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**Toward a consistent accounting of water as a resource  
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products: Methodological development and application  
to a perennial cropping system**

Sandra Payen

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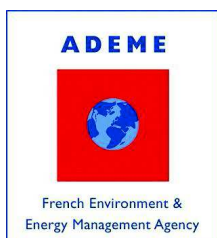
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Spécialité : **Science de la Terre et de l'Eau**

Présentée par **Sandra PAYEN**



Toward a consistent accounting of water as a  
resource and a vector of pollution in the LCA of  
agricultural products:  
Methodological development and application to  
a perennial cropping system

Directeur de thèse : Mr Sylvain PERRET, HDR, Cirad

Soutenue le 16 Décembre 2015 devant le jury composé de

Mme Cécile BULLE, Professeure, CIRAIG

Mr Patrick DURAND, Directeur de recherche, INRA

Mme Claudine BASSET-MENS, HDR, Cirad

Mr Christian GARY, Directeur de recherche, INRA

Mr Mustapha ZEMZAMI, PhD, Les Domaines Agricoles

Mr Vincent COLOMB, Ingénieur, ADEME

Rapportrice

Rapporteur

Examinatrice

Examinateur

Invité

Invité







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---

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## Table des matières

---

INTRODUCTION .....	2
<b>WHY DOING LIFE CYCLE ASSESSMENT OF AGRICULTURAL SYSTEMS?</b>	
1.1. FEEDING THE PLANET WITHOUT DESTROYING IT .....	2
1.1.1. Agriculture is feeding the planet, but has many impacts on the environment .....	2
1.1.2. An increasing pressure.....	2
1.1.3. Identify the environmental hot spots and mitigation options.....	3
1.2. LCA OF AGRICULTURAL SYSTEMS: CHALLENGES .....	3
1.2.1. LCA methodology.....	3
1.2.2. The cause and effect chain or environmental “pathway” .....	5
1.2.3. Why applying LCA to agricultural systems is relevant? .....	6
1.2.4. The inventory: a crucial LCA stage for agricultural systems .....	7
1.2.5. One limitation of LCA relates to the modelling of freshwater use impacts.....	7
1.3.2. The importance of scales.....	8
1.4. CONCLUSION .....	9
REFERENCES INTRODUCTION .....	9
CHAPTER 1.....	12
<b>HOW TO ASSESS THE IMPACTS ASSOCIATED WITH WATER USE IN AGRICULTURAL LCA?</b>	
1.1. WATER FOOTPRINTS TERMINOLOGIES .....	12
1.2. AN OVERVIEW OF THE DIFFERENT METHODS.....	13
1.3. INVENTORY SCHEMES: WATER QUANTITY, QUALITY AND SOURCE .....	15
1.3.1. The inventory scheme depends on the impact assessment method.....	15
1.3.2. The water sources must be distinguished as they might face different scarcities/availabilities .....	16
1.3.3. Water quality has to be inventoried as quality degradation may contribute to water deprivation .....	17
1.4. MIDPOINT IMPACT ASSESSMENT: WATER SCARCITY OR AVAILABILITY? .....	17
1.4.1. Water indices are used as characterisation factors .....	17
1.4.2. Water scarcity indicators vs. water availability indicators .....	18
1.4.3. Open questions on characterisation factors.....	18
1.4.4. Spatial and temporal scales: consistency with the goal and scope of the study .....	19
1.4.5. Water indices versus fate and effect modelling .....	19
1.5. ENDPOINT IMPACT ASSESSMENT: GAPS AND OVERLAPPING .....	20
1.6. OPERATIONALIZATION .....	23
1.7. WATER IS A RESOURCE, BUT ALSO A VECTOR OF POLLUTANTS, NUTRIENTS AND SALTS.....	24
1.8. THESIS SPECIFIC OBJECTIVES .....	25
REFERENCES CHAPTER 1.....	27



CHAPTER 2.....	34
----------------	----

## SALINISATION IMPACTS IN LIFE CYCLE ASSESSMENT: A REVIEW OF CHALLENGES AND OPTIONS TOWARDS THEIR CONSISTENT INTEGRATION

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ABSTRACT .....	34
2.1. INTRODUCTION.....	35
2.2. SALINISATION ENVIRONMENTAL MECHANISMS.....	36
2.2.1. Salinity.....	36
2.2.2. Human interventions causing soil and water salinisation .....	37
2.2.3. Water and soil salinisation damages to Ecosystems, Human health and Resources.....	40
2.2.4. Complexities related with salinisation in space and time.....	40
2.3. CRITICAL ANALYSIS OF SALINISATION IMPACT ASSESSMENT METHODS IN LCA.....	42
2.3.1. Salinisation associated with irrigation: Feitz and Lundie (2002).....	42
2.3.2. Salinisation associated with overuse of a water body: Amores et al. (2013) .....	43
2.3.3. Salinisation associated with brine disposal: Zhou et al. (2013b) .....	44
2.3.4. Salinisation associated with salt release: Leske and Buckley (2003; 2004a; 2004b).....	44
2.3.5. Lack of consistent frameworks .....	45
2.4. TOWARDS A CONSISTENT FRAMEWORK FOR SALINISATION IMPACTS ASSESSMENT IN LCA: METHODOLOGICAL ISSUES AND RECOMMENDATIONS .....	47
2.4.1. Context of LCIA for assessing salinisation impacts.....	47
2.4.2. Modelling options for the different salinisation types.....	47
2.4.3. Toward operationalisation .....	59
2.5. CONCLUSION .....	55
ACKNOWLEDGMENTS .....	56
REFERENCES CHAPTER 2.....	57

CHAPTER 3.....	66
----------------	----

## INVENTORY OF FIELD WATER FLOWS FOR AGRI-FOOD LCA: CRITICAL REVIEW AND RECOMMENDATIONS OF MODELLING OPTIONS

ABSTRACT .....	66
3.1. INTRODUCTION.....	67
3.2. CRITICAL ANALYSIS OF WATER INVENTORY DATABASES .....	68
3.2.1. Water inventory and agri-food LCA databases.....	68
3.2.2. Limitations of water inventory and agri-food LCA databases.....	74
3.3. MODELLING OPTIONS FOR FIELD WATER FLOWS INVENTORY IN AGRICULTURAL LCA STUDIES .....	77

3.3.1. Model specifications description .....	77
3.3.2. Modelling approach selection .....	81
3.3.3. Further developments of models, tools and databases .....	85
3.4. CONCLUSION .....	86
ACRONYM LIST.....	89
REFERENCES CHAPTER 3.....	89
 CHAPTER 4.....	 98

## E.T.: AN OPERATIONAL FIELD WATER AND SALT FLOWS MODEL FOR AGRICULTURAL LCA ILLUSTRATED ON CITRUS

ABSTRACT .....	98
4.1. INTRODUCTION.....	99
4.2. MATERIAL AND METHOD: FIELD WATER AND SALT FLOWS MODEL PRESENTATION.....	100
4.2.1. Model specifications and general principles .....	100
4.2.2. E.T. Model description .....	101
4.2.3. Validity domain of E.T. ....	108
4.2.4. Model testing.....	109
4.3. RESULTS AND DISCUSSION: MODEL TESTING .....	115
4.3.1. Comparison of E.T. outputs to other model formalisms .....	115
4.3.2. Sensitivity of E.T. model outputs to parameters' variations.....	118
4.3.3. Testing of E.T. model for different scenarios of practice.....	118
4.3.4. Model with degraded data .....	119
4.3.5. Model limitations and improvement perspectives .....	121
4.3.6. Model usage recommendations .....	123
4.3.7. Model outputs comparison with databases .....	123
4.3.8. Model usage within a LCA study .....	125
4.4. CONCLUSION .....	126
ACKNOWLEDGMENTS .....	127
REFERENCES CHAPTER 4.....	127

 CHAPTER 5.....	 132
--------------------	---------

## LIFE CYCLE ASSESSMENT OF A PERENNIAL CROP INCLUDING AN IN-DEPTH ASSESSMENT OF WATER USE IMPACTS: THE CASE OF A MANDARIN IN MOROCCO

ABSTRACT .....	132
5.1. INTRODUCTION.....	133
5.2. MATERIALS AND METHODS.....	133
5.2.1. Geographical context .....	133

5.2.2. LCA goal and scope .....	134
5.2.3. Inventory of Moroccan mandarin production: from cradle-to-farm-gate.....	135
5.2.4. Life cycle impact and damage assessment .....	139
5.2.5. COMPARISON WITH PUBLISHED LCA STUDIES ON CITRUS.....	142
5.3. RESULTS .....	142
5.3.1. Market gate - midpoint .....	142
5.3.2. Farm gate - midpoint.....	144
5.3.3. Farm gate - endpoint.....	147
5.4. DISCUSSION.....	148
5.4.1. Comparison with published references on citrus .....	148
5.4.2. Water - energy nexus .....	149
5.4.3. Water use impacts .....	150
5.4.4. Perspectives.....	154
5.5. CONCLUSION .....	157
ACKNOWLEDGEMENTS.....	158
REFERENCES CHAPTER 5.....	158
DISCUSSION AND PERSPECTIVES ( <i>IN FRENCH</i> ).....	164
COMMENT MIEUX EVALUER LES IMPACTS ASSOCIES AUX FLUX D'EAU ET DE SELS ? .....	164
COMMENT REALISER UN INVENTAIRE PERTINENT DES FLUX D'EAU ET DE SELS MOBILISES DANS LES SYSTEMES AGRICOLES ?.....	167
EST-IL POSSIBLE D'APPLIQUER LE MODELE D'INVENTAIRE DES FLUX D'EAU ET LES INDICATEURS ASSOCIES POUR EVALUER DES PRATIQUES AGRICOLES ? .....	169
GENERAL CONCLUSION ( <i>IN FRENCH</i> ) .....	177
PUBLICATIONS LIST .....	180
PEER-REVIEWED PUBLICATIONS .....	180
PRESENTATIONS IN CONFERENCES.....	180
PARTICIPATION IN BOOKS AND REPORTS .....	180
ANNEXES .....	181
LCA OF LOCAL AND IMPORTED TOMATO: AN ENERGY AND WATER TRADE-OFF	
<i>Journal of Cleaner Production - DOI: 10.1016/j.jclepro.2014.10.007</i>	
ABSTRACT .....	184
1. INTRODUCTION .....	185
2. MATERIALS AND METHODS.....	186
2.1. Geographical context.....	186

2.2. LCA goal and scope .....	187
2.3. Inventory of Moroccan tomato production: from cradle-to-farm-gate .....	187
2.4. Life cycle impact and damage assessment.....	189
2.5. LCA comparison of Moroccan and French off-season tomato production.....	190
3. RESULTS AND DISCUSSION .....	190
3.1. Environmental impacts of the Moroccan off-season tomato production and delivery .....	190
3.2. LCA comparison of imported Moroccan and local French production systems .....	192
3.3. The need for a reliable inventory for accurately modelling the impacts of freshwater use.....	194
4. CONCLUSION .....	196
ACKNOWLEDGEMENTS.....	196
REFERENCES.....	197
TABLES AND FIGURES .....	201
LCA OF LOCAL AND IMPORTED TOMATO: AN ENERGY AND WATER TRADE-OFF - SUPPLEMENTARY INFORMATION .....	207

CHAPTER 1 - SUPPLEMENTARY INFORMATION - HOW TO ASSESS THE IMPACTS ASSOCIATED WITH WATER USE IN AGRICULTURAL LCA? .....	212
--	-----

CHAPTER 2 - SUPPLEMENTARY INFORMATION - SALINISATION IMPACTS IN LIFE CYCLE ASSESSMENT: A REVIEW OF CHALLENGES AND OPTIONS TOWARDS THEIR CONSISTENT INTEGRATION .....	224
--	-----

CHAPTER 3 - SUPPLEMENTARY INFORMATION - INVENTORY OF FIELD WATER FLOWS FOR AGRI-FOOD LCA: CRITICAL REVIEW AND RECOMMENDATIONS OF MODELLING OPTIONS ..	232
---	-----

CHAPTER 4 - SUPPLEMENTARY INFORMATION - E.T.: AN OPERATIONAL FIELD WATER AND SALT FLOWS MODEL FOR AGRICULTURAL LCA ILLUSTRATED ON CITRUS .....	238
--	-----

CHAPTER 5 - SUPPLEMENTARY INFORMATION - LIFE CYCLE ASSESSMENT OF A PERENNIAL CROP WITH IN-DEPTH ANALYSIS OF WATER USE IMPACTS: THE CASE OF A MANDARIN IN MOROCCO.....	243
---	-----

GLOSSARY.....	248
---------------	-----

ACRONYM LIST .....	251
--------------------	-----

## Table des figures

---

Figure 1. Schematic overview of risks associated with main agricultural production systems. (FAO Land & water (2011) - the state of the world's land and water resources for food and agriculture).....	3
Figure 2. LCA: A global method. On this figure, water footprint refers to the volumetric blue, green and grey waters (Source: P. Roux - IRSTEA).....	4
Figure 3. Four steps of the LCA methodology with simplified substances inventory and impact categories: 1. the goal and scope definition, 2. the inventory analysis, 3.the environmental impact assessment, and 4. the interpretation that should be performed at each of the three previous steps. Arrows between the different steps show that LCA is an iterative process.....	5
Figure 4. Flow diagram of the cause and effect chain for aquatic eutrophication (ReCiPe method): from human interventions to the midpoint impacts, and finally the endpoint damages on Ecosystem quality (based on ILCD Handbook (2011)).....	6
Figure 5. Technosphere (agricultural system under study) and Ecosphere (environment receiving the emissions) boundary depend on the soil status (dotted line).....	7
Figure 6. From the inventory to midpoints and damages related to water in LCA. At the inventory level, processes can use water and/or pollute water through the emission of substances. Impacts related to water use include the impact on the water resource and water pollution. Ultimately, these impacts damage the three areas of protection.....	8
Figure 1.1. Water use impact modelling framework in LCA: 1. Product life cycle modelling including many processes using and/or polluting water, 2. Inventory of water flows for each process in terms of volume and quality (inventory flows requirement depends on the method), 3. Environmental impact assessment: water footprint profile includes water deprivation impacts (based on scarcity or availability indicator), but also acidification, eutrophication, ecotoxicity and other impacts related to water degradation. Hatched boxes represent impacts not addressed in this review. ....	13
Figure 1.2. An overview of methods addressing water use in LCA with classification for the three areas of protection. At midpoint, scarcity indicators (addressing only water quantity) are differentiated from availability indicators (addressing both water quantity and quality). *Hoekstra et al. (2012): not developed specifically for LCA but compatible.....	14
Figure 1.3. Inventory schemes and characterization factors for Pfister et al. (2009, 2014) as water scarcity indicator, and Boulay et al. (2011) as water availability indicator.....	16
Figure 1.4. Cause-effect chains from the inventory to the areas of protection of human health, ecosystem quality, and resources (adapted from Bayart et al. 2010 and Kounina et al. 2013). The pathways considering water quality degradation are emphasised in pink colour. One potential missing cause-effect chain from the framework described by Kounina and colleagues (2013) is the loss of water quality damaging the AoP Resource.....	22
Figure 2.1. Human-driven salinisation environmental mechanisms and positioning of approaches proposed in the literature. Long dash lines represent controversial pathways in the scientific community.....	41
Figure 2.2. Soil salinisation impacts on Human Health, Ecosystems and Resource: fate and effect factors positioning on the cause-effect chain and relations between agricultural and non-agricultural lands.....	51
Figure 2.3. Salinisation associated with irrigation and deposition of salts: technosphere and ecosphere boundaries options and corresponding parameters to account for in the inventory or in the impact assessment.....	52

Figure 2.4. Water salinisation impacts on Human Health, Ecosystems and Resource: fate and effect factors positioning on the cause-effect chain and Boulay et al. inventory and damage to human health methods positioning.....	54
Figure 3.1. The general scheme of recent water inventory and agri-food LCA databases (Water Database (Quantis), WFDB, Ecoinvent v3): calculations steps and main assumptions to determine the different water flows. Databases are based on a water balance: input water = output water. Each water flow is geolocalised at country or large watershed scale.....	69
Figure 3.2. Scheme of the calculation methods used in water inventory databases (Pfister et al. (2011, 2014) and WaterStat). .....	71
Figure 3.3. Agri-food LCA databases and Quantis water database are using data from water inventory databases for the water elementary flows of agricultural systems. Water inventory databases are using formalisms of water balance and crop evapotranspiration estimation from Allen et al. 1998 (also denoted as: FAO N°56). .....	71
Figure 3.4. Gradient of complexity and accuracy in the different possible approaches (databases and models) for the inventory of field water flows, and associated type of LCA study.....	82
Figure 4.1. The modular structure and main formalisms for the E.T. model. NRCS: Natural Resources Conservation Services.....	101
Figure 4.2. Salt and water balances modules in the model, input data (orange), determining factors (green), and interaction between the salt and water balances (double arrows). The evaporation-transpiration module is assessed through a dual approach: soil evaporation and plant transpiration are assessed separately. ....	102
Figure 4.3. Potential evaporation and transpiration estimation methods: single and dual approaches are presented according to their historical developments and corresponding references. The dual approach implemented in the model is framed by double red boxes.....	106
Figure 4.4. Climate data for the case study Mandarin crop grown in Morocco: minimum relative humidity (RHmin), average temperature, rainfall and reference evapotranspiration (ETo). In 2007, 2008, and 2011, assumptions were required to disaggregate monthly values or fill data gaps (details in Table 4.1).....	109
Figure 4.5. Water flows estimated by the E.T model (E/T partitioning, water and salinity stresses) and other model formalisms, expressed in m3 per ton of Mandarin over seven years (from planting in 2007 to 2015).....	115
Figure 4.6. Actual evapotranspiration (ETa) estimated by the E.T model (E/T partitioning, water and salinity stresses) and other model formalisms, expressed in mm.day-1, over seven years (from planting in 2007 to 2015) .....	115
Figure 4.7. Evolution over time of salts in soil water (red line) and salts in percolating water (blue line) (in g.m-2) for the E.T. model (E/T partitioning, water and salinity stresses).....	116
Figure 4.8. Actual transpiration (Ta) estimated with water stress or with water and salinity stresses, expressed in mm.day-1, over seven years (from planting in 2007 to 2015).....	117
Figure 4.9. Scenario analysis of the E.T. model. The input variables tested are: wz= wetted zone by the irrigation, z= rooting depth, %G= percentage of ground covered by vegetation, CN= curve number (for the runoff calculation), Kcb= basal crop coefficient, ETo = reference evapotranspiration.....	119
Figure 4.10. Values of crop coefficient Kc (continuous lines) and basal crop coefficient Kcb (dotted lines) for citrus from the litterature and simulated by the E.T. model (dashed line). Year 2012/2013, 6 years-old orchard. no cc: no cover crop.....	120

Figure 4.11. ETa blue [m <sup>3</sup> .ton <sup>-1</sup> ] of small citrus provided by water inventory databases at country scale (WaterStat, Pfister et al. 2011 and 2014) and by the E.T. model for a Mandarin Nadorcott orchard located in South-West Morocco.....	124
Figure 4.12. E.T. model integration within a LCA study: input and output water flows will be converted in terms of impacts on the environment thanks to existing water impact assessment methods (e.g.: Pfister et al. (2009) and Boulay et al. (2011a, and b)).....	126
Figure 5.1. Flow diagram for the Moroccan Mandarin production and delivery to the French market (2007-2015, Bahira region, Morocco).....	135
Figure 5.2. Representation of elementary water flows and associated characterisation factors of Pfister et al.'s (2009) and Boulay et al.'s (2011a&b) methods:.....	140
Figure 5.3. Contribution analysis of 1kg of Moroccan Mandarin at French market gate. Midpoint impacts were assessed with ReCiPe Midpoint (H) V1.12.....	143
Figure 5.4. Contribution analysis of 1kg of Mandarin at farm gate in Morocco. Midpoint impacts were assessed with ReCiPe Midpoint (H) V1.12.....	145
Figure 5.5. Water deprivation impacts of 1kg of Mandarin at farm gate, calculated with the method from Pfister et al. (2009) (water scarcity indicator), and two versions of the method from Boulay et al. (2011b) with and without considering the degradative use of the water (water availability and scarcity indicators).....	147
Figure 5.6. Contribution analysis to endpoint damages of 1kg of Mandarin at farm gate in Morocco (ReCiPe Endpoint (H) V1.12). Water consumption damages were assessed with Pfister et al. (2009). .....	147
Figure 5.7. Climate change, acidification, eutrophication impacts (calculated with CML (2001)), and non-renewable energy (calculated with CED), from different LCA studies on citrus and for this study. Results are expressed per kg of fruit at farm-gate. Nota: no value available for non-renewable energy for Sanjuan et al. (2005). .....	149
Figure 5.8. Water inventory flows and water deprivation impacts of Mandarin cultivation (nursery not included). Water inventory flows (in m <sup>3</sup> .ton <sup>-1</sup> ) are estimated with E.T. model or from database, and water deprivation impacts (in m <sup>3</sup> eq.ton <sup>-1</sup> ) associated with these water flows are calculated with different assessment methods. Results are expressed per tonne of export fruits (allocation included), even for databases. ....	152
Figure 5.9. Representation of elementary water flows and associated characterisation factors (CF) of Boulay et al. (2011b) method when addressing the impacts from rainfall water consumptive and degradative use. Rainfall water, evapotranspiration from rainfall ( $\Delta ET_a$ green), and rainfall water released in the environment through runoff ( $\Delta RO_{green}$ ) and deep percolation ( $\Delta DP_{green}$ and $\Delta DP_{blue}$ ) should be quantified as the difference with the reference state. ....	154

## Table des tableaux

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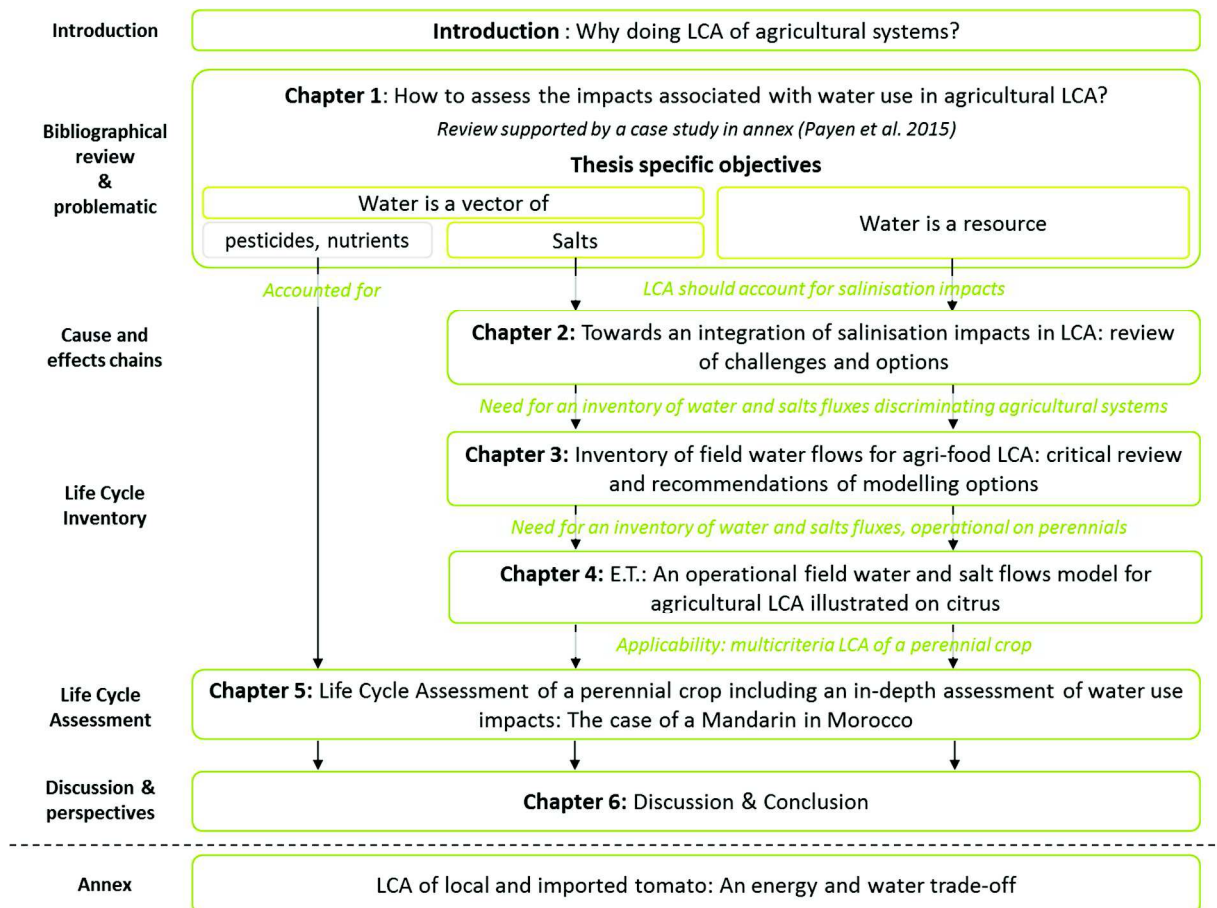
Table 2.1. Key management and biophysical factors involved in secondary salinisation, per salinisation type. .	39
Table 2.2. Inventory requirement, characterization factors and category indicator results of salinisation impact assessment methods in LCA.....	46
Table 3.1. Sources of input data of the global crop water databases WaterStat and Pfister et al. (2011) .....	73
Table 3.2. Model specifications and data sources for the inventory of water flows for the LCA-based ecodesign of cropping systems.....	79
Table 3.3. Required input variables for simulations with AquaCrop (Based on Vanuytrecht et al. (2014)) .....	85
Table 3.4. Database of modelling approaches for the assessment of water flows according to the objective of the Agri-food LCA study, the data available, the resources and time available .....	88
Table 4.1. Model input variables and parameters description, temporal resolution, determining factors, and average values for the case study (Mandarin in Morocco), and data sources and assumptions .....	110
Table 4.2. E.T. versus other model formalisms.....	113
Table 4.3. Basal crop coefficient values for citrus tested in the E.T. model .....	114
Table 4.4. Scenarios of practices tested in the E.T. model and corresponding values.....	114
Table 4.5. Percentage variations of water flows simulated with E.T. for the range values of initial conditions ( $S(i)$ and $EC_{soil\ water}(i)$ ) and for the average fraction of Total Available Water (TAW) that can be depleted from the root zone before water stress occurs ( $p$ ). .....	118
Table 5.1. Agronomic data summary for the main orchard development phases: average yield, NPK fertilisation, irrigation volumes and energy, main active substances for plant protection.....	138
Table 5.2. Data sources for volume and quality of water elementary flows, and associated characterisation factors (CF) for Boulay et al. (2011,a&b) (water availability indicator) and Pfister et al. (2009) methods (water scarcity indicator).....	141
Table 5.3. Cradle-to-farm-gate LCA results per kg of Mandarin for a selection of environmental indicators (Midpoint impacts assessment with ReCiPe Midpoint (H) V1.12) .....	146
Table 5.4. Cradle-to-farm-gate LCA results per kg of fruit for a selection of environmental indicators (ReCiPe Midpoint (H); Cumulative Energy Demand) for the Clementine and Mandarin grown in Morocco..	155
Table 5.5. Nutrient application rates for the Mandarin grown in Morocco (this study) and published citrus LCA studies.....	156



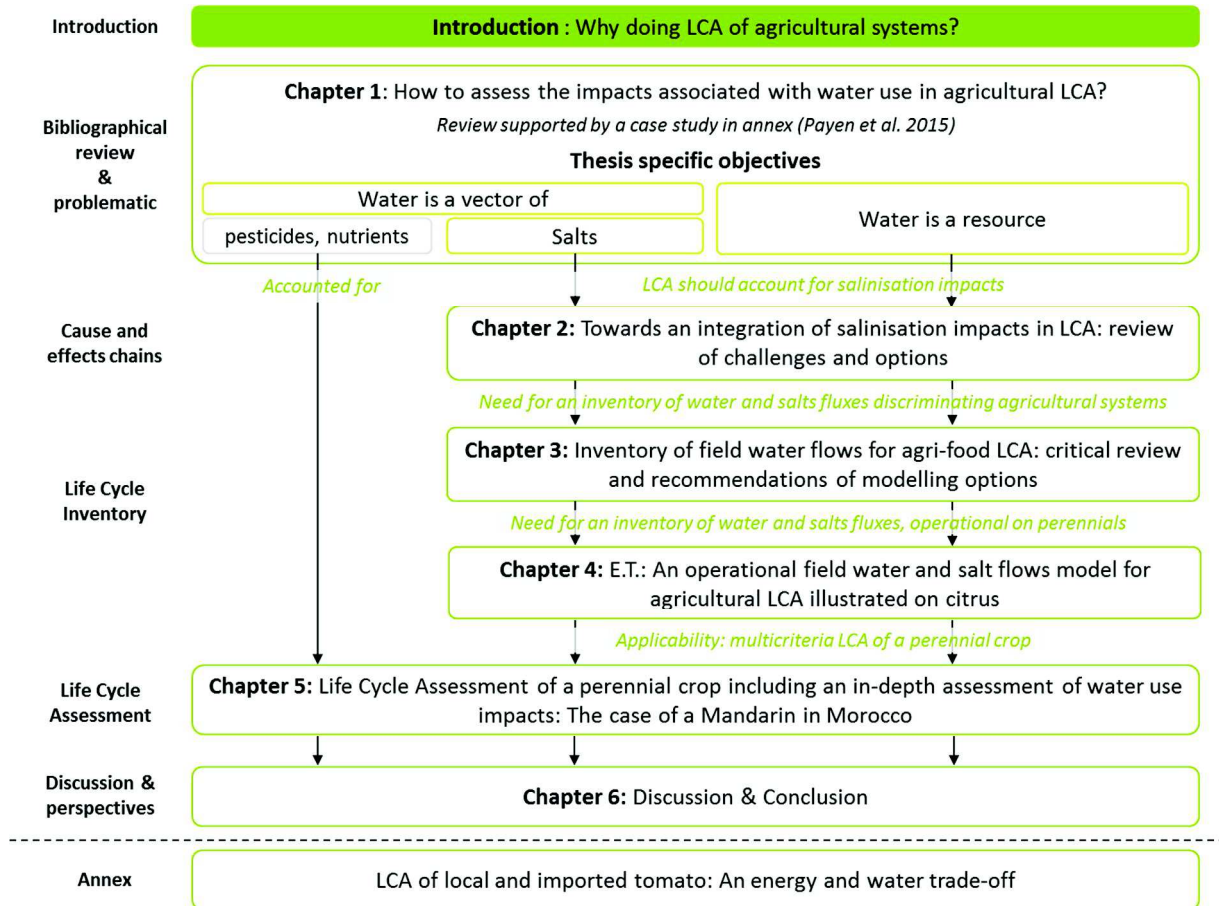


## Structure of the thesis

A brief introduction will demonstrate the importance of analysing the environmental impacts of agricultural systems and the relevance of using the Life Cycle Assessment (LCA) methodology. Then, a bibliographical review will specifically address the assessment of water use impacts in chapter 1. Through the identification of general research needs, this review will introduce the scientific questions addressed in this dissertation and its specific objectives. Next, the core of the dissertation will be organised into four main chapters including 1/ a proposal of a framework for accounting for salinization impacts in LCA, 2/ a review of available models for field water and salt flows inventory in LCA, 3/ a description of a new model for estimating these fluxes and its implementation into a complete LCA case study for citrus in Morocco. In the last section, a general discussion (in French) will then be proposed before the conclusions:







## Introduction

### Why doing Life Cycle Assessment of agricultural systems?

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#### 1.1. Feeding the planet without destroying it

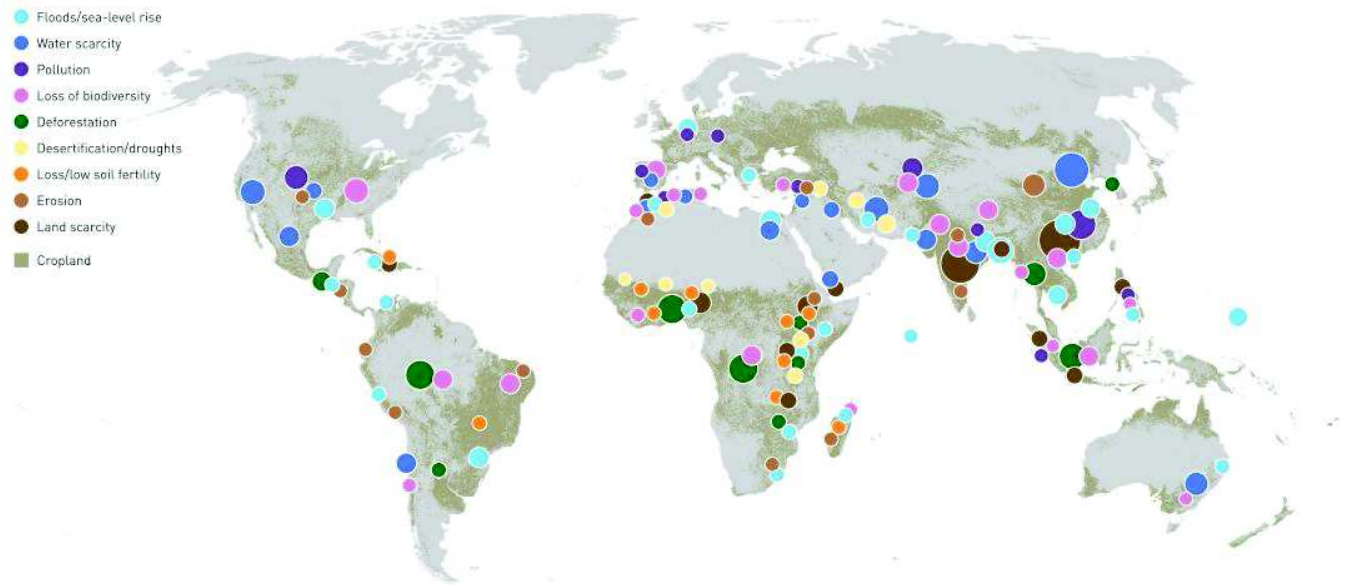
##### 1.1.1. Agriculture is feeding the planet, but has many impacts on the environment

Agriculture fulfils a function of production: providing food for human, but agriculture activity can reduce the ability of ecosystems to provide goods and services (Tilman et al. 2002). The stake is to ensure the food provision without affecting -too much- the environment, in other words, it is to have a sustainable agriculture.

Agriculture - including crops, livestock, forest, fisheries and aquaculture- is the main human activity responsible for the use of land and water resources (FAO 2013), and has many impacts on the environment (Fig. 1). Agriculture has an impact on climate change, notably through the emissions of greenhouse gases: 50% of the methane emitted into the atmosphere by human activity is due to crop and livestock production alone (FAO 2013). Agriculture has an impact on water resources through its consumption and degradation: 70% of global water withdrawals are done by agriculture (World Water Assessment Program 2009) (and irrigated crops sustain 40% of the global food production (Abdullah 2006)), and the main source of nitrate and ammonia pollution in waters come from agriculture (FAO 2013). Indeed, since agriculture started, the cycle of different elements from the soil (N, C, P) has been altered which led to their partial decrease in soil and to their accumulation in the sediments from different ecosystems. The use of inputs such as pesticides and mineral fertilizers later on added more nutrients and more pollutants into motion on earth, thus exacerbating this dual phenomenon of soil fertility decrease and environmental compartments pollution. (Fert)irrigation water is a driver of pollutions because irrigation return-flows usually carry more nutrients, salts and pesticides than in source water, impacting downstream agricultural, natural systems (Tilman et al. 2002). Thus, agricultural activity contributes to water quality degradation and causes (eco)toxicity, eutrophication and acidification impacts.

##### 1.1.2. An increasing pressure...

The pressure on agricultural systems is increasing because of the growing world population, the demand from other competing uses to food production (e.g. biofuels), and climate change (Mateo-sagasta and Burke 2010). Actually, climate change alone will have substantial impacts on the irrigation water demand, according to projections based on a set of seven global hydrological models (Wada et al. 2013). This increasing pressure is accentuated a vicious circle: impacts on the environment of agriculture have, in turn, negative impacts on agriculture production (FAO 2011).



**Figure 1. Schematic overview of risks associated with main agricultural production systems. (FAO Land & water (2011) - the state of the world's land and water resources for food and agriculture)**

### 1.1.3. Identify the environmental hot spots and mitigation options

There is a growing awareness of the importance of environmental impact of food products. European policies promote the quantification of the environmental performance of food supply chains (Peacock et al. 2011). Developing more sustainable and efficient production systems is crucial in a context where we have to produce more and pollute less. The stakes of the environmental assessment are thus considerable at a time when we are wondering how to feed the planet. This is a society issue affecting and involving politicians, farmers and consumers.

Life Cycle Assessment (LCA) is a suitable tool to evaluate the environmental impacts of the functions of agricultural systems, and a powerful decision-making tool for the different stakeholders.

## 1.2. LCA of agricultural systems: challenges

### 1.2.1. LCA methodology

LCA is a standardized and internationally recognized methodology to assess the environmental impacts of a function (product or service) over its entire life cycle (from cradle-to-grave) (ISO 2006a; ISO 2006b). Contrary to single indicator methodologies such as carbon footprint or water footprint, LCA is a multicriteria assessment method addressing a wide range of impact categories such as global warming potential, ecotoxicity, eutrophication, and acidification (Fig. 2).

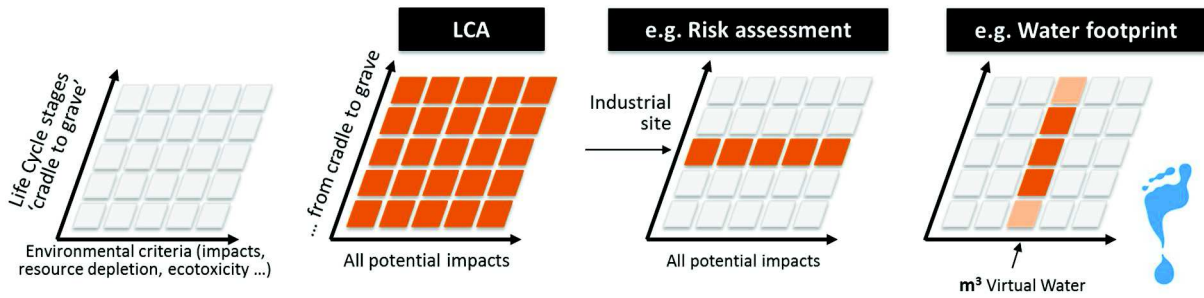


Figure 2. LCA: A global method. On this figure, water footprint refers to the volumetric blue, green and grey waters (Source: P. Roux - IRSTEA)

Risk-Assessment (site-specific) and LCA (product-specific) are complementary approaches: a product can be analysed using LCA and, at the same time, a Risk Assessment can be performed for a number of core processes in the chain, in which the emphasis is on the local environmental impacts (Guinée et al. 2002). According to the ISO norm, LCA consists of 4 steps: the goal and scope definition, the inventory, the characterization of impacts and the interpretation. In practice, all the inputs and outputs (resources extraction and emissions to the environment) associated with the product system are inventoried in the inventory stage, then, each flow is converted in environmental impacts indicators thanks to characterization factors. These impacts (at midpoint level) can be further aggregated into damage indicators on Human health, Ecosystems quality and Resources (at endpoint level) (Fig. 3). Human health, Ecosystems quality and Resources are defined in LCA as the areas of protection: the entities that we want to protect. Nevertheless, their precise definition is not fully consensual since they depend on the vision of sustainability and the underlying values (Adams 2006; Dewulf et al. 2015). This is an interesting feature (and maybe a weakness point) of LCA: LCA aims to be science-based, but involves assumptions and value choices (Guinée et al. 2002). The importance is thus to make these choices transparent while reporting a LCA study.

The inventory and environmental impacts and damages are related to the studied function of the system through the functional unit (e.g: provide 1 kilogram of tomato on the French market).

LCA is a tool presenting many assets:

- its holistic approach addresses many environmental impacts, making visible possible transfers of pollution between different technologies fulfilling the same function, and considers the whole life cycle of the product, allowing the identification of environmental hot spots,
- its functional approach allows for a more powerful eco-design regarding the service provided,
- its quantified characterisation of impacts is based on scientific modelling of environmental mechanisms,
- it is based on an international consensus and a large community of experts and scientists,
- it is supported by operational tools and databases.

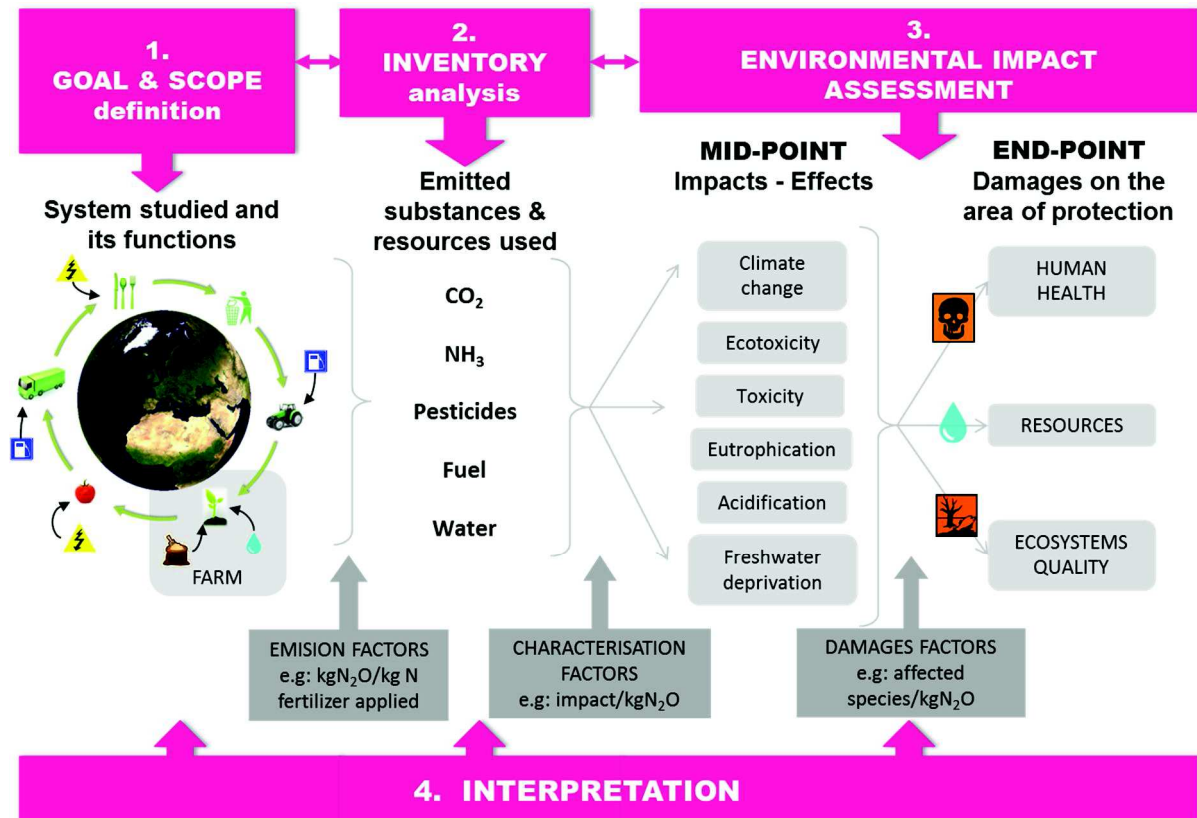


Figure 3. Four steps of the LCA methodology with simplified substances inventory and impact categories: 1. the goal and scope definition, 2. the inventory analysis, 3. the environmental impact assessment, and 4. the interpretation that should be performed at each of the three previous steps. Arrows between the different steps show that LCA is an iterative process.

### 1.2.2. The cause and effect chain or environmental “pathway”

Environmental mechanisms are complex and their modelling is a challenging task. One of the main challenges of LCA is to assess the global potential impacts of a given substance emission. The cause-effect chain is the cascade of environmental processes provoked by a substance emission (the cause), until the midpoint impacts (the effect), and finally the endpoint damages to the area of protection. Figure 4 gives an illustration of the cause-effect chain in LCA for the aquatic eutrophication potential. For a detailed and up-to-date presentation of the principles and practice of life cycle impact assessment, see Hauschild and Huijbregts (2015), part of the book series *LCA Compendium: The Complete World of Life Cycle Assessment*.



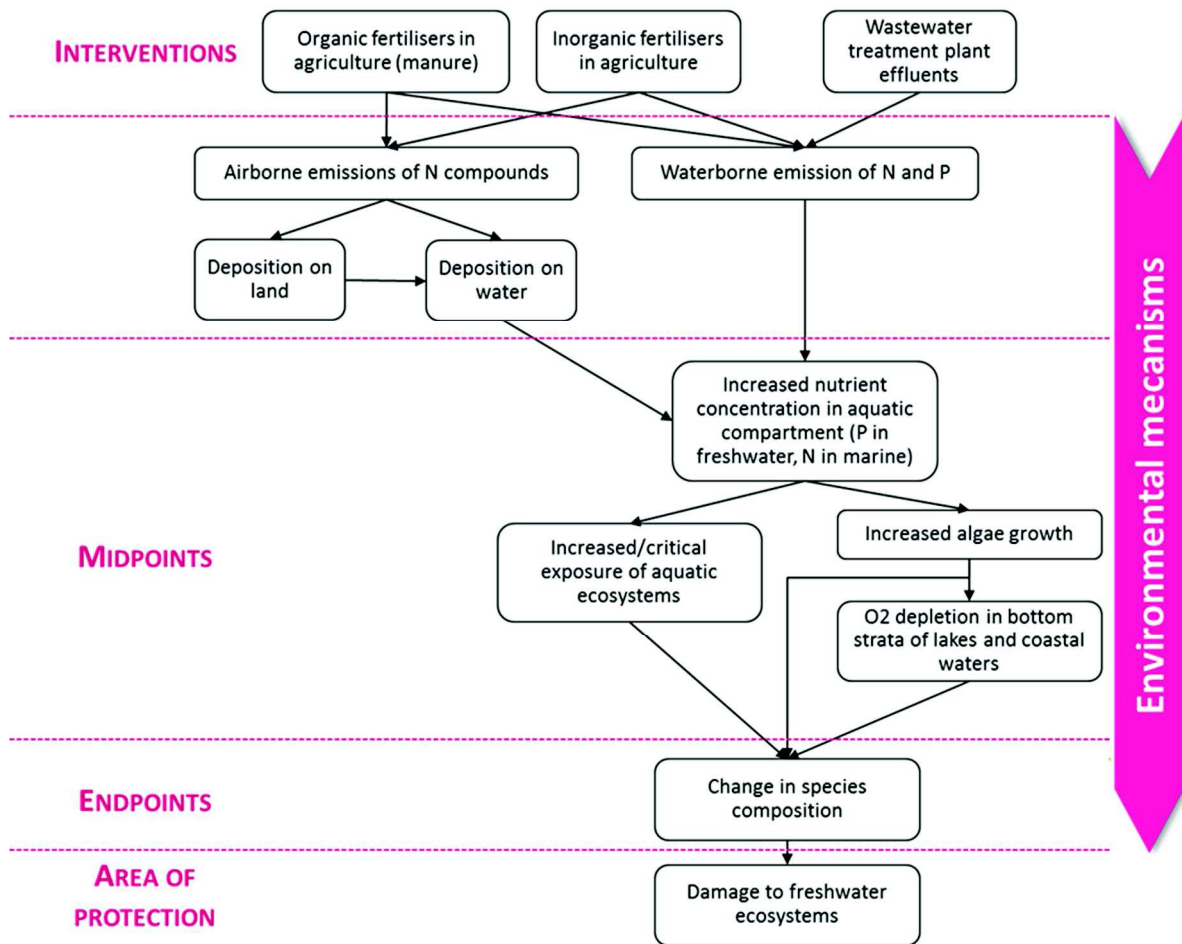


Figure 4. Flow diagram of the cause and effect chain for aquatic eutrophication (ReCiPe method): from human interventions to the midpoint impacts, and finally the endpoint damages on Ecosystem quality (based on ILCD Handbook (2011))

### 1.2.3. Why applying LCA to agricultural systems is relevant?

LCA is particularly relevant for the assessment of agricultural systems. Indeed, agriculture supply chains are globalised (Hubacek et al. 2014): imported food products rely on imported agricultural inputs such as fertilizers. LCA, through its global approach, account for all processes along the supply chains occurring all-over the world. Thus, agriculture is responsible for many impacts, at different localisations in the world, and there are impact categories for which the level of impact depends on the localisation of the emission. If global warming has a global effect on earth (the place of the greenhouse gas emission does not matter, and the impacts concern the whole planet), this is not the case for water eutrophication which is a local or regional impact (nitrate emissions will affect the local scale and the impacts will depend on the sensitivity of the environment)(Azevedo et al. 2013). Thus, we can distinguish three levels of spatial differentiation of impact assessment: site-generic, site-dependent, and site-specific assessments. Site-generic is globally valid (e.g. climate change), site-dependent operates on the regional scale, and site-specific is only locally applicable (Potting and Hauschild 2006).

Although LCA is relevant for evaluating agricultural systems, the methodology has to cope with complexities associated with agricultural systems:

### 1.2.4. The inventory: a crucial LCA stage for agricultural systems

In agricultural LCA, the objective is to relate the outputs to the inputs of the system accounting for the climate, the soil characteristics and the agricultural practices. Differentiating the practices for the inventory calculation is a major challenge because they are highly diverse and interact with soil and climate. Fluxes generated during agricultural processes are submitted to soil and climate influence and variability, but depend also on agricultural practices. It is therefore difficult to estimate field emissions such as nitrogen emissions due to fertilisation. Furthermore, it is crucial to define *what is an emission*, through the system delimitation. The definition of the temporal and physical (spatial) boundaries of the agricultural system (the technosphere) is complex because soil is both part of the agricultural system studied (the growing medium) and of the environment (Fig. 5)

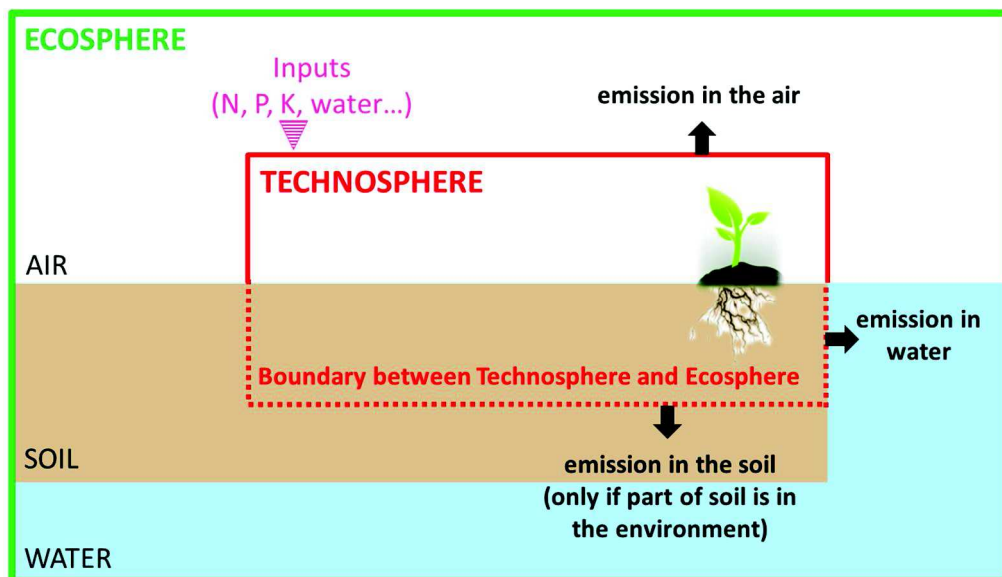


Figure 5. Technosphere (agricultural system under study) and Ecosphere (environment receiving the emissions) boundary depend on the soil status (dotted line).

### 1.2.5. One limitation of LCA relates to the modelling of freshwater use impacts.

There are deficiencies in the impact assessment associates with water use.

Water has a double status in LCA: it is an environmental compartment receiving pollutions, and a resource (Nota: the same is valid for the soil). Impacts on water as a compartment have received detailed attention over the last decades through the development of several environmental impacts (Eutrophication, Acidification, ...) (see Finnveden et al. (2009) and Pennington et al. (2004) for a review), whereas the impact from water use has only recently been considered and has not already resulted in consensual methods. Both water use (consumptive and degradative use) and emissions of substances polluting water are impacting waters (Fig. 6).

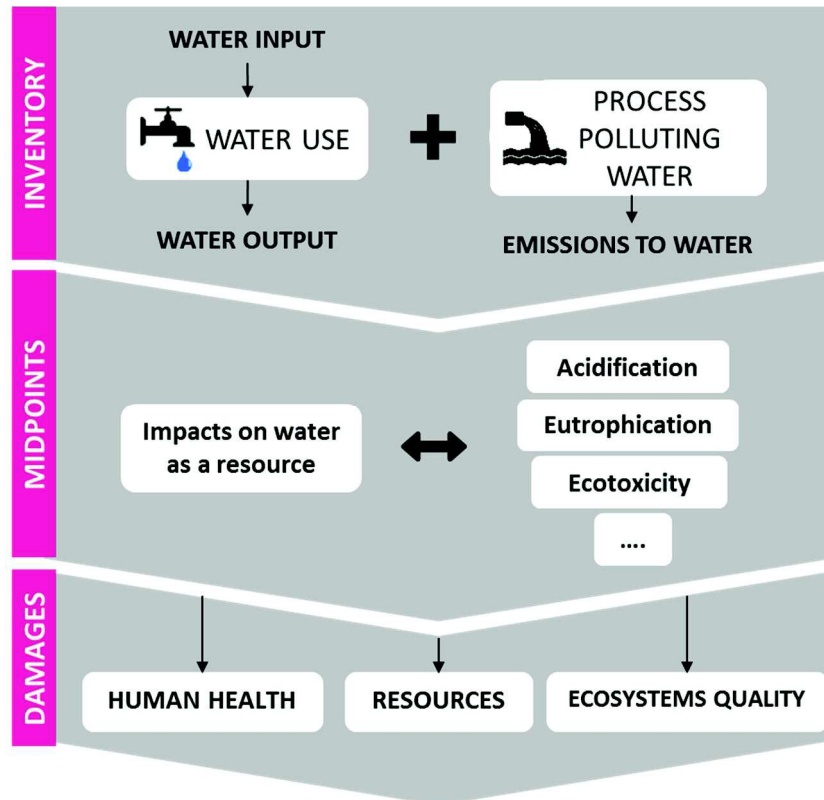


Figure 6. From the inventory to midpoints and damages related to water in LCA. At the inventory level, processes can use water and/or pollute water through the emission of substances. Impacts related to water use include the impact on the water resource and water pollution. Ultimately, these impacts damage the three areas of protection.

### 1.3.2. The importance of scales

Although the demand for water has doubled since 1960 (Millennium Ecosystem Assessment 2005), human populations did not reach the critical “planetary boundary” for freshwater use according to Rockström et al. (2009). Planetary boundaries express the limits of human pressure the whole planet can endure without critical damage. But Rockström's study failed to address the pressure on the water resource by considering it as a global resource. In 2015, an update of the planetary boundaries proposed a new boundary: freshwater use at the river-basin scale (Steffen et al. 2015). But the definition of the river-basin scale water boundary is limited by the hazardous estimation of environmental water flow requirement (water flow to preserve for the natural ecosystems).

This example is an illustration the importance of the scales when addressing water: if the global water volume remains constant at earth level over time, its potential scarcity is expressed at the local or regional scales, and at specific temporal scales depending on the contexts. Moreover, water is available at variable levels of quality which drive its possible functions and uses.

Both the expression of the water scarcity phenomenon at various non-global scales and the interaction between its qualitative and quantitative dimensions make extremely complex the modelling of water use impacts in LCA.

## 1.4. Conclusion

Identifying the environmental hot spots and mitigation options of agriculture are crucial tasks in a context where humanity has to produce more and pollute less. Life Cycle Assessment (LCA) is a powerful tool to evaluate the environmental impacts of agricultural systems, but this methodology is still fraught with shortcomings and research challenges. One limitation of LCA relates to the modelling of freshwater use impacts. This is particularly important when evaluating agricultural systems which are both consuming and degrading water.

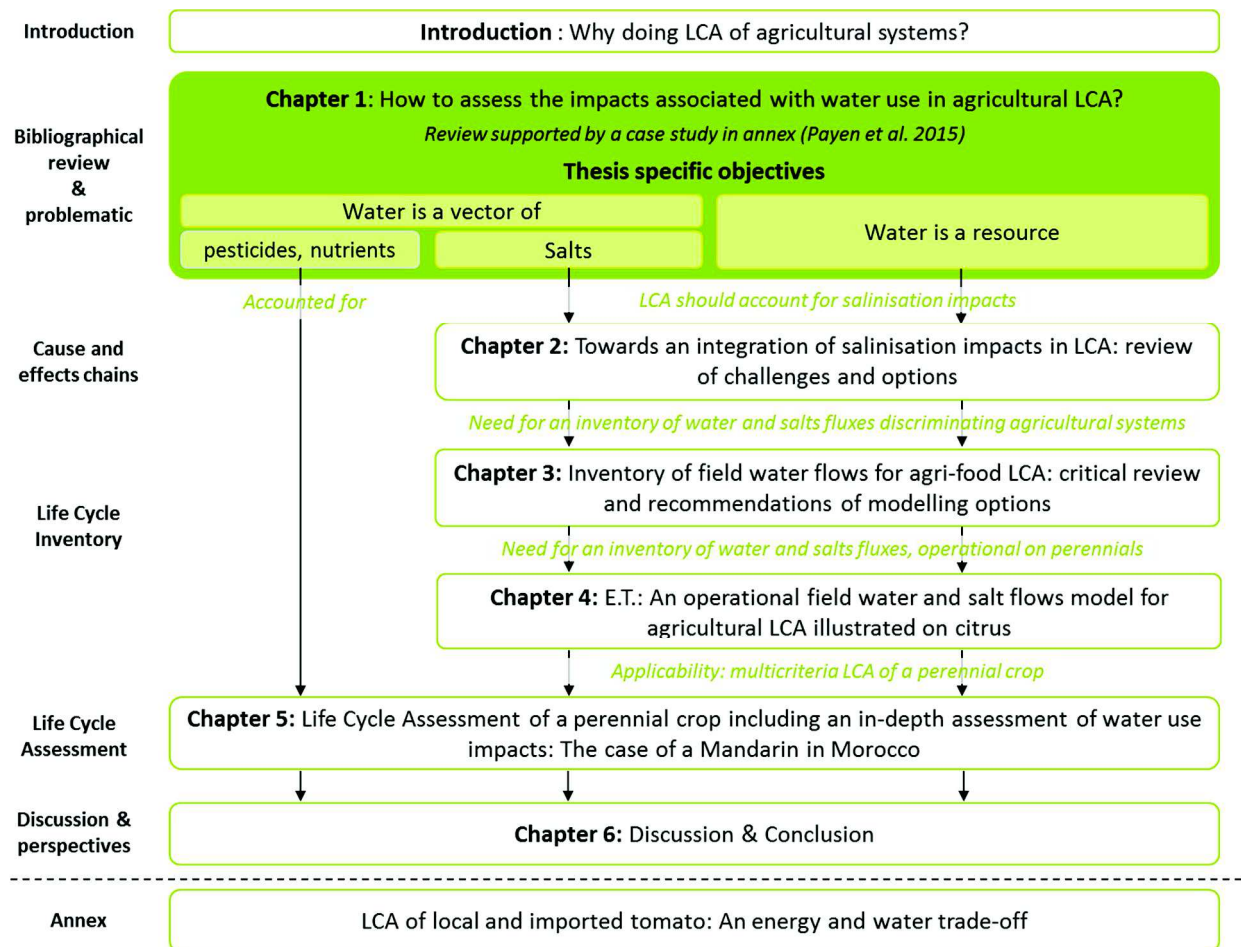
Payen et al. (2015) (See annexes) showed that accounting for water use impacts (even with a perfectible method) can radically change the outcomes of a LCA study comparing the environmental impacts of locally-grown vs. imported tomato. But the study also showed that the assessment of freshwater use impacts and damages still has shortcomings, leading to an underestimation of the impact for the Moroccan tomato. Therefore, the framework for assessing the impacts of water use in LCA will be analysed in depth in a bibliographic review in the next chapter.

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Introduction showed that LCA is a powerful but perfectible tool for evaluating the environmental impact of agricultural systems. Chapter 1 further investigates the strengths and limitations of LCA regarding the assessment of water use impacts.



## Chapter 1

### How to assess the impacts associated with water use in agricultural LCA?

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This chapter provides a synoptic literature review on the methods used in Life Cycle Assessment (LCA) for addressing water use impacts, emphasising the main differences, strengths and limitations of the methods through the successive steps of inventory, midpoint and endpoint impact assessment. The objective is to identify the research needs toward accurate LCA of agricultural products.

#### 1.1. Water footprints terminologies

It is paramount to clarify the terms, in accordance with the terminology promoted and used in the ISO (International Organization for Standardization) standards and in recent publications from the UNEP (United Nations Environment Programme) – Setac (Society of Environmental Toxicology and Chemistry) working group Water Use in LCA (WULCA) (e.g.: Boulay et al. 2015a; Boulay et al. 2015b; Boulay et al. 2015c). First, volumetric footprint must be distinguished from impact-oriented water<sup>1</sup> footprint. From an LCA perspective, the volumetric accounting of water use (m<sup>3</sup>) is not sufficient because numerically smaller footprint can cause larger impacts depending on the context (Berger and Finkbeiner 2010). The ISO standard on Water Footprint (ISO 14046 2014) provides a definition of water footprint: *“the metric(s) that quantifies the potential environmental impacts related to water”*. This impact-oriented water footprint is different from previous work on volumetric water footprint of a product or a nation from Allan (1998), Hoekstra and Hung (2002), defined as the sum of blue, green and grey water footprints. The water footprints terminologies can be confusing, especially because LCA researchers mobilised the existing concepts of blue, green and grey waters to name the water inventory flows in LCA. However, the recent ISO 14046 standard does not use these “colour” but refers to the hydrological nature of water flows. It defines actual water types as: groundwater, surface water, brackish water, seawater, fossil water and precipitation in relation to the water cycle and hydrological mechanisms.

A comprehensive water footprint assessment *“considers all environmentally relevant attributes or aspects of natural environment, human health and resources related to water, including water availability and water degradation (negative change in water quality)”* (ISO 14046 2014, 3.3.3). A water footprint profile should therefore illustrate the double status of water: because it includes impacts related to water degradation such as aquatic eutrophication or aquatic ecotoxicity, considering water as a living compartment, and also includes impacts related to water use, such as water deprivation, considering water as a resource. In practice though, two types of impact categories related to water use exist: water scarcity, referring to a consumptive use, and water availability, referring to a consumptive and degradative use. Indeed, *water quality can also influence availability* (ISO 14046 2014, 3.3.16). Thus, scarcity refers to the pressure on water resource from a quantity perspective only, whereas availability refers to the pressure on water resource due to both water quality degradation and quantity depletion

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<sup>1</sup> Note : in the following, water is used as a synonym of freshwater, unless specified

(Fig. 1.1). To sum up, a complete water footprint profile should address both the effects of water quantity and quality change on the environment, for the resource-water and living compartment-water viewpoint (Fig. 1.1). The stake of a water footprint assessment is to convert volumes of water used and/or degraded by a product or service, into potential environmental impacts: from the inventory to the impacts and damages.

In this review, we will only present the methods used to assess the impacts on the water as a resource and not on the water as a compartment (e.g. eutrophication) (hatched boxes in Fig. 1.1).

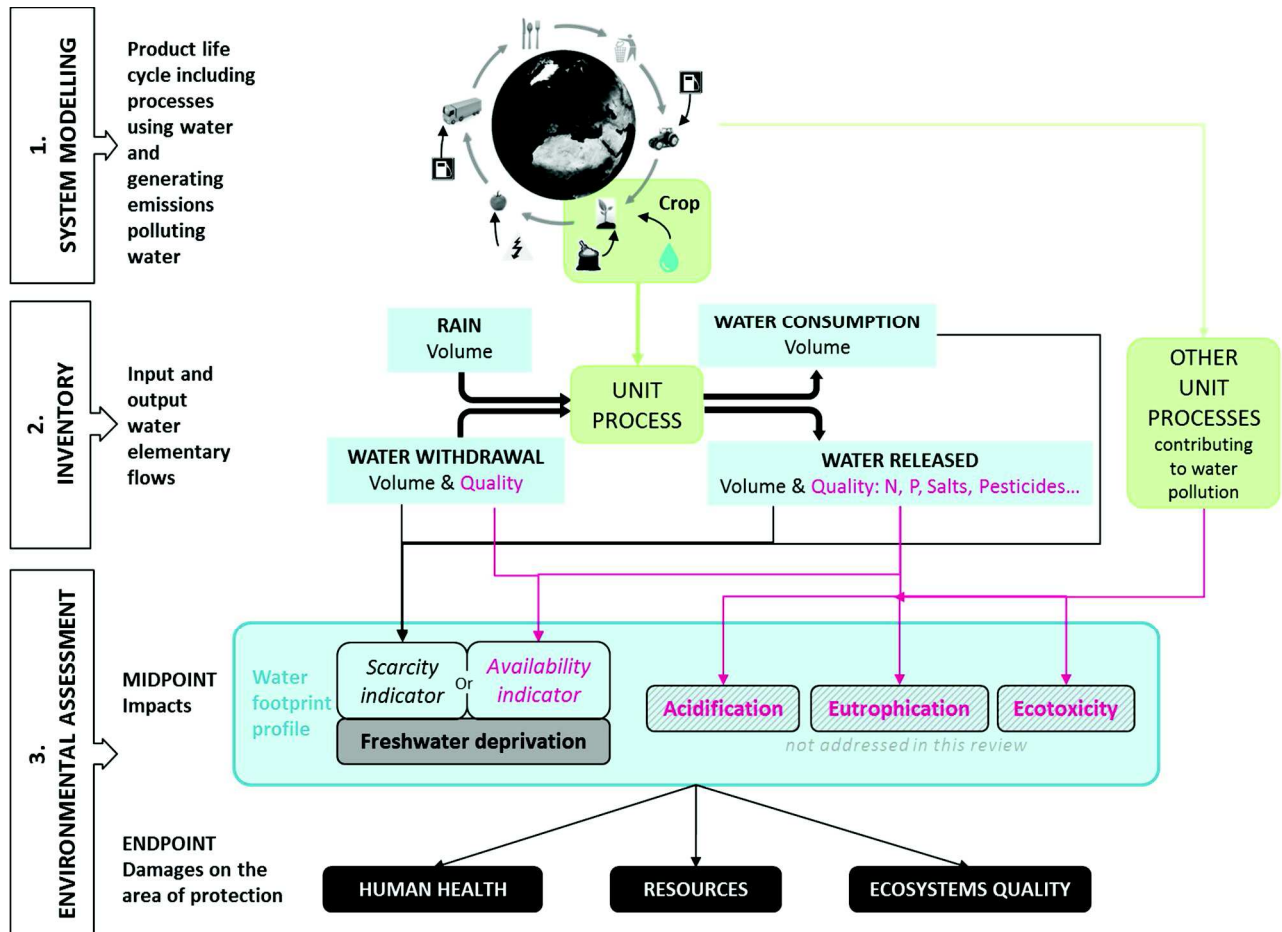


Figure 1.1. Water use impact modelling framework in LCA: 1. Product life cycle modelling including many processes using and/or polluting water, 2. Inventory of water flows for each process in terms of volume and quality (inventory flows requirement depends on the method), 3. Environmental impact assessment: water footprint profile includes water deprivation impacts (based on scarcity or availability indicator), but also acidification, eutrophication, ecotoxicity and other impacts related to water degradation. Hatched boxes represent impacts not addressed in this review.

## 1.2. An overview of the different methods

If LCA was an adolescent, water use impact analysis in LCA would be a baby. But a fast growing baby: over the last 6 years, life cycle impact assessment of water use has evolved rapidly with many new methods emerging (Tendall et al. 2013). Several reviews of these methods exist: Berger and Finkbeiner (2010), Jeswani and Azapagic (2011), Berger and Finkbeiner (2012), Kounina et al. (2013), Boulay et al.



(2015c). This chapter is based on these reviews, the original publications they are referring to, but also more recent publications.

The general framework of water use impact modelling in LCA includes: description of the product life cycle in terms of processes using water, inventory of water flows of all processes throughout the product life cycle, environmental assessment either at midpoint level through the conversion into water deprivation impacts by multiplying with characterisation factors, or at endpoint level (also called damage assessment) on the three Areas of Protection (AoP) (Fig. 1.1). Nevertheless, water use impact assessment methods are addressing different cause-effect chains and rely on different water use inventory schemes and characterisation models (Kounina et al. 2013). There are midpoint and/or endpoint oriented (Fig. 1.2). To date, no single method allows for a comprehensive impact assessment of all possible impacts due to water use. Midpoint category indicators are either scarcity (e.g. Pfister et al. 2009) or availability indicators (e.g. Boulay et al. 2011), specific to one AoP (Milà i Canals et al. 2008) or covering all AoP (e.g. Pfister et al. 2009). Across all available methods, endpoint category indicators addressing the same AoP are neither identical, nor complementary. Thus, to obtain a comprehensive water footprint profile, the compilation of several methods would be required.

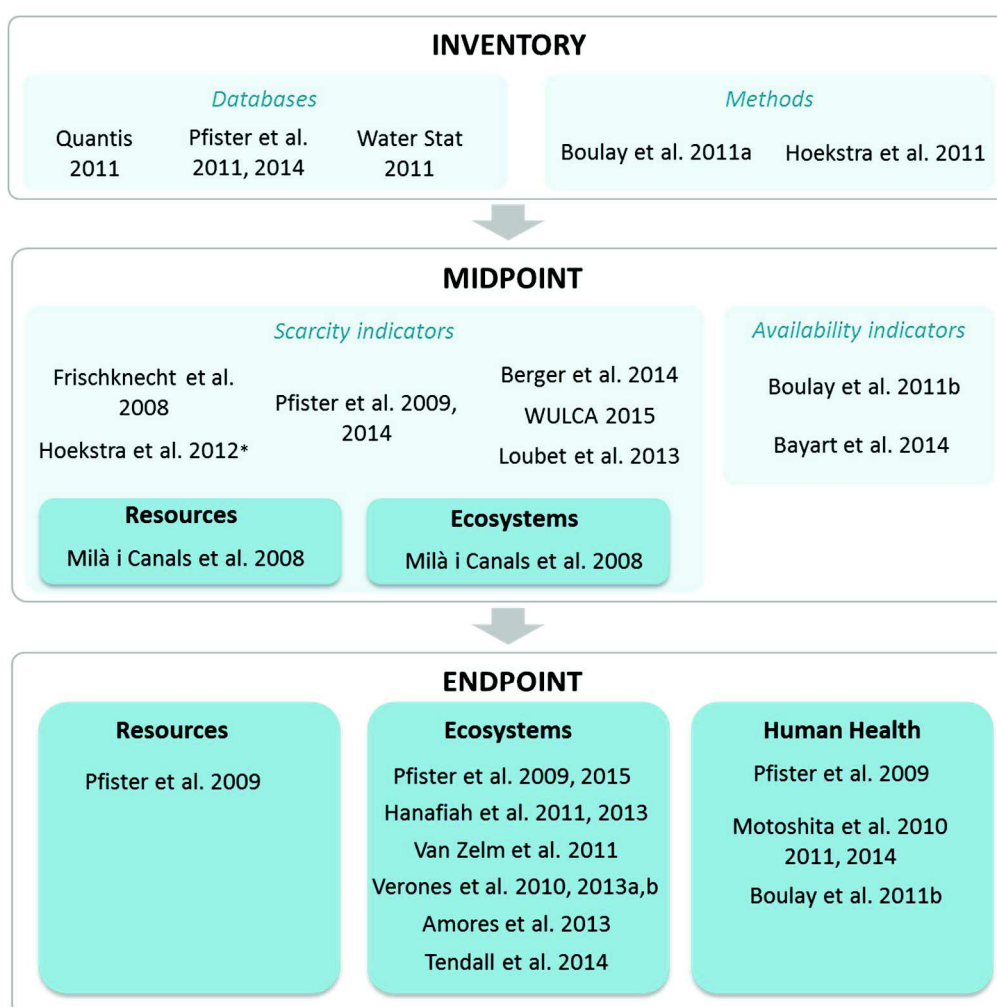


Figure 1.2. An overview of methods addressing water use in LCA with classification for the three areas of protection. At midpoint, scarcity indicators (addressing only water quantity) are differentiated from availability indicators (addressing both water quantity and quality). \*Hoekstra et al. (2012): not developed specifically for LCA but compatible.

A complete list and description of methods addressing the impacts and damages of water use in life cycle impact assessment is provided in the supplementary information. The scientific bases and specificities of each method will be further presented and discussed in sections 1.3 (inventory), 1.4. (midpoint impact assessment) and 1.5. (endpoint impact assessment), the main discrimination criteria being whether the method accounts for water quality alteration or not.

### **1.3. Inventory schemes: water quantity, quality and source**

#### **1.3.1. The inventory scheme depends on the impact assessment method**

The ISO 14046 specifies that the inventory of water elementary flows *shall include inputs and outputs from each unit process being part of the system to be studied*, while respecting a balance. Information on each water elementary flow should include quantity, source, quality, form of water use, geographical location, and temporal aspects. But in practice, the water inventory requirements differ amongst methods. On the one hand, there are methods whose inventory is based solely upon the water consumed (Hospido et al. 2012; Milà i Canals et al. 2008; Pfister et al. 2009). Water consumption is water removed from, but not returned to, the same drainage basin (e.g.: evaporation, transpiration, integration into a product). Berger and Finkbeiner (2012) pointed that the continental evaporation recycling rate is important in some areas and advocated to account for evaporative water returned to the basin (via precipitation) in his method (Berger et al. 2014). On the other hand, there are methods relying on the water withdrawal (removal of water from any water body, permanently or temporarily) and the water released (returned to the same catchment area where it was withdrawn during the same period of time) like Boulay et al. (2011a, b) (Fig. 1.3). Indeed, a method accounting for water quality degradation cannot rely on the single water consumption flow since the quality of the withdrawn and released water flows have to be compared. The recommendations for LCA practitioners and researchers are to inventory both withdrawn and released waters (Bayart et al. 2010; Kounina et al. 2013).

Rainwater has a special and complex status because it relates both to land use and water use in the water cycle: this water flow is only accessible through the soil and for plants (except in the case of rainwater harvesting). The consumption of rainwater stored in the soil profile has received poor attention in LCA because it is considered less environmentally relevant from a pure water consumption perspective (Núñez et al. 2013). Most methods consider that the use of rainwater does not have direct effect on water scarcity/availability. Yet, soil water consumption has an influence on water availability in rivers and aquifers. Only Milà i Canals et al. (2008) and Núñez et al. (2013) proposed a method to account for rainwater use through the land-use effects on the water cycle. In Núñez et al. (2013), the inventory flow is the net change in soil-water availability under the production system compared to the natural reference situation. But this approach has to cope with two issues: first, the definition of the potential natural vegetation and its water consumption, second, the fact that natural vegetation always consumes more water than an agricultural production system, thus leading to positive impact of the production system on the water availability. This shows the complexity to account for the water hydrological cycle and water flow redistribution within the LCA framework.

A detailed list of inventory requirements for each method is provided in the supplementary information.

### 1.3.2. The water sources must be distinguished as they might face different scarcities/availabilities

In theory, “the inventory flows represent a set of water types each representing an elementary flow with its own characterisation factors” (Bayart et al. 2010). Indeed, each water type (source of water in the environment) has different renewability rates and functionalities. For example, differentiating groundwater from surface water in CF calculation is important because some regions suffer much more from groundwater scarcity and others more from surface water scarcity. Such a distinction is not made in Pfister’s method where surface and ground waters are weighted with a unique CF. Only a few methods differentiate water sources (except fossil water): Hospido et al. (2012) and Boulay et al. (2011a & b). Hospido and colleagues (2012) proposed to associate a specific CF for each water type of the irrigation profile: surface, ground, desalinated and non-conventional water (based on Milà i Canals et al. 2008 method). However, in practice, they allocate the same CF for surface and ground waters, and do not consider the water that may be released to the environment. Boulay et al. (2011a) not only proposed an inventory distinguishing the water sources, but also accounted for the quality of input and output waters through water categories.

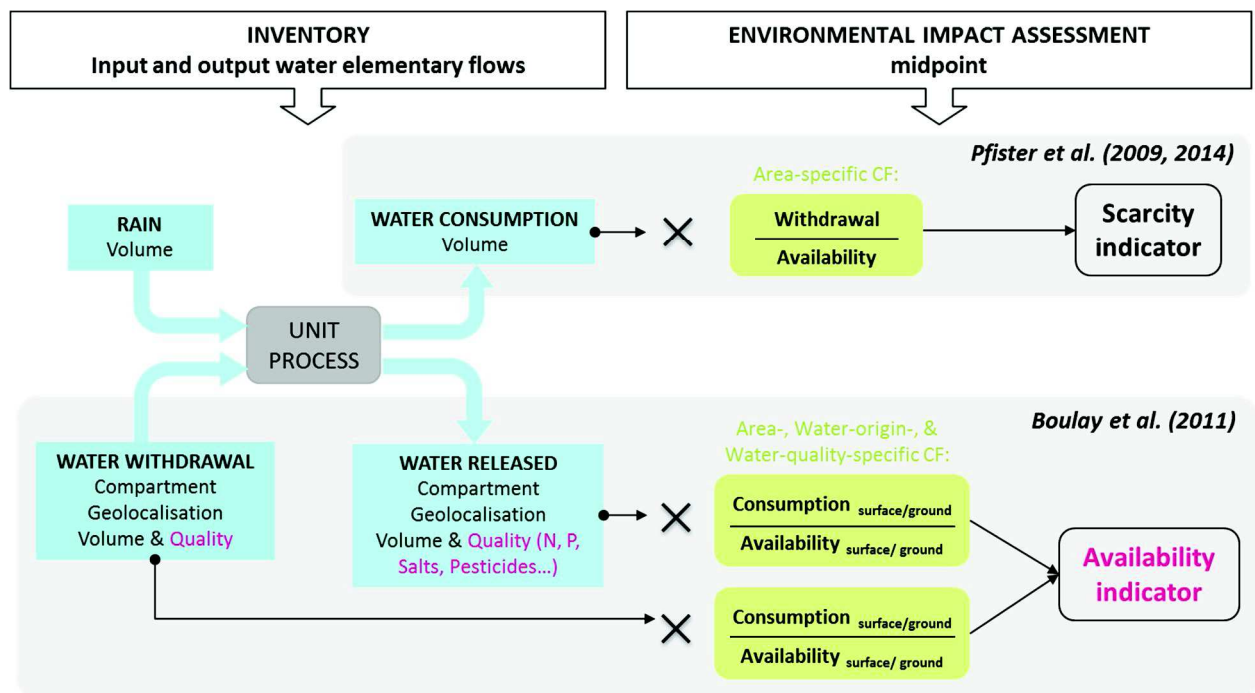


Figure 1.3. Inventory schemes and characterization factors for Pfister et al. (2009, 2014) as water scarcity indicator, and Boulay et al. (2011) as water availability indicator.

