution with better performance. We illustrate next how to analyze these curves following the strategy introduced before.



(a) (b)

Figure 9.5 Environmental profile of case study: a) shows the interval of impacts, b) displays the associated cumulative probability curves.

We first filter the cumulative curves according to the dominance concept. As observed, all the curves intersect in at least one point, meaning that there is no single solution that performs better considering the entire uncertain space. We calculate next the probabilistic metrics. Particularly, the following metrics are determined: Value at Risk (VaR), Upside Potential (UP), Expected value (EV), Risk Area Ratio (RAR), and Worst Case (WC). To determine the RAR, we take as reference curve the one associated with the minimum expected impact solution (case 4). Table 9.2 shows the values of the risk metrics for each case. This table shows the values of the metrics for each alternative being assessed along with a score ranging from 1 to 5 (being 1 the most suitable).

Table 9.2 Risk metrics criteria.

	VaR	**UP**	**RAR$_{x\ vs\ 4}$**	**WC**	**EV**
criteria	**Min**	**Min**	**Max**	**Min**	**Min**
Case 1	$6.79x10^{-03}$	$6.56x10^{-03}$	0.28	$1.03x10^{-01}$	$9.39x10^{-02}$
Ranking	2	5	3	2	4
Case 2	$8.49x10^{-03}$	$8.14x10^{-03}$	0.38	$1.02x10^{-01}$	$9.02x10^{-02}$
Ranking	**1**	4	2	**1**	3
Case 3	$2.57x10^{-02}$	$2.63x10^{-02}$	0.021	$1.49x10^{-01}$	$1.11x10^{-01}$
Ranking	5	**1**	5	3	5
Case 4	$4.25x10^{-02}$	$5.51x10^{-02}$	-	$1.68x10^{-01}$	$8.09x10^{-02}$
Ranking	4	2	**1**	4	**1**
Case 5	$5.11x10^{-02}$	$7.21x10^{-02}$	0.25	$2.02x10^{-01}$	$8.44x10^{-02}$
Ranking	3	3	4	5	2

The interpretation of the risk metrics is as follows. We are interested in alternatives with large VaR, since this implies larger probabilities of low impact. We would like the alternative to show as well small UP, since this implies low probabilities of large impacts. The expected value should be as small as possible, meaning that the solution performs well in average. Finally, the worst case should be also small, so the most unfavourable scenario is less harmful. Finally, the RAR should be as high as possible. It is very likely that a solution will not fulfil all the requirements simultaneously, so a compromise alternative should be identified. We are particularly interested in solutions that are "well-balance" meaning that they perform well in both extremes of the cumulative curves. Since the problem is multi-objective in nature, there final decision will depend on the preferences of the decision-maker.

Based on the analysis of these risk metrics, we can rule out case 5, since the relatively large probabilities of small impacts do not compensate for the large probabilities of high impacts. This conclusion can be reached by analyzing the values of the risk metrics. As observed, alternative 5 performs well in EV and VaR because it shows large probabilities of small impacts when compared to the rest of alternatives. However, as already mentioned, it leads to large probabilities of high impacts, and for this reason it shows large worst case values and a small RAR ratio, which makes it less appealing. With regard to alternative 3, it can also be easily removed because it is slightly better than 1 and 2 for large impact targets, but much worst in the low impact region of the cumulative curve. This can also be observed by analysing the value of the risk metrics associated with this solution (i.e., large EV, VaR and WC values, and very small RAR ratio).

Alternatives 1 and 2 are rather similar. They show large values of VaR and UP, low WC values, an intermediate expected impact and a small RAR ratio. Solution 2 is much more appealing, since it shows lower impacts than 1 for a wide range of probability levels, which leads to a better RAR ratio (0.38 for solution 2 and 0.28 for solution 1). Finally, solution 4 shows the best RAR value (i.e., one), but has the drawback of having a large worst case. Hence, we would recommend choosing either solution 2 or 4. A risk-averse decision-maker would keep solution 2, while a risk-taker would prefer solution 1. This is because solution 4 is better on average but leads to a higher worst case value and larger probabilities of high impacts. As observed, there is not a single "best" solution, but rather a set of "appealing" alternatives from which the most suitable ones should be identified according to the decision-makers' preferences. Our tool allows discarding solutions with poor performance, thereby facilitating the task of decision-makers and providing at the same time valuable insight into the performance of the alternatives under uncertainty.

9.4 Conclusions

A new methodology has been proposed to assess alternatives for environmental improvements under uncertainty. The framework presented makes use of stochastic modelling and financial risk metrics. We have shown how the analysis of the cumulative curves allows discarding suboptimal solutions that would be hard to identify by analyzing only confidence intervals or expected impact values. Hence, we conclude that the expected impact should be complemented by additional metrics when evaluating alternatives for environmental improvements in LCA studies under uncertainty. This is because the solution with minimum expected impact might lead to large probabilities of high impacts, that is, to a risky attitude of the decision-maker. The use of risk metrics and the analysis of cumulative curves provide a sound basis for decision-making under uncertainty in LCA.

It should be kept in mind that the case study results strongly depend on the LCA data used in the analysis. Our framework focuses on probabilistic uncertainties, but could be extended in order to deal with other types of uncertainties.

We should mention that the final alternative to be implemented in practice depends on the risk metrics employed in the analysis along with the preferences of the decision-maker. Besides the risk metrics themselves, our approach offers a comprehensive framework to assess alternatives in environmental studies under uncertainty that relies on the study of the cumulative curves of the LCA outcome. This is a step forward compared to other methodologies that focus on confidence intervals and expected values. The use of cumulative curves opens also the door for new methodologies that will rely on other probabilistic metrics based on their shape. In this regard, we should clarify that our approach is not restricted to the use of risk metrics mentioned above, and can be easily extended in order to accommodate other probabilistic metrics.

Appendix A9

The Pedrigree matrix approach (Weidema & Wesnaes, 1996; Weidema, 1998) allows translating quality indicators into quantitative information. This methodology relies on the assumption that the uncertain parameters can be described using lognormal distributions (Weidema et al. 2012). Data sources are assessed according to five items: "reliability", "completeness", "temporal correlation", "geographical correlation" and "further technological correlation" (Table A9.1).

Table A9.1 Indicator's definitions of Pedigree Matrix.

Indicators	Definition
Reliability	Relates to the sources, acquisition methods and verification procedures used to obtain the data.
Completeness	Relates to the statistical properties of the data: how representative is the sample, does the sample include a sufficient number of data and is the period adequate to even out normal fluctuations.
Temporal correlation	Represents the time correlation between the year of study (as stated in the data quality goals) and the year of the obtained data.
Geographical correlation	Illustrates the geographical correlation between the defined area (as stated in data quality goals) and the obtained data.
Further technological correlation	Concerned with all other aspects of correlation than the temporal and geographical considerations. Although data may be of the desired age and representative of the desired geographical area, it may not be representative of the specific enterprises, processes or materials under study.

Each characteristic of the data is divided into five quality levels with a score between 1 and 5. Accordingly, a set of five indicator scores is attributed to each individual input and output exchange (except the reference products), reported in a data source. Table A9.2 is known as the Pedigree matrix since the data quality indicators refer to the history or origin of the data, like a genealogical table reports the pedigree of an individual (Funtowicz & Ravetz 1990).

Table A9.2 Pedigree matrix with 5 quality indicators for environmental data used to assess the quality of the data sources (Weidema and Wesneas, 1996; Weidema, 1998).

Indicator Score	**1**	**2**	**3**	**4**	**5 (default)**
Reliability	Verified[1] data based on measurements[2]	Verified data partly based on assumptions ***or*** non-verified data based on measurements	Non-verified data partly based on qualified estimates	Qualified estimate (e.g. by industrial ex-pert)	Non-qualified estimate
Completeness	Representative data from all sites relevant for the market considered, over an adequate period to even out normal fluctuations	Representative data from >50% of the sites relevant for the market considered, over an adequate period to even out normal fluctuations	Representative data from only some sites (<<50%) relevant for the market considered ***or*** >50% of sites but from shorter periods	Representative data from only one site relevant for the market considered ***or*** some sites but from shorter periods	Representativeness unknown or data from a small number of sites ***and*** from shorter periods
Temporal correlation	Less than 3 years of difference to the time period of the dataset	Less than 6 years of difference to the time period of the dataset	Less than 10 years of difference to the time period of the dataset	Less than 15 years of difference to the time period of the dataset	Age of data unknown or more than 15 years of difference to the time period of the dataset
Geographical correlation	Data from area under study	Average data from larger area in which the area under study is included	Data from area with similar production conditions	Data from area with slightly similar production conditions	Data from unknown ***or*** distinctly different area (North America in-stead of Middle East, OECD-Europe instead of Russia)

Further technological correlation	Data from enterprises, processes and materials under study	Data from processes and materials under study (i.e. identical technology) but from different enterprises	Data from processes and materials under study but from different technology	Data on related processes or materials	Data on related processes on laboratory scale ***or*** from different technology

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[1] Verification may take place in several ways, e.g. by on-site checking, by recalculation, through mass balances or cross-checks with other sources.

[2] Includes calculated data (e.g. emissions calculated from inputs to an activity), when the basis for calculation is measurements (e.g. measured inputs). If the calculation is based partly on assumptions, the score would be 2 or 3.

According to the score, we assign a different standard deviation to the (Weidema et al. 2012). A normal uncertainty distribution is attributed to each score of the five characteristics and each one has a mean value of zero, and a variance based on Table A9.3.

Table A9.3 Uncertainty factors (variances of the underlying normal distributions) used to convert the data quality indicators of the pedigree matrix in Table A9.2 into additional uncertainty.

Indicator score	1	2	3	4	5
Reliability	0.000	0.0006	0.002	0.008	0.04
Completeness	0.000	0.0001	0.0006	0.002	0.008
Temporal correlation	0.000	0.0002	0.002	0.008	0.04
Geographical correlation	0.000	2.5e-5	0.0001	0.0006	0.002
Further technological correlation	0.000	0.0006	0.008	0.04	0.12

Part VI

Conclusions and Outlook

Chapter 10

Conclusions

The main aim of this doctoral thesis was to assess the environmental performance of different agricultural systems through the application of LCA and other complementary tools. Two main environmental impacts, water and climate change, are considered for the analysis. This chapter is a synopsis of the overall conclusions presented in sections II, III, IV and V of this dissertation. As a summary the next conclusions can be drawn:

10.1. Overall conclusions

A practical methodology is developed to evaluate agricultural water consumption and climate change contribution. The assessment of water use in agriculture has gone beyond the classical treatment given by Hoekstra et al. (2009, 2011) in order to include regionalized damage to ecosystems. Special focus has been given to applicability of assessment methods in decision making processes. In addition, several methodological aspects have been developed in order to improve environmental assessment.

A robust and effective tool in agricultural sector, eFoodPrint® has been developed to assess water and carbon footprint of crops. A special added value is given to farmers as a practical indicator to know how efficient is their water and resource management in all the life cycle of the agricultural product.

This new perspective of the environmental profile of crops, contributes to define few indicators or parameters. These indicators are not only easy to calculate but also easy to understand for the open public that can orientate the farmer and consumer for a more sustainable production and consumption, respectively. This aspect is particularly relevant in a time when the so called ecological or green market is full of false or ambiguous concepts and messages.

Furthermore, this dissertation is mainly focused to solve methodological weaknesses of biodiversity assessment, that are linked to increase in salinity because of higher water consumption, and also to know the applicability of methods in the decision making process.

A holistic conception of Urban Water Cycle has allowed proposing different scenarios of water supply as well as providing a reclaimed water scenario in order to avoid water depletion.

Finally, a first step in the development of uncertainty of different methodologies is introduced in this dissertation.

10.2. Specific Conclusions

Part II. Problem statement: water management and climate change in agriculture

Comparison of methods for calculation of Water Stress Index (Chapter 3)

Three different methods of characterizing the Water Stress Index are compared (Milà i Canals et al. 2009; Pfister et al. 2009; Smakhtin et al. 2004) with three crops (grapes, nectarines and corn) in the Ebro Basin. None of them has complete yield and precise information. Although all these methods are complementary, a method proposed by Pfister et al. (2009) provided the most accurate results, as it considers seasonality and water reservoir. However, a very interesting issue that remains relatively unexplored is how the native species in different ecosystems are affected by water use.

Significant differences between experimental and statistical data (Chapter 3)

There were considerable differences between experimental and statistical data, emphasizing the high degree of variability in results depending on the information source, crop variety, and cultivation technique. Taking into account this enormous divergence, several efforts need to encourage using real data in life-cycle assessments in order to minimize the final impact results.

Including regionalized inventory and impact assessment for water resources is crucial for crops and processes based on agricultural products (Chapter 3)

In the context of freshwater analysis, it becomes essential to provide specific guidance for future improvements of inventory data sets. Results should be based on regionalized inventory and impact assessment with spatial distinction (e.g. watershed level). Additionally, high spatial resolution can significantly decrease uncertainty related to spatial variability and it is also relevant for comparing different crops, as the location has often a higher influence on the environmental impacts than alternative crops grown under the same climate.

The discharge of more veracious maps will empower a better depiction of local-dependent environmental impacts in LCA (Chapter 3)

Using GIS tool has facilitated the gathering of location-specific LCI data to complete the inventory table for the different systems.

Agriculture is the highest impact subsystem in different pathways to produce bioethanol (Chapter 4)

The influence of the cane processing system and agricultural yield variability suggests that studies should be specific, focusing on the regional scope. Total impacts are strongly influenced by the agricultural yields and the applied allocation (e.g. economic, mass). Financial allocation becomes dependent on the relative market price of sugar/ethanol. For that, when co-products appear, economic allocation is less useful due to uncertainty in the mechanism for assigning impacts, whereas mass allocation is preferable. In this respect, measures to mitigate CO_2 emissions includes substituting fossil for biomass-based fuels in agriculture labour, using wastewater treatment at distilleries and water recycling systems during the industrial processing and using cane trash for energy instead of open burning in the fields.