### 1.3.3. Water quality has to be inventoried as quality degradation may contribute to water deprivation

The WULCA group recommended the “*use [of] water quality parameters to characterize freshwater flows*”. Boulay and colleagues (2011a) developed an inventory method whereby water quality is related to a function assessing to which users the waters withdrawn and released are functional (useful) (Bayart et al. 2010). This functionality-based water inventory considers that water quality degradation can lead to water deprivation if not suitable anymore for specific users (Boulay et al. 2011a). Thus, it allows for an impact assessment associated with water quality consumption and degradation.

### 1.4. Midpoint impact assessment: water scarcity or availability?

At midpoint, most methods are characterizing a water deprivation impact, in cubic meter equivalent, i.e. a volumetric use is adjusted against the water scarcity/availability conditions that prevail at the place of consumption or withdrawal.

#### 1.4.1. Water indices are used as characterisation factors

The Characterisation Factors (CF) used to convert the water inventory flows into impacts are based on water indices and represent the actual pressure on water resources. These water indices are originally non-LCA-based indicators and are recognized as proxies for water scarcity/availability (see supplementary information). They consist of a ratio of water use by different sectors to the water available, but vary depending on what is considered as “water use” (numerator) and water available (denominator). Indeed, they are either based on a withdrawal-to-availability ratio (Milà i Canals et al. 2008; Pfister et al. 2009; Ridoutt and Pfister 2010) or a consumption-to-availability ratio (Boulay et al. 2011b; Hoekstra et al. 2012). The CFs rely on existing water availability data from hydrological models at global scale such as WaterGAP or UNH/GRDC (Alcamo et al. 2003; Fekete et al. 2002). An alternative of global hydrological model was recently tested, using a large-scale hydrological modelling with the Soil Water Assessment Tool (SWAT)(Neitsch et al. 2009) (Scherer et al. 2015). This attempt showed that although the SWAT model outperformed the global models at large watershed scale, its use on a global scale is unlikely because of the high calibration efforts required (Scherer et al. 2015).

The related CF for each approach is then multiplied with the inventory elementary flows of water consumption (Milà i Canals et al. 2008; Pfister et al. 2009) or water withdrawal and release (Boulay et al. 2011b) (Fig. 1.3). See supplementary information for a detailed description of characterisation factors and the original water indices they rely on.

### 1.4.2. Water scarcity indicators vs. water availability indicators

To date, only the Water Impact Index (Bayart et al. 2014) and Boulay et al.'s (2011b) indicators are water availability indicators (i.e. considering that both water quantity and quality contribute to water deprivation). However, they do not model the same impact pathway since Bayart's method assesses the effect of a change in quality based on environmental standards for ambient water, whereas Boulay's method assesses the effect of a change in quality based on the functionality of water for human users (Boulay et al. 2015c). Bayart's method is ecosystem-oriented and based on a distance-to-target approach (acceptable water pollution level), whereas Boulay's method is human-oriented (functionality of water for human uses).

Since water quality degradation also contributes to water availability (ISO 14046 2014), the split between water quality & quantity for other methods such as Pfister et al. (2009) and Berger et al. (2014) is questionable. For example, Pfister et al. (2009) consider that a quality alteration do not affect water deprivation. This viewpoint divergence is important since Boulay et al. (2015c) demonstrated that calculating midpoint indicators based either on scarcity or on availability can greatly influence the results.

### 1.4.3. Open questions on characterisation factors

Regarding the CF's numerator, should scarcity be a function of water withdrawal or water consumption? (i.e.: a consumption-to-availability (CTA) or a withdrawal-to-availability (WTA) ratio?). Boulay et al. (2011b) and Berger and Finkbeiner (2012) support the use of a CTA ratio because the use of WTA tends to overestimate the water scarcity as it includes the non-consumptive use of water (e.g. cooling water, return flow from irrigation use). Moreover, the CTA ratio ensures consistency between the inventory and the impact assessment, as water consumption is multiplied by a CF also based on consumption (Kounina et al. 2013). Conversely, Pfister and colleagues support the use of a WTA. Since the water quality is not accounted for in the inventory, they are indirectly compensating it through the impact assessment: the water index is indirectly considering the water degraded in the numerator because water withdrawal includes water that may be released in the environment in a degraded state. On the contrary, Boulay et al. (2011b) clearly assess both quantity and quality in the inventory: the volume of input and output flows are multiplied with a CTA ratio specific to the water source and category (reflecting the quality) (Fig. 1.3). Payen et al. (2015) (see annexes) illustrated another issue related to CFs: when the activity under study represents a significant part of water usage in the watershed/country, its contribution to the total water consumption/withdrawal is not marginal anymore. This means that the product is directly affecting the CF which should ideally account for this significant contribution.

Regarding the CF's denominator: What is the available amount of renewable water? Most indicators only consider renewable groundwater recharge and surface runoff (Alcamo et al. 2003; Döll and Fiedler 2008) and neglect ground and surface water stocks (Berger and Finkbeiner 2012). Berger and colleagues filled this gap in their new indicator accounting for these water stocks (Berger et al. 2014). As already

explained, the method from Boulay et al. (2011b) also filled this gap but in a more relevant way: not only the ground and surface water stocks are accounted for in the CF's denominator, but surface and ground waters are also distinguished, each one having a specific characterisation factor, thus representing that different water sources have different availabilities.

#### **1.4.4. Spatial and temporal scales: consistency with the goal and scope of the study**

To convert the inventory into environmental impacts, it is necessary to consider the context of the studied system (the hydrological context, the water use competition...). The spatial resolution of the data (water scarcity indicators) and results (impacts) can be either at country, catchment or sub-catchment levels. Boulay et al. (2015c) showed that the scale at which the scarcity indicator is calculated significantly influences the indicators. Loubet et al. (2013) developed an indicator at the sub-catchment level to better account for the cascading effects that occur downstream of consumption in sub-basins, thus accounting for the location of water consumption within the basin. This spatial resolution allows considering that using water close to the outlet is less impacting (deprive less downstream users) than using water upstream at its source. Nevertheless, the question of the optimal scale remains (Boulay et al. 2015c).

Regarding the temporal scale, most water scarcity indicators are calculated at the annual scale, which seems not relevant for regions with contrasted dry and humid seasons, and more particularly for agricultural water use (Tendall et al. 2013). However, Pfister and Bayer (2014) and Hoekstra et al. (2012) developed monthly scarcity indicators. The relevant temporal scale depends on the goal and scope of the study, the studied system, and its location.

#### **1.4.5. Water indices versus fate and effect modelling**

The use of scarcity/availability indices in LCA is still debatable because the link between water scarcity, water deprivation (midpoint) and damages on the AoP still has to be demonstrated (Kounina et al. 2013). Indeed, water scarcity indices are not referring to any actual environmental mechanism such as the water cycle and the actual effect on human health and ecosystems. A few methods do not use the water indices for impact assessment but adopt a two-step modelling approach: first analysing the fate of the water flow, and second its resulting effects on the environment. Such an approach is compliant with the thorough impacts assessment scheme for an emission: fate, exposure, and effect modelling. The methods based on this framework are all endpoint-oriented: they address the effects of water use on the Ecosystems, splitting the CF in a Fate Factor (FF) and an Effect Factor (EF) (e.g. Amores et al. 2013; Verones et al. 2013a; Verones et al. 2013b; van Zelm et al. 2011). The fate factor could be considered as a midpoint indicator, but they are not comparable with one another (e.g.: wetland area change (Verones et al. 2013a), salt concentration change in the wetland (Amores et al. 2013)). Indeed, these methods are designed to provide endpoint indicators, the midpoint being only an intermediate in the assessment. But they present the advantage of describing the environmental mechanisms of the water use, contrary to the water index approaches. For example, Verones et al. (2013a, 2013b) addressed the damages of

consumptive water use onto wetland biodiversity, based on a fate factor relating water consumption to the loss of wetland area, and an effect factor quantifying the contribution of a loss in wetland area to potential global extinction of species (more details in supplementary information).

Payen et al. (2015) showed that midpoint assessment cannot clearly help identify the least environmentally-impacting option, between imported tomato from Morocco and locally produced tomato in heated greenhouses in France. As a result, there is a need to model the damages of water use on the three AoP as well, to allow a comparison of systems at endpoint level. However, most agricultural LCA studies do not calculate endpoint indicators, probably because of a larger perceived uncertainty of damage assessment.

### 1.5. Endpoint impact assessment: gaps and overlapping

Endpoint methods assess potential damages from water use on the AoP Human Health, Ecosystems and Resources (Fig. 1.2). Some cause–effect chains (pathways) are not yet covered by any methods, in particular pathways related to water quality degradation. Regarding the AoP Human Health, methods assess the impacts from a water deprivation on different sectors: agricultural users (Motoshita et al. 2010; Pfister et al. 2009), domestic users (Motoshita et al. 2011), and fisheries (Boulay et al. 2011b). The methods vary in terms of data sources for the definition of CF, socio-economic parameters accounted, and which users are considered affected (sectors are more or less sensitive to a water deprivation). A comparison of human health indicators showed that the results are greatly influenced by two model assumptions: the inclusion or not of trade effects (how food supply shortage in a country will spread to other countries through international trade) and the inclusion or not of the domestic sector as an affected user (for example, Pfister et al. (2009) considered that water deprivation generally did not affect domestic users) (Boulay et al. (2015a and 2015c)). The AoP Ecosystem received much attention with many methods addressing the damages of water consumption or degradation on aquatic and/or terrestrial ecosystems. However, these methods do not cover all cause-effect chains, and cannot be used in a complementary way because they are incompatible in their current forms. Their integration into one consistent indicator would require a harmonization process (Núñez et al. 2015).

Conversely, the AoP Resource received little attention and is not sufficiently developed (Kounina et al. 2013).

The framework proposed by the WULCA group considers that only fossil water use or overuse of renewable water can affect the Resources for future generations (Fig. 1.4). But there is a lack of clear definition of what a renewable use of water is? What is the threshold above which a water body should be considered as overused? Since the renewability rate of a water body depends on many local-specific factors, defining such a threshold is complex and will be limited by a lack of data on the state of groundwater resources. Another important aspect regarding the AoP Resource is that a loss of water quality is not considered as affecting the Resource (Fig. 1.4). Yet, some situations exist where a water body may be polluted almost irreversibly (e.g: a permanently salinised deep aquifer due to a saline

intrusion). In contrast, the impacts of water quality decrease on human health and ecosystem quality are considered as existing cause-effect chain (Fig. 1.4).

It has been argued that accounting for water quality in water use impact assessment could lead to double counting with indicators of water pollution (Berger and Finkbeiner 2012). But this is rarely the case since: (i) when water is not drinkable, human may not drink water so the ingestion route of exposure to the contaminant do not occur, (ii) the pathway leading to human health damages from water use refers to biological contamination and hygiene rather than toxicity (Boulay et al. 2015c). In the situation where toxic water was drunk, human would not suffer from water deprivation but from toxicity. Nevertheless, we cannot make the exact same reasoning for the AoP Ecosystems. Contrary to human who can (more or less) decide to use a type of water, ecosystems have to endure a type of water quality, in term of their living environment (water as a compartment), and in term of drinkable water (water as a resource). Thus, the boundary between water-compartment and water-resource is thinner for Ecosystems than for human. As a result, the double counting risk between indicators of water pollution and indicators of water availability is real regarding the impacts on Ecosystems.

To conclude, a comprehensive assessment of water use damages at endpoint level is not possible so far, but the WULCA group is working in this direction.



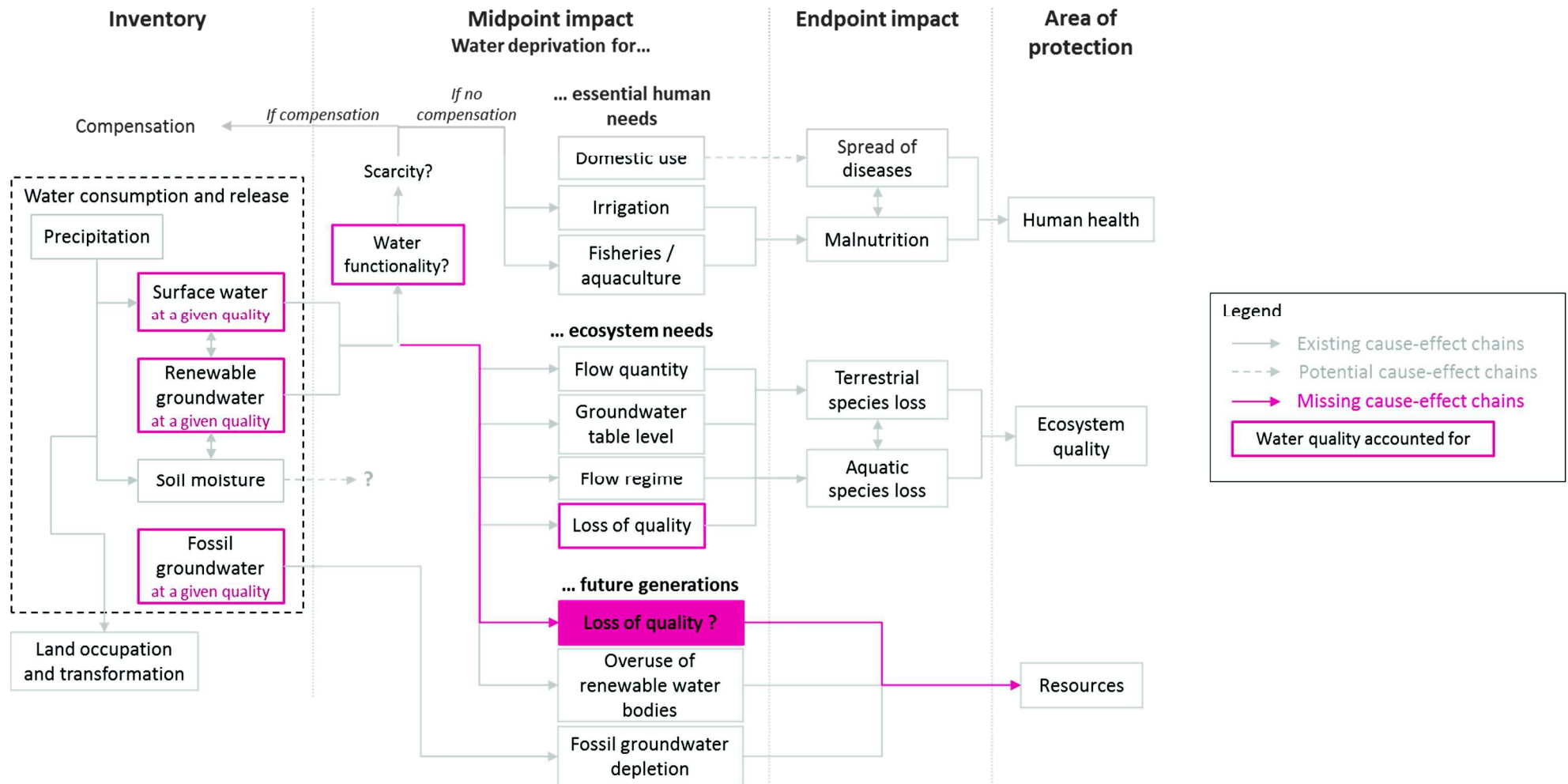


Figure 1.4. Cause-effect chains from the inventory to the areas of protection of human health, ecosystem quality, and resources (adapted from Bayart et al. 2010 and Kounina et al. 2013). The pathways considering water quality degradation are emphasised in pink colour. One potential missing cause-effect chain from the framework described by Kounina and colleagues (2013) is the loss of water quality damaging the AoP Resource.

## 1.6. Operationalization

The framework for the assessment of water use impacts in LCA is evolving quickly. Next efforts should focus on the improvement of environmental pathways coverage and method harmonization. The WULCA group is a major workforce contributing to this evolution.

We will now analyse the operationalization level of this framework: how can one calculate impacts from water use in the LCA of agricultural products/systems with current methods, databases, tools and software?

The operationalization of water impact assessment methods relies on their integration into commercial LCA software platforms, such as Simapro (PRé Consultants 2011) or Gabi. Such platforms do not allow for regionalized assessment of input and output water quality, so this is not possible so far (Boulay et al. 2015a). In Simapro 3, calculating water consumption impacts is possible only for water scarcity (i.e.: not accounting for water quality alteration) and at the country scale. As a result, applying the recent methods requires the use of Microsoft<sup>®</sup> Excel coupled with commercial LCA software.

Among the available methods, Boulay's approach is the most comprehensive and consistent method for the inventory and the midpoint impact assessment of water-related impacts since it addresses impacts associated with both water quantity loss and water quality alteration. Indeed, it explicitly assesses both water quantity and quality in the inventory: the volume of input and output flows are multiplied by a consumption-to-availability ratio specific to the water source and category (reflecting the quality). One limitation of this approach is that, up-to-now, the endpoint assessment is specific to Human Health, and the midpoint indicator is human-oriented (based on water functions for human). But one could imagine expanding this method to water functions for Ecosystems (but with double counting risks with indicators of water pollution: Cf. 1.5.).

In spite of its relevance, an analysis of recent literature on agricultural LCA revealed that Boulay's approach has not been applied so far. Conversely, the method proposed by Pfister et al. (2009) is used by LCA practitioners world-wide to assess cropping systems (Antón et al. 2014; de Figueirêdo et al. 2014; Quinteiro et al. 2014; Schmidt 2015), or animal production systems (Huang et al. 2014b; Huang et al. 2014a; Zonderland-Thomassen et al. 2014). Pfister's approach has been preferred probably due to its relative simplicity and the availability of regional water scarcity data, whereas the application of Boulay's approach requires more data on water quality that sometimes cannot be collected as part of LCA study (e.g.: Huang et al. 2014a).

Indeed, to implement the method of Boulay and colleagues, quality and quantity information on water entering and leaving each process is required which is challenging in agricultural LCA.

Although the quantity of water withdrawn is usually measured by the farmers, its quality is not always analysed. The water quality can be drawn from the classification proposed by the authors (based on GEMStat data (UNEP 2009)). Yet, the estimation of the released water flow, in terms of quantity and quality, is more complex. Regarding the volume of the released flow, the Water Database (Quantis 2015) provides the default returned water for a wide range of product/processes. However, if this database is suitable for inventorying the water elementary flows of background processes, it is not suitable for

studies on the comparison of agricultural practices including water management. If the quantity of water released is relatively easy to measure in industrial processes, this is more complex in agricultural cultivation (Tendall et al. 2013), because the irrigation return flow (part of applied water that is not consumed and that either drains or runs off) depends on many parameters of soil, climate and practice. Regarding quality of the released flow, Boulay and colleagues (2011b) recommended collecting data on water quality from the amount of chemicals emitted into water available in database such as Ecoinvent. This might be suitable for background processes, but for when agricultural systems are at the foreground, such data is not available since this is part of the inventory to estimate the emissions occurring on-field. The quality of water released in the environment by agricultural activity is crucial as it may contain pesticides, nutrients and other pollutants.

### **1.7. Water is a resource, but also a vector of pollutants, nutrients and salts**

Pesticides, nutrients and salts emissions are determined by water flows, soil, climate and agricultural practices. As a result, modelling emissions is a challenging task in LCA, and has been the purpose of several guidelines to help practitioners choose the best available methods. This is the case of the series of guidelines written by the Ecoinvent team or more recently the methodological report of the World Food LCA Database (Nemecek et al. 2015). However, in non-temperate contexts and for horticulture crops, available methods are inadequate (Bessou et al. 2013; Milà-i-Canals et al. 2008; Perrin et al. 2014). Models do exist to account for soil-plant-climate-cropping practices relationships and parameters, and to calculate emissions to the environment. But first they might prove quite complex to use and second, none of them include in a consistent way for the estimation of all inventory flows: water, nutrients, pesticides and salts flows.

Regarding pesticides, constant emission factors independent of soil, climate and practice conditions are mostly used in practice. The Pest-LCI model was the only model attempting to account for practice, soil and climate in the pesticides partitioning toward environmental compartments (air & water) (Birkved and Hauschild 2006; Dijkman et al. 2012). An international working group is currently working on the development of a consensus and default emissions factors for pesticides accounting for climate, soil and application mode (Rosenbaum et al. 2015). One additional output from this working group could be a drastic update of the PestLCI model. Regarding nutrients, STICS and STICS-derived models may be used to evaluate nitrogen emissions in contexts where default emission factors are not valid (Perrin 2013), but the application of such models is data and time-consuming and will not be associated to a proper validation due to a lack of data. SALCA-P is a possible model to estimate P loss from the field (Koch and Salou 2013).

Water flows constitute also a key driver of salts transport and accumulation in soil or water. But salts are rarely accounted for in LCA. Indeed, in a context of increasing refinement of land use and water use impact assessment methods in LCA, little attention has been given to salinisation impacts modelling. However, salinization is a global issue, which is not only threatening the agricultural production (Aragüés et al. 2011), but also biodiversity (Williams 1999). Payen et al. (2015) showed the importance of salinisation impacts in Morocco, and the shortcomings in the current LCA methodology to account for

these impacts. Indeed, as in many other areas in the world, the sustainability of agricultural production in South-western coastal Morocco is threatened by the salinization of the aquifer, due to its overexploitation. Thus, the impacts of crop production consuming this water (tomato in this case study) are certainly greater than assessed with the available methods (Payen et al. 2015).

Regarding the assessment of water flows, several LCA studies calculate the volumetric amount of water consumed by a crop using the CropWat model (Allen et al. 1998). This volume can be weighted by a water scarcity indicator for impact assessment (e.g.: Faist Emmenegger et al. 2011; Milà i Canals et al. 2008; Pfister et al. 2011; Pfister et al. 2009; Ridoutt et al. 2010). However, the water released (irrigation return flow) and its quality are not addressed. Quantifying the water released in the environment (qualitatively and quantitatively, in liquid form) is a complex and crucial task, particularly for agriculture where such water takes different forms and routes: deep infiltration, surface runoff, direct agricultural return flow.

As a result, since pesticides, nutrients and salts flows are determined by water flows, there is a need to calculate a water balance to properly estimate them. Such a quantified balance will constitute an inventory modelling: linking the input to the output flows via a model, with the aim of feeding ultimately the life cycle impact assessment methods.

## 1.8. Thesis specific objectives

The initial question was: “**How to assess the impacts associated with water use in agricultural LCA?**”

The bibliographic review showed that there are shortcomings at both the inventory and the impact assessment stages for the impact assessment of water use impacts in agricultural LCA. However, it is relevant to first focus our efforts on the inventory stage because: (i) Inventory is a prerequisite to impact assessment: an inventory of field water flows is the first step of an assessment of water-related impacts, (ii) on-going works are harmonizing the impact assessment approaches, (iii) inventory modelling is not currently adapted to recent methodological developments (in particular to implement Boulay et al 2011b), (iv) the fact that water is involved in all field emissions and a vector of pesticides, nutrient and salts is captured at the inventory stage. However, to properly position this work on the inventory stage, a general framework for the inclusion of salinization impacts in LCA is required.

This doctoral work and thesis aim to answer the following questions:

➤ **How to better assess the impacts associated with water and salts fluxes?**

As a requirement for the conception of inventory modelling, a preliminary study will be performed in **chapter 2** to assess how LCA should account for salinization impact in a complete and consistent way, from the inventory to the endpoint impact assessment, on the three areas of protection. Based on the joint work of experts from the salinization field and LCA researchers, this framework definition will allow identifying the inventory requirement for salts flows.

➤ **How to perform a relevant inventory of water and salts flows involved in agricultural systems?**

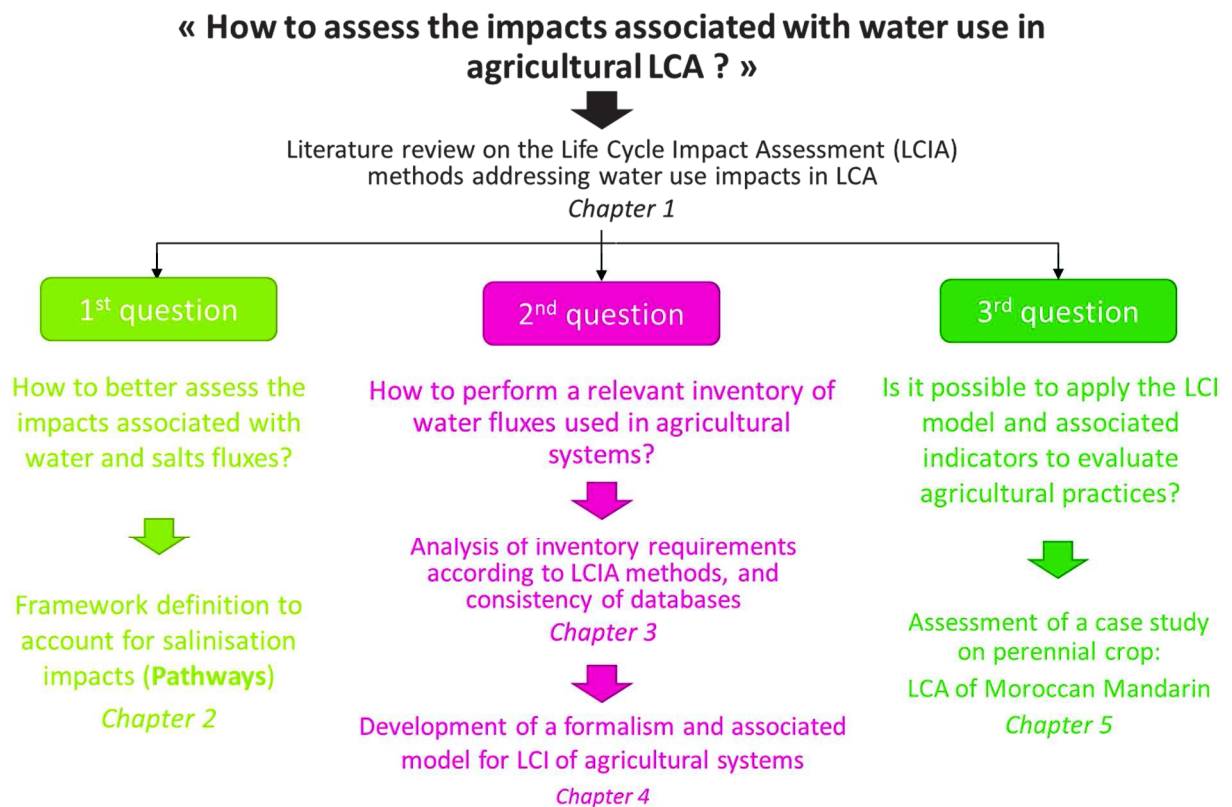
An in-depth analysis of available databases and recommendations for a consistent inventory modelling will be provided in **chapter 3**. We will explore possible modelling approaches for an inventory of field

water flows in LCA, aiming at discriminating cropping system practices: linking the input flows of water and salts, with the output flows, accounting for soil-plant-climate-agricultural practices. The modelling will be further investigated in **chapter 4** with the development of model formalisms for inventory of all agricultural systems, to cope with a lack of model appropriate for perennials.

- **Is it possible to apply the inventory model developed and associated LCIA models to evaluate agricultural practices?**

Based on a case study of Mandarin in Morocco, the applicability of the inventory of water and salts flows will be tested and discussed in **chapter 5**. Integrated in a full LCA study, water use impacts assessed through different methods will also be compared and discussed in chapter 5. The Moroccan Mandarin was selected as a case study because: (i) this perennial crop suffers from salinisation in a context where water is becoming scarcer, (ii) a local production company was motivated to contribute to this research by providing data and expertise, (iii) perennials constitute the most difficult possible testing for such an exhaustive LCA study.

To sum up, this doctoral thesis provides answers to the following questions, through the successive chapters:



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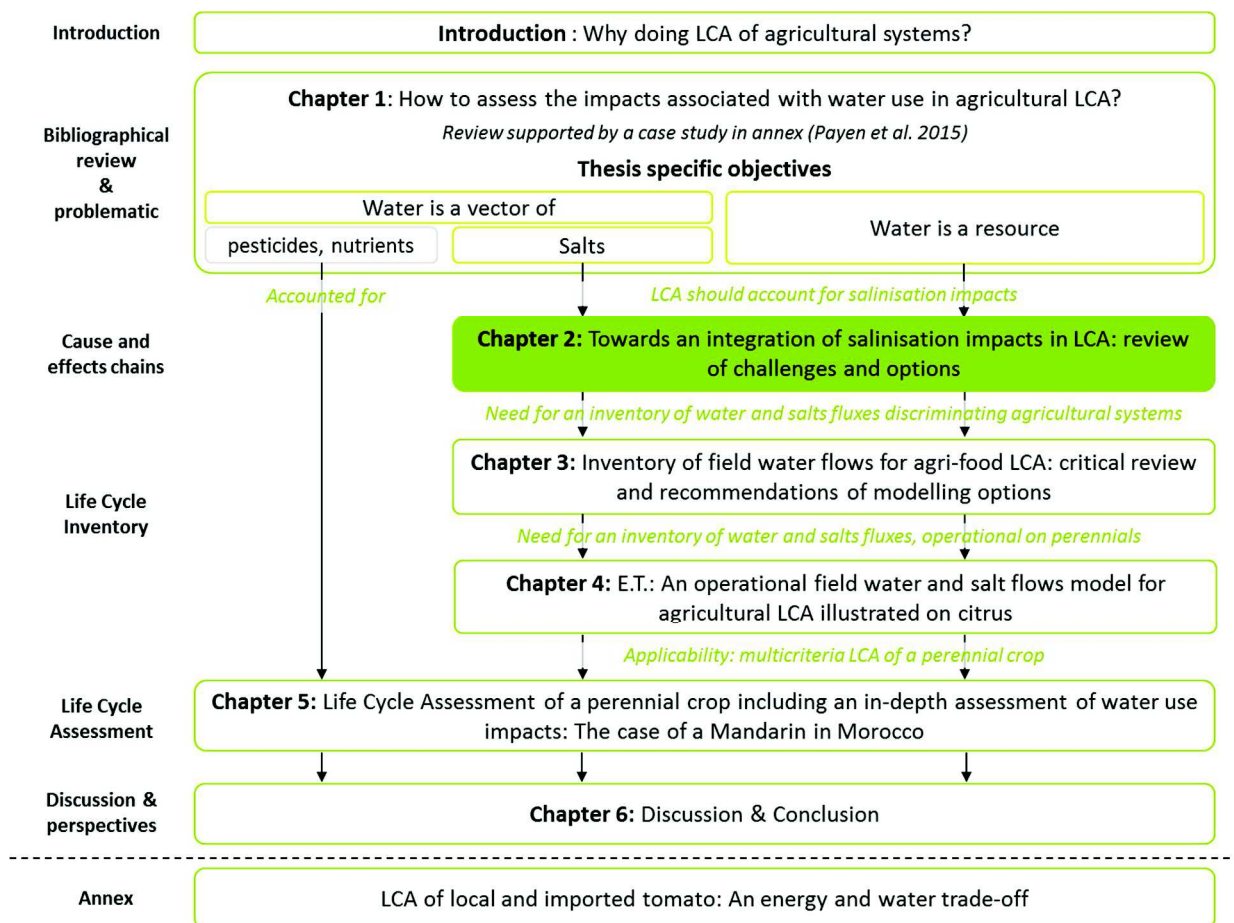
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Payen et al. (2015) showed the gap in the LCA methodology to account for salinization impacts. Furthermore, it sounds relevant to analyse the modelling options and identify the inventory requirements before tackling the questions related to the inventory of water and salts fluxes. Chapter 3 is a review of challenges and options towards an integration of salinisation impacts in LCA. This work is the result of a joint work of LCA researchers and experts in the field of salinization, notably thanks to two main workshops organized in Montpellier in June and December 2013.



## Chapter 2

### Salinisation impacts in Life Cycle Assessment: a review of challenges and options towards their consistent integration

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Sandra Payen<sup>1,2</sup>, Claudine Basset-Mens<sup>2</sup>, Montserrat Núñez<sup>3</sup>, Stéphane Follain<sup>4</sup>, Olivier Grünberger<sup>5</sup>, Serge Marlet<sup>6</sup>, Sylvain Perret<sup>7</sup>, Philippe Roux<sup>3</sup>

<sup>1</sup> ADEME, 20 avenue du Grésillé, F-49004 Angers, France

<sup>2</sup> CIRAD, UPR Hortsys, ELSA – research group for environmental life cycle sustainability assessment, Boulevard de la Lironde, F-34398 Montpellier, France

<sup>3</sup> IRSTEA, UMR ITAP, ELSA – research group for environmental life cycle sustainability assessment, 361 Rue Jean François Breton, F-34196 Montpellier, France

<sup>4</sup> Montpellier SupAgro, UMR LISAH, place Pierre Viala, F-34060 Montpellier, France

<sup>5</sup> IRD, UMR LISAH, place Pierre Viala, F-34060 Montpellier, France

<sup>6</sup> CIRAD, UMR G-Eau, INRGREF, Rue Hédi Elkarray, Tunis, Tunisia

<sup>7</sup> CIRAD, UMR G-Eau, F-34398 Montpellier, France

#### Abstract

**Purpose:** Salinisation is a threat not only to arable land but also to freshwater resources. Nevertheless, salinisation impacts have been rarely and only partially included in Life Cycle Assessment (LCA) so far. The objectives of this review paper were to give a comprehensive overview of salinisation mechanisms due to human interventions, analyse the completeness, relevance and scientific robustness of existing published methods addressing salinisation in LCA, and provide recommendations towards a comprehensive integration of salinisation within the impact modelling frameworks in LCA. **Methods:** First, with the support of salinisation experts and related literature, we highlighted multiple causes of soil and water salinisation and presented induced effects on Human health, Ecosystems and Resources. Second, existing Life Cycle Impact Assessment (LCIA) methods addressing salinisation were analysed against the International Reference Life Cycle Data System analysis grid of the European Commission. Third, adopting a holistic approach, the modelling options for salinisation impacts were analysed in agreement with up-to-date LCIA frameworks and models. **Results and discussion:** We proposed a categorization of salinisation processes in four main types based on salinisation determinism: land use change, irrigation, brine disposal and overuse of a water body. For each salinisation type, key human management and biophysical factors involved were identified. Although the existing methods addressing salinisation in LCA are important and relevant contributions, they are often incomplete with regards to both the salinisation pathways they address and their geographical validity. Thus, there is a lack of a consistent framework for salinisation impacts assessment in LCA. In analysing existing LCIA models we discussed the inventory and impact assessment boundary options. The Land Use/Land Use Change framework represents a good basis for the integration of salinisation impacts due to a Land Use Change but should be completed to account for off-site impacts. Conversely, the Land Use/Land Use Change framework is not appropriate to model salinisation due to irrigation, overuse of a water body and brine disposal. For all

salinisation pathways, a bottom up approach describing the environmental mechanisms (fate, exposure and effect) is recommended rather than an empirical or top-down approach because (i) salts and water are mobile and their effects are interconnected (ii) water and soil characteristics vary greatly spatially (iii) this approach allows the evaluation of both on- and off-site impacts and (iv) it is the best way to discriminate systems and support a reliable eco-design. **Conclusions:** This paper highlights the importance of including salinisation impacts in LCA. Much research effort is still required to include salinisation impacts in a global, consistent and operational manner in LCA, and this paper provides the basis for future methodological developments.

**Keywords:** Salinisation, Life Cycle Impact Assessment, Life Cycle Inventory, Land Use Change, Irrigation, Soil, Water, Resource

## 2.1. Introduction

Salinisation is the process leading to the accumulation of salts, not exclusively sodium chloride as it is frequently assumed, but also many other types of salts ( $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{K}^+$ ,  $\text{Na}^+$ ,  $\text{Cl}^-$ ,  $\text{SO}_4^{2-}$ ,  $\text{HCO}_3^{2-}$ ,  $\text{CO}_3^{2-}$  and  $\text{NO}_3^-$ ) (Rengasamy 2010). The salinisation process is commonly categorized in primary and secondary salinisation. Primary salinisation refers to salinisation processes mobilizing natural salts (naturally present in the environment), while secondary salinisation refers to salinisation processes increased or induced by human activity (also called anthropogenic salinisation). Although we commonly consider salinisation problems to be limited to arid and semi-arid regions (Rengasamy 2006), no climatic zone is free from salinisation. Salinisation is a worldwide problem affecting various land use types: both agricultural and non-agricultural areas, both irrigated and non-irrigated lands can be prone to salinisation (Wood et al. 2000). FAO estimates that  $83 \times 10^5 \text{ km}^2$  are affected by salinity, including  $34 \times 10^4 \text{ km}^2$  of irrigated land, and  $60\text{-}80 \times 10^4 \text{ km}^2$  affected by waterlogging and related salinity (FAO 2011). According to FAO (2003), soil salinisation is considered the second largest cause of land degradation from (and for) agricultural production. Soil salinity is a major issue because it adversely affects crop production, threatening agricultural sustainability (Aragüés et al. 2011). Moreover, salinisation is a threat not only to arable land but also to water resources (freshwater lakes and wetlands, rivers and streams) (Williams 1999). Indeed, secondary salinisation is impacting water resources in almost one-third of the world's land area. This extent is likely to increase, in particular because of global climatic change: notably through higher evaporation rates and temperatures increasing surface water salinity, and higher water demand for crop production increasing the salts brought in the soil profile due to irrigation (Duan and Fedler 2013). Since global climatic change causes are anthropogenic, the associated salinisation can be considered as secondary salinisation. Whatever the cause, the effects lead to harmful economic, social and environmental impacts (Williams 1999).

Life Cycle Assessment (LCA) is a method to quantitatively assess the environmental impacts of goods and processes from "cradle-to-grave" (Hellweg and Milà i Canals 2014). The strength of this environmental assessment tool is to identify possible burden shifting from one environmental impact category to another, by addressing all impacts occurring throughout the entire value chain. However, salinisation is missing in the range of impact categories of most LCA case studies (Finkbeiner et al. 2014). Because of the important environmental damages of salinisation, including this impact in LCA is considered a high

priority for research (JRC-IES 2011). Many LCA studies highlight this methodological gap for the environmental impact assessment of many technological processes: brine disposal from water desalination (Muñoz and Fernández-Alba 2008; Zhou et al. 2013a), water treatment processes (Friedrich and Pillay 2009), micro algae cultivation (Grierson et al. 2013), and, especially, agricultural products (Bartl et al. 2012).

Yet, only four methods are available to assess salinisation impacts in LCA: Amores et al. (2013); Feitz and Lundie (2002); Leske and Buckley (2003; 2004a; 2004b) and Zhou et al. (2013b). All methods were applied at least once in an applicability test performed by the authors, only the methods of Feitz and Lundie, valid in Australia, and the one of Amores et al. (2013), developed for a specific case in Spain, were applied in other case studies (Tangsubkul et al. 2005; Muñoz et al. 2010; Antón et al. 2014). Overall, they either focus on one salinisation type or on one specific geographical location. Therefore, there is a lack of a comprehensive approach to assess salinisation impacts due to human interventions in the LCA framework.

Our objective is to provide the scientific basis to build a relevant and complete model to assess salinisation impacts in LCA. This was done following the guidelines from Cucurachi et al. (2014), Huijbregts (2013) and Jolliet et al. (2014) for the development and critical evaluation of Life Cycle Impact Assessment (LCIA) methods. A three-steps approach was adopted (1) the setup of a comprehensive and structured overview of anthropogenic salinisation mechanisms and cause and effect chains. For this overview, we collected evidence from the literature as a measure of the importance and priority of salinisation, with support from specialists in salinisation outside the LCIA field; (2) the critical analysis of the LCIA methods modelling salinisation impacts according to the criteria proposed by the Institute for Environment and Sustainability from the European Commission (JRC-IES 2011); (3) the identification of the methodological issues and recommendations to build a consistent framework for including salinisation impacts in LCA. At this stage, recommendations are mostly of conceptual nature; operationalisation will be the aim of future research.

## **2.2. Salinisation environmental mechanisms**

The detailed analysis of the salinisation environmental mechanisms is relevant to highlight the processes involved in salinisation impacts due to human interventions. LCA addresses impacts of human interventions. Therefore, we decided to focus on anthropogenic salinisation (secondary salinisation).

### **2.2.1. Salinity**

Salinisation is the accumulation of salts. The major cations involved are sodium, calcium and magnesium and the major anions are chloride, sulfate and carbonate (Rengasamy 2010). Salinity refers to the total concentration of these salts in both soil and water samples, and is measured with the Electrical Conductivity (EC, in siemens per metre) of water or a soil saturated extract. EC is strongly correlated with the ions charges and Total Dissolved Solids (TDS) in soil water (the liquid phase of soils) (Corwin and Lesch 2005). The nature of salts involved is also important: when the sodium is in excess, an additional

process may occur: soil sodification (Ghassemi et al. 1995). Sodification is the accumulation of sodium on the soil exchange complex causing soil clays dispersion, responsible of soil structure degradation. Sodic conditions are characterized by the Exchangeable Sodium Percentage (ESP, dimensionless): the amount of sodium held in exchangeable form on the cation exchange complex, measured in soil extracts, and the Sodium Adsorption Ratio (SAR, dimensionless) measuring the relative preponderance of dissolved sodium in water compared to the amounts of dissolved calcium and magnesium (Rengasamy 2010; USDA 1954). Several classifications exist for salt-affected soils (e.g. Rengasamy 2010). Depending on the classification system, the SAR and EC thresholds values are not the same (Rengasamy 2006). From an operational point of view, farmers usually classify irrigation water according to EC measurements and crop sensitivity (USDA 1954).

### **2.2.2. Human interventions causing soil and water salinisation**

Soil and water salinisation are often studied separately: “Salinisation is the process that increases the salinity of inland waters” (Williams 1999). “Salinisation is an accumulation in the soil of dissolved salts” (Wood et al. 2000). But soil and water salinisation are inter-related, water being the agent for salt movement (Grundy et al. 2007). Salts are conservative and resistant to degradation (Schnoor 2013) but they are mobile: they can either stay in a soil at a given location or migrate with water. We distinguished four main patterns of salinisation due to human interventions associated with land use change, irrigation, brine disposal and overuse of a water body. Herein, we identified the biophysical and human management factors responsible for both soil and water salinisation for each type.

#### **Salinisation associated with land use change**

Land Use Change (LUC) modifies hydrological processes and therefore the water cycle at the catchment scale. In particular, clearance of deep-rooted perennial native vegetation and replacement with shallow-rooted crops that decrease transpiration rates and increase water infiltration rate in the vadose zone. As a consequence, saline groundwater tables can rise and reach the near soil surface in lowlands. This leads to the soil salinisation through capillary rise (Williams 1999) or artesian flow (Hammecker et al. 2012). In addition, percolation of salts can contribute to increase the salinity of the aquifer as it was observed in Australia (Williams 1999; Grundy et al. 2007; Scanlon et al. 2007) or in Thailand (Williamson et al. 1989; Hammecker et al. 2012) and the USA (Black et al. 1981; Scanlon et al. 2007). This salinisation type involves specific biophysical factors such as topography, precipitations, groundwater table level, soil geochemical and hydrodynamic profiles, and salt stock in soil; but also management factors such as a land use change modifying the evapotranspiration rates (Table 2.1).

#### **Salinisation associated with irrigation**

Irrigation and fertilization can cause soil and water salinisation. Salts provided by irrigation water have a higher tendency to accumulate in the soil in semi-arid and arid areas because of the conjunction of low



rainfall and high evapotranspiration rates (Marlet and Job 2006). Irrigation water always contains some salts, but the use of low quality water (e.g. treated wastewater) to compensate the increased scarcity of freshwater might worsen salinisation (Duan and Fedler 2013). Salts are also present in fertilizers (Scanlon et al. 2007), so fertilizers applications influence the salinity of the soil (Boman and Stover 2012). On the one hand fertilizers may cause salinity increase, on the other hand, their appropriate management helps cope with saline conditions. The development of irrigation affects the local geohydrological regime, mobilizes salts stored in the underlying substrate (Smedema and Shiati 2002) and favors salts leaching from the root zone to water bodies or underlying groundwater (Mateo-sagasta and Burke 2010). Finally, if irrigation overcomes drainage capacities, the rise of the groundwater table causes soil salinity issues through capillary action (Corwin and Lesch 2005). The rise of saline ground waters may in turn induce salinisation of some fresh waters (Williams 2001). Subsequently, poor irrigation management and inadequate drainage often lead to salinisation and waterlogging (Wood et al. 2000). Thus, there are tradeoffs, notably between salinisation of the soil in the case of insufficient salt leaching, and salinisation of the underlying aquifer if salts are leached. There are two scales involved in this salinisation context: a regional and a local one. If the spatial structure of groundwater tables is regional, associated salinisation processes act at a local scale (plot/farm). The embedded biophysical factors of this salinisation context are the water table depth, soil hydrodynamic profile, precipitation, evapotranspiration rates and salts reservoirs in the soil. The management factors are salts content in irrigation water and fertilizers, irrigation volume and drainage rates (Table 2.1).

### **Salinisation associated with brine disposal**

Many activities generate saline wastewater: e.g. mining, pumping of shallow saline aquifer and seawater desalination, therefore the problem of brine disposal is raised (Williams 2001). This is a topical question to address while many countries need to complement their water supply with seawater desalination (Zhou et al. 2013b). In coastland desalination plants, brine may be discharged in seawater (impacting the marine ecosystem), whereas in inland areas, brine discharge is more problematic because diluting brine in a water stream or discharging it directly in the soil may lead to water and soil salinisation (Sánchez et al. 2015). New alternatives are studied, such as the use of brine water for agricultural use, in combined scheme (e.g. microalgae cultivation, fish production and halophyte forage scrub irrigation). But these alternatives do not prevent the gradual salinisation of land (Sánchez et al. 2015). Brine disposal is a major cause of aquatic ecotoxic impact, and the subject is of growing interest in research (Zhou et al. 2014). This salinisation type highly depends on the salts composition of the brine, and the discharge location. Salinisation due to brine disposal will be driven by many biophysical factors relying on the geographical features of the discharge context, actually all biophysical factors identified for the other salinisation types.

### **Salinisation associated with overuse of a water body**

In many coastal areas, excessive withdrawal of groundwater and/or rivers streams leads to seawater intrusion: the decrease of the coastal aquifer table level induces seawater inflow in the aquifer, leading

groundwater to long-term salinisation (Flowers 1999; Scanlon et al. 2007; FAO 2011). The depth of the interface between freshwater and seawater is reduced when the aquifer table is decreased as illustrated by the Ghyben-Herzberg formula, a linear relationship often used to simulate seawater intrusion (for a review of methods investigating seawater intrusion processes, see Werner et al. (2013) and Sreekanth and Datta (2015)). In the estuaries and deltas, seawater intrusion happens when the freshwater flow of the river is reduced because of excessive water withdrawal upstream or the construction of impoundments (Williams 2001; FAO 2011). Sea-level rise induced by climate change is an aggravating factor of seawater intrusion (FAO 2011). In non-coastal areas, saline intrusion may result from saline water transfer from a saline aquifer to an overused aquifer. This type of salinisation happens when too much water is withdrawn from a water body, independently of the usage. However, irrigation is the principal cause because 70% of all water extraction worldwide is devoted to agricultural use (World Water Assessment Program 2009). Salinisation associated with saline intrusion involves mechanisms at the regional (e.g. fluctuating sea level) and local (e.g. well) scales (Werner et al. 2013). The biophysical factors involved are the distance to the coast or estuary, and the presence of saline aquifer. The management factors are the volume of freshwater withdrawal and the exploitation rate of the water body (river or aquifer) (Table 2.1).

**Table 2.1. Key management and biophysical factors involved in secondary salinisation, per salinisation type.**

<b>Salinisation type</b>	<b>Spatial scale</b>	<b>Management factors</b>	<b>Biophysical factors</b>
Land use change	Hydrogeological catchment	Land use transformation: ET rates modification.	Topography Soil geochemical and hydrodynamic profile Salt reservoir in soil Water table depth Precipitation
Irrigation (w or w/o shallow groundwater or poor drainage)	Local (field) within regional context (groundwater)	- Volume of irrigation water - Salts in water - Salts in fertilizers - ET rates - Drainage rates (and irrigation mode)	Soil hydrodynamic profile Precipitation Salt reservoir in soil Water table depth
Brine disposal	Local (discharge location) within regional context	- Salts in water - Discharge location	Many factors involved, relying on the geographical features of the discharge context
Overuse of a water body	Surface and underground water catchments	- Volume of water withdrawn - Water body exploitation rate	Distance to the coast or estuary Presence of saline aquifer

### 2.2.3. Water and soil salinisation damages to Ecosystems, Human health and Resources

Salinisation of soils and waters affects and ultimately damages the so-called Areas of Protection (AoP) commonly used in LCA, i.e. Ecosystems (or natural environment), Human health and Resources (Dewulf et al. 2015). Figure 2.1 depicts the salinisation environmental mechanisms (or cause-effect chains).

Soil salinisation not only affects terrestrial ecosystems and crop growth, but also degrades land more or less permanently (D'Odorico et al. 2013). Salt-affected soils have a lower fertility through three potential effects on plants: i) reduction of plant water uptake or dry out by lowering the osmotic potential, ii) toxic effect by different ions depending on the soil pH, and iii) plants nutrients uptake imbalance (Flowers and Flowers 2005). Sodic soils also have effects due to soil structure degradation and permeability reduction (Suarez et al. 2006; Rengasamy 2010; D'Odorico et al. 2013). Decrease of the soil fertility and thus of the yield production potential could result in malnutrition for poor populations (UNESCO 2003). Impacts of salinity on Ecosystems and Human health also include increased flood risk, and increased infrastructure failure risk (Grundy et al. 2007; Zhou et al. 2013a). Soil salinisation is also considered a driver for desertification and it is closely related to land degradation processes such as soil erosion and arable land abandon (D'Odorico et al. 2013). Although some measures to reduce soil salinity and sodicity can be employed, salinisation is considered irreversible in arid regions where there is not enough freshwater available to leach out the accumulated salts (Rozema and Flowers 2008), or in lowland areas of endorheic basins (i.e. closed drainage basin) with shallow and saline groundwater (D'Odorico et al. 2013). Land degradation due to salinisation might then be considered a damage to the soil resource. It is noteworthy that salinisation management techniques are simply shifting the problem by moving salts from one compartment (e.g. root zone) to another (e.g. ground water).

Salinisation of a water-body not only affects the aquatic and riparian ecosystems, but also reduces the water availability for further use. An increase of water salinity causes a change in the species composition of algae, zooplankton, and benthic communities and leads to the disappearance of macrophytes and riparian trees (Williams 1999; Schnoor 2013). It should be noticed that ecosystems may not lose diversity per se but evolve from a halosensitive biota (organisms sensitive to high salinity conditions) to a halotolerant one (organisms adapted to high salinity conditions) (Williams 1999). In addition, saline freshwater lakes, wetlands, rivers or aquifers are unfitted to serve as supplies for domestic, agricultural and other uses (Williams 1999; FAO 2011), thus resulting in water deprivation for humans and ecosystems. This quality alteration of the water resource may be irreversible, for example for a permanently saline aquifer, and thus affects the water resource for present and future generations.

### 2.2.4. Complexities related with salinisation in space and time

Salinisation processes are often inter-related (Williams 2001), and involve environmental mechanisms from different nested scales. Physico-chemical mechanisms stand at local scale and hydrological mechanisms at catchment scales. Nevertheless, the hydrological processes causing the salt mobilization are similar for all salinisation types (Zhou et al. 2013a). Although we can establish a typology of salinization contexts, in many cases the situation is complex because salinisation results from several

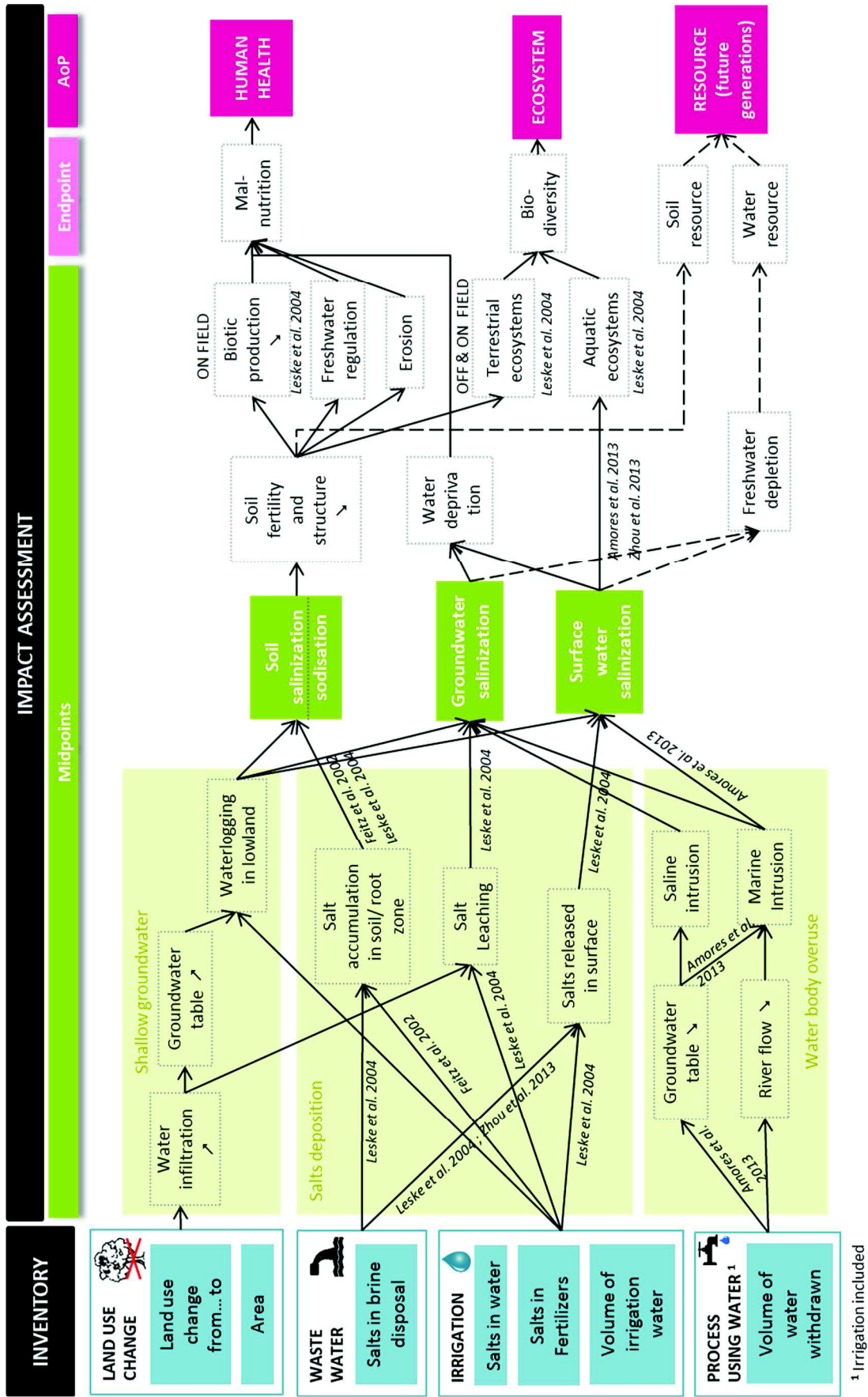


Figure 2.1. Human-driven salinisation environmental mechanisms and positioning of approaches proposed in the literature. Long dash lines represent controversial pathways in the scientific community.

causes. The combined effect of the replacement of natural vegetation by agricultural crops upstream, and the discharge of saline agricultural wastewater can lead to the salinisation of many freshwater lakes, wetlands and rivers (Williams 2001). Groundwater salinisation can be due to both seawater intrusion and the agricultural return flows (Bouchaou et al. 2008). In addition, water and soil salinisation are intimately related. The degradation of freshwater resources (surface or groundwater) has concomitant effects on the systems using these sources, and soil salinity affects in turn water resources (D'Odorico et al. 2013). The identification of the actual cause of salinisation is also difficult because its effects can be transferred in time and space from its causes (Grundy et al. 2007). It should be mentioned that other salinisation contexts with a narrower extent were not presented here (e.g. sea spray, deicing salt spreading on roads, industrial wastes). It is also important to specify that the descriptions of salinisation mechanisms provided in this article focus on the main processes involved, for the sake of clarity and concision. Since salinisation mechanisms are intimately related with the water cycle, all water flows can potentially have an influence on salinisation.

Salinisation impacts are demonstrated in literature. They are of crucial concern and the link between different human interventions and the salinisation impacts on the three AoP has been clearly established.

### **2.3. Critical analysis of salinisation impact assessment methods in LCA**

Four methods have been developed to assess salinisation impacts in the LCA framework so far (Table 2.2). These approaches are either midpoint oriented (Feitz and Lundie 2002), endpoint oriented (Zhou et al. 2013b; Amores et al. 2013), or near-endpoint oriented (Leske and Buckley 2003; 2004a; 2004b). To highlight their strengths and flaws, we analysed the methods against the criteria defined in the International Reference Life Cycle Data System Handbook procedure proposed by the Institute for Environment and Sustainability from the European Commission (JRC-IES 2011). The criteria are: completeness of scope, environmental relevance, scientific robustness and certainty, documentation, transparency and reproducibility, applicability and potential stakeholder acceptance. The detailed assessment is available in the supplementary information (Table S1).

#### **2.3.1. Salinisation associated with irrigation: Feitz and Lundie (2002)**

The midpoint soil salinisation potential developed by Feitz and Lundie (2002) assesses the propensity of irrigation water to damage soil structure and the accumulation of sodium in the soil, expressed in  $\text{Na}^+_{\text{eq}}$ . The inventory data requirements are the volume of irrigation water ( $V_i$ ) and the water sodium concentration ( $[\text{Na}]$ ). These parameters are multiplied by a soil sodiation hazard characterization factor (CF). The soil sodiation hazard is assessed through the ratio between the electrical conductivity threshold ( $\text{EC}_{\text{threshold}}$ ) representing the limit of soil structure integrity for a given Sodium Adsorption Ratio (SAR), and the EC of the irrigation water ( $\text{EC}_{\text{iw}}$ ). This method presents the advantage of assessing both salts accumulation in the soil and soil structure degradation. Apart from the applicability test done by the authors, two case studies applied this method: Tangsubkul et al. (2005) and Muñoz et al. (2010).

Muñoz and colleagues (2010) calculated, besides soil organic carbon deficit, soil salinisation potential as one indicator for soil quality impacts, to compare different water sources and water qualities for irrigation purposes. Tangsubkul et al. (2005) calculated the relative soil salinisation potential of the use of irrigation water from different water recycling technologies.

The main limitations of this method are: its restricted scope; limited to irrigated cropping systems, not prone to waterlogging, and its limited validity domain for the characterization factor, in spite of the detailed inventory data requirement. The indicator is based on a relatively ancient approach but very common and generally well accepted. However, the soil type is not accounted for, although soil texture is a key parameter in the sodicity sensitivity. For example, sandy soils do not have soil structural problems caused by high SAR, whilst clayey soils are likely to be sodic with soil structural problems (Rengasamy 2010). The indicator is only valid for soils within the validity domain of the electrolyte threshold curve, and the estimation of the Sodium Adsorption Ratio of the soil drainage water is assumed for an Australian red-brown earth. The applicability of the method may be hampered by the data requirement. Indeed, CFs have to be calculated by the practitioner because they depend on the quality (thus the composition) of the irrigation water used (Table 2.2).

### **2.3.2. Salinisation associated with overuse of a water body: Amores et al. (2013)**

Amores and colleagues (2013) evaluated the damages on biodiversity associated with a salinity increase in a Spanish coastal wetland. This salinity increase is caused by seawater infiltration in the wetland, after groundwater overexploitation for irrigation. The inventory data required is the volume of groundwater consumed by the studied crop ( $ET_{crop}$ ). To calculate the indicator result, the inventory is multiplied by the CF, composed by a fate and an effect factor. The fate factor represents the change in salt concentration in the wetland due to a change in groundwater consumption. It is calculated from yearly averaged seasonal water and salts balances for the wetland Albufera de Adra in the South of Spain (Table 2.2). The effect factor stands for the change in potentially affected fraction of native wetland species due to salinity increase. Species included in the assessment are plants, fishes, algae, and a crustacean. Apart from the applicability test done by the authors, the literature provides one case study: Antón et al. (2014) calculated damages on biodiversity from water consumption for tomato grown in the specific area where Amores and colleagues developed their method.

This method addresses a specific context of salinisation associated with overuse of a water body: seawater intrusion in a wetland (fed by the groundwater), but does not consider a seawater intrusion in the aquifer. Although the approach could certainly be applied in other contexts, the main limitation of this method is its limited geographical validity. The fate factors are based on water and salts balances relying on the specific hydrological functioning of the wetland and local hydro-climatic parameters, and the effect factor is based on specific native species of the Albufera de Adra wetland. The common endpoint unit PAF allows comparison with other methods and impacts: in the tomato case-study the biodiversity loss in the wetland due to salinisation has the same order of magnitude than terrestrial acidification and terrestrial ecotoxicity impacts (Antón et al. 2014).

### 2.3.3. Salinisation associated with brine disposal: Zhou et al. (2013b)

Zhou et al. (2013b) propose a method for assessing aquatic ecotoxicity of brine disposal from seawater desalination plants. The aquatic ecotoxic impact is the sum of the impacts generated by groups of influential chemicals. The method consists in deriving specific aquatic ecotoxic potential CFs for each group of chemicals: metals, organic and inorganic chemicals. This approach is supported by freshwater ecotoxic characterization factors from the USEtox database. Due to the lack of fate models for the inorganic salts group, a whole-effluent approach is used instead of a chemical-specific approach: CFs are estimated based on a worst case scenario.

The inventory data required is the mass of chemical group “salinity” ( $m_{\text{salinity group}}$ ) in  $1\text{m}^3$  of brine, then multiplied by the CF, composed by a fate and an effect factor, to calculate the indicator result. The residence time of most persistent chemicals is used as a fate factor. But since the persistence time of  $\text{Na}^+$  ions (millions of years) exceeds the range of the acute test (100 years), the residence time of the second most persistent chemical in the brine mixture,  $\text{Cu}^{2+}$ , is used instead (Table 2.2). The effect factor is calculated based on a worst case scenario EC50 (Yoon and Park 2012). To our knowledge, no case study has used this method yet, apart from the applicability test done by the authors.

This group-by-group approach presents the advantage of including the contribution of inorganic chemicals such as salts, which are suffering from a lack of CFs in the usual LCIA models. However, this whole effluent approach may be associated with high uncertainty if the composition of the inorganic group is highly variable. The fate factor is based on the residence time of  $\text{Cu}^{2+}$  which does not belong to the salinity group: this metal is assessed with a chemical specific approach. Regarding the effect factor, the EC50 corresponds to the salinity concentration threshold for acute toxicity of brine on four phytoplankton (Yoon and Park 2012). The EC50 (referring to growth rate) values reported in this experiment range from 40.2 to 78.7 g/L. This high variability of the EC50 and the limited number of species considered (all marine), warrant the need to use a HC50 based on a wider range of aquatic species. Recent publications in the field may now allow the use of HC50, a better alternative to EC50 (Zhou, personal communication).

### 2.3.4. Salinisation associated with salt release: Leske and Buckley (2003; 2004a; 2004b)

Leske and Buckley (2003; 2004a; 2004b) developed a salinity impact category for South African LCA. They provide salinity potential CFs for salts release in the atmosphere, surface water, natural surfaces, and agricultural surfaces compartments. The CFs stand for potential effects on aquatic ecotoxicity, materials, natural wildlife, livestock, aesthetic effects, natural vegetation and crop. Inspired by a risk assessment approach, salts fate factors are calculated with an atmospheric and hydrosalinity catchment model. Effect factors are calculated using the predicted no-effect concentration for each environmental target (Table 2.2). This method is covering several salinisation contexts and pathways, accounting for both water and soil salinisations (Fig. 2.1). However, it does not cover salinisation induced by a LUC or a saline intrusion. The main limitation of this method is its geographical validity restricted to South Africa. Indeed, the fate factors are calculated with a catchment model for South Africa, and the effect factors

are based on the South African Water Quality Guidelines. It is noteworthy that the calculated CFs for salts emissions onto the agricultural soil by far outweigh the CFs for releases into other compartments. This warrants further research to better model agricultural systems. The near endpoint indicator units TDS cannot be compared with other methods as it is specific to salinity.

### **2.3.5. Lack of consistent frameworks**

The methods provide relevant methodological approaches to salinisation impact modelling and could certainly be inspiring in other contexts. Their main limitations though are their restricted scope in terms of pathways covered (Feitz and Lundie 2002), their intensive inventory data requirement (Feitz and Lundie 2002), or their restricted geographical validity (Amores et al. 2013; Feitz and Lundie 2002; Leske and Buckley 2003, 2004, 2004b). All methods have site-specific CFs, emphasizing that salinisation impacts are highly site-dependent; especially regarding the hydrology, the climate and irrigation water quality, but are not globally applicable. Applying these methods in other contexts, requires to redevelop the whole characterization approach: a new effect factor specific of the species in the studied wetland (Amores et al. 2013), using an equivalent country-specific catchment hydrosalinity model of the studied country (Leske and Buckley 2003; 2004a; 2004b). As a result, it would be time-consuming and data-intensive to adapt the methods to other contexts. Figure 2.1 positions the contributions of these approaches on the global salinisation cause-effect chains identified, showing that a consistent framework is missing (Finkbeiner et al. 2014).



Table 2.2. Inventory requirement, characterization factors and category indicator results of salinisation impact assessment methods in LCA

Article	Life Cycle Inventory (LCI)	Characterization Factor (CF)	Category indicator result
<b>Feitz and Lundie 2002</b>	<b>Vi:</b> Irrigation water volume (L), <b>[Na]:</b> water sodium concentration (mg/L).	<b>CF</b> = $EC_{\text{threshold}}/EC_{\text{iw}}$ With: $EC_{\text{threshold}}=0.121 \times \text{SAR} + 0.033$ equation representing the clay flocculation - dispersion threshold. SAR calculation requires: [Ca], [Mg], [SO <sub>4</sub> ], [CaCO <sub>3</sub> ], pH, EC <sub>iw</sub> of irrigation water. If CF<1, no soil degradation hazard from sodisation.	<b>Midpoint</b> $\sum_i CF_i \times [Na]_i \times V_i$ For irrigation water i. <b>Unit:</b> kg Na <sup>+</sup> <sub>eq</sub>
<b>Amores et al. 2013</b>	<b>ET<sub>crop</sub>:</b> Crop groundwater consumption (m <sup>3</sup> /yr) (= Evapotranspiration of the crop: ET <sub>crop</sub> in Amores et al. 2013)	<b>CF</b> = FF*EF=Change in Potentially Affected Fraction of species (PAF) due to a change in salinity due to a change in groundwater consumption: With: FF=Fate Factor = $\Delta FGW / \Delta ET_{\text{crop}} \times \Delta C_N \cdot V_N / \Delta FGW$ FGW: fresh groundwater inflow to Lagoon, C: salinity, V: volume of the lagoon, ET <sub>crop</sub> : crop ET, Δ: change between years EF=Effect Factor = $\Delta PAF_{\text{sal}} / \Delta C_N \cdot V_N = 0.5 / HC50_{\text{sal}}^1$	<b>Endpoint</b> ET <sub>crop</sub> · CF <b>Unit:</b> PAF·m <sup>3</sup> ·year, converted into species·year considering a 7.89 x 10 <sup>-10</sup> species·m <sup>-3</sup> freshwater species density
<b>Zhou et al. 2013b</b>	<b>m<sub>salinity group</sub>:</b> mass of chemical group “salinity” (Cl <sup>-</sup> , Na <sup>+</sup> , SO <sub>4</sub> <sup>2-</sup> , Mg <sup>2+</sup> , Ca <sup>2+</sup> , K <sup>+</sup> , HCO <sub>3</sub> <sup>-</sup> ) in 1m <sup>3</sup> of brine, (kg)	For chemical group “salinity”, based on a “whole effluent approach”: <b>CF<sub>salinity group</sub></b> = FF*XF*EF = 4.62E-01 PAF·m <sup>3</sup> ·day/kg With: FF=Fate Factor =37 days; residence time of Cu <sup>2+</sup> , based on USEtox fate model XF=eXposure Factor = 1; salts 100% dissolved in water EF=Effect Factor =0.5/EC50 = 1.25E-02; EC50 <sub>salinity</sub> =40,000mg/L	<b>Endpoint</b> $\sum_j m_{\text{salinity group}} \times CF_{\text{salinity group}}$ <b>Unit:</b> PAF·m <sup>3</sup> ·day
<b>Leske and Buckley 2003; 2004a; 2004b</b>	<b>TDS released:</b> Total Dissolved Salts released (kg) in a compartment	<b>CF</b> = Total Salinity Potential (TSP) = $\sum$ Potential effects on environmental target. With: Potential effect = $\sum_i^N PEC_i - PEC_i^0 / PNEC \cdot M$ PEC <sub>i</sub> : predicted concentration in the compartment during day i after an emission of total mass M; PEC <sub>i</sub> <sup>0</sup> : predicted concentration in the compartment during day i without an emission PNEC: predicted no-effect concentration N: days in the simulation	<b>Near-endpoint</b> TDS released x TSP <b>Unit:</b> kg TDS <sub>eq</sub>

<sup>1</sup>HC50: concentration at which ≥ 50% of the species are exposed to concentrations above their EC50 (concentration where a 50% reduction in a given endpoint (e.g., growth) is observed compared to the control)

## 2.4. Towards a consistent framework for salinisation impacts assessment in LCA: methodological issues and recommendations

### 2.4.1. Context of LCIA for assessing salinisation impacts

The purpose in this section is to analyse how salinisation impacts could be modelled within the methodological framework of LCA. Answering this question raises topical methodological issues.

Since salinisation may affect soil and water resources, both often considered as limited resources at least locally, there is a need to analyse the status of the AoP Resources in LCA. Dewulf et al. (2015) identified and discussed the different status of the AoP Resources: in between the Natural Environment (their cradle) and the human-industrial environment (their application), depending on the viewpoint. This fundamental on-going debate is presented in the supplementary information (S.2). In the following, we define the AoP Resources as the protection of a resource (in sufficient quality and quantity) for future generations, while the AoP Human health and Ecosystems reflect the protection of current people and ecosystems. Since salinisation can potentially be assessed through several LCIA modelling approaches; in relation to a salt emission and/or a water use and/or a Land Use Land Use Change (LULUC), we analysed in the following the different modelling options for each salinisation types.

### 2.4.2. Modelling options for the different salinisation types

#### 2.4.2.1. Midpoint indicators

It is challenging to define the optimum midpoint indicator because “*Midpoints concern all elements in an environmental mechanism of an impact category that fall between environmental intervention and endpoints*” (Udo de Haes et al. 2002). What are the best midpoints: soil and water salinisation, or soil fertility loss and freshwater deprivation (Fig. 2.1). On the one hand, soil fertility loss and freshwater deprivation as midpoints represent a real impact and not just a concentration increase. On the other hand, soil and water salinisation as midpoints represent the end of the fate modelling. In addition, soil and water salinisation could be expressed in the same unit (e.g. TDS or EC) so we can sum up the two midpoints into one single indicator representing salinisation impacts for both soil and water. A conversion factor exists between TDS ( $\text{mg L}^{-1}$ ) and EC ( $\mu\text{S cm}^{-1}$ ):  $\text{TDS}=640 \text{ EC}$  (USDA-NRCS 2015). For these reasons, we propose to define the midpoints as the soil and water salinisation.

#### 2.4.2.2. Salinisation associated with land use change

According to JRC-IES (2011), a completed or revised land use framework may include soil salinisation in LCA. However, we believe that the LULUC framework can only partially models salinisation associated with a land use change. The human intervention is a land transformation of a given area (e.g. from forest to arable), matching perfectly with the inventory flow requirement: the area of a land use cover transformed from one type to another. Koellner et al. (2013b) provided a tiered typology of the land use

and cover categories: they can be global (e.g. “*arable*”) or more refined (e.g. “*arable, irrigated, intensive*”). These inventory flows are regionalized because the same type of land use may trigger different impacts at different locations of the globe. Regarding the CF, it reflects a difference in quality, and also a regeneration time for a LUC. In some cases, the regeneration time is exceeding the modelling horizon, thus corresponding to permanent impacts.

The LULUC framework assesses impacts on-site (at the location of the intervention), but does not model accurately in practice off-site impacts (not at the location of intervention). This is a limitation to account for the subsequent waterlogging and salinisation of soil and aquifer occurring off-site (from the LUC) in lowlands. For example, among the operational methods available, Saad et al. (2013) assesses the land use impacts on freshwater regulation, erosion regulation and water purification. However, the hydrological balance alteration downstream the location of a LUC is not modelled in Saad et al. (2013). Furthermore, soil salinity may impact freshwater regulation, erosion regulation and water purification, but this soil parameter is not accounted for in the LANCA<sup>®</sup> model used by Saad et al. (2013). LANCA<sup>®</sup> (LAND use indicator value CA l culation) is a calculation tool model assessing the influence of different land use activities on soil ecological functions (Beck et al. 2010; Saad et al. 2011). It is important to notice the strong link between land use and water use impacts, especially for irrigated agriculture because irrigation is part of the land use practices, and LUC can lead to changes in the water cycle at the catchment scale (Koellner et al. 2013b). We thus suggest the use of a hydrological model accounting for the key parameters listed in Table 2.1: linking the inventory flows (LUC) with a mechanistic fate modelling of water and salts (i.e. describing environmental mechanisms). The LANCA model is a good basis, but several limitations should be overcome (e.g. accounting for soil salinity, not considering a constant depth of aquifer). The use of a globally valid and reliable model such as SWAT (modelling the movement of pesticides, sediments or nutrients, and driven by the water balance of the watershed) (Neitsch et al. 2009) could be investigated.

In the LUC salinisation context, the main issue is related to the time frame: how to allocate the impacts when the land use change occurred many years ago? In many areas, as in Australia, dryland salinity is an on-going threat but is not easily linked to on-site agricultural management (Renouf et al. 2014).

#### **2.4.2.3. Salinisation associated with irrigation, brine disposal and overuse of a water body**

The LULUC framework is not appropriate to model salinisation associated with irrigation, brine disposal and overuse of a water body. The land use and land cover types are not refined enough to account for the key parameters affecting salinisation such as the irrigation mode or the crop type. Nevertheless, defining refined land cover types such as for example: “*citrus crop, drip irrigation, water EC=4dS.m<sup>-1</sup>*”, is not feasible. In addition, accounting for salts emission through the land use framework reduces the frontline with the salt emission-modelling framework, thus increasing double counting risks. Koellner and Geyer (2013a) highlighted this difficulty to define an integrated impact assessment where land use impacts are accounted for alongside chemical emissions, water use and climate change impacts. This overlapping risk increases with the evolution of the land use framework (more detailed land use types are claimed), in parallel with the improvement of methods based on environmental mechanisms such as for water use impact modelling (e.g. Verones et al. 2013a; 2013b). The land use framework is a good way

to assess multifactorial impacts that are complex to model and could be a good strategy in LCA where the agricultural phase is of minor importance. For LCA including an important agricultural phase, modelling complex cause and effect chains with a fine description of environmental mechanisms is preferable and is gradually being developed (e.g. pesticides, water).

The inventory flows requirement for each salinisation type will vary depending on the boundary between the technosphere and the ecosphere (this will be discussed in section 4.2.4). The minimum inventory requirements are: for salinisation associated with irrigation, the inventory flow are a salts emission (from water and fertilizers) and a volume of irrigation water: considering both water quantity and quality like Boulay et al's method (2011a), for salinisation associated with brine disposal the inventory flow are a salts emission in brine disposal, and for salinisation associated with overuse of a water body, the inventory flow is a volume of water withdrawn (Fig. 2.1).

For the LCIA, a bottom up approach will focus on the stressors (interventions responsible of the impacts) and allow to better discriminate specific human interventions and contexts, in comparison with the alternative top down approach organizing impacts according to which AoPs are affected (Udo de Haes et al. 2002). Again, we recommend an approach modelling the environmental mechanisms, in line with the ISO standard which states that category indicator shall use identifiable environmental mechanisms. The cause-effect consists in three steps: the fate of the substance in the environment until the final compartment, the exposure of the target, and the effect of this substance on the target. But following salts implies following water. The fate of salts and water, ultimately reaching soil, aquifers and surface compartments, should be calculated based on salt and water balances, involving different scales (field and/or catchment), and accounting for the key parameters listed in table 2.1. The fate modelling should allow for the discrimination of systems according to key parameters such as soil type, irrigation mode, water quality. For example, in agricultural LCA, discriminating systems according to the irrigation mode would be relevant because there is a paradoxical effect between water saving and salinisation: switching from a surface to a drip irrigation system help save water but increases the soil salinisation. Indeed, the adoption of drip irrigation results in reducing the amount of irrigation water, but salt leaching is reduced as well, thus increasing soil salinity. Another advantage of adopting an approach based on environmental mechanisms lies in the assessment of both on-site and off-site impacts thanks to the fate factor which follows the substance in the environment. However, on-site impact assessment for agricultural systems is dependent on the boundary definition between the technosphere and the ecosphere as shown for pesticides emissions by Van Zelm et al. (2014) (Cf. Part 4.2.4).

A geographical differentiation of CF is required because the fate of salts depends on climate, soil, and the hydrological context and their effect depends on the sensitivity of the target (e.g. species, capacity to desalinate water). It is paramount to develop highly-spatially explicit CF supporting aggregation over the whole life cycle (Hauschild et al. 2013; Cucurachi et al. 2014). Geo-referenced CFs should be supported by geo-referenced databases, which availability may hamper a method operationalisation (Cf. Part 4.3). Regionalized impacts in LCA can be supported by geographic information systems (e.g. Núñez et al. (2010); Boulay et al. (2011b); Saad et al. (2013); Núñez et al. (2013)). The time horizon definition is of crucial importance as well for the calculation of the salts fate factor, as it can modify the outcome of an LCA, especially when quantifying the impact of substances with a long lifetime (De Schryver et al. 2011; De Schryver et al. 2012; Huijbregts 2013). The time horizon is related to the spatial scale as it will

determine which final compartment is reached at this time. If salinity varies on a daily basis and can have visible effects within a short period (crop cycle scale), in contrast, sodicity has an effect after many years of inadequate management (several crop cycles scale). In the model selection, the challenge will be to find a trade-off between feasibility and accuracy. As an example for agricultural systems, the Aquacrop model (Steduto et al. 2012) is a salt and water balance model including a crop growth model, which can be coupled with GIS. This model is already valid for all herbaceous crops and should be updated to progressively include perennial crops (E. Fereres, pers, comm.).

The disadvantage of modelling each pathway following the fate and effect factors (bottom up approach) lies in the subsequent impacts weighting (Goedkoop and Spruiensma 2001): we can only add the impacts and miss the potential combined effects, in opposition to the top-down approach.

#### **2.4.2.4. From midpoint to endpoint**

Water and soil salinisation affect the three AoP. We analyse in the following the modelling options for each AoP. The fate factor being part of the previous step (from human intervention to midpoints), this section refers strictly to effect factors.

##### **Ecosystems**

Water salinisation affecting ecosystems could be modelled through aquatic ecotoxicity assessment. Amores et al. (2013) adopted this type of modelling in a specific context. Salts are not yet modelled in the USEtox model, a scientific consensus model providing CFs for both human health and freshwater ecotoxicity impacts (Rosenbaum et al. 2008), but future developments of this tool could include freshwater ecotoxicity CFs for salts. Another future improvement of the USEtox model is to develop regional versions because no spatial differentiation of location of the emission was considered so far (Henderson et al. 2011). Similarly, soil salinisation affecting ecosystems could be modelled through terrestrial ecotoxicity assessment. The development of terrestrial ecotoxicology CFs is also part of future developments of the USEtox model (Henderson et al. 2011).

Soil salinisation occurs on agricultural land and on non-agricultural land. It is crucial to differentiate it because non-agricultural land refers to biodiversity everywhere except in the agricultural soil (AoP Ecosystems), whereas agricultural land refers to agricultural soil biodiversity which has an intrinsic value (AoP Ecosystem) but also a crop support value through the support of soil fertility and a potential effect on malnutrition (AoP Human Health) (Fig. 2.2). Thus, salinisation on agricultural land refers to damages on biodiversity (AoP Ecosystem) and fertility of the soil (AoP Human Health). LCA should be able to reflect the importance of agricultural soil biodiversity and its paramount role in land fertility. There are no double counting risks if the discrimination between the two pathways: agricultural land salinisation effects on agricultural soil ecosystem (AoP Ecosystem) and on soil fertility (AoP Human health), is done properly.

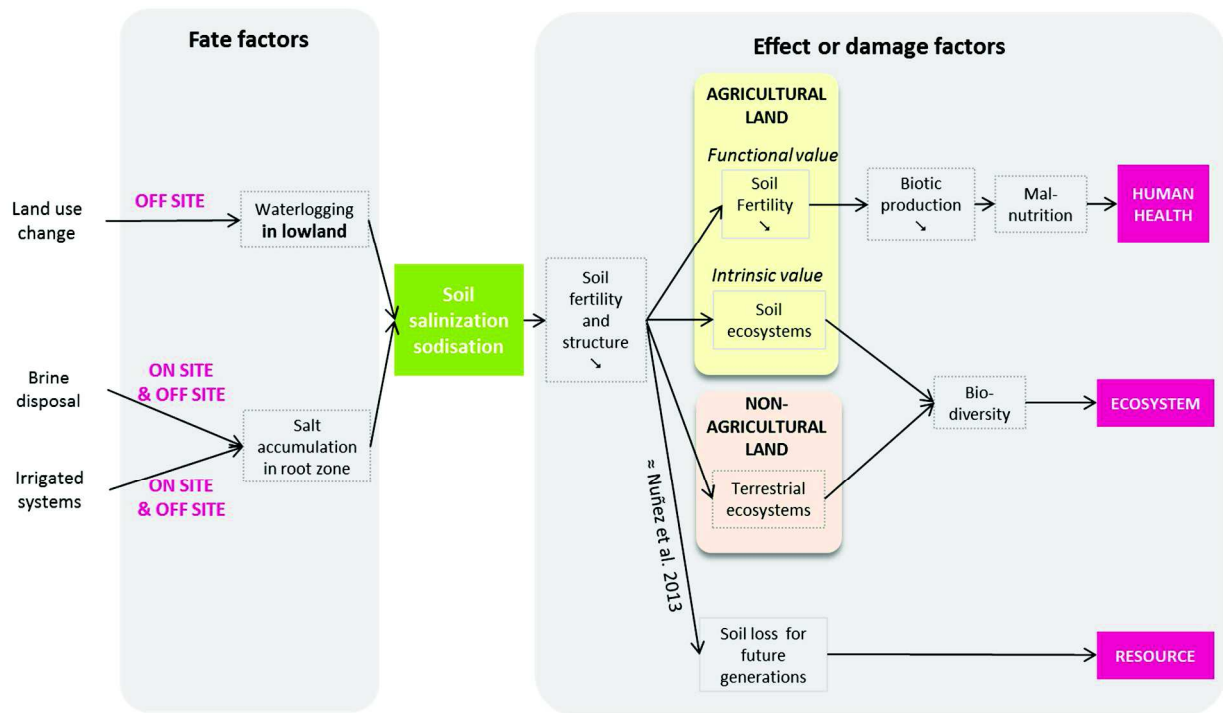


Figure 2.2. Soil salinisation impacts on Human Health, Ecosystems and Resource: fate and effect factors positioning on the cause-effect chain and relations between agricultural and non-agricultural lands.

The effects of soil salinisation on agricultural land biodiversity/ecosystems can be accounted for only if the technosphere (system studied) and ecosphere (environment) boundary allows for it, i.e. if the agricultural soil is not completely included in the technosphere or is included into it only momentarily. Indeed, several soil status options are possible because soil is both an environmental target and a part of the agricultural system. Setting the technosphere boundary will determine which parameters have to be accounted for in the inventory or to be part of the impact assessment (Fig. 2.3). Following the recommendations of Rosenbaum and colleagues (2015), this boundary should be defined according to the goal and scope of the study. In the case of salinisation associated with irrigation, if the objectives are to (i) distinguish the management practices (ii) put the stress on the human intervention on which we can act to reduce impacts, we recommend including in the technosphere the part of the soil that is influenced by the practice. Moreover, if one does not want to miss the potential impact on the agricultural land, soil should be included in the technosphere only during the time it is being used by the system and supporting its function, and considered as returned to the ecosphere in a potentially modified state afterwards (not shown in Fig. 2.3). From an operational viewpoint, this means that the discriminating factors such as drainage, irrigation mode and soil hydrodynamic profile should be accounted for in a dynamic way in the inventory stage rather than in the characterization stage assuming a steady-state (Fig. 2.3). Setting the lower boundary at the root zone limit and the temporal limit at the beginning/end of the cropping cycle is consistent with crop and water balance models.

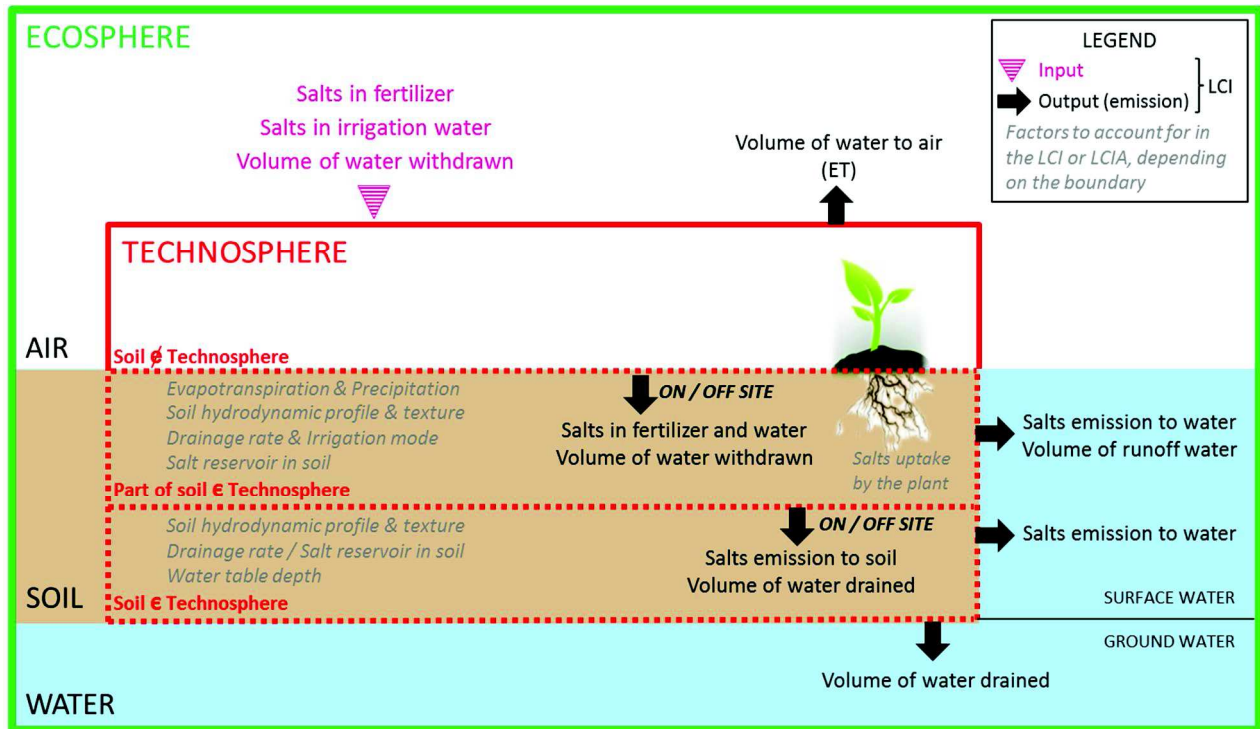


Figure 2.3. Salinisation associated with irrigation and deposition of salts: technosphere and ecosphere boundaries options and corresponding parameters to account for in the inventory or in the impact assessment. If soil is not included in the technosphere, the inventory requirements are the inputs to the technosphere: salts in fertilizer and in irrigation water, and volume of water withdrawn. If part or the whole soil is included in the technosphere, additional parameters (in grey italic) have to be accounted for at the inventory stage to calculate the output (emissions) flows, thus allowing to account for management practices.

Ecological effect factors are usually based on Species Sensitivity Distributions (SSD), relating the concentration of a pollutant in the environment with the Potentially Affected or Disappeared Fraction of species (PAF or PDF) (Huijbregts et al. 2011). This non-linear relationship requires that the fate and exposure assessment provide background dose information (Udo de Haes et al. 2002), except if the effect factor is linear (e.g. Amores et al. 2013). But there is a limited updated knowledge about the background exposure levels of salinisation: although reports of secondary salinisation abound in the literature, there is a lack of recent assessment of the levels of salinisation (Flowers 1999). An open research question is whether the effect factors should be derived following a marginal approach or an average approach (supplementary information, S.3). Salinity dose-response information abound in the literature with heterogeneous spatial coverage: e.g. freshwater fishes of south-western Australia (Beatty et al. 2011), freshwater small crustacean (Gonçalves et al. 2007), freshwater mussels in Canada (Gillis 2011), aquatic plants in Australia (Kim et al. 2013). Thus, the development of spatially-explicit effect factors on ecosystems with global coverage will be challenging and should compile all publications in the field.

Damage to ecosystem quality can be expressed as species diversity, the recommended endpoint indicator by JRC (2010). But function-related parameters, such as the biomass production of the ecosystem often estimated through the Net Primary Production NPP, might also be good endpoint indicators (Núñez et al. 2013). However, when using a NPP-based indicator, one should specify which

production of biomass is considered: either the biomass production of ecosystems referring to the AoP Ecosystem, or the biomass production of agricultural land referring to the AoP Human Health.

### **Human health**

Soil salinisation reduces productivity of agricultural and pasture lands, leading to a reduction in food availability. This, in turn, can cause malnutrition and damage human health if compensation scenarios are not possible (i.e. in developing countries). The reduction in biotic production potential is due to ecotoxicological effects of salts on crops, but also to soil physical degradation. Many studies report the salt tolerance of crops: salinity thresholds (Maas and Hoffman 1977; FAO 1985) and sodicity thresholds (Qadir et al. 2001). Soil structure alteration not only affects the biotic production potential but also freshwater regulation and erosion potential. It is noteworthy that other impact pathways exist: a review of the implications of salinity on human health shows effects on respiratory health, vector-borne disease, and mental health (Jardine et al. 2007). The pathway soil salinisation damaging Human health concerns impacts on agricultural land, which are on-site in the salinisation context of irrigated systems, and off-site in the context of a LUC (Fig. 2.2). As discussed in the previous section, a proper accounting of impact on agricultural land strongly relies on the technosphere boundary: soil should therefore be included only partially in the technosphere, and only the time it is being used to support the studied function.

Water salinisation affecting human health does not refer to toxicological effects of drinking saline water. It refers to the water quality degradation making the water inappropriate for certain usages, thus corresponding to a water deficit if compensation scenarios are not possible. The modelling of water salinisation affecting Human health requires a functional approach such as the one suggested by Boulay and colleagues, which assesses damages of a water functionality loss, accounting for the adaptation capacities (Boulay et al. 2011b). Total Dissolved Solids, Bicarbonate, Chloride, Chlorides/nitrites, Sodium and Sulfate in water are already parameters accounted for in this method to define the water categories for users (Boulay et al. 2011a). However, the method proposed by Boulay et al. (2011a, 2011b) cannot be applied in its present form due to a scale modelling issue: the water salinized should be an inventory flow which is the result of a balance between water input and water output (with associated salinity increase). Boulay's method can only be applied in the case of salinisation of drainage water induced by irrigation (Fig. 2.4), and if the soil is included in the technosphere (at least temporarily); i.e. the saline drained water is considered an "emission". Furthermore, the CF is based on a water stress index, not referring to any environmental process. Damage to Human health can be expressed as DALY (Disability-Adjusted Life Years): the most used unit by current LCIA methods addressing damages to Human Health (e.g. Pfister et al. (2009); Motoshita et al. (2011); Boulay et al. (2011b)).



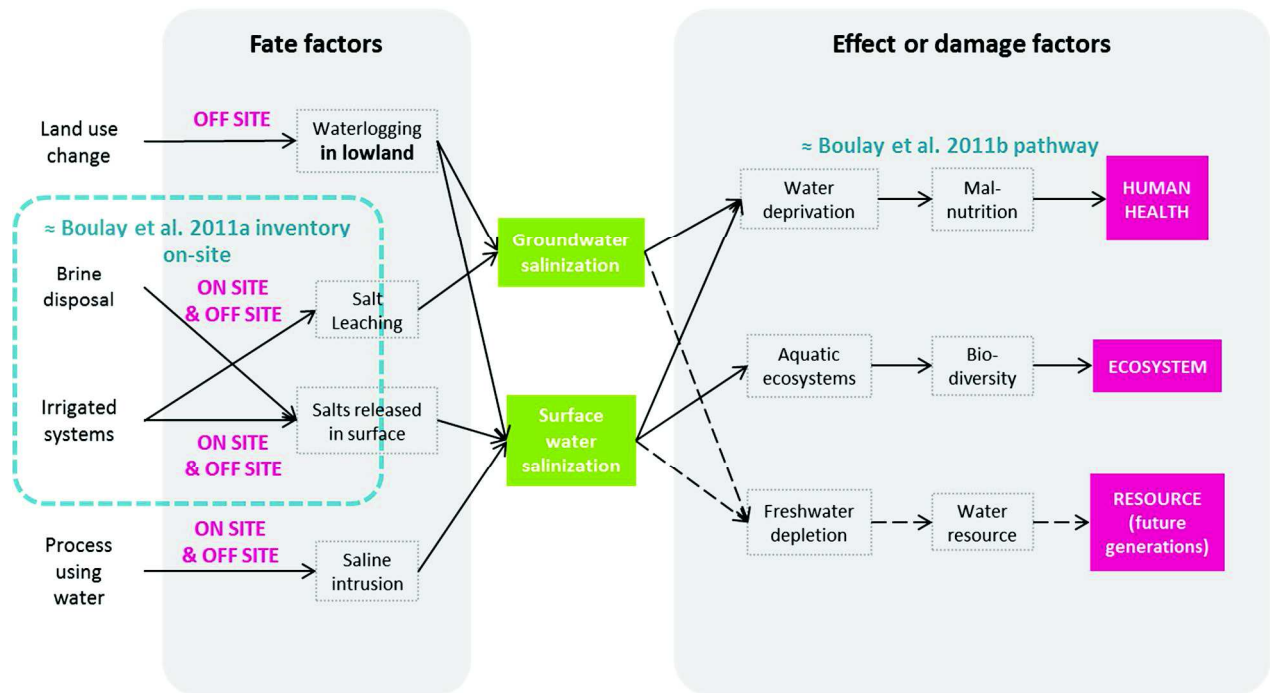


Figure 2.4. Water salinisation impacts on Human Health, Ecosystems and Resource: fate and effect factors positioning on the cause-effect chain and Boulay et al. inventory and damage to human health methods positioning.

## Resource

Water and soil salinisation affecting water and soil as resources are debatable pathways because the AoP Resources is not always considered as an intrinsic AoP (Cf. part 4.1). However, for the sake of completeness and to put forward soil and water resource preservation stake (Renouf et al. 2014), we suggest considering that permanent water or soil quality degradation represents a damage to Resource for future generations. There is no risk of double counting if one clearly defines the AoP Resources as the protection of a resource (in sufficient quality and quantity) for future generations, while the AoP Human health and Ecosystems reflect the protection of current people and ecosystems. Permanent degradation of soil or a loss of soil through erosion reduces soil availability as a future resource (Núñez et al. 2013). A high salinity area in a very dry climate could be barren for an indefinite time period and corresponds to a permanent impact (Koellner et al. 2013b). In particular, the soil structure alteration in case of sodisation is almost irreversible. Soil salinisation damages on Resources is close to the pathway modelled by Brandão and Milà i Canals (2013), (a slightly modified version of Milà i Canals et al. (2007) recommended by ILCD (2010)), consisting in a midpoint indicator of soil quality based on soil organic carbon. This impact is damaging the AoP Resources, but there is no endpoint indicator developed so far, and salinisation is not accounted for, in spite of its influence on biomass production potential. Salinisation damages on Resources should rather be modelled through a framework based environmental mechanisms modelling, close to the pathway developed by Núñez et al. (2013) for soil erosion: an intermediate between LULUC and an approach based on environmental mechanisms, involving damage and effect factors (Fig. 2.2). Water quality alteration affecting Resources is not considered in the water use impact framework (Kounina et al. 2013), although a quality alteration is affecting the availability of these resources for further uses. The framework proposed by the UNEP-SETAC Water Use in LCA (WULCA) working group considers that only fossil water use or renewable water overuse can affect the

resources (Cf. Fig. 1 in Kounina et al. 2013), although a permanently degraded freshwater represents a loss of water resource for future generations. For example, in the case of permanently saline aquifers, we can consider that future generations will be deprived of water in that specific location. It is therefore paramount to address this question in future research. Damage to Resources can be expressed as energy needed to make the resource available in the future: in energy units (megajoule equivalents), such as in EcoIndicator99, or emergy units (megajoule solar equivalents), such as in Núñez et al. (2013) for soil depletion. Other approaches express damage to Resource in monetary equivalent, such as in ReCiPe, but this unit is confronted to the cost variability of a technology.

### 2.4.3. Toward operationalisation

The recommendations provided in this article are mostly conceptual. The aim is to build a consistent and comprehensive framework which is not available through the existing methods addressing salinisation impacts. The need of a common framework regarding the technosphere and ecosphere boundary, the status of the areas of protection and the modelling approach (top-down vs. bottom up) are paramount. Our recommendations aim at overcoming limitations that existing methods were confronted with: regarding the need for a global coverage (the characterisation model should be applicable globally and to not miss any important parameter involved), and regarding the need for accounting for all potential impacts pathways (without gaps or overlapping). That is why we recommend starting from an understanding of the environmental mechanisms, driven by the water cycle, at a scale going beyond the plot. Such a bottom-up and mechanistic (i.e. describing environmental mechanisms) approach is the best way to discriminate the studied systems, therefore allowing eco-conception, one of the core application of LCA. Nevertheless, in the operationalisation process, we will have to cope with the lack of globally available data for the development of characterisation models. Indeed, spatial explicit impact assessment requires the use of local, regional or country specific information. This concerns all impact categories. The lack of data available with a global coverage may hamper the operationalisation of a method, both for the inventory and the impacts assessment. Thus, method developer should provide geo-referenced databases for inventory parameters (according to the technosphere boundary) and background system assessment, and build geo-referenced CF. Among the parameters involved in salinisation impact assessment (Table 2.1), reliable geological and groundwater level data are lacking (Zhou et al. 2013a). Generally, data availability and accuracy vary according to the country. But the development of geo-referenced databases is significant, notably thanks to remote sensing data acquisition (Bastiaanssen et al. 2007). Thus, we can reasonably think that in the medium term these objectives of global modelling are reachable. As a next step of this article, on-going work is developing an inventory tool operational for agricultural systems: water and salt flows model, compliant with the recommendations provided in this article.

## 2.5. Conclusion

Including salinisation impacts in LCA is of high priority. Assessing salinisation impact is particularly relevant in food LCA, because agricultural systems are both the main affected targets and causes of

salinisation, but not exclusively: water body overexploitation, brine disposal or a land use change are also major contributors to salinisation worldwide.

Although the existing methods addressing salinisation in LCA are important and relevant contributions, they are incomplete in terms of spatial and environmental mechanisms coverage. The modelling complexities lie in the diversity of salinisation mechanisms, at both local and regional scales, and the status of soil and water in LCA which are both resources and living environments. An analysis of the modelling options in agreement with the LCA framework has been proposed in this paper. We identified and categorized the key biophysical and management factors involved for each salinisation types and discussed the inventory and impact assessment boundary options. The land use framework might be suitable to partially model salinisation impacts from a LUC but should be completed with a mechanistic approach (based on environmental mechanisms modelling) to account for off-site impacts. An approach modelling environmental mechanisms (i.e. based on fate, exposure and effect factors) should also be preferred to model salinisation related with irrigation, brine disposal and water body overexploitation. For all salinisation pathways, a bottom up approach (rather than empirical or top-down approach) is recommended because: (i) salts and water are mobile and their effects are interconnected (ii) this is in line with the ISO norm stating that one shall relate a consequence with a cause (i.e. model environmental mechanisms) (iii) this approach allows the evaluation of both on and off-site impacts and (iv) it is the best way to discriminate systems in which the agricultural stage is predominant and support a reliable eco-design which is the core aim of LCA. Regarding the boundary between the technosphere and ecosphere (i.e. inventory and impact assessment), we recommend to consider the part of the soil that is influenced by farmers practices in the technosphere, and only during the time it is being used by the system because it allows discriminating the agricultural practices and their effects (including impacts on agricultural land), and it is consistent with many crop and water balance models (that we recommend to use within a fate modelling).

By discussing paramount methodological issues, this paper provides the basis for future method developments, and shows that much research effort is still required to include salinisation impacts in a global, consistent and operational manner in LCA. Next steps include the testing of fate and exposure models, fed by global databases, with the background issue to find a trade-off between accuracy and feasibility. To do so, it is important that LCIA scientists join their efforts together with salinisation experts to build a consensual model (Huijbregts 2013).

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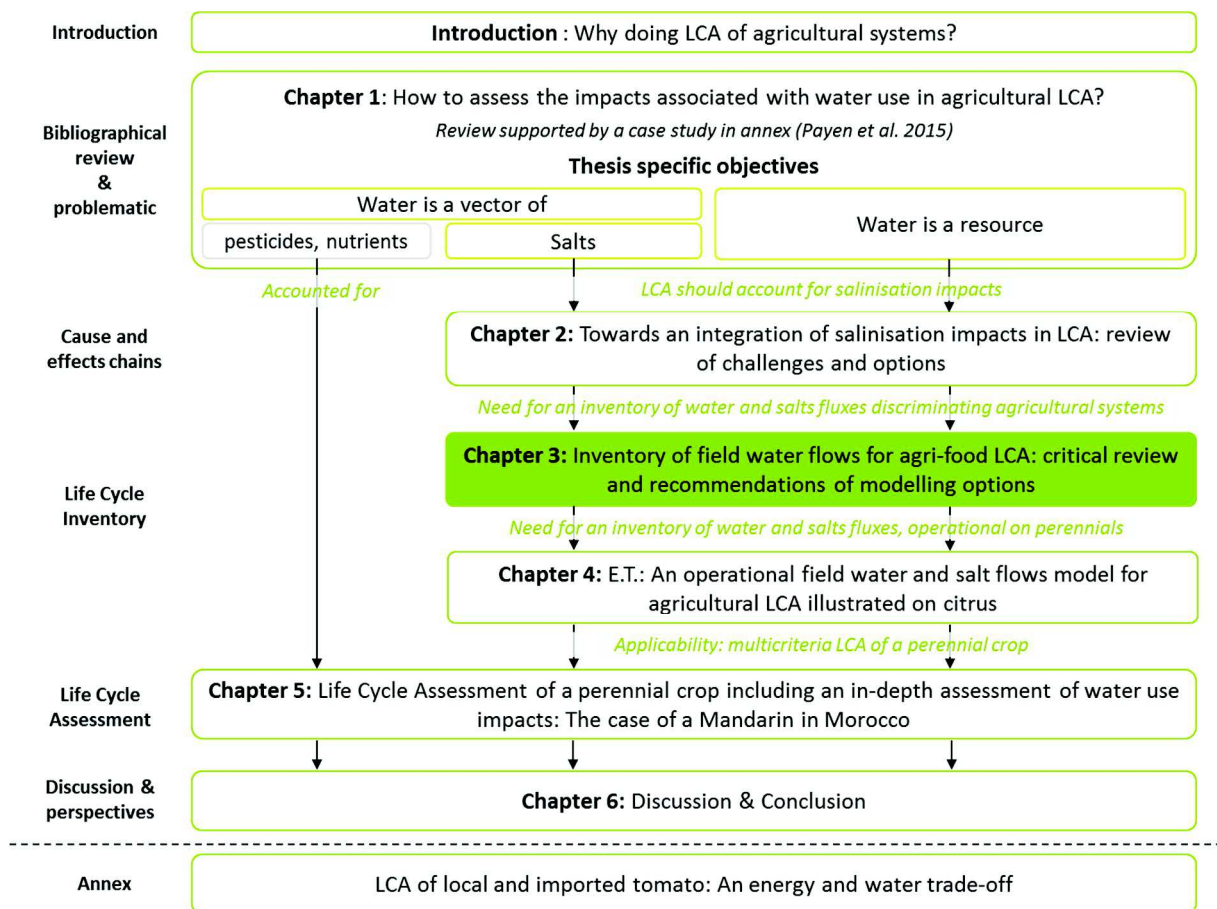


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Chapter 2 showed that when a comprehensive assessment of salinisation impacts relies on an inventory of water and salts flows, consistent with a mechanistic and “bottom-up” approach, allowing the discrimination of systems. After this reflexion process on the whole cause and effect chains, chapter 3 and 4 will address the inventory stage by answering the question: Which inventory for field water and salt flows?



## Chapter 3

### Inventory of field water flows for agri-food LCA: critical review and recommendations of modelling options

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*Sandra Payen*<sup>1,2</sup>, *Claudine Basset-Mens*<sup>2</sup>, *François Colin*<sup>3</sup>, *Pauline Roignant*<sup>2</sup>

<sup>1</sup> ADEME, 20 avenue du Grésillé, F-49004 Angers, France

<sup>2</sup> CIRAD, UPR Hortsys, ELSA – research group for environmental life cycle sustainability assessment, Boulevard de la Lironde, F-34398 Montpellier, France

<sup>3</sup> Montpellier SupAgro, UMR LISAH, place Pierre Viala, F-34060 Montpellier, France

#### Abstract

**Purpose:** In a context of flourishing eco-labelling programs and environment policy for food products, LCA applied to agricultural systems faces the challenges of being operational, accurate and exhaustive. Indeed, LCA has to include all major environmental impacts such as water deprivation. This is particularly challenging for the young water use impact assessment in LCA, with many methods recently developed. Water inventory databases (e.g. WaterStat) and agri-food LCA databases (e.g. World Food LCA Database) contain default water elementary flows for average crop and animal products. These databases should support the assessment of water use impacts, to some extent. However, these different databases are not adapted to compare specific agricultural practices with LCA. To allow all LCA practitioners fulfil their diverse agri-food LCA objectives, a proper review of available methods for field water flows inventory and recommendations are needed.

**Methods:** We critically analysed the models on which water inventory and agri-food LCA databases rely. Then, we explored alternative modelling approaches for an inventory of field water flows in LCA aiming at discriminating cropping system practices. We finally made recommendations to help practitioners identify the most appropriate method for field water flows according to their LCA study objectives and constraints.

**Results and discussion:** Water inventory and agri-food LCA databases provide estimates of theoretical water consumed by a crop and rely on data and methods presenting limitations, making them suitable only for background agricultural LCAs. In addition, databases do not support the application of methods assessing the impacts of both consumptive and degradative water use. For the LCA-based Ecodesign of cropping systems, the inventory of water flows should be based on a model simulating evapotranspiration, deep percolation and runoff accounting for crop specificities, pedo-climatic conditions and agricultural managements. In particular, the model should account for possible water, saline and nutrient stresses; assess evaporation and transpiration separately, estimate runoff and drainage according to the systems specificities. Yield should not be estimated with a model but a primary data. Recommended and default data sources are provided for each input parameters. **Conclusion:** The FAO Aquacrop model represents an optimal balance between accuracy, simplicity and robustness. However, this model is not applicable for perennial crops yet.

**Keywords:** Agriculture, Life Cycle Inventory, Water flows, Salts, Model

### 3.1. Introduction

Agriculture is by far the main user of freshwater in the world (World Water Assessment Program 2009). Agriculture not only consumes water, but also pollutes water. Thus, in a context of increasing world population and food demand, agriculture is on the hot seat. To characterise and mitigate damages caused by agriculture and food products on the environment, Life Cycle Assessment (LCA) is increasingly recognised as the most relevant tool. This standardised methodology (ISO 2006a; ISO 2006b) was selected as the reference method to support environmental policy, eco-design in business, eco-labelling programs in many countries around the world. The French Eco-labelling program constitutes a recent example (Cros et al. 2010) with the development of specifications for the food sector (BPX-30-323-15 guidelines). At the European level (EU 2013), the food sustainable Consumption and Production Roundtables co-supervised by the European Commission and food companies has led to the ENVIFOOD Protocol in 2012 (Food SCP RT 2013). Within this context, incentives for standardization and normalisation of all methods needed for implementing LCA to agricultural systems are strong as illustrated for water use impacts by the ISO 14046 water footprint guidelines (ISO 14046 2014), or the WULCA consensual water deprivation indicator.

Moreover, beyond the well-known Ecoinvent database, new dedicated Agri-food LCA databases such as the AgriBalyse® database in France or the World Food LCA database are being developed as operational tools to implement public and private policies for eco-labelling and eco-design of food products. Although the AgriBalyse database aims at supporting eco-design of agricultural systems by including several technical alternatives for more than 40 animal and vegetal products, most databases only provide default references for so-called average systems and fail to address the complexity and diversity of agricultural systems worldwide. Furthermore, performing an accurate LCA of cropping systems remains a challenge for certain impact categories related to water and land use.

Methods and data need to be adapted to the objectives of each agri-food LCA study. Two categories of LCA can be defined: the first corresponds to LCA studies where the agricultural production system is a background process (agri-food LCA), while the second corresponds to LCA studies where the agricultural production system is a foreground process (agricultural LCA). In agri-food LCA studies, the target will be consumers or policy makers where LCA results will help them adapt their consumption behaviours and subsidies programs. In agricultural LCA studies, the targets are directly the farmers and associated stakeholders such as extension services. The LCA results will help them adapt their practices in relation to environmental impacts.

Regarding the environmental impacts related to water use, many Life Cycle Impact Assessment (LCIA) methods have emerged recently (Tendall et al. 2013), and continue to be improved. To apply these methods, an inventory of water flows is required. The water inventory requirements are varying according to the Life Cycle Impact Assessment method used (Kounina et al. 2013). The water flows to estimate are either the water consumed (part of water use emitted to air, not available at the catchment of withdrawal anymore), or the water withdrawn (from surface or ground water) and released (to surface or ground water). These flows have to be provided on a yearly or monthly basis, with or without pollutant data and origin (surface water or groundwater). For example, the methods of Frischknecht et al. (2008), Pfister et al. (2009), Verones et al. (2013a; 2013b) and Berger et al. (2014) require only the

volume of water consumed annually, Pfister and Bayer (2014) on a monthly basis, whereas Boulay et al. (2011a; 2011b) require the volume, origin and quality of the water withdrawn and released annually. Thus, the efforts in the data collection are different depending on which impact assessment method we want to apply. In particular, to assess the impacts of water use from water degradation (in addition to water consumption), information on the volume of water flows is not sufficient since their quality is essential. To support this task, water inventory databases are available: they contain the elementary flows for water. However, to which extend the databases are supporting the application of the different water use impact assessment methods? Reviews on water use impact framework (Kounina et al. 2013; Boulay et al. 2015a; Boulay et al. 2015b) do not provide a detailed description and critical analysis of these databases, and do not provide clear guidelines on alternative models available to perform this inventory of water flows. Yet, this is a prerequisite for assessing the impacts related to water use.

To allow all practitioners fulfil their diverse agri-food LCA objectives be it for eco-labelling, adaptation of consumption pattern or eco-design of production systems, a proper review of available methods for field water flows inventory is needed.

The objectives of this article are to provide both LCA analysts and researchers with:

- A critical analysis of available water inventory and agri-food LCA databases, with a focus on crops. We analysed the strengths and flaws of the models they rely on, in terms of accuracy for estimating water flows, and consistency with water impact assessment methods,
- A critical analysis of available models and a definition of model specifications for an inventory of field water flows in agricultural LCA studies, aiming at discriminating practices of cropping systems.

To sum-up, this work aims at clarify the gradient of complexity and accuracy in the different approaches (databases and models) for the inventory of water flows of cropping systems, and finally provide a matrix of criteria for the selection of the appropriate method for the inventory of field water flows in Agri-food LCA studies taking account of their study objectives and their data and resources available.

## 3.2. Critical analysis of water inventory databases

### 3.2.1. Water inventory and agri-food LCA databases

Water inventory databases are used in agri-food LCA databases where water elementary flows are provided next to other elementary flows such as fertiliser application rates and field emissions for crops and animal products. Figure 3.1 shows that water inventory and agri-food LCA databases are based on a water balance with the same calculation steps. The volume of water consumed is the only flow estimated with a modelling approach ①. Water withdrawal is simply deduced from water consumption accounting for additional water losses depending on irrigation efficiency ②, and water released is the resulting balance between water withdrawal and water consumed ④. Surface and ground water partitioning is based on statistics ③ or estimates ⑤.

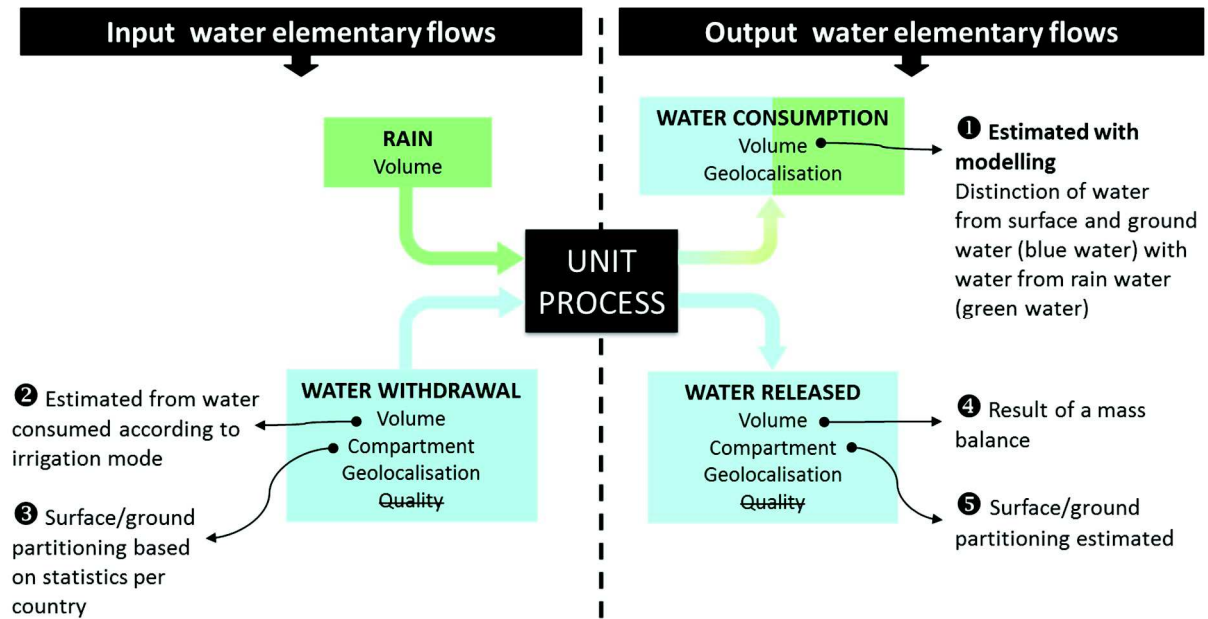


Figure 3.1. The general scheme of recent water inventory and agri-food LCA databases (Water Database (Quantis), WFDB, Ecoinvent v3): calculations steps and main assumptions to determine the different water flows. Databases are based on a water balance: input water = output water. Each water flow is geolocalised at country or large watershed scale.

In the following, we analyse the two different crop water inventory databases providing estimates for the water consumption flows for crops: WaterStat (Water Footprint Network) and Pfister et al. (2011).

### 3.2.1.1 Crop water inventory databases

Pfister et al. (2011) and the WaterStat databases are both estimating crop water consumption based on FAO concepts from Allen et al. (1998), but with a few differences (Fig. 3.2). Note that in the following,  $ET_a$  refers to the actual evapotranspiration whereas  $ET_c$  refers to the potential evapotranspiration. Refer to the glossary for a detailed definition, and to the acronym list for the abbreviations used.

*Pfister and colleagues' database* (Pfister et al. 2011; Pfister and Bayer 2014) provides the production-weighted average blue<sup>2</sup> water consumption for 160 crops, at country scale. The expected blue water consumption ( $ET_{\text{expected blue}}$ ) is the arithmetic mean of the full irrigation ( $ET_{\text{blue}}$ ) and deficit irrigation<sup>3</sup> ( $ET_{\text{deficit blue}}$ ) blue water consumption. The full irrigation blue water consumption is based on Allen et al. (1998), and corresponds to the maximum evapotranspiration ( $ET_c$ ) minus the effective precipitation ( $P_{\text{eff}}$ , the precipitation share actually available to crops) (Fig. 3.2). Crop coefficients ( $K_c$ ) are crop specific, and are based on Chapagain and Hoekstra (2004) (Table 3.1). Effective precipitation is calculated with two empirical equations, but they do not discriminate precipitation loss from runoff or from deep percolation. Deficit irrigation blue water consumption is calculated by multiplying full irrigation blue water consumption with the reported proportion of irrigation (Fig. 3.2). The proportion of irrigated cropland is obtained with the combination of two maps: the proportion of irrigated area (Siebert et al.

<sup>2</sup> The blue water refers to the volume of surface and groundwater consumed; the green water refers to the rain-water consumed.

<sup>3</sup> Pfister et al. 2011 define deficit irrigation as the actual water consumption in situations where less water than irrigation water requirement is applied, due to lack of irrigation facilities or limited water availability.



2007) and cropland share (Ramankutty et al. 2008). The water consumption is then divided by the yield (Monfreda et al. 2008), and expressed in  $\text{m}^3\cdot\text{ton}^{-1}$ . Pfister and colleagues acknowledged that the uncertainty of the input data is high and not quantified, the most critical data sets being the crop distribution and yield (Monfreda et al. 2008), and the applied irrigation (Siebert et al. 2007). In an updated version of the database (Pfister and Bayer 2014), two improvements were made. The geometric mean was considered more suitable than the arithmetic mean, since it represents a lognormal distribution and places the emphasis on smaller values. And in contexts where no irrigation is reported, expected water consumption is not considered nil anymore but equal to 5% of full irrigation, to account for a lack of accuracy in the maps reporting proportion of irrigated areas.

*In the WaterStat database* (Water Footprint Network), the Water Footprint Network assesses the green, blue and grey water footprints<sup>4</sup> of crops and derived crop products, biofuels, but also farm animals and animal products, at national and sub-national levels. The method used is described in the Water Footprint Assessment Manual (Hoekstra et al. 2011), and more details are provided in Mekonnen and Hoekstra (2011). They perform a daily water balance, based on Allen et al. (1998), to calculate the crop water requirements, actual crop water use, and yield, taking into account local climate and soil conditions. For rain-fed crops, blue water use is zero and green water use is the actual crop evapotranspiration ( $\text{ET}_{a \text{ green}}$ ) accounting for a possible water stress, through the stress coefficient ( $K_s$ ).  $K_s$  is a dimensionless transpiration reduction factor dependent on available soil water and the crop. For irrigated crops, the green water use is the actual crop evapotranspiration of non-irrigated crops, estimated with a water balance without irrigation but using crop parameters of irrigated crops (e.g. rooting depth). The blue water use of irrigated crops is a combination of two water balances: the crop water irrigation requirement (assuming full irrigation) minus the green crop water use. It is noteworthy that this approach assesses water consumption under non-standard conditions (e.i. accounting for a possible water stress) but only for rainfed crops (Fig. 3.2). The actual yield is calculated with the FAO original *water production function*, where relative yield reduction is related to the corresponding relative reduction in evapotranspiration assuming a linear relationship. This approach and the calculation procedures were published in the FAO Irrigation and Drainage Paper No. 33 (Doorenbos and Kassam 1979) and involve two concepts: the maximum yield ( $Y_{\text{max}}$ ) and the crop yield response factor ( $K_y$ ) (Table 3.1). Finally, the green and blue water footprints of crops (in  $\text{m}^3$  per ton) are calculated by dividing the total volume of green and blue water use, respectively, by the quantity of production. The grey water footprint provides information about the quality of the released water, because it is calculated by multiplying the nitrogen application by the fraction of nitrogen that leaches or runs off to water bodies, and dividing this by the maximum acceptable concentration of nitrogen and by the actual crop yield. The calculation of this “necessary dilution volume” is based on the standard recommended by EPA (2005) for nitrate in drinking water, and assumes that nitrate background concentration is negligible (Chapagain et al. 2006).

Other global *water databases* exist (e.g. Liu and Yang (2010), Hanasaki et al. (2010)), but they contain a limited set of agricultural crops, and are not detailed in this article. They have in common the assumption that full irrigation is considered in irrigated areas (Hoff et al. 2010).

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<sup>4</sup> The grey water footprint of a product refers to the volume of freshwater that is required to assimilate the load of pollutants based on existing ambient water quality standards

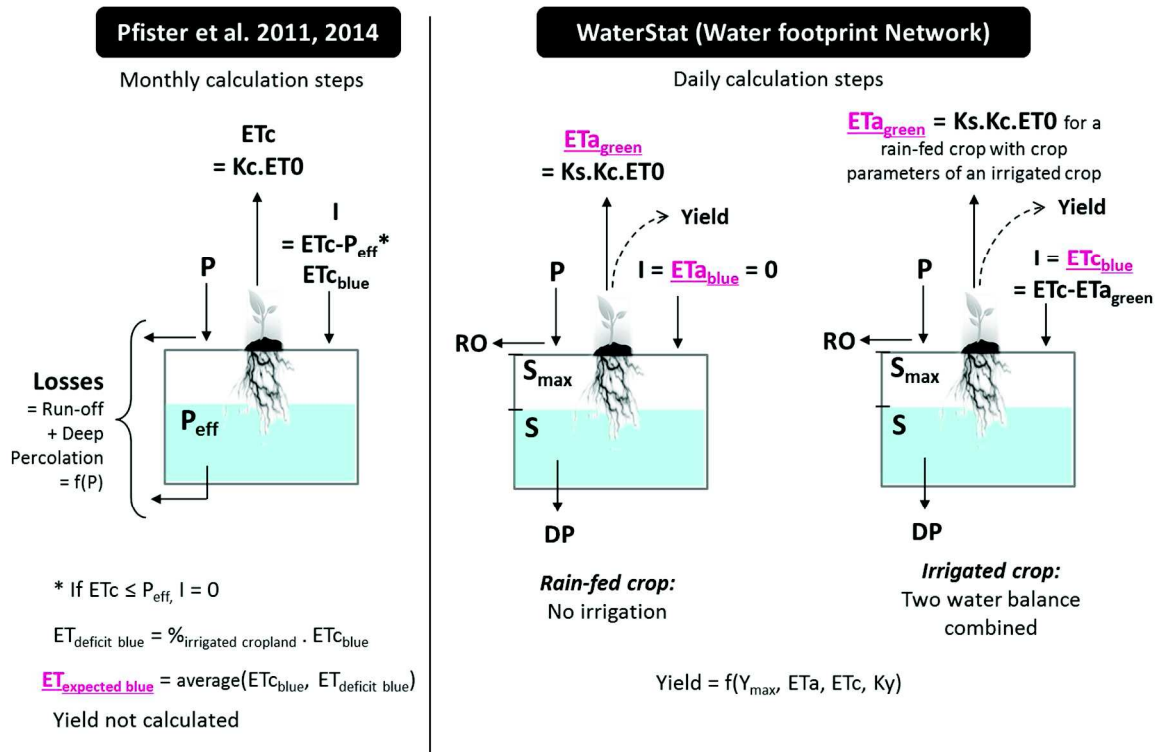


Figure 3.2. Scheme of the calculation methods used in water inventory databases (Pfister et al. (2011, 2014) and WaterStat).

Pink underlined water flows are the water elementary flows available in the databases. RO is the runoff water flow, DP the deep percolating water flow, S the soil water stock.  $ET_0$  is the reference evapotranspiration.  $ET_c$  is the crop evapotranspiration under standard conditions whereas  $ETa$  is the actual crop evapotranspiration. Green and blue indices refer to evapotranspiration of green and blue water.  $ET_{deficit,blue}$  is deficit irrigation blue water consumption,  $ET_{expected,blue}$  is the full irrigation blue water consumption through evapotranspiration, according to Pfister et al. (2011).  $K_c$  is the crop coefficient,  $K_s$  is the water stress coefficient,  $K_y$  is the crop yield response factor.  $P_{eff}$  is the precipitation share actually available to crops (according to Pfister et al. (2011)).  $Y_{max}$  is the maximum yield.

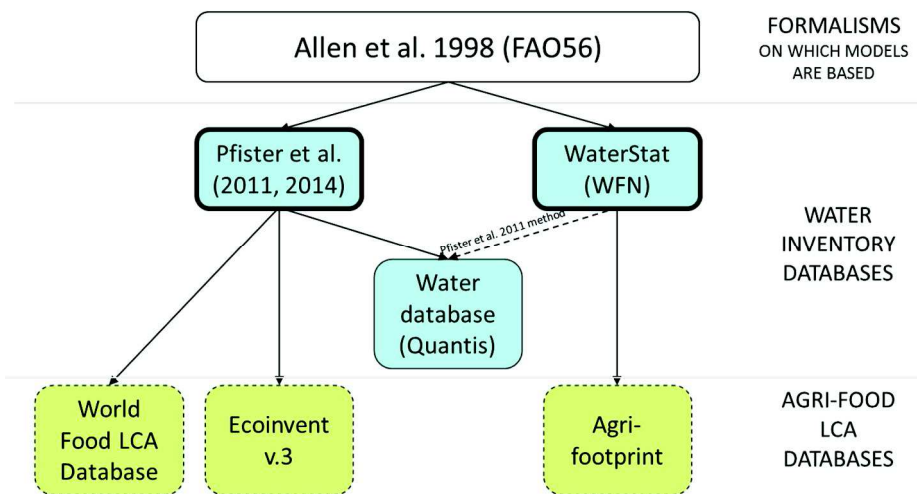


Figure 3.3. Agri-food LCA databases and Quantis water database are using data from water inventory databases for the water elementary flows of agricultural systems. Water inventory databases are using formalisms of water balance and crop evapotranspiration estimation from Allen et al. 1998 (also denoted as: FAO N°56).

### 3.2.1.3. Agri-food LCA databases

Agri-food LCA databases rely on either WaterStat (Water Footprint Network) or Pfister et al. (2011) databases for water elementary flows (Fig. 3.3).

The freely available *Agri-footprint*<sup>®</sup> database (Blonk Agri-footprint 2014) is reporting the water use for irrigation of 30 crops, at country scale, based on the blue water footprint from Mekonnen and Hoekstra (2011). The water withdrawal and the water origins are not reported.

The *World Food LCA Database (WFLDB)* (Nemecek et al. 2015) reports water withdrawal, water consumed, water released to ground and surface water, and wastewater sent to treatment for a minimum of 25 crops. The database is made available only to the project partners so far, but a document is describing the methodological approaches adopted to model the datasets (Nemecek et al. 2015). The water consumed is based on the expected blue water consumption calculated by Pfister et al. (2011). The water withdrawal is the water consumed increased by the water losses due to conveyance and field application, depending on the irrigation technique. To assess the share of irrigation technique, different levels of details are possible, but it is unclear to which extent the different levels will be applied to the crops. Regarding the origin of water, country-specific shares of ground, surface water and non-conventional sources are retrieved from Siebert et al. (2010). The water emitted to surface and ground water is deduced thanks to a simple water balance: water withdrawal minus the water consumed and water content of crop. The surface - ground water default partitioning is 80% - 20%, based on Lévová and Pfister (2012), but we could not find any justification for these values. The *Ecoinvent v2.2* database (Frischknecht et al. 2007) is reporting water withdrawal, distinguishing the water origin, but for many crops, the irrigation water and water releases are not reported. In *Ecoinvent v3*, the additional flows reported are water to air, water to surface and ground water, water embedded in product, and wastewater. Thus, all the water flows are reported (Lévová 2014). However, a recent statistical analysis showed that about 45% of the Ecoinvent 3.1 processes do not respect the water balance, with a difference between water emitted to air and water withdrawal minus water released flows greater than 10% (Liao Xun, EPFL Blog, 2015). For the crops, data are based on Pfister et al. (2011) (Pfister 2012).

The *Quantis Water Database* (Quantis 2015) is reporting, similarly to Ecoinvent v3 and WFLDB, all water flows following as much as possible an ideal water balance. The water consumption of crop products is based on Pfister et al. (2011), and when data are not available for a crop or to be more specific to a given region, the Pfister et al. (2011) model is applied to the blue water footprint published by the Water Footprint Network (Bayart, pers. comm.). Water withdrawal is then deduced accounting for irrigation efficiency, and its origin is based on national statistics. Water released is the result of the water balance, and the surface groundwater partitioning based on estimates (no further details are provided). The crops covered are the same as in Ecoinvent v2 plus additional crops. To our knowledge, there is no freely available documentation about the modelling principles of the database.

The *AgriBalyse*<sup>®</sup> database (ADEME 2015) is reporting irrigation water use for 25 crops grown in France or imported. However, the exact physical meaning of this water is not clear for all crops. As part of the *AgriBalyse*<sup>®</sup> 2 project, efforts are made to update the database regarding water flows.

A detailed description of the databases is available in the Electronic Supplementary Information.

**Table 3.1. Sources of input data of the global crop water databases WaterStat and Pfister et al. (2011)**

Reference	Crop coefficients :		Climate data :			Soil data	Fertilizer	Yield	Irrigation data
	K <sub>c</sub>	Growing phases and date	Precipitation	Temperature	ET <sub>o</sub>				
<b>Pfister et al (2011 and 2014)</b>	Allen et al. (1998)	Defined according to the climate (six climate zones) (Chapagain and Hoekstra 2004)	Monthly precipitation: CRU 2.0 TS database (Mitchell and Jones 2005)	-	Global map of monthly reference evapotranspiration (FAO 2004)	-	-	Effective yield in the year 2000 on a 5 arc-minutes grid (Monfreda et al. 2008)	% irrigated cropland: combining % cropland share (Ramankutty et al. 2008) and % irrigated area (Siebert et al. 2007)
<b>Water Stat, Mekonnen and Hoekstra (2011)</b>	Allen et al. (1998)	- Crop calendar tool (FAO 2008) - Crop planting dates (Sacks et al. 2010) - Global monthly irrigated and rainfed crop areas Mirca2000 (Portmann et al. 2010) - World crop areas and climatic profiles (USDA 1994) - Defined according to the climate (Chapagain and Hoekstra 2004)	Daily precipitation: based on monthly average values (Mitchell and Jones 2005). using the CRU-dGen daily weather generator model (Schuol and Abbaspour 2007)	CRU-TS-2.1 (Mitchell and Jones 2005), period 1996–2002, spatial resolution of 30 by 30 arc minute	Global map of monthly reference evapotranspiration (FAO 2004) Monthly average data converted to daily values by curve fitting to the monthly average through polynomial interpolation	Total available water capacity of the soil: average value of the five soil layers of ISRIC-WISE database (Batjes 2006)	Country-specific nitrogen fertilizer application rates by crop, based on : (Heffer 2009) (FAO 2006) (FAO 2009)	Actual yields are calculated by the model. (Relies on: maximum yield = national average x 1.2 (Reynolds et al. 2000) and crop yield response factor (Doorenbos and Kassam 1979))	Irrigated fraction of harvested crop areas for 24 major crops : Mirca2000 (Portmann et al. 2010) For the other 102 crops: data for “other perennials” and “other annual crops” in Mirca2000 database

## 3.2.2. Limitations of water inventory and agri-food LCA databases

### 3.2.2.1. Input data

All databases rely on the FAO 56 approach with a single crop coefficient:  $ET_c = K_c \times ET_o$  (Equation 1) (Allen et al. 1998)<sup>5</sup> (Fig 3.3). This approach relies on two main hypotheses:  $ET_o$  represents all effects of weather, and  $K_c$  varies predominantly with crop characteristics and only marginally with climate.

Regarding the reference evapotranspiration, Pfister et al. (2011) and Mekonnen and Hoekstra (2011) are using monthly  $ET_o$  from the FAO (2004) global map (Table 3.1). These values are monthly averages for the period 1961-1990, thus they do not include recent climate data. They were obtained from the FAO Penman-Monteith method, but with the *limited climatic data approach*, described in Allen et al. (1998). It means that missing climatic data are estimated thanks to known climatic data (e.g.: radiation data can be derived from air temperatures), or extrapolated from a nearby weather station. Allen et al. (1998) recommended estimating missing climatic data, and use the FAO Penman-Monteith equation, instead of using another  $ET_o$  calculation procedure requiring limited climatic parameters such as the Hargreaves equation (Hargreaves and Samani 1985) used by Liu and Yang (2010). The monthly  $ET_o$  data are then converted to daily values by curve fitting to the monthly average through polynomial interpolation by Mekonnen and Hoekstra (2011).

Regarding the crop coefficient,  $K_c$  should represent an aggregation of the physical and physiological differences between crops and the hypothetical reference crop<sup>6</sup> (Allen et al. 1998).  $K_c$  values on which databases are based are from Chapagain and Hoekstra (2004) (Table 3.1). If the planting date and calendar were adjusted according to the climate (six global climate zones are considered), the  $K_c$  values however are the same whatever the climate, and originate from Table 12 in Allen et al. (1998). But these widely used  $K_c$  values are only valid for “*non-stressed, well-managed crops in subhumid climates: daily minimum relative humidity  $\approx$  45%, wind speed at 2 m above ground surface  $\approx$  2 m/s*”. Thus,  $K_c$  largely varies depending on climatic variables (wind speed, radiation, and air vapour pressure deficit) (Katerji and Rana 2014), in particular for tall crops for which the  $K_c$  value may vary by up to 30% (Allen et al. 1998). In addition, the origin of  $K_c$  values proposed by Allen et al. (1998) are not correctly identified (Katerji and Rana 2014) and are average values of contradictory data (Doorenbos and Pruitt 1977). As a result, these widely used  $K_c$  values should be considered with caution.

It is important to bear in mind the validity domain of  $K_c$  and  $ET_o$ , in particular because they have a strong influence on the results. Indeed, Zhuo et al. (2014) showed that the crop blue water footprint is most sensitive to  $ET_o$  and  $K_c$ . For example, in a case of a +15% change in  $K_c$  values, the blue water footprint of maize can change by up to +53%. This study also showed that uncertainties in total water footprint as a result of all uncertainties considered (i.e.: precipitation,  $ET_o$ ,  $K_c$  and crop calendar) are on average  $\pm 30\%$ . Nevertheless, this study was not addressing the uncertainty and variability associated with the irrigation volume, or the model uncertainty, which can also influence a lot the results.

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<sup>5</sup> The single crop coefficient is used in the Cropwat model.

<sup>6</sup> The hypothetical reference crop has a crop height of 0.12m, a fixed surface resistance of  $70 \text{ s.m}^{-1}$  and an albedo of 0.23. It resembles closely to an extensive surface of green grass of uniform height, actively growing, completely shading the ground and with adequate water.

### 3.2.2.2. Modelling approach

First, it is important to notice that calculations are based on the assumption that soil is homogeneous since it is considered as a uniform reservoir. Actually, soil is heterogeneous and could be segmented in several soil horizons for the purpose of modelling.

Second, it is paramount to discuss the differences between crop water use modelling under standard vs. non-optimal conditions. The water consumption of irrigated crops is calculated based on standard conditions in all databases: assuming that crop evapotranspiration  $ET_a$  is equal to potential crop evapotranspiration  $ET_c$ , but this is inappropriate for any crop growing under non-standard conditions. Pfister et al. (2011) attempted to account for possible deficit irrigation, but the calculation mode was rather rough. When plants are stressed (due to a lack of water or nutrient, or an excess of salts), transpiration slows down so that actual ET is less than potential ET (Perry 2014). Thus, many estimates of crop water consumption are excessive, and the over-estimation is assigned to the blue water component because the contribution of precipitation is fixed (Perry 2014). Conversely, the opposite situation may happen: if additional water is brought to lixiviate salts (to avoid crop saline stress), the crop water consumption is underestimated. Actually, the non-inclusion of possible water, saline or nutrient stresses correspond to the non-inclusion of water quality in the modelling approach.

Third, databases lack information about the quality of the water flows. This raises two limitations: the water quality is not accounted for as a factor influencing the volume of water loss through evapotranspiration, and this is hampering the assessment of water degradation impacts (with Boulay et al. (2011b) method for example). Indeed, the input water quality is not considered in databases. Regarding the out-coming water flows, pieces of information might be found. Thanks to the grey water footprint, Mekonnen and Hoekstra (2011; 2014) provide data on nitrate loss in drainage water from the applied nitrogen. However, this is a rough estimate not considering factors influencing leaching and runoff such as precipitation, soil properties, or processes of the nitrogen cycle responsible for nitrogen emissions in the air (Mekonnen and Hoekstra 2011). They assume 10% of applied nitrogen fertilizers is reaching free flowing water bodies, following Chapagain et al. (2006), but this assumption is based on a study on cotton approximating that 20% of the applied nitrogen is leached, lost through runoff or denitrified (Silvertooth et al. 2001). It is not possible to deduce from the Water Stat database a concentration of nitrate in the runoff and leaching water because no information is given on the volume of water released: the authors only mention that part of the irrigation water applied can percolate or run-off. In Ecoinvent v3, water quality data can be retrieved from emissions to water and resource use from water for each elementary process. Mekonnen and Hoekstra (2011) mention an innovative approach that could be useful for future assessment of water quality: an estimation of global nitrogen flows of 6 nitrogen inputs and 5 nitrogen outputs including nitrogen leaching at high resolution (Liu et al. 2010).

Finally, the estimation of water flows withdrawn and released relies on several important assumptions to keep in mind (in the case of an application of Boulay and colleagues' method for example). In all databases, water withdrawal is estimated from water consumption thanks to irrigation efficiency. However, the volume of water actually withdrawn depends on irrigation management: the farmer's expertise and potential environmental constraints. Regarding the released water flows, only *Quantis Water Database* (Quantis 2015) and *Agri-footprint®* (Blonk Agri-footprint 2014) databases are

distinguishing runoff and drainage water flows. But the partitioning is based on estimates without justifications. Overall, the water flows redistribution (e.g. runoff water going to another field) is not accounted for, yet, agricultural practices on field may influence a lot the redistribution of water flows at the watershed scale (e.g. grass strips vs. bare soil in buffer zones).

### 3.2.2.3. Yield

The crop yield is either based on publication (Pfister et al. 2011) or calculated by the model (Mekonnen and Hoekstra 2011). Mekonnen and Hoekstra (2011) estimate actual yield based on a model accounting only for the effect of water deficit on yield reduction, neglecting other factors such as cultivar, salinity and nutrients (Mekonnen and Hoekstra 2011). This model is based on the assumption that reduction in evapotranspiration due to water stress (from  $ET_c$  to  $ET_a$ ), is linearly correlated with yield reduction (from  $Y_{max}$  to  $Y$ ), with a certain sensitivity represented by the crop yield response factor ( $K_y$ ). This calculation relies on two controversial concepts:  $Y_{max}$  and  $K_y$ . The maximum yield is, from an agronomic viewpoint, a theoretical concept, and its estimation is very hazardous. Mekonnen and Hoekstra (2011) computed  $Y_{max}$  by multiplying the national average yield of the crop by a factor 1.2 which comes from Reynolds et al. (2000). However, the factor 1.2 approximated maximum yields of maize in Kenya (Reynolds et al. 2000), and is probably not appropriate for other crops in other locations. Then, the calculated actual yield is scaled up to fit national average yield statistics (from FAOSTAT, referring to 1996-2005 period), thus actual yield becomes closer to national average.

The  $K_y$  concept is also controversial.  $K_y$  values can be found in Doorenbos and Kassam (1979), and represent the effect of a reduction in evapotranspiration on yield losses. However,  $K_y$  cannot be applied for making accurate estimate of yield response to water because one empirical value cannot integrate the complex and highly variable mechanisms involved between production and water use for crop production. Studies determining  $K_y$  values have shown a wide range of within-crop variations (depending on the environment, growth stage and the “memory effect”), sometimes as large as the variation between crops (Steduto et al. 2012).

### 3.2.2.4. Validity domain of databases

In summary, databases are providing estimates of water consumed by a crop based on the theoretical crop water consumption, and not on the water actually withdrawn and consumed, and rely on data and modelling approaches presenting important limitations regarding the implications of farming practices and the complex mechanisms of field water flows. Moreover, the water inventory available in databases does not allow the assessment of water use impact with all current life cycle impact assessment methods especially those addressing the impacts of both water consumption and degradation (Boulay et al. 2011b; Bayart et al. 2014), since water quality is not provided. These databases are relevant and can be used when the cropping system is at the background level, keeping in mind the limitations mentioned above. But they are too generic to distinguish precise causes of impacts due to the cropping system itself as needed in most agricultural LCA studies where it is at the foreground level (e.g. LCA of conventional vs. organic crop in a given country). Thus another approach is required. We analysed in the following alternative modelling options and provide recommendations.

### 3.3. Modelling options for field water flows inventory in agricultural LCA studies

The above-described databases were based on models and do not require any data collection except knowing the crop and the country. But they are not fulfilling the objectives of an agricultural LCA. Tillotson et al. (2014) encourage the use of on-site measurements for a more reliable and accurate water footprint. However, this is not possible for LCA studies, even when the agricultural system is at the foreground level. In the following sections, we first describe the model specifications to perform the inventory of water flows in agricultural LCA. Then we analyse the suitability of various model categories showing a gradient of accuracy and complexity against the constraints of the study in terms of resources (technical skills, time) and data available. Two important aspects are distinguished in this analysis: the model rationale and the source of the data (model inputs and parameters).

#### 3.3.1. Model specifications description

##### 3.3.1.1. General objectives of the modelling

The main characteristics of a model designed for estimating field water flows for agricultural LCA are: (i) ability to simulate ET, deep percolation and runoff (ii) flexibility for the simulation of different crops (all crops) under a variety of pedo-climatic conditions and agricultural managements, and (iii) minimum data requirements. The complexity lies in the fact that the model should be generic (not crop specific) but capable of accounting for soil, climate and agricultural management specificities without requiring too much data. It is important to notice that our objective differs from the initial objectives of most water flows and crop models. We do not want to determine the crop water requirement to maximize the yield, but, knowing the amount of water applied by the farmer and the actual yield, we want to know how much is actually consumed and released back to the environment.

The key parameters that the model should account for and the modelling formalisms depend on the objectives of the LCA. For example, in a comparative LCA of two orange production systems in the same region, differing only from their irrigation mode (e.g.: drip versus surface irrigation), it is necessary to be able to discriminate the two systems and thus account for the effect of the irrigation mode on the water flows, especially in an ecodesign procedure. Conversely, the effect of the irrigation mode might be negligible in a comparative LCA of orange juices where orange production systems contrast in their location (arid Morocco vs. humid-subtropical Florida), management technique (conventional vs. organic), crop variety and yield, especially in a food ingredient sourcing procedure. In the following we will focus on agricultural LCA studies where the objective is eco-design. We will afterwards put things into perspective to propose a matrix of criteria for the selection of the appropriate method for the inventory of field water flows in Agri-food LCA studies taking account of their study objectives and their data and resources available.

##### 3.3.1.2. Model specifications for cropping system eco-design

For the eco-design of cropping systems with LCA, the model should provide estimates of water consumed (evaporated and transpired), water released in surface water (runoff) and groundwater (deep



percolation), and the variation of water stock in the soil. Among the parameters that the model should account for, we highlight in the following the crucial ones that were not accounted for in the reviewed databases (Cf. part 3.2). First, the amount of water supply should be an input of the model (and a primary data) because it depends on the irrigation management. Second, since ET can be affected by stress factors (nutrient, water and salinity stress), it is crucial that the model accounts for these stresses as much as possible. This means that the modelling should account for water quality and nutrient inputs (Table 3.2), and a nutrient and saline budget may also be required. Third, regarding the released water, the partitioning between drainage and runoff flows is important since they will reach different water compartments. This partitioning should be based on soil characteristics (Table 3.2). Finally, as long as key crop parameters are estimated properly (e.g. rooting depth), there is no real need for a crop model since the crop yield is not an expected output of the model. Regarding the spatial scale, it is questionable whether the model should support calculations at the field scale or basin scale. Field scale seems appropriate for an inventory modelling because it corresponds to the technosphere boundary (the technosphere represents the systems under study, emitting substances to and consuming resources from the environment). However, in this case the redistribution of water flows beyond the field (e.g. a modified river flow rate) is not accounted for and should be part of the impact assessment modelling. The spatial discretisation may vary from one single unit (e.g. single soil layer), to many virtual units (e.g. several soil layers), depending on the unit at which parameters are considered homogeneous. Regarding the temporal scale, most models run at a time-step that is shorter (usually daily) than the time-step of LCA analysis (usually yearly). A daily time-step seems adequate to account for crucial parameters variations (e.g.: precipitation). Table 3.2 summarises our model specifications, and provides the recommended and default data sources for each input and parameter.

**Table 3.2. Model specifications and data sources for the inventory of water flows for the LCA-based ecodesign of cropping systems.**

	<b>Flow / parameter or input description</b>	<b>Data sources :</b>	
		<b>Recommended</b>	<b>Default</b>
<b>OUTPUTS:</b>	Water consumed : ET	Model outputs	
	Water released in groundwater: deep percolation		
	Water released in surface water: runoff		
	Water stock : soil water stock variation		
<b>ACCORDING TO:</b>			
<b>Initial conditions</b>	e.g: soil water content	Primary: measurement	Depending on the starting period for the simulation, a soil at field capacity or saturation may be assumed
<b>Crop parameters</b>	Crop coefficient $K_c$	Primary: known by the farmer or cultivar specific bibliography	$K_c$ reported value from <u>literature specific to the crop</u> : FAO 66 (Steduto 2012)
	Planting date, and length of crop cycle		Siebert and Döll (2010)
	Plant density		
<b>Irrigation management</b>	Water supply	Primary: recorded by farmer	Default value not appropriate
	Water withdrawal and irrigation efficiency (alterative to water supply)	Primary: recorded by farmer	Default value not appropriate, irrigation efficiency
	Irrigation mode	Primary: observation/ known by the farmer	Default value not appropriate
	Salts	Primary: water analysis (frequently done by farmers) or specific bibliography	- Saline aquifer: WHYMAP (BRG & UNESCO 2008); - Surface and ground water quality: EC, TDS, Major ions: Gemstat (UNEP 2009)
N, P	- Water pollution level for N and P in the world's river basins (Liu et al. 2012) - Surface and groundwater quality, heterogeneous depending on the country: UNEP (2009) GEMStat		
<b>Nutrient inputs</b>	N, P, K	Primary: recorded by the farmer	- Fertiliser Use by Crop at the Global Level : (Heffer 2013), - FAO FertiStat database, - International Fertilizer Association database (IFA 2013)
	Salts in fertilizers	Bibliography	

<b>Soil physical parameters</b>	Saturated hydraulic conductivity	Primary: soil analysis	- Soil hydraulic parameters deduced from soil texture (pedotransfer functions): (Saxton and Rawls 2006)
	Soil water content at saturation, field capacity and permanent wilting point	Primary: soil analysis	- Maximum soil moisture retention, topsoil and subsoil CEC clay, texture, bulk density...,30 arc seconds: Harmonized World Soil Database (FAO/IIASA/ISRIC/ISS-CAS/JRC 2012) - Soil water capacity in the effective root zone at 5 and 30 arc minute: ISRIC (Batjes 2006)
<b>Climate</b>	Precipitation	Primary: meteorological station	- Monthly values of major climatic parameters with a spatial resolution of 30 arc minute: CRU TS-2.1 (Mitchell and Jones 2005) - Estimates of climate when no observations are available: New LocClim software (FAO 2005) - 30-year averages climatic data for CropWat model: (FAO 2010) - Mean monthly climate: precipitation, temperature, relative humidity, windspeed: (New et al. 2002)
	Temperature, Radiation, Wind, Humidity	Primary: meteorological station	- Idem Precipitation - Estimated with the FAO56 procedure:(Allen 1998)
	ET <sub>o</sub> (alterative to Temperature, Radiation, Wind, Humidity)	Primary: meteorological station	- Calculated with Pemann-Montheith equation described in Allen (1998) - FAO Global map of monthly reference evapotranspiration at 10 arc minutes, (FAO 2004) - An ET <sub>o</sub> calculator is available, data can be given in a wide variety of unit: ET <sub>o</sub> calculator (FAO 2012)

### 3.3.2. Modelling approach selection

#### 3.3.2.1. Model types

To represent the soil-plant-atmosphere continuum, a wide spectrum of models of the unsaturated zone are available (Bastiaanssen et al. 2007). These models are water flows models, including or not a crop growth model. In their review of twenty-five years modelling of irrigated and drained soils, Bastiaanssen and colleagues (2007) distinguished typical model categories: water flows can be estimated through a bucket model (using soil reservoir cascade theory) (e.g.: CropWat Allen 1998), or through Richard's equations (e.g.: Hydrus, Simunek 1998), and may include a crop production model (e.g.: Epic (bucket model)). The inclusion of water quality requires that the model accounts for both water and solute transport processes, and even geochemical processes. Model coupling is also frequent. There is a gradient in terms of model complexity and data requirement. Since a versatile and universally appropriate model does not exist, Bastiaanssen and colleagues (2007) recommend selecting a model according to the specific objectives of the study and data available. Our strategy is to find the modelling approach meeting our above-described specifications, with the minimum data requirement. Therefore, the use of the most advanced models will not be the ultimate objective because there are only useful if sufficient data is available to calibrate them properly (Bastiaanssen et al. 2007). In comparison, simpler models, having fewer assumptions, can provide a comparable degree of certainty, but with less effort (Bastiaanssen et al. 2007). Beyond a single model selection, this is a modelling approach that we aim to characterize. Thus, we did not review exhaustively all available models, but rather identified among the model categories (illustrated with example) the appropriate approach according to our objectives and constraints.

#### 3.3.2.2. Possible modelling approaches

Figure 3.4 illustrates the gradient of complexity and accuracy in the possible modelling approaches that will be discussed.

##### 3.3.2.2.1 *Pre-parametrised model*

As an intermediate between databases and models, pre-parameterized models allow a more specific assessment of water flows, with a minimum data requirement. Such a tool has been developed by Dourte et al. (2014) to easily calculate the water footprint of a crop in the U.S. This tool is based on EPIC crop growth model (Sharpley and Williams 1990) within the SWAT hydrology model (Neitsch et al. 2009), and account for climate, crop characteristics, soil, irrigation management, fertilizer application and tillage. Yield can be an input or an output data. Such a model is very relevant but should be run within the validity domain of its parameterisation. To date, this tool was developed for the U.S and 71 annual crops. The model could be adapted for use in other regions of the world, and the addition of certain perennial crops is under development (Dourte et al. 2014), but its extension for a global coverage would require tremendous data and time.

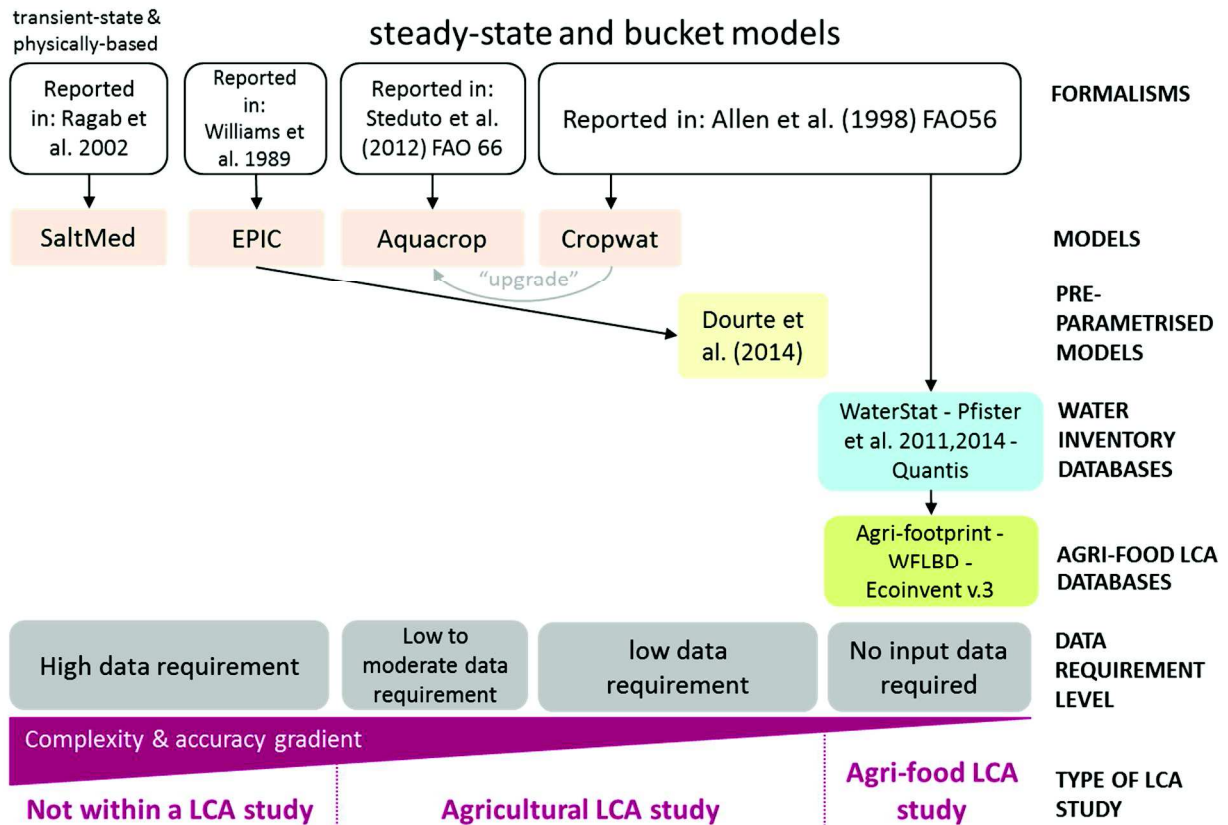


Figure 3.4. Gradient of complexity and accuracy in the different possible approaches (databases and models) for the inventory of field water flows, and associated type of LCA study.

### 3.3.2.2.2. Model gradient

Out of the validity domain of available pre-parameterized models, a model has to be selected to calculate the required water flows.

In their review of water impact assessment methods, Kounina et al. (2013) only specified that method developers should provide a method to estimate water flows with a crop model, based on input data on climate, soil, and crop characteristics. Milà i Canals et al. (2008, 2010), used the **CropWat** model to estimate crop water requirements (with climate data from CLIMWAT), and compared this value with the water supply to the plant (total irrigation supply minus irrigation efficiency). Water consumption was the minimum between water requirement and supply, and the water released was equal to the total supply minus supply to the plant. In the Water Footprint Assessment manual, Hoekstra et al. (2011) recommended using CropWat with the “irrigation schedule option”, and  $ET_{a,blue}$  was the minimum between total net irrigation and actual irrigation requirement. Using CropWat with input data on irrigation management is a better alternative to current methods used in databases since it allow accounting for actual irrigation supply. Nevertheless, the number of irrigation data inputs is limited by the software. In addition, since CropWat model is based on formalisms described in Allen et al. (1998)<sup>7</sup>, many limitations highlighted in part 3.2.2 remain: saline and nutrient stresses are not accounted for;

<sup>7</sup> It is noteworthy that only part of the formalisms described in Allen et al. 1998 (FAO56) are actually implemented in CropWat model

runoff and drainage are not dissociated<sup>8</sup>. Besides, it is important to notice that when computing the crop evapotranspiration using the CropWat model at the farmland scale, the irrigation water losses during the transport process from the water sources to cropland are not taken into account (Tillotson et al. 2014). Thus, such an approach does not allow sufficient discrimination of agricultural systems required for agricultural LCA studies. As a result, we analysed the “most advanced model” category.