Advancement in the standardization of protocols to calculate impacts based on LCA of products (Chapter 3, 4)

The international market of products considered as commodities, requires common frames to evaluate the sustainability of products addressed to final consumer.

Providing an uncertainty analysis should be mandatory to judge the significance of LCA results (Chapter 3, 4)

Uncertainty is not very often considered in LCA studies, especially in case of agricultural products's LCA which hampers the reader's ability to interpret the results. Most of the results in conventional LCA studies are presented by the expected values by underestimating uncertainty data. When confidence intervals of results are to be included as a measure of uncertainty, two alternatives are possible, use local values with high degree of accuracy or use average regional values with less precision. In the first case we obtain accurate results valid for a small local area and in the second we use average values with less accuracy but the results are valid for all the regions considered for the study.

Part III. Decision making tool in agriculture

Environmental indices focus on water and carbon footprint to make strategically decisions (Chapter 5)

We have provided environmental indices to integrate the managers' dashboard decisions. Water footprint should not be considered solely for decision making, but combined with other footprints such as carbon for covering an essential part of the environmental impacts. Moreover, if a more comprehensive assessment of environmental impact is required, a full LCA should be considered covering further impact categories. Additionally, economic and social appraisals have to be considered for a comprehensive sustainability assessment.

Development of a farm-gate environmental decision tool "eFoodPrint®" (Chapter 6)

eFoodPrint® has been proposed to help agronomical stakeholders in their water management and agricultural practices as a decision-making tool. This practical tool is based on actual field data collected from farmers that calculates a set of on-farm environmental indices in irrigated agriculture. It gives a product profile with information on efficiency in agronomical performance, and water and carbon footprints. As a novelty, this also tool provides a report within graphical and numerical results and some recommendations which can be exported.

Uncertainties associated to eFoodPrint® tool (Chapter 6)

The greatest level of uncertainty in relation to global warming potential is being widely acknowledged as on-field emissions. Particularly factors like N_2O emissions, emission factors, variability of environmental conditions and the resultant contribution to climate change. Future versions of eFoodPrint® should include an uncertainty analysis in its results.

Part IV. Scientific development and application of water use in agriculture

Further completion of existing LCIA methodologies has been developed to assess the salinity impact in wetlands (Chapter 7)

This dissertation introduces the first Characterization Factor (Cf) for salinity impacts in a coastal wetland, defined as the change in the Potentially Affected Fraction (PAF) of species due to a change in salinity which is caused by the changed amounts of groundwater used for crop irrigation.

Considered assumptions and simplifications could vary the final results (Chapter 7)

This new methodology contains its own assumptions and simplifications which may make difficult the comparison among different impact categories. To convert impacts to the unit of species yr, we have considered the species density factors for freshwater system provided by ReCiPe (Goedkoop et al. 2008). That is an order of magnitudes lower than our actual species density in the wetland of the salinity case study. For that, if this calculated density would have been applied for converting the impacts from PAF to species, the impact from salinity would have been dominant. We need to be aware that species diversities depend on the type of ecosystems where applying one global value could be risky.

Accurate data and precise information would be needed in order to decrease uncertainties (Chapter 7)

Although uncertainty factors of the model for regionalized salinity impact assessment are explicitly provided, data quality and coverage would have been improved by accurate information of wetlands: exact geographical coordinates, which water bodies are feeding them, hydrogeological parameters (e.g. thickness, depth, transition zone, hydraulic conductivity) and the biodiversity' sensitivity (e.g. species dead by salinity change).

Reclaimed water obtained from the urban water cycle provides a water supplier for irrigation in agriculture (Chapter 8)

A holistic conception of the urban water cycle by the inclusion of the case study based on real operation data has been displayed in order to propose a solution of saving water resources for agriculture use. Three scenarios has been developed taking into account a current case study including a water supply system of a city, considering the cycle from water abstraction to wastewater treatment and two additional scenarios of reclaiming water and desalination for drought situations. Reclaimed water scenario (scenario II) shows slightly higher impact in Global Warming Potential category than current urban water cycle (scenario I) due to the energy consumption in tertiary treatment. Nevertheless, this scenario II represents approximately an 83% reduction of freshwater ecosystems impact compared to scenario I due to water reutilization for non-potable uses (e.g. irrigation in agriculture). Furthermore, the incorporation of desalination plant (scenario III) as additional source of potable water has beneficial values due to saving of freshwater consumption but also negative effects with a high value in Global Warming Potential due to the increase in energy consumption.

Part V. Uncertainty analysis

New framework for decision-making under probabilistic uncertainties in LCA (Chapter 9)

This new approach offers a comprehensive framework by considering stochastic modelling and financial risk metrics to assess alternatives in environmental studies under uncertainty that relies on the study of the cumulative curves of the LCA outcome. When evaluating alternatives for environmental improvements in LCA studies under uncertainty, the solution with the minimum expected impact is not always the most suitable option for the decision maker since it might lead to large probabilities of high impacts. For that, the use of other complementary metrics such as risk metrics and the analysis of cumulative curves help during decision-making of uncertainty in LCA.

Chapter 11

Outlook

Deja que el instinto guie tus pasos…y que sean tus pies los que te marquen el camino… ellos te llevarán donde más quieres…

From the outcomes and conclusions of this dissertation outlined in Chapter 10, several possibilities for further improvement are identified.

Part II. Problem statement: water management and climate change in agriculture

- As observed in Chapter 3 every crop adapts differently in each zone in accordance with the climatic parameters (solar radiation, temperature, evapotranspiration, precipitation…). Efforts are needed by using optimization techniques to propose a model to identify where is the most suitable watershed to grow a crop taking into account different availability parameters and impact damage (Human Health, Ecosystem Quality and Resources Depletion).
- More research is required to build a regionalized Life Cycle Inventory framework to introduce green and grey water in LCA. Regarding Green water, there are many locally-related parameters (climate, soil, soil-water availability, land slope) as well as variables depending on crop (type of crop, crop variety, length of growing periods) that strongly influence on water consumption, and there are still many essential open questions to be solved for a consistent integration of green water in the LCA methodology. Although Grey water (emissions to water) are partially covered in LCA by impact categories such as eutrophication and ecotoxicity (emissions), there is still a lack of knowledge assessing this type of water in LCA.
- Further development is also required in site-dependent characterization factors for regional impact categories to obtain more accurate impact from processes and to reduce models uncertainties. It is recommended a standardized approach to address uncertainty linked to spatial differentiation over the diverse impact categories in a consistent way. In addition, whenever possible the spatial resolution should exceed country

level towards meaningful hydrological units such as watersheds but if data availability allows, this level of detail might be further accurate to sub-basins and combined with local impact assessment.

Part III. Decision making tool in agriculture

- Concerning application of LCA in agriculture, more work has to be done in the choice of the time horizon that should be harmonized, because some variables like carbon sequestration, soil maturity, avoided environmental load in organic fertilizing, depends on the time frame. Additionally, to compare results, it is necessary to have homogeneity in the standards applied in agricultural studies (e.g. functional unit, system boundaries, emissions models and allocation). Also, more accurate databases are required to build sets of reliable life cycle inventory data and to provide consistent characterization factors.

Part IV. Scientific development and application of water use in agriculture

- Saltwater intrusion is a present problem in different coastal regions around the world and it can be a threat to coastal wetlands. Further develop on the methodology for impacts from salinity becomes essential since as we have seen in the salinity case study in Spain (Chapter 7), impacts from increasing salt contents can be of relevance for coastal wetlands. For that, the approach developed in the case study should be broadened to a global level, to be able to assess salinity impacts in coastal regions around the world. To extrapolate the proposed characterization factor, fate and effect model need to be further adapted. Fate model has been developed for this specific area, so a general fate factor needs to be constructed taking into account different hydro geological characteristics of several coastal wetlands, and additionally, it should model how water consumption leads to an increase in salinity (amount of reduced/increased freshwater inflows or infiltrations take place by freshwater consumption for instance in agriculture). Regarding the effect factor, the procedure of calculating this can be directly taken from the salinity case study in Albufera de Adra (chapter 7), but it should be enhanced and calculated with a spatially explicit manner, since different species are found in different coastal wetland systems around the world of e.g. the Mediterranean, American or the Australian coastline.

- Nowadays, the use of endpoint impact assessment methods has been gradually increasing in LCA. Endpoint indicators contributing to one specific Area of Protection (AoP) and they are expressed with different units (e.g. effects on biodiversity could be founded in literature as NPP·m^2·yr, PDF m^2yr, PAF·m^2·m^{-3}), which hinder the comparison between different impact categories and the grouping at the damage level. Further research on endpoint methods should therefore tend to use common units (e.g. species-year). Additionally, another important point to be addressed to improve the consistency of endpoint methods is related to the procedure to convert from midpoint to endpoint indicators.
- This dissertation provides an example of water reuse from Wastewater Treatment plants (WWT) in an Urban Water Cycle as an alternative scenario to supply water to agriculture in case of water scarcity. Future research based on optimization techniques could be implemented in the current Urban Water Cycle model to obtain a truly integrated system including the use of water saving technology in households, industry and agriculture as well as provide a correct water management. In addition, other characteristics and parameters such as, the needs of the population, the availability of resources and the environmental loads of the different stages of the urban water cycle should take into account.

Part V. Uncertainty analysis

- Although this dissertation provides a framework for decision-making under probabilistic uncertainties in LCA, further research focus on model uncertainties in LCA should be both, qualitatively and quantitatively addressed. The tool presented in this work to assess water and carbon footprint of crops, will be probably used to compare different type of cultures or to assess improvements in the same cultivation area. At this point it will be useful to have a sound information regarding the accuracy of the results to decide which is the better solution among different alternatives.
- In addition, in agricultural LCA studies (e.g. monitorizing the main sources of N_2O and other GHG and developing emission factors for specific conditions), variability is very high, even within the same cultivation area, mainly because the weather conditions are never the same. Moreover, the wide ranges of minimal and maximal estimates for water consumption of crops are largely due to the high variability in management options and orography. The presented methodologies to calculate water and carbon footprint of crops, can be complemented by a systematic uncertainty analysis, based on measured accuracy of inputs and emissions combined with values of inventory data and impact factors with known precision. Conventional Monte-Carlo analysis or other existing techniques must be used to evaluate the uncertainty of the final results

There is still a long way to go in research to address impacts from water consumption and climate change mitigation in agriculture. However, substantial steps have already been implemented in this dissertation and current research is already underway to answer some of the research questions raised in this outlook. One of the main results of this dissertation is the eFoodPrint® calculator which satisfies the local needs of agricultural stakeholders but a new version 2.0 of this tool needs to be launched including an uncertainty framework with confidence intervals. That allows farmers to take safer decisions. Finally, we can foresee that scientific knowledge will always provide innovative solutions to improve environmental concern of society.

References

3GSolar. "LCA Calculator." from http://www.ecn.nl/lca/(S(clmbslifpwjwkv45vqgi2pjo))/Default.aspx

Agardy, T., Alder, J. (2005) Coastal Systems. Millennium Ecosystem Assessment, Ecosystems and Human Well-Being: Current State and Trends Island Press: Washington, DC, 1, 513–549.

Agència de Gestió d 'Ajuts Universitaris i de Recerca (AGAUR). Projecte VALOR00008-Model d'Us Sostenible de l'Aigua a l'Agricultura (MUSA).

Agence d l'Environnement et de la Maîtrise de l'Energie, (2006) Bilan de carbone. http://www.bilancarbonepersonnel.org/

Agriculture, food and rural actions department of Catalan Government (2010) Basic course of irrigation. Technician handbooks. 1st Edition. ISBN 978-84-393-8432-8

Ahmed, HB., Ammar, DB., Zid, E. (2008) Physiology of salt tolerance in Atriplex halimus L. In Biosaline Agriculture and High Salinity Tolerance, Abdelly, C.; Öztürk, M.; Ashraf, M.; Grignon, C., Eds. Birkhäuser Basel: 107-114.

Alcalá, FJ., Solé, A., Creus, C., Domingo, F. (2008) Urban groundwater contribution to regional water bodies and groundwater dependent ecosystems. Case of the seaside town of Adra (SE Spain) (spanish). López-Geta, J. (Ed.), VII Simposio del Agua en Andalucía, Seville (Spain), 699-708.

Alcamo, J., Doll, P., Heinrichs, T., Kaspar, F., Lehner, B., Rosch, T., Siebert, S. (2003) Development and testing of the WaterGAP 2 global model of water use and availability. Hydrol Sci J 48: 317–337.

Alcamo, J., Henrichs, T., Rösch, T. (2000) World water in 2025 – Global modeling and scenario analysis for the World Commission on Water for the 21st Century. Report A0002, Center for Environmental Systems Research, University of Kassel, Kassel, Germany.

Aldenberg, T. (1993) ETX 1.3a, A program to calculate confidence limits for hazardous concentrations based on small samples of toxicity data. RIVM report 719102 015. National Institute of Public Health and Environment, Bilthoven, the Netherlands.

Allen, R., Pereira, L., Raes, D., Smith, M., (2006). Evapotranspiración del cultivo. Guías para la determinación de los requerimientos de agua de los cultivos. FAO Riego y Drenaje, 56. ISSN 0254-5293.