Considering the quality of water flows opens a wide range of complex models. Since a proper accounting of water quality should be based on a geochemical model which cannot be used within a reasonable time in LCA, we limited water quality accounting to the effect of water quality on the balance of water flows (i.e.: a low quality water reducing ET, thus affecting the whole water balance). We restricted the quality parameters to the two potential stressors: salts and nitrate, assuming a good plant protection (no stress induced by pests and diseases). But many other pollutants are transported through water: Boulay et al. (2011) defined 136 parameters to describe water quality. It is of course not possible to model the fate of all these parameters with a single model within reasonable time-frame for a LCA study. In the most advanced model category (from a scientific relevance viewpoint), **SaltMed** (Ragab 2002; Ragab et al. 2005) is an interesting approach that could meet our requirements. It is a transient-state and physically-based model: the water and solute flows (salt and nitrogen) are mathematically described using Richard’s equations, and crop water uptake is a function of water quality. Soil nitrogen dynamics are accounted for (plant N uptake, leaching, denitrification...). This model allows an estimation of water flows accounting for water, nitrate and salt dynamics in the soil. However, this accuracy comes at a high price of fine tuning and data collection for model calibration and validation. Owing to the constant data scarcity and limited time available, we will investigate more functional approaches: steady-state and bucket models. The crop model **EPIC** (Williams et al. 1989), used by Liu and Yang (2010), presents several advantages: it is currently parameterized for approximately 80 crops, determines surface runoff (accounting for soil hydraulic properties, but not ground cover as shown by Mottes et al. (2014)), and subsurface flow volumes, estimates the severity of stresses caused by water, nutrients, and its impacts on crop growth. It also computes evaporation from soils and plants transpiration separately (Sharpley and Williams 1990). It can be coupled with SWAT, GIS. However, EPIC, presents a high level of data requirement and is difficult to use without sufficient qualifications. UNFCCC (2009) estimates 3 to 4 days of intensive training are required to be able to use the model reliably. Moreover, the EPIC model is performing well for conventional crops but there is a lack of performance reported for crops such as cassava, potato, sugar beet, groundnut, cotton, cowpea and pasture (Liu and Yang 2010). In comparison, the **CropWat** model presents the advantage of being less data intensive and is parameterized for many crops. However, it presents several limits highlighted in section 3.2.2. These limitations were overcome in the **Aquacrop** model: “a new version of CropWat” (UNFCCC 2009). So far, AquaCrop was parameterized for herbaceous crops only. AquaCrop is described in the FAO Irrigation and Drainage Paper No 66 (Steduto et al. 2012), considered as an update of FAO 33. Indeed, AquaCrop evolves from

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<sup>8</sup> Note that in the CropWat model, irrigation losses correspond to the share of net irrigation water deep percolating, and precipitation losses correspond to the share of total irrigation lost through deep percolation and runoff. Beware of a possible confusion regarding effective precipitation: the soil water balance is computed using total precipitation and not effective precipitation because DP and RO are estimated respectively as a function of soil water content and maximum infiltration rate. Effective precipitation is only calculated for the whole cropping period to calculate the crop water requirement.

the original *water production function* in FAO 33 by separating ET in non-productive soil evaporation and productive crop transpiration, and estimating biomass production from transpiration thanks to a water productivity parameter (instead of linking yield with evapotranspiration through  $K_y$ ). In addition, normalization of transpiration with reference evapotranspiration allows the application of the biomass-transpiration relationship to different climatic contexts. The calculation of crop water use in daily time steps (instead of only as the final ET) allows a more realistic accounting of water stress and crop response (Steduto et al. 2012). The model is water-driven because biomass production is determined by transpiration. The soil is considered as a reservoir whose water content changes daily according to water flows inputs (rainfall, irrigation and capillary rise) and outgoing (runoff, evaporation, transpiration, and deep percolation) (Vanuytrecht et al. 2014). This water balance depends on soil physical characteristics, climate, irrigation management practices (e.g.: irrigation mode affects soil evaporation), crop parameters and other field management practices (e.g. soil mulches). While performing the water balance, AquaCrop also performs the salt balance. AquaCrop is designed to simulate crop responses first to water, but also to nutrients and salinity stresses thanks to an indirect approach (avoiding the simulation of nutrient balances and their complex cycles that would make the model too complex). Stresses are expressed through stress coefficients ( $K_s$ ) specific of each basic growth expression, accounting for the varying sensitivity of the crop depending on the development stage. Crop phenology is simulated based either on calendar days or growing degree days which allows a more easily transferable application whatever the location of the crop. Performance review of the model concludes that in comparison with more complex models, Aquacrop performs well (Steduto et al. 2011). One limitation of the model is the yield prediction in saline conditions (Vanuytrecht et al. 2014), but this is not a problem if the objective is only to simulate water flows. Without a loss in simulation quality, Aquacrop requires less data than other complex models (Table 3.3) such as Cropsyst (Stöckle et al. 2003), Wofost (Boogaard et al. 2014), DSSAT (Jones et al. 2003) (Steduto et al. (2011). Its ease of use and limited parameterization makes it suitable for users not comfortable with the use of modelling tools (Steduto et al. 2011; Thorp et al. 2014). Nevertheless, several studies showed that calibration is a critical step in the model reliability. The calibrations already provided by the FAO for the different crops may require additional local refinements, especially in the cases of severe water stresses (Steduto et al. 2011). However, the Aquacrop modelling approach is well documented and transparent, thus fine tuning is possible. Thus, Aquacrop represents the optimal balance between accuracy, simplicity and robustness. Details of the simulated processes are provided in the report FAO 66 (Steduto et al. 2012), and in the reference manual updated regularly (Raes et al. 2012a; Raes et al. 2012b).

The main issue with AquaCrop is its limitation to herbaceous crops: barley, cotton, maize, potato, quinoa, rice, soybean, sugar beet, sugar cane, sorghum, sunflower, tef, tomato and wheat. Since tree crops present additional complexities, only guidelines regarding water management and yield estimation are provided in Steduto et al. (2012). Nevertheless, future integration of perennial crops in the model is planned.

Accounting for the salts brought by fertilizers is important since it contributes to saline stress. However, to our knowledge, such models do not exist because the interaction between fertilisers, soil-water salinity and crop are complex (this will be further discussed in the next chapter).

**Table 3.3. Required input variables for simulations with AquaCrop (Based on Vanuytrecht et al. (2014))**

<b>Parameters describing initial conditions at start of simulation</b>	Initial soil water content and soil salinity at various depths in the soil profile (soil segmentation is optional)
<b>Crop parameters likely to require adjustments for cultivar and local environment and management</b>	Planting date Plant density Maximum canopy cover Length of crop cycle Maximum effective rooting depth and time to reach it
<b>Parameters describing irrigation management practices</b>	Irrigation method Application depth and time of irrigation events Salinity of the irrigation water
<b>Soil physical parameters of the distinctive (up to 5) soil horizons</b>	Soil water content at saturation, field capacity, and permanent wilting point Saturated hydraulic conductivity Depth of layer restricting root deepening
<b>Climate</b>	Daily, 10-daily or monthly max and min temperature and rainfall Daily, 10-day or monthly ET <sub>o</sub> Mean annual CO <sub>2</sub> concentrations
<b>Groundwater table</b>	Depth and salinity of the groundwater table
<b>Parameters describing field management practices</b>	Maximum relative dry aboveground biomass expected in a fertility-stressed environment compared to stress-free conditions Cover and type of soil mulches Height of soil bund Surface runoff: On or Off

### 3.3.3. Further developments of models, tools and databases

Regarding the evolution of existing water inventory databases, the Water Footprint Network is planning to adopt the Aquacrop model to feed their database. By changing from CropWat to Aquacrop, they aim at simulating stress from limited availability of nutrients (*Hoekstra, personal communication*). However, this will not apply to perennials. For the moment, the Water Footprint Assessment Manual (Hoekstra et al. 2011) recommends for perennial crops the use of the average annual crop water use over the life span of the crop divided by the total yield over the lifespan.

Dourte et al. (2014) stated that the AgroClimate WaterFootprint tool will include perennial crops in the future with no more precise deadline.

Regarding the evolution of models, Aquacrop will progressively support the modelling of perennial crops, olive tree being the first one to be included (E. Fereres, pers. comm., 2015), thanks to the compilation of many studies. For example, Rallo et al. (2012) provided the parameters for the water stress function (the critical threshold of soil water content below which a reduction in olive tree



transpiration occurs), and defined the curve shape thanks to experimental data. Another key evolution of water flows and crop model would be their adaptation to better fulfil environmental assessment study objectives, which is not the usual objective for this community of model designers. The status of yield is particularly emblematic of this divergence in modelling objectives. Water flows and crop models were designed for maximising yield with limited water resources, or study the effect of climate change on yield, but also as a learning tool. This is different from estimating the water flows to the environment based on given agricultural practices. Since the mechanisms are the same, we can hopefully expect that these models could be adjusted to fit our requirements and, in particular, use yield as an input data to be assimilated by the model.

Beyond model development, efforts should focus on data to feed these models (Bastiaanssen et al. 2007). Remote sensing data are a promising and suitable alternative to field data and can provide required input variables for models (Bastiaanssen et al. 2007), which is particularly relevant in contexts where data are very scarce. The distribution of remote sensed data presents a high temporal frequency across vast spatial areas, are publicly available and well documented (Karimi and Bastiaanssen 2014). But using such data raises the challenge of appropriately interfacing remotely sensed data with the models (Thorp et al. 2014). A solution was provided for Aquacrop: Although the model is designed to run at field scale, features to plug the model in GIS and remote sensing platforms are developed to support simulations up to the regional level (Vanuytrecht et al. 2014, Steduto et al. 2011). Lorite et al. (2013) recently developed two tools for managing the inputs and outputs of AquaCrop, named AquaData and AquaGIS, respectively. Remote sensing is playing an increasing role in the expansion of  $K_c$  database (cultivar and climate dependent), next to measurements (ET measured by lysimeter, eddy covariance, Bowen ratio) (Pereira et al. 2015).

Another important achievement is to make simulation models trustworthy and easy to use by farmers. Because there are the ones who have decisive actions and technical solutions (Bastiaanssen et al. 2007). Toward this direction, a simplified interface of Aquacrop is being developed for its diffusion and use by producers (E. Fereres, pers. communication, 2015). However, we would like to temporise the use of models for farm management. In their review of twenty-five year modelling of irrigated and drained soil, Bastiaanssen et al. (2007) reminded us not to forget that *the rural communities' indigenous knowledge is often more finely tuned to the local conditions than is a numerical model*.

Finally, it is noteworthy that this work did not address some LCA methodological questions related to the use of models. For example, the status of crop residues within the crop rotation is a crucial methodological question that cannot be answered by a water flow model but by the LCA practitioner himself. This was considered beyond the scope of this paper.

### 3.4. Conclusion

This work provides LCA practitioners and researchers with a matrix of criteria for the selection of the appropriate method for the inventory of field water flows in Agri-food LCA studies, taking account of their study objectives and their data and resources available (Table 3.4).

Water inventory and agri-food LCA databases are supporting the application of methods to assess the impacts of water consumption (but not water degradation), for agricultural systems at the background level. They provide only theoretical crop water consumptions that may differ a lot from water actually withdrawn and consumed, and rely on data and approaches presenting important limitations, making them inappropriate for agricultural LCA studies where an adaptation of practices is sought. For such LCA studies, we investigated adapted modelling approaches for a proper inventory of field water flows. Not all available models were reviewed exhaustively, but we rather analysed the various model categories illustrated with examples. For this LCA category, we provided recommendations for a consistent inventory of field water flows of cropping systems, ensuring the assessment of water use impacts related to both water use and degradation. The estimation of water flows should be based on a model simulating ET, deep percolation and runoff accounting for crop specificities, pedo-climatic conditions and agricultural managements. Thus, the modelling approach should be able to account for possible water, saline and nutrient stresses since this is affecting crop growth and thus water flows. The yield is also affected but we recommended to not use simulated yield, even if this is one of their initial objectives. Yield is a crucial data that should be collected as a primary data, and should not be estimated with a model, in particular when non-optimal conditions cannot be accounted for. An accurate estimate of ET relies on a separate assessment of Evaporation and Transpiration; furthermore, distinguishing productive water (transpiration, related with yield) from non-productive water (evaporation) is also a valuable information for water management, and is in accordance with the ISO norm on water footprint (ISO14046, 2014). The water flows released in surface and groundwater should be distinguished (since these water compartments may suffer from a different water scarcity), and estimated according to the systems specificities.

The water flows and crop growth model presenting the optimal balance between accuracy, simplicity and robustness was the FAO Aquacrop model: the updated version of CropWat. Its minimum data requirement makes this model suitable for a use in LCA, and its future evolution, supported by a large scientific community, will improve its performances and coverage (both in terms of crops parameterized and flexibility for use with SIG and remote sensing data).

The use of a simple water flows model to serve LCA is promising because it can not only help identifying mitigation options to reduce the impacts of water use, but also improve communication between scientists, farmers and other stakeholders (Van der Laan et al. 2015). For Herbaceous crops, Aquacrop constitutes a relevant and operational model, but no dedicated model is available to-date for perennial crops. Based on the present work, the design of a tailored model for the inventory of field water flows in perennial cropping systems will be investigated in future research.

Table 3.4. Database of modelling approaches for the assessment of water flows according to the objective of the Agri-food LCA study, the data available, the resources and time available

	WATER INVENTORY DATABASES	PRE-PARAMETRISED MODELS	MODELS		
			Functional model: CropWat	Functional model: e.g. Aquacrop	Physical model: e.g. SaltMed
<b>Objective of the LCA study</b>	<ul style="list-style-type: none"> <li>- Agri-Food LCA: Agricultural systems as background process (e.g. cotton tee-shirt)</li> <li>- Impacts assessment of water consumption (and water degradation possible but with default assumption).</li> <li>- If background agricultural systems turns to be a hotspot: need for a more accurate assessment</li> </ul>	<ul style="list-style-type: none"> <li>- Agricultural LCA studies: discrimination of agricultural systems characteristics (depending on the model parametrisation), for the context/country where the model was parametrised</li> <li>- Impacts assessment of water consumption, and if the model allows it, water degradation</li> </ul>	<ul style="list-style-type: none"> <li>- Agricultural LCA studies: discrimination of agricultural systems main characteristics (e.g. average soil type)</li> <li>- Impacts assessment of water consumption (&amp; water degradation possible but with default assumption for quality and returned flow partitioning).</li> </ul>	<ul style="list-style-type: none"> <li>- Agricultural LCA studies: fine discrimination of agricultural management practices (e.g. irrigation mode, soil layer type)</li> <li>- Impacts assessment of water consumption and water degradation (application of Boulay et al. 2011a, b method)</li> </ul>	<ul style="list-style-type: none"> <li>- Agricultural LCA studies: fine discrimination of agricultural management practices (e.g. irrigation mode)</li> <li>- Impacts assessment of water consumption and water degradation (application of Boulay et al. 2011a, b method)</li> <li>- Estimation of nitrogen field emissions</li> </ul>
<b>Data available</b>	<ul style="list-style-type: none"> <li>Very low data availability:</li> <li>- Crop and country</li> <li>- Yield not known</li> </ul>	<ul style="list-style-type: none"> <li>- Minimum data availability e.g. location, soil (texture, organic matter), crop, rooting depth, tillage type, irrigation (depth, frequency), yield simulated or user input (Dourte et al. 2014)</li> <li>- Possible use of databases for missing data</li> </ul>	<ul style="list-style-type: none"> <li>Low data availability:</li> <li>- Default input data possible</li> <li>- Yield known (even if calculated by the model, do not use it)</li> </ul>	<ul style="list-style-type: none"> <li>- From low to moderate data availability, depending on the way the model is run. (e.g: Table 3.3)</li> <li>- very limited model calibration and validation</li> <li>- Default input data possible</li> <li>- Yield known (even if provided by the model, do not use it)</li> </ul>	<ul style="list-style-type: none"> <li>Measurements are required to calibrate and validate the model in the context of use over one crop cycle (at least)</li> <li>- Default input data possible</li> <li>- Yield known (even if calculated by the model, do not use it)</li> </ul>
<b>Resources (time &amp; technical skills)</b>	<ul style="list-style-type: none"> <li>- Minutes to collect the data from the databases, additional time to collect default data on water quality if required</li> <li>- No technical skills</li> </ul>	<ul style="list-style-type: none"> <li>- Hours: simplified user interface</li> <li>- No technical skills</li> </ul>	<ul style="list-style-type: none"> <li>- Hours: very friendly and intuitive model interface</li> <li>- No technical skills or expertise required</li> </ul>	<ul style="list-style-type: none"> <li>- Day: very friendly and intuitive model interface</li> <li>- Very low technical skills required</li> </ul>	<ul style="list-style-type: none"> <li>- Weeks to learn how to use the model, months to settle measurements to have data to calibrate and validate it for the specific crop &amp; pedo-climatic context</li> <li>- Expertise and technical skills required</li> <li>- Will not generally be used in LCA</li> </ul>

### Acronym list Chapter 3

- DP: Deep percolating water flow [mm]  
ET<sub>o</sub>: reference evapotranspiration [mm per unit time]  
ET<sub>c</sub>: crop evapotranspiration under standard conditions [mm per unit time]  
ET<sub>a</sub>: actual crop evapotranspiration [mm per unit time]  
ET<sub>green</sub>: evapotranspiration of green water [mm per unit time]  
ET<sub>blue</sub>: evapotranspiration of blue water [mm per unit time]  
ET<sub>deficit blue</sub>: deficit irrigation blue water consumption through evapotranspiration, according to Pfister et al. (2011)  
ET<sub>expected blue</sub>: full irrigation blue water consumption through evapotranspiration, according to Pfister et al. (2011)  
K<sub>c</sub>: crop coefficient [dimensionless]  
K<sub>s</sub>: transpiration reduction factor due to water, salinity or nutrient stresses [dimensionless]  
K<sub>y</sub>: crop yield response factor [dimensionless]  
P<sub>eff</sub>: effective rainfall [mm]  
RO: Runoff water flow [mm]  
S: Soil water stock [mm]  
Y<sub>max</sub>: maximum yield [tonne/ha or kg/ha]  
Y: actual yield [tonne/ha or kg/ha]

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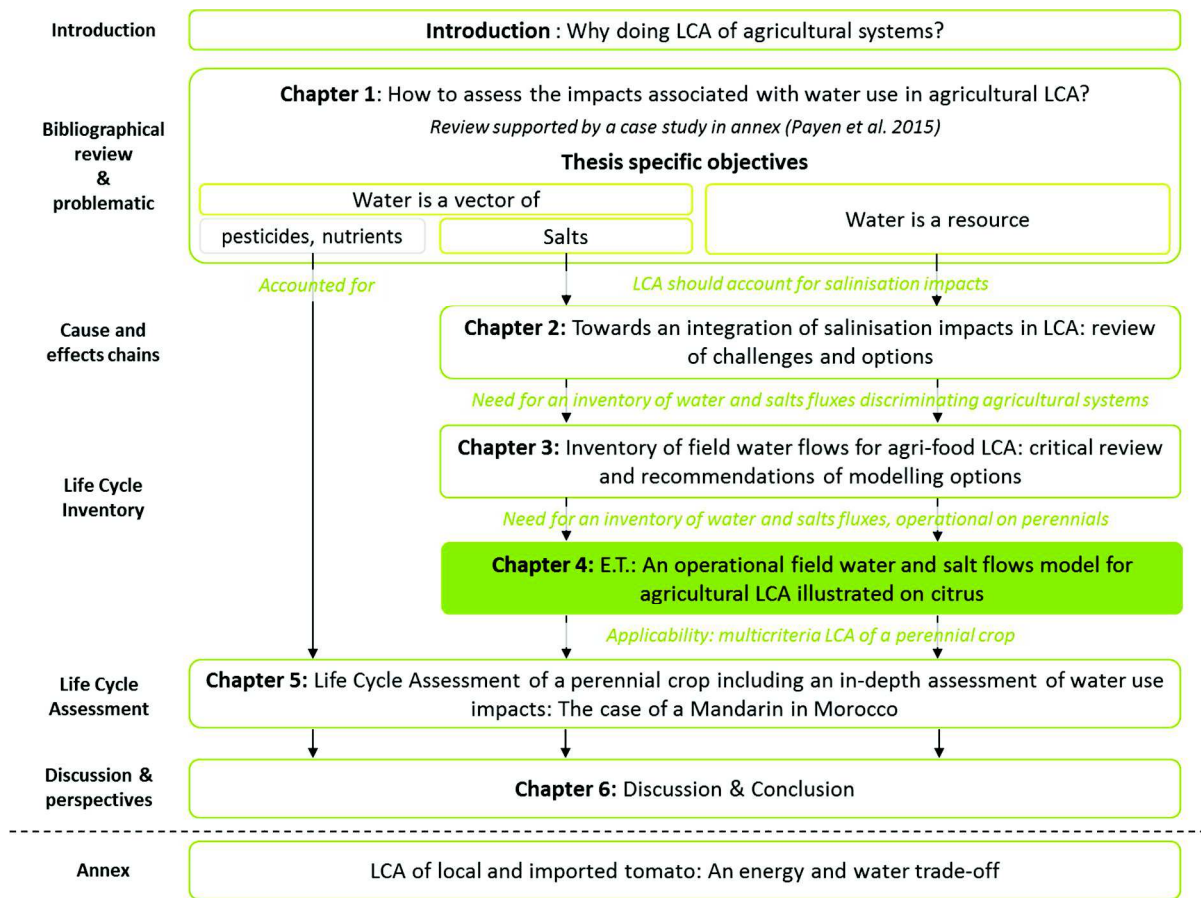


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Many water flows and crop models are available for annual crops, but for perennial crops the models are scarcer. In practice, there is a need for an operational model for the inventory of water flows for perennial crops in order to feed the LCA of these crops. Chapter 4 address the question: how to do a simple inventory of field water and salt flows for perennial crops accounting for key parameters for these crops?



## Chapter 4

### E.T.: An operational field water and salt flows model for agricultural LCA illustrated on citrus

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Sandra Payen <sup>1,2</sup>, Claudine Basset-Mens <sup>2</sup>, François Colin <sup>3</sup>, Pauline Roignant <sup>2</sup>, Sylvain Perret <sup>4</sup>

<sup>1</sup>ADEME, 20 avenue du Grésillé, F-49004 Angers, France

<sup>2</sup>CIRAD, UPR Hortsys, ELSA – research group for environmental life cycle sustainability assessment, Boulevard de la Lironde, F-34398 Montpellier, France

<sup>3</sup>Montpellier SupAgro, UMR LISAH, place Pierre Viala, F-34060 Montpellier, France

<sup>4</sup>CIRAD, UMR G-Eau, F-34398 Montpellier, France

#### Abstract

**Context and purpose:** In agricultural LCA, the estimation of field water flows (in terms of volume and quality), should be based on a water flow model since water inventory and agri-food LCA databases are not suitable. Indeed, databases do not support the application of methods assessing the impacts of consumptive and degradative water use, and are not appropriate for LCA-based Ecodesign of cropping systems. For herbaceous crops, the FAO Aquacrop model constitutes a relevant and operational model, but no dedicated model is available to-date for perennials. **Methods:** We elaborated a tailored model, so called E.T. model, for the inventory of field water and salt flows for use in LCA of cropping systems (annual and perennial crops). The model is a simple daily water and salts balance, accounting for specific soil, climate and agricultural practices (notably the irrigation mode and actual irrigation supply) and accounting for possible water and salinity stresses. The E.T. model is a modular and original integration of existing and robust concepts for water balance with more recent modules for transpiration estimation. We explored and discussed the model relevance and robustness in a case study of Mandarin crop grown in Morocco, based on farm primary data, and compared the model outputs with the literature and databases. Finally, we presented the model improvement options for the future. **Results and discussion:** The E.T. model is operational and estimate evaporation, transpiration, deep percolating water and runoff water, in terms of volume and salinity. The E.T. model outputs compared well with literature and measurements, and allowed the simulation of scenarios of agricultural practices. This first testing of the model demonstrated the discriminating power of the model, its low sensitivity to key parameters, and the importance of the crop coefficient value. Its validity domain, in terms of agricultural practices and natural environment (aquifer depth, salinity level), and accuracy could be extended thanks to the recommendations provided. A comparison with crop water consumption estimates from databases highlighted the major issue of green and blue water partitioning imposed by the framework of water use impact assessment in LCA and in the Water Footprint Network. **Conclusion:** E.T. model outputs are water elementary flows that can serve as a basis for assessing the impacts of consumptive and degradative water use in LCA, and should be the next step of this work.

**Keywords:** Model, Water flows, Salts, Evapotranspiration, LCA, Perennials

## 4.1. Introduction

As all other food products traded globally, fruits and other perennials' products are under growing scrutiny regarding their environmental impacts (Bessou et al. 2013; Cerutti et al. 2014).

Among other methodological gaps and challenges, Bessou and colleagues (2013) identified in their review of LCA studies for perennial cropping systems that water use and water use impacts are seldom calculated although most of them are irrigated. When water use is included, very little is said about the method and data used for its inventory. Even among the most recent LCA studies of perennial crops, the Brazilian cashew (Brito de Figueirêdo et al. 2015) or the Californian wine (Steenwerth et al. 2015) only reported the volumetric water consumption and did not provide the water use impacts. In these studies, the estimation of water consumption was based on a percentage of maximum crop evapotranspiration (Steenwerth et al. 2015), or was not detailed in Brito de Figueirêdo et al. (2015). Similarly on citrus, the second largest traded fruits throughout the world, the study from Sanjuan et al. (2005) on Spanish orange only reported the volumetric water consumption and did not provide the water use impacts (CIRAD 2012).

Water inventory and agri-food LCA databases are being developed to support private and public innovation for more eco-friendly agri-food products. However, in a previous work, a thorough critical review of the models and data they rely on revealed their inadequacy for supporting LCA-based eco-design of agricultural products especially related to water use impacts. Indeed, databases provide estimate of crop water consumption based on theoretical conditions, and do not provide the volume of water flows released in the environment. Furthermore, since quality data of the water flows is not considered in databases, the application of methods assessing the impacts of water degradation such as Boulay's method (2011b) is not possible. Accounting for the quality of water flows is crucial because: (i) it has an influence on the crop transpiration, then on the whole water balance (e.g.: high salinity reduce crop transpiration), and (ii) released water flow may have a degraded quality (e.g.: high salinity of percolating waters). Indeed, among the substances that water transport, salts are critical because they not only reduce yield, but also has impacts on the environment (Chapter 2). The salinity level of soil water and released water flows are crucial information for the environmental impact assessment since they contribute to agricultural soil degradation and water bodies' pollution. Citrus, a salt-sensitive crop, illustrate perfectly these productivity and environmental issues (Maas and Hoffman 1977; Syvertsen and Garcia-Sanchez 2014). A review of water flows models (chapter 3), also revealed that the recent FAO model Aquacrop is a relevant tool for estimating water and salts flows for the inventory of agricultural LCA, thanks to an optimal balance between accuracy, simplicity and robustness. Nevertheless, this model is operational for annual crops, but no similar model exists to-date for perennial crops.

Why there is a lack of model for water and salt fluxes for perennials? Not only perennials have a more complex behaviour than the major annual crops, but they have also been less studied. That is why it was not straightforward to build a simple and robust dynamic simulation model of the yield response to water for perennials similarly to Aquacrop for annual ones (Steduto et al. 2012). Regarding citrus, the usefulness of the yield to water consumptive use function (so called water production function) is

limited because most published studies of the impact of irrigation on citrus production only report applied water (many do not even report that), and do not attempt to quantify consumptive use (Steduto et al. 2012). Since estimating the yield was not our objective, we could have used only the “water balance module” of Aquacrop. Unfortunately, the model does not handle a variable transpiration coefficient (or basal crop coefficient  $K_{cb}$ ) for the moment, which seems crucial for the estimation of transpiration for citrus (Taylor et al. 2015). The inclusion of perennials in Aquacrop is planned, the first most probably being Olive tree (Ferreles, E., pers. communication).

Nevertheless, the need for an operational tool for field water flows in perennial cropping systems is urgent, even in contexts of data scarcity. In a previous work (Chapter 3) we defined the model specifications for an accurate and operational inventory of field water flows to be used in agricultural LCA studies and compatible with all LCIA methods. In the present work, we thus developed the E.T. model which is operational for all crops. The E.T. model accounts for key parameters driving field water flows in terms of practice, soil and climate. In particular, water and salinity stresses are accounted for.

The objectives of this article are to:

- present to LCA practitioners the E.T. model for the inventory of field water and salt flows for use in LCA of cropping systems (annual and perennial crops),
- explore and discuss the model relevance and robustness in a case study of Mandarin crop grown in Morocco.
- present the model improvement options for the future.

## **4.2. Material and method: field water and salt flows model presentation**

### **4.2.1. Model specifications and general principles**

The E.T. model is a modular and original integration of existing and robust concepts for water balance with more recent modules for transpiration estimation (Fig. 4.1). Thus, we did not create a new model from scratch, but elaborated a tailored model fulfilling our objectives as recommended by Affholder et al. (2012) and implemented by Langevin et al. (2015) for a specific application of LCA to slurry application techniques. Our model objectives were: estimating field water and salts flows for use in an LCA-based eco-design of cropping systems. More specifically, as defined in chapter 3, the model should: (i) simulate Evapotranspiration (ET), deep percolation and runoff (ii) allows the simulation of all crops (annuals and perennials) under a variety of pedo-climatic conditions and agricultural managements, and (iii) have minimum data requirements. Starting from existing models, we got rid of crop growth module for yield estimation (surveyed primary data) and added modules relevant for our objectives (e.g. E and T partitioning, water application as a daily input, Runoff (RO) and Deep Percolation (DP) partitioning). One paramount model feature is the assessment of water flows based on known irrigation water application or withdrawal. Indeed, the effective irrigation water applied to the field might be greatly different from the theoretical crop water need. Of course, this is not relevant for rain-fed crops.

Regarding the water quality, the model should account for salinity through two aspects: its effect on the water balance, and its effects on the environment through emissions (this will then be considered in the

impact assessment when LCIA models will be available – see chapter 2). Indeed, salinity of soil water may reduce evapotranspiration (due to osmotic effect), thus affecting all other water flows. The effect of salinity on plant nutrition is not accounted for since this would require a nutrient budget and the modelling of complex interaction of salinity, nutrient, water and the crop (this will be further discussed in part 4.3.5.2.). The model should also consider the salinity of the water flows released in the environment (runoff and deep percolation) and salinity of the soil.

Since complexity is not a guarantee for quality, simplicity was a guideline in the model elaboration. As demonstrated by Stirzaker et al. (2010) *“it is often better to have a simpler model which we understand, and understand the limitations of, than a complex one we do not understand”*. In addition, with a complex model describing accurately the processes, there is a greater risk of introducing errors through incorrect parameterisation (van der Laan et al. 2015). The model was developed in Microsoft Excel for the sake of transparency, ease of use, and future adaptation. This is a paramount aspect: this model answers a question at a specific time, and is meant to evolve with future developments.

### 4.2.2. E.T. Model description

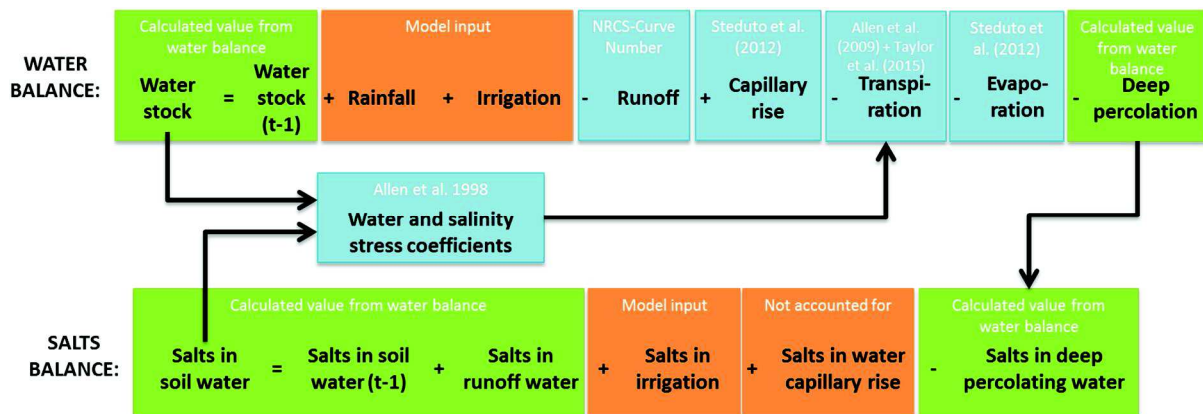


Figure 4.1. The modular structure and main formalisms for the E.T. model. NRCS: Natural Resources Conservation Services

In a schematic way, soil is considered as a uniform reservoir in which the soil water  $S(t)$  and salt content  $S_{\text{salts}}(t)$  changes as a result of incoming and outgoing water and salts flows (Fig. 4.1). The model consists in daily water and salts balances. The soil upper boundary is the atmosphere, and lower boundary is the rooting depth. This is a physically-based model since it is based on mass conservation equations<sup>9</sup>, but also includes empirical formalisms (i.e. equations and relations historically based on experimentation) for the different terms in the balance described below.

<sup>9</sup> By definition, a hydrologic physical model should rely on both mass and energy conservation. Here, the energy conservation is only partial since only gravitational forces are accounted for (excess water at the end of day-simulation will be lost through deep percolation), capillary rise is not accounted for (but can be easily added in a context of shallow aquifer).



### 4.2.2.1. Water balance

The incoming water flows are irrigation  $I(t)$ , precipitation  $P(t)$  and capillary rise  $CR(t)$ , and the outgoing flows are runoff  $RO(t)$ , deep percolation  $DP(t)$ , evaporation  $E(t)$  and transpiration  $T(t)$  (Fig. 4.2). All water flows are expressed in depth [ $\text{mm}\cdot\text{day}^{-1}$ ].

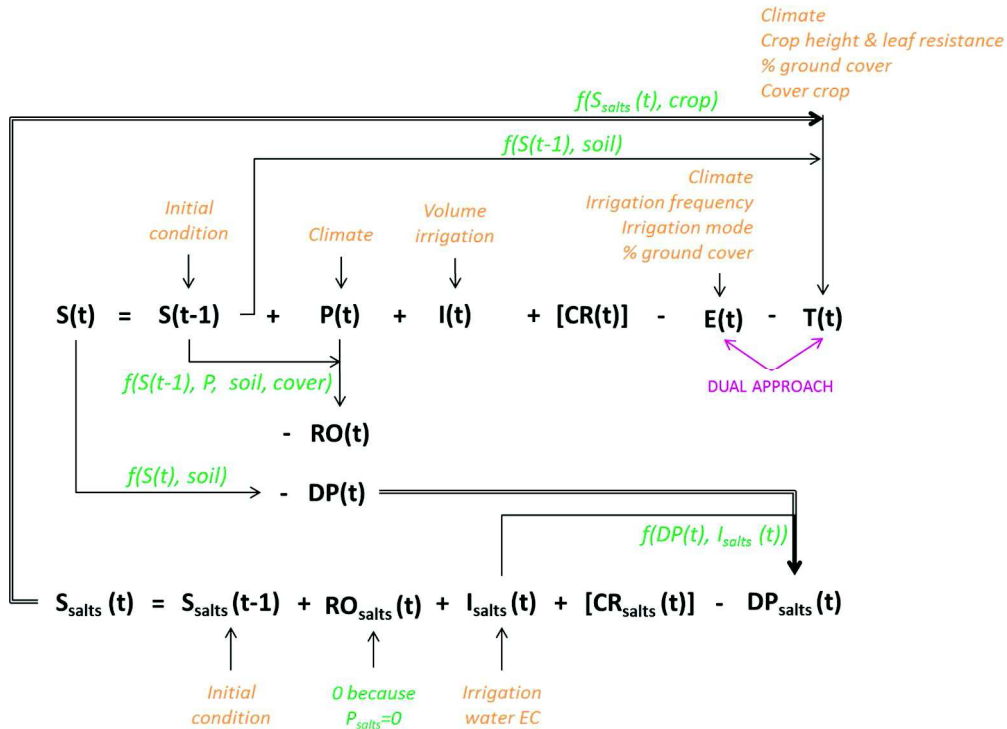


Figure 4.2. Salt and water balances modules in the model, input data (orange), determining factors (green), and interaction between the salt and water balances (double arrows). The evaporation-transpiration module is assessed through a dual approach: soil evaporation and plant transpiration are assessed separately.

**Irrigation  $I(t)$  and precipitation  $P(t)$**  amount are model inputs. Irrigation depth and frequency should be primary data specific of the studied systems. Irrigation is the net irrigation infiltrating in the soil. Precipitation data can be easily obtained from existing databases if more specific data from a nearby weather station are not available.

**Capillary rise  $CR(t)$**  is an upward water flow, function of aquifer depth and soil texture (Raes et al. 2012). The equation is not included in the model yet since this flow can be neglected if the aquifer depth is beyond one meter below the root zone (Allen et al. 1998). Nevertheless, an “empty module” is included in the excel model and can be easily completed following Raes et al. (2012) (Cf. Supplementary Information).

**Surface Runoff  $RO(t)$**  occurs when the precipitation rate is greater than the infiltration rate.  $RO$  is estimated based on the Natural Resources Conservation Services (NRCS) Curve Number (CN) (Mishra and Singh 2003). The NRCS-CN method accounts for most of the runoff producing watershed characteristics, such as soil type, land use, hydrologic conditions, and antecedent moisture condition. This method, first developed in 1954, is considered to have reached maturity (Ponce and Hawkins 1996), and is continuously refined (USDA 2012). This method is used in the Aquacrop model (Raes et al. 2012). Runoff might result from both precipitation and irrigation. However, we considered only rainfall in the runoff

equation in the case of localised irrigation such as drip irrigation. Irrigation is contributing to runoff indirectly because it increases the water stock in the soil, and the value of the Curve Number is adjusted accordingly. RO is thus determined by the antecedent soil water stock, rainfall, cover and soil characteristics (Fig. 4.2).

$$\begin{aligned} \text{If } P(t) > 0.2 S_{cn}: RO(t) &= \frac{(P(t) - 0.2 S_{cn}(t))^2}{P(t) + S_{cn}(t) - 0.2 S_{cn}(t)} \quad \text{in [mm.day}^{-1}\text{]} & (1) \\ \text{If } P(t) > 0.2 S_{cn}: RO(t) &= 0 \end{aligned}$$

With  $0.2 S_{cn}$  the initial abstraction [mm],  $P(t)$  the daily precipitation depth, and  $S_{CN}(t)$  the potential maximum retention after runoff begins [mm], defined by the Curve Number (CN) (Raes et al. 2012):

$$S_{CN}(t) = 254 \times \left( \frac{100}{CN(t)} - 1 \right) \quad \text{in [mm.day}^{-1}\text{]} \quad (2)$$

Water **deep percolation**  $DP(t)$  is water percolating below the root zone (model lower boundary), and is the last term calculated in the water balance. If the soil water stock at the day of calculation  $S(t)$  is exceeding the maximum soil water retention (depending on the soil moisture at field capacity:  $\theta_{FC}$  in  $[m^3.m^{-3}]$ , and the rooting depth:  $z$  in [m]), deep percolation occurs:

$$DP(t) = S(t) - 1000 \times \theta_{FC} \times z \quad \text{in [mm.day}^{-1}\text{]} \quad (3)$$

This equation supposes free-drainage conditions (lower boundary) and quick drainage.

**Evaporation**  $E(t)$  and **transpiration**  $T(t)$  are crucial modules in the model since they depend on the soil water status (so the above mentioned water flows), and are the most difficult terms to estimate. The model calculates evapotranspiration with a dual approach: partitioning evaporation from transpiration, as defined first by the FAO 56 report (Allen et al. 1998). Indeed, in the previous chapter (chapter 3) we showed the limitation of the single approach where the potential evapotranspiration ( $ET_c = K_c \times ET_o$ ) rely on a single crop coefficient  $K_c$  representing averaged evaporation and transpiration from a typical cropped surface for typical irrigation frequencies (Pereira et al. 2015). As a result, in the model the  $K_c$  is partitioned in basal crop coefficient ( $K_{cb}$ , describing plant transpiration) and an evaporation coefficient ( $K_e$ , describing evaporation from the soil surface) (Allen et al. 1998). See figure 4.3 for a representation of chronological developments of single and dual approaches.

$$ET_c = (K_e + K_{cb}) \times ET_o \quad \text{in [mm.day}^{-1}\text{]} \quad (4)$$

With  $ET_o$  the reference evapotranspiration expressing the evaporative demand of the atmosphere (Allen et al. 1998).

This E/T partitioning is particularly relevant for orchards or vineyard where evapotranspiration is more complex than a uniform herbaceous crop because there are different components contributing to water consumption. In addition to tree transpiration, there is transpiration from the cover crop or weeds (if any), and evaporation from the soil. Soil evaporation should be further divided in two components in the

case of micro irrigation: evaporation from the soil areas wetted by the irrigation emitters, and evaporation from the rest of the soil surface which is only wetted by rainfall (Steduto et al. 2012). Thus, evapotranspiration is the sum of four components in the model: transpiration from the tree  $T_{tr}$ , transpiration from the cover crop  $T_{cc}$ , evaporation from the soil wetted emitters  $E_{wz}$ , and evaporation from the rest of the soil  $E_{dz}$ .

$$ET_c = T_{tr} + T_{cc} + E_{wz} + E_{dz} \quad \text{in [mm.day}^{-1}\text{]} \quad (5)$$

We adopted this approach because there are now methods available to measure tree transpiration independently of evapotranspiration. As a result, this approach will certainly become a standard in the future, once the models used for estimation of the transpiration of each crop will be validated, such as for olive tree (Steduto et al. 2012).

**Transpiration** - Tree transpiration is determined by the amount of radiation intercepted by the tree canopy and by the behaviour of stomata. As a result, the calculation of tree transpiration must be specific to the tree species (Steduto et al. 2012):

$$T_{tr} = K_{cb} \times ET_o \quad \text{in [mm.day}^{-1}\text{]} \quad (6)$$

Historically, the FAO first introduces in 1998 the fact that the basal crop coefficient ( $K_{cb}$ ) depends on the fraction of soil covered by vegetation ( $G$ ). Then, in 2009, Allen and Pereira (2009) further develop the estimation of  $K_{cb}$  for row crop, orchard and vine using a density coefficient ( $K_d$ ) accounting for planting densities, canopy sizes, and for when the soil is bare or covered by active vegetation. Eventually, further  $K_{cb}$  sophistications are made for a specific crop. For example, in 2015, Taylor and colleagues (2015) adjusted the equations provided by Allen and Pereira (2009) for citrus, to account for the specific stomata control of this tree (Fig. 4.2). Thus, the basal crop coefficient in the case of no active cover crop is (Allen and Pereira 2009). (In the case of active cover crop, the  $K_{cb}$  equation provided by Allen and Pereira (2009) is similar (Cf. Supplementary information for details):

$$K_{cb} = K_{cmin} + K_d \times (K_{cbfull} - K_{cmin}) \quad (7)$$

With  $K_{cmin}$  the minimum basal  $K_c$  for bare soil ( $K_{cmin}=0.15$  under typical agricultural conditions),  $K_d$  the density coefficient describing the increase in  $K_c$  with increase in amount of vegetation (function of LAI or fraction of ground covered by vegetation, see supplementary information for details), and  $K_{cbfull}$  the basal  $K_c$  during peak plant growth for conditions having nearly full ground cover:

$$K_{cbfull} = Fr (\min(1+0.1h, 1.20) + [0.04(u_2-2)-0.004(RH_{min}-45)] (h/3)^{0.3}) \quad (8)$$

With  $h$  the mean height of the crop,  $u_2$  the mean value for wind speed, and  $RH_{min}$  the mean value for minimum daily relative humidity during the mid-season.  $Fr$  is a downward adjustment ( $Fr \leq 1$ ) if the vegetation exhibits more stomatal control on transpiration than is typical of most annual agricultural

crops. Fr is a function of climate (average monthly wind speed and temperature) and the mean leaf resistance  $r_{leaf}$  representing the stomatal control of a specific crop (Allen and Pereira 2009).

$$Fr \approx [\Delta + \gamma (1+0.34 u_2)] / [\Delta + \gamma (1+0.34 u_2 (r_{leaf}/100))] \quad (9)$$

The standard value for Fr is 1.0 for most annual agricultural crops, because  $r_{leaf}$  is often approximately 100 s.m<sup>-1</sup> (Allen and Pereira 2009), however, this is not valid for citrus as demonstrated by Taylor et al. (2015). Taylor and colleagues (2015) worked on citrus orchards in South Africa and proved that the model developed by Allen and Pereira (2009) was not suitable for citrus due to a greater stomatal control, resulting in an overestimation of citrus transpiration. They obtained the best transpiration estimates with a dynamic leaf resistance ( $r_{leaf}$ ), function of the ET<sub>o</sub>, that we implemented in E.T. model:

$$r_{leaf} = 316 * ET_o^{-61} \quad (10)$$

Thus, the model transpiration module includes a dynamic estimation of the leaf resistance specific to citrus, because we will test the model on a Mandarin crop.

Regarding the transpiration of cover crop or weeds, FAO suggests an approach accounting for the cover crops density:

$$T_{cc} = K_{cc} \times ET_o \times f_{cc} \quad \text{in [mm.day}^{-1}] \quad (11)$$

Where  $f_{cc}$  is the fraction of the orchard ground surface occupied by the cover crop, and  $K_{cc}$  is a cover crop coefficient that varies according to the cover crop density (3 classes and values are proposed) (Steduto et al. 2012).

**Evaporation** - Evaporation from soil was calculated based on an empirical approach. Surface evaporation from the soil wetted by the emitters ( $E_{wz}$ ) was considered equivalent to 60% of the ET<sub>o</sub>. This first approximation is based on measurements and models and is recommended by FAO (Steduto et al. 2012) for soil area in orchard wetted frequently (every 1-2 days):

$$E_{wz} = 0.6 \times ET_o \times wz \quad \text{in [mm.day}^{-1}] \quad (12)$$

With  $wz$  the fraction of the soil surface wetted by the emitters.

Surface evaporation from the rest of the soil surface outside the emitter wetting pattern ( $E_{dz}$ ) was estimated with an empirical equation derived from research on an olive orchard ET (Orgaz et al. 2006; Testi et al. 2006), recommended by FAO as a first approximation for orchard (Steduto et al. 2012):

$$E_{dz} = [0.28 - 0.18 \times G - 0.03 \times ET_o + (3.8 \times F \times (1-F)/ ET_o)] \times (1-wz) \times ET_o \quad \text{in [mm.day}^{-1}] \quad (13)$$

With G the ground cover fraction of the tree canopy, and F the monthly frequency of rainy days.

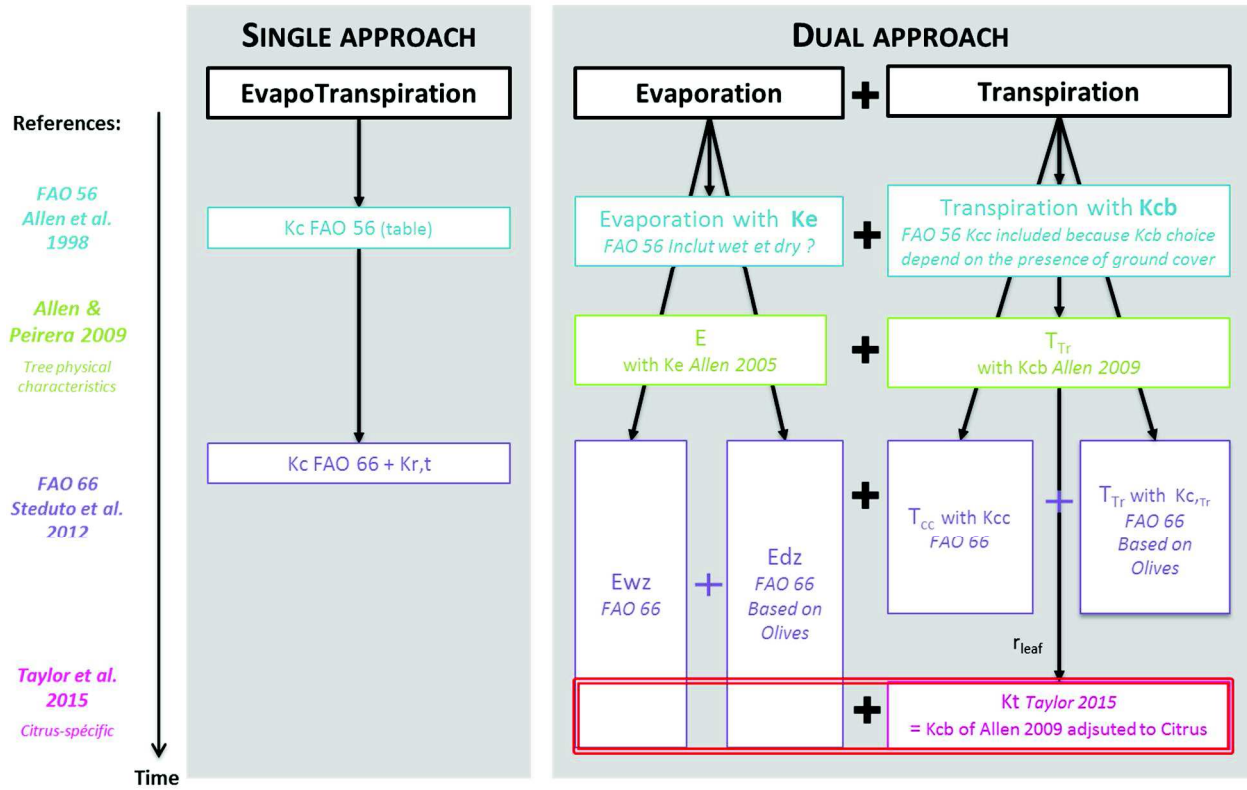


Figure 4.3. Potential evaporation and transpiration estimation methods: single and dual approaches are presented according to their historical developments and corresponding references. The dual approach implemented in the model is framed by double red boxes.

The maximum potential evapotranspiration may be reduced by possible stresses due to a lack of water or nutrients, or an excess of salts (Allen et al. 1998). This aspect of the computation will be detailed in part 4.2.2.3.

#### 4.2.2.2. Salts balance

The salts balance, expressed in grams per surface unit [ $\text{g}\cdot\text{m}^{-2}\cdot\text{day}^{-1}$ ], is computed as (Fig. 4.2):

$$S_{\text{salts}}(t) = S_{\text{salts}}(t-1) + I_{\text{salts}}(t) - DP_{\text{salts}}(t) - RO_{\text{salts}}(t) + CR_{\text{salts}}(t) \quad \text{in } [\text{g}\cdot\text{m}^{-2}\cdot\text{day}^{-1}] \quad (14)$$

The initial condition of salts stored in the soil profile  $S_{\text{salts}}(t=0)$  rely on measurements of soil salinity if available. **Salts brought by irrigation**  $I_{\text{salts}}(t)$  are determined by the irrigation water volume and electrical conductivity  $EC_{\text{irri}}$  in [ $\text{dS}\cdot\text{m}^{-1}$ ]:

$$I_{\text{salts}}(t) = EC_{\text{irri}} \times 0.64 \times I(t) \quad \text{in } [\text{g}\cdot\text{m}^{-2}\cdot\text{day}^{-1}] \quad (15)$$

With  $I(t)$  the irrigation water depth in [ $\text{mm}\cdot\text{day}^{-1}$ ] (equivalent to  $\text{L}\cdot\text{m}^{-2}\cdot\text{day}^{-1}$ ), 0.64 is a conversion factor from  $\text{dS}\cdot\text{m}^{-1}$  to  $\text{g}\cdot\text{L}^{-1}$  (USDA-NRCS). **Salts lixiviated** through drainage  $DP_{\text{salts}}(t)$  are determined by the percolating water and rely on the electrical conductivity of the soil water the day before  $EC_{\text{soil water}}(t-1)$  in [ $\text{dS}\cdot\text{m}^{-1}$ ]:

$$DP_{\text{salts}}(t) = DP(t) \times EC_{\text{soil water}}(t-1) \times 0.64 \quad \text{in [g.m}^{-2}\text{.day}^{-1}] \quad (16)$$

With  $DP(t)$  the deep percolating water in [mm.day<sup>-1</sup>].

Since we are under non steady state conditions, the  $EC_{\text{soil water}}$  has to be adjusted every day:  $EC_{\text{soil water}}$  in [dS.m<sup>-1</sup>], is related to  $S_{\text{salts}}(t)$  in [g.m<sup>-2</sup>.day<sup>-1</sup>], through the maximum water stock  $S_{\text{max}}$ <sup>10</sup>

$$EC_{\text{soil water}}(t) = \frac{S_{\text{salts}}(t)}{S_{\text{max}}} \times \frac{1}{0.64} \quad \text{in [dS.m}^{-1}] \quad (17)$$

With the salts stock in the soil water  $S_{\text{salts}}(t)$  the result of the salts balance:

$$S_{\text{salts}}(t) = S_{\text{salts}}(t-1) + I_{\text{salts}}(t) - DP_{\text{salts}}(t) \quad \text{in [g.m}^{-2}\text{.day}^{-1}] \quad (18)$$

**Salts losses through runoff**  $RO_{\text{salts}}(t)$  are considered nil because runoff is determined only by rainfall, and we assume that rainfall does not contain salts. **Salts brought by capillary raise**  $CR_{\text{salts}}(t)$  are considered nil except in the case of an underlying shallow saline aquifer. **Salts brought by fertilizers** is low in comparison with salts brought by irrigation water, and the influence of fertiliser on soil salinity relies on dissolution and precipitation mechanisms (Jurinak and Wagenet 1981). Due to this low contribution and complexity, salts brought by fertilizers were not considered in the model. This limitation is further discussed in part 4.3.5.2. As illustrated in Fig. 4.1 and 4.2, the water and salt balances are intimately linked because the amount of salts leached below the root zone depends on the deep percolating water, and the saline stress coefficient ( $K_s$  salt) is reducing the potential evapotranspiration  $ET_c$ . Indeed, the amount of salts in the root zone in [g.m<sup>-2</sup>.day<sup>-1</sup>] is converted in terms of electrical conductivity in [dS.m<sup>-1</sup>] to calculate the saline stress coefficient ( $K_{s \text{ salinity}}$ ).

#### 4.2.2.3. Salinity and water stresses

A lack of water or an excess of salts in the soil water can reduce crop water consumption. Thus, to estimate actual evapotranspiration  $ET_a$ , the model computes stress coefficients ( $K_s$ ) that reduce the potential evapotranspiration  $ET_c$  (if  $K_s < 1$ ) according to Allen et al. (1998):

$$ET_a = ET_o (K_e + K_{cb} \times K_s) \quad (19)$$

With  $K_s$  a stress coefficient [0;1] related to water  $K_{s \text{ water}}$ , salinity  $K_{s \text{ salinity}}$  or both water and salinity stresses  $K_{s \text{ water\&salinity}}$ :

$$K_{s \text{ water}} = \frac{TAW - Dr}{TAW - RAW} \quad (20)$$

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<sup>10</sup>  $EC_{\text{soil water}}$  is defined as the electrical conductivity of the soil water solution after the addition of a sufficient quantity of distilled water to bring the soil water content to saturation. That is why this is the maximum soil water stock  $S_{\text{max}}$  which is considered in the equation.

With TAW the Total Available soil Water in the root zone in [mm], RAW the Readily Available soil Water in the root zone ( $RAW = TAW \times p$ , with  $p$  the average fraction of TAW that can be depleted from the root zone before water stress occurs (Cf. Supplementary materials), and  $D_r$  the root zone depletion.  $D_r$  refers to the amount of water required to bring the water amount in the root zone back to the field capacity (Raes et al. 2012):

$$D_r = 1000 \times \theta_{FC} \times z - S(t-1) \quad (21)$$

The salinity stress is expressed as:

$$K_{s \text{ salinity}} = \frac{b}{100} (EC_{\text{soil water}} - EC^*_{\text{soil water}}) \quad (22)$$

With  $EC_{\text{soil water}}$  the mean electrical conductivity of the saturation extract for the root zone in [ $dS \cdot m^{-1}$ ], and  $EC^*_{\text{soil water}}$  the threshold of  $EC_{\text{soil water}}$  when crop yield first reduces below  $Y_{max}$  in [ $dS \cdot m^{-1}$ ], and  $b$  the reduction in yield per increase in  $EC_{\text{soil water}}$ , given in FAO 56 table 23 or computed through the following equation:

$$b = \frac{100}{EC_{\text{soil water at 0\% yield}} - EC_{\text{soil water at 100\% yield}}} \quad (23)$$

From the original  $K_{s \text{ salinity}}$  equation (Allen et al. 1998), the obsolete crop yield response factor term (see chapter 3) was based on expert advice (E. Fereres, personal comm. 2015).

The combined effect of water and salinity stresses was a product of the two stress coefficients:

$$K_{s \text{ water\&salinity}} = K_{s \text{ salinity}} \times K_{s \text{ water}} \quad (24)$$

### 4.2.3. Validity domain of E.T.

The E.T. model is based on old and robust formalisms having a large validity domain: all climatic and soil conditions are covered. In particular, we used the NRCS-CN method to estimate runoff because of its numerous successful applications worldwide since its first publication in 1956, and because curve number values are amended with new measurements (USDA 2012). This proof of robustness and stability is consistent with our objectives of applicability in all pedo-climatic contexts. Nevertheless, soil having a very low saturated conductivity and situation of shallow aquifer are contexts where the model should be modified. In situations of shallow aquifer the capillary rise empty module should be completed (can be easily done with details provided in Supplementary information), and the deep percolation (DP) equation should be modified to account for the absence of free drainage at the lower boundary. Since all water exceeding soil storage capacity is considered lost by DP, the model is not appropriate for soil with a very low saturated hydraulic conductivity.

Regarding the crop, the E.T. model is operational for citrus because transpiration estimation is based on a recent development specific to citrus crop. But the validity domain is potentially on all crops if the leaf resistance is adjusted accordingly (see part 4.3.6. for model usage recommendations).

Regarding the technology (agricultural practices) validity domain, this model is tailored for localised irrigation modes. Three modifications would be required to model other irrigation modes: (i)  $I(t)$  should be considered as potentially generating runoff, then it should be added next to rainfall in the runoff equation (eq. 1), (ii) part of salts brought by irrigation water should be considered lost through runoff (salinity of runoff water could be assumed to be equal to the salinity of irrigation water as a first approximation), and (iii) evaporation estimate should be based on another approach. The extension of the E.T. validity domain will be discussed in part 4.3.5.2.

## 4.2.4. Model testing

### 4.2.4.1. Case study

The model was applied to a perennial crop: a Mandarin grown in Morocco, over seven crop cycles: from planting in October 2007 to April 2015. Table 4.1 describes all input data and parameters needed to run the model, and the corresponding values and sources. Most data are primary data collected in a 225 ha farm of Mandarin Nadorcott located one hour North-West from Marrakech (in the Bahira plain, central Morocco): notably the irrigation volume, rainfall and reference evapotranspiration. However, a few data gaps in climate data were filled following specific rules depending on the climatic parameter (Table 4.1). The evolution of climate data over time is presented in Figure 4.4.

There is no cover crop in the studied orchard, so no transpiration from cover crop was considered.

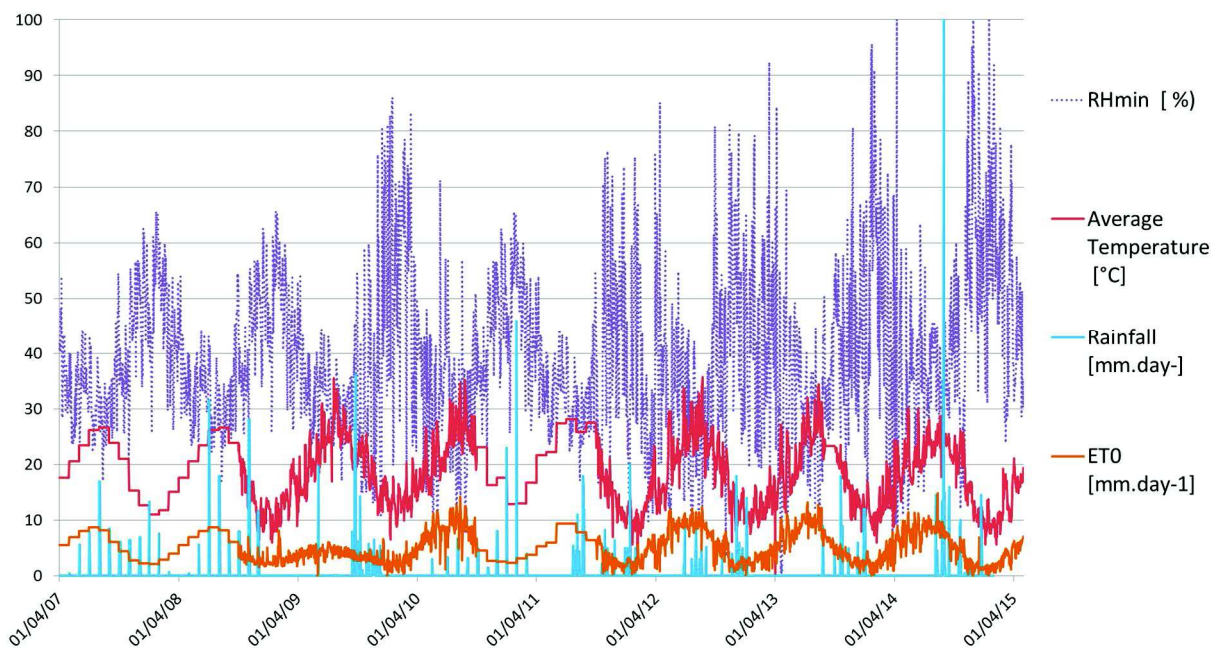


Figure 4.4. Climate data for the case study Mandarin crop grown in Morocco: minimum relative humidity (RHmin), average temperature, rainfall and reference evapotranspiration (ETo). In 2007, 2008, and 2011, assumptions were required to disaggregate monthly values or fill data gaps (details in Table 4.1).



**Table 4.1. Model input variables and parameters description, temporal resolution, determining factors, and average values for the case study (Mandarin in Morocco), and data sources and assumptions**

Model input variable/ parameter	Description	Temporal resolution	Function of	Average values for Mandarin from Morocco			Data source
				Non-productive	Growing yield	Full production	
<b>S</b>	initial soil water stock [mm]	Initial condition	soil texture, antecedent practices	Field capacity			Farm antecedent practices
<b>EC<sub>soil water</sub></b>	initial soil water electrical conductivity [dS.m <sup>-1</sup> ]	Initial condition	climate, soil, practices	1.46			Farm soil analysis
<b>p</b>	fraction of TAW that can be depleted before water stress occurs	Constant value	crop	0.5			Allen et al. 1998
<b>EC*<sub>soil water</sub></b>	lower thresholds at which crop growth starts to be affected	Constant value	crop	1.7			AquaCrop v4.0 Reference manual, 2012, Annex III. Based on FAO report N°29, 48, 56.
<b>EC<sub>soil water X</sub></b>	upper threshold at which crop growth cease	Constant value	crop	8			
<b>ML</b>	multiplier describing the effect of canopy density on shading and on maximum relative ET per fraction of ground shaded	Constant value	crop	1.5			Value recommended for citrus by Allen and Pereira 2009
<b>z</b>	rooting depth [m]	Annual for perennials	crop and water management	0.35	0.4	0.4	Farm for adult tree, adjusted for young tree following Allen et al. 1998
<b>K<sub>c</sub></b>	crop coefficient (single approach ET)	Daily/Monthly	crop, climate, soil and practices	From 0.3 to 0.6	From 0.35 to 0.65	From 0.4 to 0.65	K <sub>c</sub> used for irrigation management by the farmer (results of several years of expertise and orchard monitoring) Cf. part 4.3.4.2
<b>%G</b>	ground cover fraction of the tree canopy (ground shaded by vegetation) [%]	Monthly/Seasonal/Annual	crop and practices	20	50	70	Allen et al. 1998 and Taylor 2015
<b>h</b>	mean height of the vegetation [m]	Seasonal/Annual	crop and practices	2	2.5	3	Observation at farm and Allen et al. 1998
<b>r<sub>leaf</sub></b>	mean leaf resistance [s.m <sup>-1</sup> ]	Daily	crop and climate	316* ET <sub>o</sub> -61			Citrus orchards : Taylor et al. 2015
<b>I(t)</b>	Irrigation water [mm]	Daily/Monthly disaggregated	crop and practices	Total irrigation: 6053			Daily farm records from 2007 to 2015. From Apr 2007 to Apr 2010, and Apr 2014 to Apr 2015: Monthly irrigation disaggregated in daily value assuming equal repartition

<b>wz</b>	fraction of the soil wetted by the irrigation emitters [%]	Annual/crop lifetime	water management	10%			Farm irrigation mode: subsurface drip irrigation. Based on drip irrigation values from Allen et al. 1998 reduced for subsurface
<b>Cover description</b>	description of the land cover : Row/Terraced/crop residue... etc	Annual/crop lifetime	practices and natural environment	Straight row crops, no crop residue			Classification for determination of SCS-Curve Number (Mishra et al. 2003), based on farm characteristics
<b>Hydrologic condition: poor/good</b>	hydrologic condition is based on combination factors affecting infiltration and runoff	Annual/crop lifetime	practices and natural environment	Poor			
<b><math>\gamma</math></b>	Psychrometric constant [kPa/°C]	Crop lifetime	altitude	0.064			Allen et al. 1998 (table 2.2)
<b>EC<sub>iw</sub></b>	Electrical conductivity of irrigation water [dS.m <sup>-1</sup> ]	Daily/Monthly/Annual	water quality	1.258	1.258 - 1.165	1.187	Irrigation water analysis: 2007 -2010: average of 4 analysis made in 2010, 2011-2013: analysis of 2011, 2013-2015: analysis of 2014
<b><math>\theta_{FC}</math></b>	soil water content at field capacity [m <sup>3</sup> . m <sup>-3</sup> ]	Crop lifetime	Soil	0.15			Allen et al. 1998 table 19: average for loamy sand soil texture
<b><math>\theta_{WP}</math></b>	soil water content at wilting point [m <sup>3</sup> . m <sup>-3</sup> ]	Crop lifetime	Soil	0.065			
<b>ET<sub>o</sub></b>	reference evapotranspiration [mm]	Daily/Monthly	Climate	Cf. Fig. 4.3			Weather station at farm. Apr 2007-Oct 2008: Average ET <sub>o</sub> of known years for same month. Oct 2010 -Sept 2011: monthly ET <sub>o</sub> disaggregated in Daily ET <sub>o</sub> assuming equal repartition. Sep 2013: average daily ET <sub>o</sub> of 2009, 2010, 2011, 2012 and 2014
<b>P(t)</b>	rainfall [mm]	Daily/Monthly	Climate	Total rainfall: 1566 (details: fig. 4.3)			Weather station at farm. From Apr 2007 to Mar 2009 and Oct 2010 to Jun 2011: monthly rainfall disaggregated in daily rainfall based on average number of rainy day per month
<b>F</b>	monthly frequency of rainy days	Monthly	Rainfall	Cf. Fig. 4.4			Deduced from rainfall
<b>RH<sub>min</sub></b>	mean value for minimum daily relative humidity during the mid-season [%]	Annual	Climate	33.6%	33.0%	33.9%	Weather station at farm. Mean value for mid-season calculated with June to September daily value. From Oct 2008 to Sept 2009 and Oct 2010 to Sept 2011: daily average based on the known years.
<b>u<sub>2</sub></b>	average monthly wind speed during the mid-season [m.s <sup>-1</sup> ]	Monthly average	Climate	1.99	1.88	1.98	Weather station at farm. From Oct 2007 to Sep 2009 and Oct 2010 to Sep=2011: average of average wind for the corresponding month in 2009, 2011, 2012, 2013, 2013
<b><math>\Delta</math></b>	Slope of the saturation vapour pressure versus air temperature curve [kPa/°C]	Daily or monthly	Average temperature	0.14	0.15	0.14	Allen et al.1998: $\Delta$ function of temperature. Temperature from weather station at farm: Oct 2007 - Oct 2008: daily temperature based on average monthly for known years. Oct 2010 - Sept 2011: daily temperature based on average monthly. Sept 2013: monthly average of Sept 2009, 2012, 2014
<b>Total Yield</b>	Total crop yield [ton.ha <sup>-1</sup> ]	Annual	crop, climate, soil, practices	249.5			Farm records : yield from 2008 to 2015

#### 4.2.4.2. E.T. model output variables

The E.T. output variables are water flow volumes and quality (salinity). Additional calculations are required to convert the hydrological water flows in terms of water inventory flows usable in LCA. The total actual evapotranspiration ( $ET_a$ ) is further divided into its green (rain water) and blue (surface and ground water) components<sup>11</sup> for two reasons: (i) to compare the model outputs with water inventory databases, and (ii) because in LCA, blue water consumption and green water consumption do not have the same impacts.

- The green water component of actual evapotranspiration ( $ET_{a\ green}$ ) corresponds to the effective rainfall ( $P_{eff}$ ) as defined by Pfister et al. (2011) and CropWat for the computing of crop water requirement.  $P_{eff}$  is the part of the rainfall which is actually available to the crop for evapotranspiration: rainfall minus losses through runoff and deep percolation:

$$ET_{a\ green}(t) = P_{eff}(t) = P(t) - RO(t) - DP_{green}(t) \quad \text{in [mm.day}^{-1}] \quad (25)$$

The difficulty here is to estimate  $DP_{green}$ : the share of deep percolating water originating from rainfall. We estimated it through the ratio of rainfall infiltrated to irrigation and rainfall infiltrated:

$$DP_{green}(t) = DP(t) \times [(P(t) - RO(t)) / (P(t) - RO(t) + I(t))] \quad \text{in [mm.day}^{-1}] \quad (26)$$

Nota :  $DP_{green}$  is not green water but originate from rainfall water

- Finally, the blue component of actual evapotranspiration ( $ET_{a\ blue}$ ) is the difference between total  $ET_a$  and the effective rainfall:

$$ET_{a\ blue}(t) = ET_a(t) - P_{eff}(t) \quad \text{in [mm.day}^{-1}] \quad (27)$$

Actual evaporation and transpiration partitioning in green and blue waters was computed assuming the same percentage of green and blue waters in T and E than in ET (e.g.:  $T_{blue} = T_a \times (ET_{a\ blue} / ET_a)$ ).

- Similarly, the water flows released in the environment are divided according to their blue and green components, because only the blue component is characterized in LCA. Thus, deep percolating blue water toward ground waters ( $DP_{blue}$ ) is estimated similarly to  $DP_{green}$ . In our case study, the water flow released toward surface waters (RO) is only green water since we considered that drip irrigation was not directly generating runoff.
- Regarding water flows quality, the variable of interest are the deep percolating water salinity ( $DP_{salts}$ ) and the soil water salinity ( $S_{salts}$ ).

The output variables of the model were confronted to field measurements (from bibliography and case study) and water inventory databases ((Water Footprint Network; Pfister et al. 2011; Pfister and Bayer 2014).

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<sup>11</sup> Refer to the glossary for a complete definition

#### 4.2.4.3. Sensitivity of model outputs to model formalisms

We compared the outputs of the E.T. model with three other model formalisms over the 7 years of perennial cycle of our mandarin case study.

In addition to the E.T. model including the dual approach and possible water and salinity stresses (*E/T partitioning, water and salinity stress*), the formalisms tested were: two single approach versions accounting for possible water stress (*ET, water stress*) and salinity stress (*ET, water and salinity stress*), respectively and a dual approach version accounting for possible water stress only (*E/T partitioning, water stress*) (Table 4.2). For the single approach version, the crop coefficient  $K_c$  is a model input and is based on the  $K_c$  monthly values used by the farmer to manage irrigation. This  $K_c$  is the result of the farmer expertise and reflects the farm specificities in terms of climate, crop, and management strategy. In the case of E/T partitioning (dual approach), the crop coefficient is the basal crop coefficient (or transpiration coefficient) and is calculated as a function of crop characteristics, climate and management following Allen et al. (2009) and Taylor et al. (2015).

**Table 4.2. E.T. versus other model formalisms**

Model version	Crop Coefficient	E/T Partitioning	Water Stress	Salinity Stress	Water Quality (salts)
<b>E.T. model: E/T partitioning, water and salinity stresses</b>	<b><math>K_{cb}</math> calculated</b>	<b>yes</b>	<b>yes</b>	<b>yes</b>	<b>yes</b>
E/T partitioning, water stress	$K_{cb}$ calculated	yes	yes	no	no
ET, water stress	$K_c$ farm	no	yes	no	no
ET, water and salinity stresses	$K_c$ farm	no	yes	yes	yes

#### 4.2.4.4. Sensitivity of E.T. outputs to parameters' variations

The sensitivity of E.T. model outputs was assessed against model parameters range testing, based on realistic values rather than arbitrary values for: initial conditions of soil water stock ( $S(i)$ ) and soil water salinity ( $EC_{soil\ water}(i)$ ), and the average fraction of TAW that can be depleted before water stress occurs ( $p$ ). We tested the minimum and maximum values for each parameter based on the case study soil texture and corresponding hydrologic characteristics (permanent wilting point  $< S(i) <$  field capacity), on extreme soil water salinity measured ( $0.07 < EC_{soil\ water}(i) < 2.49$ ), and on  $p$  value range provided by Allen et al. (1998) ( $0.1 < p < 0.8$ ).

Moreover, we also tested the sensitivity of the model to the use of degraded input data for reference evapotranspiration ( $ET_o$ ) from Climwat (FAO 2010), and citrus basal crop coefficient reported in the literature ( $K_{cb}$ ). The basal crop coefficient is an intermediate variable calculated in the model (eq. 7) based on the model from Allen et al. (1998) and Taylor et al. (2015). Nevertheless, we tested other  $K_{cb}$  values for citrus reported in Allen and Pereira (2009) and Villalobos et al. (2013) (Table 4.3).

**Table 4.3. Basal crop coefficient values for citrus tested in the E.T. model**

Reference	Crop	Winter	Spring	Summer	Autumn
<b>Allen &amp; Pereira (2009)</b>	Theoretical - Citrus, no ground cover, high density (G=70%)	0.85	0.85	0.85	0.85
<b>Villalobos et al. (2013)</b>	Measurements - Oranges Lane Late, Spain	0.4	0.3	0.32	0.35

#### 4.2.4.5. Testing of E.T. model for different scenarios of practice

The E.T. model was used to simulate different scenarios of practice through the use of different input data (Table 4.4). We simulated:

- a larger wetted zone by irrigation, through the parameter  $wz$ , based on the maximum value of  $wz$  for drip irrigation provided by Allen et al. (1998),
- deeper rooting depth for adult trees through the parameter  $z$ , based on the maximum value provided by Allen et al. (1998) for citrus,
- bigger (+20%) and smaller (-50%) tree canopy size and plantation density through the parameter %G (percentage of ground covered by vegetation),
- a land use type less favourable to runoff through the Curve Number value : a contoured row crops with crop residue and good hydrologic condition, instead of straight row crops and poor hydrologic condition.

**Table 4.4. Scenarios of practices tested in the E.T. model and corresponding values**

	Reference	Scenario 1	Scenario 2
wz - wetted zone by irrigation	subsurface drip irrigation - 10%	drip irrigation maximum wetted zone area: 40%	-
z - rooting depth (adult tree)	Low rooting depth (due to farm irrigation practices) 0.4m	maximum rotting depth for citrus: 1.5m	-
%G - percentage of ground covered	Non-productive: 20% - Growing yield: 50% - Full production: 70%	+20%	-50%
CN - Curve Number	straight row crops, poor hydrologic condition: CN=91	contoured row crops, crop residue, good hydrologic condition: CN=85	-

### 4.3. Results and discussion: model testing

#### 4.3.1. Comparison of E.T. outputs to other model formalisms

The water flows estimated by E.T. and the three other model formalisms over a perennial cycle of 7 years are presented in Figure 4.5. The evolution on a daily basis over the various seasons of the actual evapotranspiration flow was analysed in more details and is presented in Figure 4.6. Figure 4.6 also presents the effects of accounting for salinity stress in addition to water stress, and the effect of evapotranspiration partitioning.

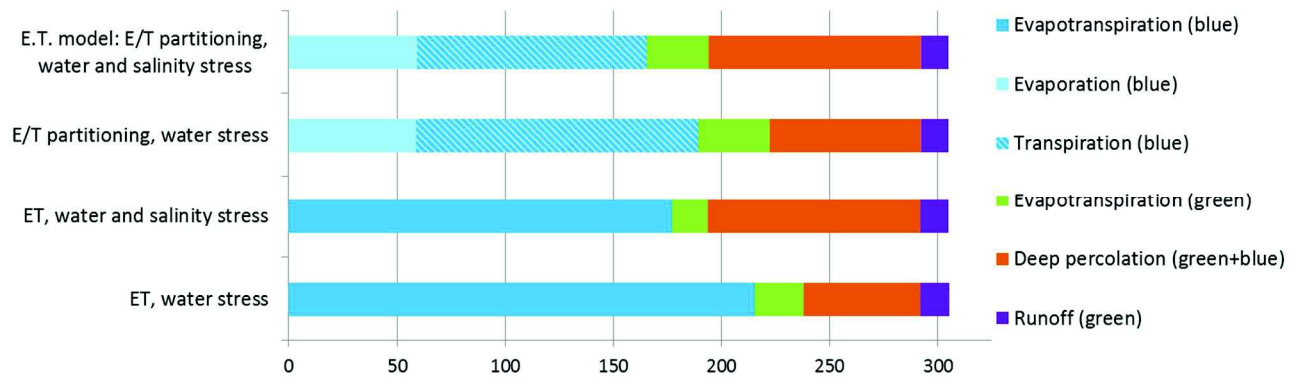


Figure 4.5. Water flows estimated by the E.T model (E/T partitioning, water and salinity stresses) and other model formalisms, expressed in m3 per ton of Mandarin over seven years (from planting in 2007 to 2015)

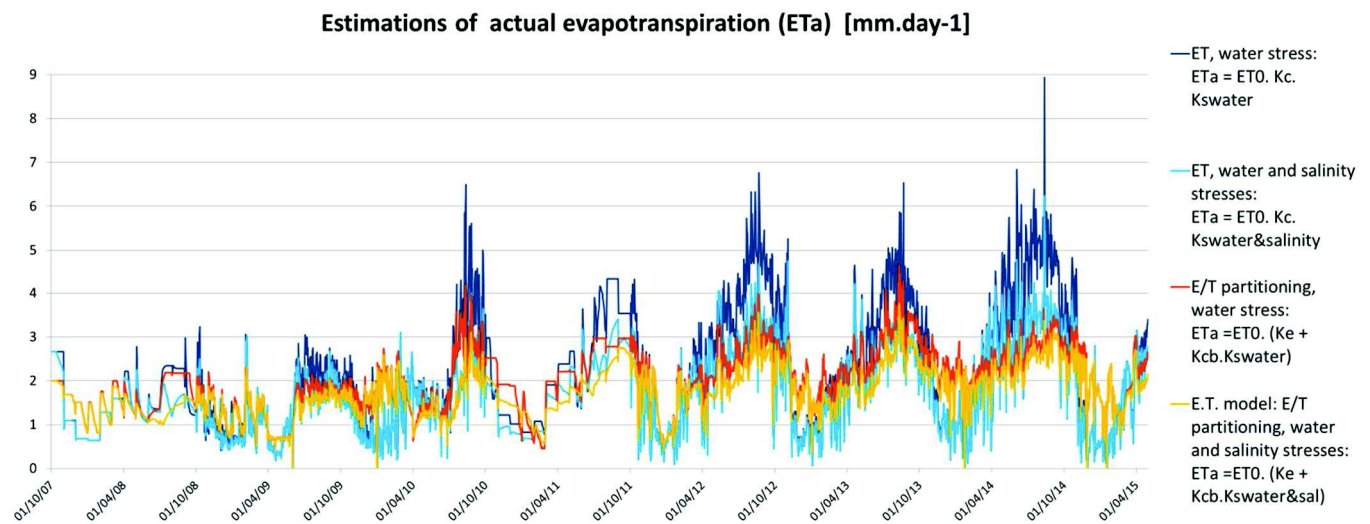


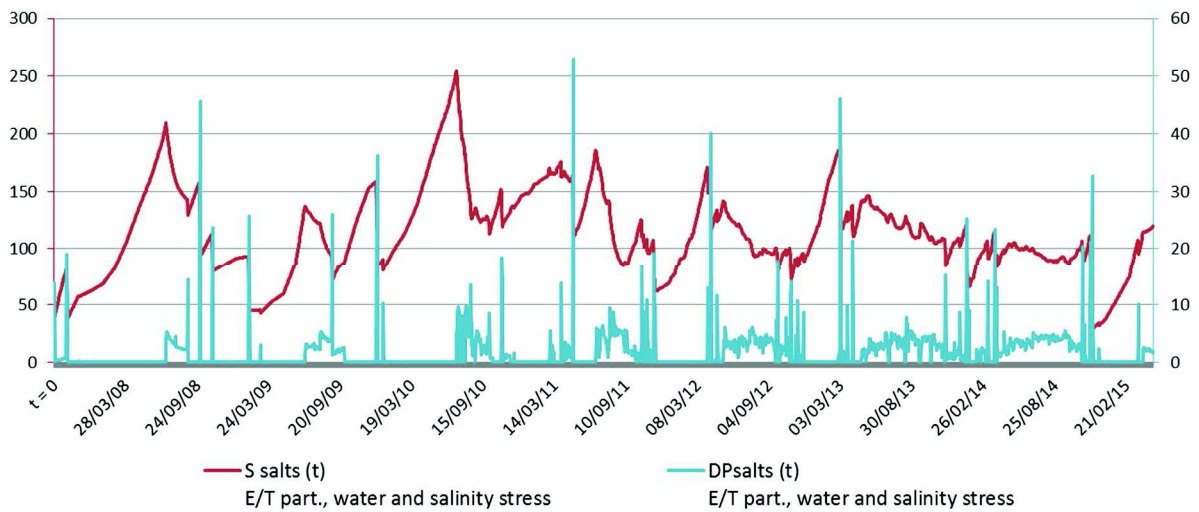
Figure 4.6. Actual evapotranspiration (ETa) estimated by the E.T model (E/T partitioning, water and salinity stresses) and other model formalisms, expressed in mm.day-1, over seven years (from planting in 2007 to 2015)

##### 4.3.1.1. Effect of saline stress

When salinity stress was not accounted for, the evapotranspiration water flow was greater:  $ET_{a\text{ blue}}$  was increased by 14.2% when comparing the E.T. model with the formalisms accounting only for water stress

(E/T partitioning, water stress). More precisely, in the absence of saline stress this is the transpiration which is increased by 22.3%, evaporation remaining unchanged. This  $ET_{a\text{ blue}}$  increase was greater (21.6%) when comparing with the single approach model formalisms (ET, water and salinity stresses vs. ET, water stress) (Fig. 4.5). As a result of the increased transpiration, less water is percolating below the root zone (-29.1% for E/T partitioning formalisms). RO is not affected because it depends mainly on the rainfall and secondarily on the soil water stock.  $ET_{a\text{ green}}$  (representing the effective rainfall:  $P_{\text{eff}} = P - RO - DP_{\text{green}}$ ) is thus increased because the green percolating water is smaller.