Alliance Keystone. (2012). Field to Market. www.fieldtomarket.org/fieldprint-calculator/

Allison I., Bindoff, NL., Bindschadler, RA., Cox, PM., de Noblet, N., England, MH., Francis, JE, Gruber, N., Haywood, AM., Karoly, DJ., Kaser, G., Le Quéré, C., Lenton, TM., Mann ME., McNeil, B.I., Pitman, AJ., Rahmstorf, S., Rignot, E., Schellnhuber, HJ., Schneider, SH., Sherwood, SC., Somerville, RCJ., Steffen, K., Steig, EJ., Visbeck, M., Weaver, AJ. (2009). The Copenhagen Diagnosis. Updating the World on the Latest Climate Science. The University of New South Wales Climate Change Research Centre (CCRC), Sydney, Australia. 60pp.

Amores, MJ., Verones, F., Raptis, C., Juraske, R., Pfister, S., Stoessel, F., Antón, A., Castells, F., Hellweg, S. (2013) Biodiversity impacts from salinity increase in a coastal wetland. Environ Sci Technol (Submitted in November 2012)

Andalucía, Junta de, (2011) Anuario de la agricultura almeriense. In La Voz de Almería, www.juntadeandalucia.es

Andalucía, Junta de, (2012) Anuario de estadísticas agrarias y pesqueras de Andalucía. www.juntadeandalucia.es

Antón, MA. (2004) Utilización del análisis del ciclo de vida en la evaluación del impacto ambiental del cultivo bajo invernadero mediterráneo. PhD thesis, Universitat Politècnica de Catalunya, Barcelona, Spain.

Antón, MA., Montero, JI., Lorenzo, P., Muñoz, P., Castells, F. (2005) Use of water in LCA: case study involving protected horticulture in the Mediterranean. In: Castells F, Rieradevall J (eds) LCM2005. Innovation by life cycle management. Barcelona, Spain, pp 394– 398

Asal, S., Marcus, R., Hilbert, JA. (2006) Opportunities for and obstacles to sustainable biodiesel production in Argentina. Energy for Sustain. Dev. 10(2): 48-58

Asano, T., Burton, F., Leverenz, H., Tsuchihashi, R., Tchobanoglous, G., (2007). Water Reuse: Issues, Technologies, and Applications. New York: McGraw-Hill Professional.

Ascough, JC., Maier, HR., Ravalico, JK., Strudley, MW. (2008). Future research challenges for incorporation of uncertainty in environmental and ecological decision making. Ecol. Model. 219: 383-389

Assies, JA. (1992), State of art. Life- cycle assessment, 1-20, Workshop Report, Leiden, December 1991.SETAC

Assies, JA. (1997) Risk Indicators for Use in Life-Cycle Impact Assessment: An Approach Based on Sustainability. University of Groningen, Center for Energy and Environmental Studies (IVEM), Groningen, The Netherlands.

Aseeri A, Bagajewicz MJ (2004) New measures and procedures to manage financial risk with applications to the planning of gas commercialization in Asia. Comput. Chem. Eng. 28 2791–2821.

Athena. "EcoCalculator." from http://www.athenasmi.org/our-softwaredata/ ecocalculator

Audsley. E (1997) "Harmonisation of environmental life cycle assessment for agriculture". Final Report, Concerted Action AIR3-CT94-2028. European Commission, DG VI Agriculture, 139 pp.
Available online: http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=CELEX:32000L0060:EN:NOT

Azapagic, A., Stichnothe, H., Gujiba, H., Morgan, A., Espinoza Orias, N., Rong, Y., Jeswani, H. and Amienyo, D. (2010). "CCaLC, Carbon Footprint Tool." From http://www.ccalc.org.uk/.

Barbaro, AF., Bagajewicz, M. (2004) Managing Financial Risk in Planning under Uncertainty. AIChE Journal. Vol 50 (5) 963-989.

Barman, UK., Jana, SN., Garg, SK., Bhatnagar, A., Arasu, ART. (2005) Effect of inland water salinity on growth, feed conversion efficiency and intestinal enzyme activity in growing grey mullet, Mugil cephalus (Linn.): Field and laboratory studies. Aquacult. Int. 13, (3), 241-256.

Barrios, R., Siebel, M., van der Helm, A., Bosklopper, K., Gijzen, H., (2008). Environmental and financial life cycle impact assessment of drinking water production at Waternet. J Clean. Prod. 16:471-476.

Bartolomé, M., D'ors, A., Sánchez-Fortún, S. (2009) Toxic effects induced by salt stress on selected freshwater prokaryotic and eukaryotic microalgal species. Ecotoxicology, 18, (2), 174-179.

Bayart J-B (2008) Quantification des impacts reliés à l'utilisation de la ressource eau en analyse de cycle de vie: définition d'un cadre d'étude et dévelopement de facteurs de caractérisation (Maîtrise és Sciences Appliquées). EcolePolytechnique de Montréal, Montréal, Canada.

Bayart, JB., Bulle, C., Deschenes, L., Margni, M., Pfister, S., Vince, F., Koehler, A. (2010) A framework for assessing off-stream freshwater use in LCA. Int J Life Cycle Assess 15 (5):439–453

Beavis, P., Lundie, S. (2003). Integrated environmental assessment of tertiary and residuals treatment- LCA in the wastewater industry. Water Sci. Technol. 47 (7–8), 109–116.

Beccali, M., Cellura, M., Iudicello, M., Mistretta, M., (2010). Life cycle assessment of Italian citrus-based products. Sensitivity analysis and improvement scenarios. J of Environ. Manag. 91, 1415-1428.

Benetto, E., Dujet, C., Rousseaux, P. (2006) Possibility theory: a new approach to uncertainty analysis? Int J Life Cycle Assess 11:114– 116

Brentrup, F., Küsters, J., Lammel, J., Kuchlmann, H., (2000). Methods to estimate On-field Nitrogen emissions from crop production as an input to LCA studies in the agricultural sector. Int. J. Life Cycle Assess 5, 349-357.

Biegler, LT. (2010). Nonlinear Programming. Concepts, Algorithms, and Applications to Chemical Processes. Society for Industrial and Applied Mathematics and Mathematical Society, Optimization.

Björklund, AE. (2002) Survey of Approaches to Improve Reliability in LCA. Int. J. Life Cycle Assess. 7: 2, 64-72.

Boesch, M., Hellweg, S., Huibregts, M., Frishknecht, R. (2007) Applying cumulative exergy demand (CExD) indicators to the ecoinvent database. Int J Life Cycle Assess 12 (3): 181-190

Bonton, A., Bouchard, C., Barbeau, B., Jedrzejak, S. (2011). Comparative life cycle assessment of water treatment plants.Desalination 284(2012): 42-54.

Boscaiu, M., Ballesteros, G., Naranjo, M., Vicente, O., Boira, H. (2007) Responses of halophytes to salt stress. Buletinul USAMV-CN, 64.

Bosco, S., di Bene, C., Galli, M., Remorini, D., Massai, R., Bonari, E., (2011). Greenhouse gas emissions in the agricultural phase of wine production in the Maremma rural district in Tuscany, Italy. Italian Jof Agron. 6, 93-100.

Boulay, AM., Bayart, JB., Bulle, C., Margni, M., Deschênes, L. (2010) Using GIS to evaluate regional human health impacts from water use. In 20th SETAC Europe annual meeting, 23-27 May 2010, Seville, Spain.

Boulay, AM., Bouchard, C., Bulle, C., Deschênes, L., Margni, M. (2011a) Categorizing water for LCA inventory. Int J Life Cycle Assess 16 (7):639-651

Boulay, AM., Bulle, C., Bayart, JB., Deschênes, L., Margni, M. (2011b) Regional characterization of freshwater use in LCA: modelling direct impacts on human health. Environ Sci Technol 15:45(20):8948-57.

Boustead, I. (1992) Eco-balance methodology for commodity thermoplastics, a report for the European Center for Plastics in the Environment, Brussels.

Bright, DA., Addison, J. (2002) Derivation of Matrix Soil Standards for Salt under the British Columbia Contaminated Sites Regulation.

British Standard Institution (BSI) (2008a) PAS 2050:2008- Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. British Standard Institution, London, UK. www.bsigroup.uk

British Standard Institution (BSI) (2008b) Guide to PAS 2050:2008 - How to assess the carbon footprint of goods and services. British Standards Institution, London, UK. www.bsigroup.uk

British Standard Institution (BSI) (2011a) PAS 2050:2011 - Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. British Standard Institution, London, UK. www.bsigroup.uk

British Standard Institution (BSI) (2011b) PAS2060:2010. Specification for the demonstration of carbon neutrality. British Standard Institution, London, UK. www.bsigroup.uk

British Standard Institution (BSI) (2012) PAS 2050-1:2012 Assessment of life cycle greenhouse gas emissions from horticultural products – Supplementary requirements for the cradle-to-gate stages of GHG assessments of horticultural products. British Standard Institution, London, UK. www.bsigroup.uk

Bruinsma J. (ed) (2009). The Resource Outlook To 2050: By How Much Do Land, Water And Crop Yields Need To Increase By 2050? Expert Meeting On How To Feed The World In 2050 Food And Agriculture Organization Of The United Nations. Economic And Social Development Department. Rome.

Brundtland, G. (1987) Our Common Future: The World Commission on Environment and Development; Oxford University Press: Oxford

Burke, JJ., Moench, M. (2000). Groundwater and Society, Resources, Tensions and Opportunities: in Themes in Groundwater Management for the 21st Century. United Nations, New York. 170pp.

Burney, JA., Davis, SJ., Labell, DB. (2010) Greenhouse gas mitigation by agricultural intensification. PNAS 107 (26) 12052-12057.

Bussieck, MR., Vigerske, S. _MINLP Solver Software (2010) Accepted for publication inWiley Encyclopedia of Operations Research and Management Science.

Calheiros, CSC., Silva, G., Quitério, PVB., Crispim, LFC., Brix, H., Moura, SC., Castro, PML.

(2011) Toxicity of High Salinity Tannery Wastewater and Effects on Constructed Wetland Plants. International Journal of Phytoremediation, 14, (7), 669-680.

Cambrollé, J., Mateos-Naranjo, E., Redondo-Gómez, S., Luque, T., Figueroa, ME. (2011) Growth, reproductive and photosynthetic responses to copper in the yellow-horned poppy, Glaucium flavum Crantz. Environmental and Experimental Botany, 71, (1), 57-64.

Caro, R., Scandaliaris, J., Romero, ER., Casem, S., De Boeck, G., Giardina, JA. (2009) Preliminary results of Project CIUNT 26/A 428: Analysis of productivity and sustainability of sugarcane as energy crop in Tucumán and NOA. Universidad Nacional de Tucumán.

Cavalett, O., Junqueira, TL., Dias, MOS., Jesus, CDF., Mantelatto, PE., Cunha, MP., Franco, H., Cardoso, T., Maciel, R., Rosell, C., Bonomi, A. (2012) Environmental and economic assessment of sugarcane first generation biorefineries in Brazil. Clean Technol. and Environ. Policy, 1-12.

Cederberg, C., Stadig, M. (2003) System expansion and allocation in life cycle assessment of milk and beef production, Inter. J. Life Cycle Assess., Vol. 8(6) pp. 350-356.

Centre for Environmental Studies (2001) CML 2 baseline method 2000. University of Leiden, Leiden, The Netherlands.

Cellura, M., Ardente, F., Longo, S., (2012). From the LCA of food products to the environmental assessment of protected crops districts: A case-study in the south of Italy. J of Environ. Manag. 93, 194-208.

Centro azucarero argentino (2010) Índice de Precios Mayoristas del Azúcar en el Mercado Interno. http:// www.centroazucarero.com.ar/precio-mayoris.html. December 1st, 2012.

Cerutti, AK., Bagliani, M., Beccaro, GL., Bounous, G., (2010). Application of Ecological Footprint Analysis on nectarine production: methodological issues and results from a case study in Italy. J of Clean. Product. 18, 771-776.

Céspedes, AJ., García, MC., Cuadrado, IM. (2009) Caracterización de la explotación hortícola almeriense. In Fiapa y Fundación Cajamar.

Chapagain, AK., Hoekstra, AY., Savenije, HHG., Gautam, R., (2006). The water footprint of cotton consumption: An assessment of the impact of worldwide consumption of cotton products on the water resources in the cotton producing countries. Ecol. Econ. 60, 186

Chapagain, A.K., Orr, S., (2009). An improved water footprint methodology linking global consumption to local water resources: A case of Spanish tomatoes. J of Environ. Manag. 90, 1219.

Chauhan, MK., Varun, Chaudhary, S., Kumar, S., Samar (2011) Life cycle assessment of sugar industry: A review. Renew and Sustain. Energy Reviews, 15(7), 3445-3453.

Cheng, L., Subrahmanian, E., Westerberg, AW. (2003). Design and planning under uncertainty: Issues on problem formulation and solution. Comput. Chem. Eng., 27(6), 781–801.

Ciroth, A., Fleischer, G., Steinbach, J. (2004). Uncertainty Calculation in Life Cycle Assessments - A Combined Model of Simulation and Approximation. Int J Life Cycle Assess 9 (4): 216 – 226.

Club of Rome (1972) A blueprint for survival, Ecologist, 2 (1), 1-44.

Coello, CAC., Lamont,GB.,Veldhuizen, V. (2007) Evolutionary Algorithms for Solving Multi-Objective Problems. Springer 2nd Edition.

Cortez, LAB., Brossard, LE. (1997) Experiences on vinasses disposal. Part III: Combustion of vinasse-6 fuel oil emulsions. Braz. J. Chem. Eng., 14(1). doi: 10.1590/S0104-66321997000100002.

Cowan, CE., Mackay, D., Feijtel, TCJ., et al., eds.(1994). The Multi-Media Fate Model. A Vital Tool for Predicting the Fate of Chemicals. SETAC Press, Pensacola, FL, USA

De Fraiture, C. (2005). Assessment of Potential of Food Supply and Demand Using the Watersim Model. Columbo: International Water Management Institute.