As demonstrated, salinity stress (when  $K_{s\text{ salinity}} < 1$ ) is limiting evapotranspiration and is greater during the summer season (Cf. Supplementary Information for  $K_{s\text{ salinity}}$  evolution). This is due to salts accumulation in the soil profile, when there is not sufficient deep percolating water to lixiviate them (Fig. 4.7).



**Figure 4.7. Evolution over time of salts in soil water (red line) and salts in percolating water (blue line) (in g.m-2) for the E.T. model (E/T partitioning, water and salinity stresses).**

The model provides estimate of  $EC_{\text{soil water}}$  ranging from 0.77 to 6.63  $dS.m^{-1}$ . This is above the electrical conductivity measurements at farm, ranging from 0.07 to 2.49  $dS.m^{-1}$ , but the average value 3.0  $dS.m^{-1}$  provided by the model compares well with measurements made close to soil surface (from 0 to 5 cm soil depth) for several plots. However, the comparison and use of model estimates should not go further (e.g. daily time step) since salinity is highly variable in time and space. Indeed, the model considers soil as a single reservoir but salinity varies a lot with depth. Soil analysis revealed that from 0 to 30 cm depth, salinity can vary up to 3000%. Thus, we use the model salinity outputs for two purposes: (i) estimating the reduction of evapotranspiration due to salinity stress and (ii) estimating the average amount of salts percolating toward the aquifer through deep percolating water.

Regarding the quality of the water flows, the E.T. model estimated an average salt concentration of deep percolating water of 1.88  $g.L^{-1}$ . This concentration is 7% smaller when estimated with a single approach formalism.

### 4.3.1.2. Effect of evapotranspiration partitioning

When ET was not partitioned, the total  $ET_a$  estimated was not significantly different (less than 7%) (Fig. 4.5). However,  $ET_{a\text{ blue}}$  and  $ET_{a\text{ green}}$  were greatly modified. The  $ET_{a\text{ green}}$  decreased by 41.7% when considering salinity stress, and 32.0% when not considering salinity stress (Fig. 4.5). The  $ET_{a\text{ blue}}$  increased by 7% when considering salinity stress, and by 13.9% when not considering salinity stress. By definition, we cannot analyse separately evaporation from transpiration in the single approach. Thus, we assume that the reduction in  $ET_a$  is mainly due to the reduction in transpiration because this is a feature of the model: accounting for the greater stomatal control of citrus thanks to the equations provided by Allen and Pereira (2009) and Taylor et al. (2015). Tree transpiration is decreased during the hottest time of the year as a result of the dynamic leaf resistance proportional with the climatic demand ( $ET_o$ ) (Fig. 4.6). This observation is supported by literature on citrus stating that stomata are closing under dry, hot, windy conditions and opening under the opposite conditions (Steduto et al. 2012). Nevertheless, it is important to notice that these estimates of water flows should not be used on a daily basis for irrigation scheduling because Taylor and colleagues (2015) specified that their approach provided good seasonal estimates. This is why we recommend using the water flows estimates from our model over one season, one year or several years.

Evaporation is the sum of two components ( $E_{wz}$  and  $E_{dz}$ ), which have an opposite behaviour: during summer evaporation from the wetted zones is higher, whereas evaporation from the rest of the soil is at its lowest value. But the resulting total evaporation is not subject to seasonal variation contrary to transpiration (Fig. 4.8) and represents on average 36% of total evapotranspiration.

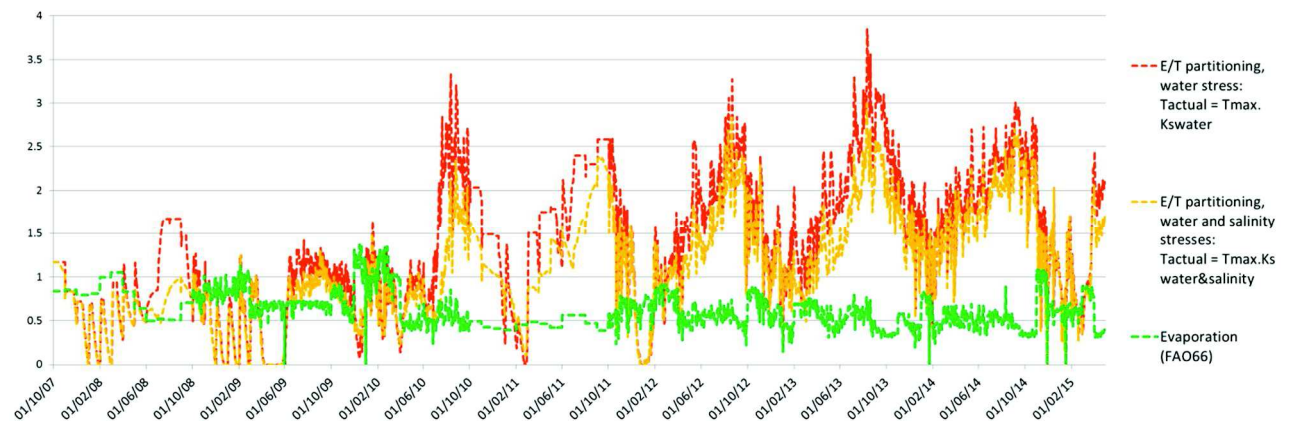


Figure 4.8. Actual transpiration ( $T_a$ ) estimated with water stress or with water and salinity stresses, expressed in  $\text{mm}\cdot\text{day}^{-1}$ , over seven years (from planting in 2007 to 2015)

The estimations of evaporation and transpiration from the E.T. model were in agreement with literature references on citrus. Villalobos et al. (2009) measured ET (using Eddy covariance) and soil evaporation (using micro-lysimeters) for a Mandarin orchard cultivated in the south of Spain (drip-irrigation, no cover crop, sandy soil, 17 years old trees). Average ET was  $2.6\text{ mm}\cdot\text{day}^{-1}$  in August ( $2.8\text{ mm}\cdot\text{day}^{-1}$  over the last 3 years for the E.T. model), and  $2.1\text{ mm}\cdot\text{day}^{-1}$  in May ( $2.0$  for the E.T. model over the last 3 years). Regarding the contribution of evaporation to total ET, it may seem surprising that even when the soil is dry and the irrigation system wetted a very small fraction of soil, the evaporation contributes 36% of



total orchard evapotranspiration. This model output is supported by Villalobos and colleagues (2009) whose measurements also showed a 36% contribution of evaporation in August.

It is important to notice that beyond the improved assessment of evapotranspiration thanks to its partitioning (Pereira et al. 2015), this approach also allows distinguishing productive (T) from non-productive water (E).

### 4.3.2. Sensitivity of E.T. model outputs to parameters' variations

The sensitivity of the model outputs to initial conditions and parameter  $p$  was low which demonstrated the robustness of the E.T. model (Table 4.5). When assuming initial soil water stock ranging from permanent wilting point to field capacity, the model outputs were not affected (the maximum variation observed was +1.8% for  $ET_{a\ green}$ ). When assuming initial soil water electrical conductivity ranging from 0.07 to 2.49  $dS.m^{-1}$ , the model outputs were not affected either (the maximum variation observed was 0.2% for deep percolating water). For  $p$  values ranging from 0.1 to 0.8, the model outputs were slightly affected: the maximum variation observed was  $ET_{a\ green}$  varying from -5.4% to +2.5%.

**Table 4.5. Percentage variations of water flows simulated with E.T. for the range values of initial conditions (S(i) and  $EC_{soil\ water}(i)$ ) and for the average fraction of Total Available Water (TAW) that can be depleted from the root zone before water stress occurs (p).**

Parameter	Value	Source	$ET_{a\ blue}$	$ET_{a\ green}$	Percolating water	Runoff water
S(i)	Permanent wilting point	Function of soil texture	-0.3%	1.8%	-1.1%	-0.5%
$EC_{soil\ water}(i)$	2.49 $dS.m^{-1}$	Farm soil analysis from 2007 to 2014	-0.1%	-0.1%	0.2%	0.0%
$EC_{soil\ water}(i)$	0.07 $dS.m^{-1}$		0.1%	0.0%	-0.2%	0.0%
$p$	0.8	range values for $p$ (Allen et al. 1998)	0.1%	2.5%	-0.9%	0.0%
$p$	0.1		-0.3%	-5.4%	2.1%	0.0%
<b>Reference:</b> S(i)=Field capacity, $EC_{soil\ water}(i)$ =1.46 $dS.m^{-1}$ , $p$ = 0.5						

### 4.3.3. Testing of E.T. model for different scenarios of practice

After showing the relevance of E.T. model formalisms and its low sensitivity to parameters, we analysed its discriminating power for different agricultural practice scenarios (Fig. 4.9).

**Irrigation mode (wz)** - When considering the upper value for the wetted zone by the drip irrigation ( $wz=40\%$ )(Allen et al. 1998), evaporation increased by 129.2% whereas transpiration decreased by 41.1% (Fig. 4.9). Deep percolating water decreased by 35.6%. As a result, soil water salinity was much higher (4.64  $dS.m^{-1}$  on average).

**Rooting depth (z)** - When considering deeper roots depth (up to 1.5 m instead of 40 cm for an adult tree), following Allen et al. (1998), transpiration increased by 3.4%, decreasing deep percolation water by 9.7%.  $ET_{green}$  increased by 10.7% because more soil water is potentially available to the plant. This result corresponded to more water being available for the plant thanks to its roots. Rooting depth depends mainly on the irrigation management, but citrus active roots are generally located at a very low

depth. Only one pivot root is going deeper for settling the tree. Therefore, the use of Allen et al. (1998) values for rooting depth should be considered as an extreme maximum for citrus not reflecting the depth of available soil water.

**Percentage of ground covered by vegetation (%G)** - %G represents the amount of solar radiation intercepted by plants for potential conversion into evapotranspiration. When dividing by two the percentage of ground covered by vegetation (from 70% to 35% for an adult tree) transpiration decreased by 18.9% whereas evaporation and deep percolation increased by 16.2% and 12.7%, respectively. The evolution was less and opposite when %G was increased (Fig. 4.9).

**Curve number (CN) (for the runoff calculation)** - When considering a contoured row crops with crop residue and good hydrologic conditions instead of straight row crops and poor hydrologic conditions, the runoff was reduced by 47.9%, and transpiration increased by 1.5%. The small influence of the infiltrating rainfall on transpiration was due to the small share of rainfall in the total water input (only 25%).

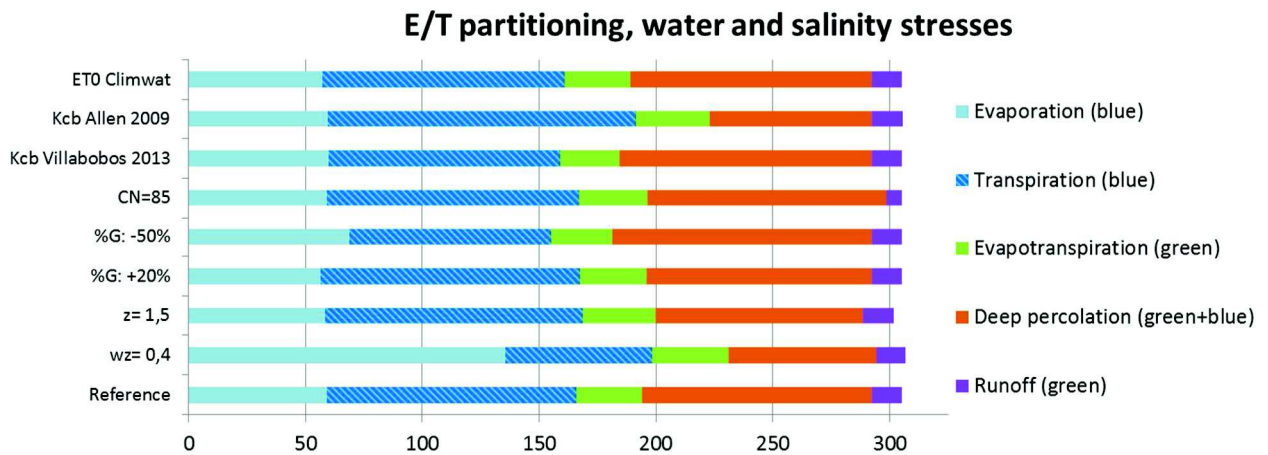


Figure 4.9. Scenario analysis of the E.T. model. The input variables tested are: wz= wetted zone by the irrigation, z= rooting depth, %G= percentage of ground covered by vegetation, CN= curve number (for the runoff calculation), Kcb= basal crop coefficient, ETo = reference evapotranspiration.

#### 4.3.4. Model with degraded data

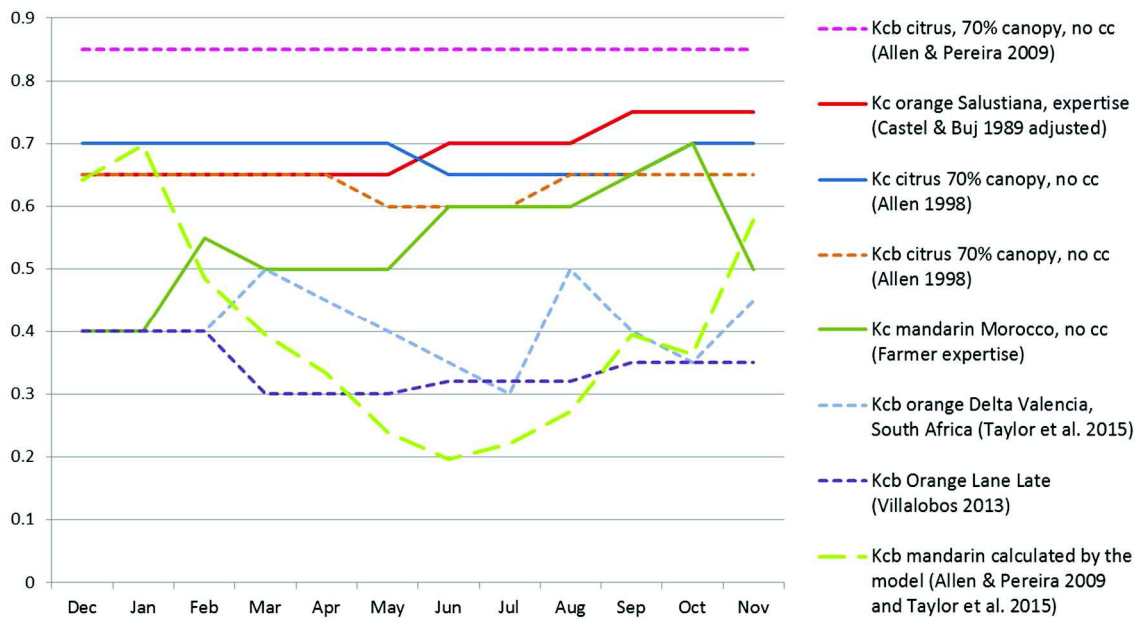
The model was further assessed with the use of degraded input data for crop coefficient and reference evapotranspiration.

##### 4.3.4.1. Reference evapotranspiration

ET<sub>o</sub> values from Climwat (long-term monthly mean values for at least 15 years of data, Marrakech station) were below the ET<sub>o</sub> values from the farm weather station (calculated according to the Penman-Montheit equation), except in summer 2009. Overall, ET<sub>o</sub> from Climwat was 19.8% lower than the ET<sub>o</sub> measured on farm. However, the effect of using default ET<sub>o</sub> from Climwat on model outputs was low: transpiration increased by 2.6%. This transpiration increase was due to the lower leaf resistance of citrus at low evaporative demand (eq. 10), and the lower water stress of the crop. The use of default ET<sub>o</sub> data from Climwat is therefore acceptable if specific crop values are not available.

#### 4.3.4.2. Basal crop coefficient

We analysed the effect of different basal crop coefficient ( $K_{cb}$ ) values on the model outputs by replacing the calculated  $K_{cb}$  with values found in the literature. When using the theoretical  $K_{cb}$  from Allen et al. (2009), the tree transpiration was greater (+23.0%). On the contrary, when using the  $K_{cb}$  from Villalobos et al. (2013), based on measurements, the tree transpiration was reduced by 6.9%. The results confirmed the observation from Villalobos et al. (2013) and Taylor et al. (2015) stating that the method of Allen for perennials is overestimating transpiration for citrus tree. The  $K_{cb}$  calculated by the model following Taylor et al. (2015) was closer to  $K_{cb}$  based on measurements in citrus orchards (Taylor et al. 2015, Villalobos et al. 2013) (Fig. 4.10).



**Figure 4.10.** Values of crop coefficient  $K_c$  (continuous lines) and basal crop coefficient  $K_{cb}$  (dotted lines) for citrus from the literature and simulated by the E.T. model (dashed line). Year 2012/2013, 6 years-old orchard. no cc: no cover crop.

Citrus crop coefficients found in the literature showed huge variations (Fig. 4.10) either due to their definition or their way of estimation. First, there is the  $K_c$ , including both the transpiration and evaporation components, and the  $K_{cb}$ , focusing on crop transpiration. Generally,  $K_c$  values are higher than  $K_{cb}$  values, except for Allen and Pereira (2009). Second, there are measured ( $T$  to  $ET_o$  ratio), and theoretical crop coefficients. In particular, the  $K_c$  used by the farmer is not a measure but an estimation and is used as an irrigation management tool. This  $K_c$  is the result of farmer's expertise and management because the farmer adjusts the  $K_c$  (originally from FAO table) years after years based on observations of the tree (e.g. fruits size) and soil water status (with tensiometers and soil profiles). Thus, this crop coefficient integrates both plant physiology and irrigation management strategy. This  $K_c$  is used for irrigation planning: monthly  $K_c$  is multiplied with daily  $ET_o$  (from the weather station) to calculate the crop water requirement.

### 4.3.5. Model limitations and improvement perspectives

#### 4.3.5.1. A limited model sensitivity assessment

Comparing the variations of model outputs with the variations of model inputs showed that:

- all water flows were very sensitive to basal crop coefficient value: this highlighted the importance of an accurate calculation of  $K_{cb}$  specific to crop and practices,
- $ET_{a\ green}$ , RO and DP were very sensitive to the runoff estimation (through the curve number),
- $ET_{a\ blue}$  (in particular transpiration) was very sensitive to the canopy size and planting density (through %G),
- $E_{blue}$  was very sensitive to wetted zone by the drip irrigation (wz) (Cf. Supplementary information for details).

This first testing of the model demonstrated the discriminating power of the model, its low sensitivity to key parameters, and the importance of the crop coefficient value. Nevertheless, beyond this first testing of the model and its robustness, a proper sensitivity analysis and a Monte Carlo analysis would be warranted in future work, notably to assess the effects of uncertainties interaction and cumulative effects. In particular, we should investigate the model discriminating power regarding the environmental context (in particular the soil texture which is influencing several parameters:  $\theta_{FC}$ ,  $\theta_{PWP}$ , CN, S(i)), but also we should test the use of default rainfall data from the Climwat database.

#### 4.3.5.2. Formalisms and mechanisms

It is important to notice that the salts balance is a rather simplified approach since several mechanisms are neglected: salts precipitation and dissolution (one possible solution would be to use a steady state model such as WATSUIT), and salts uptake by plants. In addition, salt conversion factors from electrical conductivity to concentration are only an approximate equivalence factor (USDA-NRCS).

Regarding the stresses of the crop, we only considered water and salinity stresses, this is valid in conditions where nutrients are not a limiting factors. Stress coefficients are approximate estimates of salinity and water impacts on ET (Allen et al. 1998). Salinity effects on crop do not include specific ion toxicity effects. Nutrient stress and its effect on plant evapotranspiration (and thus water flows) were not considered. As a result, the effect of fertilisers on soil salinity and crop growth was not considered. Indeed, the interactions between soil salinity, fertilisers and crop growth are complex and addressing the crop response to the interaction of fertilisation and salinity is would require sophisticated modelling. In fact, fertilizers are modifying the nature and quantity of soil salts presents in the soil, and will not only influence the osmotic potential of soil solution (affecting plant water consumption), but also affect the uptake of nutrients by the crop (affecting plant nutrition). Depending on the fertiliser form, the potential increase in soil salinity varies. If the addition of nitrogen fertilisers can increase soil salinity because nitrogen salts are soluble, on the contrary, the addition of phosphorus fertilizers can slightly reduce soil salinity because of the formation of insoluble precipitate (Jurinak and Wagenet 1981). However, it is impossible to generalise the interactive effects of soil fertility and salinity on crop because it depends on the crop type, fertilizers forms and quantity, soil type, irrigation water and soil salinity (Jurinak and Wagenet 1981). As a result, accounting for the effect of fertilisers on crop growth and thus on water fluxes would require a more complex model and is relevant only for critical crop growth development

stages. As a result, the E.T. model focuses on a salt and water balance for now, but the nutrient part could be added in the future.

The model could be improved regarding the inclusion of salinity stress and evaporation mechanisms.

**Salinity stress** -  $K_s$  salinity estimation (eq. 21) may not be valid at high salinity level when  $EC_{\text{soil water}} > EC^*_{\text{soil water}} + 50/b$  according to Allen et al. (1998). This limited validity domain was confirmed by Katerji and colleagues (2011): they assessed the validity of  $K_s$  salinity equation against measured values of ET on potato and broad bean in Italy. They showed that  $K_s$  salinity was overestimated at high salinity level, thus leading to an underestimation of actual ET ranging from 4 to 20% as a function of soil salinity. This is due to the combined effect of soil water shortage and water quality. In our simulation with the E.T. model, high salinity levels were reached during 136 days of the simulation (during summer 2008 and 2010). Thus, we were outside the validity domain of the equation during those days and can reasonably assume an underestimation of  $ET_a$ . As a result, the  $K_s$  calculation should be revised to account for:

- the effect of climate and growth development stage, as in the case of crop coefficient  $K_c$  (Katerji et al. 2011). To do so,  $K_s$  should be defined for each crop growth development stage, accounting for its sensitivity level. This is what is implemented in the Aquacrop model (Raes et al. 2012).
- the effect of salinity on the water stress threshold: salinity is lowering the minimum soil water content at which crop will start to be negatively affected. Indeed, the approach assumes that RAW ( $RAW = p \times TAW$ ) does not change with increasing salinity (Allen et al. 1998). Yet, salinity has an osmotic effect on the soil water. To account for it, Raes et al. (2012) proposed to reduce the parameter  $p$  by multiplying it with the  $K_s$  salinity of the corresponding development stage.
- the possible non-linear curve response factor to stresses. Indeed, the effect of a stress is not necessarily directly proportional to the relative stress, but can be convex or logistic depending on the crop (Raes et al. 2012).

**Evaporation estimation** - The major complication in computing evaporation from an orchard is that the soil is partially and dynamically shaded by the crop canopy. Evaporation estimate is based on empirical equations validated on Olive tree in the Mediterranean context. This method was considered appropriate to apply on another evergreen Mediterranean tree (citrus) grown under semi-arid climate. However, other ways to estimate evaporation should be tested in particular when the irrigation mode is not drip irrigation: Allen et al. (1998; 2005a; 2005b) described estimation of the evaporation component, notably accounting for effects of surface mulching, and Raes et al. (2012) and Steduto et al. (2012) described a two-step approach for the estimation of evaporation accounting for the 2 successive evaporation mechanisms.

#### 4.3.5.3. Inputs or parameters sources

Although the E.T. model was designed to cope with data scarcity (hampering the use of more complex models, see chapter 3), one model limitation might still be the input data requirement under constrained circumstances. The assets of this model lie in the fact that irrigation water volumes are specific of the actual water management practices. Thus, this should be primary data. Nevertheless, other input data can be retrieved from default data sources (See Table 4.1 and Table 3.2). Irrigation water quality information is probably the most difficult data to obtain if water analyses are not available. Regarding precipitation data, if daily volumes are not available, a desagregation of monthly precipitation is required. This was done in the model testing for 32 months (over 90 in total) where daily data were

missing: monthly rainfall were disaggregated in daily rainfall based on average number of rainy day per month (Table 4.1). It is paramount to notice that this may be a high source of uncertainty on the water balance.

Future evolution of databases and new data acquisition will support E.T. model use. Notably, thanks to remote sensing, one will have access to crop coefficients (both single and dual) estimated from vegetation indices (Pereira et al. 2015).

#### 4.3.6. Model usage recommendations

We recommend the partitioning of evaporation from transpiration as long as crop transpiration data are available for the studied crop. According to Allen et al. (2009),  $r_{leaf}$  values for plants can be found in the literature, allowing the computation of  $K_{cb}$ . When data on crop transpiration are not available for E/T partitioning, we recommend using the single approach model version (ET, water and salinity stresses) but the  $K_c$  should be selected with caution. Since  $K_c$  is a crucial value which varies a lot in the literature, values issued from specific measurements of the studied crop should be preferred.

We recommend accounting for salinity stress because it can significantly affect the water flows. However, in situation of high salinity levels, we recommend to also calculate the higher ET value without salinity stress, to have a range of ET values: the minimum value when salinity stress is accounted for and the maximum value when salinity stress is neglected (like in figure 4.11).

#### 4.3.7. Model outputs comparison with databases

Two water inventory databases provide an estimation of water consumption by crop at a country scale with global coverage (Water Footprint Network; Pfister et al. 2011; Pfister and Bayer 2014) (Cf. chapter 3 for a detailed description). The only E.T. model outputs we could compare with these databases were  $ET_{a\ blue}$ , and  $ET_{a\ green}$  (because no information about water quality or water released was provided). The estimates of  $ET_{a\ blue}$  were 60% higher in the WaterStat databases than the values provided by the E.T. model (average of  $ET_{a\ blue}$  with and without salinity stress included:  $177.4\ m^3 \cdot ton^{-1}$ ) (Fig 4.11). When comparing  $ET_{a\ green}$ , the difference was greater (597%). Pfister and colleagues estimates were closer to E.T. model estimation, but the database update in 2014 considerably decreased the initial value of 2011.

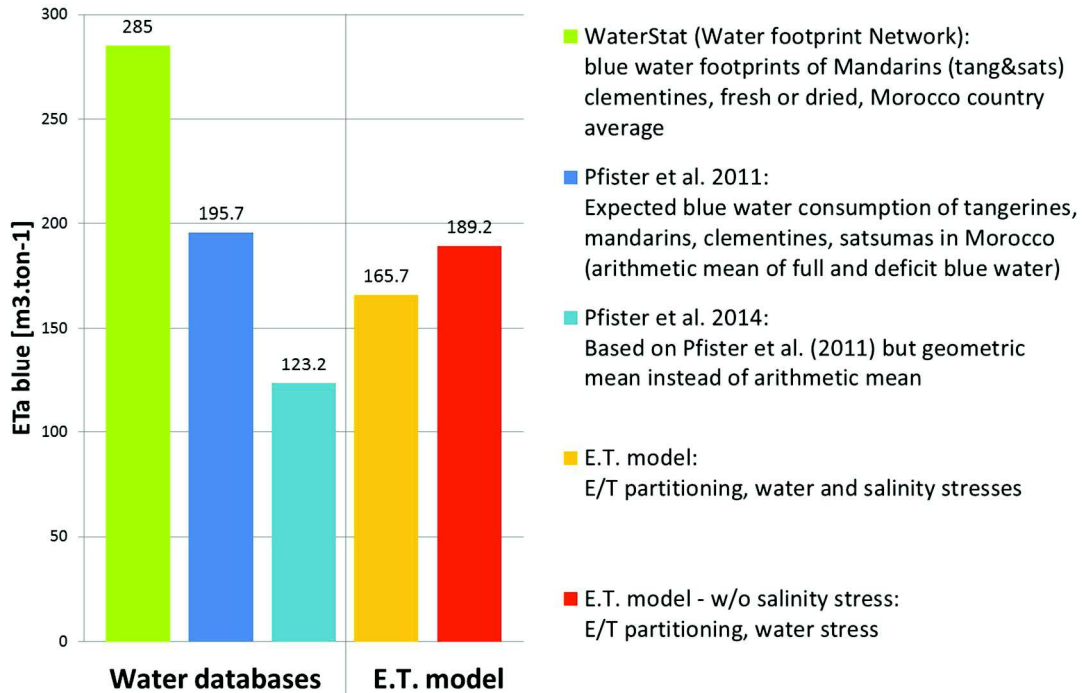


Figure 4.11. ETa blue [m3.ton-1] of small citrus provided by water inventory databases at country scale (WaterStat, Pfister et al. 2011 and 2014) and by the E.T. model for a Mandarin Nadorcott orchard located in South-West Morocco.

This comparison should be considered with caution because of the green and blue water partitioning: the assumptions in the three approaches were different. If estimating the actual evapotranspiration of a crop is complex, further dividing it in terms of blue and green waters is questionable from a hydrological perspective. First, it is important to remind that the green and blue water partitioning is required for the application of water use impact assessment methods in LCA. That is why we addressed this question in this study. Pfister and colleagues (2011) calculated green water thanks to empirical equations from USDA and FAO which are used in Cropwat 8.0 model to compute the crop water requirement (once the potential crop evapotranspiration has been assessed) (FAO 2009). These equations estimate the effective rainfall (rainfall minus losses through runoff and percolation) as a function of rainfall intensity. According to the water Footprint Network, the green water can be determined by a water balance of a non-irrigated crop (with only rainfall) but with the characteristics of an irrigated crop (e.g. rooting depth) (Mekonnen and Hoekstra 2011). However, since RO depends on the water saturation of the soil, calculating such a water balance is questionable because RO will be underestimated without irrigation input. That is why we used a different approach for estimating green water: we considered that  $ET_{a, green}$  is the infiltrating water ( $P - RO$ ), minus the green percolating water ( $DP_{green}$ ). Because DP contains a mixture of green and blue waters, we estimated  $DP_{green}$  as a share of deep percolating water originating from infiltrating rainfall.

In addition, the definition of effective rainfall itself is questionable. As emphasised in the FAO report N°25, this term is interpreted differently not only by specialists in different fields but also by different workers in the same field (Dastane 1978). From an agricultural production viewpoint, if we consider effective rainfall as the “*portion of rainfall which is useful directly and/or indirectly for crop production*” (Dastane 1978), the water use to leachate salts can be considered as “effective” since this is a condition

for crop growth. Thus, including deep percolating water ( $DP_{green}$ ) in the effective rainfall would reduce  $ET_{a\ blue}$  by 13%.

It may seem contradictory but using more blue water for irrigation increases the green water consumption because the crop roots are more developed and explore a wider soil space (this result is visible with simulation of deeper rooting depth with the E.T. model in part 4.3.3.).

Partitioning blue from green water is a non-sense from a hydrological viewpoint. Dividing a single water stock in the soil available for crop is very artificial and gives the illusion that green and blue water are separate “water stocks” (Perry 2014). Yet, exploiting more green water through water harvesting for example will reduce blue water availability. The use of green water is not considered to have a direct impact in LCA. The attempt to account for green water use impacts leads to results showing that agricultural systems consume less water than the natural ecosystem (Núñez et al. 2013). But this neglects the other positive effects of natural vegetation: high evaporative rates reconstitute more rainfall nearby. Furthermore, the use of green water has an influence on the water cycle and potentially reduces the amount of blue water needed because green water will become blue water if not lost through evapotranspiration. These observations on the inconsistencies of dividing water flows in blue and green components are arguments in favour of a more hydrological approach of water flows considering water as part of a balanced water cycle. The impacts of water use should be addressed by accounting for effects of water use on the hydrological water flows at (sub)catchment scale, within the global water cycle.

### 4.3.8. Model usage within a LCA study

#### 4.3.8.1. Model use for field emission estimation

Since water is the vector of many substances to the water compartment (pollutants, nutrients...etc), the water flows estimated with the E.T. model may enhance the assessment of other field substance emissions. By combining the E.T. model outputs with the outputs of a nitrogen balance for instance, one can estimate nitrate leaching: the leachable nitrate in the soil and the deep percolating water will provide an estimation of nitrate leaching. By combining the E.T. model outputs with a pesticide partitioning (post application) model, one can estimate the amount of pesticide in runoff water as well.

#### 4.3.8.2. Model use for water use impact assessment

Not only the model outputs are providing useful data regarding the water use efficiency and the water flows repartition in the field, but they are also meant to be used for water use impact assessment in LCA. They will serve as water elementary flows and will be multiplied by the corresponding characterisation factors. Depending on the water use impact assessment method applied, the water flows required will be different (Fig. 4.12).



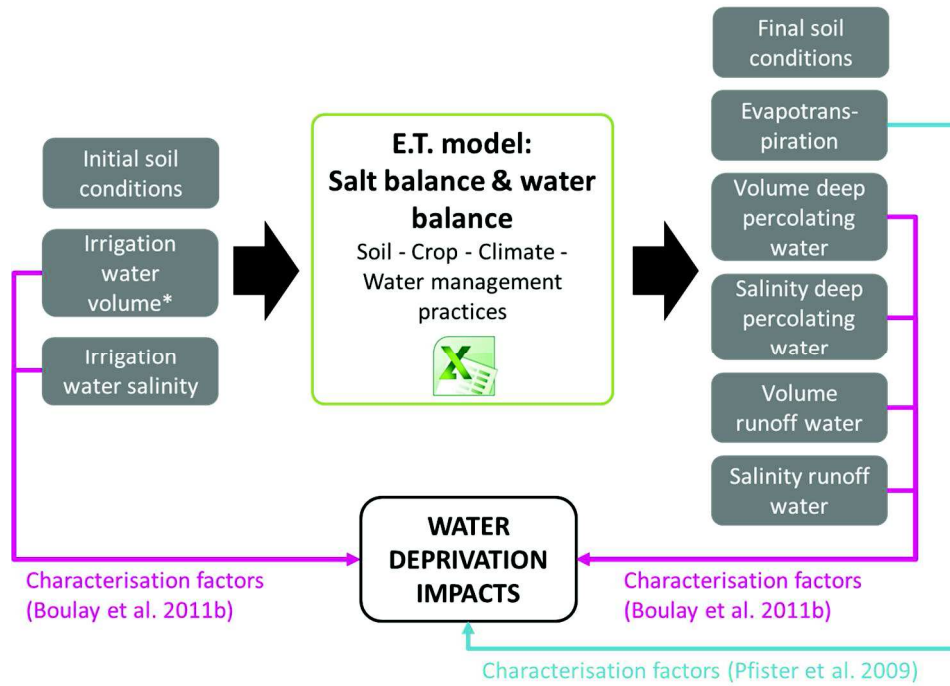


Figure 4.12. E.T. model integration within a LCA study: input and output water flows will be converted in terms of impacts on the environment thanks to existing water impact assessment methods (e.g.: Pfister et al. (2009) and Boulay et al. (2011a, and b))

The model outputs will be used for the LCA of a Moroccan Mandarin in the next chapter to test its relevance and applicability.

#### 4.4. Conclusion

The E.T. model was developed to fill a gap: the lack of a simple water and salt flows model for perennials, and meet an objective: doing the inventory of field water and salt flows for the LCA of a cropping system. The E.T. model is a modular and original integration of old and robust concepts for water balance with more recent modules for transpiration estimation. This is a tailored model rather than a new model. Its assets are its simplicity, transparency and flexibility. It meets the requirements of estimating evaporation, transpiration, deep percolating water and runoff water accounting for possible water and salinity stresses, and based on effective irrigation supply and cropping systems characteristics. It also provides information about the quality of water flows: salinity of deep percolating water and soil water stock. When applied to a perennial crop (Mandarin grown in Morocco), the E.T. model outputs compared well with literature and measurements, and allowed the simulation of scenarios of agricultural practices. Its validity domain (in terms of agricultural practices and natural environment (aquifer depth, salinity level)) and accuracy could be extended thanks to recommendations provided in this work. The E.T. model outputs will serve as water inventory elementary flows to assess the impacts of water use. The use of E.T. model for estimating field water and salt flows ease the application of water use impact assessment methods, including the method addressing both consumptive and

degradative water use (e.g. Boulay et al. 2011). The next steps are an uncertainty analysis of the model outputs, and its application within a full LCA study.

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## References Chapter 4

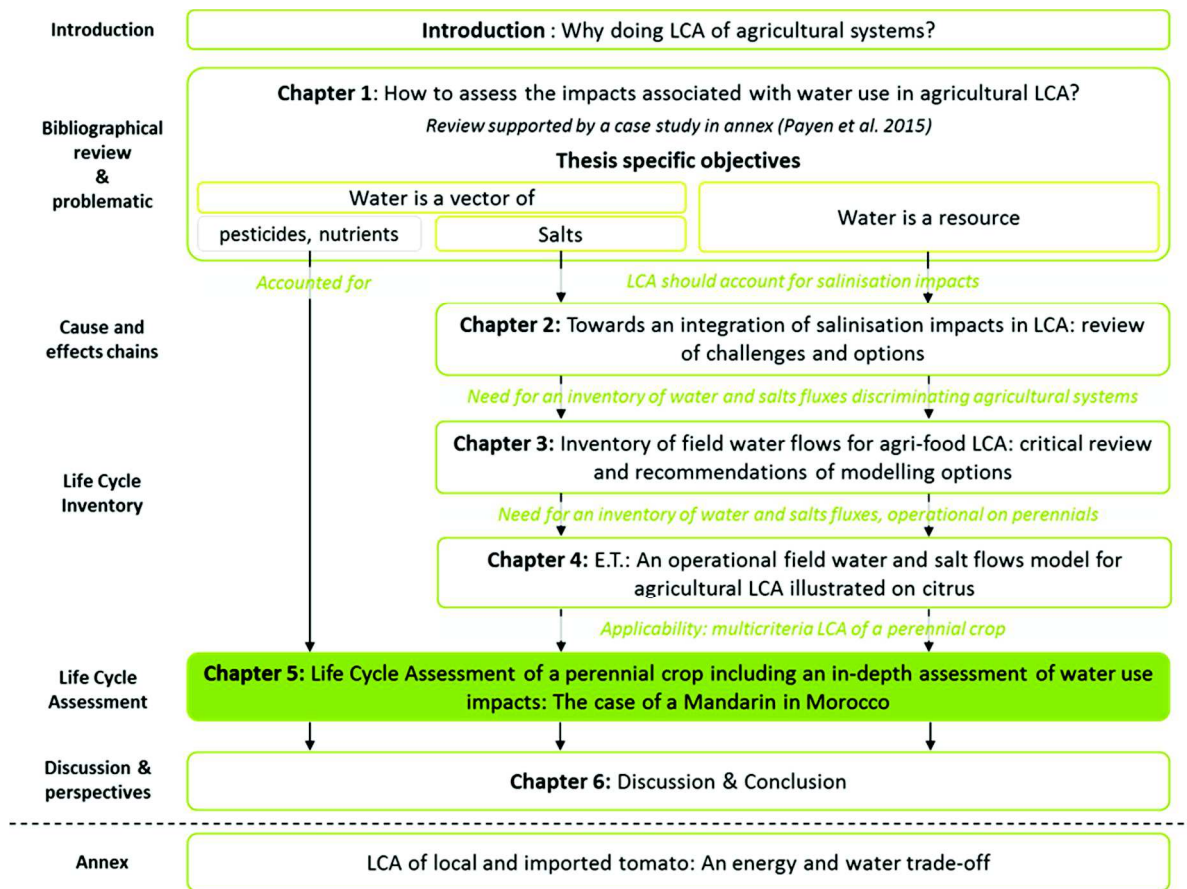
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We proposed a simple model for the inventory of water and salt fluxes of cropping systems, suitable for perennials: the E.T. model. In the following, we will test its applicability and relevance within a LCA study, and analyse the effect of different (i) water flows inventory and (ii) water impact assessment methods on the outcomes of a LCA.



## Chapter 5

### Life Cycle Assessment of a perennial crop including an in-depth assessment of water use impacts: The case of a Mandarin in Morocco

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Sandra Payen <sup>1,2</sup>, Claudine Basset-Mens <sup>2</sup>, Henri Vanni re <sup>2</sup>

<sup>1</sup>ADEME, 20 avenue du Gr sill , F-49004 Angers, France

<sup>2</sup>CIRAD, UPR Hortsys, ELSA – research group for environmental life cycle sustainability assessment, Boulevard de la Lironde, F-34398 Montpellier, France

#### Abstract

**Context and purpose:** Although citrus is the second largest fruit traded throughout the world and an irrigation-dependent crop in most producing countries, existing LCA studies do not assess its impacts associated with water use. The objectives of this article are (i) to assess the environmental impacts of producing and exporting a Mandarin grown in Morocco with a cradle-to-market-gate LCA, (ii) to calculate the impacts related to water use thanks to an inventory based on a field water and salts balance model (E.T. model), (iii) to analyse the influence of the inventory method and impact assessment method on the water use impacts. **Methods:** The 25-year perennial crop cycle of a large commercial Mandarin orchard (8-year old) was modelled from nursery to end-of-life, based on primary data and a projection scenario. The life cycle of the Mandarin was modelled up to the French market, accounting for post-farm production stages and transportation from Morocco to France. The E.T model provided estimates of water and salt elementary flows required to calculate impacts related with consumptive water use with Pfister et al. (2009) and with both consumptive and degradative water use with Boulay et al. (2011 a&b). **Results and discussion:** The contribution analysis showed for most impact categories the major contribution of energy required for pumping water in the deep aquifer, revealing a water-energy nexus. Water availability indicator from Boulay’s method (addressing both consumptive and degradative use) was slightly higher than water scarcity indicator from Pfister’s method. The water availability impact with and without accounting for a salinity stress in the inventory of field water flows was between 189 and 212 m<sup>3</sup><sub>eq</sub> per ton Mandarin for the cultivation stage, and 29.9 m<sup>3</sup><sub>eq</sub> per ton for the nursery, packaging and transportation stages all together. Overall, water impacts had a major contribution to damages on the three areas of protection. The assessment of water consumption and degradation impacts proposed by Boulay et al. (2011a&b) is limited by: (i) the lack of good quality data on the groundwater resource limiting the reliability of its characterisation factor, (ii) the complexity to assess rainfall water degradation impacts which rely on the definition of a “reference state”, (iii) the absence of regionalised assessment supported by LCA software. **Conclusion:** the major contribution of water impacts to damages on the three areas of protection and the water-energy nexus warrant the application of other damage assessment methods at endpoint level. Further development of impact assessment method related to water use is required but is notably constrained by the data availability on the state of the world freshwater resources.

**Keywords:** Mandarin, LCA, water, availability, scarcity.

## 5.1. Introduction

Citrus (including orange, lime, mandarin, pomelo) are the second largest traded fruits throughout the world. Morocco is one of the major producers of the Mediterranean area, with an increasing cultivated area (CIRAD 2012). A favourite Mandarin variety of the country, the Nadorcott (Afourer), showed tremendous harvest with 75 000 tons in 2013-2014. However, this irrigated crop production is evolving in a context where the pressure on freshwater resource is increasing dramatically (ABHT 2007). Citrus is one of the most studied fruit in LCA with published studies on: Orange grown in Spain (Sanjuan et al. 2005), Sicily (Beccali et al. 2009; Pergola et al. 2013; Lo Giudice et al. 2013), or Brazil (Knudsen et al. 2011); Lime grown in Sicily (Beccali et al. 2009; Pergola et al. 2013), and small citrus grown in Morocco (Basset-Mens et al. 2015). In spite of the limited availability of water in most areas and the use of water for irrigation, none of these studies addressed the impacts associated with water use. There are solely reporting the volume of irrigation water. Although demonstrated by Bessou et al. (in press) as having a large effect on the LCA results for perennials, the orchard life cycle modelling was quite different from one study to another: it was either based on one single productive year (Sanjuan et al. 2005; Beccali et al. 2010), or on the productive years (4 to 20 years old orchard in (Knudsen et al. 2011), or on the whole orchard life like in this study (Lo Giudice et al. 2013; Pergola et al. 2013; Basset-Mens et al. 2015). Bessou et al. (2013) formalised the different options to model the perennial cropping systems depending on the objective of the study and data availability.

Mandarin production in the water-scarce Bahira plain in Morocco is an interesting case study illustrating (i) the challenge of a perennial crop modelling in LCA, (ii) high valuable crop production in water scarce area, (iii) cultivation of a crop sensitive to salinity in an area prone to high soil salinity. Indeed, in this area the pressure on water resource is worsened by salinity issues of the aquifer (El Mokhtar et al. 2012). Thus, this cropping system was considered a perfect case study for testing the use of water inventory flows provided by the E.T. model (Chapter 4) to calculate water use impacts within a complete LCA study (multicriteria).

The objectives of this paper are to:

- assess the environmental impacts of producing and exporting a Mandarin grown in Morocco with LCA, accounting for the whole perennial crop cycle,
- calculate the impacts related to water use thanks to an inventory of water flows based on a field water and salts balance model (E.T. model developed in chapter 4) and discuss its feasibility,
- analyse the influence of the inventory method and impact assessment method on the water use impacts of a Mandarin

## 5.2. Materials and methods

### 5.2.1. Geographical context

The studied farm is located in the Bahira plain (central Morocco), and counts 225 hectares of Nadorcott Mandarin orchard (Satellite image in supplementary materials). The area is characterised by a



continental arid climate (MEMEE): a low average rainfall (208 mm.year<sup>-1</sup>), a high potential evaporation (1755 mm.year<sup>-1</sup>) and average daily temperatures ranging from 0°C in winter to 36°C in summer (meteorological measurements on-farm from 2007 to 2015). The farm is isolated from other citrus agricultural farms to avoid any cross-pollination and have seed-less Mandarin.

100% of irrigation water is withdrawn from the underlying aquifer. Groundwater withdrawals are regulated by the Water Basin Agency, but in spite of the limitation of the number of well, the aquifer level is decreasing (ABHT 2007). The pressure on groundwater resource is worsened by an increased salinity (El Mokhtar et al. 2012). Thus, this Mandarin orchard relies on the freshwater resource of an aquifer which water quantity and quality are decreasing.

### 5.2.2. LCA goal and scope

With the goal of producing a complete LCA for a Moroccan export Nadorcott mandarin for the French market, we defined the functional unit as 1 kg of fresh mandarin delivered at the Saint-Charles International Market entry gateway (French distribution hub for fruits and vegetables). The system boundaries (Fig. 5.1) were from cradle to market (i.e., from raw material extraction to the market entrance gate) and included all direct inputs for seedling production, orchard establishment, mandarin production, packaging and transportation to the French market, but generally excluded capital items (except for fertigation materials).

Primary data were collected during in-depth field surveys in one seedling nursery, the perennial crop cycles (from 2007 to 2015) at farm level, and one packaging station. We used primary data for the consumption of agricultural inputs (fertilisers, pesticides), water, electricity and fuels, the amount of materials (packaging components, irrigation system components...), the use of agricultural machineries, the amount of final products for the nursery, the mandarin cultivation and the packaging stages. Following recommendations from Bessou et al. (2013, 2015), the whole orchard life was modelled thanks to primary data from plantation to present (2015), and based on projection scenario until expected uprooting, based on the producer and mandarin experts' knowledge. We considered a life span of 25 years. Table 5.1 shows key farm inventory data for the past period and the projection scenario. Secondary data such as input transportation and manufacturing, fuel consumption for truck refrigeration and freight ship container, were obtained from the literature and from the Ecoinvent 2.2 database (Frischknecht et al., 2007) referring to the average European context. Indeed, most packaging, fertilisers and pesticides are manufactured in Europe. Transport mode and distance of farm inputs were adjusted according to their actual origin. The LCA modelling was performed with Simapro 8 software (PRé Consultants 2011).

The farm is producing mandarin for exportation (80%) but grade-out mandarins are sold on the Moroccan local market (20%). An economic allocation was used, the volume of mandarin exported and sold locally were weighted by their price (6 times more important for the exported mandarin on average).

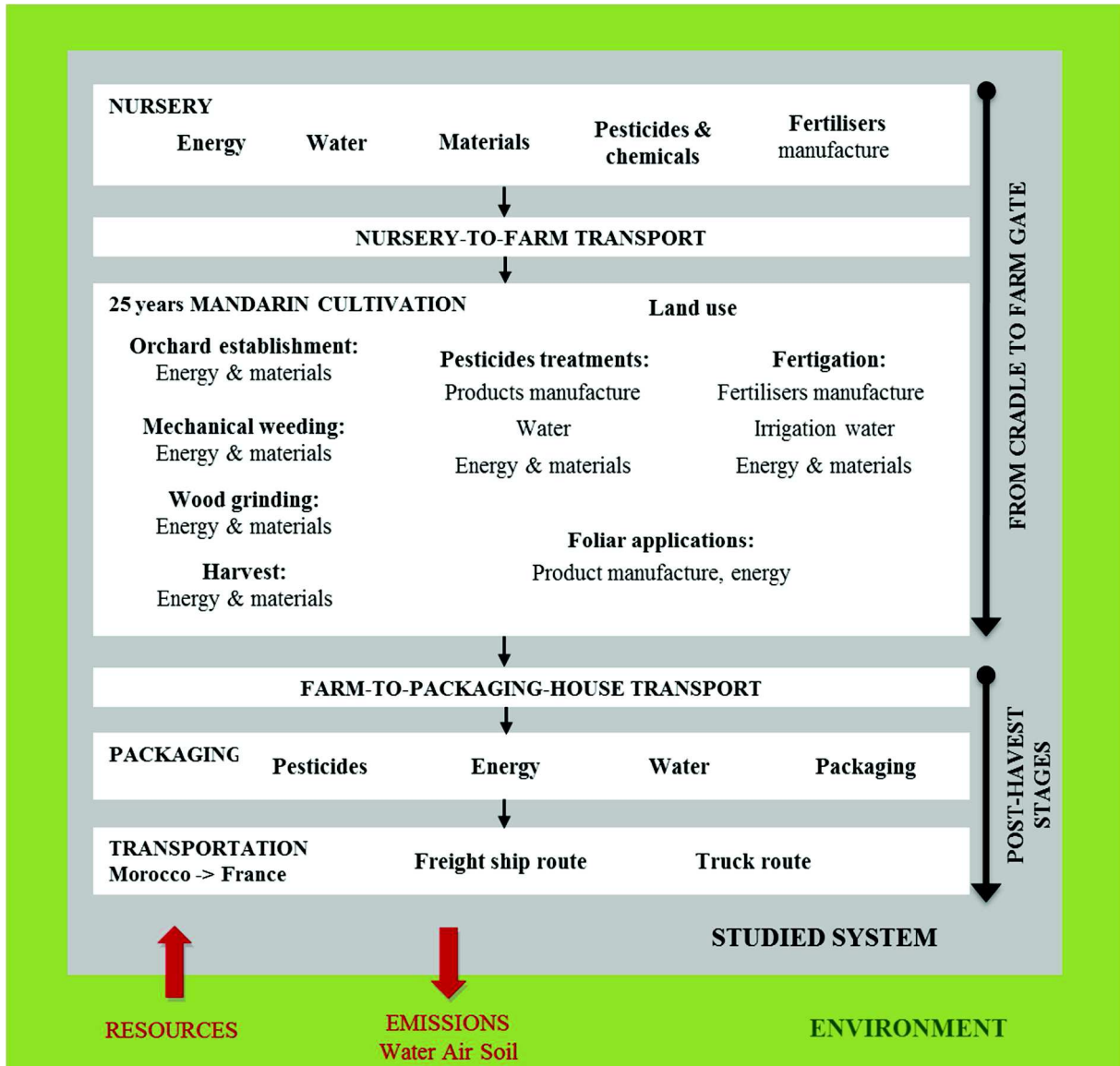


Figure 5.1. Flow diagram for the Moroccan Mandarin production and delivery to the French market (2007-2015, Bahira region, Morocco).

### 5.2.3. Inventory of Moroccan mandarin production: from cradle-to-farm-gate

#### 5.2.3.1 Nursery and mandarin cultivation

Mandarin production is based on grafted plants: a combination of a rootstock presenting good resistance to biotic and abiotic stresses (5 different rootstocks are present on farm, see supplementary information), and a productive scion (Afourer- Nadorcott). Plant grafting occurs at a nursery located in the Souss Massa, already surveyed in 2010 (Heitz 2010). Mandarin plant production at nursery requires energy, water, fertilisers, and pesticide treatments. After approximately 18 months, seedlings can be planted.

The orchard was planted in 2007 and 2008. The orchard establishment requires deep tillage, and the digging of a 55,000m<sup>3</sup> water basin and trenches. The energy consumption and the engines used during this phase were accounted for based on descriptions made by the farmer.

Harvest occurs from mid-December to the end of February. Average yield of the studied farm was 50 ton.ha<sup>-1</sup> for adult trees at full production (Table 5.1). This is above the range of reported yields for other mandarins because Nadorcott Mandarin is a highly productive one (Nadori 2005). After harvesting, mandarins are treated on-farm for conservation, transported to the packaging station in Casablanca where export quality mandarins are separated from non-export quality ones, and export quality mandarins are then transported to France by boat or truck.

Farmers estimate the water irrigation requirements through the calculation of the potential evapotranspiration of the crop. The theoretical crop water requirement (the maximum evapotranspiration) is adjusted according to the soil water status (analysed with tensiometers and soil profiles). The farm has six electric-pumps withdrawing water in the aquifer at 140 m depth (dynamic pumping depth), the aquifer being at a piezometric level of 75m. Irrigation is combined with fertilisation (fertigation) and managed through a computer. Initial fertilisers supply plans are adjusted according to soil and leaves analyses, and the farmer's expertise. The fertigation materials (well pumps, fertilizer mixing pumps, pressure pumps and irrigation hoses) were accounted for in the assessment.

Crop protection management is based upon pest monitoring. 15 active ingredients of pesticides were included in the study: their manufacture and emission after application were included.

Regarding energy consumption, the farm uses diesel and electricity for fertigation and pesticide treatments. Pumps are used for water extraction from wells, for water and fertilisers mixing and pressurisation in the drip irrigation system. Pesticides are applied using knapsack sprayers and atomisers. Energy consumption was calculated based on the pumps' specifications and their recorded operating time.

The electricity mix of Morocco of 2007 was used, in the absence of more recent data.

Regarding the end-of-life treatment, irrigation hoses, pesticides and fertilisers packaging are stored in the farm so far, while waiting for a reliable company to handle their recycling. Organic wastes such as rotten fruits, shoots and leaves are used for composting.

### **5.2.3.2. Projection scenario**

To model the future orchard life, projection scenarios were built from present (2015, 8-year-old orchard) to orchard uprooting (25-year-old orchard), based on the farmer expertise (Table 5.1). The yield used for projection (50 t.ha<sup>-1</sup>) is an average accounting for the alternating yields.

### **5.2.3.3. Water and salt flows inventory**

Field water flows were simulated with the E.T. model developed in chapter 4. Based on actual water supply (volume and salinity) and the soil, climate and practices specificities, the model estimated the water consumed through evapotranspiration, and the water released in the environment through deep percolation and runoff. The model also provided the salinity of deep percolating water and soil water. The modelling was based on primary data from planting to present (April 2015), whereas the projection

scenario until orchard end-of-life was based on an average of the last 3 years. For more details on the E.T. model, see chapter 4.

#### **5.2.3.4. Field emissions**

Nitrogen oxides, ammonia, phosphates and pesticides emissions were calculated according to Nemecek and Kägi (2007), nitrous oxide according to IPCC (Guidelines for National Greenhouse Gas Inventories) (2006). Phosphorus emissions through water erosion were not considered because the topography was flat. Following recommendations from Brentrup (2000), nitrate leaching was assessed by multiplying the nitrogen available for leaching with the leaching coefficient. The nitrogen available for leaching was estimated as the N surplus of the nitrogen budget: inputs N include mineral and organic fertilisers, and nitrates naturally present in water, whereas N outputs include N field emissions (ammonia, nitrous oxides, nitrogen gas), and N exported in the tree (function of the volume of wood) and in the fruits. The leaching coefficient was calculated based on Bockstaller and Girardin (2003) and Pervanchon et al. (2005), using the deep percolating water flow estimated thanks to E.T. model. See supplementary information for more details on nitrate leaching assessment.

Pesticides were assumed to be emitted to the soil (Nemecek and Kägi 2007). Temporary biogenic carbon fixation in biomass was not accounted for since its inclusion has no implication on the results due to a lack of associated characterisation factor.

Table 5.1. Agronomic data summary for the main orchard development phases: average yield, NPK fertilisation, irrigation volumes and energy, main substances for plant protection

Intervention	Unit	Non-productive: 0 - 2 years	Growing yield: 3 - 6 years	Full production		Projection scenario description, based on farmer expertise	
				7 - 8 years	9 - 25 years: projection		
<b>Yield</b>	t.ha <sup>-1</sup>	0	33	60	50	Average accounting for the alternating yield (between 50 and 55 t.ha <sup>-1</sup> ), and the uprooting of half of the trees (over 4 years: between 15 to 18-year old)	
<b>Fertilisation</b>	N	kg.ha <sup>-1</sup>	161.1	311.3	216.4	237.6	Average of nutrient supply for a 6 & 7-year old orchard (2012/2013 & 2013/2014)
	P <sub>2</sub> O <sub>5</sub>	kg.ha <sup>-1</sup>	21.6	38.6	21.6	49.9	
	K <sub>2</sub> O	kg.ha <sup>-1</sup>	91.9	218.2	214.4	179.7	
<b>Irrigation</b>	Water	m <sup>3</sup> .ha <sup>-1</sup>	2263	7602	11413	11000	Irrigation water is limited to 11000 m <sup>3</sup>
	Energy	MJ.ha <sup>-1</sup>	8858	32005	47114	42623	Based on the pumps power and flow, and irrigation water volume
<b>Plant protection (main active substances)</b>							
<i>California red scale</i>	<i>Spirotetramat*</i>	kg.ha <sup>-1</sup>	0	0	0	2.085	From 2016 to 2023: 1/3 of the farm area treated each year, from 2024 to uprooting: 1/2 of the farm area treated each year
<i>Citrus leaf miner</i>	<i>Imidacloprid*</i>	kg.ha <sup>-1</sup>	0.380	0.778	0.467	0	
<i>Citrus leaf miner</i>	Abamectin	kg.ha <sup>-1</sup>	0	0	0.010	0	Abamectin proved to be not efficient
<i>Mediterranean fruit fly</i>	Malathion	kg.ha <sup>-1</sup>	0.075	0.156	0.522	0.522	Average of a 7 & 8-year old orchard (2013/2014 & 2014/2015)
<i>Mediterranean fruit fly</i>	Lambda-cyhalothrin	kg.ha <sup>-1</sup>	0.017	0.021	0.062	0.062	Average of a 7 & 8-year old orchard (2013/2014 & 2014/2015)
<i>Mediterranean fruit fly</i>	Dimethoat	kg.ha <sup>-1</sup>	0	0	0.008	0	Dimethoat is now prohibited
<i>Spider mite</i>	Clofentezin	kg.ha <sup>-1</sup>	0	0	0.006	0.006	Average of a 7 & 8-year old orchard (2013/2014 & 2014/2015)
<i>Spider mite</i>	Dicofol	kg.ha <sup>-1</sup>	0.121	0.508	0	0	Dicofol is now prohibited
<i>Snail</i>	<i>Metalddehyd*</i>	kg.ha <sup>-1</sup>	0	2.259	2.932	2.932	Average of a 7 & 8-year old orchard (2013/2014 & 2014/2015)
<i>Herbicide</i>	Glyphosate	kg.ha <sup>-1</sup>	3.085	1.801	1.444	1.444	Average of a 7 & 8-year old orchard (2013/2014 & 2014/2015)
<i>Fungicide</i>	Fosetyl-Aluminium	kg.ha <sup>-1</sup>	0	0.73	0	0.196	Average of a 7 & 8-year old orchard (2013/2014 & 2014/2015)
	Total herbicides	kg.ha <sup>-1</sup>	3.085	1.801	1.444	1.444	
	Total insecticides	kg.ha <sup>-1</sup>	0.593	3.723	4.006	5.606	
	Total fungicides	kg.ha <sup>-1</sup>	0	0.73	0	0	
	TOTAL pesticides	kg.ha <sup>-1</sup>	3.678	6.252	5.646	7.245	

\* Active substances not characterised in the impact assessment due to a lack of Characterisation Factors

### 5.2.3.5. Inventory of post-farm production stages

Data related to packaging were collected from a packaging station located in Casablanca (Heitz 2010). Mandarins are successively treated with a fungicide (Imazalil), washed, dried, sorted, packed in wooden boxes, palletised, and stored in cold rooms before transport. Recently, the fungicide treatment with Guazatine usually done at the packaging station was displaced to the farm. But since it is a post-harvest treatment, it was included in the “packaging stage” for contribution analysis.

Export mandarin to France takes two possible routes: by ship in refrigerated containers (20%) or by truck in a 38- to 44-ton refrigerated truck (80%). Freight ship goes from the port of Casablanca (Morocco) to Port-Vendres (France), including truck drives to and from the ports. Trucks go from Casablanca to the Saint-Charles market in France, including a ferry through Gibraltar Detroit. The fuel consumption for traction and refrigeration for refrigerated trucks was taken from Tassou et al. (2009). With the sea route, the products are transported by freight ship in forty-foot refrigerated containers. The fuel consumption for ship propulsion and for container cooling was based on reports from the International Maritime Organisation (Buhaug et al., 2009) and Wild et al. (1999, 2005).

### 5.2.4. Life cycle impact and damage assessment

The impact assessment phase was performed using the ReCiPe life cycle impact assessment method (Goedkoop et al. 2013), adopting the Hierarchist perspective. The following environmental impact categories were considered: climate change (100 years; kg CO<sub>2eq</sub>); terrestrial acidification (g SO<sub>2eq</sub>); freshwater and marine eutrophication (g P<sub>eq</sub> and g N<sub>eq</sub> respectively, based on the nutrient-limiting factor of the aquatic environment); terrestrial, freshwater, and marine ecotoxicity (g 1,4-DB<sub>eq</sub>: 1,4-dichlorobenzene); human toxicity (g 1,4-DB<sub>eq</sub>); agricultural land occupation (m<sup>2</sup>.year); metal and fossil depletion (g Fe<sub>eq</sub> and kg oil<sub>eq</sub>).

In addition, the impacts of water use were assessed with the method of Pfister et al. (2009)(water scarcity indicator), and two versions of Boulay’s method (Boulay et al. 2011a&b) (water scarcity and availability indicators), expressed in m<sup>3</sup><sub>eq</sub> deprived per ton. For each method, the calculation of freshwater deprivation is based on a different inventory water flows requirement as illustrated on figure 5.2 (see Chapter 4 for details). Water elementary flows were based on primary data (water withdrawn), and on modelling (water flows released and consumed) (Table 5.2). The Characterisation Factors (CF), so called *water stress index (WSI)* are basin-specific for the nursery, Mandarin cultivation and packaging life cycle stages, and global for transportation and background processes (e.g.: fertilizer manufacture) (Table 5.2). Regarding the definition of water categories for the Mandarin cultivation stage (required for applying Boulay’s method), the category was defined based on 3 parameters: salinity (through total dissolved salts), total nitrogen, and nitrates content. The water quality category was driven by salinity for the withdrawn water, and by the nitrogen content for the released water. Details of CF for each method and elementary flows are provided in table 5.2.

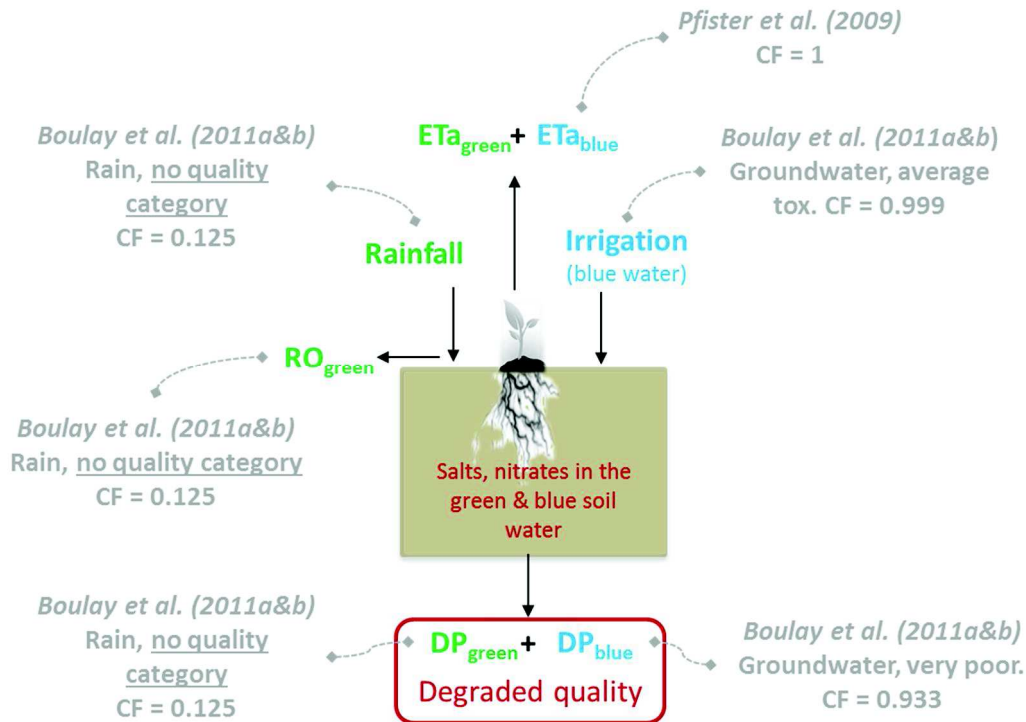


Figure 5.2. Representation of elementary water flows and associated characterisation factors of Pfister et al.'s (2009) and Boulay et al.'s (2011a&b) methods:

water flows from irrigation and rainfall (based on primary data), evapotranspiration from green ( $ETa_{green}$ ) and blue ( $ETa_{blue}$ ) water, and water released in the environment through runoff ( $RO_{green}$ ) and deep percolation ( $DP_{green}$  and  $DP_{blue}$ ) (estimated with the E.T. model). In the soil, water from irrigation and from rainfall are mixed with salts and nitrates, thus deep percolating water has a degraded quality while Boulay et al.'s (2011a&b) method proposes no characterisation factor for degraded rain water.

Water consumption of the background processes was quantified using the Ecoinvent 2.2 database (Frischknecht et al., 2007). Adopting a conservative approach, this water was considered as consumed. We then explored the aggregation of impacts into damages and analysed the contribution of freshwater deprivation to damages on the three areas of protection: Human health, Resources and Ecosystems. Water consumed during background processes was assigned the global average damage factor for each process.

Owing to the scope of the article, and preliminary studies on citrus and tomato production in Morocco showing that water and energy consumption are the hotspots for the crop production (Basset-Mens et al. 2015; Payen et al. 2015), we will focus on the sensitivity regarding the inventory modelling and impact assessment methods.

**Table 5.2. Data sources for volume and quality of water elementary flows, and associated characterisation factors (CF) for Boulay et al. (2011,a&b) (water availability indicator) and Pfister et al. (2009) methods (water scarcity indicator).**

Life Cycle Stage	Water elementary flow	Data source for volume and quality of water flows	Boulay et al. (2011, a& b)			Pfister et al. (2009)	
			Water flow category	CF [m <sup>3</sup> <sub>eq</sub> /m <sup>3</sup> ]	Spatial scale of CF	CF [m <sup>3</sup> <sub>eq</sub> /m <sup>3</sup> ]	Spatial scale of CF
Nursery	Water, from ground water, Souss Massa (Morocco)	Volume of water withdrawn: primary data Assumption: all water consumed, no quality degradation	G2a: "groundwater, good"	0.999	Morocco, big basin scale, ground water	1	Morocco, basin scale
Mandarin cultivation (25 years)	Water, from ground water, Bahira (Morocco)	Volume and quality of water withdrawn: primary data	G2c: "groundwater, average tox."	0.999	Morocco, big basin scale, ground water	1	Morocco, basin scale
	Water, to ground water, Bahira (Morocco)	Volume and quality of water released: E.T. model, and nitrogen budget	G4: "groundwater, very poor"	0.933	Morocco, big basin scale, ground water		
	Rain	Volume of rainfall: primary data	No category: Rain	0.125	Morocco, big basin scale, ground water	-	-
	Rain	Volume of rainwater released: E.T. model, No quality degradation was considered because Boulay et al. (2011b) and Pfister et al. (2009) do not address rain quality degradation	No category: Rain	0.125	Morocco, big basin scale, rain	-	-
Packaging	Water, river Casablanca	Volume of water withdrawn: primary data Assumption: all water consumed, no quality degradation	S3: "Surface water, poor"	1	Morocco, big basin scale, surface water	0.9999	Morocco, basin scale
Background processes, not geolocalised	Water, lake	Volume of water from Ecoinvent database, Assumption: all water consumed	No quality data, CF of consumptive use only	0.658	Global	0.669	Global
	Water, river			0.658		0.669	
	Water, unspecified natural origin			0.710		0.669	
	Water, well, in ground			0.694		0.669	



### 5.2.5. Comparison with published LCA studies on Citrus

Apart from very recent LCA studies on small citrus (Basset-Mens et al., 2015), most published LCA studies on citrus were based on the CML (2001) (Guinée et al. 2002) and the Cumulative Energy Demand (CED) (Frischknecht, 2007) impact assessment methods, we also calculated the impact of the Mandarin using the same methods for comparison. As a result, we carefully compared our cradle-to-farm-gate Mandarin results with the cradle-to-farm-gate LCA results of integrated orange from Spain (Sanjuan et al. 2005), conventional orange from Italy and Brazil (Beccali et al. 2010; Knudsen et al. 2011), based on CML (2001) and the CED. Nevertheless, the comparison of results from different LCA studies should be taken with caution because they may have different scope and objectives, and may rely on different assumptions. The impact categories included for the comparison were: climate change (100 years, in kg CO<sub>2eq</sub>), non-renewable energy consumption (in MJ<sub>eq</sub>), eutrophication (g PO<sub>4</sub><sup>3-eq</sup>), terrestrial acidification (g SO<sub>2eq</sub>). It is noteworthy that this impact assessment using an “out-dated” method such as CML (2001) is only meant for comparative analysis and the indicators results should not be used as such. Finally, results for our Mandarin from Morocco were compared with results for an average Clementine from Morocco as part of the Agribalyse program (Basset-Mens et al., 2015) using the same ReCiPe Midpoint (H) method.

## 5.3. Results

We analysed the hot-spots contribution from cradle-to-market gate and from cradle-to-farm gate for both Midpoint and Endpoint results.

### 5.3.1. Market gate - midpoint

We analysed the impacts of the Mandarin at French market gate, and the contribution of the cultivation, packaging, and transportation stages (from cradle-to-market gate) (Fig. 5.3). Mandarin cultivation was the main contributor to most impact categories: metal depletion, water deprivation (Pfister et al. 2009), agricultural land occupation, terrestrial ecotoxicity, marine and freshwater eutrophication, terrestrial acidification and human toxicity, ranging from 52.9% to 98.2% (Fig. 5.3). Mandarin transportation by truck was the main contributor to climate change, freshwater and marine ecotoxicity, and fossil depletion, ranging from 43.1% to 52.2% (Fig. 5.3). This is mainly due to carbon dioxide emissions and crude oil resource use. The packaging had a non-negligible contribution to (eco)-toxicities, mainly due to the use of fungicides (Imazalil and Guazatine).

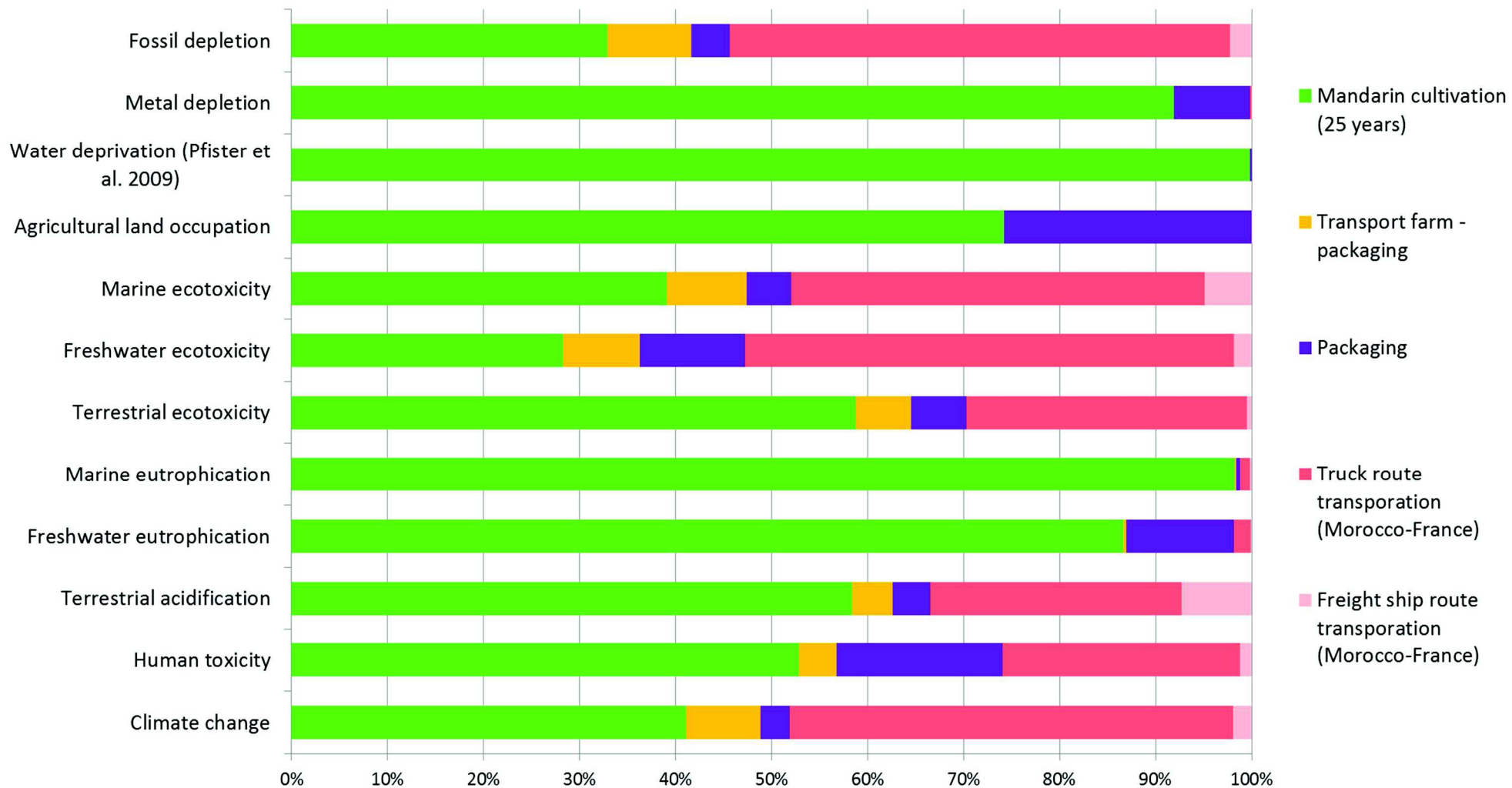


Figure 5.3. Contribution analysis of 1kg of Moroccan Mandarin at French market gate. Midpoint impacts were assessed with ReCiPe Midpoint (H) V1.12

### 5.3.2. Farm gate - midpoint

We further analysed the impacts of the main contributor: the Mandarin cultivation stage to most impact categories (from cradle-to-farm gate) (Fig. 5.4, Table 5.3). The contribution analysis showed that fertigation was the main contributor to most impact categories.