De Smet, B., Hemming, C., Baumann, H., Cowell, S., Pesso, C., Sund, L., Markovic, V., Moilanen, T, Postlethwaithe, D. (1996) Life-cycle assessment and conceptually related programs, working group draft, SETAC-Europe, Brussels.

Directive 2000/60/EC of the European Parliament and of the council of 23 October 2000 establishing a framework for community action in the field of water policy. The European Parliament and the Council of the European Union.

Dixon, A., Simon, M. and Burkitt, T., (2003). Assessing the environmental impact of two options for smallscale wastewater treatment: Comparing a reedbed and an aerated biological filter using a life cycle approach. Ecol. Eng. 20 (4), 297–308.

Döll P (2009) Vulnerability to the impact of climate change on renewable groundwater resources: a global-scale assessment. Environ Res Lett 4 (3):1-12

Droogers, P., Aerts, J. (2005) Adaptation strategies to climate change and climate variability: a comparative study between seven contrasting river basins. Physics and Chemistry of the Earth 30, pp339–346.

Edwards, R., Larive, JF., Beziat, JC. (2011) Well-to-wheels Analysis of Future Automotive Fuels and Power trains in the European Context Report. European Commission Joint Research Centre, Institute for Energy ISBN 978-92-79-21395-3. http://publications.jrc.ec.europa.eu/repository/handle/111111111/22590 Last accessed December 1st, 2012.

El Instituto Andaluz de Investigación y Formación Agraria, P., Alimentaria y de la Producción Ecológica (IFAPA)
http://www.juntadeandalucia.es/agriculturaypesca/ifapa/web

El-Sayed Mohamed Mahgoub, M., van der Steen, N.P., Abu-Zeid, K., Vairavamoorthy, K., (2010). Towards sustainability in urban water: a life cycle analysis of the urban water system of Alexandria City, Egypt. J Clean. Prod. 18(10-11): 1100-1106.

Ematsa Staff (2009) Personal communication with water technicians, EMATSA, Empresa Municipal Mixta d'Aigües de Tarragona, S.A. Tarragona, Spain.

Emerton, L., Bos, E. (2004). Value. Counting Ecosystems as an Economic Part of Water Infrastructure. IUCN, Gland, Switzerland and Cambridge, UK. 88pp.

Epp, A., Reichenbach, A. (1999). "ULME. Umweltfolgen von Lebensmitteleinkäufen." from http://www.ulme.ethz.ch/.

Ercin, A., Aldaya, M., Hoekstra, A., (2011). Corporate Water Footprint Accounting and Impact Assessment: The Case of the Water Footprint of a Sugar-Containing Carbonated Beverage. Water Resourc. Manag. 25, 721-741.

Espinar, JL., García, LV., Clemente, L. (2005) Seed storage conditions change the germination pattern of clonal growth plants in Mediterranean salt marshes. American J. of Botany, 92, (7), 1094-1101.

ESRI (2011) ArcGis Deckstop. ArcView v10.1. www.esri.com/software/arcgis/index.html

EUPHOROS (2008-2012). "Efficient use of inputs in protected horticulture." <http://www.euphoros.wur.nl/UK/>.

European Commission (1996). Technical Guidance Document in Support of Commission Directive 93/67/EEC on Risk Assessment for New Notified Substances and Commission

Regulation (EC) No1488/94 on Risk Assessment for Existing Substances. Luxembourg

European Comission (2012) Sustainable agriculture for the future we want. http://ec.europa.eu/agriculture/events/2012/rio-side-event/brochure_en.pdf Last accessed December 20th, 2012

European Union Water Framework Directive (EUWFD). Directive 2000/60/EC of the European Parliament and the council. October 23, 2000. http://ec.europa.eu/environment/water/water-framework/index_en.html. Accessed July 2012.

Faist, M., Pfister, S., Koehler, A., de Giovanetti, L., Arena, AP., Zah, R. (2011) Taking into account water use impacts in the LCA of biofuels: An Argentinean case study. Int J Life Cycle Assess, 16 (9) 869-877.

Falkenmark, M., Lundqvist, J., Widstrand, C. (1989) Macro-scale water scarcity requires micro-scale approaches. Aspects of vulnerability in semi-arid development. Nat Resour Forum 13 (4):258-267

Falkenmark, M., Rokström, J. (2004) Balancing water for humans and nature. The new approach in ecohydrology.Earthscan, London, UK. FAO (Food and Agriculture Organization of the United Nations) (2008) Estimated world water use, http://www.fao.org/nr/water/infores_maps.html

Farrell, AE., Plevin, RJ., Turner, BT., Jones, AD., O'Hare, M., Kammen, DM. (2006) Ethanol can contribute to energy and environmental goals. Science 311(5781):506–508

Farreny, R. , Gasol, CM. (2012). "CO2ZW: Carbon Footprint Tool for Waste Management in Europe." from http://co2zw.eu.sostenipra.cat/

Fazil, AM. (2005) A primer on risk assessment modelling: focus on seafood products. Food and Agriculture Organization of United Fisheries technical Paper, Rome

Fernández, C., Pérez, J., (2004) Caracterización de los invernaderos de la provincia de Almería. Cajamar.

Fernández, MD., Orgaz, F., Ferreres, E., López, JC., Céspedes, A., Pérez, J., Bonachela, S., Gallardo, M.(2001) Programación del riego de cultivos hortícolas bajo invernadero en el sudeste español. In CAJA MAR (Caja Rural de Almería y Málaga).

Fessehaziona, MK., Stirzaker, RJ., Annandale,JG., Everson, CS. (2011) Improving nitrogen and irrigation water use efficiency through adaptive management: A case study using annual ryegrass. Agriculture, ecosystems, environment 141 (2011): 350-358

Finkel, AM (1990). Confronting uncertainty in risk management— A guide for decision-makers. Washington, DC: Resources for the Future.

Finnveden, G. (2005) Site-dependent life-cycle impact assessment in Sweden. Int. J. Life Cycle Assess. 10 (4), 235–239.

Finnveden, G., Hauschild, M., Ekwall, T., Guinée, J., Heijungs, R., Hellweg, S., Kowhler, A., Pennington, D. and Suh, S., (2009). "Recent developments in Life Cycle Assessment." Journal of Environmental Management 91, pp. 1-21.

Fischer, G., Tubiello, FN., van Velthuizen, H., Wiberg, DA. (2007) Climate change impacts on irrigation water requirements: Effects of mitigation, 1990–2080. Technological Forecasting & Social Change 74, 1083–1107.

Foley, JA., Ramankutty, N., Brauman, KA. (2011) Solutions for a cultivated planet. Nature 478, 337-342.

Food and Agriculture Organization of the United Nations (FAO) (2003) World agriculture: towards 2015/2030. , in: Bruinsma, J. (Ed.), An FAO perspective, Earthscan Publications Ltd London.

Food and Agriculture Organization of the United Nations (FAO) (2004). Economic valuation of water resources in agriculture: from a sectoral to a functional perspective of natural resource management. FAO Water Report 27. 186pp

Food and Agriculture Organization on the United Nations (FAO- AQUASTAT) (2006a): FAO's information system on water and agriculture. Land and water development division, food and agriculture organization,
http://www.fao.org/ag/agl/aglw/aquastat/dbase/index.stm

Food and Agriculture Organization on the United Nations (FAO- FAOSTAT) (2006b): FAO's information system on water and agriculture. Land and water development division, food and agriculture organization,
http://faostat.fao.org/site/567/DesktopDefault.aspx?PageID=567#ancor

Food and Agriculture Organization of the United Nations (FAO) (2008). Options for Decision Makers. Expert Meeting on Climate Change Adaptation and Mitigation. FAO Headquarters, Rome, http://www.fao.org/fileadmin/user_upload/foodclimate/presentations/EM1/OptionsEM1.pdf

Food and Agriculture Organization of the United Nations (FAO) (2011a) The state of the world's land and water resources for food and agriculture (SOLAW) - Managing systems at risk. Food and Agriculture Organization of the United Nations, Rome and Earthscan, London.

Food and Agriculture Organization of the United Nations (FAO) (2011b) Climate change, water and food security. FAO water reports, Nº36. ISBN 978-92-5-106795-6, Rome

Food and Agriculture Organization of the United Nations (FAO) (2012) www.fao.org

Food Database (FDDB) (2012)
http://fddb.info/db/en/food/natural_product_sugar/index.html.

Friedrich, E. (2001). The Use of Environmental Life Cycle Assessment for the Production of Potable Water. M.Sc. thesis, Pollution Research Group, University of Natal, Durban, South Africa.

Frischknecht, R., Jungbluth, N. (2003) Implementation of Life Cycle Impact Assessment Methods. ecoinvent report Nº3, v2.0 Swiss Centre for Life Cycle Inventory, Dübendorf, Switzerland.

Frischknecht, R., Jungbluth, N., Althaus, HJ., Doka, G., Dones, R., Heck, T., Hellweg, S., Hischier, R., Nemecek, T., Rebitzer, G., Spielmann, M. (2004) The ecoinvent Database: Overview and Methodological Framework. Int J LCA 10 (1):3-9

Frischknecht, R., Steiner, R., Braunschweig, A., Egli, N., Hildesheimer, G. (2006) Swiss Ecological Scarcity Method: The New Version 2006. Swiss Federal Office for the Environment (FOEN), Zurich and Bern, Switzerland

Frischknecht, R., Jungbluth, N., Althaus, HJ., Doka, G., Heck, T., Hellweg, S., Hischier, R., Nemecek, T., Rebitzer, G., Spielmann, M., Wernet, G. (2007) Overview and Methodology. Ecoinvent report N1. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.

Funtowicz, S., Ravetz, J. (1990). Uncertainty and Quality in Science for Policy. Dordrecht, Netherlands: Kluwer

GAMS Development Corporation (2012), W., DC, USA, GAMS.

Gan, Y., Liang, C., Campbell, C. A., Zentner, R. P., Lemke, R. L., Wang, H., Yang, C.(2012a) Carbon footprint of spring wheat in response to fallow frequency and soil carbon changes over 25 years on the semiarid Canadian prairie. European J of Agron. 43: 175-184.

Gan, Y., Liang, C.,May, W.,Malhi, S. S.,Niu, J.,Wang, X (2012b) Carbon footprint of spring barley in relation to preceding oilseeds and N fertilization. Inter Jof Life Cycle Assess. 17 (5): 635-645

García, CA., Fuentes, A., Hennecke, A., Riegelhaupt, E., Manzini, F., Masera, O. (2011) Life-cycle greenhouse gas emissions and energy balances of sugarcane ethanol production in Mexico. Appl. Energy, 88(6), 2088-2097.

Gasol, C.M., Gabarrell, X., Rigola, M., González-García, S. and Rieradevall, J. (2011) Environmental assessment: (LCA) and spatial modelling (GIS) of energy crop implementation on local scale. Biomass and Bioenergy 35(7):2975-2985.

Geisler, G., Hellweg, S., Hungerbühler, K. (2005) Uncertainty Analysis in Life Cycle Assessment (LCA): Case Study on Plant-Protection Products and Implications for Decision Making. Int. J. Life Cycle Assess. 10 (3) 184 – 192.

Geyer, R., Stoms, DM., Lindner, JP., Davis, FW. and Wittstock, B. (2010) Coupling GIS and LCA for biodiversity assessments of land use. The international journal of life cycle assessment 15(5):454-467.

Gleick PH (1996) Basic Water Requirements for Human Activities: Meeting Basic Needs. Water Int 21 (2):83-92

Glenn, E., Thompson, TL., Frye, R., Riley, J., Baumgartner, D., (1995) Effects of salinity on growth and evapotranspiration of Typha domingensis Pers. Aquatic Botany, 52, (1–2), 75-91.
Goedkoop M., Spriensma, R. (1999). The Eco-Indicator 99: A damage orientated method for life cycle impact assessment. VROM, Ministry of Spatial Planning, Housing and the Environment, The Hague, The Netherlands.

Global G.A.P. stands for Good Agricultural Practice www.globalgap.org

Goedkoop, M. and Spriensma, R. (2000) The Eco-Indicator 99. A damage oriented method for life cycle impact assessment. PRé Consultants B.V., Amersfoort, the Netherlands.

Goedkoop, M., Heijungs, R., Huijbregts, MAJ., De Schryver, A., Struijs, J., van Zelm, R. (2009) ReCiPe 2008. A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. First edition, Report I: Characterisation, the Hague, the Netherlands.

Greenwood, M. Predicting the effects of salinity on three dominant macrophytes: An anticipatory approach to the restoration of degraded coastal wetlands in NSW, Australia.

Guinée, J., Heijungs, R. (1993). A proposal for the classification of toxic substances within the framework of life cycle assessment of products. Chemosphere 26, 1925-1944.

Guinée, J., Heijungs, R., van Oers, L. (1996) LCA impact assessment of toxic releases. Ministry of Housing, Spatial Planning and Environment (VROM), The Hague, The Netherlands.

Guinee, JB., de Bruijn, H., van Duin, R., Gorree, M., Heijungs, R., Huijbregts, MAJ., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Sleeswijk, AW., Suh, S., Udo de Haes, HA. (2001) Life Cycle Assessment—An Operational Guide to the ISO Standards; Centre of Environmental Science, Leiden University (CML): Leiden, The Netherlands.

Guinée, JB., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H., Bruijn, H., van Duin, R. and Huijbregts, M. AJ., (2002). Handbook on life cycle assessment. Operational guide to the ISO standards. Kluwer, Dordrecht, The Netherlands.

Guinée, JB., van Oers, L., de Koning, A., Tamis, W. (2006) Life cycle approaches for conservation agriculture. CML report 171, Institute of Environmental Sciences, Department of Industrial Ecology & Department of Environmental Biology, Leiden, the Netherlands.
Guma, IR., Padrón-Mederos, MA., Santos-Guerra, A., Reyes-Betancort, JA. (2010) Effect of temperature and salinity on germination of Salsola vermiculata L. (Chenopodiaceae) from Canary Islands. J of Arid Environ 74, (6), 708-711.

Guldimann, T. (2000). The story of risk metrics. Risk, 13(1), 56–58.

Guma, IR., Padrón-Mederos, MA., Santos-Guerra, A., Reyes-Betancort, JA., (2010) Effect of temperature and salinity on germination of Salsola vermiculata L. (Chenopodiaceae) from Canary Islands. J of Arid Environ, 74, (6), 708-711.
Halberg, N., (2004a). DIAS report. Life Cycle assessment in the Agri-food sector. Proceedings from the 4th International Conference, October 6-8, 2003. Danish. Institute of Agricultural Sciences, Bygholm, Denmark. nº 61.

Hanafiah, M., Xenopoulos, M., Pfister, S., Leuven, RS., Huijbregts, M. (2011) Characterization factors for water consumption and greenhouse gas emissions based on freshwater fish species extinction. Environ Sci Technol 45 (12):5272–5278

Harris, S., Narayanaswamy, V. (2009) A Literature Review of Life Cycle Assessment in Agriculture. Rural Industries Research and Development Corporation. ISBN 1-74151 833-4

Hauschild, M., Potting, J. (2005) Spatial differentiation in life cycle assessment - the EDIP 2003 methodology; Danish Environmental Protection Agency: Copenhagen, Denmark http://www2.mst. dk/udgiv/publications/2005/87-7614-579-4/pdf/87-7614-580-8.pdf.

Heijungs, R. (1995). Harmonization of methods for impact assessment. Environ Sci Pollut Res Int 2:217–24.

Heijungs, R., Guinée, JB., Huppes,G., Lankreijer,RM., Udo de Haes, HA., Wegener Sleeswijk,A., Ansems,AMM, Eggels,PG., van Duin, R., de Goede, HP. (1992). Environmental life cycle assessment of products. Guidelines and backgrounds. Centre of Environmental Sciences, Leiden, the Netherlands.

Hertwich, E., Matales, SF., Pease, WS., (2001). Human toxicity potentials for life-cycle assessment and toxics release inventory risk screening. Environ Toxicol Chem 20:928–39

Hertwich, EG., McKone, TE., Pease, WS. (2000). A Systematic Uncertainty Analysis of an Evaluative Fate and Exposure Model. Risk Analysis 20 (4): 439–454.

Heuvelmans, G., Muys, B., Feyen, J. (2005) Extending the life cycle methodology to cover impacts of land use systems on the water balance. Int J Life Cycle Assess 10:113–119

Hillier, J., Hawes, C., Squire, G., Hilton, A., Wale, S., Smith, P., (2009). The carbon footprints of food crop production. Int J of Agricult Sustain 7, 107-118.

Hillier, J., Water, C., Malin, D., Garcia-Suarez, T., Mila-i-Canals, L. and Smith, P., (2011). "A farm-focused calculator for emissions from crop and livestock production." Environ Model & Soft. 26, pp. 1070-1078.

Hoekstra, A. Y. (ed) (2003) 'Virtual water trade: Proceedings of the International Expert Meeting on Virtual Water Trade', 12–13 December 2002, Value of Water Research Report Series No 12, UNESCO-IHE, Delft, Netherlands, www.waterfootprint.org/Reports/Report12.pdf

Hoekstra, A., Chapagain, A., Mekonnen, M., (2005). Extended calculator of Water Footprint Network. www.waterfootprint.org

Hoekstra, AY., Chapagain, AK., Aldaya, MM., Mekonnen, MM. (2009) Water Footprint Manual: State of the Art 2009, Water Footprint Network, Enschede, the Netherlands www.waterfootprint.org/downloads/WaterFootprintManual2009.pdf

Hoekstra, AY., Chapagain, AK., Aldaya, MM., Mekonnen, MM. (2011) The water footprint assessment manual: setting the global standard. Water Footprint network. Enschede, the

Netherlands.

Hoffman, GJ., Evans, RG., Jensen,ME., Martin,DL., Elliott, RL. (Eds.). (2007). Design and operation of farm irrigation systems. 2nd Edition. ABABE, MI, USA.

Hofstetter, P. (Ed.) (1998). Perspectives In Life Cycle Impact Assessment. A structured approach to combine models of the technosphere, ecosphere and valuesphere, Dordrecht, Kluwer Academic Publishers.

Hogan, L., Beal, R., Hunt, R. (1996). Threshold inventory interpretation methodology: a casestudy of three juice container systems. Internat J LCA 1:159–67

Hollis, GE. (1992) The causes of wetland loss and degradation in the Mediterranean. pp83–90.

Hong, J., Shaked, S., Rosenbaum, RK., Jolliet, O. (2010). Analytical uncertainty propagation in life cycle inventory and impact assessment: application to an automobile front panel. Int J Life Cycle Assess 15:499–510.

Hospido, A., Moreira, M. T., Fernández-Couto, M., Feijoo, G., (2004) Environmental performance of a municipal wastewater treatment plant. Int J Life Cycle Assess 9(4): 261–271.

Hugot, E. (1986) Handbook of Cane Sugar Engineering. Elsevier. Amsterdam, The Netherlands. International Organisation for Standardisation (2006a) ISO 14040: Environmental management- life cycle assessment principles and framework. Geneva, Switzerland.

Huijbregts, MAJ., Gilijamse, W., Ragas, AMJ., Reijnders, L. (2003). Evaluating uncertainty in environmental life-cycle assessment a case study comparing two insulation options for a Dutch one-family dwelling. Environ Sci Technol 37(11):2600–2608.

Huijbregts, MAJ. (1998). Application of uncertainty and variability in LCA. Part I: A general framework for the analysis of uncertainty and variability in life cycle assessment. Int J Life Cycle Assess 3 (5): 273-280.

Huijbregts, MAJ., Thissen, U., Guinée, J. (2000a). Priority assessment of toxic substances in life cycle assessment. Part I: Calculation of toxicity potentials for 181 substances with the nested multi-media fate, exposure and effects model USES-LCA. Chemosphere 41:541–73

Huijbregts, MAJ., Schopp, W., Verkuijlen, E., Heijungs, R., Reijnders, L. (2000b) Spatially Explicit Characterization of Acidifying and Eutrophying Air Pollution in Life-Cycle

Assessment. J. Ind. Ecol. 4 (3), 75–92.

Huijbregts MAJ, Norris G, Bretz R, Ciroth A, Maurice B, Von Bahr B, Weidema B, De Beaufort ASH (2001) Framework for modelling data uncertainty in life cycle inventories.Int. J. Life Cycle Assess. 6: 127-132.

Huijbregts MAJ, Gilijamse W, Ragas AMJ, Reijnders L (2003). Evaluating uncertainty in environmental life-cycle assessment a case study comparing two insulation options for a Dutch one-family dwelling. Environ Sci Technol 37(11):2600–2608.

Humbert, S., Manneh, R., Shaked, S., Wannaz, C., Horvath, A., Deschênes, L., Jolliet, O., Margni, M. (2009) Assessing regional intake fractions in North America. Sci. Total Environ. 407 (17), 4812− 4820.

Hung, ML., Ma, H. (2009) Quantifying system uncertainty of life cycle assessment based on Monte Carlo simulation. Int. J. Life Cycle Assess., 14, 19-27.

Hutchinson, I., Salinity Tolerance of plants of estuarine wetlands and associated uplands.

Ierapetritou MG, Pistikopoulos EN (1994). Simultaneous incorporation of flexibility and economic risk in operational planning under uncertainty. Comput. Chem. Eng. 18(3), 163–189.

Ingram, D.L., Thomas Fernandez, R., (2012). Life cycle assessment: A tool for determining the environmental impact of horticultural crop production. HortTechnology 22, 275-279.

International Organisation for Standardisation (2006a) ISO 14040: Environmental management- life cycle assessment principles and framework. Geneva, Switzerland. www.iso.org.
International Organisation for Standardisation (2006b). ISO 14044: Environmental management— life cycle assessment—requirements and guidelines. Geneva, Switzerland.www.iso.org.

International Organisation for Standardisation (2006c) ISO 14064-1 Greenhouse gases - Part 1: Specification with guidance at the organization level for quantification and reporting of greenhouse gas emissions and removals. Geneva, Switzerland. www.iso.org.

International Organisation for Standardisation (2011a) ISO/DIS 14067-2. Carbon footprint of products - Requirements and guidelines for quantification and communication. Geneva, Switzerland. www.iso.org.
International Organisation for Standardisation (2011b) ISO/PRF TR 14069. Greenhouse gases (GHG) - Quantification and reporting of GHG emissions for organizations (Carbonfootprint

of organization) -- Guidance for the application of ISO 14064-1. Geneva, Switzerland. www.iso.org.

International Organisation for Standardisation (2011c) ISO 14021:2001+A1:2011: Environmental labels and declarations. Self-declared environmental claims (Type II environmental labelling) www.iso.org.

International Organisation for Standardisation (2012) ISO /TR 14047:2012 Environmental management -- Life cycle assessment -- Illustrative examples on how to apply ISO 14044 to impact assessment situations. www.iso.org.

International Organisation for Standardisation (under development) ISO/DIS14046-Environmental Management- Water Footprint- Principles, requirements and guidelines.

International Water Management Institute (2007) Water for Food, Water for Life: A Comprehensive Assessment of Water Management in Agriculture; Earthscan: London, UK.

Intergovernmental Panel on Climate Change (IPCC), Summary for Policymakers. In: Climate Change (2007): The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC) [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M.Tignor and H.L. Miller (eds.)]. 2007, Cambridge, University Press. www.ipcc.ch.

Jefferies, D., Muñoz, I., Hodges, J., King, V.J., Aldaya, M., Ercin, A.E., Milà i Canals, L., Hoekstra, A.Y. (2012). Water Footprint and Life Cycle Assessment as approaches to assess potential impacts of products on water consumption. Key learning points from pilot studies on tea and margarine. J of Clean Prod 33, 155.

Joint Research Centre - Institute for Environment and Sustainability (2010a) ILCD Handbook Analysing of existing Environmental Impact Assessment methodologies for use in Life Cycle Assessment. Luxembourg

Joint Research Centre - Institute for Environment and Sustainability (2010b) ILCD Handbook General guide for Life Cycle Assessment – Recommendations for Life Cycle ImpactAssessment in the European context - based on existing environmental impact assessment models and factors. Luxembourg

Jolliet O, Margni M, Charles R, Humbert S, Payet J, Rebitzer G, Rosenbaum R (2003) IMPACT 2002+: a new life cycle impact assessment methodology. Int J of Life Cycle Assess 8 (6): 324-330.

Jolliet, O., Müller-Wenk, R., Bare, J., Brent, A., Goedkoop, M., Heijungs, R., Itsubo, N., Peña, C., Pennington ,D., Potting, J., Rebitzer, G., Stewart, M., Udo de Haes, H., Weidema, B.

(2004) The LCIA midpoint-damage framework of the UNEP/SETAC Life Cycle Initiative. Inter J of Life Cycle Assess 9 (6): 394-404.

Jolliet O. (1995). Experts need experts' judgement: Fate in LCA impact assessment. LCA News, SETAC-Europe 5:4–6

Jolly, I. D.; McEwan, K. L.; Holland, K. L. (2008) A review of groundwater–surface water interactions in arid/semi-arid wetlands and the consequences of salinity for wetland ecology. Ecohydrology, 1, (1), 43-58.

Joseph, K. (2010) Gain Report: Biofuels Annual Argentina, USDA Foreign agricultural Service.

Jungbluth, N., Chudacoff, M., Dauriat, A., Dinkel, F., Doka, G., Faist, Emmenegger, M., Gnansounou, E., Kljun, N., Schleiss, K., Spielmann, M., Stettler, C., Sutter, J. (2007) Life cycle inventories of bioenergy. Ecoinvent report no. 17. Swiss Centre for Life Cycle Inventories, Dübendorf.

Klepper, O., Traas, TP., Schouten ,A., Korthals, GW., de Zwart, D. (1999). Estimating the effect on soil organisms of exceeding no-observed effect concentrations (NOECs) of persistent toxicants. Ecotoxicology, 8: 9–21.

Koehler, A. (2008) Water use in LCA: managing the planet's freshwater resources. Int J Life Cycle Assess 13(6):451-455

Koehler, A., Aoustin, E. (2008) UNEP/SETAC Life Cycle Initiative SETAC Europe Annual Meeting, Project Group on assessment of use and depletion of water resources within LCA. Warsaw, Poland.

Kostin, AM., Guillén-Gosálbez, G., Mele, FD., Bagajewicz, MJ., Jiménez, L. (2011a) A novel rolling horizon strategy for the strategic planning of supply chains. Application to the sugarcane industry of Argentina. Comput and Chem. Eng. 35:2540– 2563.

Kostin, AM., Guillén-Gosálbez, G., Mele, FD., Bagajewicz, MJ., Jiménez, L. (2011b) Design and planning of infrastructures for bioethanol and sugar production under demand uncertainty. Chem. Eng. Res. and Des. doi:10.1016/j.cherd.2011.07.013

Kottek, M., Grieser J., Beck C., Rudolf B., Rubel F. (2006): World Map of the Köppen-Geiger climate classification updated. Meteorol. Z., 15, 259-263. DOI: 10.1127/0941-2948/2006/0130.