For fossil depletion, freshwater eutrophication, terrestrial acidification, human toxicity and climate change, electricity consumption for fertigation was the main contributor (from 67.3% for fossil depletion to 46.9% for human toxicity) (Fig. 5.4). For marine and freshwater ecotoxicity, electricity had the second greatest contribution with 29% and 29.8%, respectively with important contributions from Vanadium released during electricity production in oil power plant and Bromine emission to water during extraction of crude oil (Phyllis 2001). For marine ecotoxicity and metal depletion, the manufacturing of fertilisers was the main contributor with 61.1% and 46.8% respectively, because of zinc emission to air during the manufacture of zinc monosulphate, and manganese use for the manufacture of manganese sulphate. Manganese is an important oligoelement for plant, in particular for citrus (Obreza and Morgan 2008). The manufacturing of fertilisers was the second most important contributor for fossil depletion, terrestrial ecotoxicity, human toxicity and climate change ranging from 18.3% to 40.5%. For terrestrial ecotoxicity, the pesticides application (active substance manufacture and emissions) was the main contributor with 69%. Field emission of Lambda-cyhalothrin contributed 27.2%, Fosetyl-aluminum 22.5%, and Malathion 14.8%. For marine eutrophication, field emissions due to fertiliser application represented the main contributor with 98.9%, due to nitrate leaching emission to water (98.6%). Agricultural land occupation was largely dominated by the land occupation of the orchard. For water deprivation, irrigation water of the orchard was the main contributor for both methods (Pfister et al. 2009 and Boulay et al. 2011), with 86.2% and 86.8 %, respectively. The contribution of background processes was negligible (less than 1%) (Fig. 5.5). The two impact assessment methods (including two versions for Boulay et al. (2011a, b)) provide different water deprivation results: impacts from water consumptive use assessed with Pfister et al. (2009) ( $209.4 \text{ m}^3_{\text{eq}}.\text{ton}^{-1}$ ), are greater than assessed with Boulay et al. (2011a, b) ( $121.3 \text{ m}^3_{\text{eq}}.\text{ton}^{-1}$ ). Impacts from water consumptive and degradative use ( $217.7 \text{ m}^3_{\text{eq}}.\text{ton}^{-1}$ ), are greater than impacts from consumptive use only when assessed with Boulay et al. (2011a, b) (Fig. 5.5).

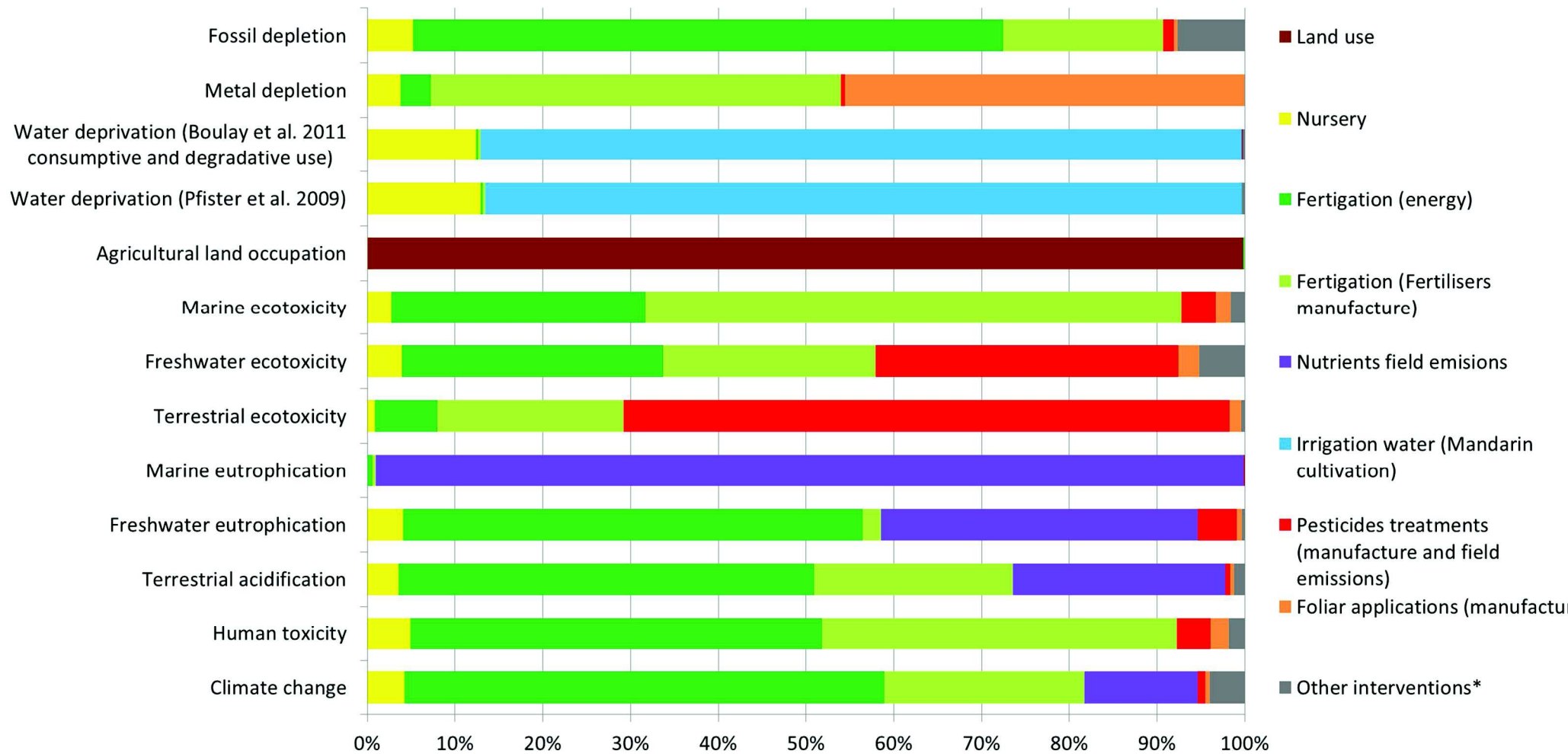


Figure 5.4. Contribution analysis of 1kg of Mandarin at farm gate in Morocco. Midpoint impacts were assessed with ReCiPe Midpoint (H) V1.12

Table 5.3. Cradle-to-farm-gate LCA results per kg of Mandarin for a selection of environmental indicators (Midpoint impacts assessment with ReCiPe Midpoint (H) V1.12)

Impact category	Unit	Total	Land use	Nursery	Fertigation: energy	Fertigation: Fertilisers manufacturing	Nutrients field emissions	Irrigation water (Mandarin cultivation)	Pesticides treatments: manufacturing and emissions	Foliar applications: manufacturing	Other interventions *
Climate change	kg CO <sub>2</sub> eq	<b>2.74E-01</b>	0.00E+00	1.17E-02	1.50E-01	6.28E-02	3.55E-02	0.00E+00	2.29E-03	1.25E-03	1.11E-02
Human toxicity	kg 1,4-DB eq	<b>9.35E-03</b>	0.00E+00	4.60E-04	4.38E-03	3.78E-03	0.00E+00	0.00E+00	3.60E-04	1.96E-04	1.67E-04
Terrestrial acidification	kg SO <sub>2</sub> eq	<b>1.86E-03</b>	0.00E+00	6.64E-05	8.80E-04	4.22E-04	4.50E-04	0.00E+00	9.79E-06	7.44E-06	2.34E-05
Freshwater eutrophication	kg P eq	<b>2.09E-05</b>	0.00E+00	8.51E-07	1.09E-05	4.37E-07	7.54E-06	0.00E+00	9.30E-07	1.28E-07	6.32E-08
Marine eutrophication	kg N eq	<b>3.83E-03</b>	0.00E+00	2.05E-06	1.89E-05	1.57E-05	3.78E-03	0.00E+00	2.71E-06	9.04E-07	4.41E-07
Terrestrial ecotoxicity	kg 1,4-DB eq	<b>6.50E-05</b>	0.00E+00	5.48E-07	4.65E-06	1.38E-05	0.00E+00	0.00E+00	4.49E-05	8.41E-07	2.80E-07
Freshwater ecotoxicity	kg 1,4-DB eq	<b>7.22E-05</b>	0.00E+00	2.85E-06	2.15E-05	1.75E-05	0.00E+00	0.00E+00	2.49E-05	1.71E-06	3.76E-06
Marine ecotoxicity	kg 1,4-DB eq	<b>2.82E-04</b>	0.00E+00	7.71E-06	8.19E-05	1.73E-04	0.00E+00	0.00E+00	1.08E-05	4.95E-06	4.52E-06
Agricultural land occupation	m <sup>2</sup> .a	<b>2.76E-01</b>	2.75E-01	5.17E-05	2.92E-04	3.43E-05	0.00E+00	0.00E+00	3.84E-06	1.22E-06	1.64E-04
Water deprivation (Pfister et al. 2009)	L H <sub>2</sub> Oeq	<b>2.09E+02</b>	0.00E+00	2.70E+01	5.64E-01	6.34E-01	0.00E+00	1.81E+02	7.72E-03	5.42E-03	6.25E-01
Water deprivation (Boulay et al. 2011 consumptive & degradative)	L H <sub>2</sub> Oeq	<b>2.18E+02</b>	0.00E+00	2.70E+01	5.61E-01	6.70E-01	0.00E+00	1.89E+02	7.89E-03	5.58E-03	6.25E-01
Metal depletion	kg Fe eq	<b>1.43E-02</b>	0.00E+00	5.43E-04	4.93E-04	6.70E-03	0.00E+00	0.00E+00	7.39E-05	6.52E-03	8.81E-06
Fossil depletion	kg oil eq	<b>6.23E-02</b>	0.00E+00	3.24E-03	4.19E-02	1.14E-02	0.00E+00	0.00E+00	7.08E-04	3.25E-04	4.75E-03

\*Other interventions include: orchard establishment, fertigation (materials), mechanical weeding and wood grinding, pesticides treatments (water and materials), pesticides treatments (energy), foliar applications (energy), and fruit harvesting (energy & materials).

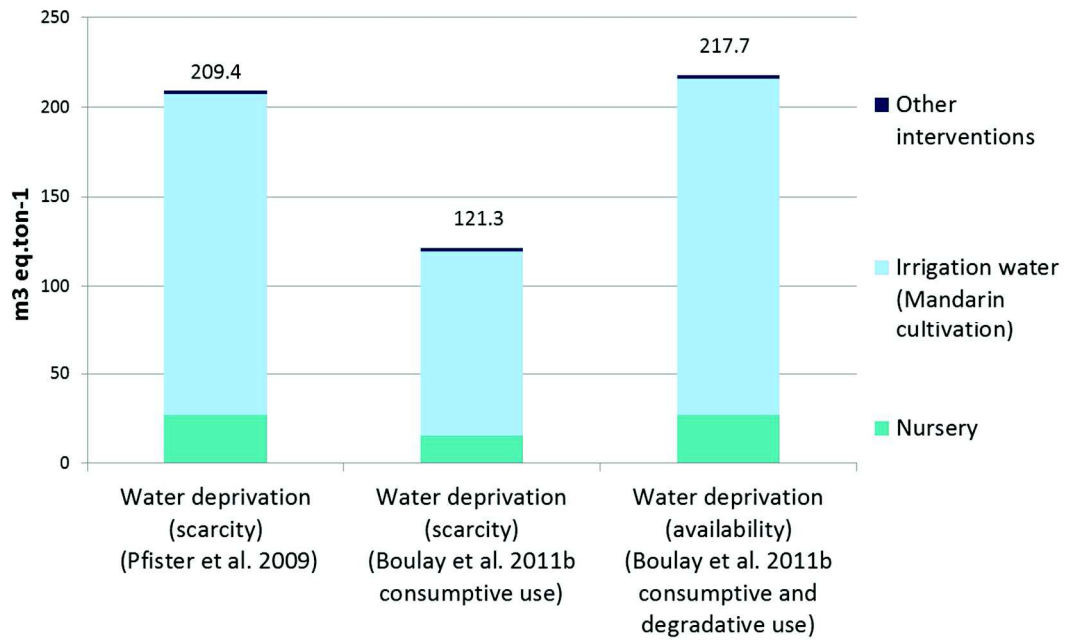


Figure 5.5. Water deprivation impacts of 1kg of Mandarin at farm gate, calculated with the method from Pfister et al. (2009) (water scarcity indicator), and two versions of the method from Boulay et al. (2011b) with and without considering the degradative use of the water (water availability and scarcity indicators).

### 5.3.3. Farm gate - endpoint

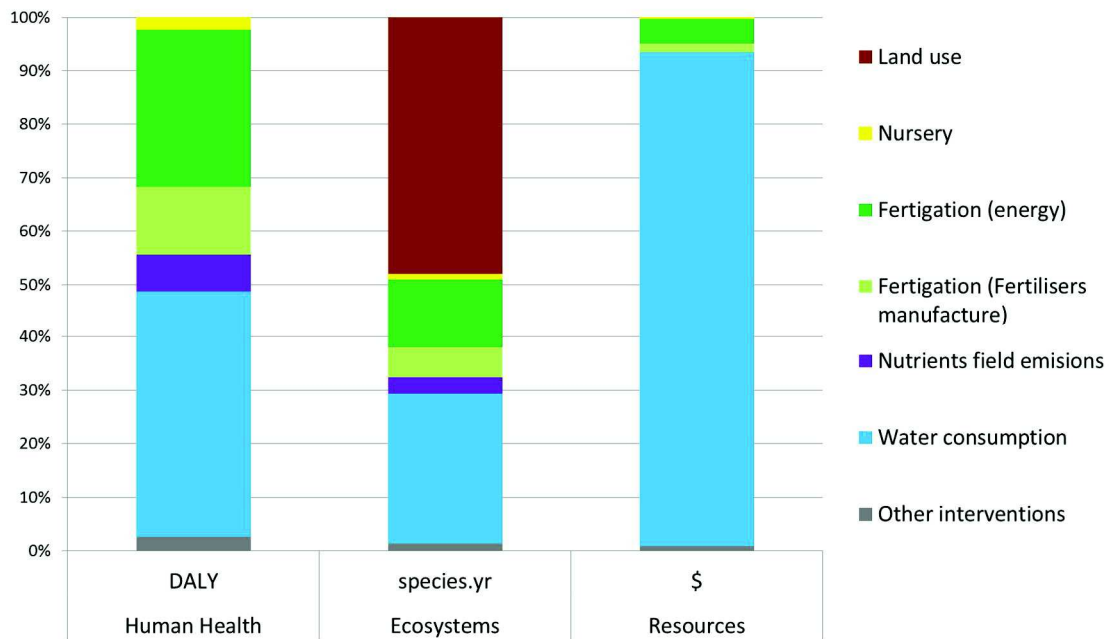


Figure 5.6. Contribution analysis to endpoint damages of 1kg of Mandarin at farm gate in Morocco (ReCiPe Endpoint (H) V1.12). Water consumption damages were assessed with Pfister et al. (2009).

Calculating endpoint damages allows analysing the contribution of water consumption to damages on the three areas of protection when using Pfister et al. (2009). Water consumption was the main

contributor to damages to Resources and Human health, with 92.5% and 46.1%, and the second most important contributor to damages to Ecosystems with 28% after land use (Fig. 5.6).

## 5.4. Discussion

Results showed the great contributions of energy for fertigation, water use, and field emissions to environmental impacts. After a comparison with published references on citrus, we will further analyse and discuss these contributions.

### 5.4.1. Comparison with published references on citrus

Since no water use impacts were calculated in previous LCA studies on citrus, we could not compare water deprivation results with references from the literature. In terms of irrigation water volume, the Clementine from Agribalyse (Basset-Mens et al. 2015) and our Mandarin from Morocco used 286 and 278 m<sup>3</sup>.ton<sup>-1</sup>, respectively, which was approximately 100 m<sup>3</sup>.ton<sup>-1</sup> greater than other citrus grown in Italy and Spain (Sanjuan et al. 2005, Beccali et al. 2010, Lo Giudice et al. 2013). No irrigation was reported for the orange grown in Brazil (Knudsen et al. 2011).

Non-renewable energy consumption was similar for Clementine and Mandarin grown in Morocco (Basset-Mens et al. 2015 and this study) and grown in Italy (Beccali et al. 2010) (around 3.4 MJ<sub>eq</sub>), but lower for the orange from Brazil (Knudsen et al. 2011) (1.26 MJ<sub>eq</sub>) (Fig. 5.7). Eutrophication impacts were similar for the orange grown in Spain (Sanjuan et al. 2005) and for our Mandarin at about 2 g PO<sub>4</sub><sup>3-</sup><sub>eq</sub> per kg citrus (Fig. 5.7). Sanjuan and colleagues (2005) estimated nitrate leaching based on previous measurements of nitrate leaching on citrus orchards from the same region reported in Ramos et al. (2002). For acidification and climate change, results were within the same range for both Moroccan Clementine and Mandarin, but lower for the other citrus. See supplementary information for absolute values for category indicators.

This comparison is limited by the different scopes, objectives and assumptions of each study. In addition, the fruits compared are different (orange, mandarin, small citrus) and present different yield levels and fruit quality (flavour, shape and grade). In particular, the fruit quality for Mandarin and Clementine is very high because they are meant to be eaten raw, whereas oranges from Italy and Brazil have a lower quality because they are destined to be pressed in juice. This raises the question of the relevance of the functional unit kilogram of fruits, and calls for a fruit quality-based functional unit. Another important aspect concerns the scope of the study: this study provided the environmental impacts of a Mandarin grown in a specific farm in Morocco, whereas the study from Basset-Mens et al. (2015) provided the environmental impacts for an average Clementine grown in Morocco.

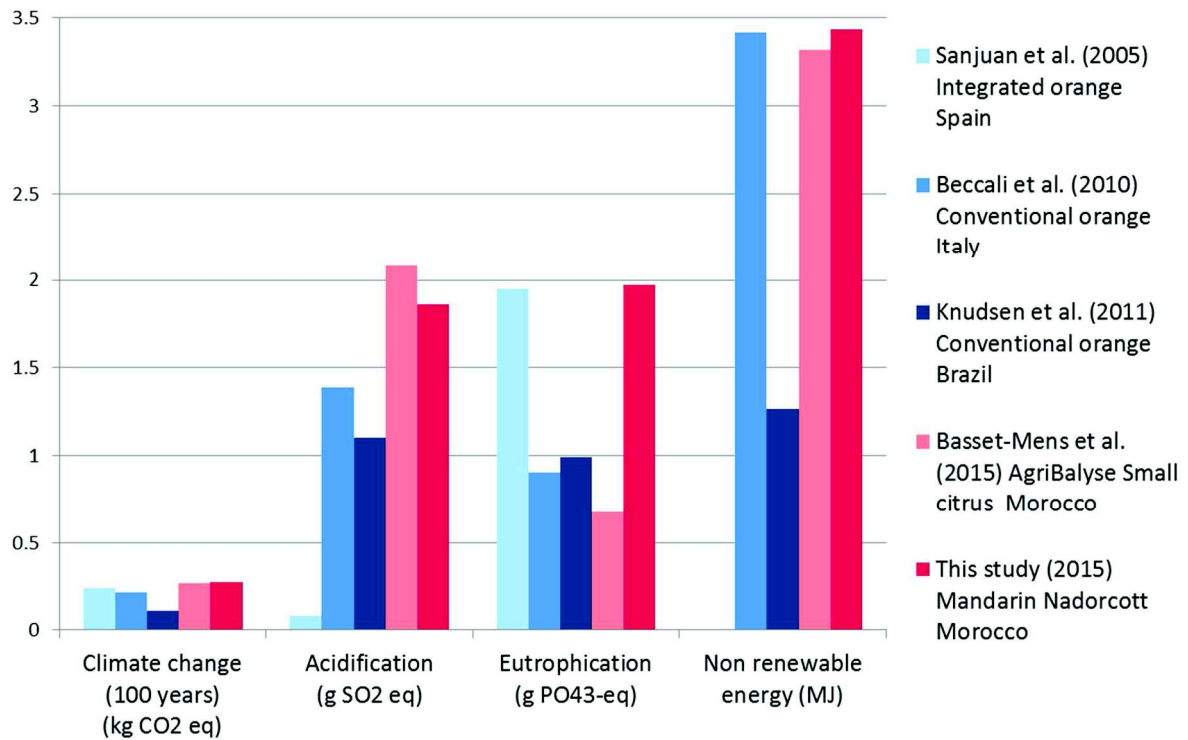


Figure 5.7. Climate change, acidification, eutrophication impacts (calculated with CML (2001)), and non-renewable energy (calculated with CED), from different LCA studies on citrus and for this study. Results are expressed per kg of fruit at farm-gate. Nota: no value available for non-renewable energy for Sanjuan et al. (2005).

#### 5.4.2. Water - energy nexus

The major contribution of electricity to many impacts can be explained by two main reasons. First, Mandarin cultivation requires a lot of water that has to be withdrawn from a deep aquifer, thus requiring the use of electricity-intensive pumps. Second, the Moroccan electricity mix relies for more than 50% on fossil energy. This water-energy nexus is frequent in water-scarce countries, and was already observed for Clementine by Basset-Mens et al. (2015) and by Payen et al. (2015) for tomato. Several government initiatives are currently being developed to solve this double problem. First, renewable energies are being massively developed in the country (e.g.: a 2500 ha solar power plan is being developed nearby Ouarzazate (MASEN - Moroccan Agency for Solar Energy 2010), with the objectives of reaching 42% of renewable energy in the Moroccan electricity mix by 2020. The total energy production capacity of the country is expected to double by 2020, but the share of electricity from coal will remain around 25% (MEMEE 2010). Second, other water resources are explored through colossal projects such as the creation of a “water highway” transporting water from the north to the centre and the south of the country. At farm level, mitigation options can be found in the direct use of renewable energies. The use of pumps powered by solar panel could potentially reduce the impacts on climate change by up to 67.4%. Nevertheless, the manufacturing, maintenance and end-of life of the solar panel should be included in the impact assessment. Using more solar energy seems relevant in a



region where sun is shining throughout the year. However, solar pumps represent an economic investment for which most farmers will need financial support.

It is important to notice that the use of database default value for energy consumption for pumping water, would have underestimated the energy consumption in our Mandarin case study. Withdrawing water at 140 m depth requires on average 0.7 kWh per cubic meter (calculation based on the operating time and specifications of the well-pumps, and the volume of water withdrawn). This value is greater than the default energy consumption proposed in the World Food LCA database (0.239 kWh.m<sup>3</sup>), because it considers a 48 m pumping depth (Nemecek et al. 2015).

### 5.4.3. Water use impacts

#### 5.4.3.1. Applicability

We applied two different methods assessing the impacts related to water use (Pfister et al. 2009; Boulay et al. 2011a&b). To satisfy the water inventory requirements for a perennial crop, we had to develop and use a model of field water flows: estimating the consumed and released water flows according to actual water supply and system specificities (soil-climate-practices). The E.T. model can be used for other crops by adjusting the transpiration module, and within the validity domain described in chapter 4.

In comparison with a LCA study reporting only water volume, the additional efforts for data collection is reasonable since most farmers record irrigation supply volume, at least on a monthly basis. Cross-checking monthly irrigation volumes with irrigation frequency allows to disaggregate data at a daily time-scale. The critical data are water quality data. As highlighted by Boulay et al. (2015), this is the weakest point of methods addressing degradative use of water (Boulay et al. 2011b; Bayart et al. 2014). When water quality analyses are not available, global datasets can be used such as GEMStat, providing surface and ground water quality data over 100 parameters (UNEP 2009), or the Global Nutrient Export from WaterSheds (NEWS), providing phosphorus and nitrogen loads in river basins (Liu et al. 2012). Other data sources may be found at the regional or national scale (e.g.: groundwater nitrate concentrations in France (ONEMA 2014)). Other E.T. model input data (climate, soil...etc) should be preferably primary data, but default values can also be used if necessary (for default data sources refer to Table 3.2 and Table 4.1).

Regarding the impact assessment, since neither the regionalisation, nor the impact assessment method of Boulay et al. (2011b) (consumptive and degradative use) was supported by LCA software, it was necessary to use Simapro and Microsoft Excel complementarily which still remained acceptable in terms of feasibility.

### 5.4.3.2. Relevance

#### **Water deprivation impact results depending on inventory and impact assessment methods**

Since Pfister and colleagues provided both inventory water flows in databases (Pfister et al. 2011; Pfister and Bayer 2014), but also a LCIA method for calculating impacts (Pfister et al. 2009), we will always specify “database” or “method” to avoid any confusion.

We first compared water inventory flows estimates of the E.T. model (with or without salinity stress accounted for), with databases (Pfister et al. 2011; Pfister and Bayer 2014). We focused on the main contributor to water deprivation impacts: the Mandarin cultivation stage. Results showed that depending on the inventory method used, the evapotranspiration blue water (water consumed from irrigation water) varies a lot (Fig. 5.8). The lowest consumption estimate is from Pfister and Bayer database (2014) with  $149 \text{ m}^3.\text{ton}^{-1}$ , and the higher estimate is from Pfister database (2011) with  $237 \text{ m}^3.\text{ton}^{-1}$ , whereas our best estimate with the E.T. model (accounting for salinity and water stresses) is  $181 \text{ m}^3.\text{ton}^{-1}$ .

Then, we compared water deprivation impact results based on different inventory methods. Estimating the released water flows with the E.T. model allows the application of Boulay’s method, which is not possible with Pfister databases (since it is not providing the withdrawn and release water), unless assuming a rough estimate of water withdrawal and released (based on irrigation efficiency like in water inventory database). Thus, we can assess the effect of using either databases or E.T. model outputs only on impacts calculated with the Pfister’s method (Pfister et al. 2009). Since Pfister’s characterisation factors are maximum for the cultivation stage in Morocco (equal to 1), the water deprivation results are equal to their corresponding water inventory flows (Fig. 5.8). Results based on the E.T model inventory are within the extreme values proposed in Pfister database. When the inventory is based on the E.T. model version not accounting for salinity stress, impacts results are greater. As discussed in chapter 4, since evapotranspiration might be underestimated at high salinity levels, we recommend estimating crop evapotranspiration without accounting for salinity stress, and consider it as an upper value (in this case, the deep percolating water flow is lower, and should be considered as a minimum).

Finally, we compared water deprivation impacts using the E.T. model for the inventory for different water Life Cycle Impact Assessment (LCIA) methods. Surprisingly, the water deprivation impacts estimated with Pfister et al. (2009) was 74% higher than estimated with Boulay et al. (2011b) (consumptive). Although they are both water scarcity indicators (considering only consumptive water use), the characterisation factor (CF) is maximum for Pfister et al. (2009) at 1, whereas it is 0.565 for Boulay et al. (2011b). Both CFs are based on water consumption and availability data taken from the WaterGap 2.2 model (Alcamo et al. 2003), and are using similar low and upper scarcity thresholds and logistic function (S-curve) (Boulay et al. 2015). But several reasons can explain this difference: Boulay’s CF is specific to groundwater and based on a ratio of water consumption to renewable groundwater available, whereas Pfister’s CF is based on a ratio of water withdrawal to total water available. The difference between scarcity indicators specific to surface water and ground water (Boulay et al. 2011b), versus a generic scarcity indicator (such as Pfister et al. 2009) is high in Morocco (between 10 to 50% difference) as demonstrated by Boulay et al. (2015). It is thus relevant to use the ground-water specific

CF from Boulay et al. (2011b) in Morocco, which is lower than the surface-water specific CF equal to 1 from Pfister et al. (2011), because surface water scarcity is much higher than ground water scarcity. Nevertheless, the reliability of ground-water specific CF is questionable since data on groundwater resources do not have a sufficient quality in hydrological model (on which the characterisation factors are based) (Boulay et al. 2015). Indeed, we know from the Water Basin Agency that the pressure on the groundwater resource in this area is high (ABHT 2007), but no quantitative and accurate data on the groundwater resource availability and increasing pressure is available. Thus, Boulay’s scarcity CF for ground-water is probably too low (0.565), and, surprisingly, corresponds to the availability CF for the worst ground water quality (category G5 “unusable”).

When considering both consumptive and degradative water use (Boulay et al. 2011a&b), water availability impacts results are similar to water scarcity impacts results from Pfister et al. (2009) since the impact due to degradative water use from Boulay’s method increases the pressure on the water resource.

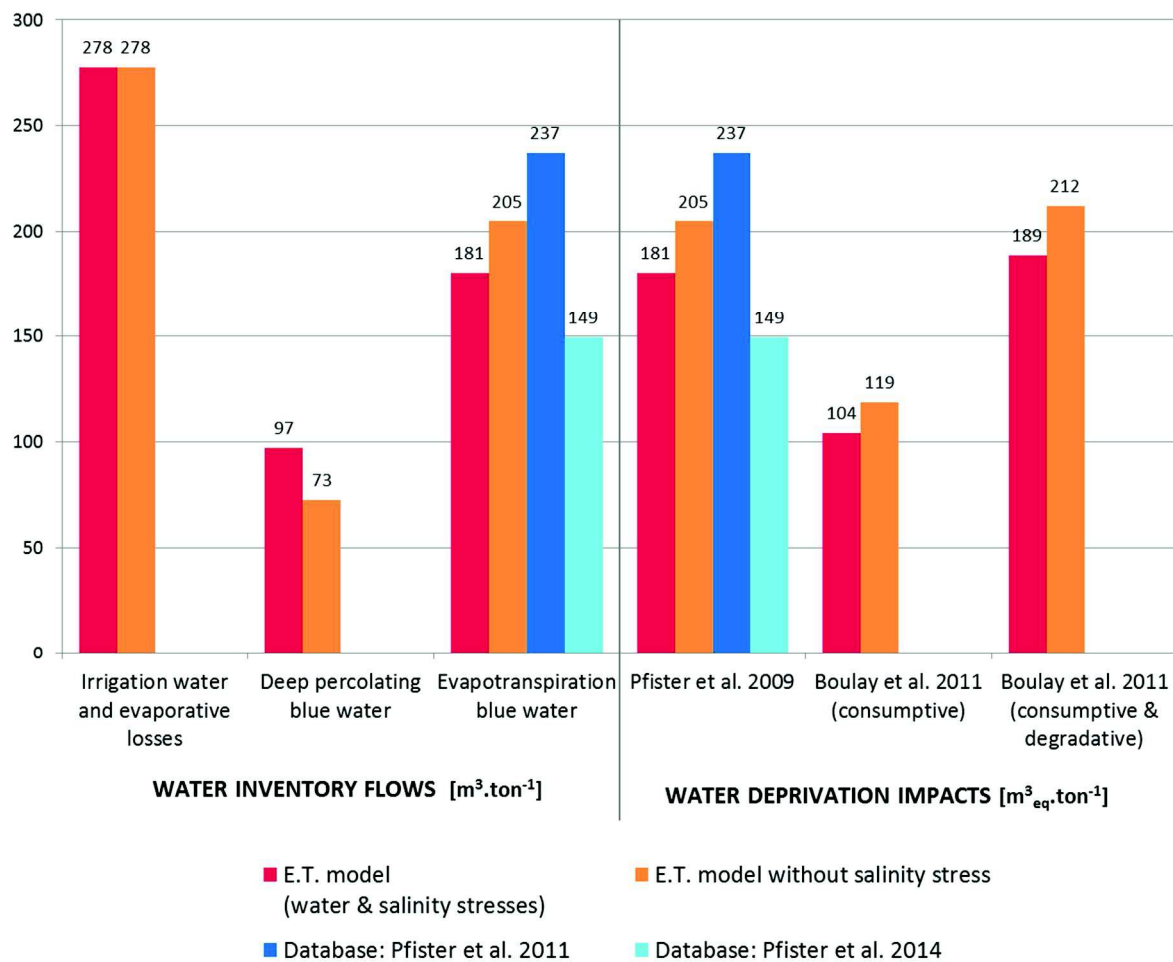


Figure 5.8. Water inventory flows and water deprivation impacts of Mandarin cultivation (nursery not included). Water inventory flows (in m<sup>3</sup>.ton<sup>-1</sup>) are estimated with E.T. model or from database, and water deprivation impacts (in m<sup>3</sup><sub>eq</sub>.ton<sup>-1</sup>) associated with these water flows are calculated with different assessment methods. Results are expressed per tonne of export fruits (allocation included), even for databases.

The most scientifically-sound result for water deprivation impacts is the one based on water flows estimated with E.T. model (accounting for stresses), and characterised with Boulay et al. (2011a and b), accounting for both consumptive and degradative water use. Nevertheless, as explained above, the upper value assuming no salinity stress should also be provided. As a result, the water deprivation impact result should be given as a range: 189-212 m<sup>3</sup><sub>eq</sub>.ton<sup>-1</sup>.

### **LCIA methods limitations**

In our application of Boulay and colleagues' (2011b) method, the impact from rainfall use was limited to the consumptive use of rainfall (Fig. 5.2). Nevertheless, this assessment does not consider that rainfall use can be associated with water quality degradation. However, the cropping system is not only consuming rainfall water (through evapotranspiration), but also contributing to its degradation because rain water is mixed with salts, pesticides and nutrients in the soil. As a result, the part of rainfall not being consumed is not released in the environment with the same quality but with salts (average of 1.78 g.L<sup>-1</sup>) and nitrates (average of 175.7 mg.L<sup>-1</sup>). Accounting for rainwater consumption and quality degradation is possible according to Boulay et al. (2011b), but the methodology is complex and relies on the critical definition of a "reference natural state". Indeed, we cannot consider the actual runoff water (RO<sub>green</sub>) and deep percolating water (DP<sub>green</sub>) originating from rainfall (estimated with the E.T. model) as elementary water flows: only the change in the released water flows due to the presence of the crop compared to the so-called reference natural state should be considered (Fig 5.9). This is equivalent to quantify the surplus water evapotranspirated by the crop, in comparison with the evapotranspiration by the reference state if the crop was not in place. The question is how to define the reference vegetation state, and the corresponding evapotranspiration? The second difficulty is to translate the rainfall partitioning between runoff and infiltration in terms of equivalent surface and ground waters respectively. This can be easily done thanks to the E.T. model providing RO and DP water fluxes. Nevertheless, this runoff and infiltration partitioning depends on the soil water stock, which strongly depends on irrigation water supply. The characterisation factors associated should correspond to the best available surface and ground water in the region (Fig 5.9).

In brief, Boulay and colleagues' (2011b) method has the advantage of proposing a method accounting for rainfall water use impacts. However, its complexity and limitations make it very difficult and not so relevant to apply. Therefore, this rainfall and irrigation water partitioning approach constitutes as already said an important drawback in the assessment of water use impacts.

The application of other water impact assessment methods should be the object of future work, to further investigate the effect of calculating water impact at a monthly scale with (Pfister and Bayer 2014), and to test the new WULCA consensual scarcity indicator (WULCA 2015).

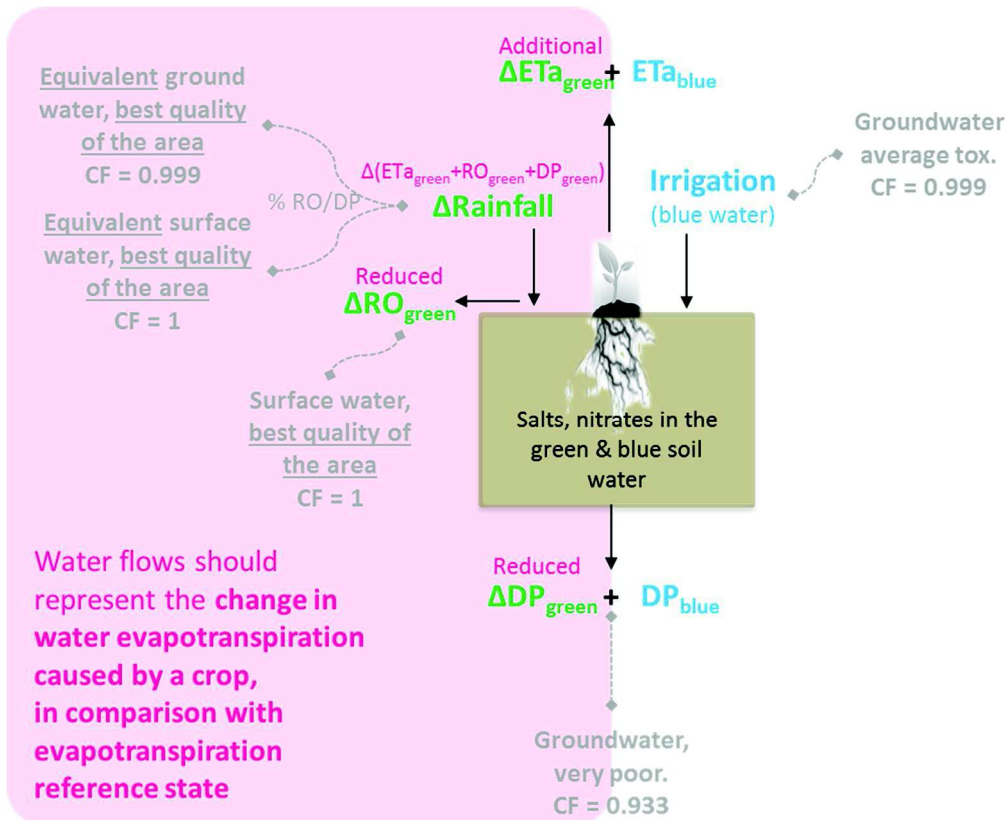


Figure 5.9. Representation of elementary water flows and associated characterisation factors (CF) of Boulay et al. (2011b) method when addressing the impacts from rainfall water consumptive and degradative use. Rainfall water, evapotranspiration from rainfall ( $\Delta ETa_{green}$ ), and rainfall water released in the environment through runoff ( $\Delta RO_{green}$ ) and deep percolation ( $\Delta DP_{green}$  and  $\Delta DP_{blue}$ ) should be quantified as the difference with the reference state.

### Water impacts contribution in comparison with other impacts

The endpoint assessment showed the major contribution of water damages to Resources and Human health. Nevertheless, the relative importance of water use impacts compared to other impact category results can only be assessed with Pfister et al. (2011) method which is the only method proposing endpoint damage assessment on the three areas of protections. The contribution of water impact to damages on Human health alone, could still be further investigate with other methods (Boulay et al. 2011b; Motoshita et al. 2014) in future work.

#### 5.4.4. Perspectives

In addition to salts, water is a vector of pesticides and nutrients. Improving the modelling of field water flows implies improving the field emission depending on it: in particular nitrogen and pesticides responsible for eutrophication and toxicity impacts.

#### 5.4.4.1. Nitrate leaching

In our Mandarin case study, nitrate leaching estimates varied between 38.7% and 64.3% of total nitrogen applied throughout the years, and was the main contributor to eutrophication impacts. This nitrogen leaching rate was much higher than the default 10% of nitrogen applied reaching water bodies, provided by the Water Footprint Network (Hoekstra et al. 2011). In this study, nitrate leaching estimate was based on an annual nitrogen balance and the annual deep percolating water estimated with the E.T. model (details in Supplementary information). But ideally, the daily water balance should have been coupled with a daily nitrogen balance. Considering a nitrogen budget at an annual scale is a limitation of this approach, leading to a possible overestimation of nitrate leaching because we considered that all nitrate leachable annually can potentially leach (Pervanchon et al. 2005). Nevertheless, the high average nitrate leaching of 134 kg N-NO<sub>3</sub>N.ha<sup>-1</sup> for a nitrogen application of 235.4 kg N-NO<sub>3</sub>N.ha<sup>-1</sup> (emission fraction of 57%) is consistent with the high annual nitrate leaching measurements made on a Mandarin orchard under drip irrigation in Spain of 188 kg N-NO<sub>3</sub>N.ha<sup>-1</sup> for an application rate of 210 kg N.ha<sup>-1</sup> (emission fraction of 90%) (Castel et al. 1996). This result is contrasting with the zero nitrate leaching emission factor of IPCC Tier1 for drip-irrigated crops in semi-arid climate. This default assumption is clearly not appropriate for crop requiring high volumes of water, especially in a context where salts leaching is necessary. In the absence of appropriate model in 2010 and as part of the AgriBalyse® program, the IPCC zero nitrate emission factor was used for the Clementine from Morocco (Basset-Mens et al. 2015). As a result, eutrophication impacts from the Mandarin (this study) are 2.8 times greater than the representative Moroccan Clementine (Table 5.4).

**Table 5.4. Cradle-to-farm-gate LCA results per kg of fruit for a selection of environmental indicators (ReCiPe Midpoint (H); Cumulative Energy Demand) for the Clementine and Mandarin grown in Morocco**

Reference	Product	Climate change (kg CO <sub>2</sub> eq)	Acidification (g SO <sub>2</sub> eq)	Eutrophication (g PO <sub>4</sub> <sup>3-</sup> eq)	Non renewable energy (MJ)	Human toxicity (kg 1,4-DBeq)	Aquatic freshwater ecotoxicity (kg 1,4-DBeq)	Terrest. eco-toxicity (kg 1,4-DBeq)	Fossil depletion (kg oil-eq)
This study (2015)	Mandarin Nadorcott	0.274	1.859	1.673	3.44	0.0093	0.00007	0.00006	0.0623
<b>Basset-Mens et al. (2015)</b>	Small citrus	0.269	2.27	0.437	3.32	0.0783	0.00616	0.00699	0.0667

Reducing the nitrogen application rate could be a mitigation option to reduce eutrophication impact. The nitrogen application is high in comparison with other studies (Table 5.5). Nevertheless, we cannot recommend applying less nitrogen fertilizers based on our model simulations which do not capture the interactions between crop, salinity and nutrients. Yet, the nutrient supply to the crop is complex and should integrate that the crop has to cope with salinity stress. Future research is warranted on these complex interactions.

**Table 5.5. Nutrient application rates for the Mandarin grown in Morocco (this study) and published citrus LCA studies**

References	Product	Country	Reference period	N kgN.ha <sup>-1</sup>	P205 kgP <sub>2</sub> O <sub>5</sub> .ha <sup>-1</sup>	K20 kgK <sub>2</sub> O.ha <sup>-1</sup>
Sanjuan et al. (2005)	Integrated orange	Spain	Adult orchard	300	65	135
Beccali et al. (2010)	Conventional orange	Italy	Average productive year	137	57	103
Knudsen et al. (2011)	Conventional orange	Brazil	Adult orchard	111	53	102
Basset et al. (2015)	Small citrus	Morocco	Adult orchard (9-25 years)	180	45	180
This study (2015)	Mandarin Nadorcott	Morocco	Adult orchard (9-25 years)	235	47	183

#### 5.4.4.2. Pesticide emissions

Pesticide emissions were a major contributor to ecotoxicity impacts. Pesticide emission modelling was based on a conservative assumption: 100% of pesticides were considered emitted in the soil. In the absence of surface water bodies in the area and owing to the depth of aquifer, we can assume that active substance would degrade before reaching water.

The (eco)toxicity impact assessment is incomplete since promising new molecules are not characterised yet. Indeed, characterisation factors are missing for several active substances: Imidaclopride (controlling Mediterranean fruit fly, *Ceratitis capitata*), Spirotetramat (controlling California red scale) and Métaldéhyde (controlling snail), thus ecotoxic impacts are underestimated. When comparing with the Clementine from Morocco, based on an antecedent reference period (2000-2010) (Basset-Mens et al. 2015), (eco)toxicity impacts are much lower for the Mandarin (Table 5.5). This can be explained by the recent change in several active substances and the absence of characterisation factors for new ones. Ecotoxicity impacts of the Clementine were essentially due to the use of Chlorpyrifos-ethyl for controlling California red scale, and Methomyl for controlling citrus leaf miner. Methomyl now has a restricted use (not authorized in France, but authorized in Morocco), and Imidacloprid is used instead in the studied farm. The California red scale is one of the most feared pest in Citrus orchard, and a new active substance Spirotetramat is now used (since 2013) notably because its application requires less work than Chlorpyrifos-ethyl. The toxicity reported for Spirotetramat is lower than for the Chlorpyrifos-ethyl (e-phyl 2015), thus we can expect a lower contribution to impacts.

To improve the toxicity impact assessment of this LCA study in the future, we should apply the upcoming version of the PestLCI model as updated through the on-going pesticide consensus building workshop to determine pesticides partitioning, and use the USEtox method completed with the characterisation factors of recent active substances (Rosenbaum et al. 2008; Henderson et al. 2011; Dijkman et al. 2012).

#### 5.4.4.3. Next steps

In future work, it would be interesting to analyse the sensitivity of LCA results associated with the input variables of the E.T. model, based on the variation ranges and expertise from the producer. Furthermore, the simulation of different projection scenarios, the simulation of a replacement of electricity pump with solar pumps, and the comparison with other Nadorcott farms would be relevant

for the farmer. Other water impact assessment methods could be tested, including the very recent AWaRe indicator which is based on the unused water in a basin rather than on a ratio of demand to availability (WULCA 2015). Finally, water damages assessment should be performed with different endpoint methods to analyse if the relative contribution of water remains large whatever the method.

## 5.5. Conclusion

This work presented the LCA of a Mandarin crop grown in Morocco with an in depth assessment of water use impacts. The E.T. model, estimating field water and salts flows (Chapter 4) was used in a full cradle-to-market-gate LCA study to test its relevance and feasibility in the assessment of water use impacts.

The 25-year perennial crop cycle was modelled from nursery to orchard end-of-life, based on primary data and a projection scenario for the future. The life cycle of the Mandarin was modelled up to the French market gate, accounting for post-farm production stages and transportation from Morocco to France. For most impact categories, the contribution analysis showed the major contribution of energy required for pumping water in the aquifer, revealing a water-energy nexus. The E.T model outputs allowed the assessment of consumptive and degradative water use impacts, which was not possible with current water databases.

Water use impacts were different depending on the impact assessment method used. The water availability impact calculated with Boulay et al. (2011a and b) with and without accounting for a salinity stress for field water flows was between 189 (with) and 212 (without)  $\text{m}^3_{\text{eq}}$  per ton Mandarin for the cultivation stage. An additional 29.9  $\text{m}^3_{\text{eq}}$  per ton should be added when accounting for the nursery, packaging and transportation stages. The water impact result was provided as a range (with and without salinity stress) to account for the uncertainty associated with the E.T. model that may underestimate water evapotranspiration at high salinity level. Water availability indicator results from Boulay et al. (2011a&b) addressing both consumptive and degradative use) were close to water scarcity indicator estimated with Pfister et al. (2009): 181  $\text{m}^3.\text{ton}^{-1}$  and 189  $\text{m}^3.\text{ton}^{-1}$ , respectively. However, the quality degradation of deep percolating water originating from rainfall was not accounted for in our implementation of the Boulay's method because it relies on the estimation of the evapotranspiration of a "reference state" if the crop was not in place. The rainfall and irrigation water partitioning (so-called green and blue waters) is arbitrary and fails to properly represent the water cycle. It constitutes an important drawback in the assessment of water use impacts. Characterisation factors specific to the water source (surface or groundwater) are very relevant in Morocco, but their quality is hampered by the lack of good quality data on groundwater resource state in the global hydrological models they use. Water use impact assessment showed limitations in terms of applicability: regionalisation is not supported by software, and the estimation of the quality of the released water flows requires data on the quality of irrigation water. The quality of input water can be retrieved from databases if water analyses are not available, and the volume and salinity of the deep percolating and runoff waters can be estimated thanks to the E.T. model.



This study should be completed with a comparative LCA with other Mandarin farms, the application of other water use impact assessment methods at both midpoint and endpoint levels, and an uncertainty assessment. Finally, the daily field water flows provided by the E.T. model opens new possibilities for an improved assessment of other field emissions such as nitrate and pesticides.

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## References Chapter 5

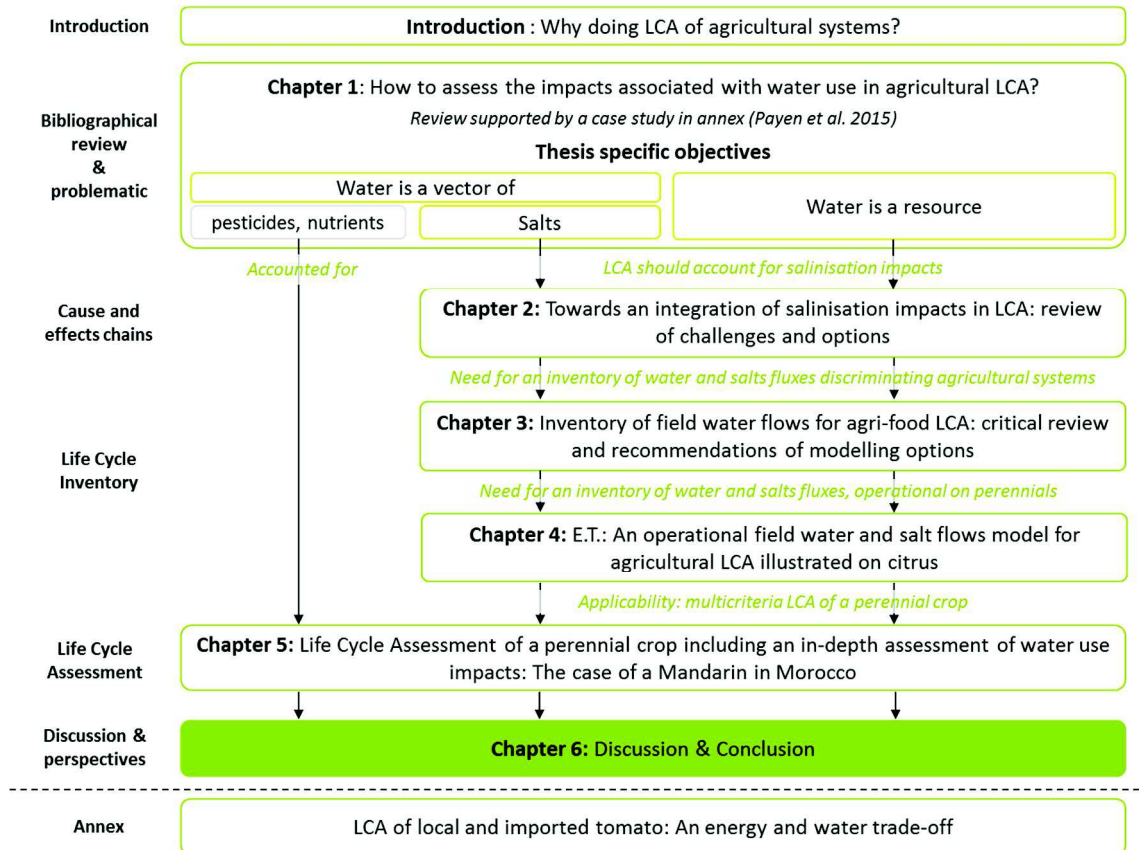
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## Discussion and perspectives (in French)

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La discussion est organisée autour des questions de la thèse énoncés en fin de chapitre 1 :

- Comment mieux évaluer les impacts associés aux flux d'eau et de sels ?
- Comment réaliser un inventaire pertinent des flux d'eau et de sels mobilisés dans les systèmes agricoles ?
- Est-il possible d'appliquer le modèle d'inventaire des flux d'eau et les indicateurs associés pour évaluer des pratiques agricoles ?

### Comment mieux évaluer les impacts associés aux flux d'eau et de sels ?

#### Une synthèse bibliographique de plus sur l'eau ?

La synthèse bibliographique présentée en chapitre 1 capitalise sur des synthèses existantes. Elle a néanmoins la particularité de mettre l'accent sur le double statut de l'eau (à la fois ressource et milieu de vie), ce qui se traduit au niveau de l'évaluation d'impacts par des catégories distinctes : les impacts liés à des émissions de polluants vers l'eau (eutrophisation, acidification, écotoxicité) et les impacts liés à l'usage de l'eau (usages consommateurs et dégradants de l'eau). Le double statut de l'eau est donc illustré tant au niveau des impacts et dommages (avec les faibles risques de doubles comptages évoqués pour la santé humaine), qu'au niveau de l'inventaire : on caractérise l'eau en termes de volume et de qualité. Pour être exhaustif, l'inventaire des flux d'eau doit donc refléter le fait que l'eau est un vecteur de polluants en prenant en compte sa qualité. La synthèse bibliographique inclut également un effort de clarification des termes relatifs aux « empreintes eau ». Il y a eu en effet un foisonnement de terminologies, à la fois contradictoires et redondantes, ce qui a malheureusement apporté de la confusion (Perry 2014). Heureusement, la récente norme ISO 14046 (ISO 14046, 2014 ) va permettre d'harmoniser l'usage des termes et le chapitre 1 est une contribution à cette harmonisation.

#### La salinisation : une catégorie d'impact qui cristallise beaucoup de challenges de l'ACV

Le chapitre 2 (Article en révisions majeures pour Int. J. LCA en octobre 2015), décrit les bases de la construction d'un cadre de prise en compte des impacts liés à la salinisation. Cette catégorie d'impact a probablement été négligée à cause de la complexité et de la diversité des mécanismes environnementaux de la salinisation, qui ont lieu à des échelles différentes et emboîtées, mais aussi parce que la salinisation touche aussi bien les eaux que les sols, qui ont tous deux ce double statut de « ressource » et de « milieu de vie ». Dans la perspective d'un cadre de prise en compte exhaustif (tous les contextes de salinisation) et cohérent (pas de double comptage ou d'oubli), une clarification était donc nécessaire sur les cadres de modélisation ACV possibles : le cadre associé à un usage ou changement d'usage des terres (« Land use/Land use change » en anglais) versus le cadre associé à une émission vers l'environnement. Le cadre « usage ou changement d'usage des terres » souffre d'un manque d'harmonisation entre les méthodes et ne permet pas (pour l'instant) d'évaluer les impacts ayant lieu en aval du lieu de l'intervention. Ceci est particulièrement problématique lorsqu'on s'intéresse à des impacts dont l'eau est le vecteur déterminant (exemple de la salinisation dans les bas-fonds, due à

un changement d'usage des terres). L'article fournit des arguments en faveur d'une modélisation décrivant des mécanismes environnementaux du type : devenir des sels, exposition de la cible environnementale aux sels, et effet de la salinité sur la cible environnementale. Ce cadre classique en ACV de devenir/exposition/effet est pertinent car : (i) les sels sont mobiles dans l'environnement et leurs effets sont interconnectés, (ii) cela permet d'évaluer les impacts ayant lieu sur le lieu d'intervention et au-delà, (iii) cela repose sur un inventaire qui permet une meilleure discrimination des systèmes (plutôt qu'une « simple » catégorie d'usage des terres), nécessaire dans une démarche d'éco-conception des systèmes de culture.

L'évaluation des impacts liés à la salinisation illustre donc bien les défis méthodologiques auxquels l'ACV doit faire face, ce qui en fait un « cas d'école ». Cette thèse apporte une contribution méthodologique en décrivant les points de blocages ACV qui nécessitent discussion et consensus. En particulier, le statut des aires de protection et la frontière entre technosphère et écosphère. Statuer -ou du moins débattre- de ces aspects critiques pourrait être l'objet d'ateliers de travail comme le premier atelier sur la qualité des sols en ACV qui a eu lieu récemment (organisé le 30 août 2015 à Bordeaux par le CSIRO, l'ADEME et le CIRAD). Ces lieux d'échanges privilégiés entre experts ACV doivent permettre de débattre de façon constructive pour idéalement aboutir à un consensus comme c'est le cas pour la modélisation de l'émission de pesticides (Rosenbaum et al. 2015). Néanmoins, il est essentiel de travailler conjointement avec des experts des disciplines concernées. Pour aboutir à la revue présentée en chapitre 2 des types de salinisation et l'analyse critique des méthodes existantes, agronomes experts en salinisation, pédologues et chercheurs ACV ont joint leurs connaissances dans le cadre d'un groupe de travail qui s'est réuni à plusieurs reprises sur une année. Ce mode de travail très courant en ACV passant par l'appropriation d'une nouvelle discipline, son éventuelle simplification (e.g. typologie des salinisations) et en tout cas son adaptation au cadre conceptuel de l'ACV permet également l'identification de fronts de science (e.g. modélisation des interactions engrais-salinité, ou modalités d'irrigation-salinisation, ou encore le manque de données du niveau de salinité des sols et des eaux à une échelle globale).

Cette interaction et ouverture au-delà de la sphère ACV est un challenge puisqu'elle requiert un réel échange de connaissances, parfois contraint/ralenti par des terminologies ou jargons spécifiques (voire contradictoire dans le cas de l'eau en ACV). Mais c'est un des apports de la thèse que de participer à la diffusion et à la formation à l'ACV auprès de nos pairs de disciplines connexes (hydrologie, agronomie).

Le cadre général proposé sur la salinisation pourrait représenter une base pour un projet de thèse ou un post-doc dédié à la production de facteurs de caractérisation idéalement spatialisés en mobilisant les SIG. Une couverture globale sera cependant confrontée à la grande hétérogénéité des données disponibles sur le niveau de salinité des sols et des eaux dans le monde. Une étude préliminaire en collaboration avec des chercheurs travaillant à l'acquisition de données sur les niveaux de salinité (hydrologues, hydrogéologues, agronomes) sera nécessaire pour évaluer la faisabilité du travail à une échelle globale. La seconde limitation à la production de facteurs de caractérisation est qu'il n'existe pas de modèle de flux de sels à l'échelle du bassin versant (Ragab R., comm. pers). Cela passera donc par une modélisation des flux d'eau, couplé à un bilan salin.

De plus, une meilleure évaluation des impacts associés aux flux d'eau et de sels doit passer par un travail plus abouti d'affectation des dommages à chaque aire de protection mais aussi par un effort de



consensus sur la définition de la frontière entre l'inventaire et l'évaluation d'impact (la frontière techno/écosphère). C'est ce que nous analysons plus en détail ci-dessous.

### Le statut des aires de protection

Le statut des aires de protection structure les chaînes de cause à effet (ou « pathways » en anglais) modélisés en ACV. Une définition non consensuelle constitue donc un frein au développement de ces chaînes sans risque de double comptage (Hauschild et al. 2013). C'est en particulier le statut de l'aire de protection Ressource qui fait débat. Il est nécessaire de définir si l'on considère les Ressources comme un « objet à protéger » intermédiaire qui n'a de fonction que pour les Ecosystèmes et la santé humaine, ou bien si on lui confère une valeur intrinsèque au même niveau que les autres aires de protection. C'est notamment à cause de cette ambiguïté que les chaînes de cause à effet de l'usage de l'eau sur les Ressources ont reçu aussi peu d'attention. La nécessité de clarifier la définition des aires de protection va bien au-delà des impacts liés à l'eau. Le problème est identique pour l'usage des terres où les impacts sont tantôt liés à l'aire de protection Ressource (Milà i Canals et al. 2007), l'aire de protection Ecosystèmes (Goedkoop et al. 2013) ou encore aux services écosystémiques, intermédiaires aux trois aires de protection (Koellner et al. 2013). Dans la définition du cadre de modélisation des impacts salinisation (chapitre 2), nous proposons de considérer que l'aire de protection Ressource représente la protection de ressource en quantité et qualité suffisante pour les générations futures, alors que les aires de protection Santé Humaine et Ecosystèmes représentent la protection des personnes et des écosystèmes actuels. L'arrivée imminente de la nouvelle méthode d'évaluation d'impacts IMPACT World<sup>+</sup> participera à ce débat en apportant une définition intéressante de l'aire de protection Ressource. Cette aire de protection est en effet définie comme l'ensemble *des impacts sur la société humaine sans conséquences directes sur la santé*, tels que l'utilisation de ressources abiotiques et la dépréciation de services écosystémiques (IMPACT World<sup>+</sup> 2015).

Enfin, une définition claire et exhaustive des aires de protection ne peut pas ignorer les liens avec l'Analyse Sociale du Cycle de Vie (ASCV) et l'Analyse du Coût du Cycle de Vie (ACCV). Car si l'on souhaite aboutir à un cadre cohérent pour l'évaluation de la durabilité, il est important d'analyser si les aires de protection doivent être intégrées pour ses trois piliers (économique, social et environnemental) (Dreyer et al. 2006; Kloepffer 2008). Sonnemann et al. (2015) considèrent que la seule façon de définir et évaluer correctement l'aire de protection Ressource et d'intégrer le concept de criticité des ressources (dans lequel les ressources en eau et en terres sont considérées au même titre que les ressources minérales) dans un cadre d'Analyse de la Durabilité du Cycle de Vie. Un des arguments est que les dimensions sociaux-économiques et géopolitiques liés aux ressources naturelles sont essentielles (Sonnemann et al. 2015). L'aire de protection Santé Humaine est elle aussi intimement liée aux dimensions à la fois économiques et environnementale. Une intégration des cadres de l'Analyse Sociale et Environnementale du Cycle de Vie serait donc pertinente mais requiert d'identifier les risques de double comptage entre Santé Humaine et Bien-être Humain (Dreyer et al. 2006).

### Le statut du sol en ACV

Le statut du sol en ACV doit être défini pour permettre l'analyse de systèmes agricoles : comment prendre en compte le fait que le sol doit être préservé, mais que c'est aussi un outil de production ?

Cette question n'est pas résolue pour bon nombre de flux (pesticides, nutriments, sels) et donc d'impacts. Dans le cadre de la réflexion sur la prise en compte de la salinisation, nous proposons de considérer qu'une partie du sol (zone d'action du producteur) est incluse dans la technosphère, et ce, uniquement le temps de son utilisation (cycle de culture). Ceci permet de modéliser les pratiques agricoles dans l'inventaire (appuyé par des modèles agronomiques, de bilan hydrique ou autre), et de prendre en compte le changement de qualité du sol entre le début et la fin de son utilisation. Cette proposition est valable pour la salinisation mais aussi pour toute autre catégorie d'impact. Il y a un besoin de clarification et de mise en cohérence de tous les flux d'inventaire, et de mise en cohérence avec les modèles d'évaluation d'impact. La thèse apporte donc une contribution à cette discussion en étudiant l'eau, le vecteur commun de très nombreux flux, et les sels, qui constituent l'un de ces flux.

## **Comment réaliser un inventaire pertinent des flux d'eau et de sels mobilisés dans les systèmes agricoles ?**

### **La recherche d'opérationnalisation**

Un des mandats du Cirad, organisme de recherche agronomique pour le développement, est de participer au développement durable des régions tropicales et méditerranéennes. L'usage de l'ACV comme outil d'évaluation de cette durabilité et d'aide à la décision est un atout grâce à sa vision holistique, à son caractère intégrateur. Néanmoins, il implique une recherche d'opérationnalité dans des contextes très contraints notamment en données : un compromis entre faisabilité (imposée par la disponibilité des données d'inventaire et modèles d'impact appropriés) et fiabilité des résultats. Lorsque l'étude ACV de la tomate produite au Maroc a révélé l'importance des impacts liés à l'usage de l'eau, et démontré que ces impacts étaient probablement sous-estimés, le besoin d'amélioration de l'inventaire des flux d'eau au champ et l'évaluation des impacts salinisation s'est imposé. L'enjeu consiste à associer précision dans la discrimination des pratiques agricoles qui ont un impact (lié à l'objectif de développement et l'engagement auprès du partenaire de terrain), et faisabilité pour les cas d'études à venir (objectif promu par l'ADEME, l'Agence de la maîtrise de l'énergie et de la maîtrise des déchets). Le modèle E.T. apporte une réponse à ce besoin et à cet enjeu, avec l'atout d'être opérationnel et applicable sur toutes cultures y compris les cultures pérennes.

### **Modélisation sur-mesure**

En l'absence de modèle de flux d'eau et de sels simple et applicable pour les cultures pérennes (Chapitre 3), ce travail de thèse a développé un modèle « sur mesure », une approche efficace en agronomie (Affholder et al. 2012). Cette modélisation capitalise sur des formalismes anciens et éprouvés dont la validité est vaste (e.g. Allen et al. (1998); USDA (2012)), se débarrasse des formalismes inutiles à l'objectif (modélisation du rendement), et utilise des formalismes plus récents pour raffiner certains modules cruciaux pour notre objectif (estimation de la transpiration). Le développement du modèle sous Microsoft Excel vise à favoriser sa transparence et sa flexibilité. L'objectif étant que : (i) l'usage de ce modèle ne requiert pas de compétences supplémentaires à celle requises pour la réalisation d'une ACV,

(ii) ce modèle puisse évoluer en termes de complétude et de prise en compte des spécificités de la transpiration de la culture.

Ce modèle devra être confronté avec des mesures de terrain car cela n'a pas été possible dans le cadre de la thèse. On pourra pour cela se baser sur un jeu de donnée existant issus d'un dispositif expérimental mesurant la transpiration des arbres par flux de sève (« sap flow » en anglais) et comparer avec les estimations de transpiration fournies par le modèle.

### Utilisation de modèles en ACV

Ce n'est pas la première fois que l'on fait appel à des modèles de flux au champ pour les inventaires d'ACV agricoles. Des exemples sont : l'usage de modèle d'émission azoté pour l'estimation des flux d'ammoniaque ( $\text{NH}_3$ ) et de protoxyde d'azote ( $\text{N}_2\text{O}$ ) suite à l'épandage de lisier (Langevin et al. 2015), l'usage d'un modèle de culture et d'émission azotée pour l'estimation des flux d'azote et du rendement sur culture de tomate au Bénin (Perrin 2013), ou encore l'usage d'un modèle de culture pour simuler la consommation d'eau, le rendement et le taux de sucre sur culture de canne à sucre en Afrique du Sud (van der Laan et al. 2015). D'autres études utilisent des modèles hydrologiques pour la caractérisation des impacts. C'est le cas de l'application du modèle SWAT pour le calcul de facteurs de caractérisation (indice de rareté de l'eau) à l'échelle du bassin versant (Scherer et al. 2015). Certes le modèle SWAT se montre bien plus performant que les modèles hydrologiques globaux actuellement utilisés (e.g. WaterGAP, Alcamo et al. (2003)), mais les efforts de calibration sont si importants qu'il n'est pas envisageable de l'appliquer à une échelle globale (Scherer et al. 2015). Il arrive donc que l'usage d'un modèle soit trop lourd pour une application à grande échelle (pour le calcul de facteurs de caractérisation), ou pour une application « en routine » au stade de l'inventaire. Néanmoins, ces applications sont de formidables outils de connaissance qui nous permettent d'identifier les limites de nos approches et de s'enrichir des disciplines concernées. Plus de dialogue avec les disciplines de l'agronomie et de l'hydrologie permet aussi une rétroaction sur les modèles en faisant émerger auprès de nos pairs nos besoins spécifiques en évaluation environnementale, comme illustré au Chapitre 3 avec un intérêt pour l'utilisation du rendement comme donnée assimilée par les modèles pour l'estimation de la transpiration.

### Quantifier l'incertitude associée à l'usage du modèle : une priorité

La sensibilité du modèle E.T. a été testée d'une part à la variation des données d'entrée mais aussi au choix des formalismes utilisés dans le modèle et a révélé sa pertinence sur un plan scientifique et sa robustesse pour simuler les flux d'eau et de sels pour différentes pratiques agricoles. Cependant, une des lacunes de ce travail de thèse est la quantification de l'incertitude cumulée du modèle associée aux variables d'entrée. Une telle analyse d'incertitude doit être une priorité afin d'être en mesure d'associer une estimation d'incertitude aux sorties du modèle pour ses futurs usages. Etant donné qu'accéder à des volumes d'eau d'irrigation journaliers peut être difficile dans certaines situations et représente un frein à l'application du modèle, une analyse de la sensibilité à l'usage de données mensuelles désagrégées serait aussi nécessaire.

Les pistes d'améliorations de la robustesse et du domaine de validité du modèle sont discutées dans le chapitre 4. Notamment via la prise en compte du flux d'eau de remontées capillaires, une meilleure

estimation du coefficient de stress dû à la salinité distinguant les phases critiques de croissance, et une estimation de l'évaporation non restreinte à une irrigation localisée. Ces pistes d'amélioration se basent sur le modèle Aquacrop développé par la FAO et présenté comme la nouvelle version de CropWat.

## **Est-il possible d'appliquer le modèle d'inventaire des flux d'eau et les indicateurs associés pour évaluer des pratiques agricoles ?**

### **Une faisabilité démontrée sur un cas d'étude exigeant**

L'enjeu était d'évaluer la faisabilité de calculer les impacts liés à la consommation d'eau avec la méthode de caractérisation la plus exigeante (Boulay et al. 2011a; 2011b), sur un cas d'étude exigeant. Il était donc pertinent de travailler sur une culture pérenne, qui plus est destinée à l'exportation, dans une exploitation en milieu non-contrôlé (non-expérimentale), et dans un contexte où la rareté de l'eau et la salinisation sont des enjeux majeurs. La réalisation de l'ACV d'une Mandarine produite au Maroc et exportée vers la France (Chapitre 5) a été possible grâce au partenariat avec les Domaines du Maroc qui ont mis à disposition à la fois leur personnel, leurs données et leur expertise du terrain. Notre étude prouve qu'il est possible de faire l'ACV d'une culture pérenne (incluant l'évaluation d'impact lié à la consommation et la dégradation de qualité de l'eau). La réalisation de futures ACV est maintenant facilitée par la disponibilité du modèle E.T.. Néanmoins, la modélisation du cycle pérenne complet reste un travail considérable, non seulement parce qu'il faut remonter le temps depuis l'implantation du verger, mais aussi parce qu'il faut prévoir le futur jusqu'à son arrachage. Des stratégies de modélisation des systèmes pérennes en ACV doivent être mises au point en fonction des objectifs de chacune des ACV. Bessou et al. (2013 et 2015) ont récemment proposé une première formalisation de ces stratégies. Par ailleurs, si l'usage du modèle E.T. ne nécessite pas de connaissances spécifiques, la définition de scénarios de projection nécessite en revanche des connaissances et un appui par des experts de la culture étudiée (les producteurs eux-mêmes et les agronomes) et la sensibilité des résultats mérite également d'être testée aux différents scénarios de projection et aux différentes hypothèses possibles pour leur construction.

### **Un compromis entre précision et applicabilité**

Comme toujours, il existe un compromis entre la qualité scientifique des résultats et l'applicabilité des méthodes d'évaluation d'impact. Ce constat est fait notamment par De Boer et al. (2013) pour l'évaluation des impacts liés à la consommation d'eau pour la production de lait en Allemagne. Ce constat a aussi été fait dans les premières phases de travail de cette thèse où l'usage d'un modèle gourmand en données s'est retrouvé contraint par le manque de données. En effet, après une prise en main du modèle SaltMed (Ragab 2002) avec son développeur le Dr. Ragab R. (au Center for Ecology and Hydrology, UK), une confrontation avec les données de terrain disponibles a révélé une insuffisance pour sa calibration et sa validation. Ce modèle de flux d'eau, de sels et de l'azote satisfaisait nos objectifs d'estimation des flux pour l'inventaire, mais n'était pas approprié aux contraintes de données et temps disponibles, pour son usage dans un cadre ACV. De plus, des compétences en agronomie étaient

nécessaires. C'est donc pour cette raison que cette thèse s'est attachée à analyser les outils/modèles d'inventaire appropriés en fonction des objectifs de l'ACV et des moyens disponibles (Chapitre 3).

### Une contribution méthodologique pour de futures productions de références

Il est important de rappeler que le cas d'étude ne se veut pas représentatif de la Mandarine Marocaine : cette ACV n'a pas été précédée d'une analyse de la diversité des pratiques pour la production de Mandarine au Maroc, contrairement à l'ACV de la tomate (Payen et al. 2015), qui a été précédée d'une phase de typologie des exploitations de tomate marocaine destinée au marché français. En effet, la contribution de la thèse est avant tout méthodologique, dans un objectif à plus long terme de participer à la production de références et d'alimenter les bases de donnée d'ACV agricoles comme AgriBalyse® (ADEME 2015). Néanmoins, une comparaison avec l'ACV de la clémentine du Maroc montre des résultats d'impact du même ordre de grandeur, sauf pour les impacts eutrophisation, toxicité et écotoxicité. Cette différence observée démontre que : (i) la méthode d'estimation de la lixiviation des nitrates ne doit pas se baser sur le facteur d'émission par défaut de l'IPCC pour des cultures en micro-irrigation sous climat aride, et que (ii) il y a un besoin de caractériser les nouvelles substances actives des produits phytosanitaires. Le modèle E.T., par son estimation du flux d'eau de drainage et de stock d'eau dans les sols, contribue à une meilleure estimation des nitrates lixiviés. Comme discuté en chapitre 5, cette amélioration doit se poursuivre par un meilleur couplage du modèle de bilan hydrique et salin avec un bilan d'azote journalier ou mensuel, et non pas annuel. En effet, un indicateur de lixiviation de nitrates est moins pertinent si la résolution du calcul est à un pas de temps annuel, notamment parce qu'il ne tient pas compte de l'adaptation du producteur aux contraintes du climat.

### Un outil de connaissance pour le producteur

Au-delà de la contribution méthodologique, le résultat de cette étude ACV constitue une information valorisable par le producteur. Une étape supplémentaire de vulgarisation a été nécessaire, avec un effort de communication et de représentation des résultats. Cette transmission des résultats et le suivi de leur interprétation fait partie intégrante de l'étude. Quatre aspects majeurs ont émergés des échanges avec le partenaire sur les résultats d'ACV de la Mandarine: (i) la forte consommation d'électricité pour pomper l'eau dans la nappe est déjà bien connue du producteur qui avait étudié la possibilité de remplacer les pompes électriques par des pompes solaires (mais leur cout est très élevé), (ii) la compréhension de la notion « d'impacts potentiels » en ACV est complexe et souvent confondue avec les impacts réels locaux, (iii) le souhait de comparer avec une autre exploitation pour évaluer la performance environnemental relative est très forte, et (iv) la motivation de compléter cette analyse environnementale par une ACV sociale est très grande. La comparaison des résultats d'impact de la Mandarine a été faite avec la Clémentine moyenne Marocaine (Basset-Mens et al. 2015) (Chapitre 5), mais comparer avec une autre exploitation en appliquant le modèle E.T. et les impacts associés à la consommation d'eau, permettrait non seulement d'identifier le pouvoir discriminant du modèle (au-delà des simulations réalisées au chapitre 4), mais aussi d'identifier si la ferme étudiée présente ou non un avantage environnemental. Ces échanges entre chercheurs et producteurs n'en sont donc qu'à leur début puisque cette étude a engagé une émulation au sein de l'entreprise productrice et exportatrice qui y voit non seulement un outil de connaissance mais aussi un outil de valorisation de leurs produits.

Les Domaines souhaitent d'ailleurs aujourd'hui internaliser la compétence ACV et que l'on mette à leur disposition un outil d'ACV simplifié pour leur permettre de tester des scénarios de pratiques et de gestion logistique.

### Diffusion du modèle - usage du modèle

Le potentiel d'usage du modèle E.T. ne se borne pas au CIRAD ou à l'ADEME, mais potentiellement à toute étude ACV de système agricole. Néanmoins, cela appelle à sa diffusion et son utilisation. Il existe un délai entre le développement et l'application d'une « innovation » issue de la recherche, à cause de plusieurs freins successifs : (i) la communication : il y a eu une explosion du nombre de publications et de praticiens en ACV, (ii) la compréhension : la vulgarisation n'est pas toujours possible ou bien faite, (iii) la faisabilité et le support des logiciels et outils, et enfin (iv) l'acquisition : la réalisation proprement dite.

Après les étapes décrites ci-dessus de confrontation du modèle avec des mesures et d'analyse d'incertitude, la première étape de la diffusion du modèle passera par la publication d'un article. En parallèle, la diffusion doit se réaliser au sein de l'équipe de recherche (groupe ELSA), de l'ADEME (co-financier de ce travail de thèse). La diffusion auprès des instituts techniques permettra aussi de faire évoluer le modèle en adaptant le module de transpiration spécifiquement à d'autres cultures.

### La rareté des données

La rareté des données ou leur difficulté d'accès est fréquente, et reste un frein pour beaucoup d'études ACV. Soit à cause d'absence de données (ex: pas d'archivage des doses de fertilisation), soit par réticence car il n'est pas simple de s'engager dans une démarche d'évaluation environnementale. Cette rareté des données a été une contrainte intégrée tout au long du travail de thèse, en particulier pour la recherche d'un modèle d'inventaire de flux au champ. L'application réussie prouve la faisabilité d'appliquer le modèle dans le cadre d'une ACV. Néanmoins, il est important de rappeler que ce cas d'étude bénéficiait d'un atout: la présence d'une station météo dont les données ont été enregistrées (à peu près régulièrement). De futures mises à l'épreuve du modèle devront donc évaluer sa sensibilité à des données climatiques issues à 100% de bases de données faciles d'accès (telle Climwat, FAO (2010)). Pour le moment, seulement l'utilisation de données d'évapotranspiration de référence ( $ET_0$ ) par défaut a été testée et montre une faible influence sur les résultats (chapitre 4). Nous tendons vers un développement et une harmonisation des bases de données globales (sur les sols, la qualité des eaux...), cela facilitera l'accès aux données et l'application de modèles tels que E.T.. L'acquisition de nouvelles données par imagerie satellitaire doit contribuer à alimenter ces bases de données globales (développé ci-dessous)

### Vers une ACV toujours plus spatialisée

L'application des méthodes d'évaluation des impacts liés à l'usage de l'eau est freinée par l'absence de support de la régionalisation par les logiciels ACV tels que Simapro (PRé Consultants 2011). Par conséquent, le calcul des impacts liés à l'eau doit être réalisé avec Microsoft Excel, sans pour autant faire l'économie d'utiliser un logiciel d'ACV (couplé avec une base de donnée) afin d'en extraire les volumes d'eau consommés par tous les processus d'arrière-plan/amont (lors de la fabrication des engrais par exemple) (Chapitre 5). L'évaluation d'impacts régionalisés pour l'eau peut devenir complexe lorsque l'on

souhaite appliquer des facteurs de caractérisation spécifiques au lieu d'usage, qui plus est à une résolution spatiale différente selon le niveau du processus étudié dans l'ACV : soit de premier plan (résolution fine au bassin versant), soit d'arrière-plan (résolution plus grossière au pays ou globale). Ceci est d'autant plus complexe qu'il n'y a pas que la catégorie d'impacts liée à l'eau qui est concernée. De façon générale, les catégories d'impacts sur lesquelles les ACV agricoles se focalisent (eutrophisation, usage des terres, toxicité) dépendent souvent de caractéristiques spécifiques du lieu d'émission ou de prélèvement (ex : Koellner et al. (2013); Azevedo et al. (2013)). Les méthodes d'évaluation d'impact évoluent dans cette direction et permettent de plus en plus de prendre en compte les spécificités locales et régionales, ce qui est crucial pour les ACV agricoles. Il existe donc un réel besoin de développement des logiciels ACV afin qu'ils supportent la régionalisation. L'enjeu est de pouvoir lier un inventaire spatialisé, à une évaluation d'impact régionalisée à différentes échelles spatiales selon la méthode, et ce pour le cycle de vie complet du produit ou service étudié. Les Systèmes d'Information Géographique (SIG) sont le meilleur moyen de gérer des données spatialisées et sont déjà employés par plusieurs auteurs pour le calcul de facteurs de caractérisation régionalisés ou la génération d'inventaires spatialisés (ex : Núñez et al. 2010). Plusieurs tentatives réussies d'ACV régionalisées assistées par des SIG existent, et ce, grâce à des logiciels open source (Open LCA, Brightway) (Mutel et al. 2012; Rodríguez et al. 2014). Une ACV régionalisée nécessite un effort supplémentaire mais présente un gain considérable notamment en réduisant l'incertitude sur les impacts (Mutel et al. 2012). De plus, ces efforts seront progressivement réduits grâce au développement d'inventaires et de méthodes d'évaluation d'impact régionalisés, et d'une méthodologie pour faire le lien entre les deux. L'ACV régionalisée basée sur des SIG pourra alors être réalisée en routine et ne sera plus réservée à la recherche (Mutel et al. 2012).