Kounina, A., Margni, M., Bayart, JB., Boulay, AM., Berger, M., Bulle, C., Frischknecht, R., Koehler, A., Milà i Canals, Ll., Motoshita, M., Núñez, M., Peters, G., Pfister, S., Ridoutt, B., Van Zelm, R., Verones, F., Humbert, S. (2012) Review of methods addressing freshwater availability in life cycle inventory and impact assessment. Int J Life Cycle Assess. DOI 10.1007/s11367-012-0519-3

Krewitt, W., Trukenmüller, A., Bachmann, TM., Heck, T. (2001) Country-specific damage factors for air pollutants. Int. J. Life Cycle Assess., 6 (4), 199–210.

Kurukulasuriya, P. and Rosenthal, S. (2003). Climate Change and Agriculture: A Review of Impacts and Adaptations. World Bank Climate Change Series. Vol. 91. World Bank Environment Department, Washington.

Lassaux, S., Renzoni, R., Germain,A., (2007). Life Cycle Assessment of water from the pumping station to the wastewater treatment plant. In J LCA, 12 (2): 118-126.

Law AM, Kelton WD (2000) Simulation Modeling & Analysis, 3rd Edition; McGraw-Hill: New York.

Lewandowska A, Zenon F, Andrzej P (2004) Comparative lca of industrial objects part1: lca data quality assurance — sensitivity analysis and pedigree matrix. Int. J. Life Cycle Assess. 9 (2): 86-89.

Lillebø, AI., Pardal, MA., Neto, JM., Marques, JC. (2003) Salinity as the major factor affecting Scirpus maritimus annual dynamics: Evidence from field data and greenhouse experiment. Aquat Botany, 77, (2), 111-120.

Lim, SR., Park, JM., (2008). Environmental impact minimization of a total wastewater treatment network system from a life cycle perspective. J Environ Manag, 90 (2009): 1454–1462

Lissner, J., Schierup, HH. (1997) Effects of salinity on the growth of Phragmites australis. Aquat. Botany, 55, (4), 247-260.

Lloyd, SM., Ries, R. (2007) Characterizing, propagating, and analyzing uncertainty in life-cycle assessment: a survey of quantitative approaches. J. Ind. Ecol.11:161–179

Losa, TM., S., R. G., Estudio florístico y geobotánico de la provincia de Almería. Archivos Instituto Aclimatación de Almería. 13 (1), 5-111.

Lundie, S., Peters, G. M., Beavis P.C., (2004). Life Cycle Assessment for Sustainable Metropolitan Water Systems Planning. Environ Sci Technol 38(13): 3465-3473.

Lundie, S., Peters, G. M., Beavis P.C., (2005). Quantitative sytems analysis as a trategic planning approach for metropolitan water service providers. Water Sci. Technol.52 (9):11-20.

Lundin, M., Bengtsson, M., Molander, S., (2000). Life cycle assessment of wastewater systems: Influence of system boundaries and scale on calculated environmental loads. Environ Sci Technol 34(1): 180–186.

Lundin, M., Morrison, G.M., (2002). A life cycle assessment based procedure for development of environmental sustainability indicators for urban water systems. Urban Water 4:145-152.

Luo, L., van der Voet, E., Huppes, G. (2009) Life cycle assessment and life cycle costing of bioethanol from sugarcane in Brazil. Renew. and Sustain. Energy Reviews, 13(6-7), 1613-1619.

Ma, BL., Liang, BC., Biswas, DK., Morrison, MJ, McLaughlin, NB. (2012) The carbon footprint of maize production as affected by nitrogen fertilizer and maize-legume rotations. Nutrient Cycling in Agroecosystems: 1-17

Mackay D and Seth R. (1999). The role of mass balance modelling in impact assessment and pollution prevention. In: Sikar SK and Diwekar U (eds), Tools and Methods for Pollution Prevention, pp 157–79. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Maendly, R., Humbert, S. (submitted) Empirical characterization model and factors assessing aquatic biodiversity damages of hydropower water use. Int J Life Cycle Assess.

Ministry of agriculture, food and environment (MAGRAMA). Spanish Government. Reports of surfaces and agricultural productions from 2006 to 2011 (In Spanish). http://www.magrama.es/.

Maheswarappa, HP., Srinivasan, V., Lal, R. (2011). Carbon footprint and sustainability of agricultural production systems in India. J of Crop Improv 25, 303-322.

Manneh, R., Margni, M., Deschênes, L. (2010) Spatial Variability of Intake Fractions for Canadian Emission Scenarios: A Comparison between Three Resolution Scales. Environ. Sci. Technol 44 (11), 4217–4224.

Margni, M., Pennington, DW., Bennett, DH. (2004). Cyclic exchanges and level of coupling between environmental media: Intermedia feedback in multimedia fate models. Environ Sci Tech 38:5450–7

Martinez-Blanco (2012). Sustainability assessment of municipal compost use in horticulture using a life cycle approach. PhD Doctoral dissertation by Universitat autònoma de Barcelona, Barcelona, Spain.

McKone, TE, Nazaroff, W W., Berck, P., Auffhammer, M., Lipman, T., Torn, M. S., Masanet, E., Lobscheid, A., Santero, N., Mishra, U., Barrett, A., Bomberg, M., Fingerman, K., Scown, C., Strogen, B., Horvath, A. (2011) Grand Challenges for Life-Cycle Assessment of Biofuels. Environ. Sci. Technol, 45 (5), 1751−1756.

Meador, MR., Kelso, WE., (1989) Behavior and Movements of Largemouth Bass in Response to Salinity. Transact of the American Fisheries Soc., 118, (4), 409-415.

Meadows, DH., Meadows, DL., Randers, J., Behrens, W. (1972) The limits to growth: a report for the Club of Rome’s Project on the Predicament of Mankind, ISBN 0-330-241699, Universe Books, New York.

Mekonnen MM, Hoekstra AY (2010) The green, blue and grey water footprint of crops and derived crop products. Value of Water Research Report Series No. 47 (with supporting information), UNESCO-IHE, Delft, the Netherlands. http://www.waterfootprint.org/Reports/Report47-WaterFootprintCrops-Vol1.pdf

Mekonnen, M., Hoekstra, A., (2011). The green, blue and grey water footprint of crops and derived crop products. Hydrol. and Earth System Sci. 15(5), 1577

Mele, FD., Kostin, A., Guillén-Gosálbez, G., Jiménez, L. (2011) Multiobjective Model for More Sustainable Fuel Supply Chains. A Case Study of the Sugarcane Industry in Argentina. Industrial & Eng. Chem. Res. 50, 4939–4958

Meneses, M., Pasqualino, J. and Castells, F., (2010a). Environmental assessment of urban wastewater reuse: Treatment alternatives and applications. Chemosphere. 81 266–27210.1016/j.chemosphere.2010.05.053

Meneses, M., Pasqualino, J., Céspedes, R. and Castells, F., (2010b). Alternatives for reducing the Environmental Impact of the main residue from a desalination plant. J. Ind. Ecol. appl. and implement., Volume 14, Number 3.

Messac, A, A Ismail-Yahaya, & C Mattson. The Normalized Normal Constraint Method for Generating the Pareto Frontier. Structural and Multidisciplinary Optimization 25: (2003) 86-98.

Milà i Canals, L., Chenoweth, J., Chapagain, A., Orr, S., Anton, A., Clift, R. (2009) Assessing freshwater use impacts in LCA: Part I-inventory modelling and characterization factors for the main impact pathways. Int J Life Cycle Assess 14 (1):28-42

Millennium Ecosystem Assessment (MEA) (2005) Ecosystem services and human well-being: wetlands and water synthesis. World Resources Institute, Washington DC. 68pp.

Ministerio de Medio Ambiente, (2010). Programa AGUA. Programa A.G.U.A. (Acciones para la Gestión y la Utilización del Agua) http://www.mma.es/secciones/agua/entrada.htm. Last accessed July 3rd, 2012 [In Spanish]

Ministerio de Planificación Federal Inversión Pública y servicios (2010) Secretaria de Energía. Precios de biocombustibles. http://energia3.mecon.gov.ar/contenidos/verpagina.php?idpagina=3033. Accessed December 1st, 2012.

Mohapatra, PK., Siebel, MA., Gijzen, HJ., van der Hoek, JP., Groot, CA., (2002). Improving ecoefficiency of Amsterdam water supply: a LCA approach, J Water Supp Res Technol Aqua 51 (4): 217–227.

Molden, D., Vithanage, M., de Fraiture, C., Faures, J., Gordon, L., Molle, F., Peden, D., (2011) Water Availability and Its Use in Agriculture. Elsevier B.V.

Molina, F. (2006) Ficha Informativa de los Humedales de Ramsar (FIR). In Consejería de Medio Ambiente de la Junta de Andalucía (España).

Morgan, M., Henrion, M. (1990). Uncertainty Cambridge University Press, NY NY.

Morrison, J., Schulte, P. (2010) Corporate Water Accounting. An Analysis of Methods and Tools for Measuring Water Use and Its Impacts. United Nations Environment Programme, United Nations Global Compact, Pacific Institute ISBN-13: 978-1-893790-23-0 Available online: http://www.unep.fr/scp/publications/details.asp?id=WEB/0164/PA

Motoshita, M., Itsubo, N., Inaba, A. (2010a) Damage assessment of water scarcity for agricultural use. Proceedings of 9th international conference on EcoBalance. Towards & Beyond 2020. D1-1410 National Institute of Advanced Ind. Sci. and Technol. (AIST).

Motoshita, M., Itsubo, N., Inaba, A. (2010b) Development of impact factors on damage to health by infectious diseases caused by domestic water scarcity. Int J Life Cycle Assess 16(1):65-73

Mourad, AL., Coltro, L. Oliveira, PA. P. L. V., (2007). "A Simple Methodology for Elaborating the Life Cycle Inventory of Agricultural Products." Inter J Life Cycle Assess 12 (6), pp. 408-413.

Msangi, A., Palazzo, M., Batka, M., Magalhaes, R.,Valmonte-Santos, M., Ewing and Le D.

(2009). Climate Change Impact on Agriculture and Costs of Adaptation. IFPRI. Washingon DC.

Mufarrege, M., Di Luca, G., Hadad, H., Maine, M., (2011) Adaptability of Typha domingensis to high pH and salinity. Ecotoxicology, 20, (2), 457-465.

Muñoz, I., Milà i Canals, L., Fernández-Alba, AR. (2010) Life Cycle Assessment of Water Supply Plans in Mediterranean Spain. The Ebro River Transfer versus the AGUA Programme. J Ind Ecol 14 (6):902-918

Muñoz, I., Rodríguez Fernández-Alba, A., (2008). Reducing the environmental impacts of reverse osmosis desalination by using brackish groundwater resources. Water Res. 42: 801–811.

Muñoz, I., Milà i Canals, Ll., Fernández-Alba, A., (2010). Life Cycle Assessment of Water Supply Plans in Mediterranean Spain: The Ebro River Transfer versus the AGUA Programme. J. Ind. Ecol. 14 (6): 902-918.

Muñoz, I., Rodríguez A., Rosal, R. and Fernández- Alba, A.R., (2009). Life cycle assessment of urban wastewater reuse with ozonation as tertiary treatment: A focus on toxicity related impacts. Sci. Total Environ. 407(4): 1245–1256.

Model de l'Us Sostenible d'Aigua a l'Agricultura (MUSA) (2011-2012). Projecte VALOR2010-00008. Agència de Gestió d'Ajuts Universitaris i de Recerca (AGAUR). Generalitat de Catalunya.

Mutel, CL., Hellweg, S. (2009) Regionalized Life Cycle Assessment: Computational Methodology and Application to Inventory Databases. Environ. Sci. Technol 43 (15), 5797−5803.

Mutel, CL., Pfister, S., Hellweg, S. (2012) GIS-Based Regionalized Life Cycle Assessment: How Big Is Small Enough? Methodology and Case Study of Electricity Generation. Environ. Sci. Technol 46 (15), 1096−1103.

Nansai, K., Moriguchi, Y., Suzuki, N. (2005) Site-Dependent Life-Cycle Analysis by the SAME Approach: Its Concept, Usefulness, and Application to the Calculation of Embodied Impact Intensity by Means of an Input-Output Analysis. Environ. Sci. Technol, 39 (18), 7318−7328.

Nelson, G., Rosegrant, M., Koo, J., Robertson, R., Sulser, T., Zhu, T., Ringler C., Msangi S., Palazzo A., Batka M., Magalhaes M., Valmonte-Santos R., Ewing, M., Le D. (2009). Climate

Change Impact on Agriculture and Costs of Adaptation. IFPRI. Washingon DC.

Nemecek, T., Heil, A., Huguenin, O., Meier, S., Erzinger, S., Blaser, S., Dux, D., Zimmermann, A. (2007) Life Cycle Inventories of Agricultural Production Systems. Ecoinvent report No. 15, v2.0. Agroscope FAL Reckenholz and FAT Taenikon, Swiss Centre for Life Cycle Inventories, Dübendorf

Nguyen, TLT., Gheewala, SH. (2008) Life cycle assessment of fuel ethanol from cane molasses in Thailand. Int J Life Cycle Assess, 13(4), 301-311.

Nuñez, M. (2011) Modelling location-dependent environmental impacts in life cycle assessment: water use, desertification and soil erosion. Application to energy crops grown in Spain. Doctoral Thesis. Universitat Autònoma de Barcelona i Institut de Recerca i Tecnologia Agroalimentària . (Barcelona)

Núñez, M., Pfister, S., Antón, A., Muñoz, P., Hellweg, S., Koehler, A., Rieradevall, J. (2012) Assessing the environmental impacts of water consumption by energy crops grown in Spain. J Ind Ecol. DOI: 10.1111/j.1530-9290.2011.00449.x

Nuñez, M., Civit, B., Muñoz, P., Arena, A., Rieradevall, J., Antón, A. (2010) Assessing potential desertification environmental impact in life cycle assessment. Int. J. Life Cycle Assess., 15 (1), 67–78.

Ohlsson, L. (2000) Water conflicts and social resource scarcity.PhysChem Earth, Part B: Hydrology, Oceans and Atmosphere 25 (3):213-220

Olsen, SI., Christensen, FM., Hauschild, M., Pedersen, F., Larsen, HF., Toerslov, J. (2001), Life-cycle impact assessment and risk assessment of chemicals- a methodological comparison, Environ.Impact Assess. Rev., 21, 385-404.

Okala Life Cycle Analysis Calculator." from http://www.risdpedia.net/lca/

Olsson L.Ed. (2007) Biofuels. Advances in Biochemical Engineering and Biotechnology. Springer: Berlin- Heidelberg, vol. 108

Ometto, AR., Hauschild, MZ., Roma, WNL (2009) Lifecycle assessment of fuel ethanol from sugarcane in Brazil. Int J Life Cycle Assess 14:236–247

Organization for Economic Co-operation and Development (OECD) (1996) /IUCN Guidelines for aid agencies for improved conservation and sustainable use of tropical and sub-tropical wetlands; Paris, France, OECD.

Organization for Economic Co-operation and Development (OECD) (2001). Environmental indicators for agriculture. Paris, OECD.

Organization for Economic Co-operation and Development (OECD) (2003) Descriptions of selected key generic terms used in chemical hazard/risk assessment. ENV/JM/MONO (2003). 15 OECD Series on Testing and assessment 44.

Organization for Economic Co-operation and Development (OECD) (2008), Environmental Performance of Agriculture in OECD Countries Since 1990, Paris.

Organization for Economic Co-operation and Development (OECD) (2010) Sustainable Management of Water Resources in Agriculture. ISBN 978-92-64-083455.

Owens, JW. (2001). Water resources in life-cycle impact assessment: Considerations in choosing category indicators. J. Ind. Ecol. 5(2): 37– 54.

Padgham, JEd. (2009). Agricultural Development under a changing climate: opportunities and challenges for adaptation. Joint Departmental Discussion Paper - Issue 1. World Bank. Washington.

Page, G., Ridoutt, B., Bellotti, B., (2012). Carbon and water footprint tradeoffs in fresh tomato production. Journal of Cleaner Production 32, 219-226.

Palisade, @RISK Version 5.0. In 2012.

Palme, U., Lundin M., Tillman A. M., Molander S., (2005). Sustainable development indicators for wastewater systems: Researchers and indicator users in a co-operative case study. Resour. Conserv. Recycl. 43(3): 293–311.

Panichelli, L., Dauriat, A., Gnansounou, E. (2009) Life cycle assessment of soybean-based biodiesel in Argentina for export. Int J Life Cycle Assess 14:144–159

Pankratz, T. (2004). Desalination technology trends. Bienal Report on Seawater Desalination: Volume 2. Austin, TX: TexasWater Development Board.

Paracuellos, M., (2009) How does the semiarid wetlands of the Iberian Peninsula evolve? The case of the Almería province (Spanish). Oxyura, 12, (1), 25-39.

Pasqualino, J., Meneses, M. and Castells, F. (2009). "JocAPQUA: Joc de la compra sostenible." from http://jocapqua.urv.es/.

Pasqualino, J., Meneses M., Abella M., F. Castells., (2009). LCA as a decision support tool for the environmental improvement of the operation of a municipal wastewater treatment plant. Environmental Science and Technology 43(9): 3300– 3307.

Pasqualino, J., Meneses, M. and Castells, F., (2011). Life Cycle Assessment of Urban Wastewater Reclamation and Reuse Alternatives. J. Ind. Ecol. 15(1): 49-63.

Passuello AC (2010) Development of Environmental tools for the management of sewage sludge on agricultural soils. Doctoral thesis by Universitat Rovira i Virgili.

Pathak, H., Jain, N., Bhatia, A., Patel, J., Aggarwal, P.K., (2010). Carbon footprints of Indian food items. Agricult, Ecosystems and Environ 139, 66-73.

Paustian, K., Cole, CV., Sauerbeck, D. and Sampson, N. (1998). CO_2 mitigation by agriculture: An overview. Clim. Change 40:135–162.

Payet, Y. (2004). Assessing toxic impacts on aquatic ecosystems in Life Cycle Assessment (LCA). Doctoral Thesis. École Polytechnique Fédérale de Lausanne.

PE (2011) GaBi software and database. Available at:http://www.gabi-software.com

Pedersen, B. (1993), Environmental Assessment of Products, UETP-EEE, IBSN951-9110-836.

Penning de Vries FWT, Acquay H, Molden D, Scherr SJ, Valentin C, Cofie O (2003) Integrated land and water management for food and environmental security – Comprehensive assessment of water management in agriculture. Research Report 1, Comprehensive Assessment Secretariat, Colombo, Sri Lanka.

Pennington, DW., Margni, M., Payet, J., Jolliet, O. (2006): Risk and Regulatory Hazard-Based Toxicological Effect Indicators in Life-Cycle Assessment (LCA), Human and Ecological Risk Assessment: An Inter J, 12:3, 450-475.

Pennington, DW., Potting, J., Finnveden, G. (2004a). Life cycle assessment Part 2: Current impact assessment practice. Environ Internat 30:721–39.

Pennington, DW., Payet, J., Hauschild, M. (2004b). Aquatic ecotoxicological measures in life cycle assessment (LCA). Environ Toxicol Chem 23(7):1796-1807.

Pereira, CLF., Ortega, E. (2010) Sustainability assessment of large-scale ethanol production from sugarcane. J of Clean. Prod, 18(1), 77-82.

Pérez, D., Fandos, C., Scandaliaris, J., Mazzone, L., Soria, F., Scandaliaris, P. (2007) Estación Experimental Agroindustrial Obispo Colombres (EEAOC) Technical Report 34. Current state and evolution of the productivity of sugarcane cultivation in Tucuman and Argentinean Northwest in 1990-2007.

Peters, G., Wiedemann, S., Rowley, H., Tucker, R. (2010) Accounting for water use in Australian red meat production. Int J LCA 15 (3):311-320.

Pfister, S., Koehler, A., Hellweg, S. (2009) Assessing the Environmental Impacts of Freshwater Consumption in LCA. Environ Sci Technol 43 (11):4098-4104.

Pfister, S., Bayer, P., Koehler, A., Hellweg, S. (2011) Environmental Impacts of Water Use in Global Crop Production: Hotspots and Trade

Pidd, M., (1996). Five simple principles of modelling, The 1996 Winter Simulation Conference, Coronado, California, USA.

Point, E., Tyedmers, P., Naugler, C., (2012). Life cycle environmental impacts of wine production and consumption in Nova Scotia, Canada. Journal of Cleaner Production 27, 11-20.

Posthuma, L., Suter, II. Traas, TP. (2002). Species Sensitivity Distributions in Ecotoxicology. Lewis Publishers, Boca Raton, FL, USA

Potting, J., Hauschild, M. (2006) Spatial Differentiation in Life Cycle Impact Assessment: A decade of method development to increase the environmental realism of LCIA. Int. J. Life Cycle Assess 11 (1), 11−13.

Pulido, A., Morales, G., Benavente, J., (1988) Hidrogeology of the Adra River delta (spanish). Estudios Geológicos, 44, 429-443.

Quantis (2011) Quantis Water Database - Technical report. Lausanne, Switzerland. Available at: http://www.quantis-intl.com/waterdatabase.php

R., S. Flora de Almería. Plantas vasculares de la provincia.; 1987.

Raluy, G., Serra, L., Uche, J., (2006). Life cycle assessment of MSF, MED and RO desalination technologies. Energy 31 (13), 2025-2036.

Raluy, R. G., Serra, L., Uche, J., Valero A., (2004). Life cycle assessment of desalination

technologies integrated with energy production systems. Desalination 167, 445–458.
Ramírez O., (2009). L'aigua a Tarragona, 25 anys d'EMATSA. Editorial SILVA. Tarragona. ISBN: 978-84-92465-17-0.

Raskin, .P, Gleick, P., Kirshen, P., Pontius, G., Strzepek, K. (1997) Water Futures: Assessment of Long-range Patterns and Prospects. Stockholm, Sweden.

Reap, J., Roman, F., Duncan, S., Bras, B. (2008) A survey of unresolved problems in life cycle assessment. Part 2. Impact assessment and interpretation. Int. J. Life Cycle Assess. 13:374–388

Rebitzer, G., Ekvall, T., Frischknecht, RD., et al. (2004) Life cycle assessment: Goal & scope definition, inventory analysis, and applications (Part 1). Environ Internat 30:701–20.

REE (Red Eléctrica Española), (2008). El sistema eléctrico español 2008. www.ree.es/sistema_electrico/informeSEE-2008.asp. Accessed July 2012.

Refsgaard, JC., van der Sluijs, JP., Højberg, AL., Vanrolleghem, PA. (2007) Uncertainty in the environmental modelling process – A framework and guidance. Environ. Model. & Softw., 22, 1543-1556.

Renou, S., Thomas, J. S., Aoustin, E. and Pons, M. N., (2008). Influence of impact assessment methods in wastewater treatment LCA. J. Clean. Prod. 16, 1098-1105.

Renouf , MA., Pagan, RJ., Wegener, MK. (2011) Life cycle assessment of Australian sugarcane products with a focus on cane processing. Int J Life Cycle Assess 16:125–137.

Renouf, MA., Wegener, MK., Nielsen, LK. (2008) An environmental life cycle assessment comparing Australian sugarcane with US corn and UK sugar beet as producers of sugars for fermentation. Biomass Bioenergy 32:1144–1155.

Ridoutt, BG., Eady, SJ., Sellahewa, J., Simons, L., Bektash, R., (2009). Water footprinting at the product brand level: case study and future challenges. Journal of Cleaner Production 17, 1228.

Ridoutt B (2011) Development and application of a water footprint metric for agricultural products and food industry in: Finkbeiner, M. Towards life cycle sustainability management. Springer, Berlin, Germany

Ridoutt, BG., Pfister, S. (2010) A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. Glob Environ Ch 20(1):113-120.

Riggs, JL. (1968). Economic decision models for engineers and managers. New York: McGraw

Rijsberman, FR. (2006) Water scarcity: fact or fiction? Agric Water Manag 80 (1–3): 5–22.

Ritz, C., (2010) Toward a unified approach to dose–response modeling in ecotoxicology. Environ. Tox. and Chem., 29, (1), 220-229.

Ritz, C., Streibig, J. (2005) Bioassay analysis using R. J Stat Softw, 12, 1–22.

Rodríguez-Rodríguez, M., (2007) Hydrogeology of ponds, pools, and playa-lakes of southern Spain. Wetlands, 27, (4), 819-830.

Rodríguez-Rodríguez, M.; Benavente, J.; Alcalá, F. J.; Paracuellos, M., (2011) Long-term water monitoring in two Mediterranean lagoons as an indicator of land-use changes and intense precipitation events (Adra, Southeastern Spain). Estuarine Coastal and Shelf Sci. 91, (3), 400-410.

Rodríguez-Rodríguez, M.; Benavente, J.; Moral, F., (2004) Hydrological regime and physical limnology in the lagoons of Adra (Almería, Spain) (spanish). In: VIII Simposio de Hidrogeología, Zaragoza (Spain) pp 409-419.

Röling, N. (1993). Agricultural knowledge and environmental regulation: The crop protection plan and the Koekoekspolder. Sociologia Ruralis, 33 (2), 212-231.

Röling, N. (1994a). Platforms for decision making about ecosystems. In L. Fresco (Ed.), The future of the land. Chichester: John Wiley and Sons.
Röling, N. (1994b). Creating human platforms to manage natural resources: First results of a research programme.In Proceedings of the International Symposium on Systems Oriented Research in Agriculture and Rural Development (p. 391-395). Montpellier, France, November21.25.1994

Röös, E., Sundberg, C., Hansson, PA., (2010). Uncertainties in the carbon footprint of food products: A case study on table potatoes. Int J of Life Cycle Assess 15, 478-488.

Rosenbaum, R.; Bachmann, T.; Gold, L.; Huijbregts, M. J.; Jolliet, O.; Juraske, R.; Koehler, A.; Larsen, H.; MacLeod, M.; Margni, M.; McKone, T.; Payet, J.; Schuhmacher, M.; Meent, D.; Hauschild, M., (2008) USEtox—the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. Int J of Life Cycle Assess, 13, (7), 532-546.

Rovira, J., Nadal, M., Domingo, J.L., Tanaku, T., Suciu, N.A., Trevisan, M., Capri, E., Seguí, X., Darbra, R.M., Schuhmacher, M., (2012). A Revision of Current Models for Environmental and Human Health Impact and Risk Assessment for Application to Emerging Chemicals. Springer Berlin Heidelberg, pp. 1-18.

Rubel, F., and M. Kottek, (2010): Observed and projected climate shifts 1901-2100 depicted by world maps of the Köppen-Geiger climate classification. Meteorol. Z., 19, 135-141. DOI: 10.1127/0941-2948/2010/0430.

Rural Industries Research and Development Corporation (RIRDC, Australian Government) A Literature Review ofLife Cycle Assessmentin Agriculture (2009) ISBN 1 74151 833 4.RIRDC Publication No. 09/029

Saad, R., Margni, M., Koellner, T., Wittstock, B., Deschênes, L. (2011) Assessment of land use impacts on soil ecological functions: development of spatially differentiated characterization factors within a Canadian context. Inter J of Life Cycle Assess 16 (3): 198-211.

Sahinidis, NV (2004) Optimization under uncertainty: state-of-the-art and opportunities. 28: 971–983

Schuytema, GS., Nebeker, AV, Stutzman, TW. (1997) Salinity Tolerance of Daphnia magna and Potential Use for Estuarine Sediment Toxicity Tests. Archives of Environ Contamination and Toxicol, 33, (2), 194-198.