L'acquisition grandissante de données par télédétection va favoriser le développement d'inventaires régionalisés. La télédétection présente deux avantages prometteurs : elle donne accès à des informations qui étaient auparavant non-(ou difficilement) accessibles, comme les coefficients de transpiration par exemple (Bastiaanssen et al. 2007; Pereira et al. 2015), sous une forme géo-référencée qui plus est. La télédétection peut aussi contribuer à combler le manque de données sur l'état de salinisation des terres (Abbas et al. 2013). Ces développements de base de données géo-localisées constituent des sources très pertinentes pour alimenter les inventaires, les modèles d'inventaire et les facteurs de caractérisation. Tendons-nous vers des ACV où le praticien n'aura plus qu'à renseigner la latitude et la longitude du lieu de l'émission/consommation pour que les facteurs de caractérisation soient générés ? C'est déjà le cas pour le modèle LANCA qui détermine les impacts d'usage de terres sur les services écosystémiques (purification de l'eau, régulation des eaux et érosion), simplement basé sur la surface et la durée d'un type d'usage de sol et sa localisation géographique (Saad et al. 2013).

### Replacer l'eau dans son cycle

Nous avons discuté et démontré (Chapitre 2) qu'une modélisation pertinente des impacts liés à la salinisation doit se fonder sur un cadre que l'on pourrait qualifier de « mécaniste » (car décrivant des mécanismes environnementaux) suivant une chaîne de causes à effets de devenir des sels, exposition de la cible environnementale aux sels et d'effet de la salinité sur la cible environnementale. Les difficultés rencontrées pour la modélisation des impacts liés à l'eau montrent qu'il devrait en être de même pour les impacts liés à la consommation d'eau. En effet, partitionner l'eau bleue de l'eau verte (ou l'eau de

pluie des eaux de surface et souterraines) pour les systèmes agricoles est non seulement associé à une incertitude importante à cause de la méthode d'estimation, mais est aussi critiquable d'un point de vue hydrologique et agronomique. Il faut tout d'abord être vigilant sur le mode d'estimation de l'eau verte car elle est souvent calculée comme étant la pluie efficace, alors que la définition même de la pluie efficace varie beaucoup. En effet, par définition, on pourrait inclure dans la pluie efficace la part d'eau qui est destinée à lixivier les sels puisque cela est nécessaire (donc efficace) à la production agricole (Dastane 1978). Mais alors dans ce cas, la pluie efficace n'est pas égale à l'eau verte puisque cette dernière n'inclut pas l'eau drainée (Cf glossaire). Partitionner l'eau originaire de la pluie de l'eau de celle originaire des eaux de surface et souterraine donne l'impression que ce sont deux stocks distincts (Perry 2014). Et ce, d'autant plus qu'on leur assigne des facteurs de caractérisation très différents en ACV.

Par conséquent, il semble cohérent de renouer avec les principes de l'hydrologie et évaluer les flux d'eau comme étant partie intégrante d'un cycle de l'eau (ou plutôt d'un petit cycle de l'eau à l'échelle du bassin versant, et d'un grand cycle de l'eau global) gouverné par les lois de conservation de masse et d'énergie. Les facteurs de caractérisation doivent correspondre à une caractérisation du devenir du flux d'eau (hydrologique) de l'inventaire, puis de son effet. Il est notamment crucial de prendre en compte le fait que la contribution du flux d'inventaire (ex : eau consommée) puisse être non marginale comme c'était le cas pour la production de tomate dans le Souss Massa : un contributeur majeur aux consommations d'eau dans la région (Payen et al. 2015).

Cela participerait à la mise en cohérence des cadres d'évaluation de la consommation et de la pollution de l'eau (comme l'eutrophisation ou l'écotoxicité) qui sont déjà basées sur ce cadre. Une illustration des efforts d'harmonisation des méthodes est le travail de Veronesi et al. (2015) permettant d'évaluer les impacts combinés liés à l'usage des terres et de l'eau sur la biodiversité.

Une question persiste sur un cadre de prise en compte des impacts liés à la consommation d'eau de pluie. Notamment, comment prendre en compte le fait que consommer de l'eau de pluie a un effet sur la disponibilité des eaux de surface et souterraines ? Se baser sur la consommation additionnelle du système étudié par rapport à un système naturel de référence est complexe et hasardeux, car lié à la définition de la végétation naturelle de référence et de l'estimation de sa consommation d'eau.

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## General conclusion (In French)

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L'écoconception des systèmes agricoles et alimentaires appuyée par l'ACV est aujourd'hui une forte demande sociale et politique. Cependant, la mise en œuvre de l'ACV à ces systèmes complexes implique le développement d'outils d'inventaires opérationnels et pertinents et de modèles d'impact liés à l'usage des ressources naturelles comme le sol et l'eau. Les ressources sol et eau possèdent un double statut à la fois de compartiment susceptible d'être dégradé par des pollutions et de ressource ce qui rend leur prise en compte complexe dans le cadre conceptuel de l'ACV. De plus, à l'échelle de la parcelle irriguée l'eau représente une ressource mais aussi un vecteur de pollution qu'il s'agisse de nutriments, de pesticides ou de sels. En effet, les phénomènes de salinisation représentent à l'échelle mondiale une menace considérable pour la fertilité des sols agricoles et pour la qualité de la ressource en eau, et ne sont pourtant pas encore modélisés en ACV.

Basés sur une revue de la littérature approfondie qui a donc notamment révélé un manque de prise en compte des phénomènes de salinisation en ACV et d'outils d'inventaire opérationnels pour nourrir les méthodes de caractérisation les plus récentes, les objectifs de cette thèse étaient :

- La proposition d'un cadre général de prise en compte des phénomènes de salinisation en ACV,
- L'analyse des besoins en outil de modélisation des flux d'eau au champ à des fins d'écoconception appuyée par l'ACV et permettant l'application des modèles de caractérisation des impacts les plus récents,
- Le développement d'un modèle d'inventaire pertinent et opérationnel des flux d'eau et de sels au champ pour toutes cultures y compris les pérennes,
- La mise en œuvre de ce modèle dans une ACV complète pour une culture pérenne pour une prise en compte approfondie des impacts liés à l'usage d'eau douce,
- La discussion des contributions de la thèse et l'identification de perspectives de recherche pour la suite

Ce travail de thèse propose à la fois plusieurs contributions de nature conceptuelle et méthodologique ainsi que des mises en œuvre concrètes et discutées sur un cas d'étude.

Tout d'abord, ce travail de thèse a fourni les bases d'un cadre de prise en compte des impacts salinisation, en fournissant une description des mécanismes environnementaux impliqués et des facteurs responsables, ainsi qu'une discussion approfondie sur ces questions clés qui nuisent à l'harmonisation des méthodes. En particulier, l'harmonisation du statut des aires de protection et du statut du sol est cruciale. La mise en cohérence des inventaires des émissions au champ est incontournable afin qu'ils se basent sur les mêmes hypothèses de délimitation du sol entre technosphère et écosphère. Une limite flexible selon les études ACV nuit non seulement à la compatibilité des résultats, mais introduit aussi un biais en omettant certains impacts. En particulier, les impacts sur le sol lui-même en tant que support de la culture ne doivent pas être négligés. Ceci est

d'autant plus pertinent à l'heure où l'agro-écologie<sup>12</sup> est un terme qui vient de faire son entrée officielle dans la langue française. La contribution de cette phase du travail a donc été essentiellement méthodologique et conceptuelle, l'objectif n'étant pas de produire des références mais bien de donner un cadre pour mieux produire de futures références.

L'analyse des méthodes et données utilisées dans les bases de données eau et inventaire agricoles existantes a révélé leur inadéquation pour un usage de l'ACV à des fins d'écoconception. Les principales caractéristiques attendues pour un modèle de flux d'eau au champ dédié à cette application étaient : une estimation de l'eau consommée par évapotranspiration, de percolation profonde et de ruissellement, basée sur les caractéristiques spécifiques de sol, climat et pratiques agricoles. Si le modèle Aquacrop a été identifié comme opérationnel et pertinent pour l'estimation des flux d'eau au champ en cultures annuelles, aucun modèle correspondant n'était disponible pour les pérennes.

Le modèle d'inventaire des flux d'eau et de sels proposé et appelé E.T. répond donc à un manque d'outil simple pour l'inventaire des cultures pérennes, permettant de discriminer les pratiques. Le modèle estime des flux d'eau et de sels au champ, en prenant en compte les pratiques réelles du système étudié, les caractéristiques du sol, le climat et la culture. Il est pertinent dans une démarche d'écoconception des systèmes agricoles, sans être trop compliqué à prendre en mains pour autant. Cette recherche de compromis entre faisabilité et précision a été la ligne directrice de la thèse. Une analyse des moyens d'inventaire à mettre en œuvre en fonction des objectifs de l'étude et des ressources disponibles (données, temps, compétence) a également été proposée. Il est par ailleurs important de rappeler que le bilan salin réalisé ne considère pas la nature des sels en présence, et néglige donc les phénomènes impactant la structure du sol. Une telle prise en compte serait contrainte par le besoin en données supplémentaires à renseigner sur la nature des sels présents dans l'eau d'irrigation.

Le modèle E.T. a finalement été mis en œuvre avec succès dans l'ACV d'une mandarine marocaine produite sur une ferme réelle et conduite du berceau à la porte du marché en France. Pour la majorité des catégories d'impact, l'énergie consommée pour pomper l'eau d'irrigation est apparue comme le principal contributeur révélant un lien étroit entre l'eau et l'énergie dans le profil d'impact de cette culture irriguée. La sensibilité des résultats d'ACV à plusieurs méthodes de caractérisation des impacts liés à l'usage d'eau douce a également été testée et a notamment révélé l'importance de calculer des facteurs de caractérisation spécifiques pour les eaux de surface et de nappe profonde. Enfin, les dommages engendrés par l'usage d'eau douce sont des contributeurs majeurs aux dommages sur la Santé Humaine, les Ecosystèmes et La Ressource.

Les perspectives de recherche de cette thèse résident d'abord dans la mise en œuvre opérationnelle du cadre proposé pour la prise en compte de la salinisation et pour laquelle l'utilisation des SIG est une

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<sup>12</sup> agro-écologie : un ensemble de pratiques agricoles privilégiant les interactions biologiques et visant à une utilisation optimale des possibilités offertes par les agrosystèmes (définition selon le Journal Officiel de la République française du 19 août 2015).

piste prometteuse. Ensuite, ces perspectives concernent l'amélioration du modèle E.T. et l'élargissement de son domaine de validité, sa diffusion et son application sur d'autres cas d'étude (notamment dans le cadre du partenariat avec les Domaines Agricoles du Maroc). Son utilisation conjointe avec une estimation des flux de nutriments et de pesticides représente une autre piste prometteuse pour la pertinence de l'estimation de ces flux et pour la cohérence d'ensemble des méthodes d'inventaire en ACV agricoles. Toutes ces perspectives requerront des interactions entre chercheurs ACV, hydrologues et agronomes.

La prise en compte des impacts liés à la consommation d'eau en ACV agricole possède encore une marge de progrès importante, mais bénéficie d'une émulation qui a fait évoluer ce domaine rapidement en quelques années seulement. Ce travail de thèse prône de façon appuyée une vision plus hydrologique et mécaniste de l'évaluation des impacts liés à l'eau en ACV, pour plus de cohérence entre l'évaluation des impacts de la pollution et de l'usage de l'eau. Les impacts salinisation en sont un excellent exemple car ils sont liés à la fois aux volumes d'eau en mouvement (liés aux usages d'eau et des terres), aux sels en solution dans ces eaux et aux émissions de sels dans l'eau. Le modèle d'inventaire E.T. est cohérent avec cette vision, au stade de l'inventaire.

## Publications list

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### Peer-reviewed publications

**Payen S.**, Basset-Mens C., Perret S., 2015. **LCA of local and imported tomato: An energy and water trade-off**. Journal of Cleaner Production 87 (2015) 139-148. doi: 10.1016/j.jclepro.2014.10.007

**Payen, S.**, Basset-Mens, C., Follain, S., Grünberger, O., Marlet, S., Núñez, M., Perret, S. Roux, P (2016) Salinisation impacts in Life Cycle Assessment: a review of challenges and options towards their consistent integration. International Journal of Life Cycle Assessment. doi:10.1007/s11367-016-1040-x

Basset-Mens C, Vannièrè H, Grasselly D, Heitz H, Braun A, **Payen S**, Koch P, Biard Y, 2016 **Environmental impacts of imported and locally-grown fruits for the French market**. Fruits (2016) 71(2) 93-104 doi: 10.1051/fruits/2015050

### Presentations in conferences

Payen, S., Basset-Mens, C., Follain, S., Grünberger, O., Marlet, S., Núñez, M., Perret, S. 2014. **Pass the salt please! From a review to a theoretical framework for integrating salinization impacts in food LCA**. The 9<sup>th</sup> International Conference on Life Cycle Assessment in the Agri-Food sector, 8 - 10 October 2014, San Francisco, USA.

Basset-Mens, C., Payen, S., Braun, A., Vannièrè, H., Jannoyer, M. 2014. **Including tropical fruits in the French eco-labelling scheme: The case of mango from Brazil**. The 29<sup>th</sup> International Horticultural Congress 17 – 22 August 2014- Eco-efficiency in the life cycle of horticultural production, Brisbane, Australia.

Payen, S., Basset-Mens, C. 2013. **Environmental assessment with LCA of tomato produced in Morocco and exported to France: the importance of freshwater use impacts**. International conference on sustainable water use for securing food production in the Mediterranean region under changing climate, 10 – 15 March 2013, Agadir, Morocco.

### Participation in books and reports

Bessou C., Basset-Mens C., Benoist A., Biard Y., Burte J., Feschet P., **Payen S.**, Tran T., Perret S. **L'analyse de cycle de vie pour élucider les liens entre agriculture et changement climatique**. Dans : Changement climatique et agricultures du monde, Edité par Torquebiau E. 2015. Edition Quae. (Available in English soon)

Koch P. and Salou T., 2015. **AGRIBALYSE® : Methodology** – Version 1.2. Ed ADEME, Angers, France.

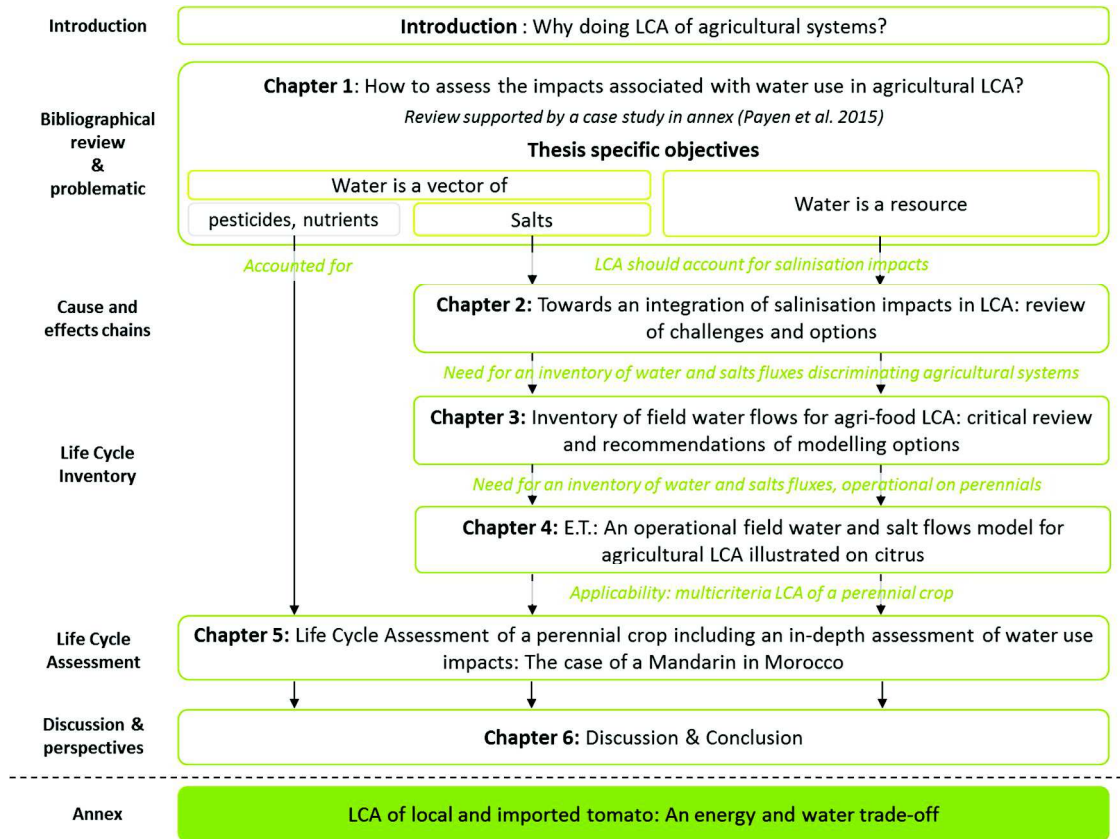
## Annexes

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LCA of local and imported tomato: An energy and water trade-off.....	184
Chapter 1 - Supplementary information - How to assess the impacts associated with water use in agricultural LCA?.....	212
Chapter 2 - Supplementary information - Salinisation impacts in Life Cycle Assessment: a review of challenges and options towards their consistent integration.....	224
Chapter 3 - Supplementary information - Inventory of field water flows for agri-food LCA: critical review and recommendations of modelling options.....	232
Chapter 4 - Supplementary information - E.T.: An operational field water and salt flows model for agricultural LCA illustrated on citrus .....	238
Chapter 5 - Supplementary information - Life Cycle Assessment of a perennial crop with in-depth analysis of water use impacts: The case of a Mandarin in Morocco .....	243







## LCA of local and imported tomato: An energy and water trade-off

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Sandra Payen<sup>1,2,\*</sup>, Claudine Basset-Mens<sup>2</sup>, Sylvain Perret<sup>3</sup>

<sup>1</sup>ADEME, (Agence de l'Environnement et de la Maîtrise de l'Energie), 20 avenue du Grésillé - BP 90406, 49004 Angers, France

<sup>2</sup>CIRAD, UPR Hortsys, ELSA, 34398 Montpellier, France

<sup>3</sup>CIRAD, UMR G-Eau, 34398 Montpellier, France

### Abstract

The environmental impact of imported fresh agricultural products, such as off-season vegetables transported over long distances, is under growing scrutiny. We hypothesised that the environmental Life Cycle Assessment (LCA) ranking between local and imported vegetables might change depending on the impact category considered. We focused on the case study of off-season tomatoes produced in Morocco under unheated greenhouses in a water-scarce area, which covers 68% of the fresh tomatoes imported to France. First, we performed a cradle-to-market gate LCA of the Moroccan production using primary data based on a field survey. Second, we applied the same Life Cycle Impact Assessment (LCIA) method to published cradle-to-farm-gate results of the French tomato cropping system, which also provides off-season tomatoes to the French market and which is characterised by heated greenhouses with a high level of inputs. In addition to typical environmental impact categories, the freshwater use impact was included. The ranking between imported and local tomatoes was different depending on the impact category. Freshwater use had greater impacts under the Moroccan arid climate: 28.0 LH<sub>2</sub>Oeq.kg<sup>-1</sup> of Moroccan tomato and 7.5 LH<sub>2</sub>Oeq.kg<sup>-1</sup> of French tomato. Conversely, the higher level of artificialisation of the French production resulted in greater impacts on total energy consumption, global warming, and eutrophication, even including transport to France for the Moroccan tomato. This reveals a trade-off between freshwater use impacts and the usual/other impacts, mostly energy-related. At the farm gate, we found that the Moroccan tomato water consumption highly contributed to the total damages to Human Health (14%), and Ecosystems (20%) (contribution to Resources depletion was only 2%). Therefore, ignoring the impacts of freshwater use in LCA also underestimates the damages. Moreover, we showed that the assessment of freshwater use impacts and damages still has shortcomings, leading to an underestimation of the impact for the Moroccan tomato case. These results emphasised the importance of considering all of the impact categories when performing an agricultural LCA and the need for a more comprehensive method for assessing the impacts of freshwater use. In particular, the use of an operational tool for estimating water and solute fluxes at the field level is recommended to feed freshwater impact assessment methods.

### Keywords

Environmental impacts; LCA; Water deprivation; Off-season tomato; France; Morocco.

## Highlights

- LCA of fresh tomato production in water scarce southwest Morocco was performed.
- Moroccan imported and French locally produced off-season fresh tomato are compared.
- From a freshwater resource perspective, French local production performs better.
- From a carbon and energy perspective, Moroccan imported production performs better.
- Methodological development is needed for a comprehensive water use impact assessment.

## 1. Introduction

In Northern developed countries, the consumption of fresh agricultural products is currently regular and diversified throughout the year (Freshfel, 2012). In Europe, to meet consumers' year-round demand for fresh vegetables, off-season fresh products are either imported or produced in artificialised cropping systems, such as heated greenhouses. In this context, the environmental impacts attached to the year-round supply of fresh vegetables are receiving increasing attention (Sim et al., 2007; Webb et al., 2013). This is particularly important when imported vegetables are water-demanding crops grown in water-scarce areas.

The case of fresh tomatoes marketed in France in winter is a typical illustration of these issues. The tomato is the most consumed fresh vegetable in France, and its production requires much water. Off-season tomatoes are either produced locally in heated greenhouses or imported from Morocco and Spain. Morocco (North Africa) is the primary supplier of the French market, with 68% of the imported off-season tomatoes (French customs); production for export is located in the Souss-Massa region (West Southern Morocco). This region has a favourable warm climate for off-season production, but water scarcity is a major natural constraint because of low annual precipitation and high evaporation (Bouchaou et al., 2008). In such an arid climate, the assessment of water use efficiency and impacts of agricultural systems is paramount. However, to our knowledge, the environmental impacts of Moroccan tomato production system for export have never been assessed.

Life Cycle Assessment (LCA) is a standardised (ISO, 2006a, 2006b) multicriteria decision support tool for the environmental assessment of products. LCA was chosen by the French government as the reference method for the environmental labelling of food products as part of the Grenelle law 2 (Cros et al., 2010). Nevertheless, the LCA methodology still has shortcomings for the modelling of freshwater use impacts (Kounina et al., 2013). These shortcomings are of particular concern when evaluating irrigated agricultural systems, knowing that 70% of all water extraction worldwide is destined for agricultural use (World Water Assessment Program, 2009). Life-cycle impact assessment of water consumption has evolved rapidly over the past five years, with many new methods improving the completeness of pathway coverage (Tendall et al., 2013), but it has not yet resulted in a single consensus method. The UNEP-SETAC Life Cycle Initiative established an international working group called Water Use in LCA (WULCA) to evaluate the latest methodological developments and make recommendations to fill this gap (Bayart et al., 2010; Kounina et al., 2013). There are several reviews on LCA methods for the modelling of freshwater use impacts (Berger and Finkbeiner, 2012, 2010; Jeswani and Azapagic, 2011; Kounina et al., 2013). The most commonly applied method is that from Pfister et al. (2009), who

proposed the first operational approach for assessing the impacts of freshwater consumption accounting for local freshwater scarcity.

Recent research has shown that the impacts of water use for vegetable production are crucial in the choice of vegetable sourcing. Stoessel et al. (2012) studied a wide range of vegetables, including tomato, and concluded that, from a carbon footprint viewpoint, it is often better to import vegetables produced in warm Southern countries during periods when Northern production requires heating. However, from a water perspective, sourcing vegetables from water-scarce Southern countries is questionable. Page et al. (2011) studied the tomatoes supplied to the Sydney market and also highlighted a trade-off between carbon and water footprints between different tomato production sites in Australia. However, such studies are not multicriterion LCA studies because they only focus on carbon and water footprints. Recent LCA studies have investigated the environmental impacts of French, Italian and Spanish tomato production, surprisingly without considering the impacts of freshwater use (Anton et al., 2005; Boulard et al., 2011; Cellura et al., 2012; Martínez-Blanco et al., 2011; Torrellas et al., 2012). In their recent comparison of locally produced tomatoes in the UK and imported tomatoes from Spain, Webb and colleagues (2013) also did not address the impacts of freshwater use.

The aim of our study was to answer the following question: does the inclusion of the impacts of freshwater use make a difference in the environmental evaluation of off-season vegetables either produced locally or imported from warm Southern countries? We addressed this question through a typical case study: the Moroccan tomato supplying the French market. Therefore, we performed a complete LCA including freshwater deprivation and identified the environmental hot-spots of off-season tomato production in Morocco and delivery to the French market in winter. We then compared these results with local French off-season tomatoes, already studied by Boulard et al. (2011), on a range of environmental impact categories, including freshwater deprivation. We lastly assessed the methodological limitations of the evaluation of freshwater use in LCA.

## **2. Materials and methods**

### **2.1. Geographical context**

In Morocco, tomato production for export to Europe is highly standardised, and 85% of the total tomatoes for export are produced in West Southern Morocco, in the Souss-Massa region (Lacombe, 2010). This alluvial basin produces more than half of Morocco's exported citrus and vegetables (Bouchaou et al., 2008). These crops consume large amounts of water. The Souss-Massa is characterised by a semi-arid climate: a low average rainfall (250 mm.year<sup>-1</sup>), a high potential evaporation (>2000 mm.year<sup>-1</sup>) and average daily temperatures ranging from 19°C in winter to 27°C in summer (Bouchaou et al., 2008). The over-exploitation of groundwater for irrigation has led to the depletion of groundwater resources and the degradation of their quality. Current and future water supplies are threatened by the groundwater level decline and the large variation in salinity of groundwater and surface water (Bouchaou et al., 2008).

## 2.2. LCA goal and scope

With the goal of producing a complete LCA for the Moroccan export tomato for the French market, we defined the functional unit as 1 kg of fresh bulk tomato delivered at the Saint-Charles International Market entry gateway (French distribution hub for fruits and vegetables). The system boundaries (Fig. 1) were from cradle to market (i.e., from raw material extraction to the market entrance gate) and included all direct inputs for seedling production, greenhouse manufacture, tomato production, packaging and transportation to the French market, but excluded capital items other than greenhouses.

Primary data refer to three annual crop cycles (2009-2010-2011) and were collected during in-depth field surveys in one seedling nursery, three farms, and one packaging station, all located in the Souss-Massa region. We used primary data for the consumption of agricultural inputs (fertilisers, pesticides), water, electricity and fuels, the amount of materials (greenhouse components, packaging components...), the use of agricultural machineries, the amount of final products for the nursery, the tomato cultivation and the packaging stages. Table 1 shows key farm inventory data provided by the producers. Secondary data such as input transportation and manufacturing, fuel consumption for truck refrigeration and freight ship container, were obtained from the literature and from the Ecoinvent 2.2 database (Frischknecht et al., 2007) referring to the average European context. Indeed, packaging, fertilisers, pesticides and most of the greenhouse components are manufactured in Europe. Transport mode and distance of farm inputs were adjusted according to the origin. Primary dataset is of high quality and secondary datasets are of basic quality when self-evaluated following the data quality assessment of ILCD (JRC-IES, 2010) (Detailed analysis in table A2a and A2b in Supplementary data). The LCA modelling was performed with Simapro 7.3.2 software (PRé Consultants, 2011).

In the case of co-product generation at the farm gate, including grade-out tomatoes provided to the Moroccan local market, a physical allocation was used (according to their mass). An economic allocation was not possible due to insufficient time series data for price. Conversely, an economic allocation at the nursery gate was used thanks to sufficient seedling price data. The energy (fuel and electricity) and water consumption of the packaging station were allocated to tomato using a physical approach.

## 2.3. Inventory of Moroccan tomato production: from cradle-to-farm-gate

### 2.3.1. Nursery and tomato cultivation

Tomato production is based on grafted plants, which are resistant to soil-borne diseases. The grafted tomato plants are produced during summer period in highly artificialised nurseries, with air-conditioning and moisture control systems, located a few kilometres from the tomato farms. After replanting, the tomato crop grows in non-heated greenhouses, in natural soil, with a drip fertigation system. The greenhouses are of the "Canarian" type, a multi-span greenhouse with a wood or metal frame covered with transparent polyethylene plastic. The crop cycle is about nine months, with planting from August to September and harvesting from October to May. There is no crop rotation. Average yield of the studied farm was 208 ton. ha<sup>-1</sup> (Table 1), this is within the range of reported yield by other producers in the area and seedling companies (Grasselly D. Personal communication) and in the north of Spain, unheated greenhouse tomato production yields reach also 200 ton.ha<sup>-1</sup> (Munoz et al., 2008). However, this yield is

higher than the tomato grown in Almeria in the south of Spain (Soto et al., 2014; Thompson et al., 2007; Torrellas et al., 2012) probably due to less favourable climate in Almeria. After harvesting, tomatoes are packed in cardboard boxes and transported to France by boat or truck. Moroccan tomatoes are exported during winter; during summer, the tariff protection enforced by the European Union is prohibitive. Farmers estimate the water irrigation requirements through the calculation of the potential evapotranspiration of the crop. The farms are part of a 18 050-ha irrigation scheme; on average, over the three cropping seasons, 50% of the irrigation water came from the Youssef Ben Tachfine dam and 50% from the aquifer through wells. Fertilisation is based on local agricultural institution recommendations for each crop growing stage, adjusted according to soil analysis and the farmer's expertise. Because drip fertigation is used, we collected water and fertiliser amounts on a daily basis, from farmers' practices and records.

Because the soil is covered with polyethylene plastic mulch, no herbicide is used. Crop protection management is based upon pest monitoring, except for the systematic soil treatment against nematodes before planting. Fifty-nine active ingredients of pesticides were included in the study; overall, 96.5% in total weight of pesticides applied were characterised.

Regarding energy consumption, the farms use diesel and electricity for fertigation and pesticide treatments. Pumps are used for water extraction from wells, for water and fertilisers mixing and pressurisation in the drip irrigation system. Pesticides are applied using motor-pump units or connection with the fertigation system. Energy consumption was calculated based on the pumps' specifications and their operating time.

The electricity mix of Morocco of 2007 was used. Regarding the end-of-life treatment, as in Morocco the wastes have at least a second life, we considered that all equipment was re-used by other systems, resulting in no environmental cost for our system. Indeed, local small producers re-use mulch plastics, greenhouse plastics and metals, and goat farmers recover the crop residues and damaged fruit to feed their herds. However, after several uses, the materials are landfilled in a wasteland and finally burned. We tested a waste incineration scenario in the sensitivity analysis (Table A3)

### **2.3.2. Field emissions**

Nitrogen oxides, phosphates and pesticides emissions were calculated according to Nemecek and Kägi (2007), nitrous oxide according to IPCC (Guidelines for National Greenhouse Gas Inventories) (2006), and ammonia emissions were based on emission factors for group I from ECETOC (European Centre for Ecotoxicology and Toxicology of Chemicals) (1994). Phosphorus emissions through water erosion were not considered because the topography is flat and the crop was greenhouse covered. Nitrate leaching was considered nil because the daily irrigation volume was below the soil field capacity. The field capacity calculation was based on the ISRIC-WISE global data set of derived soil properties (Batjes, 2006). A more conservative method was also adopted assuming a nitrate leaching of 20% of N-fertiliser (like in Boulard et al. (2011) for soil-systems, presented by Perrin et al. (2014). Results are shown in the sensitivity analysis. The pesticides were assumed to be emitted to the soil (Nemecek and Kägi 2007). Temporary biogenic carbon fixation in biomass was not accounted for since its inclusion has no implication on the results due to no characterisation factor associated. This is in line with practices from the literature on tomato (Boulard et al, 2011; Cellura et al. 2012).

### 2.3.3. Inventory of post-farm production stages

Data related to packaging were collected from a packaging station located near the farms. Tomatoes are washed, sorted according to size and colour, packed in cardboard boxes, palletised, and stored in cold rooms before transport. Export tomatoes to France take two possible routes: by ship from the port of Agadir (Morocco) to Port-Vendres (France), including truck drives to and from the ports, or by truck from the Moroccan packaging station to the Saint-Charles market in France, which is 50 km from Port-Vendres, through Spain. In a 38- to 44-ton refrigerated truck, 24 tons of packed tomatoes can be loaded. The fuel consumption for traction and refrigeration for this vehicle was taken from Tassou et al. (2009). With the sea route, the products are transported by freight ship, in forty-foot refrigerated containers. The fuel consumption for ship propulsion and for container cooling was based on reports from the International Maritime Organisation (Buhaug et al., 2009) and Wild et al. (1999, 2005). The reference scenario assessed was 67% truck and 33% ship, based on exporters' records.

### 2.4. Life cycle impact and damage assessment

The impact assessment phase was performed using the ReCiPe life cycle impact assessment method (Goedkoop et al., 2009), adopting the Hierarchist perspective. The following environmental impact categories were considered: climate change (100 years; kg CO<sub>2</sub>eq); terrestrial acidification (g SO<sub>2</sub>eq); freshwater and marine eutrophication (g Peq and g Neq respectively, based on the nutrient-limiting factor of the aquatic environment); terrestrial, freshwater, and marine ecotoxicity (g 1,4-DBeq; 1,4-dichlorobenzene); agricultural land occupation (m<sup>2</sup>.year); metal and fossil depletion (g Feeq and kg oil<sub>eq</sub>). The non-renewable energy consumption (fossil and nuclear; MJ<sub>eq</sub>) was assessed using the Cumulative Energy Demand method (Frischknecht, 2007).

In addition, the impacts of freshwater consumption were assessed with the method of Pfister et al. (2009) compatible with ReCiPe (Pfister et al., 2011b). The calculation of freshwater deprivation is based upon the inventory of consumed water flows. This includes water flows from the aquifer and the dam into the farming system for irrigation and agrochemical preparation, and the water use associated with background processes (e.g., farm inputs, manufacturing and transport). Irrigation water was assumed to be fully consumed. Water consumption of the background processes was quantified using the Ecoinvent 2.2 database (Frischknecht et al., 2007) and represented only 13% of the total water consumed from cradle to farm gate. Adopting a conservative approach, this water was considered as consumed. The mid-point freshwater deprivation (L H<sub>2</sub>O<sub>eq</sub>) is calculated through the multiplication of each instance of consumptive water use by the relevant water stress index (WSI) and then summed across the life cycle (Pfister et al., 2009). The WSI reflects the local freshwater scarcity and is based on a water withdrawal-to-availability ratio calculated with the Water GAP 2 model. The WSI of the Souss-Massa region is 1, meaning that all of the water consumed potentially contributes to freshwater deprivation. The water consumed during background processes (e.g., inputs and manufacture) was weighted with the global average WSI (0.669).

We then explored the aggregation of impacts into damages and analysed the contribution of freshwater deprivation to these damages. Thus, we calculated end-point damages to the three areas of protection: Human Health, Ecosystems, and resources, using ReCiPe and Pfister et al. (2009) end-point assessment.



The damage factors for the Souss-Massa area were as follows: 2.79E-06 DALY.m<sup>-3</sup> for Human Health, 2.77E-08 species.year.m<sup>-3</sup> for Ecosystems and 0.895 \$.m<sup>-3</sup> for Resources. Water consumed during background processes was assigned the global average damage factor for each process.

Scenario analyses were performed to test the robustness of our results. We modelled the incineration of all wastes (greenhouse and nursery materials) instead of the re-use modelled in the reference scenario; we modelled a full truck load (24 tons of tomato) instead of the Ecoinvent default average load (11.68 tons); we modelled an economic allocation instead of a physical allocation, at both farm and packaging stages (7.5 Dirham per kg of exported tomato and 2.5 Dirham per kg of locally sold tomato and the few other vegetables); we modelled a truck modernisation (from EURO 3 to EURO 4) for the tomatoes exportation to France; and we modelled different shares of means of transportation (100% ship route or 100% truck route). Relevant outcomes of these analyses will be mentioned in the results section. See Table A3 in Supplementary data for details.

## 2.5. LCA comparison of Moroccan and French off-season tomato production

We compared our cradle-to-farm-gate Moroccan results with the cradle-to-farm-gate LCA results obtained by Boulard et al (2011) for French off-season tomato production. Boulard et al (2011) defined typical cropping systems for each region of production, based on data and on the expertise of French technical extension services. The French off-season tomato crop grows during winter under soil-less conditions in heated glass or plastic greenhouses in north-western and south-eastern France and requires high levels of inputs (Tab. 1).

Boulard and colleagues gave us access to their data set and permission to recalculate the life cycle impact assessment results with the same method we applied to Moroccan production (ReCiPe including Pfister et al. (2009), at both mid-point and end-point levels).

The impact categories included for the comparison were as follows: climate change (100 years), non-renewable energy consumption (fossil and nuclear), marine and freshwater eutrophication, terrestrial acidification, and water deprivation (Pfister et al., 2009). We calculated the water deprivation impact by multiplying the volume of water consumed (adopting a conservative approach, all water was considered consumed) with the average Water Stress Index (WSI) for France (0.181) because it was not possible to precisely locate the tomato cropping systems in France. The French off-season tomato cropping systems do not consume precipitation water. The ecotoxicity data could not be recovered from Boulard et al. (2011) because they were calculated independently and with a specific method.

## 3. Results and discussion

### 3.1. Environmental impacts of the Moroccan off-season tomato production and delivery

Over the entire tomato life cycle, the tomato cultivation stage was the main contributor to the freshwater eutrophication, ecotoxicity, metal depletion and freshwater deprivation impact categories, whereas the tomato packaging stage had the largest contribution to agricultural land occupation (Tab.

2). Transport from Morocco to France was the main contributor to climate change, terrestrial acidification, marine eutrophication and fossil depletion (Tab. 2). The contribution of seedling production was small for all impact categories (less than 4%).

Climate change - Transport to France was the main contributor to climate change, responsible for 44% of the impact of tomatoes delivered to the French market (Tab. 2). This was mainly due to CO<sub>2</sub> emissions from trucks. The tomato cultivation contributed 37% to climate change impact, mainly due to CO<sub>2</sub> emissions occurring during the manufacture of greenhouse components and to electricity consumption for fertigation (Fig. 2). Tomato cardboard packaging contributed 17% (Tab. 2). The scenario analysis showed that climate change impacts were sensitive to the transport route and truck load. Indeed, tomato being entirely exported by freight ship reduced the climate change impact by about a quarter, whereas transport exclusively by truck entailed a 13% increase. Modelling the trucks as full (24 tons of tomato) reduced the impacts by 18% (Tab. A3).

Non-renewable energy use - Transport to France contributed 39% to the total non-renewable energy consumption, tomato cultivation (fertigation and greenhouse manufacture) 34% and packaging 23%.

Terrestrial acidification - Transport to France was the main contributor to terrestrial acidification, with 50% of the impact, followed by the tomato cultivation (39%) and packaging (10%). Impact was dominated by nitrogen oxides emissions during the truck transportation, sulphur dioxide emissions related to fertigation (fertiliser production and energy consumption), and ammonia emissions occurring after N-fertiliser field application (Fig. 2). A scenario considering a full truck showed a 15% reduction of impacts.

Eutrophication - Tomato cultivation was the main contributor to freshwater eutrophication, with 66% of the impact. This was primarily due to phosphate emissions during the production of fertilisers. Manufacture of packaging and transportation contributed 20% and 12%, respectively, to freshwater eutrophication. When testing an economic allocation, thus considering a higher economic value for the exported tomato than the locally sold tomato, the freshwater eutrophication impact increased by 15%. Transport to France was the main contributor to marine eutrophication, with 38% of the impact, closely followed by packaging (36%), and then tomato cultivation (26%). Contribution to marine eutrophication was dominated by the emissions of nitrogen oxides from truck use.

Ecotoxicity - Tomato cultivation was the main contributor to all ecotoxicity impact categories: 96% of terrestrial ecotoxicity, 59% of freshwater ecotoxicity, and 54% of marine ecotoxicity. Terrestrial ecotoxicity impacts were directly related to pesticide emissions (Fig. 2), more precisely to the release of Cypermethrin and Methomyl. Regarding freshwater ecotoxicity, the key contributors were the pesticide emissions, the greenhouse structure manufacturing, and the energy use for fertigation. For marine ecotoxicity, greenhouse structure manufacturing and energy use for fertigation were the main contributors, while the pesticide contribution was small (less than 3%) (Fig. 2).

Land use – Surprisingly, packaging contributed 69% of the agricultural land occupation, because of the forest area required for producing the wood-made cardboard, whereas tomato cultivation represented only 30% of the impact.

Resource depletion - Tomato cultivation contributed 69% of metal depletion, because of iron used primarily for the greenhouse structure, and secondly for the fertigation system. Regarding fossil depletion, the petrol consumption for trucking was responsible for 46% of the impact, whereas tomato

cultivation represented 34% (explained by the polyethylene composition of the plastic covering the greenhouse).

Freshwater deprivation - The tomato cultivation was responsible for 94% of the freshwater deprivation over the entire tomato life cycle due to irrigation water use. An economic allocation increased the freshwater deprivation impacts by 20%.

Additional ReCiPe impact categories are presented in the supplementary data (Tab. A4).

The high standardisation of the Moroccan tomato production system is an argument in favour of our data being representative. Nevertheless, this study would benefit from additional field survey and data for validation.

## 3.2. LCA comparison of imported Moroccan and local French production systems

### 3.2.1. When importing has lower environmental impacts: the energy and global warming evidence

Surprisingly, for the French off-season vegetable market, sourcing local tomatoes during winter in France is not the best option regarding global warming, energy use and eutrophication potential (Tab. 3). Our results reinforce the idea that food miles can be a misleading indicator (Milà-i-Canals et al., 2008; Page et al., 2011).

Indeed, regarding energy use and global warming potential, our results showed that export off-season tomatoes grown in non-heated greenhouses in Southern Morocco had less impact than local French tomatoes grown under heated greenhouses. Even considering transport to France, the energy use was three times lower for the Moroccan export tomato (Tab. 3). We explain this result by the low motorisation level of the Moroccan system and the high environmental impacts of heated crops. Comparison with the energy use of the Spanish tomato at the farm gate (Torrellas et al., 2012) showed that the tomato production in Morocco and in Spain, both under non-heated greenhouses, had similar energy use impacts: 3.61 and 4.00 MJ.kg<sup>-1</sup> tomato, respectively. These similar results highlighted that sourcing tomatoes in warm, southern countries seems more favourable from an energy perspective even if adding the extra burdens due to transport. Williams et al. (2008) and Webb et al. (2013) reached the same conclusion comparing Spanish tomato production and delivery to the United Kingdom with local tomato production.

Our results confirm the lesser impacts of Moroccan export tomato compared to local French production, with 95% and 38% less impact in terms of marine eutrophication and freshwater eutrophication, respectively. Even when packaging and transport to France are included, the impacts are reduced by 79% and 8%, respectively. When testing a conservative approach: assuming a nitrate leaching of 20% of N-fertiliser (Boulard et al. 2011), the marine eutrophication potential reaches 0.68 g Neq kg<sup>-1</sup> Moroccan tomato, but is still below the 0.96 g Neq kg<sup>-1</sup> of the French tomato at the farm gate.

Acidification potential results showed the same trend, but only at the farm gate. In contrast with previous impact categories, inclusion of the post-farm stages (packaging and transport) brought the Moroccan tomato above the French tomato (Tab. 3), primarily because of the emissions of acid particles during the transportation from Morocco to France.

### **3.2.2. When producing locally has lower environmental impacts: the freshwater use evidence**

Growing crops with high water requirements in water-scarce areas has important implications. Indeed, although the water use efficiency was similar, with 28.6L.kg<sup>-1</sup> for Moroccan tomatoes and 32.8L.kg<sup>-1</sup> for French tomatoes, Moroccan tomato freshwater deprivation was almost four times higher, with 28.0 LH<sub>2</sub>Oeq.kg<sup>-1</sup> for Moroccan tomatoes and 7.5 LH<sub>2</sub>Oeq.kg<sup>-1</sup> for French tomatoes (see Supplementary Fig. A5). This was explained by the high WSI of the Souss-Massa area. Our results confirmed those of Pfister et al. (2011a), who modelled the global water consumption of 160 crops and characterised the irrigation water volume with the WSI. Although their water inventory was not specific to the cropping system (based on FAO's CROPWAT model, global databases, and statistical data), these authors obtained the same ranking regarding freshwater deprivation for the Moroccan and French tomatoes, with 29.2 L H<sub>2</sub>Oeq.kg<sup>-1</sup> and 0.25 L H<sub>2</sub>Oeq.kg<sup>-1</sup>, respectively.

However, certain issues and limitations remain with regard to the freshwater use impact. The WSI is calculated at annual scale, which seems irrelevant for regions with distinct dry and humid seasons and more particularly for agricultural water use (Tendall et al., 2013). Therefore, Pfister and Baumann (2012) are developing monthly WSI indicators. Moreover, there is no consensus on the freshwater characterisation factors regarding both the numerator (water withdrawal (Pfister et al., 2009) or water consumption (Berger and Finkbeiner, 2012; Boulay et al., 2011b)) and the denominator (whether to include groundwater and surface water stocks (Boulay et al., 2011b; Tendall et al., 2013) or not (Pfister et al., 2009) in the total available water). The spatial resolution of the characterisation factors is also of critical importance (country, watershed or sub-watershed scale; Tendall et al., 2013). It is important to note that the tomato production for exportation is not marginal in the Souss-Massa area and represents an important part of the total water withdrawal. Hence, the function studied directly influences the numerator of the WSI defined by Pfister et al. (2009): the system studied affects the characterisation factor.

Our results demonstrated that the ranking of Moroccan export tomatoes against local French tomatoes depended on the impact category. There is a trade-off between the low impacts of energy use, global warming potential and eutrophication of winter production in Southern warm countries and the high water stress in those arid countries. Hospido et al. (2012), Page et al. (2012), and Stoessel et al. (2012) also highlighted the trade-off between water and carbon footprints, while Pfister et al. (2011a) showed the trade-off between water footprint and land use depending on the location of the crop production. These outcomes highlight the importance of including all of the potential impacts when using LCA to compare agricultural system alternatives. Standalone mid-point indicators addressing a unique environmental issue should be used with caution (Page et al., 2012).

### **3.2.3. Damage-wise comparison of Moroccan and French tomato production**

Because the ranking of systems differed depending on the impact category, setting recommendations is challenging. Indeed, which priority should be set between global warming and local (or regional) freshwater deprivation? Because the decision making process should at least in theory be based on scientific evidence, aggregation of impacts into damages seems to be a promising approach to help

decision makers. Thus, we calculated the end-points to compare the damages of the Moroccan and French tomato production and analysed the contribution of freshwater deprivation to the total damages.

The damages of the Moroccan tomato production were 79%, 74% and 88% lower than the French system at the farm gate for Human Health, Ecosystems and Resources, respectively (Fig. 3). Even when adding the Moroccan tomato packaging and transport to France, the damages remained 54%, 41% and 69% lower for the imported tomatoes.

The contribution of mid-point impact categories to end-point damages indicated that the damages from climate change and fossil depletion were most important. For the French tomato, the contribution of water deprivation to the total damages was negligible, while the contributions of climate change (more than 90% for Human Health and Ecosystems) and fossil depletion (more than 99% for Resources) predominated (Fig. 3). Although the Moroccan tomato damages are also dominated by climate change and fossil depletion, the contribution of water deprivation to the total damages was notable, with 14%, 20% and 2% contributions to Human Health, Ecosystems and Resources, respectively. Thus, excluding the freshwater use impacts would have underestimated the damages to Human Health and Ecosystems. The surprisingly low contribution of freshwater use to the Resources damages may be explained by the low damage factor for water (0.89 \$.m<sup>-3</sup> in Souss-Massa) compared with the high damage factor for crude oil (14,350 \$.m<sup>-3</sup> oil). The freshwater damage factor is based on the concept of backup technology and relies on the money required for seawater desalination (1 \$.m<sup>-3</sup>; Pfister et al., 2011b). We further investigated this aspect by expressing the Resources damages in terms of surplus energy instead of monetary value. When assessing the water contribution to Resources damages using EcoIndicator 99 (Goedkoop and Spriensma, 2001), the water impact contribution to Resources damage reached 22%. This is explained by more similarity between the damage factors for crude oil (5,08 MJ surplus.m<sup>-3</sup>) and water (9.34 MJ surplus.m<sup>-3</sup>) in this method. This outcome shows that expressing damages to Resources in terms of energy equivalent or monetary value strongly influences the results. Generally speaking, modelling the cause-effect chain up to the damages is associated with high uncertainty (Jolliet et al., 2003), particularly for freshwater damages assessment, which is still under development. Thus, it would be inappropriate to make recommendations for tomato sourcing based on end-point results. Beyond end-points, the impacts scale is crucial: water deprivation is a pressing local issue, whereas climate change is a global issue. The decision level and viewpoint of the decision makers (policy maker, consumer or farmer) will prevail in the decision-making process.

An analysis of water damages revealed that the contribution of water consumed during background processes may be important, demonstrating the importance of localising the water withdrawals to assign the region-specific WSI instead of the global WSI (Fig. A5).

### **3.3. The need for a reliable inventory for accurately modelling the impacts of freshwater use**

As shown above, considering the impacts of freshwater use in the LCA of local versus imported tomatoes is critical; ignoring them may lead to underestimating the total damages of the studied systems. However, we demonstrated that the freshwater impact and damage assessment still has shortcomings.

Extensive and comprehensive research is on-going for the modelling of impacts and damages due to freshwater use. We want to emphasise below the need for methodological improvement in the inventory stage because it is a complex task to which the impact assessment is closely related.

First, the inventory should differentiate the sources of water (from surface or groundwater) because they have different renewability rates and functionalities (Bayart et al., 2010). This is particularly relevant in the coastal Souss-Massa area, where the groundwater is threatened by over-exploitation of long residence time water (several thousands of years) and by salinisation by seawater intrusion (Bouchaou et al., 2008). The average Souss-Chtouka aquifer withdrawal to recharge ratio is 180% (Faysse et al., 2012). In this context, accounting for the groundwater resource depletion and salinisation impacts provoked by the agricultural activity would probably result in greater impacts than assessed in this study. The development of salinisation impacts pathways are needed in LCA (Stoessel et al., 2012), as salinisation may affect both water and soil (Williams, 1999; Wood et al., 2000). The first attempts to model salinisation impacts in the LCA framework (Amores et al., 2013; Feitz and Lundie, 2002) must be completed with other salinisation pathways such as the one commented in this case study.

Another important aspect regarding the water inventory concerns the water balance. Crop water consumption is often estimated through the modelling of crop water requirements, with tools such as CROPWAT (Faist Emmenegger et al., 2011; Milà-i-Canals et al., 2008; Pfister et al., 2011a, 2009; Ridoutt and Pfister, 2010). Primary data collection, as performed in this study, is preferable because the producer may use more or less water than predicted by the model due to natural and socioeconomic circumstances. For example, in the case study of Spanish tomato production, Torrellas et al. (2012) indicated that the irrigation water supply included a 25% surplus in order to counter soil salinisation. When primary data on water use refers to water withdrawals, it is necessary to subtract drainage, deep percolation, return flow and runoff, all of which return to the environment, in order to calculate the actual water consumption. The method by Pfister et al. (2009) focuses on the water consumed and does not account for the quality degradation of irrigation return flows: the loss of water quality as a loss of freshwater resources is not addressed. However, irrigation return flows carry more salts, nutrients, minerals and pesticides into surface and ground waters, impacting downstream agricultural and natural systems (Tilman et al., 2002). Indeed, water is a vector of solutes and pollutants that may degrade water quality and thus affects the resource. Contrary to the method of Pfister et al. (2009), the framework proposed by Boulay et al. (2011a & b) considers that water-quality degradation can lead to water deprivation if the quality is no longer suitable for use. They address water-related impacts accounting for both input and output water flows in terms of quantity, quality and origin, as recommended by the WULCA working group (Kounina et al., 2012). However, this method requires inventorying the volume and quality of the released water, which is a complex task for agricultural systems because it depends on local parameters of soil, climate, and practices. There is a need for a consistent inventory modelling approach: linking the input flows of water, pesticides, nutrient and salts, with the output flows via a model accounting for soil, climate, and practices. Such an operational tool would be valuable to feed current and future freshwater impact assessment methods. The central and recurring question is to find the correct balance between data requirement and accuracy.

## 4. Conclusion

This study not only produces a reference for the environmental impacts of a Moroccan tomato, but also highlights crucial issues related to the comparison of environmental impacts of food products. First, we produced a cradle-to-market LCA study, including the impacts of freshwater use, for one typical case study of the off-season supply of vegetables: off-season tomatoes, produced for the French market, grown in the arid region of West Southern Morocco under non-heated greenhouses. Over the entire life cycle studied, tomato cultivation mainly contributed to water deprivation, freshwater eutrophication, ecotoxicity and metal depletion, whereas tomato transport from Morocco to France was the main contributor to climate change, terrestrial acidification, marine eutrophication and fossil depletion. Second, we applied the same LCIA method to the French cropping systems, already studied by Boulard et al. (2011), characterised by heated greenhouses with high levels of inputs, which also provide off-season tomatoes to the French market. The comparison of the environmental impacts of the Moroccan and the French tomatoes shows that the inclusion of the impacts of freshwater use is critical, revealing a trade-off between usual impact categories, mostly energy-related, and freshwater use impacts. Indeed, sourcing tomatoes in France mitigates impacts from a freshwater resource perspective but not from carbon, energy, or eutrophication perspectives. Aggregating impacts into damages did not allow us to make recommendations due to methodological shortcomings and uncertainty in the current damage modelling. This outcome is particularly relevant for food LCA addressing the question of product sourcing: how to build a decision when assessing the best sourcing option from an environmental point of view? This study shows that it is paramount to include all relevant impacts in LCA, such as water deprivation for irrigated agricultural systems, and also identify key limitations of the current methods for freshwater use assessment. Indeed, the current freshwater impact assessment method is not complete and probably leads to underestimating the impacts for the Moroccan tomato study case. Aquifer overuse causing water depletion and salinisation is not properly addressed. In addition, impact assessment methods should be based on a reliable inventory. An operational tool estimating water fluxes both qualitatively and quantitatively would be valuable to feed current and future freshwater impacts assessment methods of agricultural products.

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Tables and figures

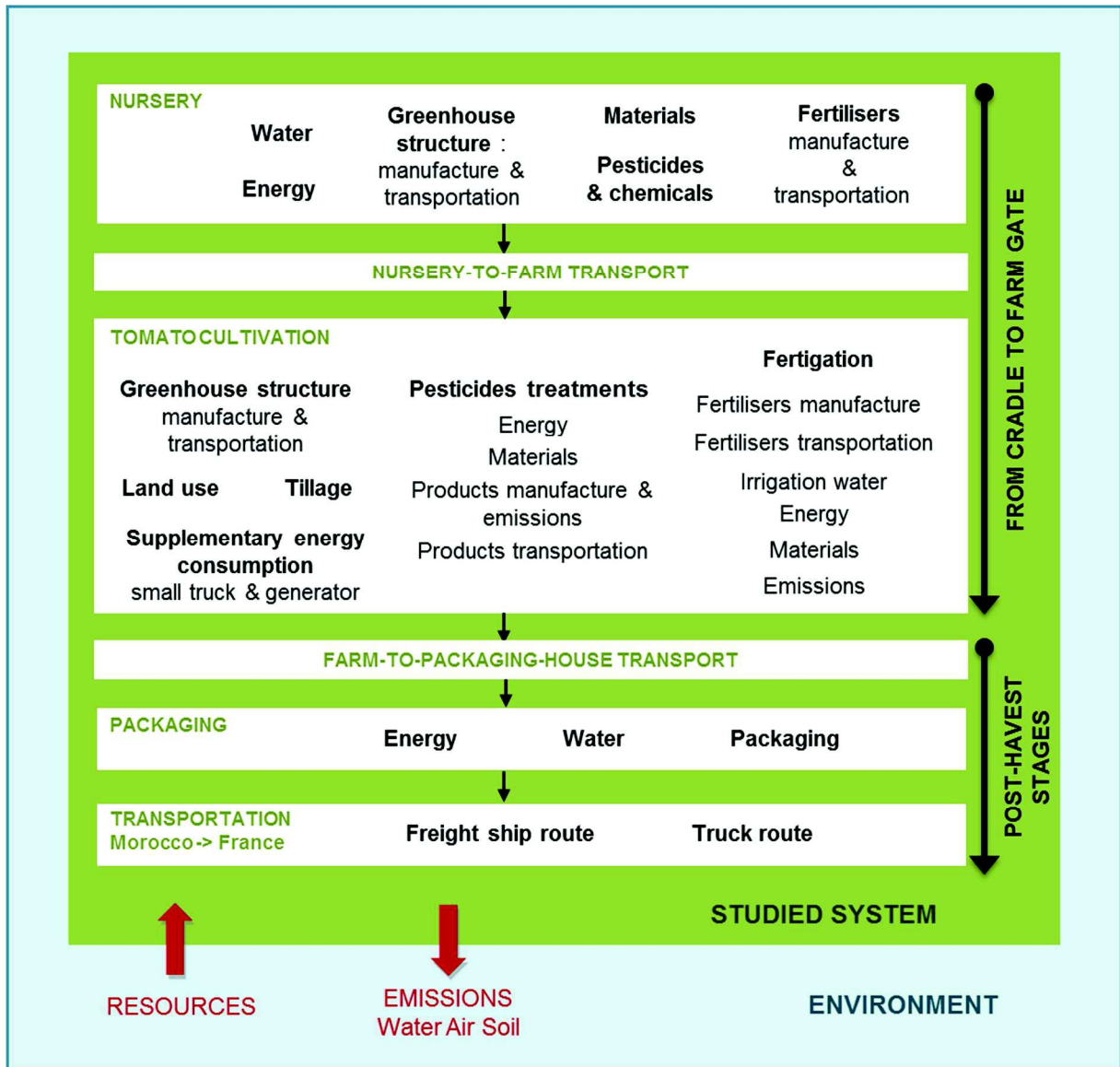


Figure 1. Flow diagram for the Moroccan off-season tomato production and delivery to the French market (2009-2011, Souss-Massa region, Southern Morocco).

**Table 1. Key inventory data for the Moroccan and French tomato cropping systems.**

Parameter	Unit	This study 3 farms average [ <i>min</i> ; <i>max</i> ]	<i>Boulard et al. 2011</i> ( <i>bulk tomato</i> )
Reporting period		2009 to 2011	2006 to 2008
Country (production site)		Morocco	France
Growing period		Winter-Spring	Winter-Spring
Greenhouse structure		Canarian plastic greenhouse	Glass or plastic greenhouse
Substrate		Soil	Rockwool
Greenhouse heating		No	Yes
CO <sub>2</sub> enrichment		No	Yes
Yield	ton.ha <sup>-1</sup>	208 [180; 234]	450
Fertilisation	kg N.ha <sup>-1</sup>	657 [473; 968]	2,561
	kg P <sub>2</sub> O <sub>5</sub> .ha <sup>-1</sup>	483 [311; 776]	1,401
	kg K <sub>2</sub> O.ha <sup>-1</sup>	1,742 [1,285; 2,458]	5,378
Irrigation water	m <sup>3</sup> .ha <sup>-1</sup>	5,591 [4,430; 6,296]	12,500
Energy consumption	kWh.ha <sup>-1</sup>	26,751 [18,414; 42,840]	2,965,000

**Table 2. Contribution analysis of 1 kilogram of tomatoes at St Charles market gate, ReCiPe midpoint impact assessment method (Hierarchist), including the water characterisation factors of Pfister et al. (2009).**

Impact Category	Unit	Total	Nursery	Tomato cultivation	Packaging	Transportation to France
Climate change	kg CO <sub>2eq</sub>	0.546	0.012	0.203	0.091	0.240
Non-renewable energy (fossil & nuclear)	MJ <sub>eq</sub>	9.131	0.391	3.220	1.588	3.932
Terrestrial acidification	g SO <sub>2eq</sub>	3.203	0.041	1.235	0.328	1.598
Freshwater eutrophication	g P <sub>eq</sub>	0.168	0.002	0.111	0.034	0.020
Marine eutrophication	g N <sub>eq</sub>	0.206	0.002	0.053	0.073	0.078
Terrestrial ecotoxicity	g 1,4-DB <sub>eq</sub>	1.408	0.006	1.347	0.021	0.034
Freshwater ecotoxicity	g 1,4-DB <sub>eq</sub>	3.126	0.059	1.830	0.675	0.562
Marine ecotoxicity	g 1,4-DB <sub>eq</sub>	2.888	0.045	1.555	0.573	0.715
Agricultural land occupation	m <sup>2</sup> .year	0.211	0.001	0.063	0.146	0.001
Metal depletion	g Fe <sub>eq</sub>	45.290	0.242	31.394	3.584	10.069
Fossil depletion	kg oil <sub>eq</sub>	0.196	0.008	0.066	0.033	0.089
Water deprivation Pfister et al. (2009)	L H <sub>2</sub> O <sub>eq</sub>	29.738	0.068	27.926	1.176	0.569

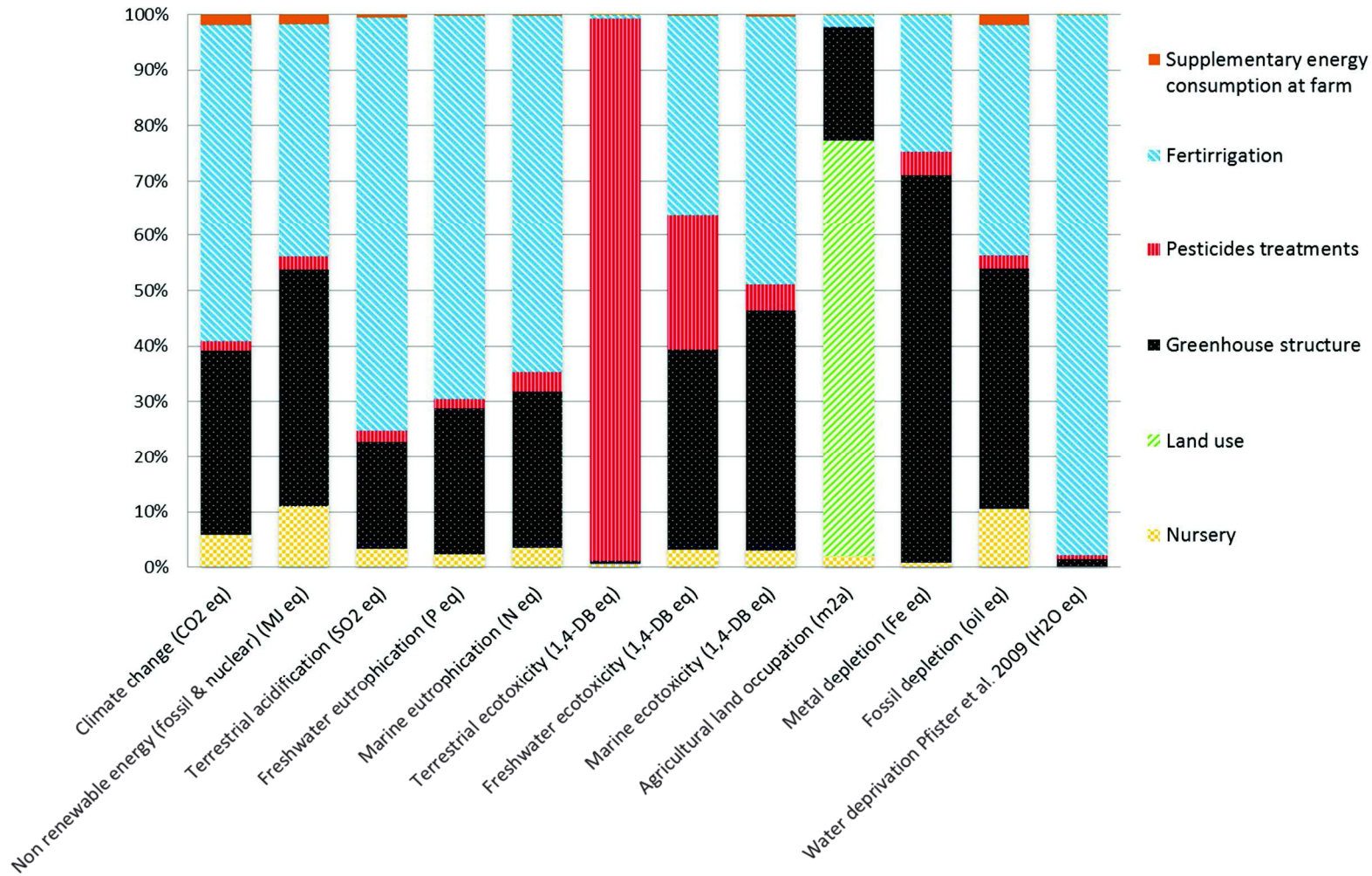


Figure 2. Contribution analysis of 1 kilogram of tomatoes at the farm gate (Morocco), with the ReCiPe midpoint impact assessment method (Hierarchist), including the water characterisation factors of Pfister et al. (2009). Nursery-to-farm transportation and tillage contributions are not visible on the chart.

Publication	Cropping system and functional unit	Climate change (100 year)	Non-renewable energy consumption	Marine eutrophication	Freshwater eutrophication	Terrestrial acidification	Water deprivation, <i>Pfister et al.</i> <i>2009<sup>1</sup></i>
		kg CO <sub>2eq</sub>	MJ <sub>eq</sub>	g N <sub>eq</sub>	g P <sub>eq</sub>	g SO <sub>2eq</sub>	L H <sub>2</sub> O <sub>eq</sub>
This study	1 kg tomato at St Charles market gate, grown in plastic greenhouse	0.55	9.13	0.21	0.17	3.20	29.7
	1 kg tomato at farm gate, grown in plastic greenhouse	0.22	3.61	0.05	0.11	1.28	28.0
Boulard et al. (2011) adapted	1 kg tomato at the farm gate, grown in glass/plastic greenhouse	1.75	30.44	0.96	0.18	2.94	7.5

**Table 3. Global Warming Potential (kg CO<sub>2</sub> eq), Energy use (MJ<sub>eq</sub>), Marine eutrophication potential (g N<sub>eq</sub>), Freshwater eutrophication potential (g P<sub>eq</sub>), Acidification potential (g SO<sub>2</sub> eq) and Freshwater deprivation potential (L H<sub>2</sub>O<sub>eq</sub>) of 1 kg of tomato for the Moroccan and French off-season tomato production systems. <sup>1</sup>Water deprivation for the water consumed during foreground and background processes.**



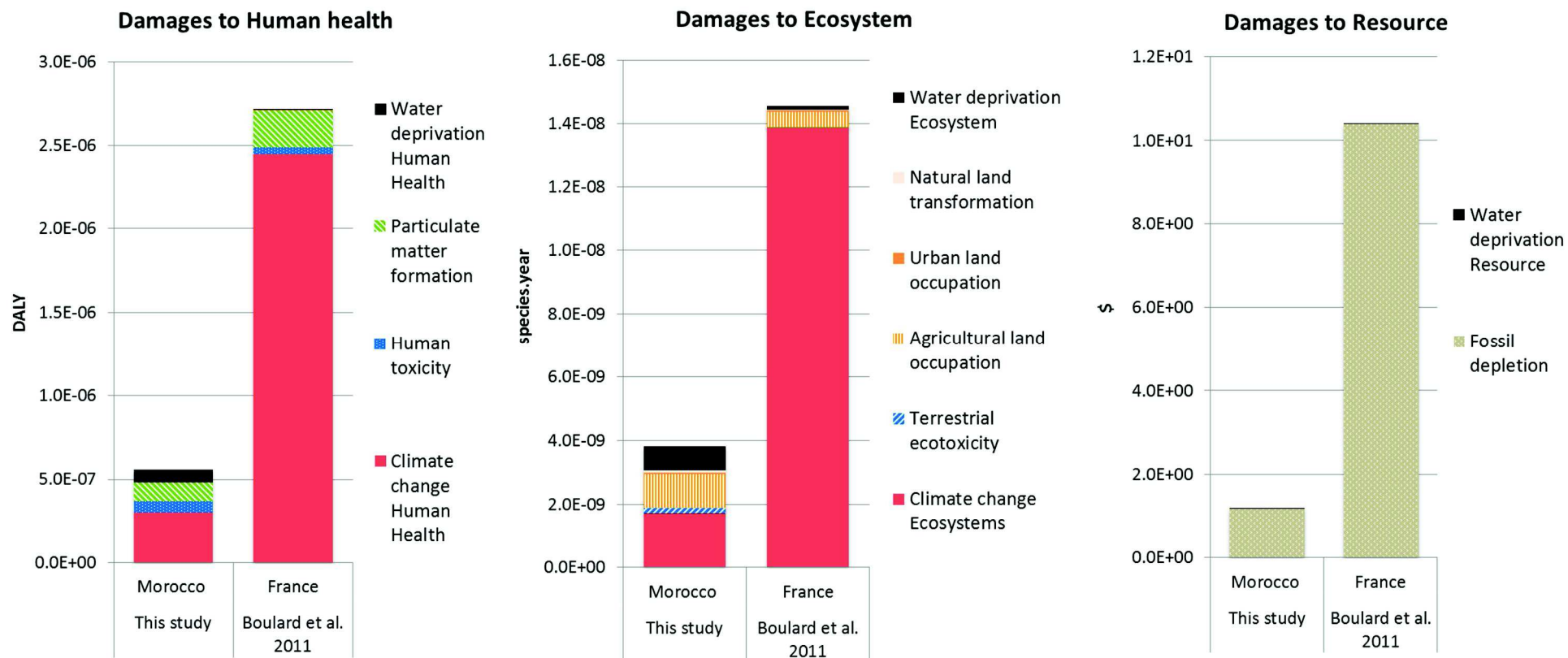


Figure 3. Human Health (DALY), Ecosystems (species. year) and Resources (\$) damages comparison for 1 kg Moroccan or French tomatoes at farm gate. Impact contribution to the total damages. The negligible contributors are not shown on the legend: Ionising radiation, Photochemical oxidant formation and Ozone depletion for Human Health, Terrestrial acidification, Freshwater eutrophication, Freshwater and marine ecotoxicities for Ecosystems, and metal depletion for Resources.

## LCA of local and imported tomato: An energy and water trade-off - Supplementary information

Sandra Payen<sup>1,2,\*</sup>, Claudine Basset-Mens<sup>2</sup>, Sylvain Perret<sup>3</sup>

<sup>1</sup>ADEME, (Agence de l'Environnement et de la Maîtrise de l'Energie), 20 avenue du Grésillé - BP 90406 49004 Angers, France

<sup>2</sup>CIRAD, UPR Hortsys, ELSA, 34398 Montpellier, France

<sup>3</sup>CIRAD, UMR G-Eau, 34398 Montpellier, France

### A1. Physical allocation at the packaging station

The energy (fuel and electricity) and water consumption of the packaging station were allocated to tomatoes using a physical approach: 71.9% of the vegetables packed at the station are tomatoes destined for exportation; the rest is outsized tomato sold on the local market (19.5%), other vegetables (8.3%), or loss (0.3%).

### A2. Data quality assessment

Primary and secondary data quality was assessed following the ILCD requirements:

**Table A2a. Primary data quality assessment against the ILCD data quality indicators, based on a self-evaluation. Primary data were obtained from a specific field survey**

Indicator	Definition	Dataset evaluation	Quality rating	Quality level
Technological representativeness	Degree to which the data set reflects the true population of interest regarding technology	The technology for tomato production is highly standardised within the population of farms exporting to Europe.	1	very good
Geographical representativeness	Degree to which the data set reflects the true population of interest regarding geography	The farms studied are located in the Souss-Massa area producing 85% of the product studied (tomato exported to France)	1	very good
Time-related representativeness	Degree to which the data set reflects the true population of interest regarding time / age of the data	Data were collected on farms specifically for the study and correspond to the practices from 2009 to 2011	1	very good
Completeness	Share of (elementary) flows that are quantitatively included in the inventory	- All known inventory flows were quantified	1	very good

Indicator	Definition	Dataset evaluation	Quality rating	Quality level
Precision uncertainty	/ Measure of the variability of the data values for each data expressed	- Data are based on farmer's records thus have a low uncertainty - Data are calculated over a 3 crop cycle average - Variability between farms and years are quantified with ranges	2	good
Methodological appropriateness and consistency	The applied LCI methods and methodological choices (e.g. allocation, substitution, etc.) are in line with the goal and scope of the data set	- Allocation choices are in line with the function studied - Best available field emissions estimation methods were selected accounting for the system specificities	2	good
<b>Overall data quality rating</b>			<b>1.6</b>	<b>High quality</b>

Table A2b. Secondary data quality assessment against the ILCD data quality indicators, based on a self-evaluation. Secondary data were obtained from the Ecoinvent database.

Secondary data based on Ecoinvent			Data quality indicator (ILCD 2010)				
			Technological, geographical and time-related representativeness	Completeness	Precision / uncertainty	Methodological appropriateness and consistency	Overall data quality rating
Input manufacture	Fertilisers	Mainly imported from Europe	"Very good" for all inputs imported from Europe. "Good" for inputs manufactured in Morocco (e.g. a few greenhouse and packaging components)	""Very good" because datasets in the Ecoinvent database include all known exchanges	A "fair" quality was given as a default because relative standard deviation for the overall environmental impact could not be quantified	"Very good" because all data in the Ecoinvent database are reviewed against data quality guidelines in accordance with ILCD handbook	<b>basic quality</b>
	Pesticides	Mainly imported from Europe					
	Greenhouse components	Imported from Europe, except the wood frame					
	Packaging	Imported from Europe or manufactured in Morocco					
Input transport	Transport mode	Transport by truck or boat					
	Distance	Additional transport from Europe to farm included					
Electricity mix		Morocco 2007					
Diesel	Manufacture	Europe					
	Combustion	SAEFL 2000					

### A3. Scenarios analysis

**Table A3. Climate change, eutrophication, acidification and water deprivation for the scenarios analysed, expressed as a percentage of the results from the reference scenario**

Parameter	Reference scenario	Simulated scenarios	Climate change	Freshwater eutrophication	Marine eutrophication	Terrestrial acidification	Water deprivation Pfister et al.
Allocation method	Physical allocation	Economic allocation	9%	15%	6%	9%	20%
Truck age	EURO 3	EURO 4	0%	0%	-8%	-7%	0%
Transport mode	1/3 Freight ship; 2/3 Truck	100% Truck	13%	4%	8%	0%	0%
		100% Freight ship	-27%	-8%	-16%	0%	-1%
Truck load	11,68 tons (Ecoinvent)	24 tons (Full load)	-18%	-4%	-14%	-15%	0%
Waste treatment	Re-use	100% incinerated	6.7%	0.1%	0.4%	0.2%	0.1%
Nitrate emission calculation	No emissions	20% of N fertilisers	0%	0%	+303%	0%	0%

### A4. Environmental impacts of the Moroccan off-season tomato production and delivery: additional impact categories

**Table A4. Contribution analysis of 1 kilogram of tomatoes at St Charles market gate, ReCiPe midpoint impact assessment method (Hierarchist), including the water characterisation factors of Pfister et al. (2009).**

Impact Category	Unit	Total	Nursery	Greenhouse tomato production	Packaging	Transportation to France
Ozone depletion	kg CFC-11 <sub>eq</sub>	6.20E-08	5.54E-10	1.33E-08	1.04E-08	3.78E-08
Human toxicity	kg 1,4-DB <sub>eq</sub>	1.54E-01	1.86E-03	9.88E-02	2.74E-02	2.58E-02
Photochemical oxidant formation	kg NMVOC	3.22E-03	5.13E-05	6.40E-04	3.08E-04	2.22E-03
Particulate matter formation	kg PM10 <sub>eq</sub>	1.18E-03	1.49E-05	4.13E-04	1.22E-04	6.33E-04
Ionising radiation	kg U235 <sub>eq</sub>	7.80E-02	1.84E-03	3.96E-02	1.87E-02	1.79E-02
Urban land occupation	m <sup>2</sup> .year	6.63E-03	2.08E-04	1.49E-03	3.01E-03	1.92E-03
Natural land transformation	m <sup>2</sup>	1.61E-04	1.49E-06	3.35E-05	3.54E-05	9.08E-05

## A5. Analysis of water damages

The end-point damages of water use alone were greater in the case of the Moroccan tomato than for the French tomato for the three areas of protection (Fig. A5). Freshwater deprivation in France did not cause damages to Human Health because this country is not vulnerable to malnutrition: the human development factor (based on the Human Development Index HDI) was nil. In contrast, with a HDI of 0.6, Morocco is vulnerable to malnutrition: the freshwater deprivation caused damage to Human Health with  $7.64 \cdot 10^{-8}$  DALY.kg<sup>-1</sup>. The water use in the French tomato caused less damage to Ecosystems ( $1.05 \cdot 10^{-10}$  species.year.kg<sup>-1</sup>) than the water use in the Moroccan tomato ( $7.62 \cdot 10^{-10}$  species.year.kg<sup>-1</sup>). The Ecosystems damage factor is the ratio between the Net Primary Production limited by water availability and the precipitation, meaning that the ecosystems are more vulnerable to water deprivation and/or the annual rainfall is less in Morocco. The damages to Resources caused by water use in the French tomato were negligible ( $8.51 \cdot 10^{-4}$  \$.kg<sup>-1</sup>) compared with the damages caused by water use in the Moroccan tomato ( $2.44 \cdot 10^{-2}$  \$.kg<sup>-1</sup>). This is because the fraction of water consumption that contributes to depletion is higher in Morocco.

The impacts of water consumed during background processes (e.g., fertiliser production) were non-negligible for the French tomato (Fig. A5). Indeed, although this background water represented only 9.5% of the total water use, its impact was calculated by multiplying by the global average WSI, which is higher than the French WSI. This outcome shows the importance of localising the water withdrawals to properly assign the region-specific WSI and avoid the use of country or global averaged WSI.