Swiss Centre for Life-Cycle Inventories, 2009. Ecoinvent V2.1. database. www.ecoinvent.org, Dübendorf, Switzerland: SCLCI.

Seabra, JEA., Macedo, IC., Chum, HL., Faroni, CE., Sarto, CA. (2011) Life cycle assessment of Brazilian sugarcane products: GHG emissions and energy use. Biofuels, Bioprod. and Biorefin., 5(5), 519-532.

Seckler, D., Amarasinghe, U., Molden, DJ., de Silva, R., Barker, R. (1998) World Water Demand and Supply, 1990 to 2025: Scenarios and Issues. IWMI Research Report vol 19. IWMI, Colombo, Sri Lanka

Seckler, D., Amarasinghe, U., Molden, DJ., de Silva, R., Barker, R. (1998) World Water Demand and Supply, 1990 to 2025: Scenarios and Issues. IWMI Research Report vol 19. IWMI, Colombo, Sri Lanka.

Seckler, D., Molden, D. Sakthivadivel, R. (2003). The concept of efficiency in waterresources management and policy. In: Kijne et al. eds.. Water Productivity in Agriculture: Limits and Opportunities for Improvement. CABI Publishing, Wallingford, UK. pp37–51.

Secretaría de Energía de la Nación. Informe del Sector Eléctrico (2009) (Part 1). http://energia3.mecon.gov.ar/contenidos/verpagina.php?idpagina=3368.

Segal, R., MacMillan, T. (2009). "Water labels on food: Issues and recommendations. Food Ethics Council, Brighton, United Kingdom.

Society of Environmental Toxicology and Chemistry (SETAC) (1993) Guidelines for Life-Cycle Assessment: a 'Code of Practice', Sesimbra/ Portugal SETAC Workshop report, SETAC Press, Pensacola, FL. SETAC-Europe 5:4–6

SIA, Integrated Water Information database. Available on-line in Spanish http://servicios2.marm.es/sia/visualizacion/descargas/mapas.jsp.

Siebert, S., Döll, P., Feick, S., Frenken, K., Hoogeveen, J. (2007). Global map of irrigation areas version 4.0.1. University of Frankfurt (Main), Germany, and FAO, Rome, Italy.

Siget, K., Klauer, B., Pahl-Wostl, C. (2010). Conceptualizing uncertainty in environmental decision-making: the example of the EU water framework directive. Ecol. Econ. 69: 502-510.

SIGPAC, Geographical Information Systems (GIS) of farms. Available on-line in Spanish http://sigpac.mapa.es/fega/visor/.

Skidmore, AK. (2002) Environmental Modelling with GIS and Remote Sensing. Edited by Andrew Skidmore. CRC Press. Pages 1-7. ISBN: 978-0-415-24170-0.

Slinger, D., Tension, K. (2005) Salinity Glove Box Guide for NSW Murray & Murrumbidgee. In Industries, N. D. o. P., Ed.

Smakhtin, V., Revenga, C., Döll, P. (2004) Taking into Account Environmental Water Requirements in Global-scale Water Resources Assessments. Comprehensive assessment of water management in agriculture Research report 2. International Water Management Institute (IWMI) Colombo, Sri Lanka. ISBN 92-9090-542-5.
http://www.iwmi.cgiar.org/Assessment/files/pdf/publications/ResearchReports/CARR2.pdf. Accessed 30 April 2012.

Smith, M., (1988). Manual for CROPWAT version 5.2. FAO, Rome. 45pp.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P.e.a., (2007). Agriculture. In:

Climate change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Sombekke, HDM. , Voorhoeve, DK. , Hiemstra, P. (1997). Environmental impact assessment of groundwater treatment with nanofiltration, Desalination 113:293–296.

Sonneman, G., Castells, F., Schuhmacher, M. (2004) Integrated Life-Cycle and Risk Assessment for industrial processes. CRC Press LLC Lewis publishers ISBN 1-56670-644-0

Sostaqua project, Evaluación de la sostenibilidad del ciclo del agua en España (TT07001C). (2008-2010). A Project funded by Spanish Ministry of Science and Technology (Spain).

Starkl, M., Brunner, N., Grasser, U., Moog, O., Stagl, S., Kärman, E., Wimmer, J., Szewieczek, R., and Haberl, R., (2005). Analysis and evaluation of methodologies to assess technical urban water systems. Water Sci. and Technol. 52 (9):43-51.

Stirzaker, RJ. (2011). Strategy, tactics and heuristics for managing solutes in horticultural crops. Proc. 6th IS on Irrigation of Horticultural crops. Ed.: S. Ortega-Farias and G. Selles. Acta Hort. 889, ISHS 2011.

Stoessel, F., Juraske, R., Pfister, S., Hellweg, S. (2012) Life Cycle Inventory and Carbon and Water FoodPrint of Fruits and Vegetables: Application to a Swiss Retailer. Environ Sci & Technol, 46, (6), 3253-3262.

Stokes, J., Horvath, A., (2006). Life cycle energy assessment of alternative water supply systems. Int J Life Cycle Ass 11: 335–343.

Striving Towards Best Available Practice. SETAC Press, Pensacola, FL, USA

Struijs, J., De Zwart, D., Posthuma, L., Leuven, RSE., Huijbregts, MAJ. (2011) Field sensitivity distribution of macroinvertebrates for phosphorus in inland waters. Integrated Environ. Assess. and Manag., 7, (2), 280-286.

Suh, YJ, Rousseaux, P. (2001). An LCA of alternative wastewater sludge treatment scenarios. Resour. Conserv. and Technol. 35 (2002):191-200.

Sullivan, C., Meigh, J., Giacomello, A. (2003) The Water Poverty Index: Development and application at the community scale. Nat Resour Forum 27 (3):189-199

Swiss Centre for Life-Cycle Inventories (2010). Ecoinvent database V2.2, Dübendorf,

Switzerland. http:// www.ecoinvent.org.
The Ramsar Conventional on Wetlands http://www.ramsar.org/

Tidaker, P., Kärrman, E., Baky, A., Jönsson, H., (2006). Wastewater management integrated with farming -an environmental systems analysis of a Swedish country town. Resour. Conserv. Recycl. 47: 295–315.

Tolón, A., Lastra, X., (2010) La agricultura intensiva del poniente almeriense. Diagnóstico e instrumentos de gestión ambiental. M+A. Revista Electrónic@ de Medio Ambiente, 8, 18-40.

Tomei, J., Upham, P. (2009) Argentinean soy-based biodiesel: An introduction to production and impacts. Energ Policy 37: 3890–3898.

Torrellas, M., Antón, A. Montero, JI. (2013) An environmental impact calculator for greenhouse production systems. Jl of Environ. Manag. (in press)

Tubiello, F., Soussana, JF, Howden, SM., (2007). Crop and pasture response to climate change. Proc Natl Acad Sci USA 104, 19686-19690.

U.S. Environmental Protection Agency (1996) Report on Revisions to 5th Edition AP-42, Section 1.8, Bagasse Combustion in Sugar Mills. Washington, DC.

UNEP DTIE (Division of Technology, Industry and Economics) (1999), Towards the global use of life- cycle assessment, technical report, United Nations Environment Program, Paris.

UNESCO-WWAP, 2009. The United Nations World Water Development Report 3: Water in a Changing World. The United Nations Educational, Scientific and Cultural Organisation, Paris.

Unilever, (2010). Cool farm tool calculator.
www.growingforthefuture.com/content/Cool+Farm+Tool

United Nations (2006) Coping with water scarcity. A strategic issue and priority for system-wide action.UN-Water Thematic Initiatives, Rome, Italy.

United Nations Framework Convention on Climate Change (1997) Kyoto Protocol. http://unfccc.int/kyoto_protocol/items/2830.php

UNEP DTIE (Division of Technology, Industry and Economics) (1999), Towards the global use of life- cycle assessment, technical report, United Nations Environment Program, Paris.

Van de Meent, D., Klepper, O. (1997) Mapping the potential affected fraction (PAF) of species as an indicator of generic toxic stress, RIVM report number:607504001.

Van de Meent, D., Huijbregts, MAJ., (2005) Calculating life-cycle assessment effect factors from potentially affected fraction-based ecotoxicological response functions. Environ Toxicol and Chem, 24, (6), 1573-1578.

Van der Sluijs, J., Craye, M., Funtowicz, S., Kloprogge, P., Ravetz, J., Risbey, J. (2005). Combining quantitative and qualitative measures of uncertainty in model-based environmental assessment: the NUSAP system. Risk Anal. 25 (2): 481-492.

Van Wijk, RJ, Van Goor, EMJ., Verkley, JAC., (1988) Ecological studies on Potamogeton pectinatus L. II. Autecological characteristics, with emphasis on salt tolerance, intraspecific variation and isoenzyme patterns. Aquat Botany, 32, (3), 239-260.

Van Zelm, R., Schipper, AM., Rombouts, M., Snepvangers, J., Huijbregts, MAJ. (2011) Implementing Groundwater Extraction in Life Cycle Assessment: Characterization Factors Based on Plant Species Richness for the Netherlands. Environ Sci Technol 45(2):629-635

Van Zelm, R., Huijbregts, M. (2007) Time horizon dependent characterization factors for acidification in life-cycle assessment based on forest plant species occurrence in Europe. Environ. Sci. Technol, 41 (3), 922–927.

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

Verbrugge, LH., Schipper, A., Huijbregts, MJ., Van der Velde, G., Leuven, REW. (2012) Sensitivity of native and non-native mollusc species to changing river water temperature and salinity. Biol Invasions, 14, (6), 1187-1199.

Verones, F., Bartl, K., Pfister, S., Jimenez, R., Hellweg, S. (2012) Modeling the local biodiversity impacts of agricultural water use: case study of a wetland in the coastal arid area of Peru. Environ Sci Technol DOI: 10.1021/es204155g

Verones, F., Hanafiah, MM., Pfister, S., Huijbregts, MAJ., Pelletier, GJ., Koehler, A. (2010) Characterization Factors for Thermal Pollution in Freshwater Aquatic Environments. Environ Sci & Technol, 44, (24), 9364-9369.

Vigon, BW., Tolle, DA., Corneby, BW, Lotham, HC, Harrison, CL, Boguski, TL, Hunt, RG, Sellers, JD (1993), Life cycle assessment: inventory guidelines and principles, conducted by

Battelle and Franklin Associates for the EPA, Ltd. Office of Research and Development, EPA/600/R-92/245.

Vince F (2007) UNEP/SETAC Life Cycle Initiative Working group: Assessment of water use and consumption within LCA. Veolia Environnement, Paris, France

Vince, F., Aoustin, E., Bréant P. and Marechal F. (2008). LCA tool for the environmental evaluation of potable water production. Desalination 220(1-3): 37–56.

Vörösmarty, C. J, McIntyre, PB., Gessner, MO., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, SE., Sullivan, (2010) Global threats to human water security and river biodiversity. Nature, 467: 555-561.

Water Footprint Network (2011) WaterStat.Enschede, the Netherlands.

Walker, W., Harremoes, P., Rotmans, J., Van der Sluijs, J., Van Asselt, M., Janssen, P., Krayer, von Krauss, M. (2003). Defining uncertainty. A conceptual basis for uncertainty management in model-based decision support. Integrated Assess. 4 (1), 5-17.

Wang, JQ., Lui, H., Po, H., Fan, L. (1997) Influence of salinity on food consumption, growth and energy conversion efficiency of common carp (Cyprinus carpio) fingerlings. Aquaculture, 148, (2–3), 115-124.

Warmink, JJ., Janssen, JAEB., Booij, MJ., Krol, MS. (2010). Identification and classification of uncertainties in the application of environmental models. Environ Model Soft 25: 1518-1527.

Waste.ideal.es (Magazine on line) http://waste.ideal.es/albuferadeadra.htm

Water Footprint Network Homepage (2011) WaterStat. Enschede, the Neteherlands. Available online: http://www.waterfootprint.org.

WBCSD (2010) The global water tool. Accessible at:www.wbcsd.org. Geneva, Switzerland

Wegener, A., Heijungs, R. (2010) GLOBOX: A spatially differentiated global fate, intake and effect model for toxicity assessment in LCA. Sci. Total Environ., 408 (14), 2817−2832.

Weidema, BP (1998) Multi-User Test of the Data Quality Matrix for Product Life Cycle Inventory. Int J. of Life Cycle Assess. 3(5):259-265.

Weidema, BP., Wesnaes, MS. (1996), Data quality management for life cycle inventories-an example of using data quality indicators. J. Clean. Prod. 4: 167–174.

Wiecek, MM, Ehrgott, M, Fadel ,G., Figueira, JR (2008). Multiple criteria decision making for engineering. Omega-International J Of Manag Sci 36, no. 3: 337-339.

Williams, AG., Audsley, E., Sandars, DL. (2006) "Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities." Main Report. Defra Research Project IS0205. Bedford: Cranfield University and Defra.

World Bank. (2009). Water and Climate Change: Impacts on groundwater resources and adaptation options. Water Unit Energy, Transport, and Water Department. Washington DC. 98pp.

World Resources Institute (WRI) and the World Business Council for Sustainable Development (WBCSD) (2001) The Greenhouse Gas Protocol: a corporate accounting and reporting standard. http://www.ghgprotocol.org/

World Resources Institute (WRI) and the World Business Council for Sustainable Development (WBCSD) (2011 a) Greenhouse Gas Protocol Corporate Value Chain (Scope 3) Accounting and Reporting Standard. http://www.ghgprotocol.org/

World Resources Institute (WRI) (2012) The Greenhouse Gas Protocol: Agriculture guidance. http://www.ghgprotocol.org/