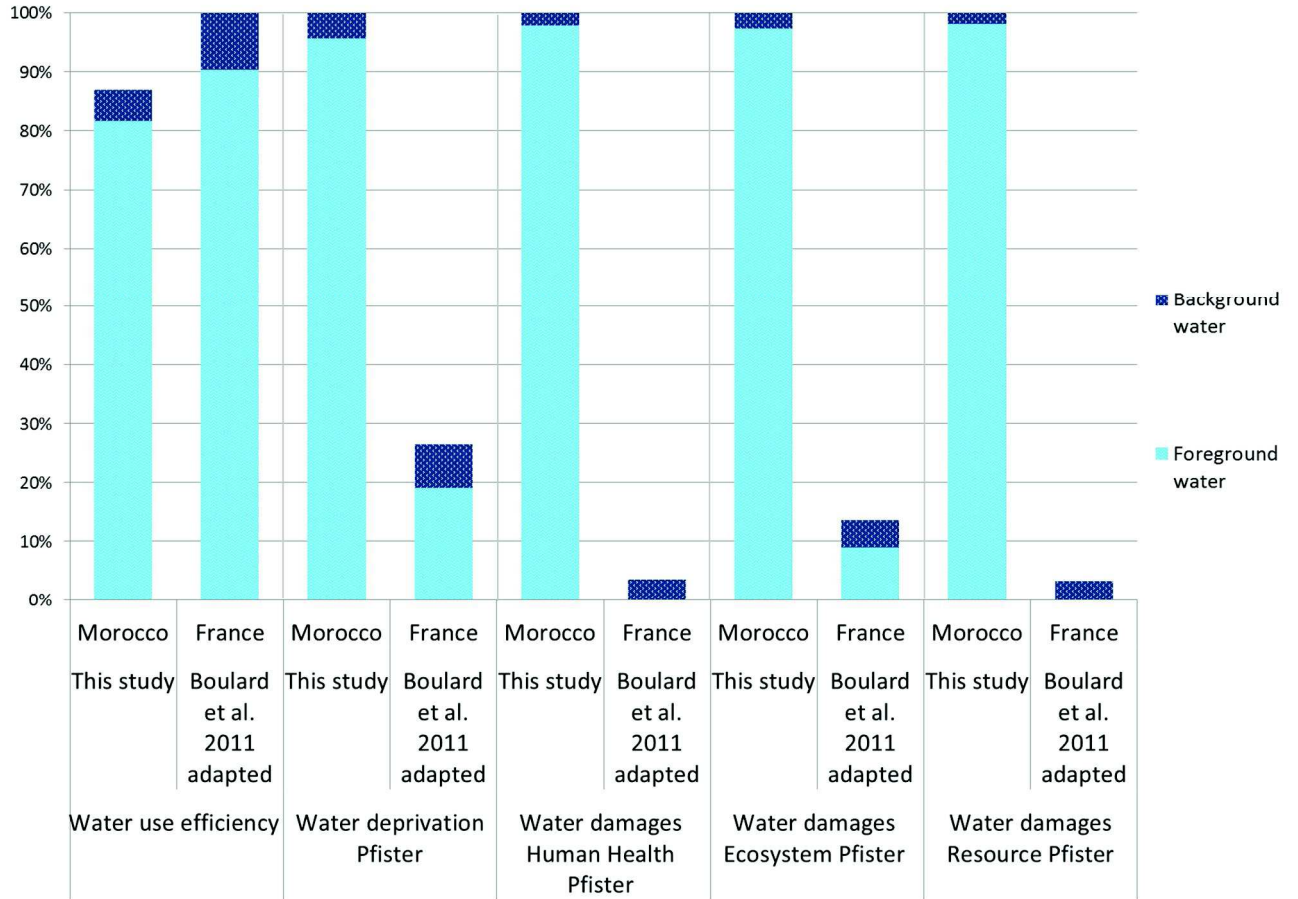


Figure A5. Comparison of water use efficiency (litres of water consumed per kg tomato), water deprivation ( L H<sub>2</sub>Oeq) and water damages to Human Health(DALY), Ecosystems (species. year) and Resources (\$) for the Moroccan, and French tomatoes. The contributions of the freshwater consumed for the foreground and background processes are distinguished.

## Chapter 1 - Supplementary information - How to assess the impacts associated with water use in agricultural LCA?

### Description of methods addressing the impacts and/or damages of water use in life cycle impact assessment

For each reference, the indicator description, impact pathway covered, characterisation factor components and inventory requirements are specified. Note that all methods require a regionalized inventory.

	Indicator	Reference	Impact pathway	Characterisation factor	Inventory, regionalized
<b>Midpoint</b>	Water scarcity	(Frischknecht et al. 2008)	Water deprivation from water consumption	Distance-to-target principle (similar to withdrawal-to-availability ratio)	Volume of water consumed or withdrawn
	Water scarcity	(Pfister et al. 2009)	Water deprivation from water consumption	Withdrawal-to-availability ratio, logistic function	Volume of water consumed
	Water scarcity	(Pfister and Bayer 2014)	Water deprivation from water consumption	From Pfister et al. 2009, monthly	Volume of water consumed monthly
	Water scarcity (Ecosystem-oriented)	(Milà i Canals et al. 2008)	Freshwater ecosystem impact: Water deprivation impact on freshwater ecosystem from water consumption	Water resources per capita (Falkenmark et al. 1989) or water use per resource (Raskin et al. 1997) or environmental water scarcity (Smakhtin et al. 2004)	Volume of water consumed and rainwater stored in soil
	Water scarcity (Resource-oriented)	(Milà i Canals et al. 2008)	Freshwater depletion: Water deprivation impact on freshwater resource from water consumption	Abiotic resource depletion potential	Volume of water consumed from over-abstracted aquifers and withdrawn from fossil aquifer
	Water scarcity	(Hoekstra et al. 2012)	Water deprivation from water consumption	Consumption-to-availability ratio, direct function	Volume of water consumed
	Water scarcity	(Loubet et al. 2013)	Water deprivation in downstream subwatersheds from water consumption	Weighting sum of consumption-to-availability ratio of downstream subwatersheds	Volume of water withdrawal in subwatershed A, and released in subwatershed B
	Water availability (Ecosystem-oriented)	(Bayart et al. 2014)	Water deprivation from water consumption and quality degradation	Withdrawal-to-availability ratio, distance to target for water pollution	Volume and quality of input and output flows of water
	Water availability (Human-oriented)	(Boulay et al. 2011)	Water deprivation from water consumption and quality degradation	Consumption-to-availability ratio, quality specific, logistic function	Volume and quality of input and output flows of water

	<b>Indicator</b>	<b>Reference</b>	<b>Impact pathway</b>	<b>Characterisation factor</b>	<b>Inventory, regionalized</b>
	Water scarcity	(Berger et al. 2014)	Water deprivation from water consumption	Consumption-to-availability ratio, logistic function	Volume of water withdrawn and released (water recycling rate per watershed provided by authors)
	Water scarcity	(WULCA 2015)	Available Water Remaining: Water deprivation from water consumption	Inverse of unused water remaining in a basin (availability minus demand of humans and aquatic ecosystems)	Volume of water consumed
	Water scarcity (Human-oriented)	(Motoshita et al. 2014)	Agricultural water scarcity causing food production shortages due to water consumption	Combine: water physical vulnerability (Pfister et al. stress index used as a default), agricultural irrigation demand and food stock compensation capacity	Volume of water consumed
<b>Endpoint</b>	Resources	(Pfister et al. 2009)	Water depletion caused by water overuse: surplus energy to make water resource available	Withdrawal-to-availability ratio and energy required for seawater desalination	Volume of water consumed
	Ecosystem: Terrestrial species loss	(Pfister et al. 2009)	Terrestrial species loss due to water use	Fraction of net primary production limited by water availability	Volume of water consumed
	Ecosystem: Thermal pollution	(Verones et al. 2010)	Impacts on species due to an increased temperature of effluent of a nuclear power plant in Switzerland (degradative use of water)	Fate factor : model of residence time of heat emissions in the river. Effect factor: loss of aquatic species diversity per unit of temperature increase	Heat energy and water volume discharged
	Ecosystem: Biodiversity in wetlands	(Verones et al. 2013a; Verones et al. 2013b)	Biodiversity loss in wetland due to water consumption	Fate factor: Area changes of surface water-fed and groundwater-fed wetlands. Effect factor: number of species lost per wetland area loss	Volume of surface and groundwater consumed
	Ecosystem: Aquatic species loss	(Hanafiah et al. 2011)	Freshwater fish species loss due to water consumption	Fate factor: relate water consumption to reductions in river discharge (one-to-one relationship). Effect factor: relationship between species richness for whole river basins to average discharge at the mouth of the basins	Volume of water consumed
	Ecosystem: Aquatic species loss	(Hanafiah et al. 2013)	Aquatic species loss due to the introduction of exotic Species (degradative use of water)	Fate factor: change in fraction of exotic species due to a change in the transportation of goods. Effect factor: impact of exotic species on native freshwater species richness	kg of transported goods by inland shipping



Indicator	Reference	Impact pathway	Characterisation factor	Inventory, regionalized
Ecosystem: Terrestrial species loss	(van Zelm et al. 2011)	Terrestrial species loss from groundwater table lowering due to water use	Fate factor: amount of time required for groundwater replenishment. Effect factor: groundwater level response curves of potential plant species richness	Volume of groundwater withdrawn
Ecosystem: Aquatic species loss	(Tendall et al. 2014)	Aquatic species loss due to river water consumption	Region-specific species-discharge relationship for Europe (inspired from Hanafiah et al. 2011)	Volume of water consumed
Ecosystem: Aquatic species loss	(Amores et al. 2013)	Aquatic species loss due to groundwater consumption causing salinity increase in a wetland in Spain (consumptive use leading to quality degradation of water)	Fate factor: change in salt concentration in the wetland due to a change in irrigation groundwater consumption. Effect factor: change in potentially affected fraction of native wetland species due to salinity increase	Volume of groundwater consumed
Ecosystem: Thermal pollution	(Pfister and Suh 2015)	Impacts on freshwater ecosystem due to thermal emission in the USA (degradative use of water)	Fate factor (local and downstream): temperature change in a volume of water during the residence time of heat emission. Effect factor: function of ambient river temperature, temperature rise and temperature tolerance intervals of different species	Heat emission rate from cooling
Human Health: Malnutrition	(Pfister et al. 2009)	Malnutrition damage due to water deprivation for agricultural users (lack of irrigation water)	Combine: scarcity indicator, agricultural users' share of water use, human development factor for malnutrition, per-capita water requirements to prevent malnutrition, damage caused by malnutrition	Volume of water consumed
Human Health: Malnutrition	(Motoshita et al. 2010)	Malnutrition damage due to agricultural water scarcity	relationship between agricultural water use, crop productivity and the undernourishment damage related to the change of food consumption	Volume of water consumed
Human Health: Infectious disease	(Motoshita et al. 2011)	Infectious disease damage due to domestic water scarcity	Correlate oral intake of unsafe water with water scarcity	Volume of water consumed
Human Health: Malnutrition	(Motoshita et al. 2014)	Malnutrition damage due to agricultural water scarcity	Combine: food production loss assessment, food supply shortage assessment and health damage Assessment. Improve previous models by incorporating economic adaptation capacity and the international food	Volume of water consumed

Indicator	Reference	Impact pathway	Characterisation factor	Inventory, regionalized
Human Health: Malnutrition and water-related diseases	(Boulay et al. 2011)	Malnutrition from water deprivation for agricultural users and fisheries, and water-related diseases associated with a lack of water for domestic use, due to water degradation and consumption	trade. Fate factor: water availability indicator (midpoint) Exposure factor: affected user by the loss of water functionality and adaption capacity. Marginal: 100% of water use will affect agriculture OR Distribution: all users affected proportionally to their use Effect factor: Agriculture and fisheries: malnutrition damages. Domestic: lack of hygiene and sanitation damages Consideration impacts associated with compensation	Volume and quality of input and output flows of water

## Water scarcity and availability indices

Based on Brown et al. (2011) and Kounina et al. (2013)

	Article	Index name	Definition	Scale	Comment
Indices Based on Human Water Requirements	(Falkenmark et al. 1989)	Water resource per capita (WRPC)	m <sup>3</sup> per capita Fraction of the total annual runoff available for human use.	country	The most widely used measure of water stress
	(Gleick 1996)	Basic Human Water Requirements	measurement of the ability to meet all water requirements for basic human needs	country	Basic water requirement (BWR) for drinking, cooking, bathing, and sanitation and hygiene = 50L/person/day Estimate the population by country without access to this BWR
	(Ohlsson 2000)	Social water scarcity index (SWSI)	Falkenmark index / human development index (HDI)	country	Highlight the importance of a society's social adaptive capacity facing the challenges of water scarcity
	(Yang et al. 2003)	Water Resources Availability and Cereal Import	In nearly all the countries < the water-deficit threshold, => increase in per capita cereal import (to compensate for the water deficit)	country	Inverse relationship between availability of land resources and cereal import. Doesn't take in to account the use of non-renewable groundwater
Water Resources Vulnerability Indices	(Raskin et al. 1997)	Water resources vulnerability index (WRVI)	Withdrawal to availability ratio Ratio of total annual withdrawals to available water resources		Made up of 3 sub-indices: (i) a use-to-resource ratio sub-index (similar to the criticality ratio) (ii) a coping capacity sub-index (iii) a reliability sub-index
	(Alcamo et al. 2000)	Criticality ratio (CR)	Withdrawal to availability ratio : $CR_i = \frac{\sum_j WU}{WA_i}$ 0 < CR < 1 i= watershed; j=users: industry + agriculture + households	watershed or country	Water availability refers to the renewable water resources generated inside the entity of interest (river discharge and the groundwater recharge)
	(Alcamo et al. 2007)	Criticality index (CI)	Withdrawal to availability ratio: combines 2 factors: the criticality ratio and the water availability per capita, into a single indicator of water vulnerability	watershed or country	Table with scores
	(Vörösmarty et al. 2005)	The Index of Local Relative Water Use and Reuse	Index of local relative water use : DIAn/QCn water reuse index : $\sum DIAn/QCn$	8km cells n	Water use = water withdrawals for the domestic (D), industrial (I), and agricultural (A) sectors. The locally generated discharge= locally generated runoff x cell area; the river corridor discharge is the sum of all local discharges (QC)
	(Chaves and Alipaz 2007)	The Watershed Sustainability Index (WSI)	WSI (0-1) average of 4 indicators : WSI=H+E+L+P/4	watershed or basin < 2500	Incorporates hydrologic H, environmental E, life (human) L; and policy P indicators (Score calculated according to pressure

	Article	Index name	Definition	Scale	Comment
				km <sup>2</sup>	and state parameters)
	(McNulty et al. 2010)	The Water Supply Stress Index (WaSSI) (similar to the WTA)	Quantitatively assess the relative magnitude of water supply and demand WaSSI= WD <sub>x</sub> /WS <sub>x</sub> WaSSI(0-12)	USA : 8-digit USGS Hydrologic Unit Code	WD=water demand, WS=water supply, x= historic or future water supply and/or demand from environmental and anthropogenic sectors
	(IWMI 2008)	Physical and Economical Water Scarcity	Country “physically water scarce” = + than 75% of river flows are withdrawn for agriculture, industry, and domestic purposes Country “economically water scarce” = adequate renewable resources with - than 25% of water from rivers withdrawn for human, but need improvements in water infrastructure	country	Portion of renewable freshwater resources available for human requirements (accounting for existing water infrastructure), with respect to the main water supply
Indices Incorporating Environmental Water Requirements	(Sullivan 2002)	Water poverty index (WPI)	Reflect both the physical availability of water and the degree to which human populations are served by that water, subject to constraints imposed by the maintenance of ecological integrity		Incorporates ecosystem productivity, community, human health, and economic welfare. Approach critically dependent on the development of standardized weights for each variables
	(Smakhtin et al. 2004)	Assessing Water Resource Supplies Using the Water Stress Indicator: Water Stress Indicator (WSI)	Withdrawal to availability ratio: WSI = Withdrawals/ (MAR–EWR) Mean annual runoff (MAR) = proxy for total water availability. Environmental water requirements (EWR) = % of long-term mean annual river runoff that should be reserved for environmental purposes	River basin	Recognizes environmental water requirements as an important parameter
Sensitivity index	(Döll 2009)	water scarcity indicator	Ratio of consumptive water use to statistical low flow Q90. This index is then used in addition to an indicator for dependence of water supply on groundwater and the Human Development Index to form a sensitivity index	0.5 by 0.5° grid cells on a monthly base	Vulnerability to the impact of decreased groundwater recharge in the 2050s are derived combining this sensitivity index with per cent groundwater recharge decrease
LCA and Water Footprint	(Pfister et al. 2009)	Water Scarcity Index (WSI)	Based on a withdrawal to availability ratio (WTA) factor calculated as a criticality ratio (Alcamo, et al., 2000) which differentiates watersheds with strongly regulated flows $WTA_i = \frac{\sum_j WU_{ij}}{WA_i}$ WA <sub>i</sub> =annual freshwater avail ability WU <sub>ij</sub> =withdrawals for different users j, for each watershed i Water stress index (WSI) is adjusted to a logistic function, 0.01<WSI< 1:	Watershed	A modified WTA (WTA*) is calculated to differentiates watersheds with strongly regulated flows WSI index is a characterization factor for water consumption

Article	Index name	Definition	Scale	Comment
		$WSI = \frac{1}{1 + e^{-6.4 \cdot WTA \cdot \left(\frac{1}{0.01} - 1\right)}}$		
(Pfister and Bayer 2014)	Monthly Water Scarcity Index (WSI monthly)	<p>Monthly variability of water availability s*month is excluded as it is explicitly covered by applying monthly WSI. Only the inter-annual variability is accounted for by the geometric standard deviation (s* year)</p> <p>WSI adjusted : the exponent factor - 6.4 -&gt; -9.8</p> $WSI_{monthly} = \frac{1}{1 + e^{-9.8 \cdot WTA'_{monthly} \cdot \left(\frac{1}{0.01} - 1\right)}}$ $WTA'_{monthly} = WTA_{monthly} \cdot s'_{year}$	watershed	<p>WTA monthly is determined by aggregating data from the 0.5 arc-degree model by Fekete et al., (2002) to watershed level and deriving factors of monthly WTA to annual WTA for each month. Then, these monthly factors are applied to the annual data from “WaterGAP” Alcamo et al., (2003) which are used in the original WSI to derive values for WTA monthly that are consistent with the annual factors.</p>
(Frischknecht et al. 2008)	Ecological water scarcity	<p>Withdrawal to availability ratio</p> <p>Ecological water scarcity is defined for each individual watershed area. Six scarcity classes are proposed to simplify life cycle inventory modelling</p>	watershed	The Swiss ecological scarcity method is a “distance-to-target” method
(Hoekstra et al. 2011)	Green, blue and grey water scarcity index	<p>Consumption to availability ratio</p> <p>Green=total green water footprint / green water availability</p> <p>Blue=blue water footprint /blue water availability</p> <p>Grey=total grey water footprint /runoff from that catchment</p>	catchment	blue water availability accounts for environmental water needs by subtracting from the total runoff the presumed flow requirement for ecological health (if=100%; all blue water has been consumed)
(Hoekstra et al. 2012)	Monthly Blue water Environmental flow requirement	<p>Consumption to availability ratio</p> <p>The monthly blue water availability in a river basin in a certain period was calculated as the ‘natural runoff’ in the basin minus ‘environmental flow requirement’</p>	River basin	Average monthly blue water footprints per river basin for the period 1996–2005 have been derived from the work of Mekonnen and Hoekstra 2011
(Loubet et al. 2013)	WDCF	<p>Local water scarcity in a sub-river basin, based on a consumption to availability ratio:</p> $CTA_i = \frac{\sum_{k=1}^i tWC_k}{WA_i}$ <p>tWCi= the total local water consumption, plus upstream water consumption: tWC 1 to (i-1)</p> <p>WAI= local water availability (runoff: Global Runoff Data Center)</p>	SUB-watershed	<p>Characterization factor for water deprivation in sub-river basin i is the weighted sum of all downstream CTA ratios:</p> $CF_{WD,i} = \frac{1}{\bar{P} \cdot N_{down}} \sum_{j=i}^n (CTA_j \cdot p_j)$
(Boulay et al. 2011)	scarcity parameter of midpoint level and endpoint model for human	<p>Scarcity parameter <math>\alpha^*</math>, based on a consumption to availability ratio, specific to the water origin:</p>	Watershed	The midpoint Water Stress Indicator (WSI), is calculated at the watershed scale and can be used for all three endpoint categories; human health,

Article	Index name	Definition	Scale	Comment
	health	$\alpha_{\text{surface},i}^* = \frac{CU \times (1 - f_g)}{Q90} \times \frac{1}{P_i}$ $\alpha_{\text{GW},i}^* = \frac{CU \times f_g}{GWR} \times \frac{1}{P_i}$ $\alpha_{\text{Rain}} = \frac{CU}{(Q90 + GWR)} \times \frac{1}{P_r}$ <p>CU= consumptive use            Q90= the statistical low flow,            fg the fraction of usage dependent on groundwater,            GWR=renewable groundwater resource available            Pi= the proportion of available water that is of category i.            The stress index (<math>\alpha_i</math>) is then modelled based on accepted water stress thresholds</p>		<p>ecosystems and resources:            WSI expresses the midpoint result in m<sup>3</sup> equivalent of water:  <math>WSI = \sum_i(\alpha_i \times V_{i, in}) - \sum_i(\alpha_i \times V_{i, out})</math>            With:            - <math>\alpha_i</math> the stress index of water category            - <math>V_i</math> (in and out) the volumes of water category i entering and leaving the process or product system</p>
(Milà i Canals et al. 2008)	WSI	<p>Withdrawal to availability ratio            The water scarcity indicator have to be chosen between:</p> <ul style="list-style-type: none"> <li>- Falkenmark et al.'s water resources per capita (Falkenmark et al. 1989),</li> <li>- Raskin et al.'s water use per resource (WUPR) (Raskin et al. 1997)</li> <li>- Smakhtin et al.'s environmental water scarcity (Smakhtin et al. 2004).</li> </ul>	River basin	The water uses considered are all evaporative uses of freshwater (including evaporated irrigation water, cooling water, evaporated water from dams and reservoirs, etc.). Milà I Canals et al. (2009) acknowledge that it can lead to an underestimation of local effects, when non-evaporative uses are considered to have no impact on freshwater ecosystem impact
(Berger et al. 2014)	WDI	<p>Based on a consumption to availability ratio:</p> $CTA^* = \frac{C}{A + SWS} \cdot AF_{GWS}$ $WDI = \frac{1}{1 + e^{-40 \cdot CTA^*} \left( \frac{1}{0.01} - 1 \right)}$ <p>C=Annual water consumption            A= annually renewable freshwater volumes within the basin            SWS=annually usable surface water            AF<sub>GWS</sub>=adjustment factor to account for ground water stocks (WHYMAP)</p>	Watershed	<p>The midpoint is the risk of freshwater depletion (RFD):</p> $RFD = \sum (WC_{\text{eff},n} \cdot WDI_n)$ <p>WF<sub>eff, n</sub>= effective water consumption in each basin, considering the effects of atmospheric evaporation recycling</p>

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## **Chapter 2 - Supplementary information - Salinisation impacts in Life Cycle Assessment: a review of challenges and options towards their consistent integration**

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*Sandra Payen, Claudine Basset-Mens, Montserrat Núñez, Stéphane Follain, Olivier Grünberger, Serge Marlet, Sylvain Perret, Philippe Roux*

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**Table S1. Evaluation of salinisation impacts assessment methods against the criteria defined in the International Reference Life Cycle Data System Handbook procedure proposed by the Institute for Environment and Sustainability from the European Commission (JRC-IES 2011).**

Criteria	Sub-criteria	Feitz and Lundie 2002 MIDPOINT	Amores et al. 2013 ENDPOINT (no mid-point)	Leske and Buckley 2003; 2004a; 2004b near-ENDPOINT	Zhou et al. 2013 ENDPOINT
<b>Complete-ness of scope</b>	<i>The impact indicator covers the majority of impact mechanisms and relevant elementary flows for the AoP Human Health, Ecosystems, and Natural Resources</i>	- The soil salinisation potential is a mid-point indicator focusing on one salinisation pathway: soil sodisation and salinisation from poor irrigation practices. Within the soil salinisation pathway, it does not account for waterlogging - This indicator is relevant for agricultural LCA only	- The biodiversity impact from salinity increase is an end-point indicator for Ecosystems. Focus on marine intrusion salinisation pathway, and more specifically on a specific marine intrusion context: groundwater use inducing seawater intrusion in the groundwater-fed wetland. Within the marine intrusion pathway, it does not consider groundwater salinisation - <i>A priori</i> relevant for irrigated systems only, but can be also applied to ground-water consuming systems	- The salinity impact indicator is a near endpoint indicator accounting for potential effects on aquatic ecotoxicity, materials, natural wildlife, livestock, aesthetic effects, natural vegetation and crop. Consider both water and soil salinisation but focus on salinisation induced by deposition of ions (salts release in a given compartment), does not consider salinisation induced by a land use change or a saline intrusion. - This indicator is not specific to agricultural LCA	- The aquatic ecotoxicity of brine disposal is an end-point indicator for ecosystems. - Focus on the effects on aquatic Ecosystems of brine disposal from seawater desalination plants.
	<i>The characterisation model is adaptable to spatial and temporal explicit evaluation</i>	The CF is specific to the irrigation water composition. The characterisation model is valid for Australian red-brown earth, and should be adapted for other soil types (see below).	The CF is site-specific The characterisation model is valid for the Albufera de Adra wetland area in Spain, and should be re-developed for application in other location (see below)	The CF is country-specific The characterisation model is valid for South Africa, adaptation for other location require the use of another model (see below)	The CF is not spatially explicit (it is a constant)
	<i>Global geographical validity preferable, separate validity for Europe beneficial</i>	Geographical validity not clearly defined, but soil-dependant: - depends on the validity domain of the electrolyte threshold curve which « may not be appropriate for some soils », - the estimation of the Sodium Adsorption Ratio of	The geographical validity is limited to the specific case study: - The fate factor is based on water and salts balance relying on the specific hydrologic functioning of the wetland and local hydro-climatic parameters. The same calculation approach can be adopted for similar wetland. - The effect factor is based on	The geographical validity is limited to South Africa: - The fate factor is calculated with a South African catchment atmospheric deposition-hydrosalinity model: calculate the predicted environmental concentration of salts in each compartment. - The effect factor is based on the	The geographical validity is not specified but assumed to be global since the characterisation factor is a constant, not geographically specific.

Criteria	Sub-criteria	Feitz and Lundie 2002 MIDPOINT	Amores et al. 2013 ENDPOINT (no mid-point)	Leske and Buckley 2003; 2004a; 2004b near-ENDPOINT	Zhou et al. 2013 ENDPOINT
		the soil drainage water is assumed for an Australian red-brown earth.	specific species native to the Albufera de Adra wetland and can be applied to other wetlands with similar species composition.	South African Water Quality Guidelines to determine the no-effect concentration	
	<i>When empirical data is used, double counting is avoided</i>	Double counting with Leske et al. 2003, 2004a &b, but not the same geographical location	No double counting with other impact methods	Double counting with Feitz et al. 2002, but not the same geographical location	No double counting with other impact methods
<b>Environmental relevance</b>	<i>All critical parts of the environmental mechanism describing the cause-effect chain are included with acceptable quality given current scientific understanding</i>	Characterisation model based on threshold electrolyte concentration concept that predicts the SAR/EC ratio at which soil will potentially disperse. This is a relatively ancient approach but very common and generally accepted Account for the irrigation water composition (but no balance is done) Do not account for: soil type (crucial regarding its texture), fertiliser load (particularly if fertirrigation), climate, accumulation of other (from Na+) toxic ions, Besides, the quality of the soil solution is buffered by slow physico-chemical mechanisms occurring over several years (Condom et al. 1999), not accounted for in the method.	The characterisation factor aim to model the complete cause effect chain from groundwater use to salinity impact on biodiversity. But the fate and effect factor calculation are simplifying the mechanisms: The effect factor is linear: calculated as the average gradient at the 50% hazardous concentration but does not account for ambient concentration The Species Sensitivity Distribution are not based on EC50s describing the same effect (e.g., survival or growth inhibition) The fate factor is not utilizing any model and is based on water and salt balance equations	The fate model is an atmospheric deposition-hydrosalinity model for a “unit South African catchment”: the land use distribution is confined to one single urban area, one single rural area and one single rural agricultural area. The model predicts environmental concentrations in all the compartments relevant to the calculation of salinity potentials: the atmospheric deposition model predicts salt deposition rates, the rainfall-runoff model predicts the soil moisture and the river flow, and the salt transport model predicts the soil moisture and river salt concentrations. Originally developed by (Pitman et al. 1973) the model was later expanded to include salinity by Herold (1981). Do not account for: soil type (only natural and agricultural soils are distinguished), land cover type -Effect factor:	The characterisation model is based on a “whole effluent approach” for salinity group (Cl <sup>-</sup> , Na <sup>+</sup> , SO <sup>42-</sup> , Mg <sup>2+</sup> , Ca <sup>2+</sup> , K <sup>+</sup> , HCO <sup>3-</sup> ) (Next to chemical-specific approach for other groups of influential chemicals) The CFs are estimated based on a fate, exposure and effect model: The fate factor is the residence time of Cu <sup>2+</sup> , used instead of Na <sup>+</sup> ion which is the most persistent chemicals in the salinity group. But since the persistence time of Na <sup>+</sup> (210 million years) exceeds the range of the acute test (100 years), the residence time of the second most persistent chemical in the brine mixture is used. However, if Cu <sup>2+</sup> is in the brine mixture, it does not belong to the salinity group. The exposure factor is one, considering that 100% salts are dissolved into water. But this is not always the case. The effect factor is calculated based on a worst case scenario: the EC50 correspond to the salinity concentration threshold for acute

Criteria	Sub-criteria	Feitz and Lundie 2002 MIDPOINT	Amores et al. 2013 ENDPOINT (no mid-point)	Leske and Buckley 2003; 2004a; 2004b near-ENDPOINT	Zhou et al. 2013 ENDPOINT
				The salts effect factors are based on the Predicted No-Effect Concentrations, a conservative approach compared with the HC50, assuming that sensitivity of an ecosystem depends on the most sensitive species. Effect factors are not calculated as a function of the background salt concentration, except for aquatic ecotoxicity.	toxicity of brine on four phytoplankton, and refers to growth rate effects (Yoon and Park 2011). The EC50 values reported in this experiment varies from 40.2 to 78.7 g/L. This high variability of the EC50 and the limited number of species considered (all marine), warrant the need to use a HC50 based on a wider range of aquatic species. Recent publications in the field may now allow it according to Jin Zhou (personal communication)
<b>Scientific robustness &amp; Certainty</b>	<i>The critical part of the model including the parameters used in the model have been peer reviewed</i>	peer-reviewed	peer-reviewed	peer-reviewed	peer-reviewed
	<i>The model reflects the latest knowledge for the cause-effect chain (the critical links are covered)</i>	<i>Cf: environmental relevance</i>	<i>Cf: environmental relevance</i>	<i>Cf: environmental relevance</i>	<i>Cf: environmental relevance</i>
	<i>Indicators can be confirmed and verified against monitoring data, if available</i>	Indicators can <u>partially</u> be verified against monitoring data because rely on observable (soil structure) and measurable ([Na <sup>+</sup> ]) components	Indicators cannot be verified against monitoring data (expressed in potentially affected fraction of species)	Indicators can <u>partially</u> be verified against monitoring data : the CFs are based on kg TDS in a given compartment	Indicators cannot be verified against monitoring data (expressed in potentially affected fraction of species)
	<i>Uncertainty estimates of the indicators are provided, justified and reported in statistical terms</i>	Model uncertainty not provided	Model uncertainty is provided through the confidence interval and the standard error of CF.	Model uncertainty is provided through a sensitivity analysis of the fate model	Model uncertainty is provided through an assumption: the CF was assumed to have an uncertainty value of ±30%
	<i>The category indicator and characterisation models are science based</i>	The model is science based	The model is science based	The model is science based	The model is science based
<b>Documen-</b>	<i>The model documentation is</i>	The model documentation,	The model documentation,	The model documentation,	The model documentation,

Criteria	Sub-criteria	Feitz and Lundie 2002 MIDPOINT	Amores et al. 2013 ENDPOINT (no mid-point)	Leske and Buckley 2003; 2004a; 2004b near-ENDPOINT	Zhou et al. 2013 ENDPOINT
<b>tation &amp; Transparency &amp; Reproducibility</b>	<i>published and accessible This must support the development of new, consistent factors by third parties.</i>	characterisation model and results are published and available	characterisation model and results are published and available	characterisation model and results are published and available	characterisation model and results are published and available
	<i>Ability for third parties to freely generate additional, consistent factors and to further develop models</i>	The model can be further developed by third parties	The model can be further developed by third parties	The model can be further developed by third parties	The model can be further developed by third parties
	<i>The characterisation factors are straightforward to apply for general LCA practitioners and in most market-relevant LCA software tools</i>	CFs are not straightforward to apply and have to be calculated by the practitioner (rely on the irrigation water composition)	CFs are available only for the specific case study: the wetland Albufera de Adra in Spain	CFs are available only for South Africa.	CFs are straightforward to apply : it is a constant whatever the location of the brine discharge
	<i>Unit comparable with other impact categories</i>	Units (Na+ equivalent ) cannot be compared with other methods	The common end-point unit PAF can be compared with other methods	Units (kg TDS eq) cannot be compared with other methods	The common end-point unit PAF can be compared with other methods
<b>Stakeholder acceptance criteria</b>	<i>The indicator is easily understood and interpretable</i>	The indicator seems not easy to interpret for non-agronomist, but can be easily understood because it is based on physical mechanisms	The indicator seems easy to understand thanks to the unit (PAF), but not to interpret in details since it is at the end of the cause-effect chain	The indicator seems not easy to interpret because of its “hybrid” position between mid and endpoints.	The indicator seems easy to understand thanks to the unit (PAF), but not to interpret in details since it is at the end of the cause-effect chain
	<i>There is an authoritative body behind the general model principles like the IPCC model</i>	The model is not endorsed by an authoritative body	The model is not endorsed by an authoritative body	The model is not endorsed by an authoritative body	The model is not endorsed by an authoritative body
	<i>The principles of the model are easily understood by non-LCIA experts</i>	The principles of the model can be easily understood by non-LCIA experts	The principles of the model can be easily understood by non-LCIA experts	The principles of the model can be easily understood by non-LCIA experts	The principles of the model can be easily understood by non-LCIA experts
<b>Overall evaluation of stakeholders acceptance criteria</b>		Acceptance of the method among LCA practitioner has been limited with only one application of the method within 12 years	Acceptance of the method will have to be evaluated after further methodological development to make global characterisation factors available. The method has been applied once but in the same location in Spain	Acceptance of the method among LCA practitioner has been limited without application of the method within 10 years (except by the authors)	Acceptance of the method will have to be evaluated after a few years owing to the young age of the publication

## **S.2. Status of the AoP Resource: a fundamental on-going debate**

There is a need for consistently defining the AoP, especially the AoP Resource (Hauschild et al. 2013; Jolliet et al. 2014). This is paramount because this is impeding the definition of clear and non-overlapping impact pathways, compatible across different LCIA methods. But it is difficult to find a consensus on the status of the AoP which depends on the vision we have of sustainability and the underlying value framework (Adams 2006).

The status of the AoP Resource is crucial in the different viewpoints found in the literature. Three different status of the AoP Resource can be found. The first and most frequent case is when resources are not considered to have an intrinsic value but rather an instrumental value (Stewart and Weidema 2005). In this case, Resource is just an intermediate towards Human health and Ecosystem damages. This is consistent with the implicit trend of recent water use LCIA methods who neglect the AoP Resource (Kounina et al. 2013) and with the frequent focus on ecosystems services function of soil (Renouf et al. 2014). A second viewpoint illustrated by the IMPACT World+™ LCIA methodology is to consider an AoP Resource next to Human Health and Ecosystem but completed with ecosystem services damages. This is consistent with the Millennium Ecosystem Assessment (MEA 2005): Ecosystem services and Resources are functional for Human (and Ecosystems), but this is not consistent with the land use framework considering ecosystem services damage as an endpoint contributing to the three AoP (Koellner and Geyer 2013). A third viewpoint (e.g Bayart et al. 2010), is to restrict the AoP Resource to damages for future generations next to actual damages on Human health and Ecosystems. Indeed, including soil and water depletion in the AoP Resource emphasizes the importance to preserve these resources for future generations, which may be relevant in many (semi)-arid countries such as Australia (Renouf et al. 2014).

## **S.3. Average or marginal effects factors?**

An open research question is whether the effect factors should be derived following a marginal approach or an average approach. When adopting a marginal approach to derive the effect factor, we focus on marginal changes only, and there is little benefit in reducing pollution loads in context with high environmental pressure (i.e: at high concentration, the slope of the SSD curve is almost zero). Huijbregts et al (2011) suggest a possible change in paradigm. They recommend exploring the use of an average approach to derive effect factors; reflecting the average distance between the current state of emission and the preferred state of the environment (that can be “zero effect”). Following an average approach would allow LCIA to focus on reaching the preferable state of the environment defined by society, and not on marginal changes (Huijbregts et al. 2011). For both approaches, spatial-explicit models are of high relevance.



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### **Chapter 3 - Supplementary information - Inventory of field water flows for agri-food LCA: critical review and recommendations of modelling options**

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- ➡ Excel file with a description of the water inventory and agri-food databases

REFERENCE DATABASES	Available at	Crop	Spatial scale	Reference period	Water withdrawal from		Waste water	Water emitted to				Waste-water send to treatment	Yield
					Surfacewater	Groundwater		air		Ground-water	Surface water		
								Blue water consumption	Green water consumption				
Pfister et al 2011 - Environmental Impacts of Water Use in Global Crop Production	<a href="http://pubs.acs.org">http://pubs.acs.org</a>	160 crops and crop groups (accounting for 99.96% of the mass of total global crop production as reported by the production data from FAOSTAT)	Data available at country scale, with global coverage Calculation made at 5 arc-minutes resolution ~10km (for the yield)	2000 for the yields	n.a	n.a	n.a	<p><b>Calculations on a monthly basis</b>  <b>The expected blue water consumption is the ARITHMETIC mean of full and deficit blue water consumption:</b>                      - <b>Full irrigation blue water consumption=ET-Peff/yield or 0 if <math>ET \leq Peff</math></b>                      with <math>ET=kc*ET_0</math> (based on CROPWAT)  <b>ET is ETmax here : optimal irrigation is assumed</b>                      - <b>Deficit blue water consumption=% irrigated cropland*Full irri blue water consumption</b>  <b>Represent the lower margin of irrigation-water demand</b>                      % irrigated cropland: obtained by combining % cropland share (Ramankutty et al. 2008) and % irrigated area (Siebert et al. 2007)</p>	Not provided in the database. But can be deduced from: Total water consumption - blue water consumption	n.a	n.a	n.a	Effective yield in the year 2000 on a 5 arc-minutes grid (Monfreda et al. 2008)
Pfister et al 2014 - Monthly water stress: spatially and temporally explicit consumptive water footprint of global crop production	<a href="http://dx.doi.org/10.1016/j.jclepro.2013.11.031">http://dx.doi.org/10.1016/j.jclepro.2013.11.031</a>	Idem Pfister et al 2011	Idem Pfister et al 2011	Idem Pfister et al 2011	n.a	n.a	n.a	<p><b>Two modifications of Pfister et al 2011</b>                      - <b>GEOMETRIC mean of full and deficit blue water consumption</b>                      - <b>To put less weight on the lower limit when no irrigation is reported:</b> if deficit blue water consumption=0, expected blue water consumption=0.05*Full irri blue water consumption (based on the assumption that 0.25% of cropland area is irrigated even if no irrigation is reported)</p>	Not provided in the database. But can be deduced from: Total water consumption - blue water consumption	n.a	n.a	n.a	idem Pfister et al. 2011

REFERENCE DATABASES	Available at	Crop	Spatial scale	Reference period	Water withdrawal from			Water emitted to				Waste-water send to treatment	Yield
					Surfacewater	Groundwater	Waste water	air		Ground-water	Surface water		
								Blue water consumption	Green water consumption				
World Food LCA DataBase - Quantis, Agroscope Nemecek et al. 2014 <b>Database currently in development, will be completed in 2015</b>	Will be published by Ecoinvent	About 25 crops (including vegetables, fruits, cereals, oleaginous), and additional 14 crops depending on time and budget	Data available at country scale, for main net-exporting countries	Same as Pfister et al. 2011	<b>Water withdrawal</b> = Evapotranspiration from irrigation (blue water consumption) / Irrigation efficiency = ET irr/EF irr - Irrigation efficiency = field application efficiency x conveyance efficiency. Default values from FAO 1989, distinguish surface, sprinkler and drip irrigation technique. LEVEL 1: The average irrigation efficiency is calculated based on the shares of irrigation techniques in each country (International Commission on Irrigation and Drainage (ICID 2012) LEVEL 3: Data from literature on specific crop (not country-specific) LEVEL 4: Expert judgment or data from literature on specific crop produced in a specific country. It is unclear to which extend the different level will be applied to the crops  - Country-specific shares of ground and surface water and non-conventional sources used for irrigation from Siebert et al. 2010			<b>Water emitted to air in [m3/t]</b> ET irr = based on <b>expected blue water consumption of Pfister et al. 2011:</b> ARITHMETIC mean of full and deficit blue water consumption	Green water is not accounted for since it does not affect environmental impacts.	Water emitted to ground water: <b>20%</b> (Water withdrawal - Water emitted to air - water in crop) Based on Lérová & Pfister 2012	Water emitted to surface water: <b>80%</b> (Water withdrawal - Water emitted to air - water in crop) Based on Lérová & Pfister 2012	Wastewater send to treatment	LEVEL 1: Yield of fresh matter/ ha from FAOSTAT, 4 years average per product per country LEVEL 2: should distinguish conventional /organic, and refer to specific system parameters (soil and climate conditions, production techniques, crop rotation) LEVEL 3: Primary data + expert consultation + medium detailing LEVEL 4: Primary data + expert consultation + high detailing

REFERENCE DATABASES	Available at	Crop	Spatial scale	Reference period	Water withdrawal from		Waste water	Water emitted to				Waste-water send to treatment	Yield
					Surfacewater	Groundwater		air		Ground-water	Surface water		
								Blue water consumption	Green water consumption				
<p>WaterStat, Water Footprint Network, Enschede, the Netherlands</p> <p>Mekonnen and Hoekstra 2011: The green, blue and grey water footprint of crops and derived crop products, and associated report (Volume 1): Mekonnen and Hoekstra (2010).</p>	<a href="http://waterfootprint.org/en/resources/water-footprint-statistics/#Cp1">http://waterfootprint.org/en/resources/water-footprint-statistics/#Cp1</a>	146 crops and >200 derived crop products	Data available at country and sub-country scale, with global coverage Calculations made at 5 by 5 arc minute grid resolution	1996-2005	n.a	n.a	n.a	<p><b>The blue water footprints of crops [m3 per ton]</b>= total volume of blue water use /quantity of the production</p> <p><b>Daily water balance model:</b>  <b>Rain-fed crops:</b>                      Blue water use of rain-fed crop = 0  <b>Irrigated crops: (two water balance are combined)</b>                      Blue water use of irrigated crop = crop water irrigation requirement (assuming full irrigation: <math>ET_c = K_c * ET_0</math>) - Green water use of irrigated crop</p> <p><b>Kc, planting date and growing phase duration :</b> Chapagain 2004  <b>ET0:</b> FAO 2008 Global map of monthly reference evapotranspiration – 10 arc minutes, GeoNetwork. Following the CROPWAT approach, the monthly average data were converted to daily values by curve fitting to the monthly average through polynomial interpolation</p>	<p><b>The green water footprints of crops [m3 per ton]</b>= total volume of green water use /quantity of the production</p> <p><b>Rain-fed crops:</b>                      Green water use of rain-fed crop = <math>ET_a = K_c * K_s * ET_0</math> : actual crop evapotranspiration accounting for a possible water stress  <b>Irrigated crops:</b>                      Green water use of irrigated crop = <math>ET_a = K_c * K_s * ET_0</math> : actual crop evapotranspiration of non-irrigated crop, using crop parameters of irrigated crop</p>	<p>No information about the volume of freshwater released                      Information about the quality (nitrate) of the water released through the grey water footprint: calculated by multiplying the fraction of nitrogen that leaches or runs off by the nitrogen application rate, and dividing this by the maximum acceptable concentration of nitrogen and by the actual crop yield.</p> <p>- Country-specific N fertilizer application rates by crop: estimated with Heffer(2009), FAO(2006,2009) and IFA (2009)                      - Fraction N leached= 10% applied fertilization (Chapagain et al. 2006)                      - Maximum acceptable concentration of N= 10 mg NO3-N per Litre (Chapagain et al. 2006)                      - Natural N concentrations: assumed to be zero</p>	n.a	<p>Actual yield calculated with a simple water production function (Doorenbos and Kassam 1979): <math>(1 - Y_a/Y_{max}) = K_y (1 - ET_a/ET_{max})</math> - maximum yield (<math>Y_{max}</math>) : national average yield x 1,2 (from Reynolds et al. 2000)                      - crop yield response factor (<math>K_y</math>) : crop specific in (Doorenbos and Kassam 1979)</p>	

REFERENCE DATABASES	Available at	Crop	Spatial scale	Reference period	Water withdrawal from			Water emitted to				Waste-water send to treatment	Yield
					Surfacewater	Groundwater	Waste water	air		Ground-water	Surface water		
								Blue water consumption	Green water consumption				
Agri-footprint - Blonk Consultants	Freely available to SimaPro users <a href="http://www.agri-footprint.com">www.agri-footprint.com</a>	30 crops	Data available at country scale	Idem Mekonnen and Hoekstra 2010	n.a	n.a	n.a	<b>“Water, unspecified natural origin”, with a specific country suffix</b> <b>Water use for irrigation [m3/ha cultivated] = blue water footprint</b> Based on the blue water footprint from Mekonnen & Hoekstra (2010) [m3/tonne of product], combined with FAO yields (2007-2011) to obtain water use for irrigation in m3/ha	n.a	n.a	n.a	n.a	FAO (2007-2011)
Quantis Water Database (Quantis 2011)	<a href="http://www.quantis-intl.com/microsites/waterdatabase.php">http://www.quantis-intl.com/microsites/waterdatabase.php</a>	Idem Ecoinvent 2 + additional crops	Country level, global coverage	Pfister and Mekonnen	= Evapotranspiration from irrigation (blue water consumption) / irrigation efficiency Origin based on national statistics	= Evapotranspiration from irrigation (blue water consumption) / Irrigation efficiency Origin based on national statistics	n.c	Based on Pfister et al. 2011. When data are not available for a crop or to be more specific in a region, the Pfister et al. 2011 model is applied to the Blue water footprint published by the Water Footprint Network. (Bayart, personal communication)	n.a	Water emitted to ground water: result of the water balance  Surface groundwater partitioning based on estimate	Water emitted to surface water: result of the water balance  Surface groundwater partitioning based on estimate	n.c	n.c
AgriBalyse	<a href="http://www.ademe.fr">http://www.ademe.fr</a>	25 crops (grown in France and imported)	Regional or country scale,	2005-2009	n.a	n.a	n.a	Irrigation water in [m3/ha] : primary data, crop specific. But the exact physical meaning of this water is not clear for all crops.	n.a	n.a	n.a	n.a	Primary data

REFERENCE DATABASES	Available at	Crop	Spatial scale	Reference period	Water withdrawal from		Waste water	Water emitted to				Waste-water send to treatment	Yield
					Surfacewater	Groundwater		air		Ground-water	Surface water		
								Blue water consumption	Green water consumption				
Ecoinvent v2.2	<a href="http://www.ecoinvent.ch/">http://www.ecoinvent.ch/</a>	About 25 crops	Country level	Depend on the crop	Elementary flows of <b>water withdrawal</b> in m3 of water. Distinguishes between water from lake; river; ground (well); unspecified natural origin; turbined water; and other non-elementary flows such as salt water from sole (e.g., produced water in oil and gas extraction); and salt water from ocean. It also has a flow referred as cooling water from unspecified natural origin. But : for many crops, the irrigation water is not reported.		n.a	n.a	n.a	n.a	n.a	n.a	
Levova & Pfister 2014 - ecoinvent v3.0 Good practice for life cycle inventories, modelling of water use DRAFT Pfister 2012 New water data in Ecoinvent v3 48th LCA DF	<a href="http://www.ecoinvent.ch/">http://www.ecoinvent.ch/</a>	Crop data based on Pfister et al. (2011)	Country level		Water, <source>, From environment Irrigation, from technosphere <source> = lake - river - unspecified natural origin	Water, <source>, From environment Irrigation, from technosphere <source> = well	Waste water, from technosphere	Water to air unspecified/urban/non-urban/..	n.a	water, to water, ground = 0.8*irrigation - Attention: "irrigation" may be confusing. In fact it is "irrigation - ETirrigation", Cf. Nemecek et al. 2014	water, to water, surface water = (1-0.8)*irrigation - Attention: "irrigation" may be confusing. In fact it is "irrigation - ETirrigation", Cf. Nemecek et al. 2014	Wastewater from...irrigation/... -> properties: pollutant content	



## Chapter 4 - Supplementary information - E.T.: An operational field water and salt flows model for agricultural LCA illustrated on citrus

### Capillary rise equation (to be added in the model in future development)

The relationship between capillary rise and the depth of the groundwater table is given by the equation:

$$CR = \exp\left(\frac{\ln(z)-b}{a}\right) \text{ in mm/day}$$

With:

- z: aquifer depth below the soil surface (m)

- a and b: parameters specific to the soil texture and hydraulic characteristics

a and b parameters are estimated with equations defined for 4 soil classes (for a given range of saturated hydraulic conductivity (K<sub>sat</sub>)), and provided in Raes et al. (2012):

**Table 3.7b** – Equation 3.7k and 3.7l for the 4 soil Classes with indication of the considered range for the saturated hydraulic conductivity (K<sub>sat</sub>) (Janssens, 2006).

Soil Class	Range K <sub>sat</sub> mm.day <sup>-1</sup>	a Eq. 3.7k	b Eq. 3.7l
<b>I. Sandy soils</b> sand, loamy sand, sandy loam	200 to 2000	$-0.3112 - 10^{-5} K_{sat}$	$-1.4936 + 0.2416 \ln(K_{sat})$
<b>II. Loamy soils</b> loam, silt loam, silt	100 to 750	$-0.4986 + 9 (10^{-5}) K_{sat}$	$-2.1320 + 0.4778 \ln(K_{sat})$
<b>III. Sandy clayey soils</b> sandy clay, sandy clay loam, clay loam	5 to 150	$-0.5677 - 4 (10^{-5}) K_{sat}$	$-3.7189 + 0.5922 \ln(K_{sat})$
<b>IV. Silty clayey soils</b> silty clay loam, silty clay, clay	1 to 150	$-0.6366 + 8 (10^{-4}) K_{sat}$	$-1.9165 + 0.7063 \ln(K_{sat})$

### Curve Number (CN) calculation according to antecedent soil moisture

Curve Number calculation accounting for Antecedent Soil Moisture (ASM)(Raes et al. 2012):

$$CN_{AMC I} = -16.91 + 1.348 CN_{AMC II} - 0.01379 CN_{AMC II}^2 + 0.0001172 CN_{AMC II}^3$$

with  $0 \leq CN_{AMC I} \leq 100$  (Eq.3.7g)

$$CN_{AMC III} = 2.5838 + 1.9449 CN_{AMC II} - 0.014216 CN_{AMC II}^2 + 0.000045829 CN_{AMC II}^3$$

with  $0 \leq CN_{AMC III} \leq 100$  (Eq.3.7h)

## Kcb equation in the case of active cover crop (Allen and Pereira 2009):

$$K_{cb} = K_{cb_{cov}} + K_d \left[ \max \left( K_{cb_{full}} - K_{cb_{cov}}, \frac{K_{cb_{full}} - K_{cb_{cov}}}{2} \right) \right]$$

With:

- $K_{cb_{cov}}$ : Kcb of the ground cover in the absence of tree foliage
- $K_d$ : density coefficient describes the increase in Kc with increase in amount of vegetation (function of LAI or fraction of ground covered by vegetation)
- $K_{cb_{full}}$ : basal Kc during peak plant growth for conditions having nearly full ground cover

## Kd (density coefficient) equation (Allen and Pereira 2009) :

The density coefficient  $K_d$  describes the increase in  $K_c$  with increase in amount of vegetation:

$$K_d = \min(1, M_L \times f_{c_{eff}}, f_{c_{eff}}^{(1/(1+h))})$$

With:

- $f_{c_{eff}}$  : effective fraction of ground covered by vegetation
- $M_L$  : describe the effect of canopy density on shading and on maximum relative ET per fraction of ground shaded
- $h$ : mean height of the crop

## Stress coefficients

The fraction of water available: parameter  $p$

The Readily Available soil Water (RAW) is usually calculated through the parameter  $p$ : the average fraction of TAW that can be depleted from the root zone before water stress (reduction in ET) occurs:

$$RAW = TAW \times p$$

$0 < p < 1$ , is provided in Allen et al. (1998) table 22 and is crop specific. However,  $p$  is defined for a reference evapotranspiration of 5 mm.day<sup>-1</sup>. Thus, a daily adjustment of  $p$  would be required to be more accurate in the water balance calculation. This can be computed like in Aquacrop following Raes et al. (2012):

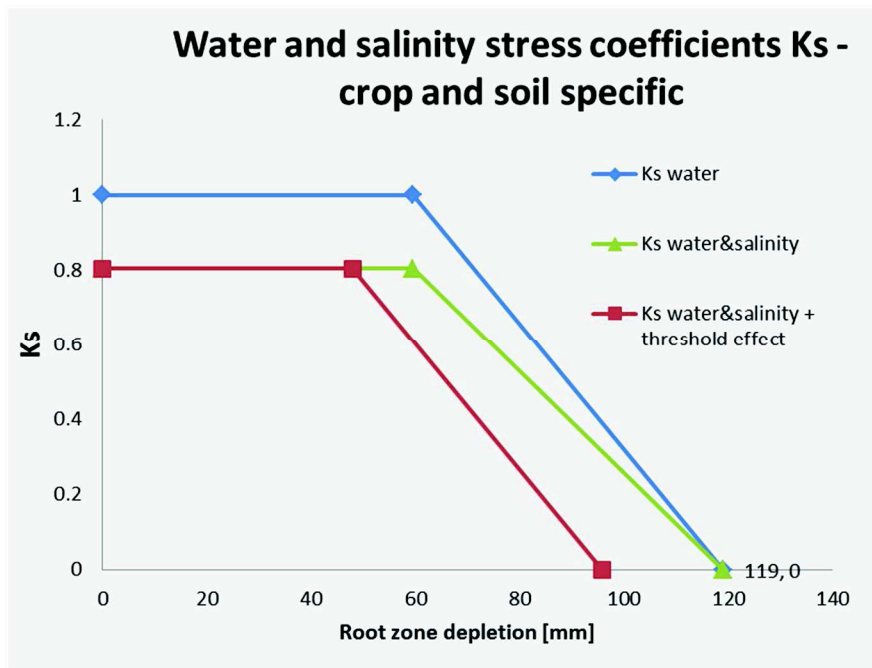
$$0 \leq p_{adj} = p_{given} + f_{adj} (0.04(5 - ET_0)) \left( \log_{10}(10 - 9 p_{given}) \right) \leq 1$$

An illustration of the stress coefficients:

The figure below illustrates the curve profile of:

- water stress coefficient  $K_{s_{water}}$ ,
- combined water and saline stresses coefficients  $K_{s_{water\&salinity}}$ ,
- combined water and saline stresses coefficients accounting for the reduced threshold of RAW' (Readily Available soil Water)  $K_{s_{water\&salinity} + \text{threshold effect}}$

The curves are provided as an example for a citrus crop on a loamy sand soil, with a soil water salinity of 3 dS.m<sup>-1</sup>:



Regarding the  $Ks_{\text{water\&salinity} + \text{threshold effect}}$ , the reduced RAW of the calculated as:

$$RAW' = TAW \times p'$$

$$\text{With: } p' = p \times Ks_{\text{salinity}}$$

The calculation of the reduced threshold  $p'$ , rely on the computation of stress coefficient of Aquacrop model (Raes et al. 2012). However, it is important to note that Aquacrop decompose the saline stress in several components to account for the varying sensitivity of the crop growth stage. Thus, Aquacrop model distinguishes stress on leaf expansion and stomatal conductance. In the calculation above, we should have considered  $Ks_{\text{salinity}}$  of stomatal conductance, but use the overall  $Ks_{\text{salinity}}$  instead because the discrimination was not possible.

## Salinity stress coefficient - E.T. model

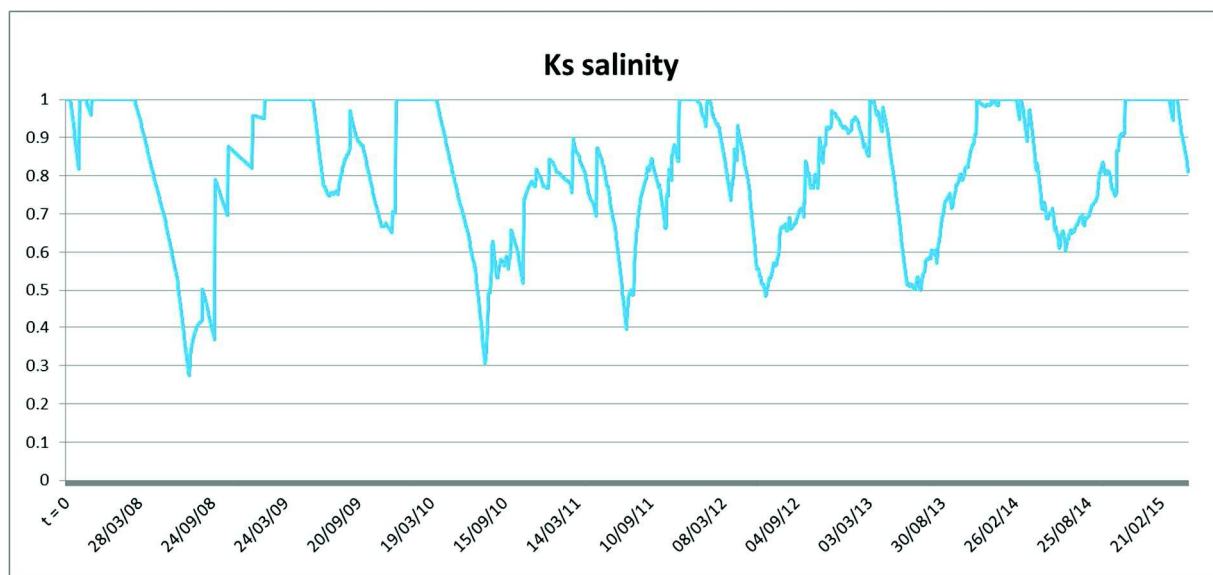


Figure 1. Salinity stress coefficient evolution (the same for the two model version E/T partitioning, water and salinity stresses, and ET partitioning, water and salinity stresses)

## Ratio % variation output/ % variation input

Table 1. Comparng the variations of model outputs with the variations of model inputs: ratio of % variation output/ % variation input

Input	% variation ETblue	% variation ETgreen	% variation percolating water	% variation runoff water	% variation E blue	% variation T blue	Total cumulated
wz= 0,4	11.9%	9.2%	-20.0%	-1.5%	<b>43.4%</b>	-6.3%	92.3%
z= 1,5	1.67%	2.2%	-4.3%	0.9%	-0.1%	2.7%	11.8%
G: +20%	-8.6%	-6.5%	15.0%	0.0%	25.0%	-28.0%	83.1%
G: -50%	<b>17.0%</b>	20.1%	-31.8%	0.0%	-32.9%	<b>45.9%</b>	147.6%
CN=85	-8.08%	<b>-37.1%</b>	<b>-69.8%</b>	<b>727.2%</b>	4.8%	-12.8%	859.7%
K <sub>cb</sub> Villalobos et al. (2013)	24.68%	<b>55.38%</b>	<b>-53.73%</b>	0.00%	-5.20%	41.98%	181.0%
K <sub>cb</sub> Allen and Pereira (2009)	67.76%	45.80%	<b>-115.44%</b>	-1.26%	2.60%	<b>105.50%</b>	338.4%
ET <sub>0</sub> Climwat	-7.53%	-14.96%	15.85%	0.00%	17.50%	<b>-22.03%</b>	77.9%

## References - Supplementary information Chapter 4

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## Chapter 5 - Supplementary information - Life Cycle Assessment of a perennial crop with in-depth analysis of water use impacts: The case of a Mandarin in Morocco

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### Geographical context



Figure 1. Google view of the studied farm (2015)

### Nursery and mandarin cultivation

Stock	Area (ha)
Volkameriana ( <i>Citrus limonia</i> Osbek)	102,50
Citrange carrizo (hybrid)	75,95
Bigaradier ( <i>Citrus Aurantium</i> L.)	28,51
Citrulemo sacaton (hybrid)	11,23
Citrus macrophylla ( <i>Citrus macrophylla</i> Wester)	5,79
Citrange C-35 (hybrid)	1,02

## Nitrate leaching estimation method

**Step 1:** A Nitrogen budget is computed annually accounting for N inputs and N outputs:

Year xxxx :			
N inputs (kg N/ha)		N outputs (kg N/ha)	
Description	Quantity (kg N/ha)	Description	Quantity (kg N/ha)
<i>Mineral fertiliser</i>		<i>Exportation in fruits</i>	
<i>Organic fertiliser</i>		<i>N fixation in wood</i>	
<i>Mineralisation in the soil</i>	equal to immobilisation	<i>Immobilisation in the soil</i>	equal to minéralisation
<i>N in irrigation water</i>		<i>N-NH3 emissions (volatilisation)</i>	
		<i>N-N2O emissions (denitrification)</i>	
		<i>N-N2 emissions (denitrification)</i>	
<b>N-NO<sub>3</sub><sup>-</sup> leachable (kg/ha) :</b>		<b>N-NO<sub>3</sub><sup>-</sup> = ∑ input - ∑ output</b>	
<b>NO<sub>3</sub><sup>-</sup> leachable (kg/ha) :</b>		<b>NO<sub>3</sub><sup>-</sup> = N-NO<sub>3</sub><sup>-</sup> * 62/14</b>	

**Step 2:** According to Pervanchon et al. (2005), the **leaching coefficient** can be estimated through:

$$\%N_{\text{leached}} = (Wd / (Wd + (Wsr/10)))^{D/2}$$

With: Wd is the average drainage (in mm) over 30 years during the drainage period, Wsr is the volumetric soil water retention (in %) and D is the rooting depth (in cm).

Instead of the 30 year average, we use the annual drainage calculated with a water and salt balance model accounting for climate, soil and agricultural practices (E.T. model, Cf. Chapter 4)

We used D instead of D/2 because we assume that N is not uniformly distributed in the soil (Bockstaller and Girardin 2003).

**Step 3:** Then, the **amount of N losses through nitrates leaching** is estimated through:

$$NO_3 \text{ losses} = 100 * ((N \text{ leachable} * \%N \text{ leached}) / Wd) * 4.42$$

With: NO<sub>3</sub> is the amount of losses of nitrogen to the environment through NO<sub>3</sub> leaching due to agricultural practices (in mg NO<sub>3</sub>.L<sup>-1</sup>), Wd is the average drainage (in mm) over 30 years, %N leached is the part of nitrogen leached to water, and N leachable is the amount of mineral residual nitrogen in soil (in kg NO<sub>3</sub>-N.ha<sup>-1</sup>.yr<sup>-1</sup>). Instead of the 30 year average, we use the annual drainage calculated with a water and salt balance model (E.T. model, Cf. Chapter 4).

## Irrigation water volumes reported in published LCA studies on citrus

**Table 1. Average irrigation water volume (in m<sup>3</sup>.ton<sup>-1</sup>) for citrus for several LCA studies**

References	Product	Country	Irrigation system	Average irrigation volume (m <sup>3</sup> .ton <sup>-1</sup> )
Sanjuan et al. (2005)	Integrated orange	Spain	surface or fertigation	183
Beccali et al. (2010)	Conventional orange	Italy	n.c	168
Knudsen et al. (2011)	Conventional orange	Brazil	no irrigation	0
Lo Giudice et al. (2013)	Integrated orange	Italy	drip irrigation	184
Basset et al. (2015) (AgriBalyse)	Small citrus	Morocco	drip irrigation	286
This study (2015)	Mandarin Nadorcott	Morocco	drip irrigation	278

## Effects of water flows inventory an impact assessment method on impacts

**Table 2. Water inventory flows and water deprivation impacts results of Mandarin cultivation (nursery not included). Comparison of water inventory flows estimated with a water and salt balance model or taken from database, and comparison of water deprivation impacts associated with these water flows with different assessment methods. Values are expressed in m<sup>3</sup>.ton<sup>-1</sup> fruits destined to exportation (allocation included), even for databases.**

Water flow inventory method :	Blue Water withdrawal :	Blue Water released :	Blue Water consumed :	Midpoint impacts :		
	Irrigation water and evaporative losses	Deep percolating blue water	Evapo-transpirated blue water	Pfister et al. 2009	Boulay et al. 2011 (consumptive & degradative)	Boulay et al. 2011 (consumptive)
Model: E/T partitioning, water & salinity stresses	277.6	97.0	180.5	180.5	188.9	104.2
Model: E/T partitioning, water stress	277.6	72.6	205.0	205.0	212.3	118.6
Database: Pfister et al. 2011	n.a	n.a	237.2	237.2	n.a	n.a
Database: Pfister et al. 2014	n.a	n.a	149.3	149.3	n.a	n.a



## Comparison with published LCA of citrus

**Table 3. Global warming Potential, non-renewable energy, eutrophication, acidification and toxicity impacts (calculated with CML 2001) from different LCA studies for citrus and for this study. Results are expressed per kg of fruit at farm-gate.**

Reference	Impact assessment method	Product	Country	Climate change (kg CO <sub>2</sub> eq)	Acidification (g SO <sub>2</sub> eq)	Eutrophication (g PO <sub>4</sub> <sup>3-</sup> eq)	Non renewable energy (MJ)
Sanjuan et al. (2005)	CML 2001	Integrated orange	Spain	0.22-0.28	0.07-0.09	1.95	-
Beccali et al. (2010)	CML 2001	Conventional orange	Italy	0.217	1.387	0.905	3.42
Knudsen et al. (2011)	CML 2001, EDIP97 for acidification	Conventional orange	Brazil	0.112	1.10	0.99	1.26
Basset-Mens et al. (2015)	CML 2001, CED	Small citrus	Morocco	0.269	2.08	0.679	3.32
<b>This study (2015)</b>	CML 2001, CED	Mandarin Nadorcott	Morocco	0.273	1.86	1.97	3.44

## References - Supplementary information Chapter 5

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## Glossary

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- Actual yield:** Actual yield of the crop under actual conditions (i.e.: under potential stresses) [tonne/ha or kg/ha] (Allen et al. 1998).  
Denoted as: Y
- Actual evapotranspiration:** (or: evapotranspiration under non-standard conditions) evapotranspiration from crops grown under management and environmental conditions that differ from the standard conditions [mm per unit time] (Allen et al. 1998).  
Denoted as:  $ET_a$
- Blue water:** Fresh surface and groundwater, i.e. the water in freshwater lakes, rivers and aquifers. (Hoekstra et al. 2011)
- Crop coefficient:** serves as an aggregation of the physical and physiological differences between crops and the hypothetical reference crop (e.g.: crop canopy and aerodynamic resistance). In the single crop coefficient approach, the difference in evapotranspiration between the cropped and reference grass is combined into one single coefficient. In the dual crop coefficient approach, the crop coefficient is split into two factors describing separately the differences in evaporation and transpiration between the crop and reference surface [dimensionless] (Allen et al. 1998).  
Denoted as:  $K_c$
- Effective rainfall/precipitation:** Term interpreted differently not only by specialists in different fields but also by different workers in the same field (Dastane 1978). As a result, the definition provided here may differ from the effective rainfall mentioned in chapter 3 and 4.  
From the agricultural production point of view (as far as the water requirement of crops is concerned), effective rainfall is that portion of total annual or seasonal rainfall which is useful directly and/or indirectly for crop production at the site where it falls. It therefore includes water intercepted by living or dry vegetation, that lost by evaporation from the soil surface, the precipitation lost by evapotranspiration during growth, that fraction which contributes to leaching, percolation or facilitates other cultural operations either before or after sowing without any harm to yield and quality of the principal crops [mm]. (Dastane 1978)  
Denoted as:  $P_{eff}$
- Evapotranspiration:** combination of two separate processes whereby water is lost on the one hand from the soil surface by evaporation and on the other hand from the crop by transpiration [mm per unit time]. (Allen et al. 1998)  
Denoted as: ET
- Green water:** The precipitation on land that does not run off or recharge the groundwater but is stored in the soil or temporarily stays on top of the

soil or vegetation. Eventually, this part of precipitation evaporates or transpires through plants. Green water can be made productive for crop growth (but not all green water can be taken up by crops, because there will always be evaporation from the soil and because not all periods of the year or areas are suitable for crop growth). (Hoekstra et al. 2011)

- Grey water footprint:** **Concept used by the Water Footprint Network.** The grey water footprint of a product is an indicator of freshwater pollution that can be associated with the production of a product over its full supply chain. It is defined as the volume of freshwater that is required to assimilate the load of pollutants based on natural background concentrations and existing ambient water quality standards. It is calculated as the volume of water that is required to dilute pollutants to such an extent that the quality of the water remains above agreed water quality standards. (Hoekstra et al. 2011)
- Maximum evapotranspiration:** (or: evapotranspiration under standard conditions) evapotranspiration from disease-free, well-fertilized crops, grown in large fields, under optimum soil water conditions, and achieving full production under the given climatic conditions crop [mm per unit time] (Allen et al. 1998).  
Denoted as:  $ET_c$
- Maximum yield:** maximum (expected) yield of the crop in absence of environment or water stresses (i.e.: under standard conditions) [tonne/ha or kg/ha] (Allen et al. 1998).  
Denoted as:  $Y_{max}$
- Reference evapotranspiration:** evapotranspiration rate from a reference surface, not short of water. The reference surface is a hypothetical grass reference crop with specific characteristics [mm per unit time] (Allen et al. 1998).  
Denoted as:  $ET_0$
- Stress coefficient:** transpiration reduction factor due to water, salinity or nutrient stresses [dimensionless] (Allen et al. 1998).  
Denoted as:  $K_s$
- Water scarcity:** extent to which demand for water compares to the replenishment of water in an area, e.g. a drainage basin, without taking into account the water quality (ISO 14046: 2014).
- Water availability:** extent to which humans and ecosystems have sufficient water resources for their needs. Water quality can also influence availability (ISO 14046: 2014).
- Water footprint profile:** compilation of impact category indicator results addressing potential environmental impacts related to water (ISO 14046: 2014).

<b>Water footprint:</b>	metric(s) that quantifies the potential environmental impacts related to water (ISO 14046: 2014).
<b>Water use:</b>	use of water by human activity. Use includes, but is not limited to, any water withdrawal, water release or other human activities within the drainage basin impacting water flows and/or quality, including in-stream uses such as fishing, recreation, transportation (ISO 14046: 2014).
<b>Water consumption:</b>	describes water removed from, but not returned to, the same drainage basin (ISO 14046: 2014).
<b>Water withdrawal:</b>	(or: water abstraction) anthropogenic removal of water from any water body or from any drainage basin, either permanently or temporarily (ISO 14046: 2014).
<b>Water released:</b>	(or: return flow) The part of the water withdrawn for an agricultural, industrial or domestic purpose that returns to the groundwater or surface water in the same catchment as where it was abstracted. This water can potentially be withdrawn and used again (Hoekstra et al. 2011). <b>Not clearly defined by the ISO norm</b> (Cf. definition of “water use”).
<b>Yield response factor:</b>	crop yield response factor: describes the reduction in relative yield according to the reduction in maximum evapotranspiration caused by soil water shortage [dimensionless]. This factor was first introduced in FAO report N°33 (Doorenbos and Kassam 1979) (Allen et al. 1998). Denoted as: $K_y$

## References Glossary

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## Acronym list

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AoP: Areas of Protection  
CF: Characterization Factors  
CTA: consumption-to-availability  
DP: Deep percolating water flow [mm]  
EC: Electrical Conductivity [siemens per metre]  
EF: Effect Factor  
ET : Evapotranspiration  
ET<sub>o</sub>: reference evapotranspiration [mm per unit time]  
ET<sub>c</sub>: crop evapotranspiration under standard conditions [mm per unit time]  
ET<sub>a</sub>: actual crop evapotranspiration [mm per unit time]  
ET<sub>green</sub>: evapotranspiration of green water [mm per unit time]  
ET<sub>blue</sub>: evapotranspiration of blue water [mm per unit time]  
ET<sub>deficit blue</sub>: deficit irrigation blue water consumption through evapotranspiration, according to Pfister et al. (2011)  
ET<sub>expected blue</sub>: full irrigation blue water consumption through evapotranspiration, according to Pfister et al. (2011)  
FF: Fate Factor  
ISO: International Organization for Standardization  
K<sub>c</sub>: crop coefficient [dimensionless]  
K<sub>s</sub>: transpiration reduction factor due to water, salinity or nutrient stresses [dimensionless]  
K<sub>y</sub>: crop yield response factor [dimensionless]  
LCA: Life Cycle Assessment  
LCI: Life Cycle Inventory  
LCIA: Life Cycle Impact Assessment  
LU: Land Use  
LULUC: Land Use/Land Use Change  
P<sub>eff</sub>: effective rainfall [mm]  
RO: Runoff water flow [mm]  
S: Soil water stock [mm]  
SETAC: Society of Environmental Toxicology and Chemistry  
TDS: Total Dissolved Solids  
WTA: withdrawal-to-availability  
Y<sub>max</sub>: maximum yield [tonne/ha or kg/ha]  
Y: actual yield [tonne/ha or kg/ha]

## Abstract

Identifying the environmental hot spots of agriculture is crucial in a context where humanity has to produce more food and pollute less. Life Cycle Assessment (LCA) is a powerful tool to evaluate the environmental impacts of agricultural systems, but is still fraught with shortcomings, notably for the evaluation of impacts of freshwater use and of salinisation of water and soil. The core complexity lies in the double status of water and soil resources in LCA which are both a resource and a compartment. The three questions answered by the thesis were: **How to better assess the impacts associated with water and salts fluxes? What model should be developed for a relevant inventory of field water and salts fluxes? Is the developed model operational for an LCA study on a perennial crop?** The first question was answered through a literature review on salinisation impacts in LCA. It revealed the main environmental mechanisms of salinisation, the factors involved, and discussed the soil and water status, notably through a consistent definition of the technosphere and ecosphere boundary. To answer the second question, a critical analysis of water inventory and agri-food LCA databases showed their inadequacy for the LCA-based ecodesign of cropping systems: they provide estimates of theoretical water consumed, rely on data and methods presenting limitations, and do not support the calculation of both consumptive and degradative water use impacts. For the LCA-based ecodesign of cropping systems, the inventory of water flows should be based on a model simulating evapotranspiration, deep percolation and runoff accounting for crop specificities, pedo-climatic conditions and agricultural managements. For herbaceous crops, the FAO Aquacrop model constitutes a relevant and operational model, but no dedicated model is available to-date for perennials. To fill this gap, a tailored and simple model, so called E.T., was elaborated for the inventory of field water and salt flows for annual and perennial crops. The model combines daily water and salts balances, accounting for soil, climate, agricultural practices and possible crop water and salinity stresses. A first testing of the E.T. model demonstrated its discriminating power for agricultural practices and its robustness. Its validity domain can be extended and its accuracy increased thanks to the recommendations provided. E.T. was also tested in the LCA of a Mandarin grown in Morocco. For most impact categories, electricity use for irrigation was the main contributor revealing a water-energy nexus. Water use had a major contribution to damages for all areas of protection. Overall, to further improve the assessment of impacts due to water use (including salinization impacts) we recommend using a more mechanistic and hydrological approach.

## Résumé

Identifier les « hotspots » environnementaux de l'agriculture est crucial dans un contexte où l'humanité doit produire plus et polluer moins. L'Analyse du Cycle de Vie (ACV) est un outil puissant pour évaluer les impacts environnementaux des systèmes agricoles, mais souffre encore de lacunes, notamment pour l'évaluation des impacts liés à la consommation d'eau douce et la salinisation des eaux et des sols. La complexité fondamentale réside dans le double statut de l'eau et du sol en ACV qui sont à la fois des ressources et des compartiments. Les trois questions auxquelles la thèse répond sont: **Comment mieux évaluer les impacts associés aux flux d'eau et de sels? Quel modèle devrait être développé pour un inventaire pertinent des flux d'eau et de sels au champ? Le modèle développé est-il opérationnel pour une étude ACV d'une culture pérenne?** La première question a été traitée grâce à une revue de la littérature sur les impacts salinisations en ACV. Cette revue détaille les principaux mécanismes environnementaux de la salinisation, les facteurs impliqués, et discute du statut du sol et de l'eau, notamment en définissant une frontière cohérente entre technosphère et écosphère. Pour répondre à la seconde question, une analyse critique des bases de données d'inventaire eau et ACV de produits agroalimentaires a montré leur inaptitude pour l'ecodesign basé sur l'ACV: elles fournissent des estimations d'eau consommée théorique, se basent sur des données et méthodes qui présentent des limites, et ne permettent pas le calcul des impacts liés à l'usage consommateur et dégradant de l'eau. Pour l'ecodesign des systèmes agricoles basé sur l'ACV, l'inventaire des flux d'eau et de sels devrait se fonder sur un modèle simulant l'évapotranspiration, la percolation profonde et le ruissèlement, prenant en compte les spécificités de la culture, les conditions pédoclimatiques et les pratiques agricoles. Le modèle Aquacrop de la FAO est un modèle pertinent et opérationnel pour les cultures herbacées, mais il n'existe pas de modèle dédié aux cultures pérennes pour le moment. Pour pallier à ce manque, un modèle simple et « sur mesure », appelé E.T., a été élaboré pour l'inventaire des flux d'eau et de sels au champ, pour les cultures annuelles et pérennes. Le modèle combine un bilan journalier de l'eau et des sels, prenant en compte le sol, le climat, les pratiques agricoles et d'éventuels stress salin ou hydrique. Un premier test du modèle a démontré son pouvoir discriminant des pratiques agricoles et sa robustesse. Son domaine de validité peut être étendu et sa précision augmentée grâce aux recommandations fournies. E.T. a aussi été appliquée dans une ACV de Mandarine cultivée au Maroc. Pour la plupart des catégories d'impacts, l'usage d'électricité pour l'irrigation était un contributeur majeur, révélant une forte connexion entre l'eau et l'énergie. L'usage d'eau avait une contribution majeure aux dommages sur les trois aires de protection. Dans l'ensemble, pour améliorer davantage la prise en compte des impacts liés à l'usage de l'eau (dont la salinisation) nous recommandons d'adopter une approche plus mécaniste et hydrologique.

